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Index to Appendix of Supporting Materials

Exhibit A

United States
Environmental Protection
Agency

Office of Water
4304T

EPA-820P25001
January 2025

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

January 2025

U.S. Environmental Protection Agency Office of Water, Office of Science and Technology, Health and Ecological Criteria Division

Washington, D.C.

Acknowledgements

Technical Analysis Leads:

Sophie Greene, Office of Water, Office of Science and Technology, Health and Ecological Criteria Division

Todd Phillips, Standards and Water Quality Branch, Water Division, US EPA Region 7

Tess Richman, Office of Water, Office of Science and Technology, Health and Ecological Criteria Division

Lisa Weber, Office of Water, Office of Science and Technology, Health and Ecological Criteria Division

David Tobias, Office of Water, Office of Science and Technology, Health and Ecological Criteria Division

Modelers:

Ted Lilly, Anne Lutes, Rohit Warrier, and Donna Womack RTI subcontracted through GLEC Contract No.: 68HERC20D0019

Reviewers:

Brittany Jacobs, Amanda Jarvis, James Justice, Casey Lindberg, Jacques Oliver, and Colleen Flaherty, Office of Science and Technology, Health and Ecological Criteria Division

EPA Peer Reviewers

Michael Brooks, Marina Evich, Ronald Hermann, and John Washington. Office of Research and Development

Norman Birchfield and Jason Mills, Office of Land and Emergency Management

Aderonke Adegbule, Marcy Card, and Dirk Young, Office of Chemical Safety and Pollution Prevention

Inter-Agency Review

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).

Sara Lupton and David Smith, Agricultural Research Service, United States Department of Agriculture

Ian Edhlund, Barry Hooberman (retired), Lynn Post, Sara Sklenka, and Amanda Wiley, Center for Veterinary Medicine, Food and Drug Administration

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.

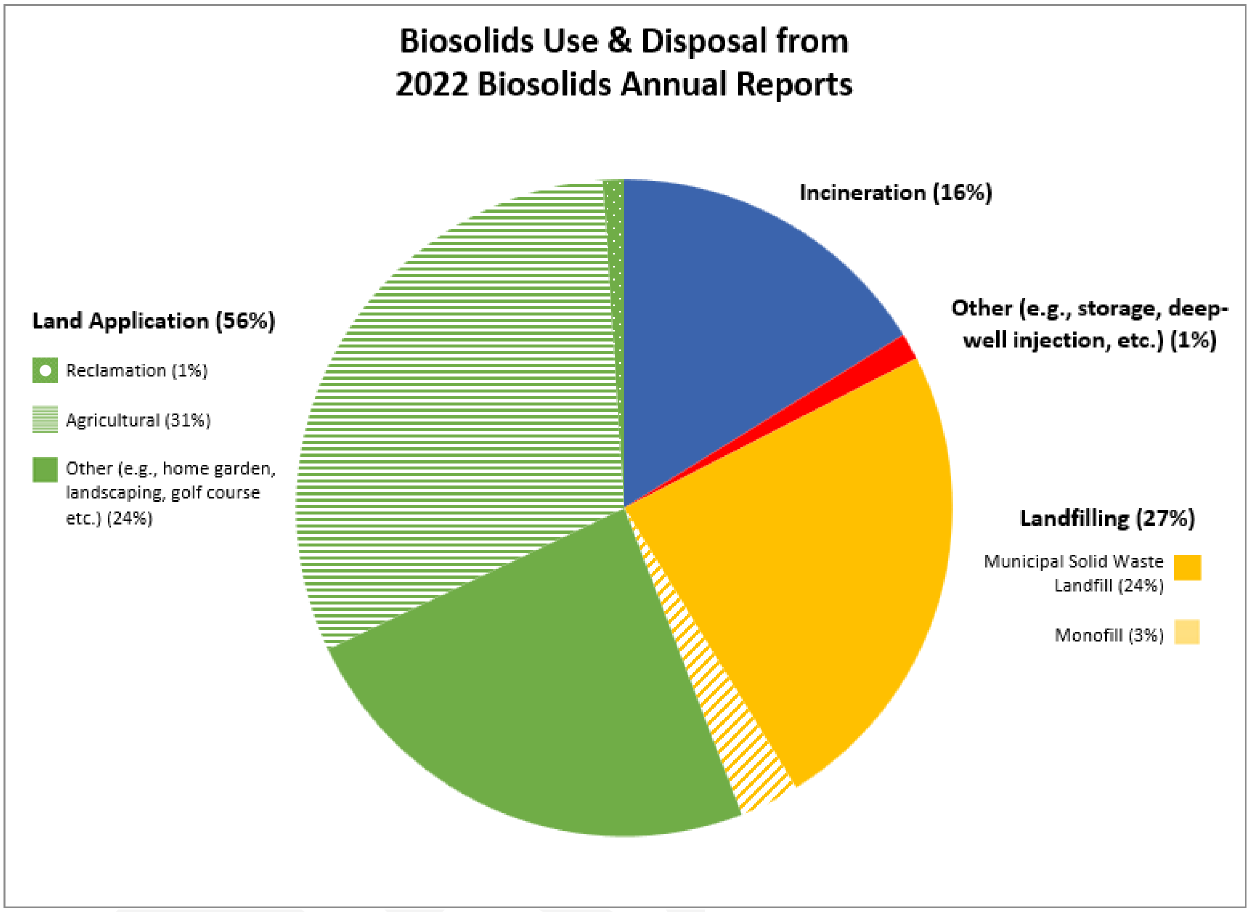


Figure 1. Distribution of sewage sludge use and disposal from Biosolids Annual Reports covering 2022 submitted to the EPA's Office of Enforcement and Compliance.

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 °C	Lide (2007); SRC (2016)
Melting Point	54.3 °C	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 °C (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	2.48×10^{-6} mm Hg at 20°C (potassium salt)	ATSDR (2021)
Henry's Law Constant	Not measurable; not expected to volatilize from aqueous solution ($< 2.0 \times 10^{-6}$)	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.

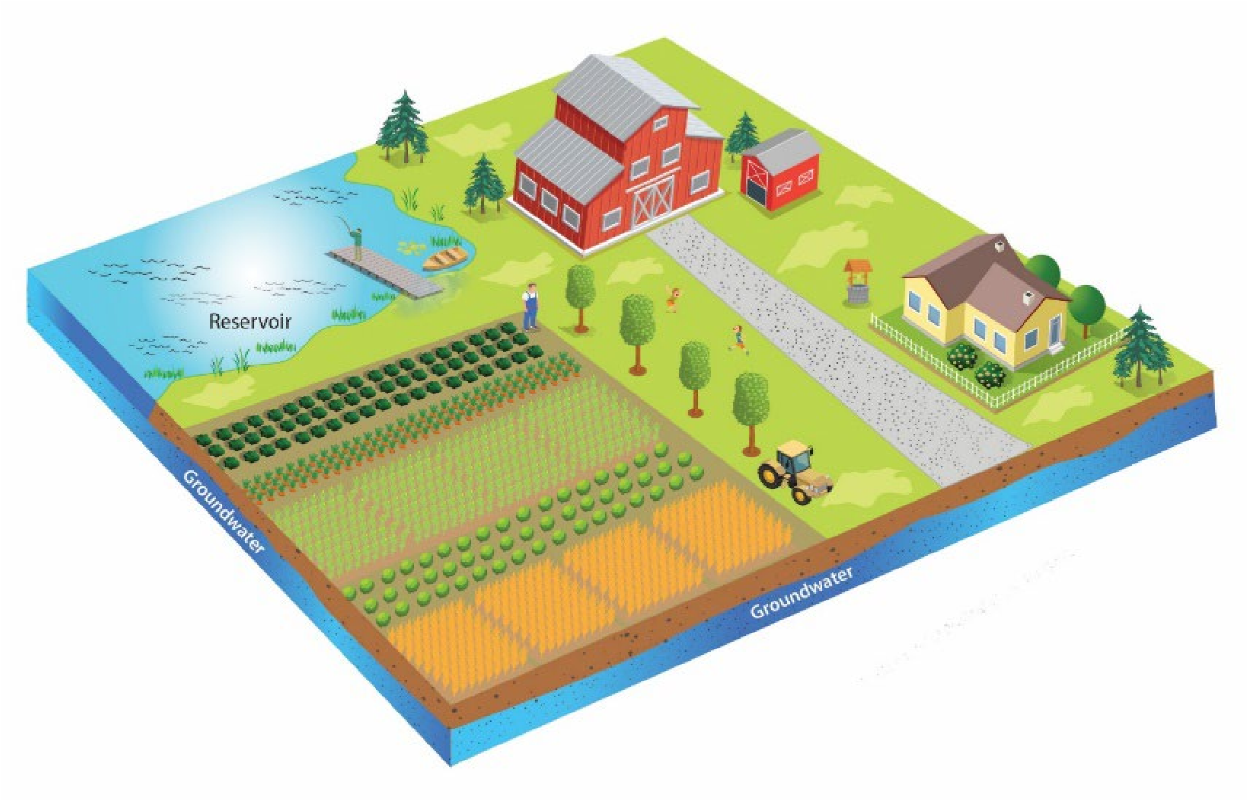
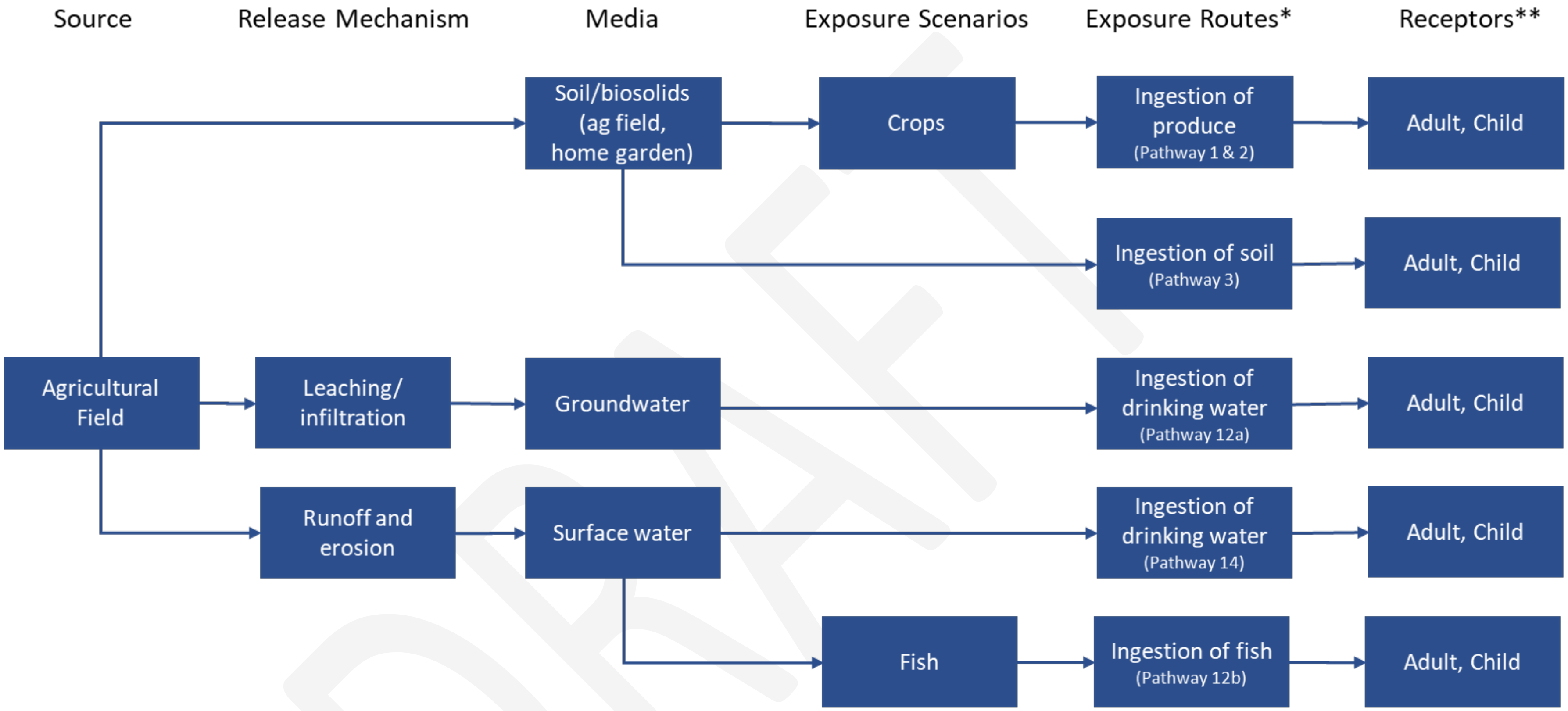


Figure 2. Conceptual visual depiction of crop farming scenario.



* 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.

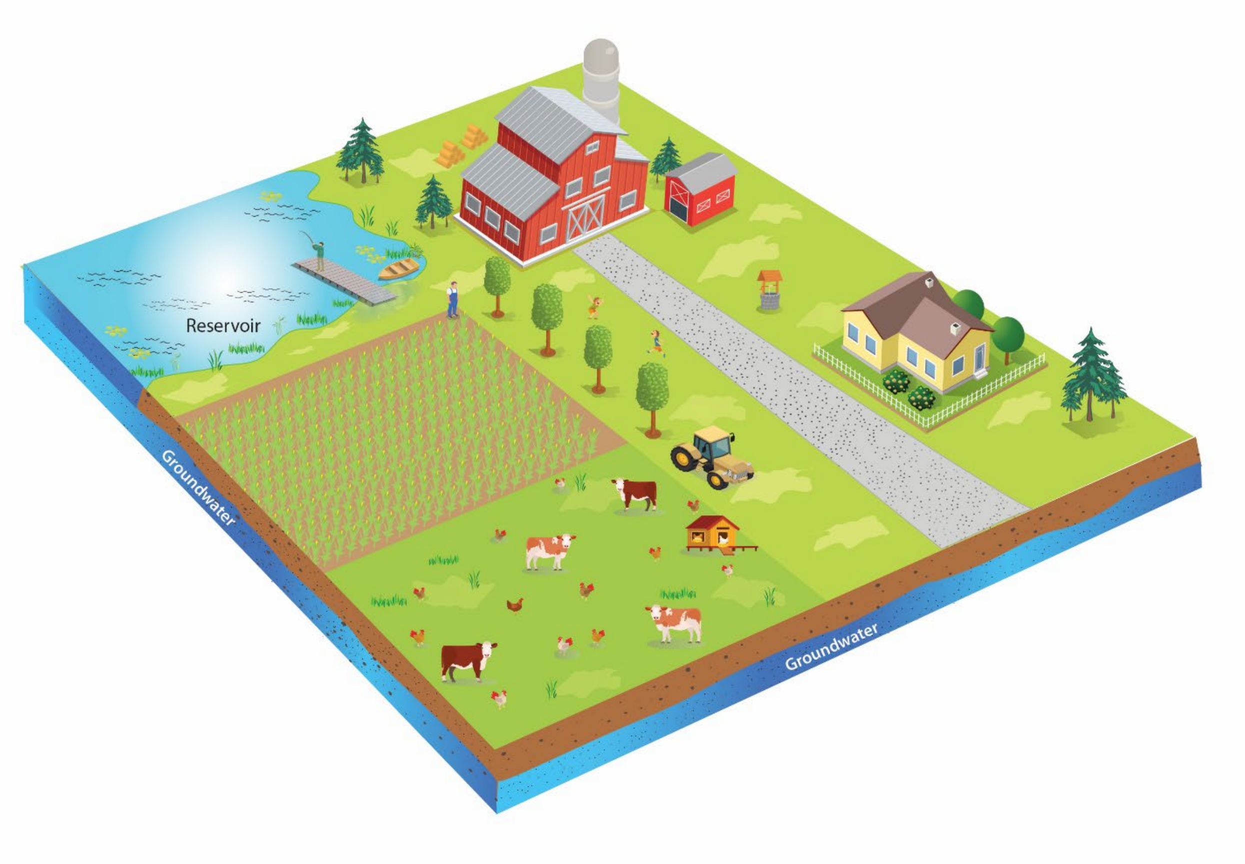
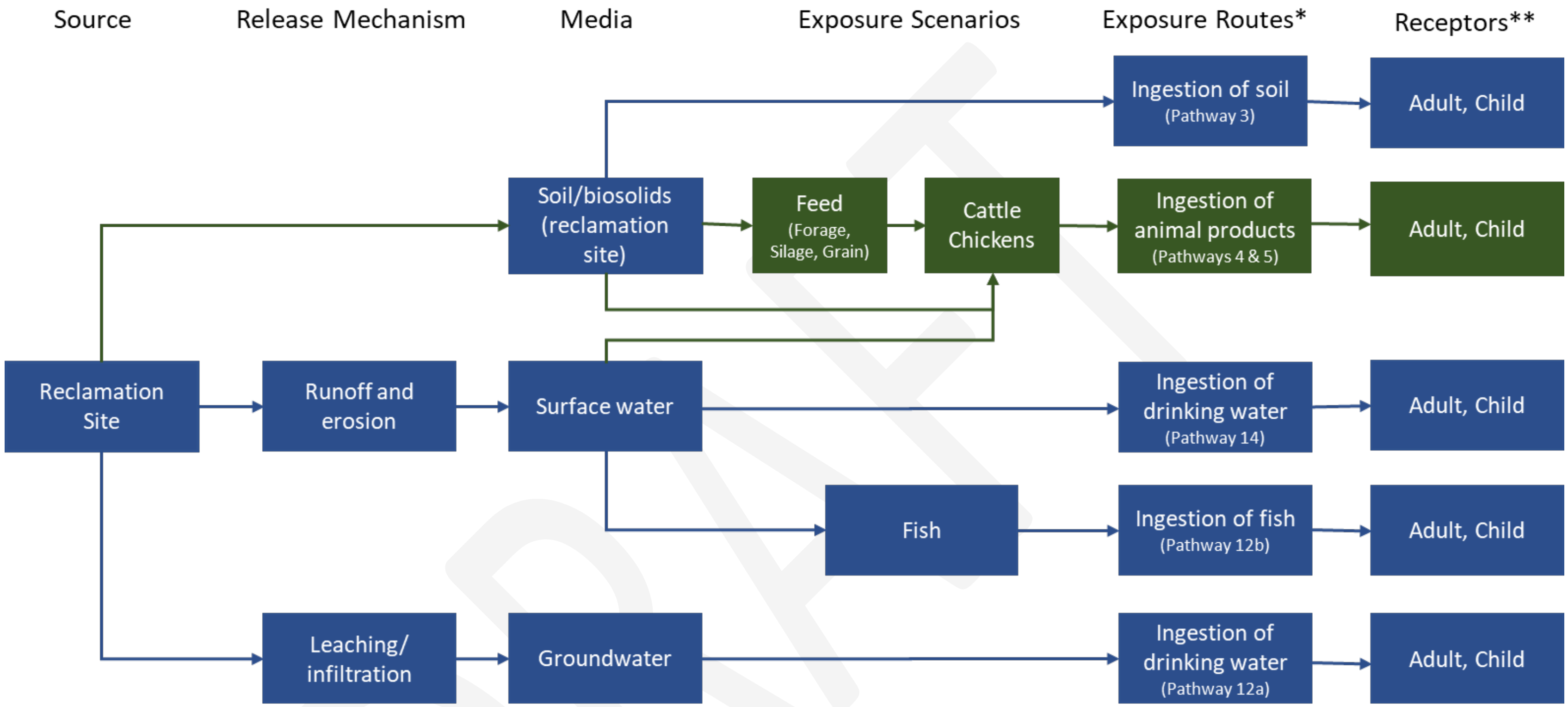


Figure 4. Conceptual visualization of pasture farm scenario.



* 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.

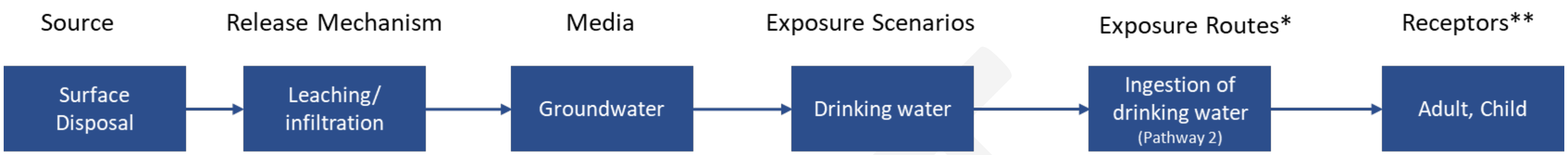
2.8.2 Land Reclamation

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).

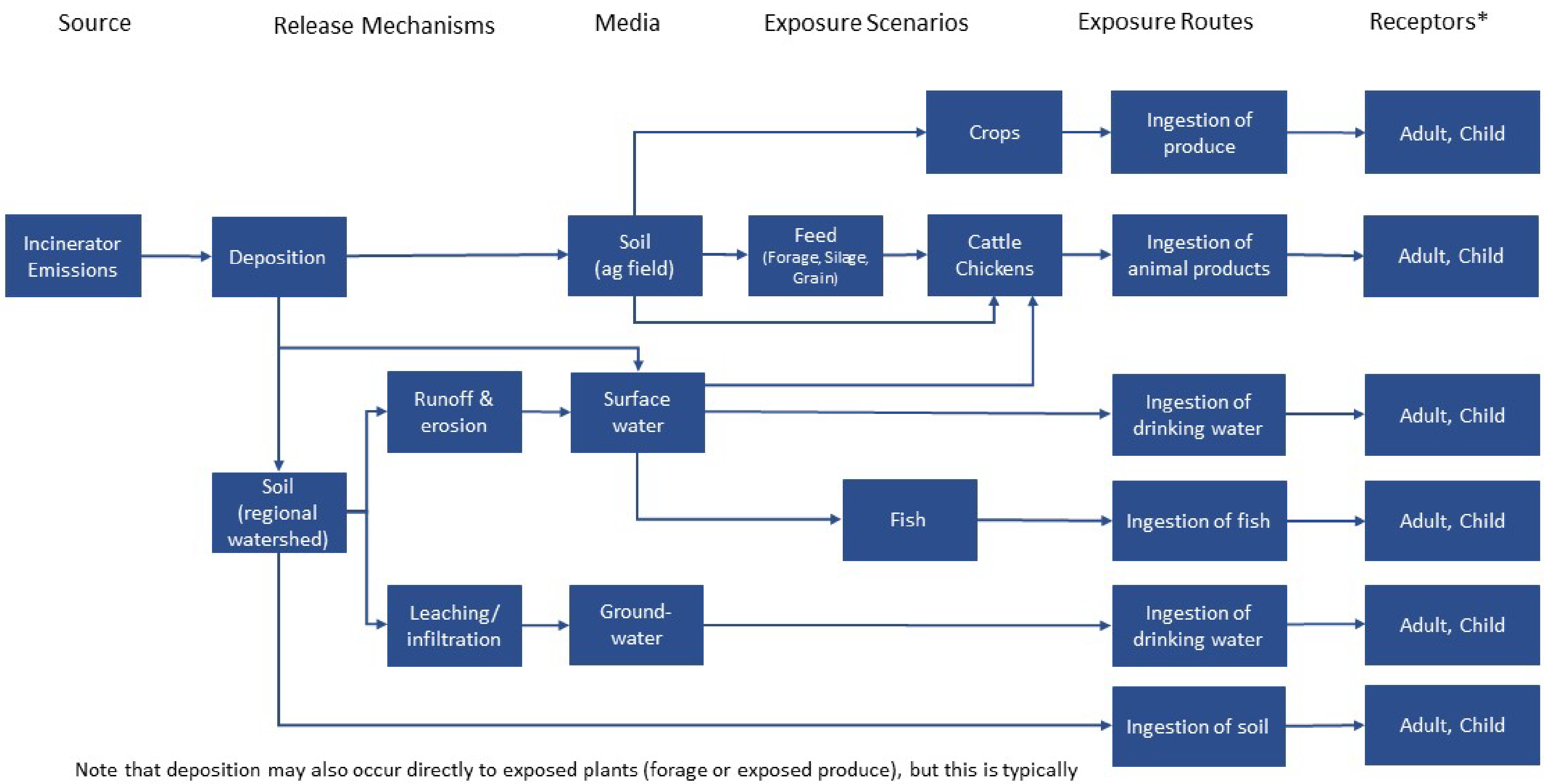
2.8.4 Incineration

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).

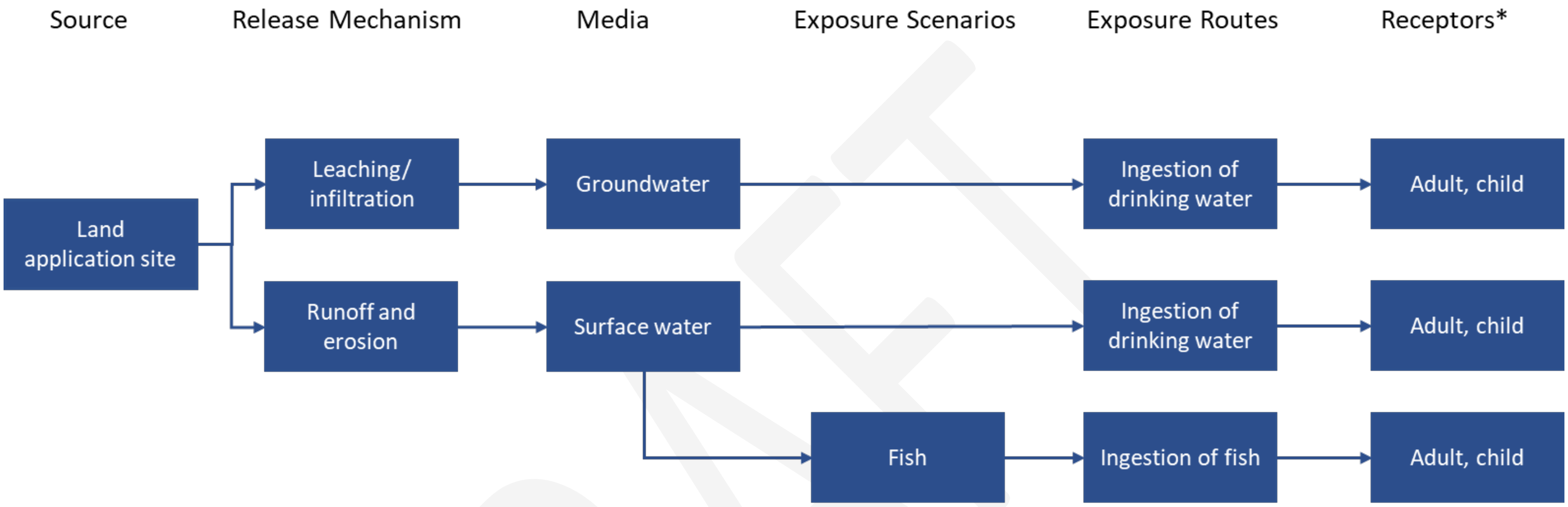


Note that deposition may also occur directly to exposed plants (forage or exposed produce), but this is typically a negligible pathway compared to soil uptake and is not shown.
* Receptor populations may include farm families, home gardeners, CSA participants, or nearby residents.

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)		
$A = BTF \times \left[I_{soil} + I_{feed} + I_{water}\right]$ where $I_{soil} = C_{soil} \times Q_{soil} \times Bs$ $I_{feed} = \sum_i P_i \times Q_i \times F_i$ $I_{water} = C_w \times Q_w \times F_w$		
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

	Low Koc				High Koc			
	Adult		Child		Adult		Child	
Pathway	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

	Low Koc		High Koc	
Pathway	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

Pathway	Low Koc				High Koc			
	Adult		Child		Adult		Child	
	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.

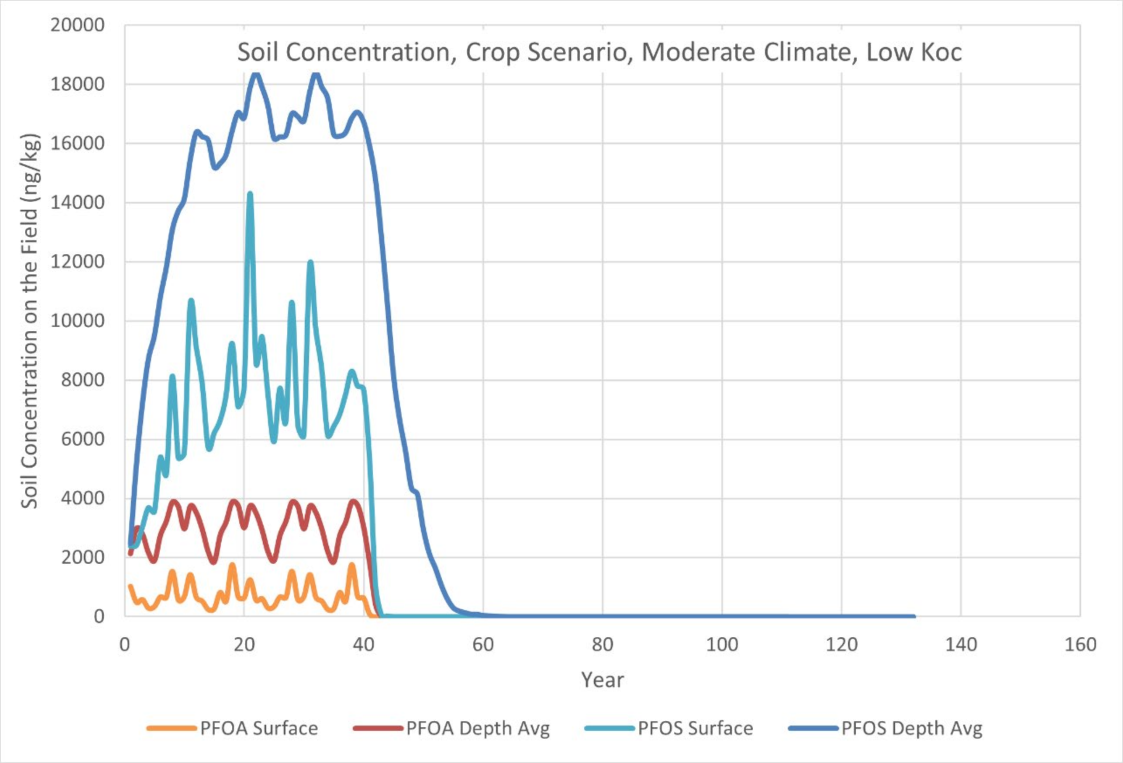

Figure 9. Plot of PFOA and PFOS concentrations over time in the "moderate" climate crop farm scenario with the low K_{oc} setting, assuming biosolids application ceases after 40 years.

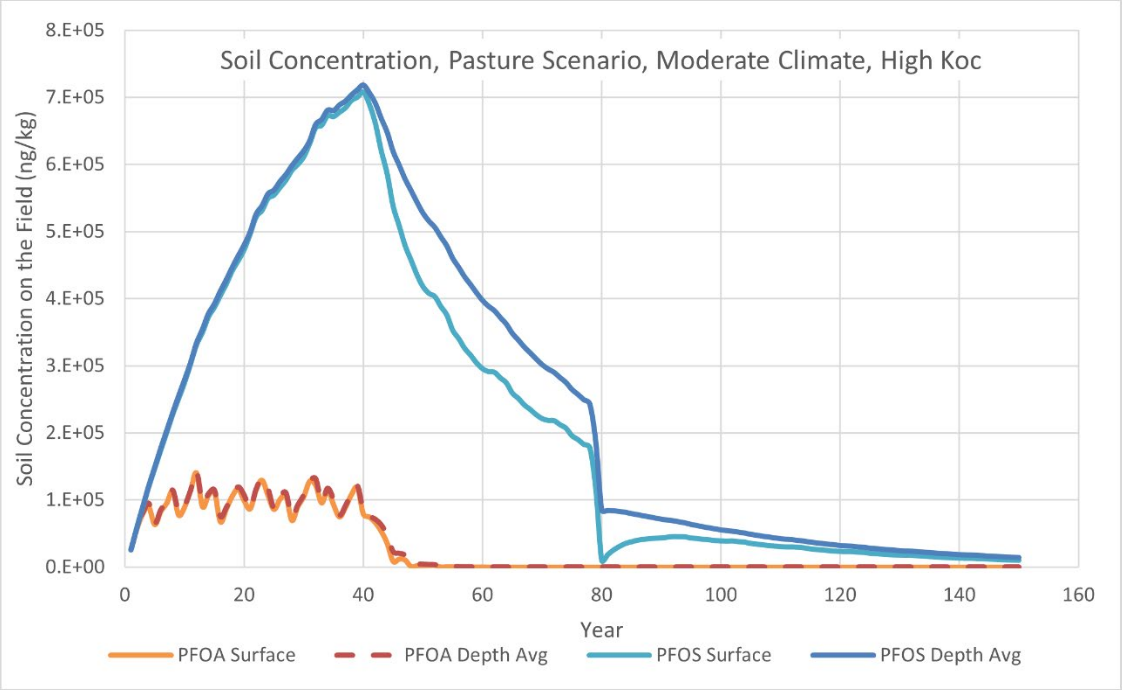

Figure 10. Plot of PFOA and PFOS concentrations in soil over time in the "moderate" climate pasture farm scenario with the high K_{oc} setting, assuming biosolids application ceases after 40 years.

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.

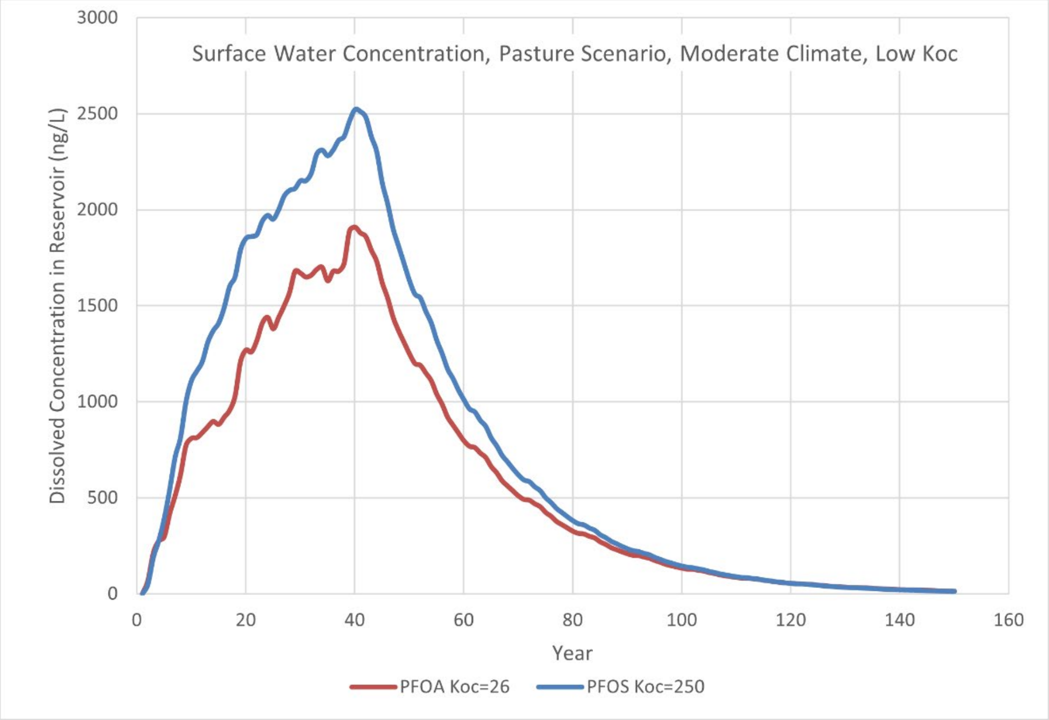

Figure 11. **PFOA and PFOS concentrations over time in the low K_{oc}, pasture farm, moderate climate setting.**

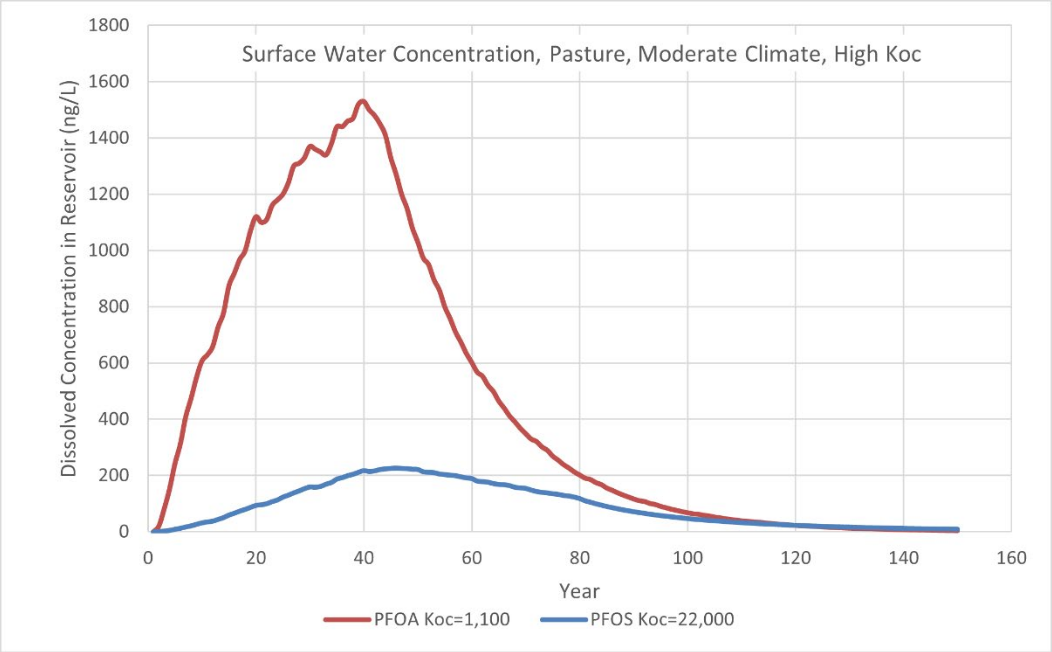

Figure 12. PFOA and PFOS concentrations over time in the high K_{oc}, pasture farm, moderate climate setting.

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.

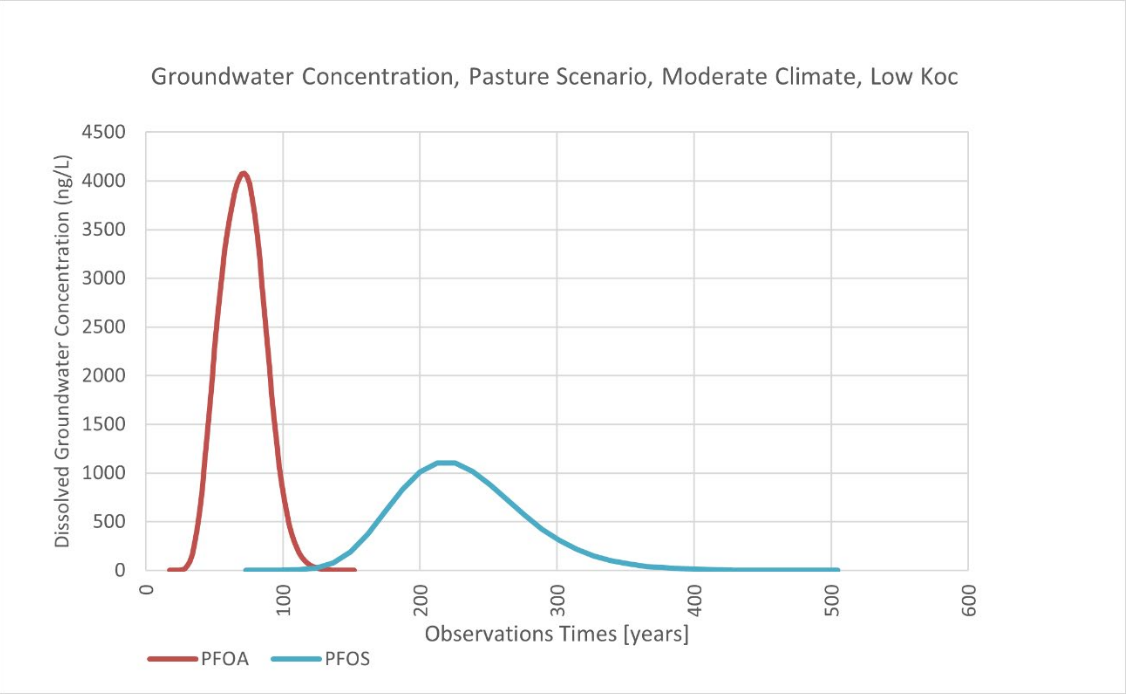

Figure 13. PFOA and PFOS concentrations over time in the low K_{oc}, pasture farm, moderate climate setting.

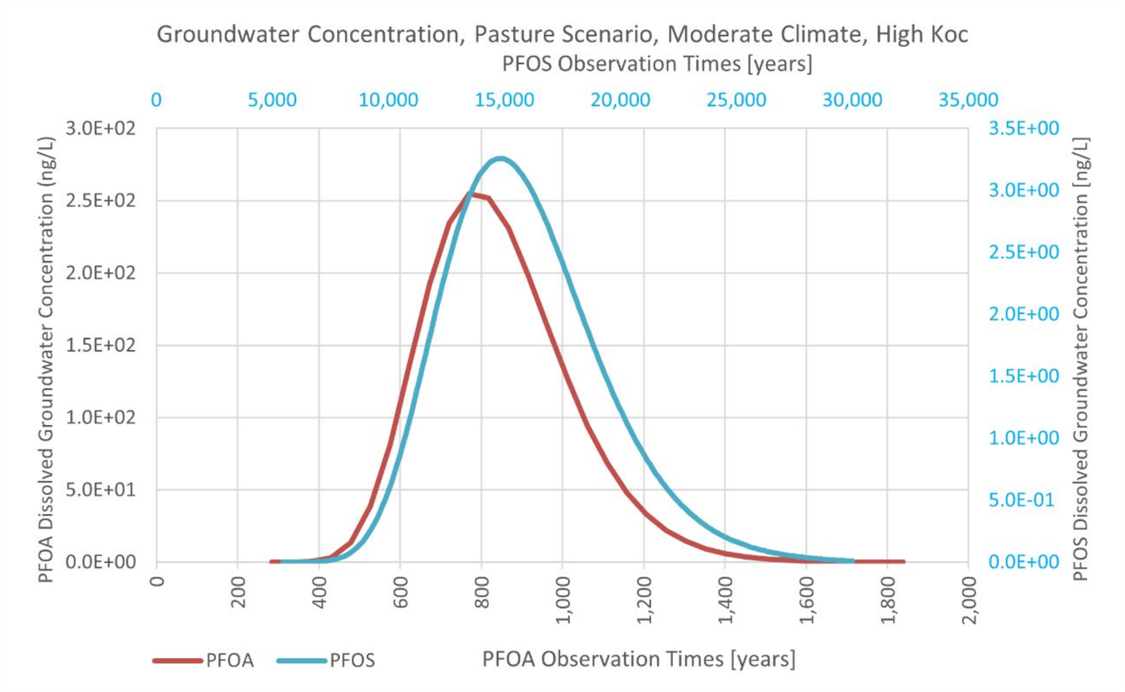

Figure 14. PFOA and PFOS concentrations over time in the high K_{oc}, pasture farm, moderate climate setting.

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

Pathway	Low Koc				High Koc			
	Adult		Child		Adult		Child	
	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

Pathway	Low Koc				High Koc			
	Adult		Child		Adult		Child	
	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

Pathway	Low Koc				High Koc			
	Adult		Child		Adult		Child	
	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

	Low Koc				High Koc			
	Adult		Child		Adult		Child	
Pathway	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.

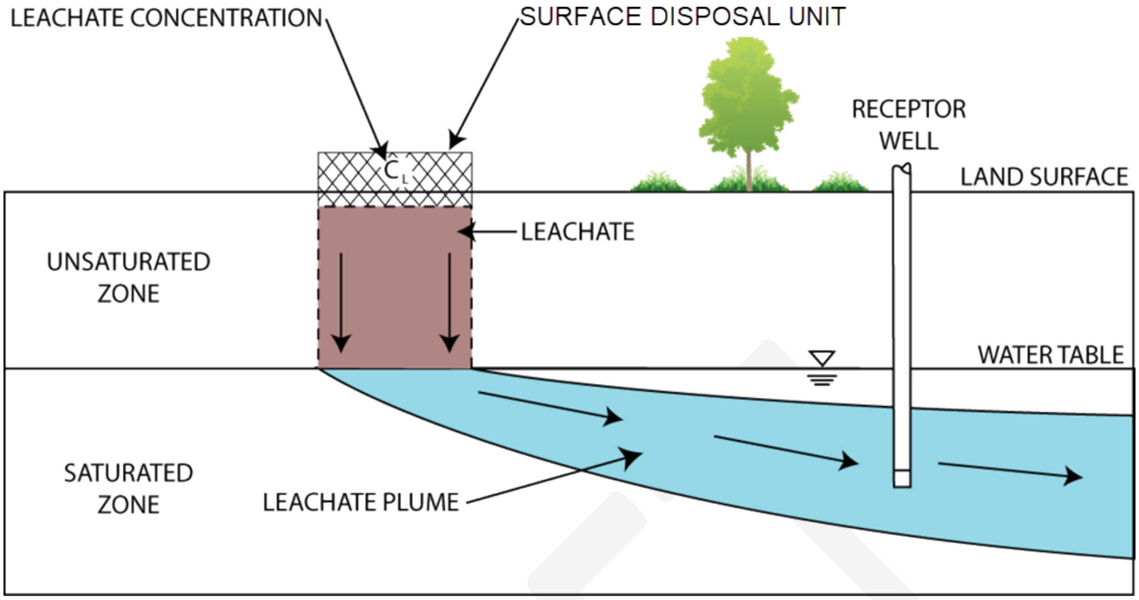


Figure C-1. Schematic diagram of groundwater modeling scenario.

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.

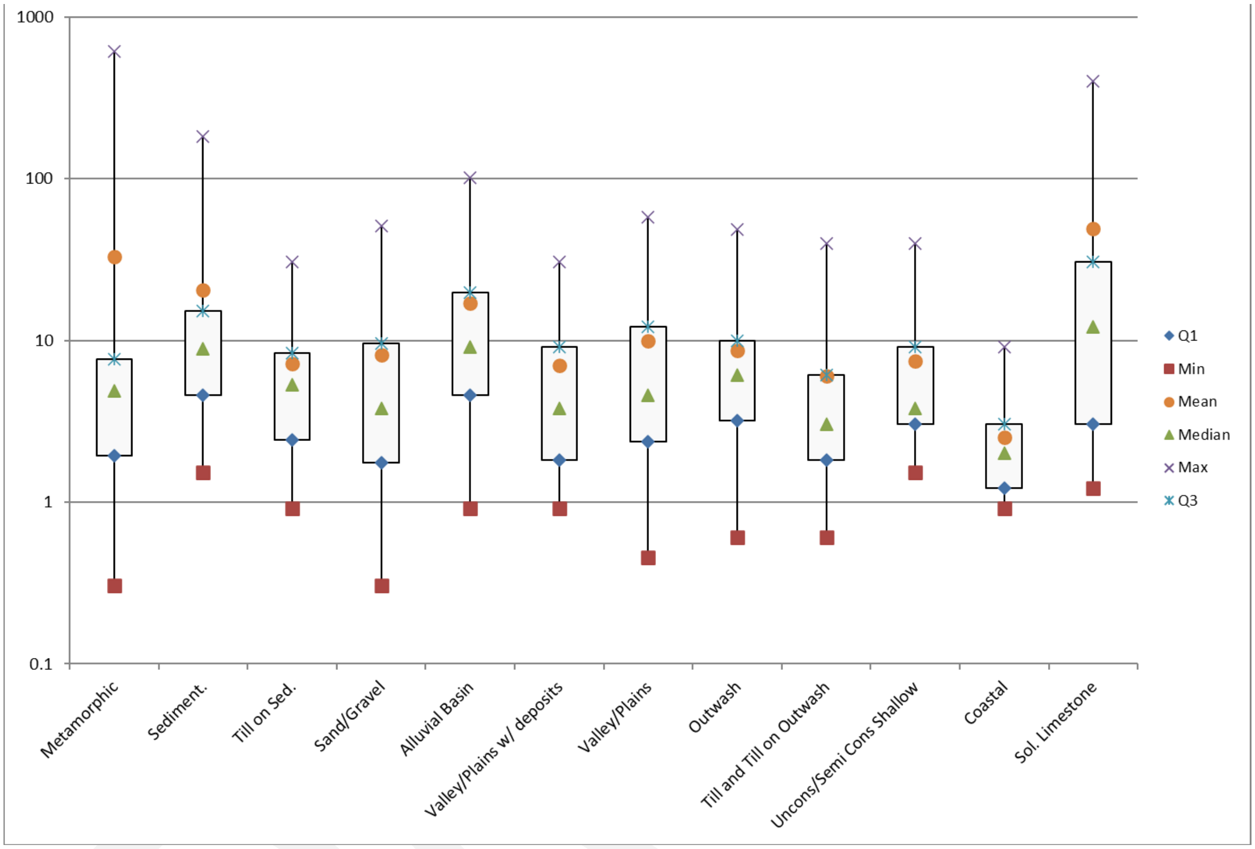


Figure C-2. Depth to water table data from HGDB.

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

Chemical Property	Notation	Units	Used in Models			Values		Reference/Notes
			EPACMTP	HYDRUS	ANALYTICAL	PFOA	PFOS	
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.

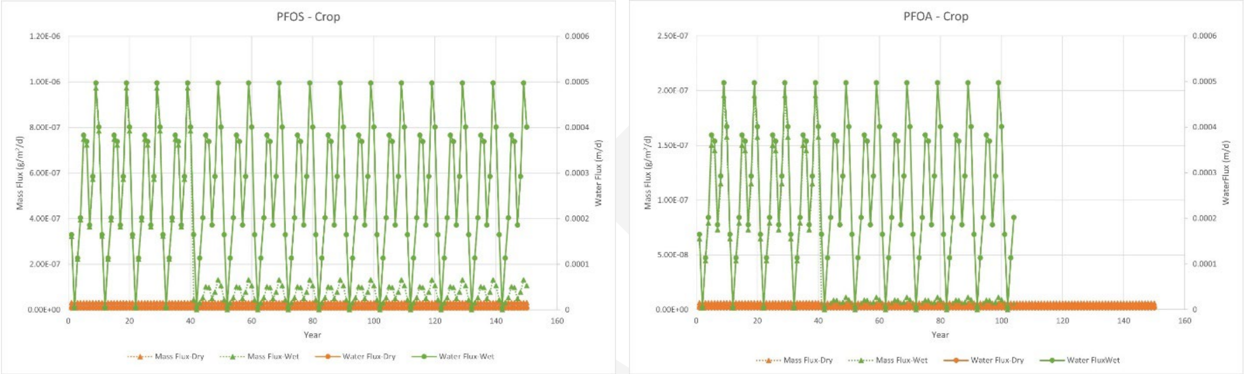


Figure C-3. Leachate flux for PFOS (left) and PFOA (right) for crop scenario.

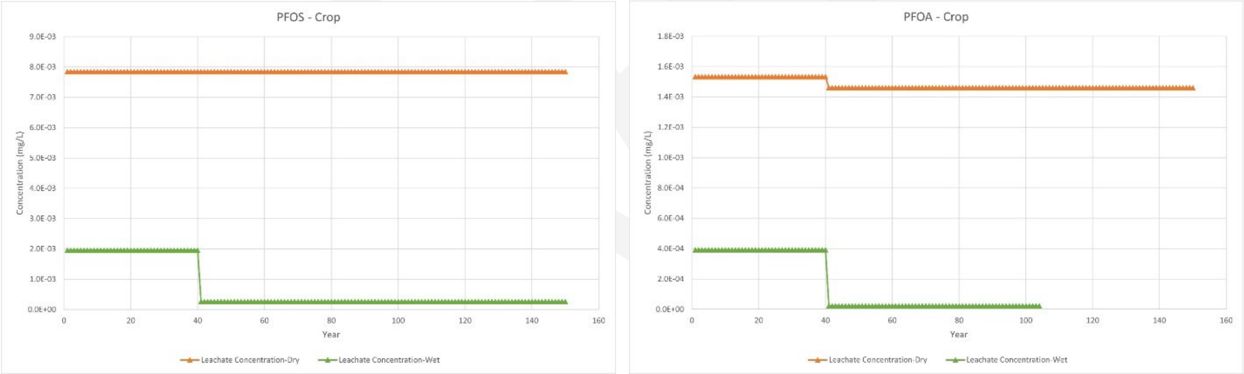


Figure C-4. Leachate concentration for PFOS (left) and PFOA (right) for crop scenario.

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.

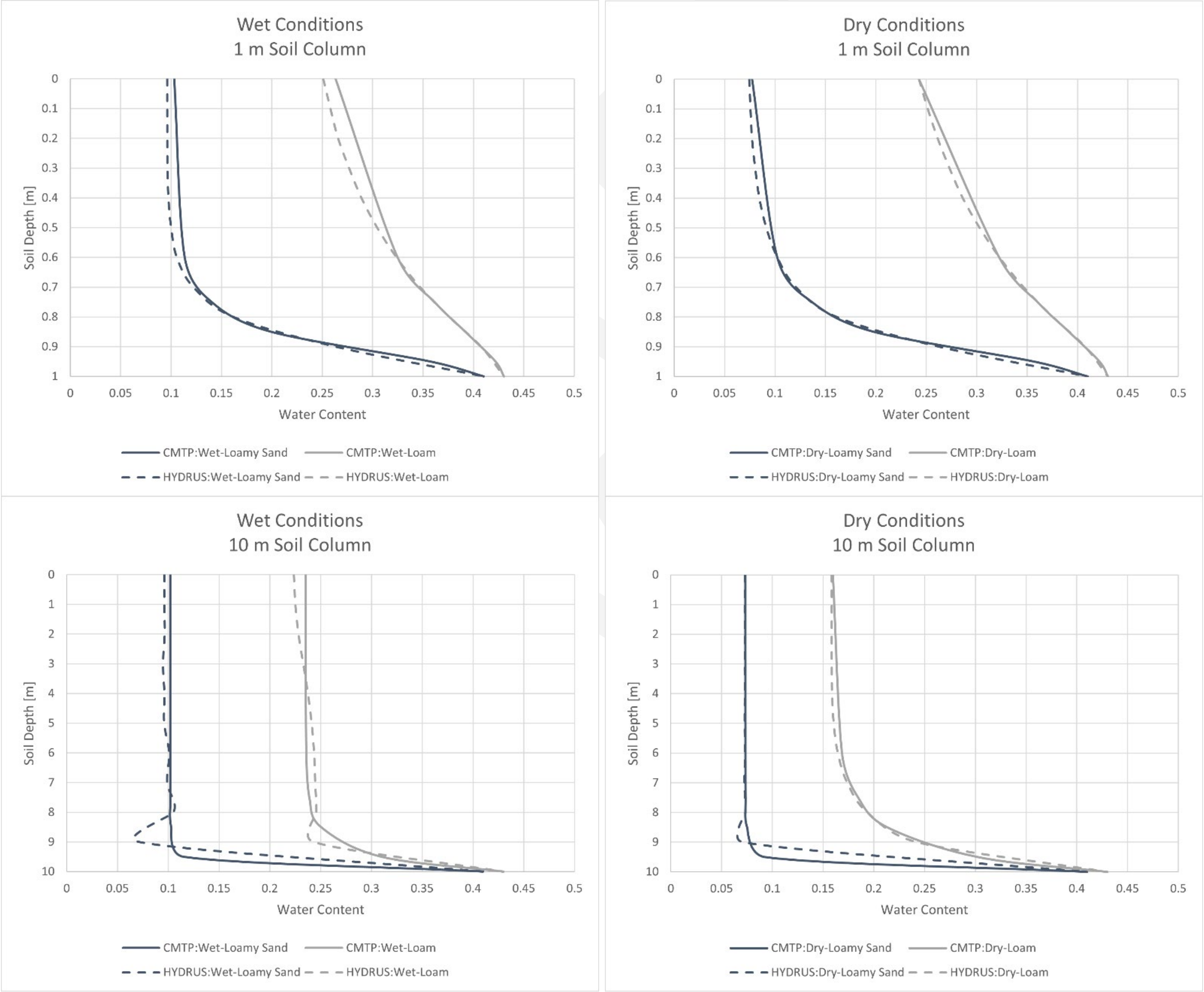


Figure C-5. **Water content profiles for 1-m (top) and 10-m (bottom) soil column for loam and loamy sand under wet (left) and dry (right) scenarios simulated using EPACMTP (solid lines) and HYDRUS (dashed lines).**

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).

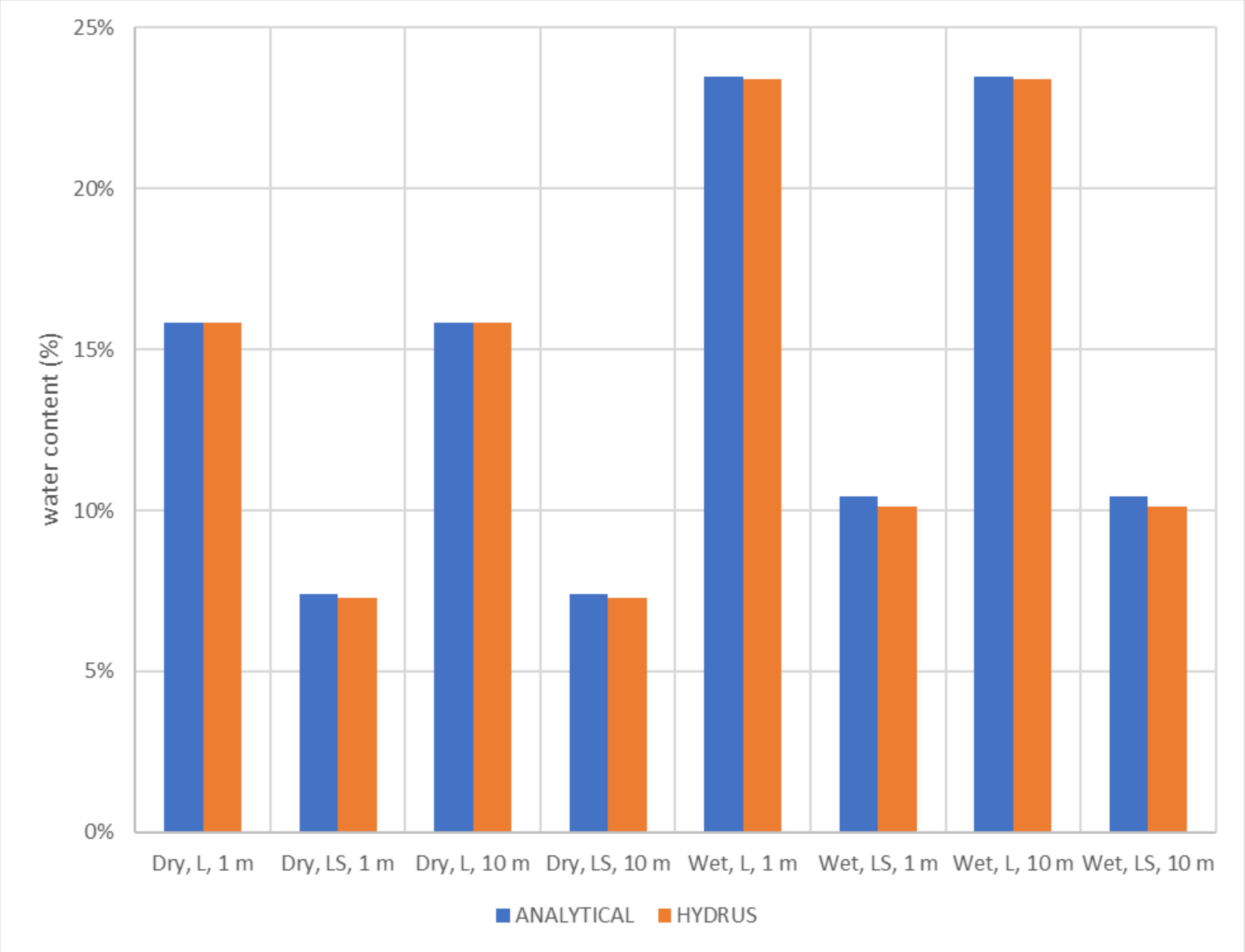


Figure C-6. Water content profiles for 1-m and 10-m soil column for loam and loamy sand under wet and dry scenarios simulated using HYDRUS and ANALYTICAL models.

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.

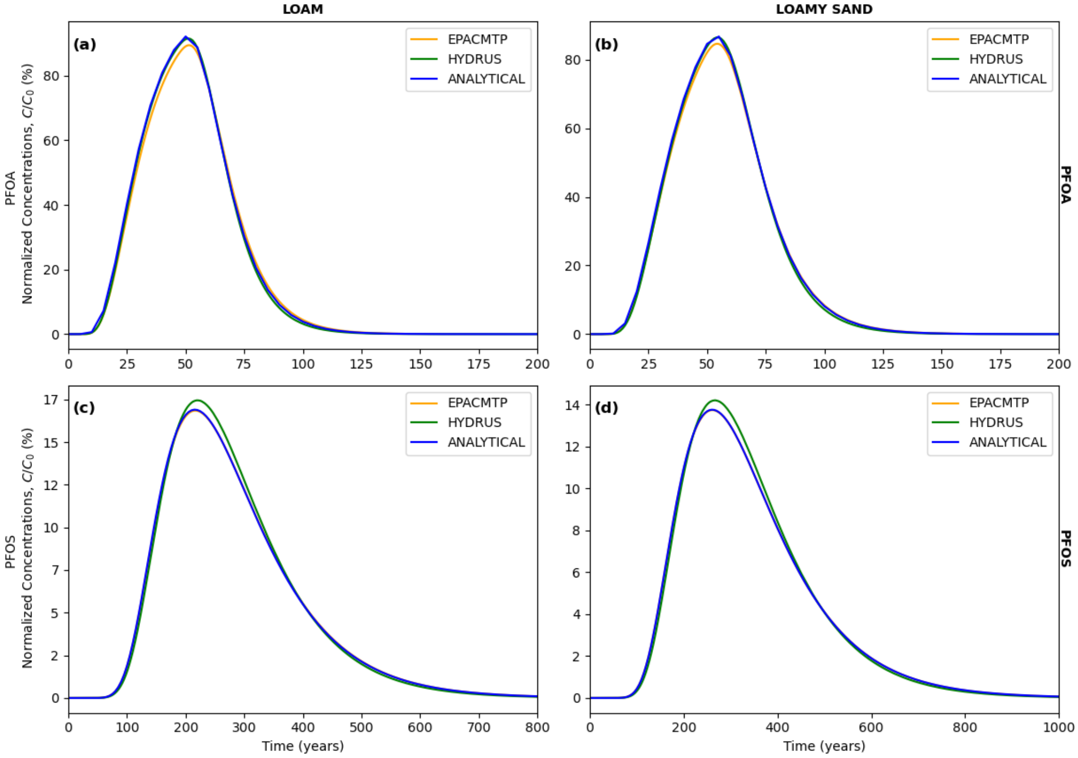


Figure C-7. Breakthrough curves at a 1-m water table depth in a wet environment using linear sorption.

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.

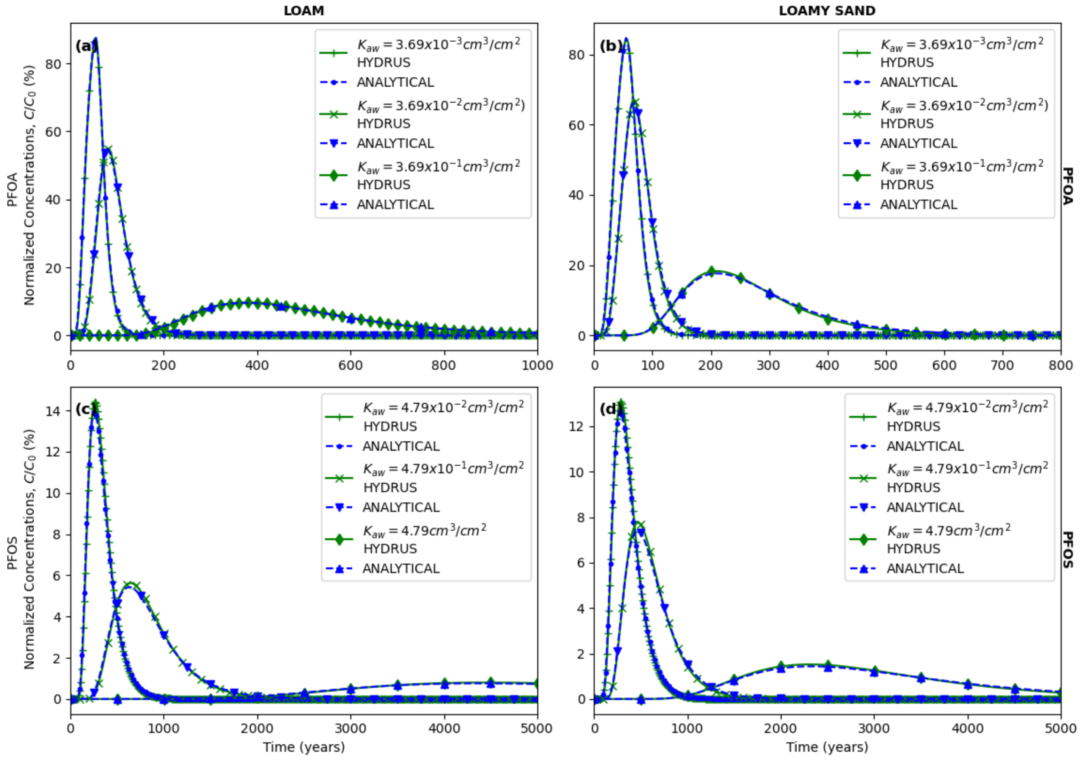


Figure C-8. Breakthrough curves at a 1-m water table depth in a wet environment using linear sorption and AWI effects for three values of Kaw (Silva et al., 2020).

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.

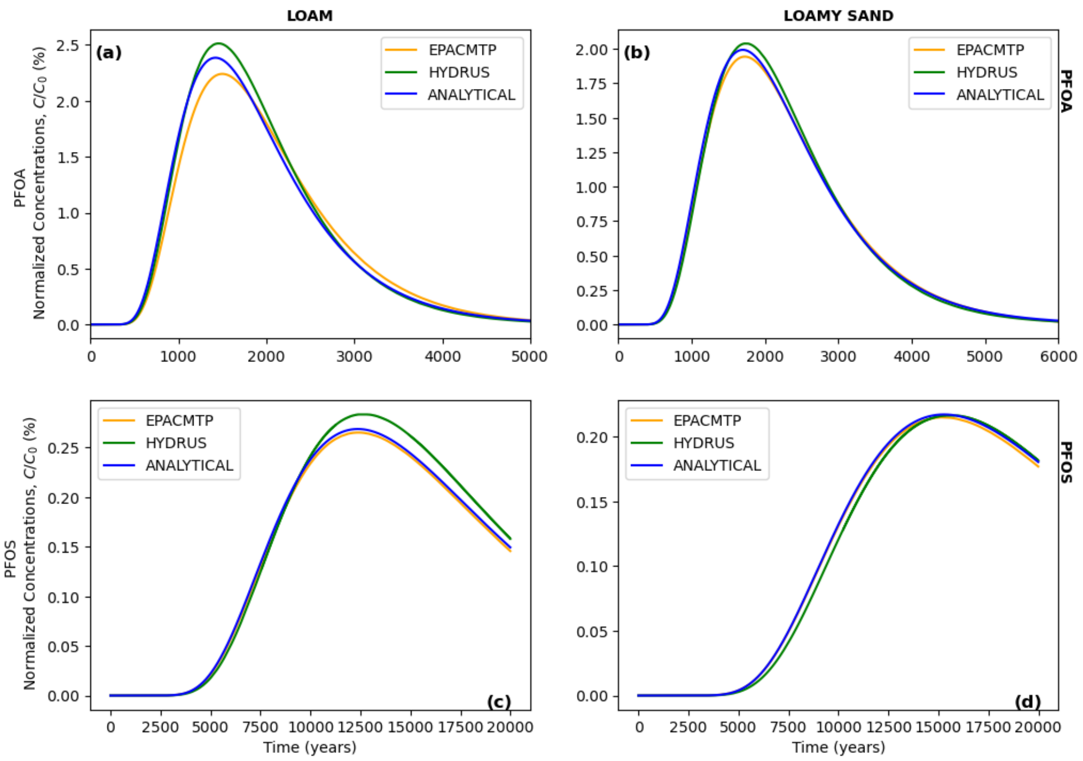


Figure C-9. Breakthrough curves at a 1-m water table depth in a dry using linear sorption.

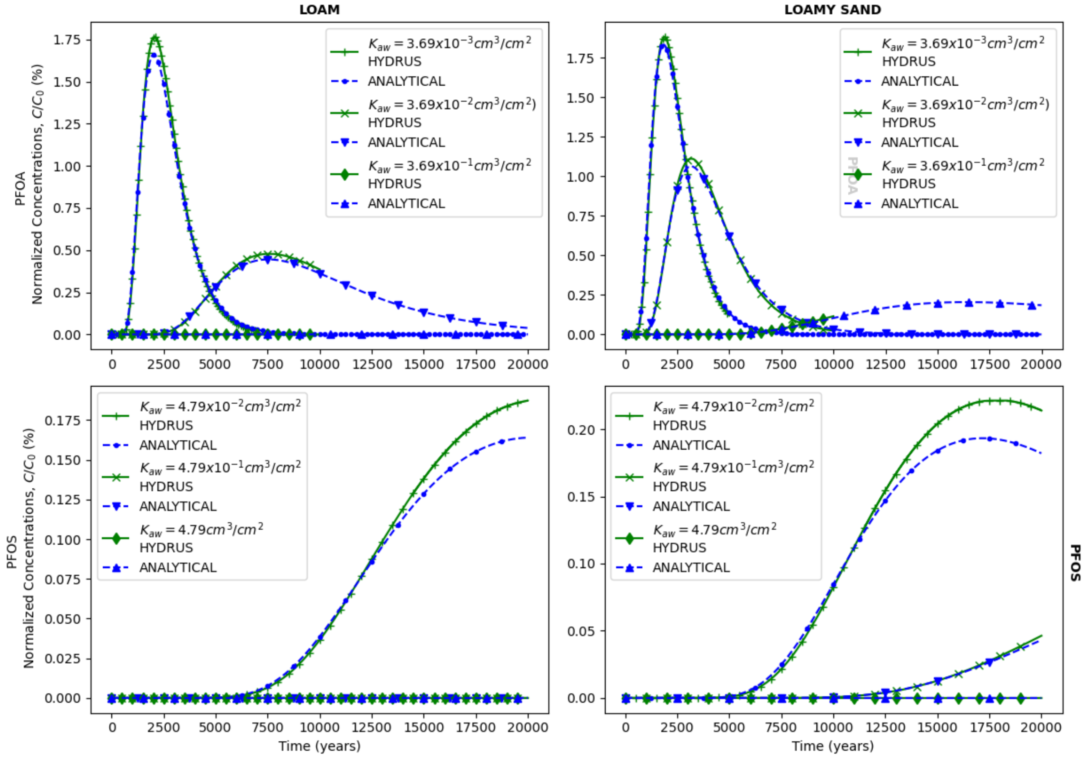


Figure C-10. Breakthrough curves at a 1-m water table depth in a dry environment using linear sorption and AWI effects for three values of Kaw (Silva et al., 2020).

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.

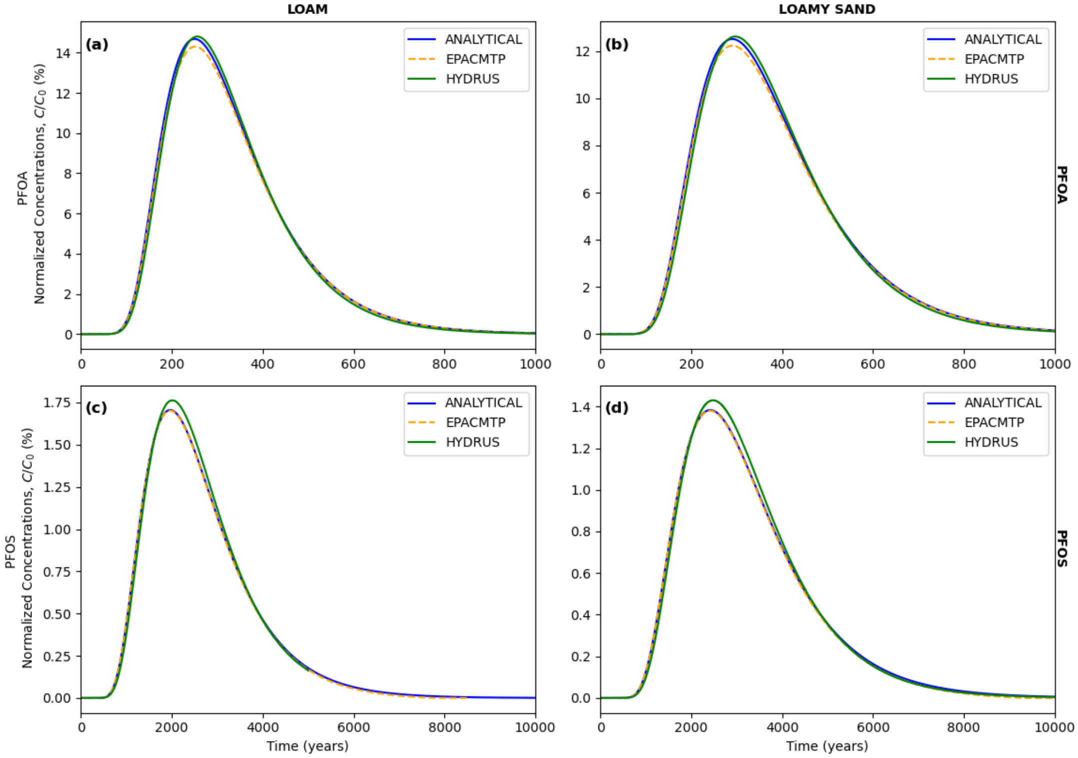


Figure C-11. Breakthrough curves at a 10-m water table depth in a wet environment using linear sorption.

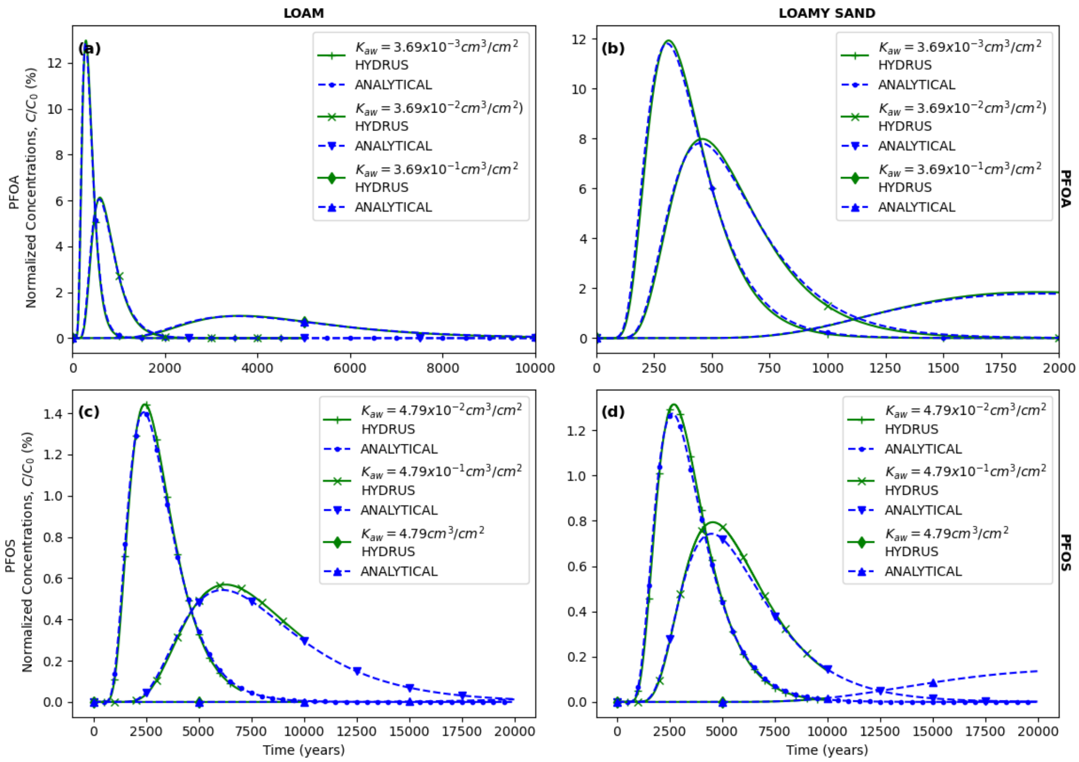


Figure C-12. Breakthrough curves at a 10-m water table depth in a wet environment using linear sorption and AWI effects for three values of Kaw (Silva et al., 2020).

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.

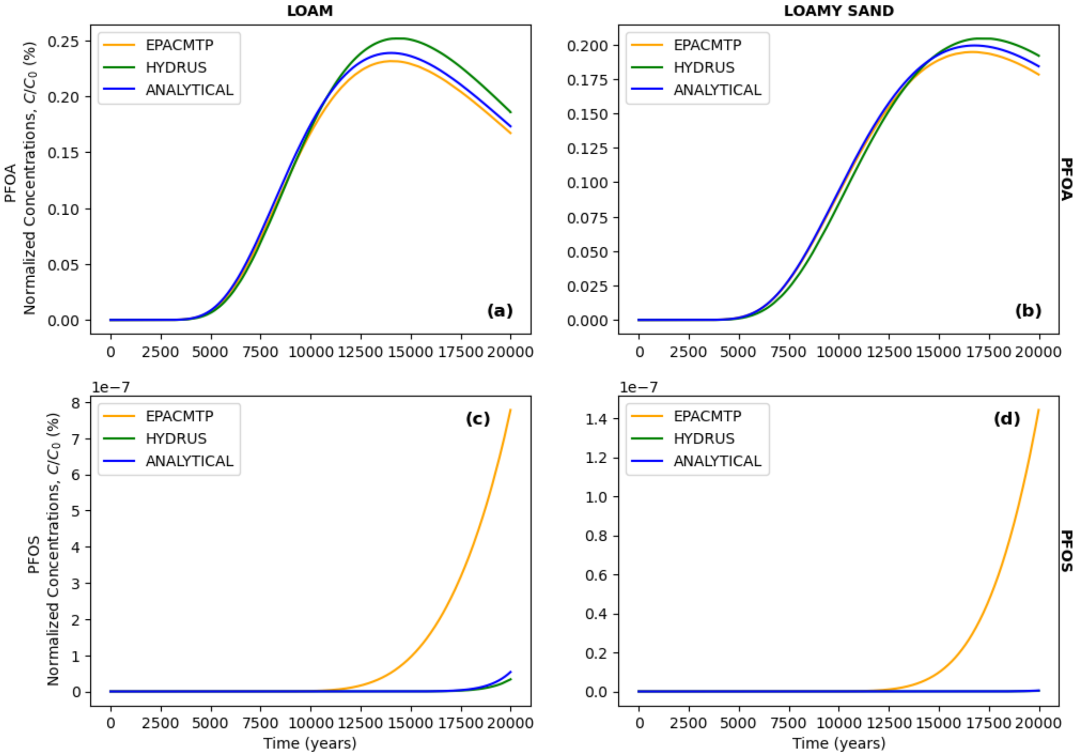


Figure C-13. Breakthrough curves at a 10-m water table depth in a dry environment using linear sorption.

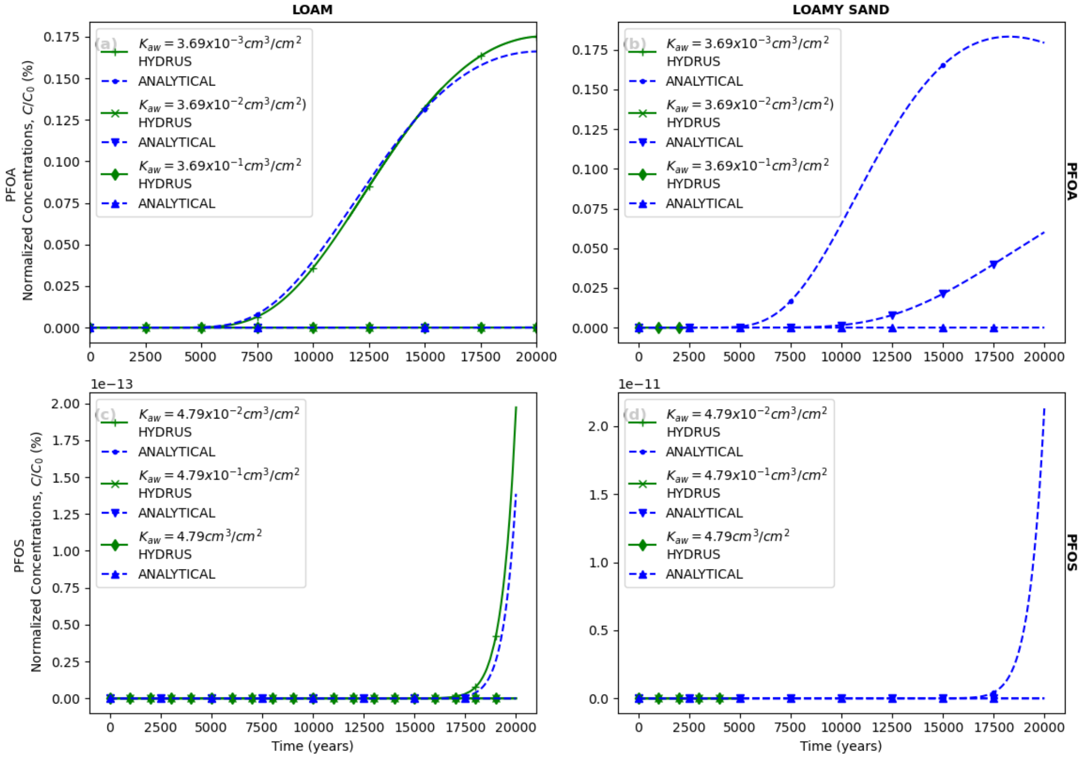


Figure C-14. Breakthrough curves at a 10-m water table depth in a dry environment using linear sorption and AWI effects for three values of Kaw (Silva et al., 2020).

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 $\mathrm{K_d}$ or $\mathrm{K_{oc}}$, 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 $\mathrm{K_{oc}}$ 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 $\mathrm{K_d}$. This behavior is reasonable as log $\mathrm{K_d}$ additionally reflects the effects of organic carbon (OC) variability in various matrices, as well as other parameters (*e.g.*, pH), whereas log $\mathrm{K_{oc}}$ 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 $\mathrm{K_{oc}}$ 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.

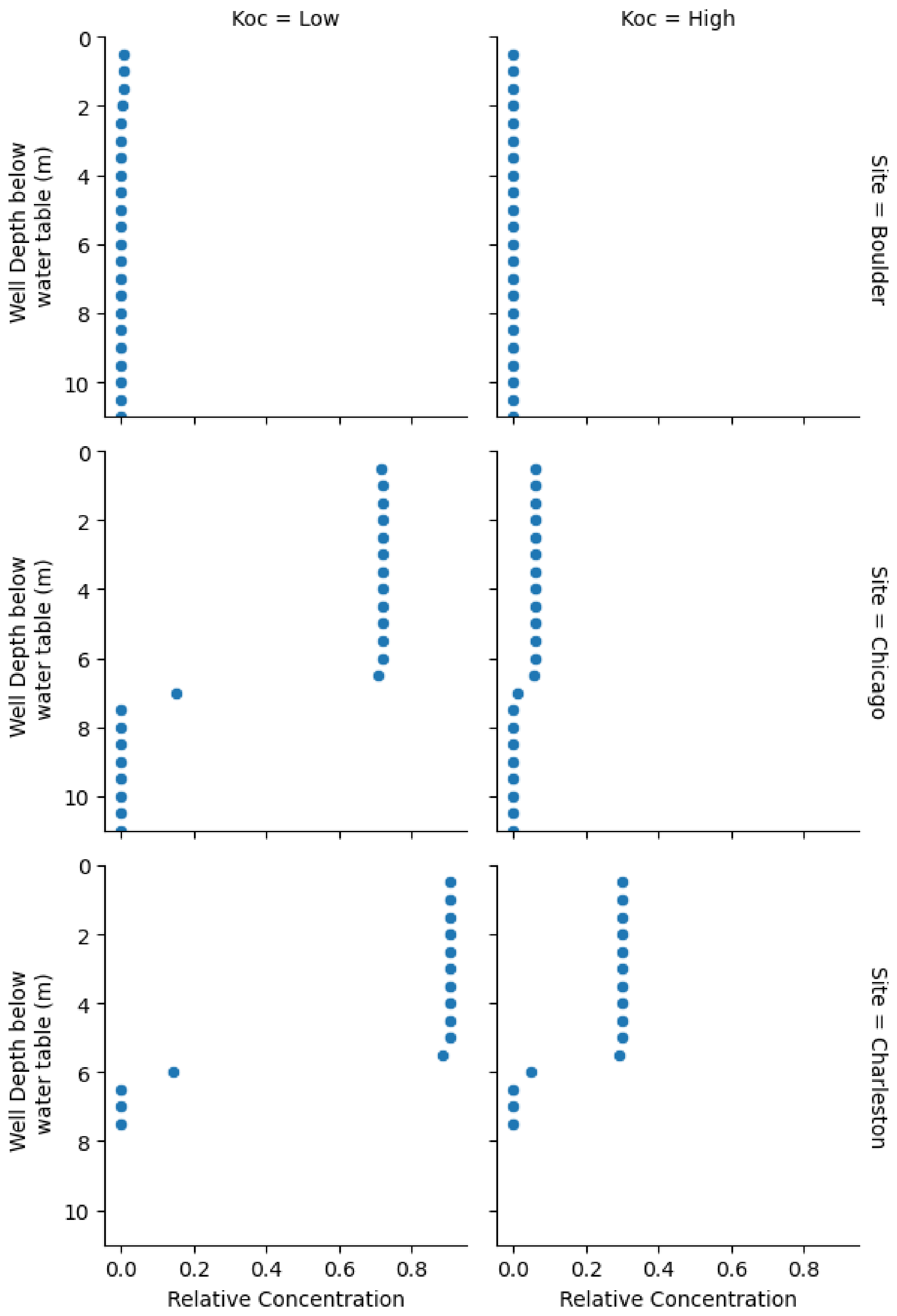


Figure C-15. Well depth below water table (m) vs. relative PFOA concentrations *for point observations (solid circles) for CROP, low Koc (left panels) and high Koc (right panels) at a well located 5 meters away from edge of field.*

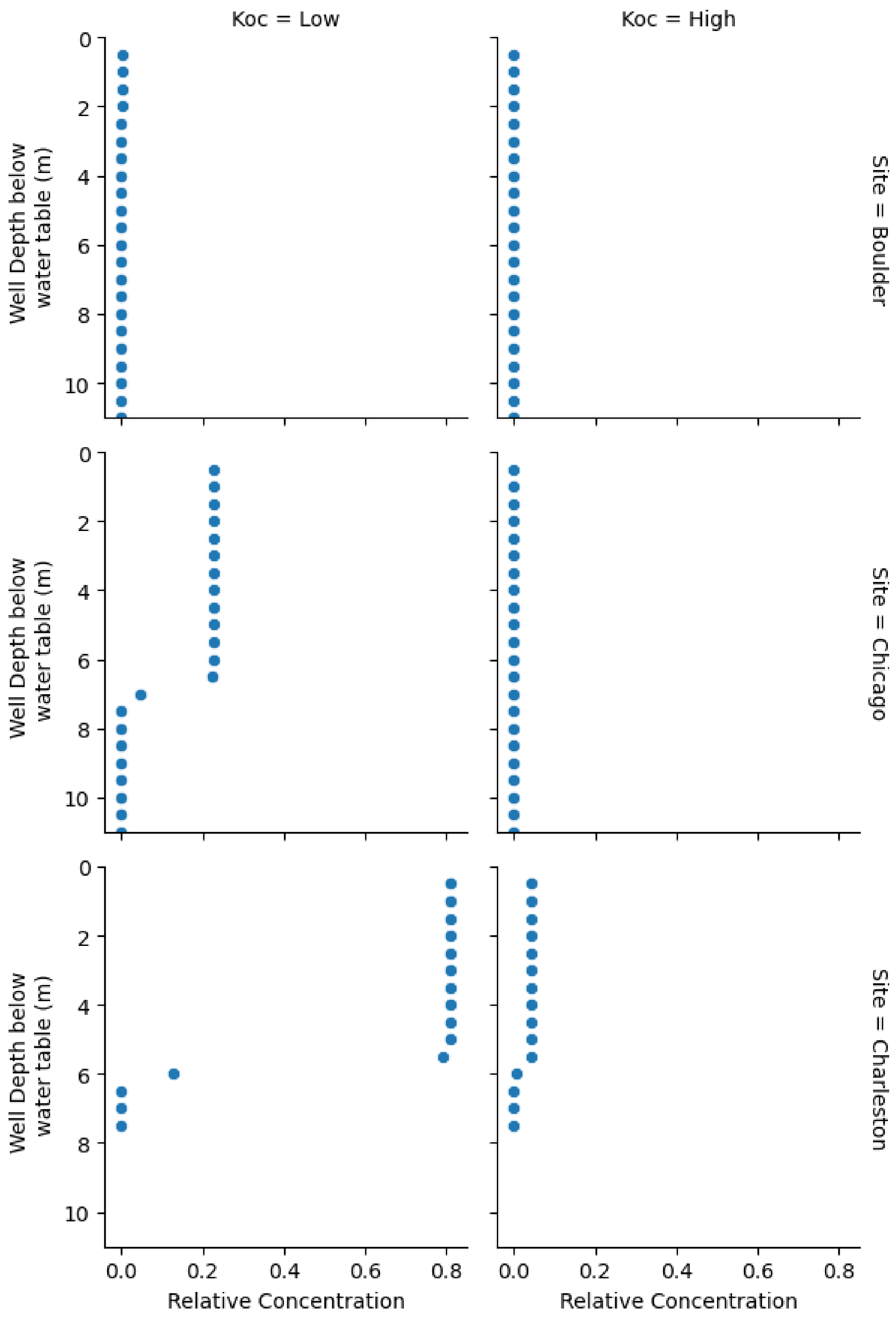


Figure C-16. Well depth below water table (m) vs. relative PFOS concentrations *for point observations (solid circles) for CROP, low Koc (left panels) and high Koc (right panels) at a well located 5 meters away from edge of field.*

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).

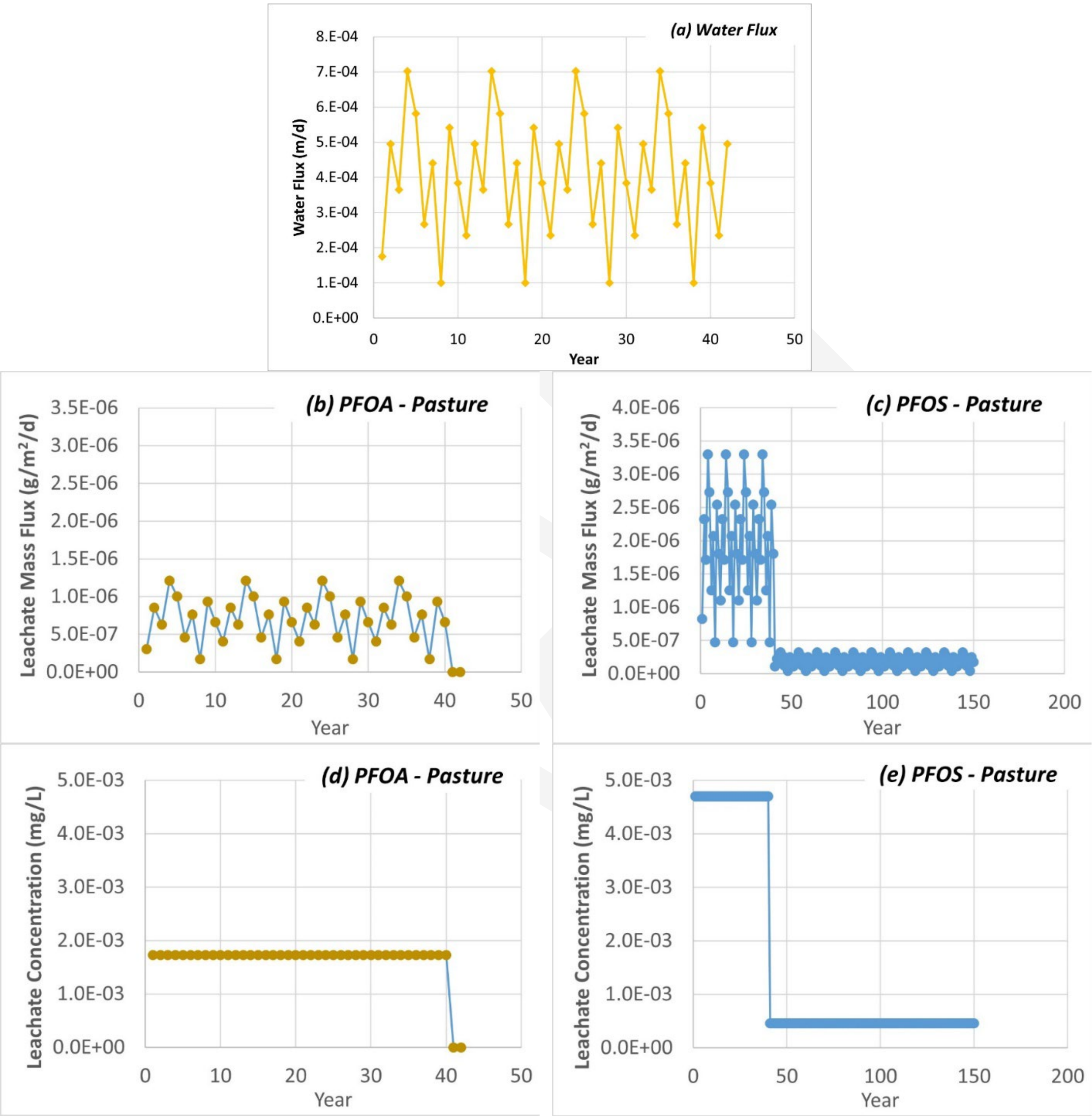


Figure C-17. Simulated time series of PFOA and PFOS mass and water fluxes generated for the pasture LAU scenario in Chicago, representing moderate meteorological conditions. (a) infiltrating water fluxes; (b) and (c) PFOA and PFOS mass fluxes leaching to the subsurface; and (d) and (e) PFOA and PFOS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

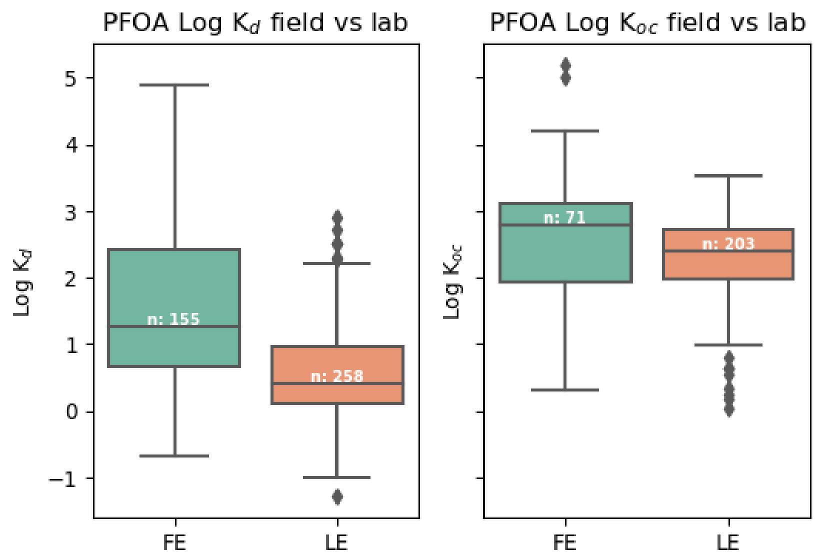
[a] Data reported by Tokranov et al. (2023).

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

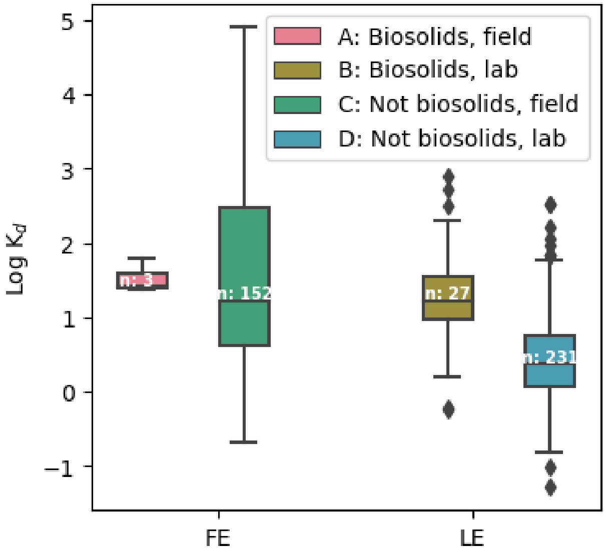
A significant finding for the purposes of identifying representative values of these parameters for modeling is that reported log K_d and log K_{oc} values span more than four orders of magnitude (**Figure C-18**). For log K_d, the median value from field data is greater than the median for laboratory 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.



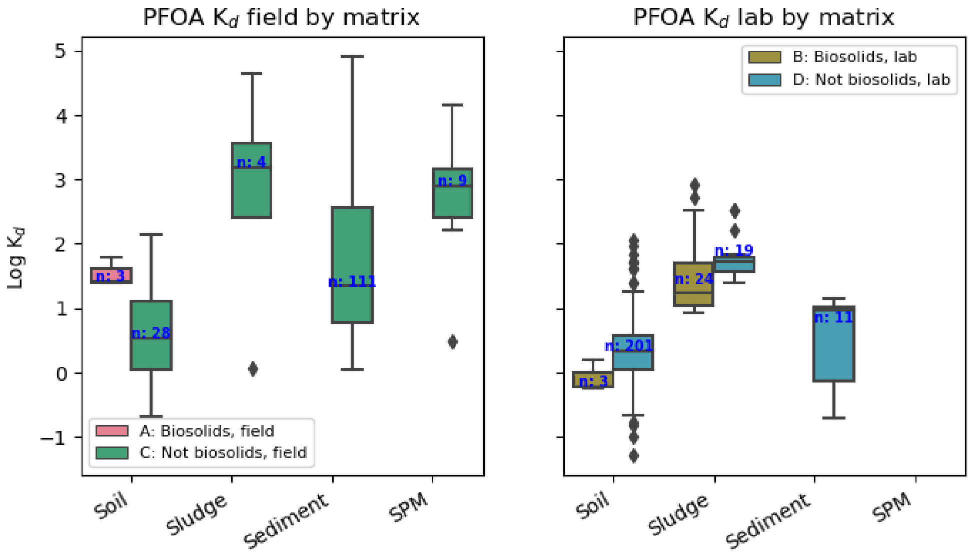
Shaded portion represents the interquartile range (IQR); whisker ends are quartile ±1.5xIQR; diamonds are outliers.
FE=Field, LE=lab

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.

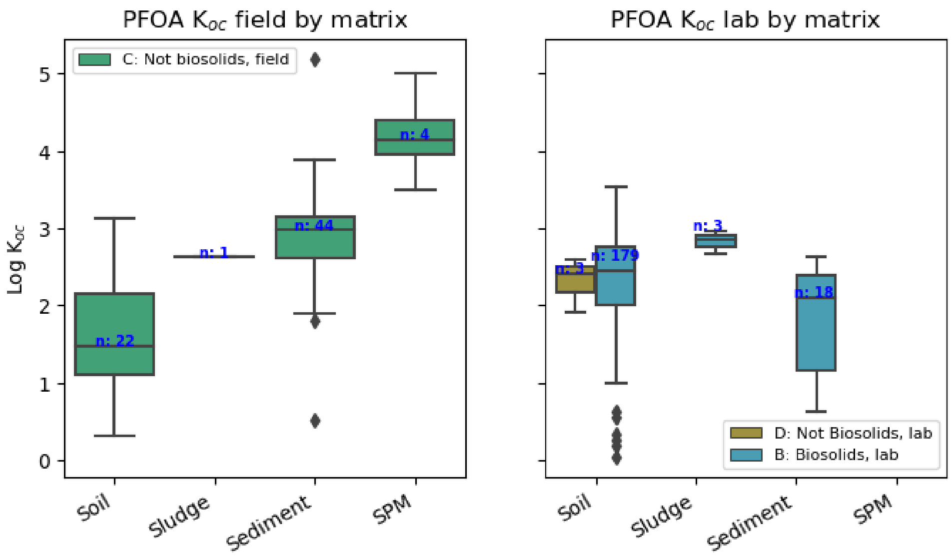


Shaded portion represents the interquartile range (IQR); whisker ends are quartile ±1.5xIQR; diamonds are outliers. SPM = suspended particulate matter.

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).

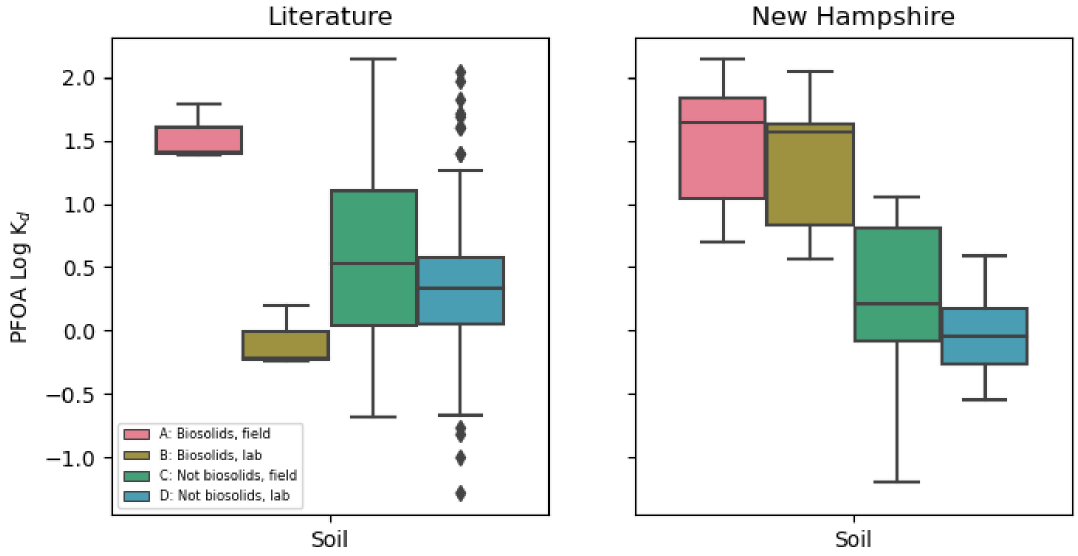
Reported log K_{oc} values for sludge are considerably less variable (log K_{oc} approximately 2.6 to 3). Values for SPM are more variable than sludge and somewhat less variable than soil and sediment. The SPM values are considerably higher than other samples of the same type (*i.e.*, field) but are 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.

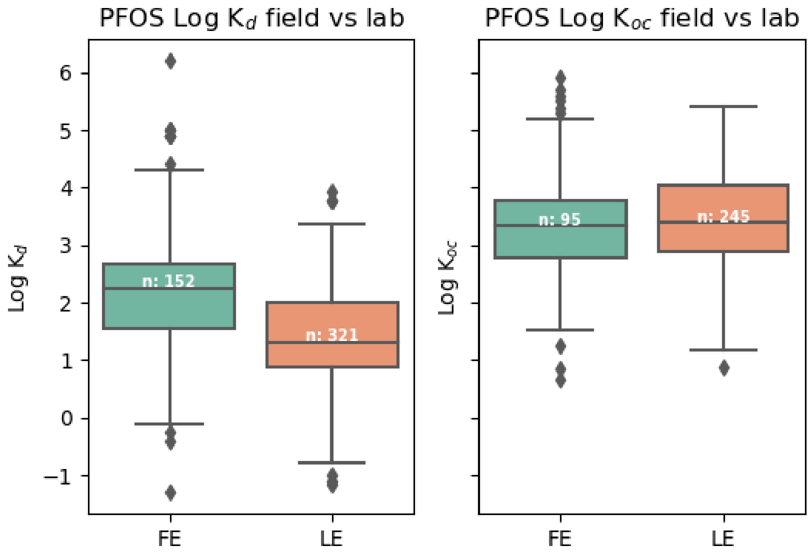


Shaded portion represents the interquartile range (IQR); whisker ends are quartile ±1.5xIQR; diamonds are outliers.

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.

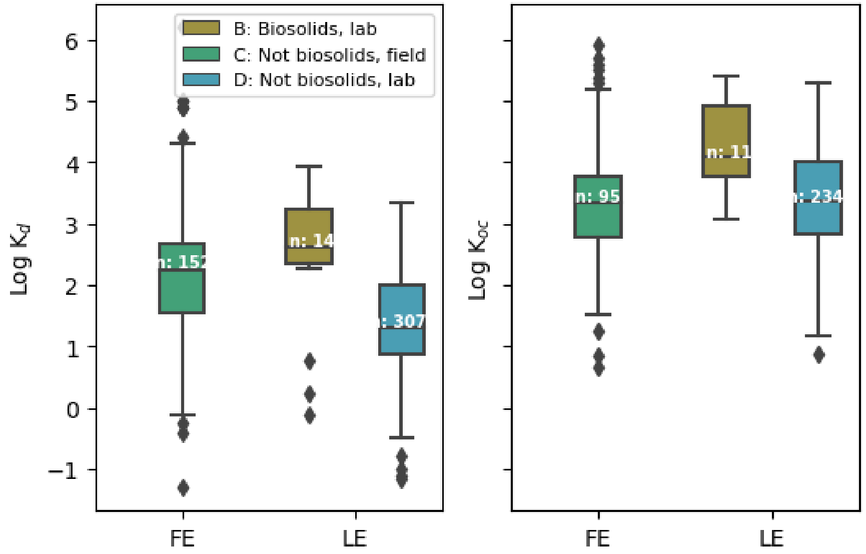
Figure C-23. Boxplots of log K_d (left) and log K_{oc} (right) values for PFOS: field (FE) vs. laboratory (LE) studies.

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}).

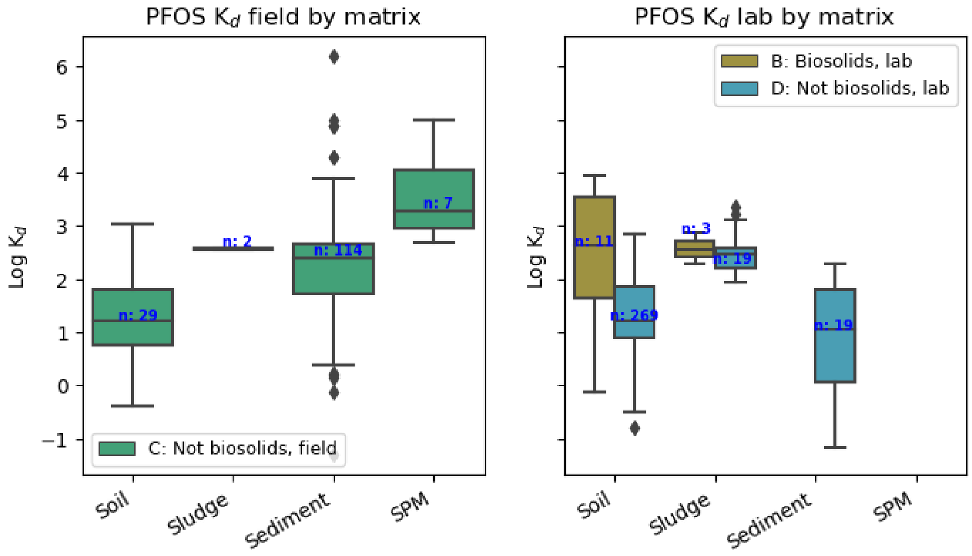
As shown in **Figure C-25**, the overall range of log K_d values in soil (approximately -0.8 to 3.9) is narrower than the range of log K_d values in sediment (approximately –1.3 to 6.2). Note that the soil values are all lab studies, as no field studies (in biosolids or otherwise) were identified for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

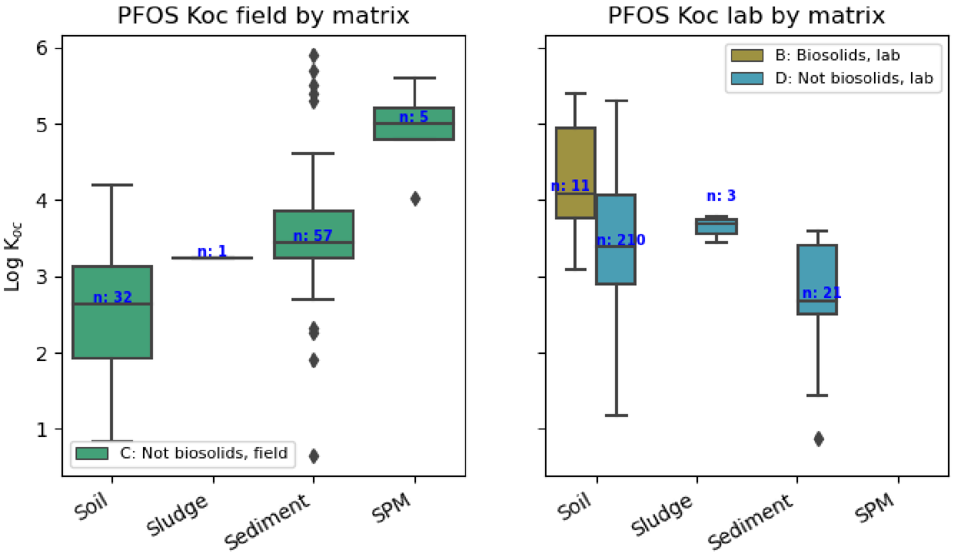


Shaded portion represents the interquartile range (IQR); whisker ends are quartile ±1.5xIQR; diamonds are outliers. SPM = suspended particulate matter.

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).

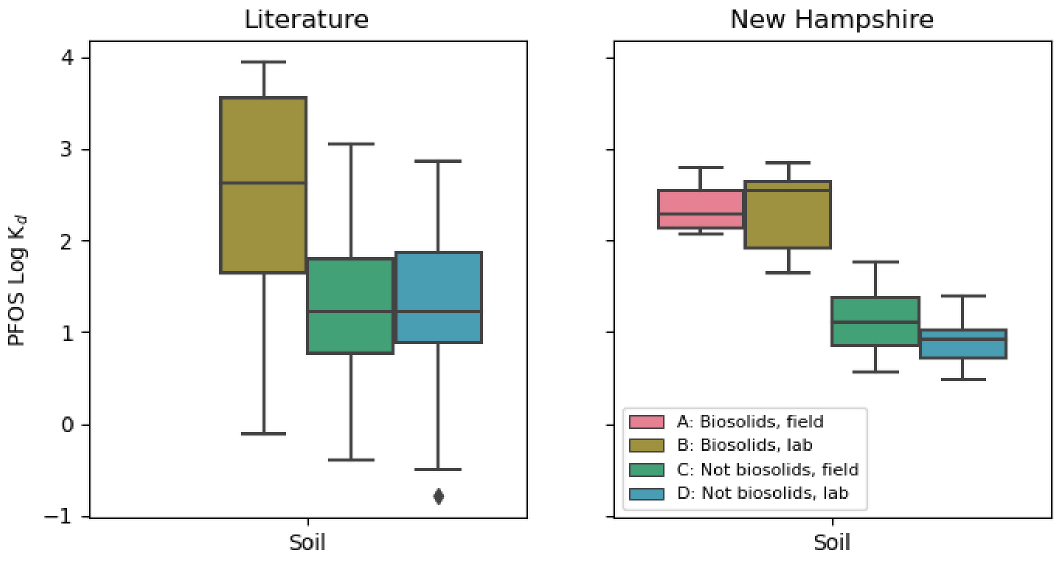
Compared to soils and sediments, log K_{oc} values reported for SPM, where available, are generally higher (Figure C-26). However, the number of studies reporting log K_d in SPM (n=5) are much fewer than soils (n=253) or sediments (n =78).



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.

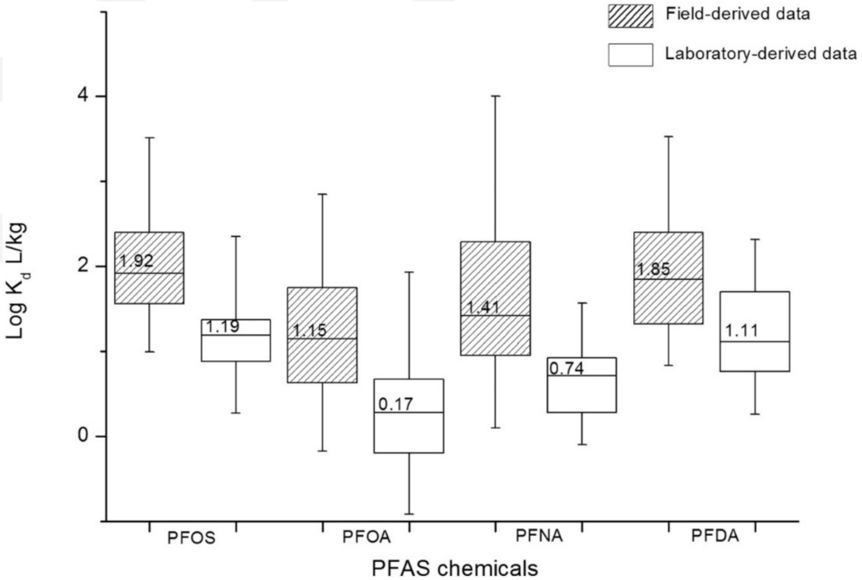


Shaded portion represents the interquartile range (IQR); whisker ends are quartile ±1.5xIQR; diamonds are outliers.

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.

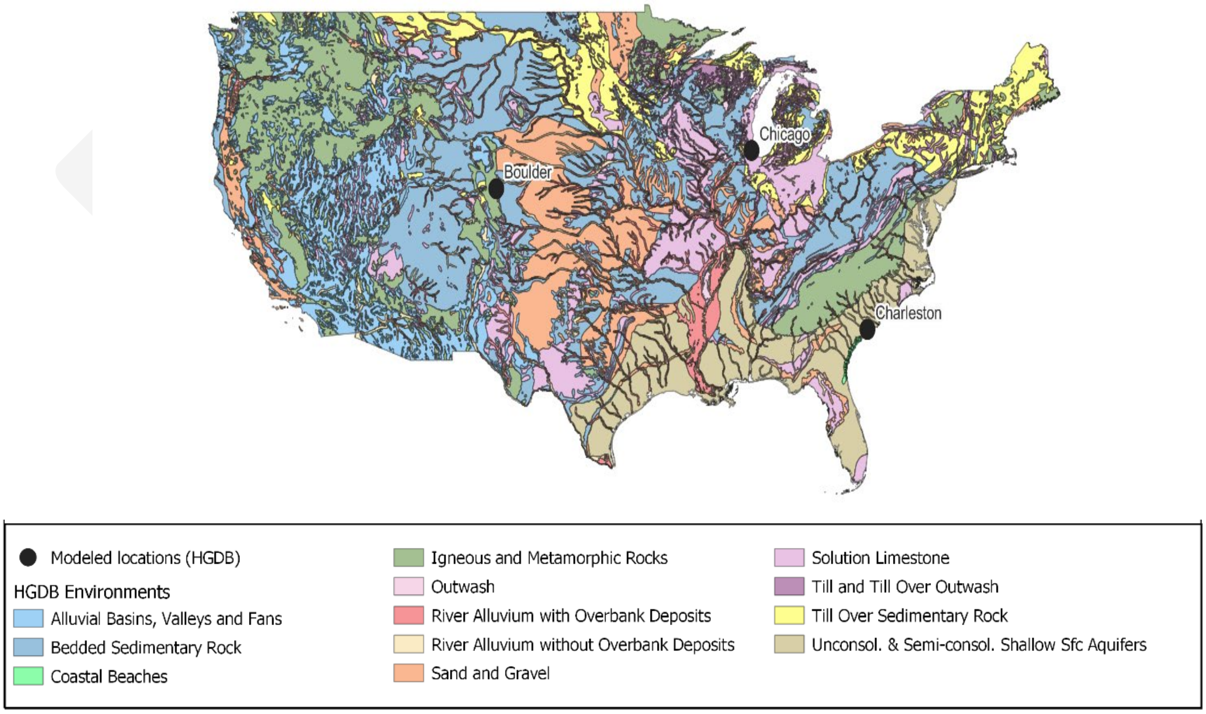

Figure C-29. National geographic map of the 12 hydrogeologic environments developed by EPA.

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)

	Low Koc			High Koc		
Scenario	**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

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[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.

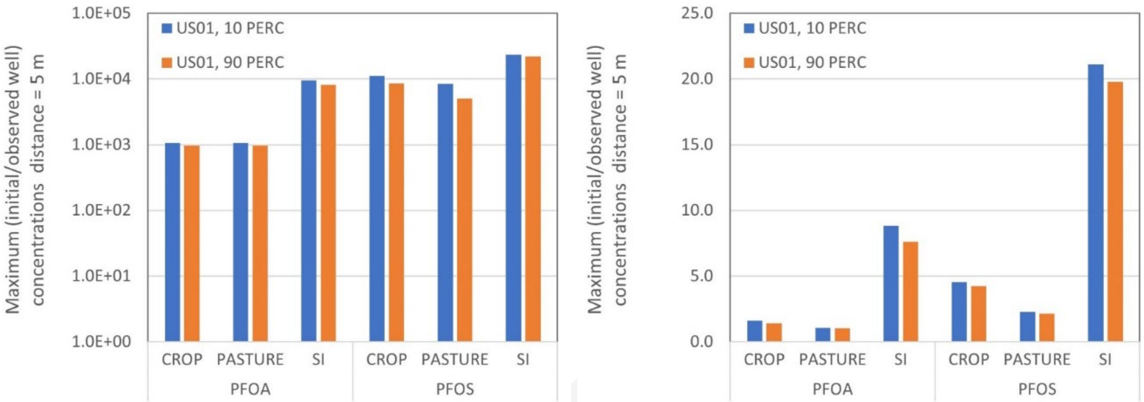


Figure D-1. Sensitivity to saturated hydraulic conductivity of the unsaturated zone for Boulder (dry climate, left) and Charleston, SC (wet climate, right).

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.

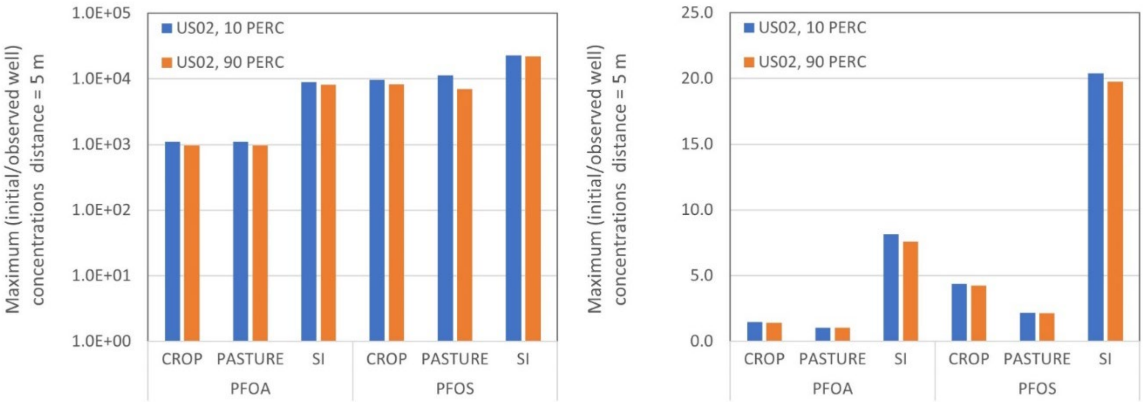


Figure D-2. Sensitivity to van Genuchten parameter alpha for Boulder (dry climate, left) and Charleston, SC (wet climate, right).

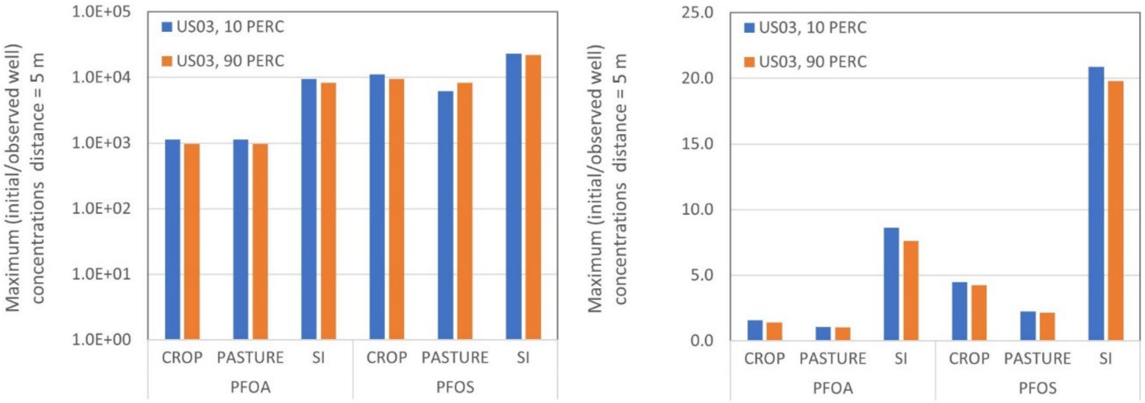


Figure D-3. Sensitivity to van Genuchten parameter beta for Boulder (dry climate, left) and Charleston, SC (wet climate, right).

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.

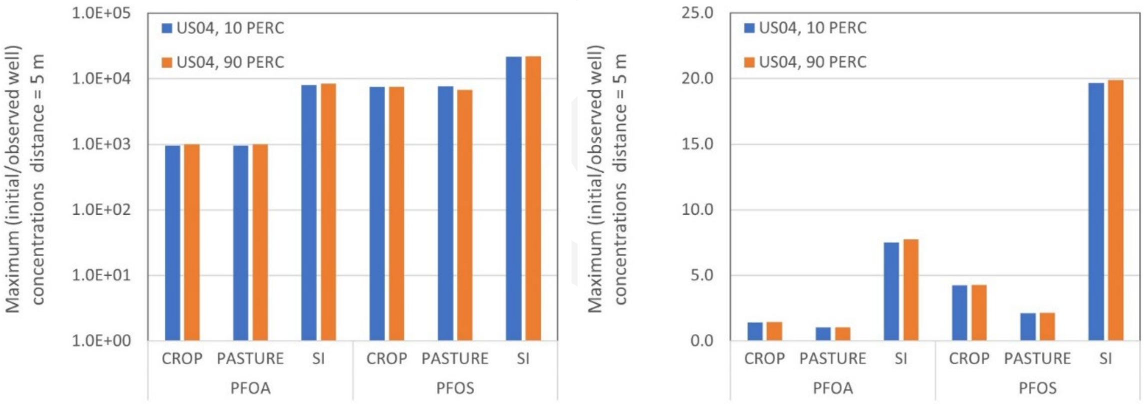


Figure D-4. Sensitivity to residual water content of the unsaturated zone for Boulder (dry climate, left) and Charleston, SC (wet climate, right).

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.

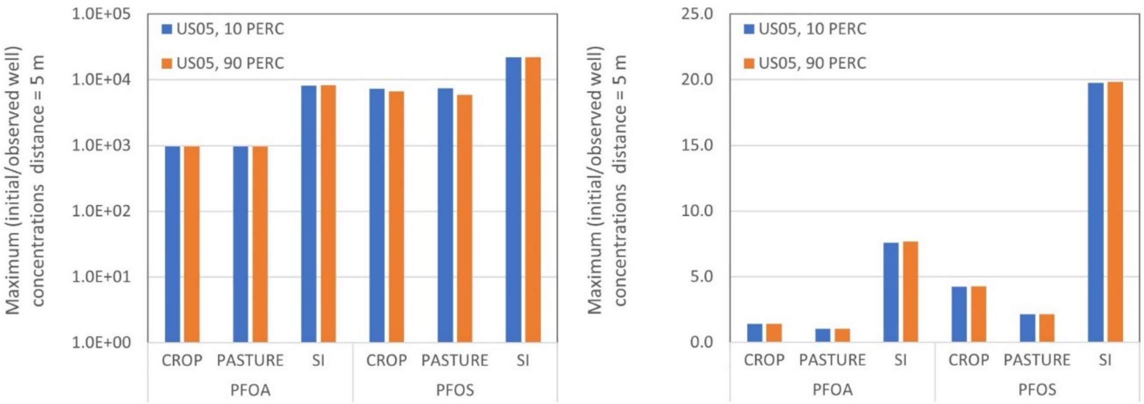
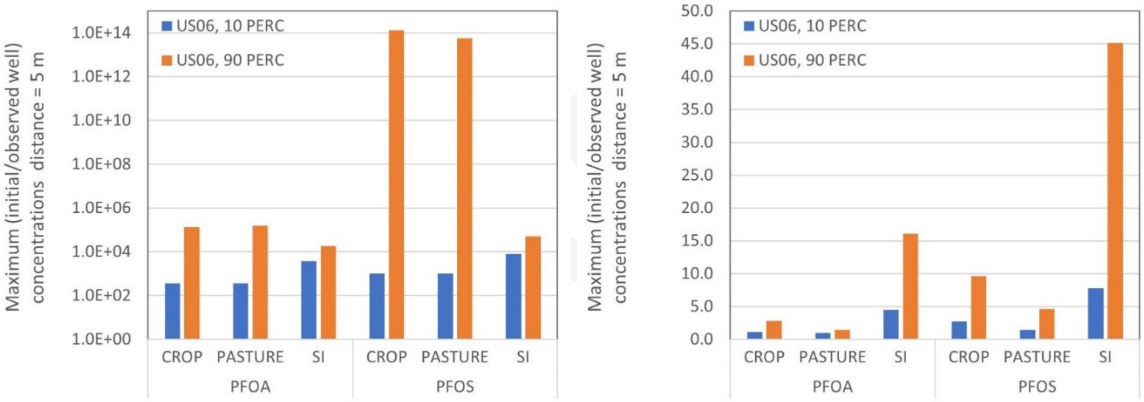


Figure D-5. Sensitivity to saturated water content of the unsaturated zone for Boulder (dry climate, left) and Charleston, SC (wet climate, right).

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.

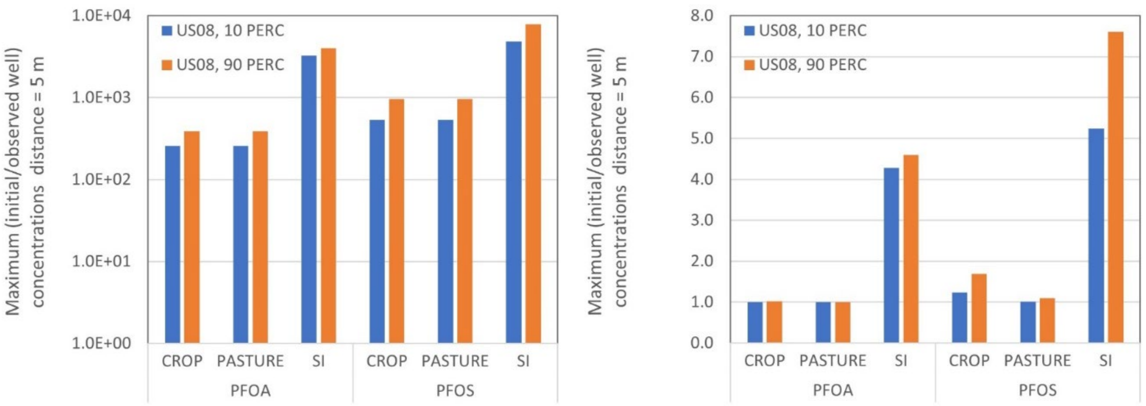


Figure D-7. Sensitivity to percent organic matter of the unsaturated zone for Boulder (dry climate, left) and Charleston, SC (wet climate, right).

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.

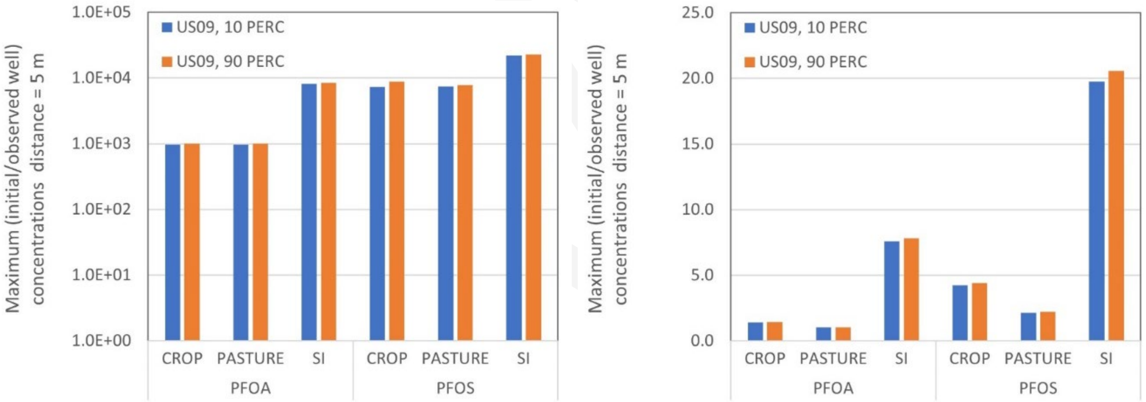


Figure D-8. Sensitivity to bulk density of the unsaturated zone for Boulder (dry climate, left) and Charleston, SC (wet climate, right).

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.

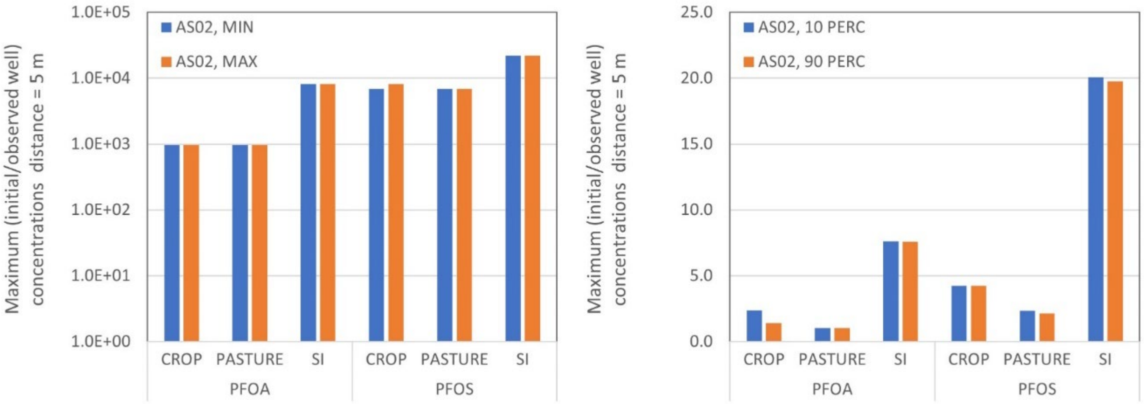


Figure D-9. Sensitivity to effective porosity of the unsaturated zone for Boulder (dry climate, left) and Charleston, SC (wet climate, right).

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.

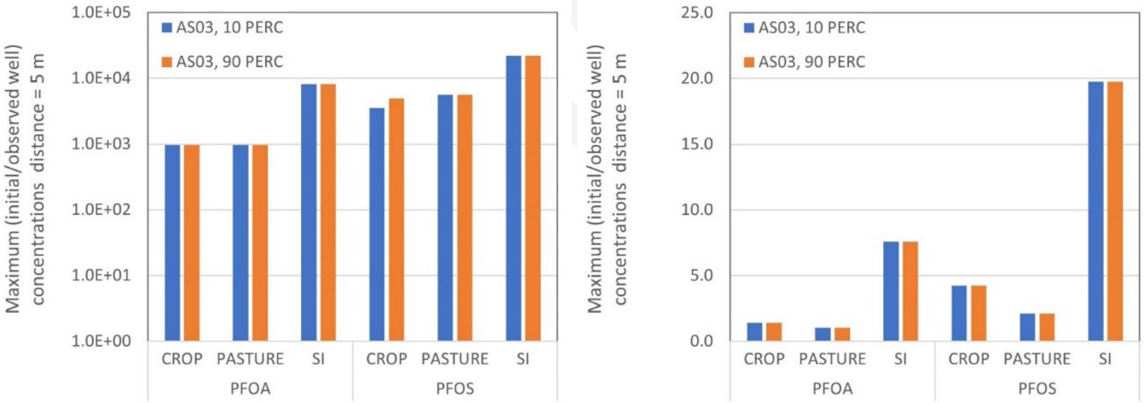
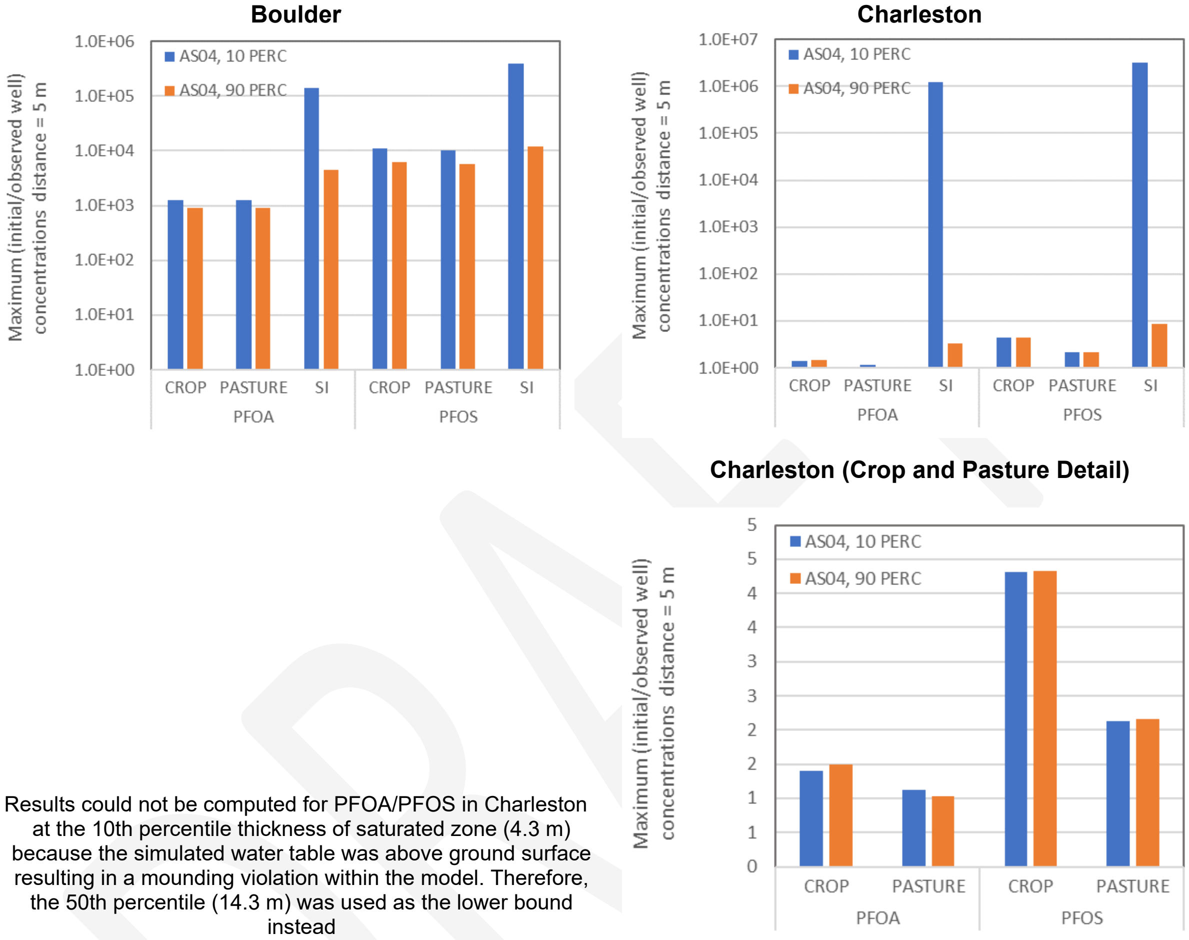


Figure D-10. Sensitivity to bulk density of the saturated zone for Boulder (dry climate, left) and Charleston, SC (wet climate, right).

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.

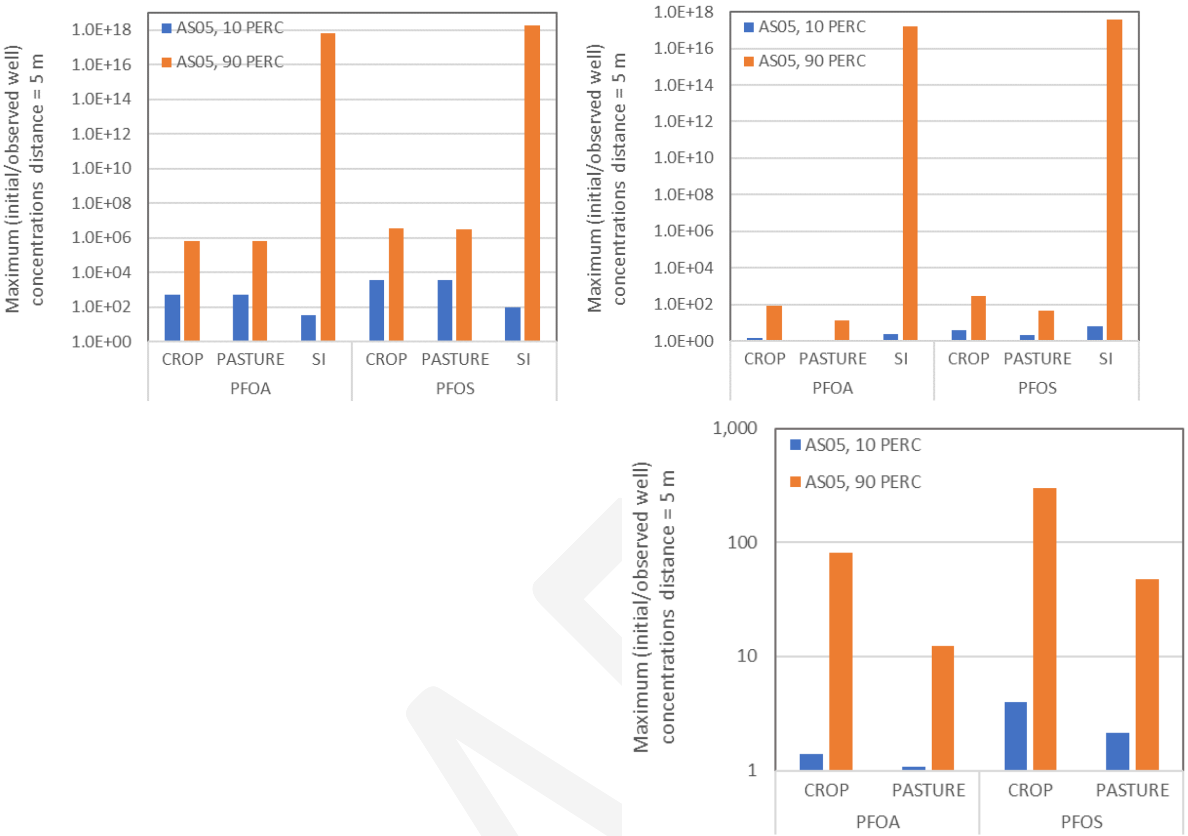


Figure D-12. Sensitivity to hydraulic conductivity of the saturated zone for Boulder (dry climate, left) and Charleston, SC (wet climate, right; top shows LAU crop and pasture and SI, lower shows detail for crop and pasture on a smaller y-axis scale).

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.

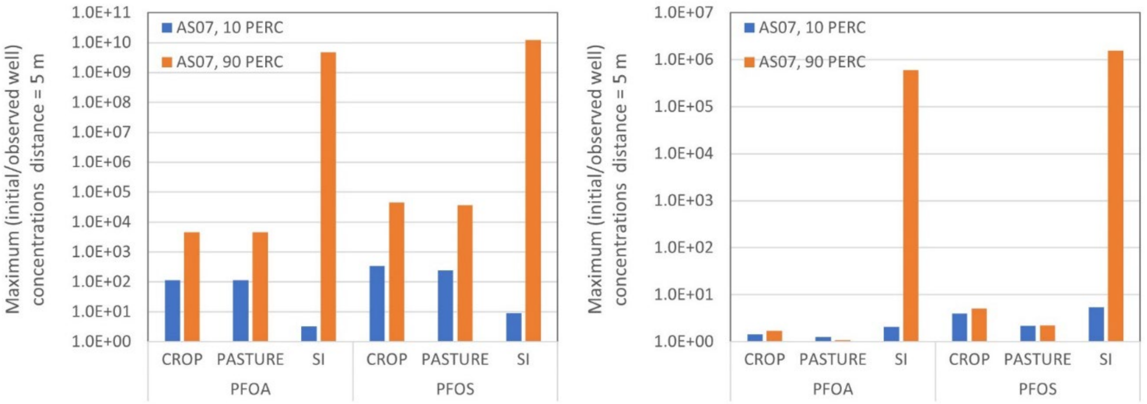


Figure D-13. Sensitivity to regional hydraulic gradient of the saturated zone for Boulder (dry climate, left) and Charleston, SC (wet climate, right).

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) = $f_{oc} \times k_{oc}$
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.

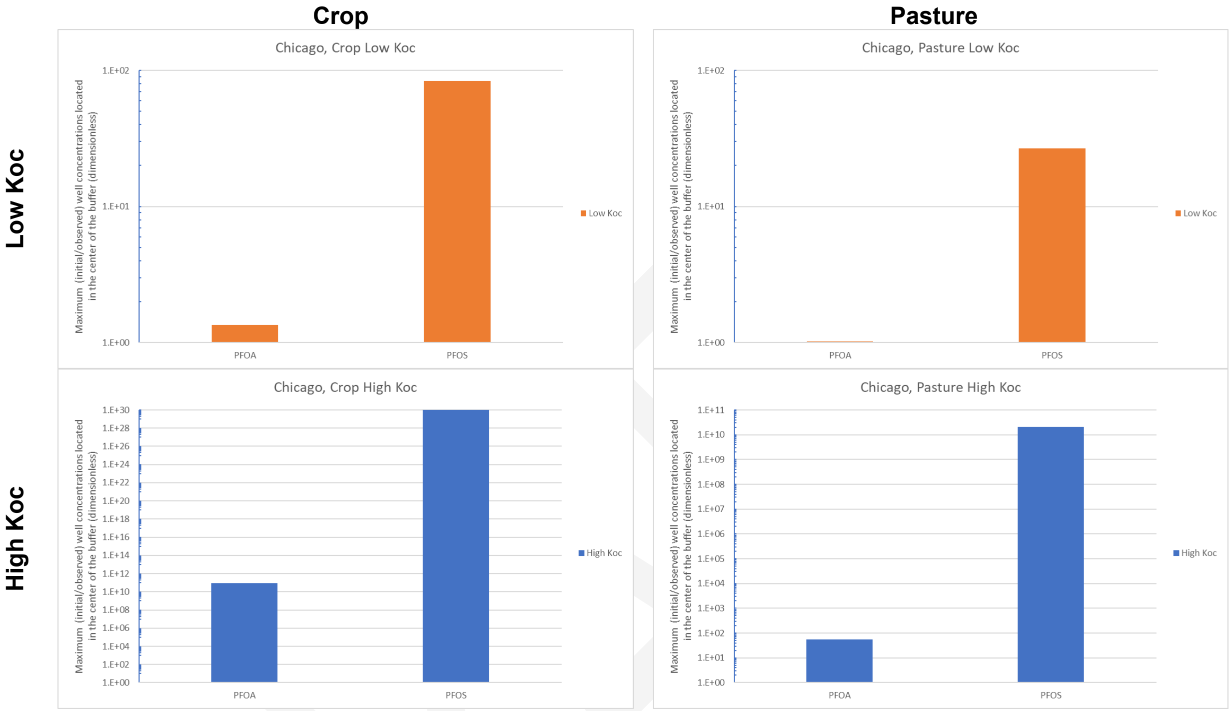


Figure D-14. Maximum leachate to well concentration ratio for land application unit for Chicago (moderate climate): crop (left) and pasture (right), low Koc (top) and high Koc (bottom).

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.

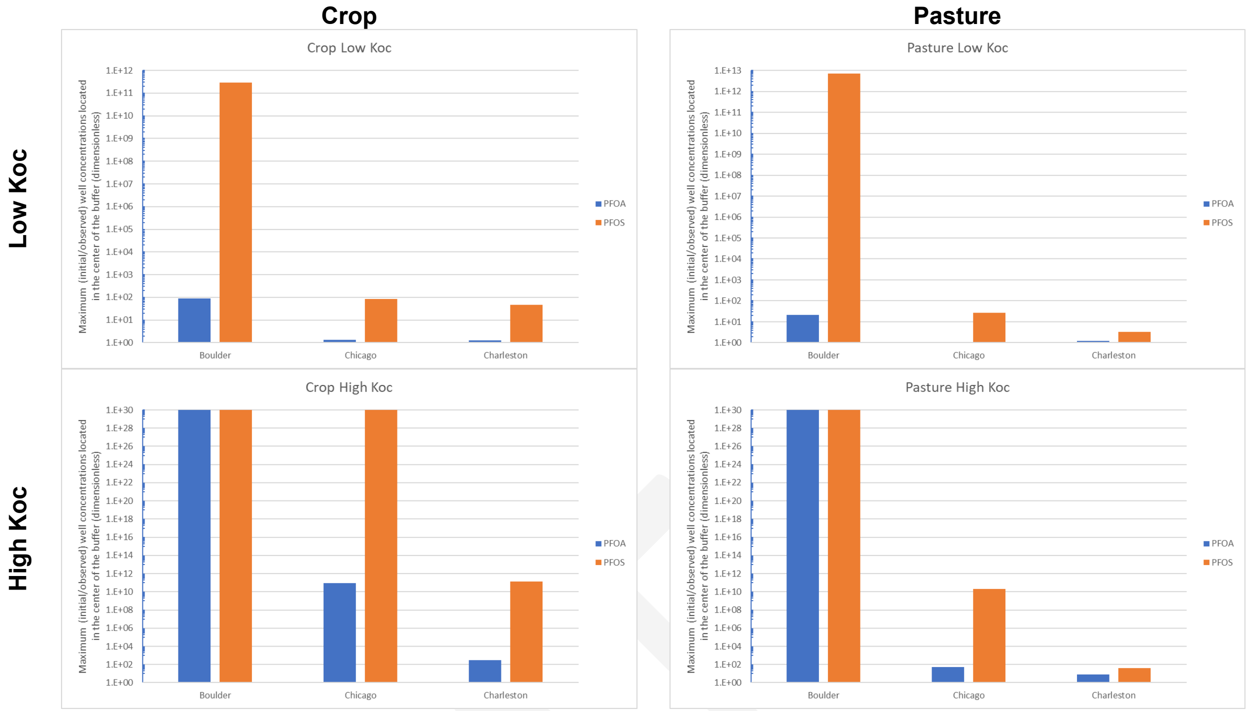


Figure D-15. Maximum leachate to well concentration ratio for land application unit by climate: crop (left) and pasture (right), low Koc (top) and high Koc (bottom).

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.

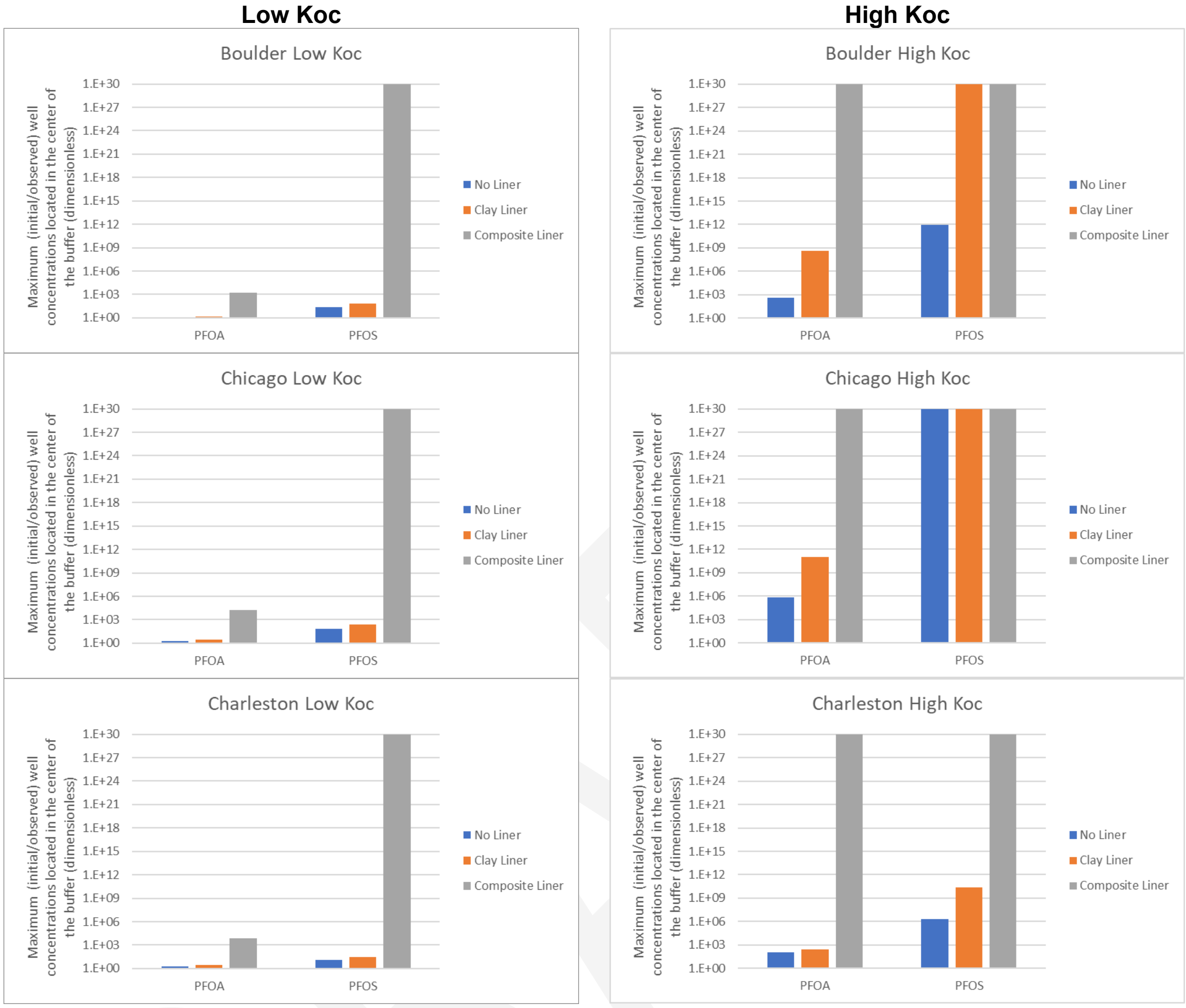


Figure D-16. Maximum leachate to well concentration ratio for surface disposal unit for low Koc (left column) and high Koc (right column) for Boulder (dry climate, top row), Chicago (moderate climate, middle row), and Charleston, SC (wet climate, bottom row).

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.

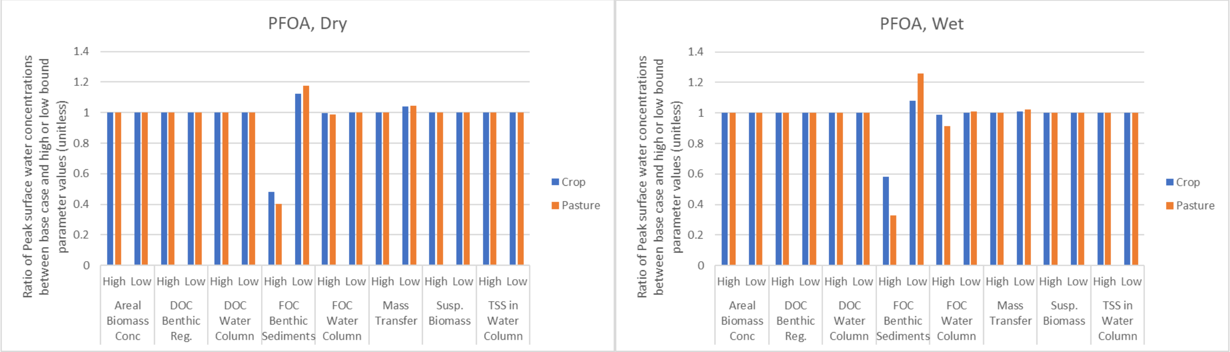


Figure D-17. Sensitivity of VVWM to media parameters for PFOA for Boulder (dry climate, left) and Charleston, SC (wet climate, right).

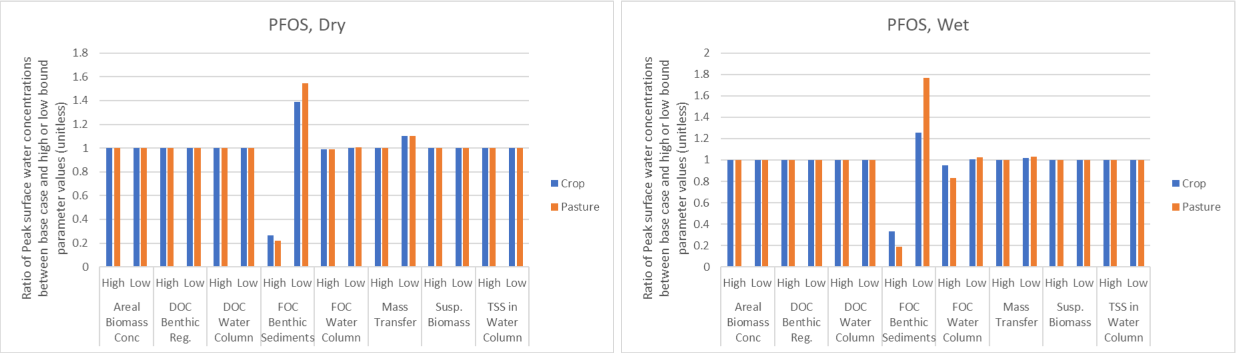


Figure D-18. Sensitivity of VVWM to media parameters for PFOS for Boulder (dry climate, left) and Charleston, SC (wet climate, right).

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.

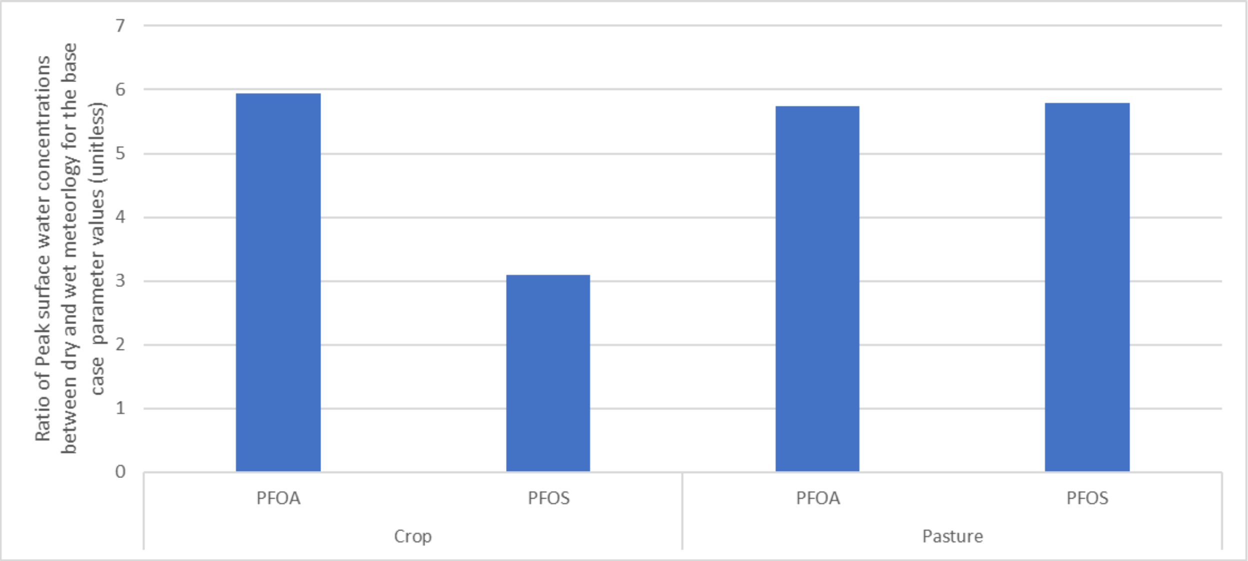


Figure D-19. Sensitivity of VVWM to meteorology for PFOA and PFOS for crop and pasture: ratio of peak surface water concentrations for dry climate to wet climate.

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.

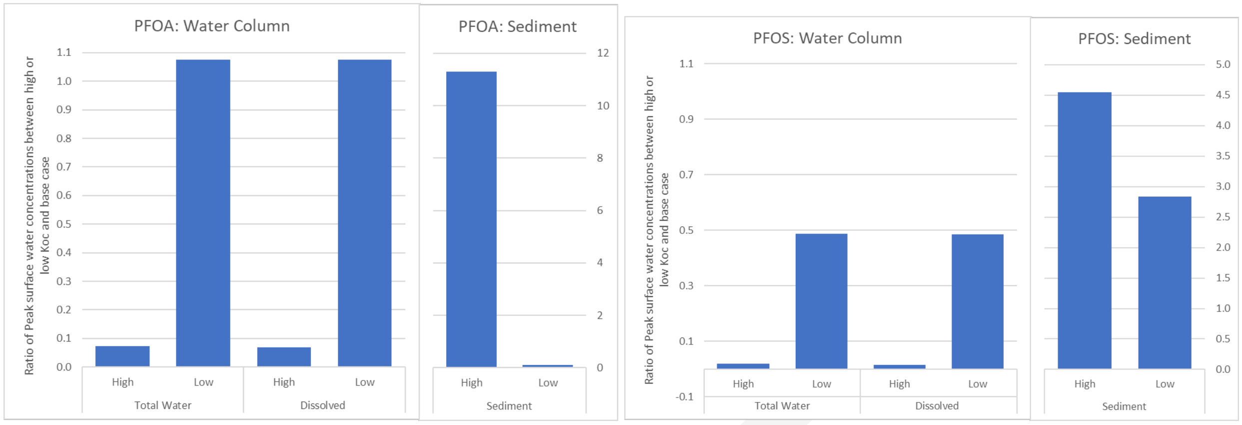


Figure D-20. Sensitivity of VVWM to Koc for PFOA (left) and PFOS (right).

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

				Dry Climate			Moderate Climate			Wet Climate		
Scenario	Receptor	Pathway	Media Conc. Units	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

				Dry Climate			Moderate Climate			Wet Climate		
Scenario	Receptor	Pathway	Media Conc. Units	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

				Dry Climate			Moderate Climate			Wet Climate		
Scenario	Receptor	Pathway	Media Conc. Units	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

				Dry Climate			Moderate Climate			Wet Climate		
Scenario	Receptor	Pathway	Media Conc. Units	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

E.3 References

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Exhibit B

EPA FACT SHEET

Draft Sewage Sludge Risk Assessment for PFOA and PFOS

January 2025

On January 14, 2025, the U.S. Environmental Protection Agency (EPA) released its Draft Sewage Sludge Risk Assessment for Perfluorooctanoic Acid (PFOA) and Perfluorooctane Sulfonic Acid (PFOS). The draft risk assessment indicates that in some scenarios, the EPA's acceptable risk thresholds may be exceeded when sewage sludge containing PFOA and PFOS is land applied for beneficial reuse or surface disposed. The draft risk assessment focuses on people living on or near impacted farms or those that rely primarily on their products. The findings presented in the draft risk assessment are preliminary. The EPA expects to publish a final risk assessment after reviewing public comments and revising the draft risk assessment accordingly. Once finalized, the risk assessment will provide information on risk from use or disposal of sewage sludge and will inform the EPA's potential future regulatory actions under the Clean Water Act (CWA). The EPA is committed to partnering with states, Tribes, territories, and wastewater treatment plants (WWTPs) to reduce risks from PFOA and PFOS that may occur through the management of sewage sludge, including the land application of sewage sludge.

What are sewage sludge and biosolids?

When sewage from households and businesses is sent to a WWTP, the liquids are separated from the solids, producing a nutrient-rich product known as "sewage sludge." The EPA typically uses the term "biosolids" to refer to treated sewage sludge that is intended to be applied to land as a soil conditioner or fertilizer. Sometimes biosolids are distributed to farms. While some states, Tribes, or counties may have additional rules around the use of biosolids, federal rules currently allow biosolids to be applied to pastures, feed crops, and crops for direct human consumption. Biosolids can also be applied to forests, tree farms, golf courses, turf farms, and other types of land. In other cases, biosolids are bagged and sold at stores to the general public and are often used on lawns or in home gardens. Not all WWTPs create biosolids for land application; some incinerate sewage sludge and others send it to a landfill. Biosolids are different from manure or industrial sludge (like pulp from a paper mill), which are also sometimes used as a soil amendment. The EPA does not regulate the land application of manure or industrial sludges in the same manner it does for biosolids.

What are PFOA and PFOS?

PFOA and PFOS are two chemicals in a large class of synthetic chemicals called per- and polyfluoroalkyl substances (PFAS). PFOA and PFOS have been widely studied, and they were once high production volume chemicals within the PFAS chemical class. 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 tend to persist in the environment for long periods of time and have been linked to a variety of adverse human health effects.[1]

PFAS manufacturers voluntarily phased out domestic manufacturing of PFOA and PFOS and the EPA restricted their use by Significant New Use Rules (SNURs) issued under the Toxic Substances Control Act (TSCA).[2] Though

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[1] *see* the EPA's Final Toxicity Assessment for PFOA and Final Toxicity Assessment for PFOS

[2] *see* the EPA's Risk Management for PFAS under TSCA

concentrations of PFOA and PFOS in people's blood have lowered since the voluntary phase out, blood levels can be elevated in communities where there is significant environmental contamination and exposure. [3]

Why is the EPA concerned about the presence of PFOA and PFOS in sewage sludge?

Although domestic manufacturing of PFOA and PFOS have been phased out and their uses restricted, multiple activities still result in PFOA, PFOS, and their precursors being released to WWTPs.[4] Traditional wastewater treatment technology does not remove or destroy PFOA or PFOS, and these chemicals typically accumulate in the sewage sludge. PFOA and PFOS have strong chemical bonds, which means they do not break down on their own in the environment or in our bodies. The chemicals can move from soils to groundwater or nearby lakes or streams, and be taken up into fish, plants, and livestock. These factors combine to raise questions about the potential risks associated with the presence of PFOA or PFOS in sewage sludge that is land applied as a soil conditioner or fertilizer (on agricultural, forested, and other lands), surface disposed, or incinerated.

What are the potential sources of PFOA and PFOS in sewage sludge?

Current and historical activities include industrial releases (*e.g.*, certain types of firefighting foam, pulp and paper plants), commercial releases (*e.g.*, car washes, industrial launderers), and down-the-drain releases from homes (*e.g.*, use of consumer products like after-market water resistant sprays, ski wax, floor finishes, and laundering of stain or water-resistant textiles with PFOA or PFOS coatings). If products containing PFOA or PFOS are disposed of at a lined municipal solid waste landfill, because the most common off-site management practice for landfill leachate is to transfer it to a WWTP, then that landfill's leachate could be a source of PFOA and PFOS to a WWTP. Studies have found PFOA and PFOS in sewage sludge even at WWTPs that only receive wastewater from residential and commercial users. At different WWTPs across the country, any of these release mechanisms may play a role in PFOA or PFOS entering the plant and contaminating the sewage sludge.

What is a sewage sludge risk assessment?

Risk assessment is a scientific process that is used to understand health risks to people, livestock, or wildlife across the country. The EPA uses sewage sludge risk assessments to help evaluate whether actions, including regulation, are needed to protect those who may experience risks from sewage sludge use or disposal. In this sewage sludge risk assessment, the EPA estimates potential human exposures and risks in modeled scenarios where sewage sludge has been land applied or surface disposed. The draft risk assessment focuses on risks to humans because available data indicate that people are much more sensitive to exposures to PFOA or PFOS than livestock or wildlife. Finally, this risk assessment does *not* assess risks to people in the general population, who often have a diversity of sources for their foods.

What does this draft sewage sludge risk assessment suggest?

The draft risk assessment focuses on those living on or near impacted sites (*e.g.,* farm families and their neighbors) 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. Based on the modeling in the draft sewage sludge risk assessment, the EPA finds that there may be human health risks exceeding the EPA's acceptable thresholds for some modeled scenarios when land-applying sewage sludge that contains 1 part per billion (ppb) of PFOA or PFOS. The EPA also finds that there may be human health risks associated with drinking

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[3] *see* the ATSDR's Resources on PFAS Exposure in Impacted Communities

[4] *see* the EPA's Preliminary Effluent Guidelines Program Plan 16 and Multi-Industry Per- and Polyfluoroalkyl Substances (PFAS) Study – 2021 Preliminary Report

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 EPA provides a qualitative description of the potential risks to communities living near a sewage sludge incinerator (SSI) in the draft risk assessment but does not provide quantitative risk estimates due to significant data gaps related to the extent to which incineration in an SSI destroys PFOA and PFOS and the health effects of exposure to products of incomplete combustion.

The draft risk calculations are not conservative estimates because (1) they model risk associated with sewage sludge containing 1 ppb 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) rather than high end exposure conditions, (3) do not take into account non-sewage sludge exposures to PFOA and PFOS (*e.g.*, consumer products, other dietary sources), (4) do not account for the combined risk of PFOA and PFOS, and (5) do not account for additional exposures from the transformation of PFOA and PFOS precursors. As such, risk estimates that account from multiple pathways, multiple sources of exposure, and multiple PFAS would be greater than presented in this draft assessment.

What does this mean for communities?

The Agency recognizes that this draft risk assessment may raise many questions, especially for those who have had biosolids applied to their farms or properties. The EPA encourages people who are concerned to learn about PFAS, including actions that may already be underway and opportunities to reduce exposure. The EPA has created answers to a list of important questions related to this announcement to help members of the public learn more.

If you are concerned about PFAS in sewage sludge, the EPA recommends you contact your state environmental agency or county government to learn about its efforts to address PFOA and PFOS, including in wastewater and sewage sludge. You may also contact your local agriculture extension program, your closest USDA Service Center, or your local wastewater utility to learn more about the biosolids applied to your property and to find out whether they have monitoring data for PFAS or can provide any specific recommendations for your community. The EPA recommends that wastewater systems that find PFOA or PFOS in their biosolids that is land applied take steps to inform the users of biosolids, undertake additional sampling to assess the level, scope, and source of contamination, and examine options for steps to limit exposure. Current science indicates that **lower levels of PFAS exposure present less risk**, so these efforts to identify and reduce PFOA and PFOS in sewage sludge help protect public health.

If you are concerned about PFAS in sewage sludge, the EPA recommends you:

- Consider contacting your state environmental agency or county government to learn about its efforts to address PFOA and PFOS, including in wastewater and sewage sludge.
  - State and regional biosolids contacts: https://www.epa.gov/biosolids/epa-regional-and-state-contacts-biosolids
  - General PFAS resources from your state: https://www.epa.gov/pfas/us-state-resources-about-pfas
- Consider contacting your local agriculture extension program or your closest USDA Service Center.
  - https://extension.org/find-cooperative-extension-in-your-state/
  - https://www.farmers.gov/working-with-us/service-center-locator
- Contact your local wastewater utility to learn more about the biosolids applied to your property and to find out whether they have monitoring data for PFAS or can provide any specific recommendations to request testing of the soil on your property.

- If you have a home drinking water well, ensure you are protecting and maintaining it: https://www.epa.gov/ground-water-and-drinking-water
- Consider testing your home drinking water well for PFOA and PFOS.
  - There is more information about testing private drinking water wells for PFAS in the EPA's factsheet for small and rural communities under the section "Information for Communities and Households Served by Privately-Owned Wells": https://www.epa.gov/system/files/documents/2024-04/pfas-npdwr_fact-sheet_monitoring_4.8.24.pdf
- Learn more about the EPA's Research on PFAS: https://www.epa.gov/chemical-research/research-and-polyfluoroalkyl-substances-pfas
- Review the EPA's Meaningful and Achievable Steps You Can Take to Reduce Your Risk: https://www.epa.gov/pfas/meaningful-and-achievable-steps-you-can-take-reduce-your-risk
- Learn more about the National Academies of Science and Medicine's Guidance on PFAS Exposure, Testing, and Clinical Follow-up: https://nap.nationalacademies.org/resource/26156/interactive/

What is the EPA doing to reduce exposure to PFOA and PFOS in sewage sludge?

The potential risks posed by PFOA, PFOS, and other PFAS demand that the EPA address the problem on many fronts using all applicable statutory authorities. The EPA continues to fund research and take actions to reduce the concentration of PFOA and PFOS discharged to wastewater treatment plants, lower the concentration of these chemicals in sewage sludge, and reduce risk from use or disposal of sewage sludge.[5] Specifically:

- The EPA has provided over twenty million dollars in research funding through the Evaluation of Pollutants in Biosolids and Research for Understanding PFAS Uptake and Bioaccumulation in Plants and Animals in Agricultural, Rural, and Tribal Communities grants.
- The EPA continues to work toward restricting industrial PFAS discharges to WWTPs using Effluent Limitations Guidelines. Current actions include:
  - Revising the Organic Chemicals, Plastics, and Synthetic Fibers Effluent Limitation Guidelines (ELGs) to address wastewater PFAS discharge from PFAS manufacturing facilities;
  - Revising the Metal Finishing and Electroplating ELGs to address wastewater discharge of PFAS from metal finishing and electroplating operations focusing on facilities using PFAS-based fume suppressants and wetting agents; and
  - Revising the Landfills ELGs to address PFAS discharges from landfill leachate.
- The EPA's upcoming Publicly Owned Treatment Works (POTW) Influent PFAS Study will also help the Agency prioritize industrial point source categories for future study and, as appropriate, ELGs.
- To better understand occurrence, the Agency has announced the next National Sewage Sludge Survey to obtain national concentration data on PFAS in sewage sludge.
- The EPA continues to track releases through Toxics Release Inventory Reporting.
- The EPA has updated the Interim Guidance on the Destruction and Disposal of PFAS and Materials Containing PFAS, which presents the state-of-the-science information on methods to remediate, dispose of, and destroy PFAS contamination.
- The Agency has published Final Ambient Water Quality Criteria for Aquatic Life for PFOA and PFOS, which can be used for WWTP effluent permitting.
- The EPA also released draft Human Health Criteria for PFOA, PFOS, and PFBS which, when finalized, can be used for WWTP effluent permitting.

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[5] Learn more about PFAS and review the Agency's PFAS Strategic Roadmap: https://www.epa.gov/pfas/pfas-strategic-roadmap-epas-commitments-action-2021-2024

While the PFOA and PFOS sewage sludge risk assessment and these agency actions are underway, the EPA recommends that states monitor sewage sludge for PFAS contamination, identify likely industrial discharges and other sources of PFAS, and implement industrial pretreatment programs where appropriate. Doing so will help reduce downstream PFAS contamination and lower the concentration of PFAS in sewage sludge as described in Section C of the EPA's December 2022 memorandum entitled, "Addressing PFAS Discharges in NPDES Permits and Through the Pretreatment Program and Monitoring Programs."

Learn more about the EPA's recent actions to address PFAS in sewage sludge.

Learn more about the EPA's Draft Sewage Sludge Risk Assessment for PFOA and PFOS.

Exhibit C

Interim Guidance on the Destruction and Disposal of Perfluoroalkyl and Polyfluoroalkyl Substances and Materials Containing Perfluoroalkyl and Polyfluoroalkyl Substances—Version 2 (2024)

INTERIM GUIDANCE FOR PUBLIC COMMENT
APRIL 8, 2024

The contents of this document do not have the force and effect of law and are not meant to bind the public in any way. This document is intended only to provide clarity to the public regarding existing requirements under the law or agency policies. This guidance is not intended to, and does not, create any right or benefit, substantive or procedural, enforceable at law or in equity by any party against the United States, its departments, agencies, or entities, its officers, employees, or agents, or any other person.

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Abbreviations

AFFF	aqueous film-forming foam
APCD	air pollution control device
ARFF	aircraft rescue firefighting
BDL	below detection limit
BMP	best management practice
C	Celsius
CAA	Clean Air Act
CaF_2	calcium fluoride
CaO	calcium oxide
$Ca(OH)_2$	calcium hydroxide
C&D	construction and demolition
CDC	Centers for Disease Control and Prevention
CDR	Chemical Data Reporting
CEJST	Climate and Economic Justice Screening Tool
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CF_4	carbon tetrafluoride
C_2F_6	hexafluoroethane
C_3F_8	octafluoropropane
CFR	Code of Federal Regulations
CHES	Clean Harbors Environmental Services
CHF_3	fluoroform
CIC	combustion–ion chromatography
CI/MS	chemical ionization mass spectrometry
CKD	cement kiln dust
DE	destruction efficiency
DoD	Department of Defense
DOE	Department of Energy
DRE	destruction and removal efficiency
ECHO	Enforcement Compliance and History Online
EJ	environmental justice
EPA	United States Environmental Protection Agency
ESP	electrostatic precipitator
ESTCP	Environmental Security Technology Certification Program
F	Fahrenheit
FAA	Federal Aviation Administration
FBC	fluidized bed combustor
FF	fabric filter
FML	flexible membrane liner
FTIR	Fourier transform infrared spectrometry
FTOH	fluorotelomer alcohol
FTS	fluorotelomer sulfonate
FY 2020 NDAA	National Defense Authorization Act for Fiscal Year 2020
GAC	granular activated carbon
GCCS	gas collection and control system
HAP	hazardous air pollutant

HF	hydrogen fluoride
HFPO-DA	hexafluoropropylene oxide-dimer acid, 2,3,3,3-Tetrafluoro-2-(1,1,2,2,3,3,3-heptafluoropropoxy) propanoic acid, ($C_6F_{11}O_3H_2$)
H_2O_2	hydrogen peroxide
HWC	hazardous waste combustor
HWI	hazardous waste incinerator
LC/MS/MS	liquid chromatography/tandem mass spectrometry
LFG	landfill gas
LWAK	lightweight aggregate kiln
MDL	method detection limit
MF	microfiltration
MnO_2	manganese(IV) oxide
MSW	municipal solid waste
MWC	municipal waste combustor
NADP	National Atmospheric Deposition Program
NASA	National Aeronautics and Space Administration
NESHAP	National Emission Standards for Hazardous Air Pollutants
NF	nanofiltration
ng/L	nanograms per liter
NIOSH	National Institute for Occupational Safety and Health
NMOC	nonmethane organic compound
NPDES	National Pollutant Discharge Elimination System
NPDWR	National Primary Drinking Water Regulation
NSPS	New Source Performance Standards
NTA	non-targeted analysis
OECD	Organization for Economic Cooperation and Development
ORD	Office of Research and Development
OSTP	Office of Science and Technology Policy
OTM	Other Test Method
PAC	powdered activated carbon
PBPK	physiologically based pharmacokinetic
PCB	polychlorinated biphenyl
PFAA	perfluoroalkyl acid
PFAS	perfluoroalkyl and polyfluoroalkyl substances
PFASTT	PFAS Thermal Treatment Database
PFBS	perfluorobutanesulfonic acid ($C_4F_9SO_3H$)
PFC	perfluorocarbon
PFHxS	perfluorohexanesulfonic acid ($C_6F_{13}SO_3H$)
PFOA	perfluorooctanoic acid ($C_8F_{15}O_2H$)
PFOS	perfluorooctanesulfonic acid ($C_8F_{17}SO_3H$)
PIC	product of incomplete combustion
PID	product of incomplete destruction
PIGE	particle-induced gamma emission spectrometry
PITT	PFAS Innovative Treatment Team
POHC	principal organic hazardous constituent
POTW	publicly owned treatment works
ppb	parts per billion
ppmv	parts per million by volume

PTFE	polytetrafluoroethylene
PVDF	polyvinylidene fluoride or polyvinylidene difluoride
RCRA	Resource Conservation and Recovery Act
RDF	refuse-derived fuel
RO	reverse osmosis
RSEI	Risk-Screening Environmental Indicators
SBIR	Small Business Innovation Research
SDA	spray dryer absorber
SDWA	Safe Drinking Water Act
SERDP	Strategic Environmental Research and Development Program
SSI	sewage sludge incinerator
TOF	total organic fluorine
TRI	Toxics Release Inventory
TRL	technology readiness level
TSCA	Toxic Substances Control Act
TWTDS	treatment works treating domestic sewage
UF	ultrafiltration
UIC	underground injection control
U.S.C.	United States Code
USDW	underground source of drinking water
UST	underground storage tank
VOC	volatile organic compound
WWTP	wastewater treatment plant
XPS	X-ray photo-electron spectroscopy

Executive Summary

This 2024 update of the U.S. Environmental Protection Agency (EPA)'s 2020 interim guidance presents currently available information on the destruction and disposal of perfluoroalkyl and polyfluoroalkyl substances (PFAS) and PFAS-containing materials. It focuses on the current state of science and associated uncertainties for three large-scale capacity technologies that can destroy PFAS or control PFAS release into the environment: thermal destruction, landfills, and underground injection. It also includes a framework for evaluating emerging technologies for PFAS destruction or disposal.

EPA published the first edition of this guidance as required by the National Defense Authorization Act (NDAA) of 2020. The NDAA directs EPA to address the destruction and disposal of PFAS and specific PFAS-containing materials, including aqueous film-forming foam, contaminated media, textiles (other than consumer goods), and various wastes from water treatment. The NDAA also requires EPA to review and update the guidance as appropriate, but no less frequently than every three years.

The primary audience of this guidance is decision-makers who need to identify the most effective means for destroying or disposing of PFAS-containing materials and wastes. The audience may also include regulators, waste managers, and the public, including affected communities. EPA is recommending that managers of PFAS wastes consider the nature of the waste, location, potential for environmental release, and other factors to determine the most appropriate destruction, disposal, or storage method. As a general approach, EPA recommends that decision-makers prioritize the use of destruction and disposal technologies that have a lower potential for PFAS release to the environment, over destruction and disposal technology options with a greater potential for environmental release of PFAS. EPA acknowledges that each destruction and disposal technology has limitations. However, managing PFAS to minimize environmental releases during destruction and disposal activities supports protection of human health and the environment.

While the purpose of an effective destruction and disposal technology is to prevent or minimize environmental releases, it is also important to distinguish between a potential environmental release and a direct human exposure. A PFAS release does not inherently imply direct and immediate human exposure and a release does not necessarily present an unacceptable risk to specific populations.

EPA has developed a fact sheet for the public, including potentially affected communities, that provides a more concise and less technical presentation of key materials.

Key Findings and Updates

EPA describes destruction and disposal options based on each technology's potential to destroy PFAS or control PFAS release into the environment. This approach is consistent with EPA's mission to protect human health and the environment. There are several different types of technologies that have been used for PFAS treatment and disposal, and their potential to destroy PFAS or control PFAS releases is described herein (see Section 1). In general, the following technologies (in no particular order) have a lower potential for environmental release of PFAS compared to other technologies in the same category and are viewed as the more protective technologies:

- **Underground Injection - Permitted Class I non-hazardous industrial or hazardous waste injection wells** are the waste management approach with a lower potential for environmental release when compared to other destruction and disposal options. These wells may help ensure that injected fluids are confined and cannot enter underground sources of drinking water. However, the limited number of wells currently receiving off-site PFAS and waste transportation logistics may significantly limit the type and quantity of PFAS-containing fluids appropriate for underground injection.

- **Landfills - Permitted hazardous waste landfills** have the most protective landfill engineering controls and practices for the containment of PFAS waste and would be more effective at minimizing PFAS release into the environment than other landfill types. EPA recommends Subtitle C landfills when PFAS levels of the waste are relatively high and landfill disposal is the selected option. However, for all landfill types, new information demonstrates landfilling could have higher PFAS releases to the environment than previously thought in 2020.
- **Thermal Treatment - Permitted hazardous waste combustors** such as commercial incinerators, cement kilns, and lightweight aggregate kilns and granular activated carbon (GAC) reactivation units with thermal oxidizers may operate under conditions more conducive to destroying PFAS and controlling related products of incomplete combustion. Research suggests that the use of higher temperatures, well mixed combustion environments, and longer residence times may be more conducive to destroying PFAS and controlling related products of incomplete combustion. Although limited data have been obtained since the 2020 version of the interim guidance (including data suggestive of adequate temperature ranges to break down PFAS), uncertainties remain about the effectiveness of thermal treatment. EPA encourages additional testing with EPA-approved or EPA-evaluated methods by waste managers of thermal treatment operations, including for products of incomplete combustion and the presence of PFAS in all associated waste streams, to evaluate whether thermal treatment technologies are minimizing potential environmental releases. EPA's new analytical method, OTM-50, will allow better characterization of products of incomplete combustion (PICs) that will help to address some of these uncertainties when additional data are collected as recommended in this guidance (see Appendix A).

This list includes the three technology categories described in Section 3 and does not preclude new or emerging technologies. When considering the bullets above, it is important to note that real-world performance and testing data are generally limited. Additional performance and testing data—including data on destruction and removal of PFAS in thermal treatment devices, ongoing research activities, and long-term performance data for landfills and underground injection—may change EPA's understanding of each technology's ability to control PFAS. Data needs and priorities are summarized in this update.

EPA continues to seek collaboration with thermal treatment facilities to conduct air emission testing during thermal treatment of PFAS and PFAS-containing materials. EPA provides guidance for this and for independent analysis in Appendix A. In addition to performance and testing data, the 2024 guidance identifies other, high-priority data gaps that need to be addressed by researchers across government, academia, and industry for EPA to provide meaningful updates on effective destruction and disposal practices.

EPA partnered with industry and academia to collect data on four emerging technologies for PFAS destruction: mechanochemical degradation, electrochemical oxidation, gasification and pyrolysis, and supercritical water oxidation. While the results from these studies show promise for PFAS destruction, further work using newly available methods is needed to more fully characterize the outputs of these processes and to evaluate their performance for a wider range of PFAS-containing materials. Managers of PFAS and PFAS-containing materials may consider these or other emerging technologies for PFAS disposal or destruction. To assist PFAS material managers with evaluating whether an emerging destruction (or disposal) technology is suitable for a particular PFAS-containing material, EPA developed a technology evaluation framework. The framework provides a transparent, consistent approach for evaluating destruction and disposal technologies for PFAS and PFAS-containing material and includes considerations for communities. EPA recommends that managers of PFAS-containing materials use the technology evaluation framework to evaluate emerging destruction and disposal technologies and to inform their decisions about managing PFAS-containing materials. EPA also encourages technology developers to generate and publicly release data that can be used to complete the framework (see Section 6).

EPA updated tools, methods, and approaches for considering the impacts of potential releases and exposure on communities located near destruction and disposal facilities, including identifying vulnerable populations overburdened by cumulative impacts, incorporating vulnerability into decision-making, and engaging the community (see Section 4).

What's Next?

EPA and other government, academic, and private institutions will continue to conduct research to better understand PFAS destruction and disposal. EPA will consider public comments it receives on this version of the guidance as well as additional advancements in PFAS research and science to revise this interim guidance within the next three years.[1] For information about the Agency's plans and actions related to PFAS, visit EPA's website pages *PFAS Strategic Roadmap: EPA's Commitments to Action 2021-2024* and *Key Actions to Address PFAS*.

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[1] EPA may explore opportunities to provide more frequent technical updates as information becomes available.

1. Introduction

This second version interim guidance provides information on technologies that may be feasible and appropriate for the destruction or disposal of perfluoroalkyl and polyfluoroalkyl substances (PFAS) and PFAS-containing materials. It represents the current state of science and the associated uncertainties for large-capacity technologies that can destroy PFAS or control PFAS release into the environment. This is an update to the interim guidance the U.S. Environmental Protection Agency (EPA) issued in 2020.

EPA published the first version of this interim guidance and made it available for public comment.[2] EPA received 77 unique public comments (out of 5,066 total comments) on the interim guidance from states, Tribes, industries, trade organizations, environmental organizations, and members of the public. This updated version incorporates this public feedback.

The 2024 interim guidance contains the following new information:

Section 1: Introduction

- Updated review of PFAS destruction and disposal technologies.

Section 2: Description of PFAS-Containing Materials Identified in the FY 2020 NDAA

- Additional details on spent water treatment materials.
- Updated information on biosolids generation and management.

Section 3: Technologies for the Destruction and Disposal of PFAS and PFAS-Containing Materials

- Additional information on thermal treatment and landfills.
- Additional information on testing and monitoring at thermal treatment facilities.

Section 4: Considerations for Potentially Vulnerable Populations Living Near Likely Destruction or Disposal Sites

- Updated list of tools for vulnerability screening.

Section 5: Research Needs and Data Gaps for Destruction and Disposal Technologies

- Increased focus on research needs to inform future updates to this interim guidance.
- Updated information on ongoing research and development activities.

Section 6: Emerging Technologies for PFAS Destruction and Disposal (NEW)

- EPA efforts to identify, review, and test emerging technologies for PFAS destruction.
- Technology evaluation framework that utilizes a multiple-lines-of-evidence approach to evaluate a technology/PFAS material combination.

Appendix A: EPA Guidance to Conduct PFAS Emissions Field Testing at Commercial Thermal Destruction Sources (NEW)

- EPA request to access commercial sources to conduct air emissions testing during PFAS thermal destruction.
- Guidance for facilities to conduct source testing prior to accepting/processing PFAS materials at full-scale thermal destruction facilities.

Appendix B: Summary of the Clean Harbors Test Data (NEW)

- Two source test campaigns conducted at a hazardous waste combustor.

Appendix C: Summary of the Chemours Thermal Oxidizer Test Data (NEW)

- Source test campaign conducted at a chemical manufacturer.

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[2] Interim PFAS Destruction and Disposal Guidance; Notice of Availability for Public Comment. U.S. EPA. 85 Federal Register 83554. December 22, 2020.

Appendix D: Summary of Costs and Considerations (NEW)
- Summary of costs and factors that should be considered when estimating costs to destroy or dispose of PFAS and PFAS-containing materials.

1.a Purpose and audience of this interim guidance

The purpose of this interim guidance is to describe technologies for managing the destruction and disposal of PFAS-containing materials and to recommend practices associated with these technologies that minimize PFAS releases to the environment. This purpose is consistent with EPA's mission to protect human health and the environment and the National Defense Authorization Act (NDAA) for Fiscal Year 2020, Public Law No: 116-92 (hereafter, "FY 2020 NDAA"), signed into law on December 19, 2019.

Section 7361 of the FY 2020 NDAA (see Figure 1-1) directs EPA to publish interim guidance on the destruction and disposal of PFAS and PFAS-containing materials no later than one year from the date of enactment of the FY 2020 NDAA. EPA published the initial interim guidance on December 18, 2020. The FY 2020 NDAA also requires EPA to review and revise the interim guidance at least every three years, if appropriate. This 2024 update fulfills that requirement.

The primary audience of this guidance is decision-makers who need to identify the most effective means for destroying or disposing of PFAS-containing materials and wastes. The audience may also include regulators, waste managers, and technology developers who can use the technology evaluation framework in Section 6 to evaluate emerging technology applicability to PFAS destruction or disposal. This guidance may also interest the general public; particularly communities near PFAS destruction and disposal sites who may have an opportunity to offer input or provide consultation to decision-makers regarding destruction and disposal decisions based on relevant decision-makers determination. EPA has developed a fact sheet for the public, including potentially affected communities, that provides a more concise and less technical presentation of this guidance.

This interim guidance meets one of many commitments EPA made under the Agency's PFAS Strategic Roadmap. The PFAS Strategic Roadmap, released in 2021, describes EPA's whole-of-Agency approach to address PFAS and identifies concrete actions EPA is taking to safeguard public health, protect the environment, and hold polluters accountable. For more information on EPA's actions to address PFAS, visit *PFAS Strategic Roadmap: EPA's Commitments to Action 2021-2024* and *Key Actions to Address PFAS*.

SEC. 7361. PFAS DESTRUCTION AND DISPOSAL GUIDANCE.

(a) IN GENERAL.—Not later than 1 year after the date of enactment of this Act, the Administrator shall publish interim guidance on the destruction and disposal of perfluoroalkyl and polyfluoroalkyl substances and materials containing perfluoroalkyl and polyfluoroalkyl substances, including—

- (1) aqueous film-forming foam;
- (2) soil and biosolids;
- (3) textiles, other than consumer goods, treated with perfluoroalkyl and polyfluoroalkyl substances;
- (4) spent filters, membranes, resins, granular carbon, and other waste from water treatment;
- (5) landfill leachate containing perfluoroalkyl and polyfluoroalkyl substances; and
- (6) solid, liquid, or gas waste streams containing perfluoroalkyl and polyfluoroalkyl substances from facilities manufacturing or using perfluoroalkyl and polyfluoroalkyl substances.

(b) CONSIDERATIONS; INCLUSIONS.—The interim guidance under subsection (a) shall—

- (1) take into consideration—
  - (A) the potential for releases of perfluoroalkyl and polyfluoroalkyl substances during destruction or disposal, including through volatilization, air dispersion, or leachate; and
  - (B) potentially vulnerable populations living near likely destruction or disposal sites; and
- (2) provide guidance on testing and monitoring air, effluent, and soil near potential destruction or disposal sites for releases described in paragraph (1)(A).

(c) REVISIONS.—The Administrator shall publish revisions to the interim guidance under subsection (a) as the Administrator determines to be appropriate, but not less frequently than once every 3 years.

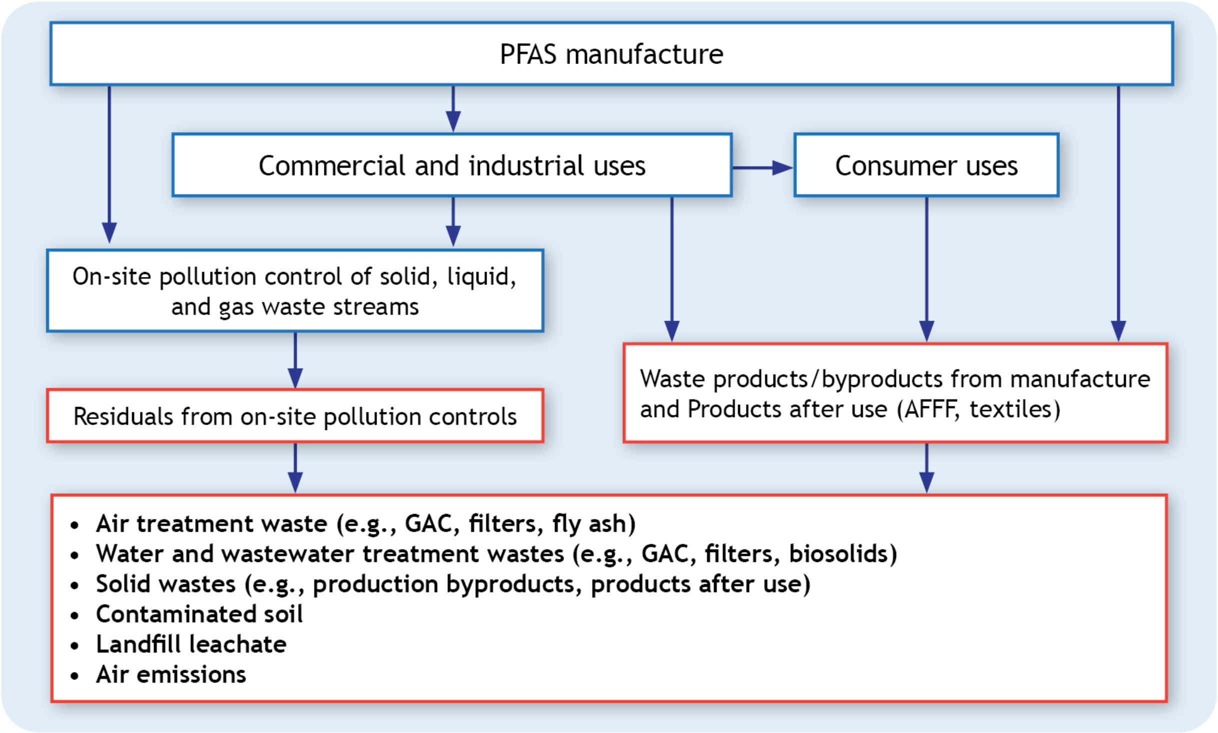
Figure 1-1. FY 2020 NDAA Section 7361.

1.b Scope and use of interim guidance

1.b.i PFAS and PFAS-containing materials in this interim guidance

Section 7361 of the FY 2020 NDAA (see Figure 1-1) lists six types of PFAS-containing materials and wastes. Although the information included in this interim guidance is likely to be suitable for other types of PFAS and PFAS-containing materials, this interim guidance addresses destruction and disposal centering on these six material types, which are described in more detail in Section 2. In this document, the PFAS-containing materials covered by the FY 2020 NDAA that will be destroyed or disposed of are referred to as “waste.”

Figure 1-2 shows conceptually how manufacturing activities could result in material streams that are within the intended scope of this interim guidance, including GAC and aqueous film-forming foam (AFFF).



Note: The red-outlined portions show where the FY 2020 NDAA material types occur in the course of manufacture, use, and disposal of PFAS and PFAS-containing materials that are within the scope of this interim guidance.

Figure 1-2. Generation of PFAS materials identified in the FY 2020 NDAA.

1.b.ii Destruction and disposal technologies addressed in this interim guidance

This interim guidance presents three widely-used, commercially available destruction and disposal technologies that may be effective for managing PFAS and PFAS-containing materials. Advances in scientific understanding allow EPA to make some recommendations, however, filling information gaps and addressing uncertainties will improve the scientific understanding and could lead to additional guidance in the future. EPA updates this guidance at least every three years.

This interim guidance describes technologies with the potential to destroy PFAS or control releases of PFAS into the environment. Other waste management options are not discussed, because they are not within the intended scope of this interim guidance. For example, land application of biosolids containing PFAS does not control PFAS releases into the environment.

Current technologies. In developing this second version of the interim guidance, EPA focused on the three widely-used, commercially available destruction and disposal technologies that may be effective for managing PFAS wastes—thermal treatment, landfills, and underground injection. EPA also focused on reviewing data and information about the ability of each of these technologies to manage PFAS-containing materials through containment or destruction (i.e., breaking carbon–fluorine bonds and mineralization) while limiting releases of PFAS to the environment. In doing so, EPA prioritized addressing the scientific issues and uncertainties highlighted in the 2020 interim guidance and, when possible, relied on publicly available, peer-reviewed data and information (see Section 3).

Emerging technologies. Concerns about the effectiveness of current, widely-used technologies for managing PFAS and PFAS-containing materials have motivated efforts to develop new technologies and apply existing technologies to PFAS materials. EPA is aware of and, in some cases, has participated in efforts to identify, develop, and test non-thermal destruction technologies for PFAS-containing materials (see Section 6).

In some cases, managers of PFAS and PFAS-containing materials may consider emerging technologies for PFAS destruction. When considering the use of emerging technologies, managers of PFAS waste should work with state, territorial, or Tribal government agencies. To assist PFAS material managers with evaluating whether an emerging destruction (or disposal) technology is suitable for a particular PFAS-containing material, EPA developed a technology evaluation framework (see Section 6). The framework provides a transparent, consistent approach for evaluating destruction and disposal technologies for PFAS materials, including considerations for communities, including vulnerable populations.

In 2024, EPA is neither recommending nor discouraging the use of any emerging technology for managing the PFAS-containing materials specified in the FY 2020 NDAA. Instead, the Agency recommends that managers of PFAS-containing materials use the technology evaluation framework to evaluate emerging destruction and disposal technologies and to inform their decisions about managing PFAS-containing materials. EPA also encourages technology developers to generate and publicly release data that can be used to complete the framework (see Section 6).

Generally, destruction and disposal of PFAS-containing materials are currently not federally regulated. PFAS-containing materials may be managed in non-hazardous and hazardous waste treatment and disposal systems. Some exceptions exist, such as risk management requirements in cases where permits for production of new chemicals specify methods to be used for destruction or disposal following the review of a Toxic Substances Control Act (TSCA) new chemicals notice. Unless governing regulations require specific actions to the contrary, this guidance recommends selecting an approach with a higher potential to control PFAS releases into the environment, as possible.

Table 1-1. Destruction and Disposal Technologies Discussed in This Interim Guidance, with Examples of PFAS-Containing Materials

Destruction and Disposal Technology, by Physical Phase of Materials	Examples of PFAS-Containing Materials (within the Scope of the FY 2020 NDAA) That Could Be Managed Using These Technologies
Solid phase: Landfill disposal Thermal treatment	• Drinking water, groundwater, and wastewater treatment residuals ○ Biosolids/sewage sludge ○ Spent GAC ○ Ion exchange resins • Air waste stream treatment residuals ○ Spent GAC ○ Fly ash • Contaminated soil • End-of-life products (e.g., textiles) • Solidified liquid wastes
Liquid phase: Underground injection Thermal treatment	• Sewage sludge (liquid) • Landfill leachate • AFFF (spent or concentrate)

Destruction and Disposal Technology, by Physical Phase of Materials	Examples of PFAS-Containing Materials (within the Scope of the FY 2020 NDAA) That Could Be Managed Using These Technologies
	• End-of-life products (e.g., spent cleaning solvents) • Pollution control residuals (e.g., concentrates) from PFAS production and use
Gas phase: Thermal treatment	• Landfill gas (LFG) • Emissions from manufacture, use, or destruction of PFAS

1.b.iii Use of this interim guidance

This interim guidance provides information and suggested considerations for evaluating destruction and disposal options. This second version of the interim guidance is based on currently available research and information in 2023 and is responsive to the scope of the FY 2020 NDAA. It presents background information on the manufacture and uses of PFAS, as well as solid, liquid, and gas waste streams containing PFAS, including those materials identified in the FY 2020 NDAA: AFFF, soils and biosolids, textiles, spent water treatment materials, landfill leachate, and PFAS waste streams from facilities manufacturing or using PFAS (see Section 2). Most significantly, it provides up-to-date information on potential releases during the destruction and disposal of PFAS and PFAS-containing materials (see Section 3) and identifies data gaps to be filled that can inform future EPA guidance (see Section 5). This version also incorporates information received during the 2020 Interim Guidance's public comment period.

This version of the interim guidance takes the following into account:

- It is based on currently available information on technology performance and capabilities for destruction and disposal of the PFAS and PFAS-containing materials specified in the FY 2020 NDAA (see Section 1.b.i).
- It focuses on three widely-used, commercially available destruction and disposal technologies that may be effective for managing PFAS wastes (i.e., thermal treatment, landfills, underground injection) (see Section 3).
- It advises how to assess and consider the impacts of potential releases and exposure on communities, including identifying vulnerable populations, incorporating vulnerability into decision-making, and performing community engagement (see Section 4). The FY 2020 NDAA states that the interim guidance should take into consideration potentially vulnerable populations living near destruction or disposal sites.
- It identifies research needs to inform future updates to this interim guidance and provides updated information on ongoing research and development activities (see Section 5).
- It discusses the development of emerging technologies and includes a framework for evaluating technology/PFAS material combinations (see Section 6).
- It does *not* establish what concentrations of PFAS in wastes, spent products, or other materials or media would necessitate destruction or disposal. Regulatory mechanisms are more appropriate for establishing such concentrations.
- It provides references to destruction and disposal facility locators that decision-makers can use to request cost estimates. EPA recommends obtaining cost estimates from applicable destruction and disposal facilities as described in Section 1.c.ii. In addition, Appendix D provides some cost

information that PFAS material managers may want to consider when evaluating different techniques to destroy or dispose of PFAS and PFAS-containing materials. Note that cost information presented in this appendix has a relatively high level of uncertainty associated with it. Moreover, the cost of PFAS destruction and disposal may change based on demand and evolution of available technologies. Additionally, overall costs may vary widely based on the type of materials or the distance to a facility.

This interim guidance is not a regulation. Any discussion of EPA's regulatory authorities is for the purpose of describing controls relevant to the destruction or disposal of PFAS and should not be considered a description of the applicability of those authorities or affect EPA's or any other agency's ability to enforce under these authorities in the future. For information about EPA's other plans and actions related to PFAS, visit EPA's *PFAS Strategic Roadmap: EPA's Commitments to Action 2021-2024* and *Key Actions to Address PFAS*.

1.c Considerations for managing PFAS and PFAS-containing materials

The information presented in this interim guidance can help managers of PFAS or PFAS-containing materials make informed decisions regarding destruction and disposal. To assist decision-makers, EPA reviews destruction and disposal options for thermal treatment, landfills, and underground injection based on the relative potential for releasing PFAS to the environment. This approach highlights destruction and disposal options with lower potential for PFAS releases to the environment compared to other options. This approach is consistent with EPA's mission to protect human health and the environment and is designed to facilitate decisions that protect human health and the environment as much as possible.

EPA recognizes that the potential to control PFAS releases to the environment is one of several factors to consider when selecting among options for destroying or disposing of PFAS and PFAS-containing materials in a way that is consistent with maximizing protection to public health and the environment. Other factors include whether it is imperative to destroy or dispose of the waste immediately, availability of interim storage or destruction and disposal options, the type and volume of waste materials, the concentrations of PFAS in the waste, and the presence of and potential impacts to communities, including vulnerable communities. In the following sections, EPA provides guidance for these decisions in terms of technology considerations and infrastructure considerations.

1.c.i Technology Considerations - Relative potential for PFAS releases to the environment

Using the information presented in Section 3, and considering the significant uncertainties that remain, EPA listed destruction and disposal options in terms of relative potential for releasing PFAS to the environment. When considering the data and information presented in Section 3, EPA considered the type of data available (e.g., laboratory-, pilot-, or field-scale studies), the amount of data available (e.g., data from different operating conditions), and the quality of the data.

When considering the order, it is important to note that available performance and testing data are generally limited. Additional performance and testing data—including data on destruction and removal of PFAS in thermal treatment devices, and long-term performance data for landfills and underground injection—are needed to support more specific guidance. When these data and information become available, EPA may reconsider and revise the order presented below.

As a general approach, EPA encourages managers of PFAS and PFAS-containing materials to prioritize destruction and disposal options with a lower potential for releasing PFAS to the environment over the use of destruction and disposal technology options with a higher potential for environmental release of PFAS. EPA also encourages testing with EPA-approved methods of thermal treatment operations, including products of incomplete combustion and the presence of PFAS in all associated waste streams (see Appendix A), to evaluate whether site-specific conditions are minimizing potential environmental releases.

While the purpose of an effective destruction and disposal technology is to prevent or minimize environmental releases, it is also important to distinguish between a potential environmental release and a direct human exposure. A PFAS release does not inherently imply direct and immediate human exposure and a release does not necessarily present an unacceptable risk to specific populations.

EPA encourages managers of PFAS and PFAS-containing materials to make decisions in consideration of factors discussed in this guidance to select current or future disposal or destruction technologies that protect human health and the environment as much as possible. Based on currently available information cited in this guidance, EPA has listed the following disposal and destruction options from lowest to highest based on their relative potential to release PFAS to the environment:

- **Interim storage with controls.** While not a destruction or disposal method, interim storage may be a short-term option if the destruction or disposal of PFAS and PFAS-containing materials is not imperative, on-site storage capacity is readily available, and interim storage has proper controls in place to reduce releases into the environment. Storage may be more appropriate for some PFAS materials than others. For example, EPA recommends interim storage of low volumes of containerized or high PFAS-concentration materials. In contrast, some materials may be less appropriate for storage because they are continuously generated or are typically high-volume. With proper engineering controls in place, interim storage can control PFAS releases in the short-term.
- **Underground injection (Class I non-hazardous industrial and hazardous waste wells).** EPA has determined the use of Class I non-hazardous industrial waste and hazardous waste wells for high concentration liquid PFAS waste has a lower potential for environmental release when compared to other PFAS destruction and disposal options, and there is relatively low uncertainty in this determination. The standards associated with the construction, operation, and monitoring of these Class I wells are designed to isolate liquid wastes deep below the land surface and ensure protection of underground sources of drinking water (USDWs). Class I hazardous waste wells operators must conduct mechanical integrity testing more often than Class I non-hazardous industrial waste operators. They also must establish and follow additional procedures for reporting and correcting mechanical integrity problems. Class I hazardous waste well operators must also develop and follow a waste analysis plan and conduct annual tests of cement at the base of the well. While the standards associated with Class I hazardous wells are more stringent than for non-hazardous industrial wells, EPA has determined that the probability of failures for both has been demonstrated to be low. While Class I wells are an option for managing PFAS-containing fluids, this technology may not be appropriate everywhere. The limited number of wells currently receiving PFAS, well location, and waste transportation logistics may significantly limit the type and quantity of PFAS-containing fluids appropriate for underground injection. Additional Class I wells may need to be constructed where geologically suitable, and existing well permits may need to be modified to meet the capacity needs for PFAS disposal (see Section 3.c.v).

- **Hazardous waste landfills (Resource Conservation and Recovery Act (RCRA) Subtitle C).** This class of landfills has the most stringent environmental controls in place for minimizing environmental releases and migration of some PFAS from disposed waste. EPA recommends Subtitle C landfills when PFAS levels of the waste are relatively high and landfill disposal is the selected option. Permitted hazardous waste landfills have relatively low quantities of landfill leachate[3], which lessens the probability of mobilizing PFAS that is reversibly bound to the waste. These landfills are required to handle and treat leachate as listed hazardous waste (hazardous waste number F039 per 40 CFR 261.31), lessening the likelihood of environmental release. Because hazardous waste landfills generally do not receive organic wastes that decompose, PFAS migration via LFG is anticipated to be relatively low but LFG collection and destruction equipment could be installed if anaerobic decomposition is shown (see Section 3.b). While recent studies have improved understanding of PFAS migration, additional research is needed to resolve uncertainties (see Section 3.b.vi).
- **Landfill disposal in all landfill types: Landfill disposal of stable polymeric PFAS**. Stable polymeric PFAS, such as polytetrafluoroethylene (Teflon™) and fluorinated ethylene propylene (FEP), are large molecules that are not susceptible to hydrolysis or oxidation. They are neither volatile nor water soluble and, as a result, they are expected to remain within the waste mass for long periods of time. This PFAS category tends not to migrate with the leachate and is unlikely to volatilize with the LFG; therefore, permitted hazardous and municipal solid waste (MSW) landfills (MSWLFs) may provide a good disposal option. Some data are available to support this assessment, but more data are needed to resolve uncertainties (see Sections 3.b.i and 3.b.vi).
- **GAC reactivation units with thermal oxidizers operating under certain conditions**. Carbon reactivation systems, with the associated use of off-gas incineration (i.e., afterburners operating at temperatures >1,100°C[4]) and gas scrubbing units, can potentially destroy PFAS without significant environmental releases, or without PFAS remaining on the reactivated carbon. However, as discussed in Section 3.a.i, more data are needed for confirmation, particularly regarding reactor conditions, differing carbons, and products of incomplete combustion (PICs). There is uncertainty associated with this option because it is based on limited research. While recent studies have improved understanding, additional research is needed to resolve uncertainties on PFAS emissions from full-scale reactivation facilities (see Section 3.a.v).
- **Thermal treatment units operating under certain conditions.** Hazardous waste combustors (HWCs), including commercial incinerators, cement kilns, lightweight aggregate kilns (LWAKs), and thermal oxidizers, that operate under certain conditions (i.e., feeding liquid PFAS-containing materials, higher temperatures >1,100°C[4], well mixed, and adequate residence time) may be more effective at adequately destroying (mineralizing) PFAS and minimizing PICs. There is uncertainty associated with this option because it is based on limited research and observations, there are limited data for the treatment of solid materials or containerized wastes, and there are limited emissions data (e.g., no data for PIC formation and the presence of PFAS in air pollution control device residuals) at full-scale facilities (see Section 3.a.v) due to previous methodology limitations.
- **Landfill disposal in MSWLFs that have composite liners and leachate and gas collection and treatment systems: Landfill disposal of volatile or water soluble PFAS.** MSWLFs receive non-hazardous waste and most have composite liners and gas collection systems. Soluble PFAS in these

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[3] EPA defines landfill leachate as the liquid that is formed when rain water filters through wastes placed in a landfill. When this liquid comes in contact with buried wastes, it leaches, or draws out, chemicals or constituents from those wastes (*https://www.epa.gov/landfills/municipal-solid-waste-landfills*).

[4] Maximum PIC destruction was observed at and above 1,090°C as described by Shields et al., 2023.

landfills are likely to be released over time in landfill leachate if they are not adequately bound or sequestered in immobile solids. Further research is needed on the ability of various solidification techniques to retain soluble PFAS for extended periods under conditions found in MSWLFs. Although MSWLFs generally have leachate collection and management processes in place, the typical practices employed are not expected to prevent further PFAS migration. In addition, MSWLFs generally receive significant quantities of organic wastes (e.g., food scraps) that decompose, and thereby can cause PFAS migration via LFG. Multiple studies have reported elevated levels of PFAS in leachate and LFG. Studies estimate that up to 15 percent of mobile PFAS disposed in MSW landfills is emitted to the environment over time. EPA does not recommend MSWLF disposal of media containing relatively high concentrations of soluble or volatile PFAS. While recent studies have improved understanding of PFAS migration in MSWLFs, additional research is needed to resolve uncertainties (see Section 3.b).

- **Thermal treatment at lower temperatures** including municipal waste combustors (MWCs), sewage sludge incinerators (SSIs), or HWCs operating at lower temperatures or feeding solid or containerized PFAS-containing materials. Some HWCs operate at temperatures near or below 1,100°C, MWCs typically operate between 850 and 1,000°C, and SSIs typically operate at even lower temperatures. There is uncertainty associated with this option because it is based on very limited research. While limited, existing datapoints suggest the potential for air emissions of PFAS and PICs orders of magnitude above detection limits at temperatures below 1,000°C (Shields et al., 2023). There are no data on the treatment of solid materials or containerized wastes at lower temperatures. There are currently insufficient data and information to conclude anything about PIC formation or PFAS air emissions from full-scale combustion units operating at these lower temperatures or when feeding solid materials or containerized wastes. Because there are insufficient data available, there is low confidence in the reliability of this technology to control PFAS releases (see Section 3.a).
- **Construction and Demolition (C&D) landfills: Landfill disposal of volatile, water soluble, or oxidizable PFAS.** This class of landfills may not have a composite liner or leachate collection system. Because of the potential for environmental releases, EPA does not recommend that PFAS waste, other than wastes containing stable polymeric PFAS, be disposed of in this type of landfill. While few studies have documented emissions of PFAS from C&D landfills, the known mobility of PFAS in landfills and lack of engineered controls at C&D landfills make them unsuitable for PFAS disposal. For these reasons, EPA does not expect that C&D landfills will minimize PFAS releases to the environment (see Section 3.b).

1.c.ii Infrastructure considerations – communities, costs, and capacity

Waste management infrastructure in the United States is a complex and varied system, crucial for managing the vast amounts of waste generated, including PFAS-containing waste. Therefore, in addition to considering potential for environmental release, decision-makers should consider site- or project-specific factors that may influence decisions about managing PFAS-contaminated wastes. Specific factors include, but are not limited to, the composition of the waste, the types and concentrations of PFAS, transportation costs, destruction and disposal costs, and impacts to communities.

EPA emphasizes the importance for decision-makers (managers of PFAS-containing materials making decisions about destruction and disposal) to consider potential impacts to nearby communities, including vulnerable populations, when evaluating destruction, disposal, and storage options. Those concerns are amplified because the uncertainties in the effectiveness of these options to limit or control environmental releases can contribute to adverse impacts, including potentially disproportionate and cumulative impacts, for vulnerable communities. Therefore, EPA recommends decision-makers screen communities located in the vicinity of potential releases from the destruction, disposal, and storage options (considering fate and transport) in order to consider the potential for community impacts, including adverse and disproportionate impacts (see Section 4), and to consider potential measures to address such impacts. Depending on site-specific circumstances (e.g., PFAS concentrations, impacted media, and potential exposure pathways), the size and shape of this area (vicinity) will vary. In deciding how to address PFAS waste, site-specific considerations and waste characteristics should factor into the decision. For example, destruction or disposal of waste with a high concentration of PFAS using a method that has a higher potential for environmental release may not be well suited for a site where there is greater likelihood of potential exposure to vulnerable populations.

The waste management infrastructure's design and operation are heavily influenced by regional needs, local regulations, and technological advancements. As a result, waste management options vary by the type of waste and by location due to factors like availability, feasibility, and transportation requirements. For instance, urban areas may incur higher disposal costs due to limited space while rural areas may face higher transportation costs to distant disposal facilities. Furthermore, technological advances specifically related to the management of PFAS-containing materials can potentially reshape the cost and operational landscape of PFAS waste management. EPA recommends that decision-makers request cost estimates directly from potentially applicable destruction and disposal facilities. In addition, decision-makers can refer to Appendix D, which summarizes costs and factors to consider when estimating costs to destroy or dispose of PFAS and PFAS-containing materials.

Decision-makers seeking lists of waste management sites in the United States can access various resources depending on the type of waste and location. It is important to note that often the most direct source is local government websites, such as county or city environmental services or public works departments. These websites typically provide comprehensive lists and maps of local facilities, along with their types, hours of operation, and accepted waste materials. For specialized waste like hazardous or medical waste, state environmental protection agencies or EPA are valuable resources. EPA's website, in particular, offers databases and tools such as *RCRAInfo*, which provides detailed information about facilities managing hazardous waste.

A general yet incomplete list of potential treatment and disposal facilities may also be available through the EPA Incident Waste Decision Support Tool (*I-WASTE DST*). This list includes landfills (hazardous waste, MSW, industrial, and C&D), HWCs, and nonhazardous combustors (sewage sludge and municipal waste) but does not include all technologies available for PFAS destruction and disposal (e.g., cement kilns, carbon reactivation facilities, and underground injection facilities). I-WASTE DST is updated periodically and is limited to information available online.

Similar to cost estimation, management capacity estimates can also be challenging. Facilities that can potentially manage PFAS-containing waste are not evenly distributed. Their number and capacity vary by region. Some areas across the United States face challenges due to limited available landfill space or capacity limits. Incineration and underground injection capacities can also be greatly influenced by technological capabilities and local regulations.

2. Description of PFAS-Containing Materials Identified in the FY 2020 NDAA

The FY 2020 NDAA identifies six types of materials that commonly contain PFAS (see Figure 1-1 and the Executive Summary). This section discusses each FY 2020 NDAA material type, its origin, potential sources of PFAS, current disposal and treatment methods, and potential releases to the environment.

Data on FY 2020 NDAA-relevant material types come from a variety of sources, and more PFAS data will be available from EPA-managed datasets in the near future. For example, EPA's Chemical Data Reporting (CDR) dataset includes production volumes for manufactured and imported amounts and conveys certain industrial processing and use activities. One data element distinguishes amounts that are recycled instead of discharged or released to a waste stream. In the 2020 CDR reporting cycle, submitters began using Organization for Economic Cooperation and Development (OECD)-based industrial processing and use codes to better harmonize those data (U.S. EPA, 2020a).

In 2022, EPA issued two memoranda recommending EPA and state National Pollutant Discharge Elimination System (NPDES) permitting authorities and publicly owned treatment works (POTW) pretreatment program authorities conduct certain activities to align wastewater and stormwater NPDES permits, pretreatment program implementation activities, and biosolids monitoring with the goals in EPA's *PFAS Strategic Roadmap* (U.S. EPA, 2022a, 2022d). The memoranda recommend that EPA and state permitting authorities use the most current sampling and analysis methods in their NPDES programs to identify known or suspected sources of PFAS and to take actions using their pretreatment and permitting authorities, such as imposing technology-based limits on sources of PFAS discharges. The memoranda will also help the Agency obtain comprehensive information through monitoring regarding the sources and quantities of PFAS discharges, informing other EPA efforts to address PFAS.

Additionally, EPA's Toxics Release Inventory (TRI) program collects data related to industrial releases and waste management of certain chemicals. The TRI dataset includes, among other information, quantities of environmental releases to all media above certain thresholds (including on-site disposals and land application), as well as quantities transferred to off-site waste management facilities, including POTW. Under Section 7321 of the FY 2020 NDAA, certain PFAS were added to the TRI list beginning with Reporting Year 2020 (U.S. EPA, 2020b). As of Reporting Year 2023 (reporting forms due by July 1, 2024), a total of 189 PFAS are on the TRI list. TRI reporting data can be accessed at *https://www.epa.gov/toxics-release-inventory-tri-program/tri-data-and-tools*.

In October 2023, EPA finalized the PFAS 8(a)(7) reporting rule under TSCA (88 FR 70516, October 11, 2023). In accordance with obligations under TSCA, as amended by the FY 2020 NDAA, the PFAS 8(a)(7) reporting rule requires any person that manufactures (including import) or has manufactured (including imported) PFAS or PFAS-containing articles in any year since January 1, 2011, to electronically report information regarding PFAS uses, production volumes, disposal, exposures, and hazards. The PFAS 8(a)(7) rule will help EPA better understand the sources and quantities of PFAS manufactured in the United States and provide EPA with the most comprehensive dataset of PFAS manufactured in the United States. EPA will begin receiving reports under the PFAS 8(a)(7) rule in 2025.

2.a Solid, liquid, or gas waste streams containing PFAS from facilities manufacturing or using PFAS

PFAS do not occur naturally. They are synthesized for use in a diverse array of industrial and commercial applications. Industrial waste streams containing PFAS stem from two main sources: (1) primary manufacturing facilities of PFAS and (2) secondary industries that use PFAS or manufacture finished products that contain PFAS. A 2009 survey by OECD identified 27 primary manufacturers and processors of PFAS globally (OECD, 2011). At the time of that survey, more than 90 percent of the global annual production of PFAS was generated by eight manufacturers (Posner et al., 2009), all of which participated in EPA's 2010/2015 Perfluorooctanoic Acid (PFOA) Stewardship Program (U.S. EPA, 2006).

The goal of the PFOA Stewardship Program was to reduce PFOA facility emissions and their use in products by 95 percent by 2010, compared with 2006 baseline levels, and to eliminate PFOA from all facility emissions and products by 2015 (U.S. EPA, 2006). Companies that did not participate in the PFOA Stewardship Program likely continue to use these chemical substances. Long-chain PFAS and their precursors may still be produced as unintentional byproducts and may persist in facility emissions and as product impurities in small quantities (3M Company, 1999; Boucher et al., 2019; Lehmler, 2009; Kissa, 2001). Perfluoroalkyl sulfonic acids containing six or more carbons and perfluoroalkyl carboxylic acids with seven or more carbons are generally considered to be long-chain PFAS. PFOA and PFOS are examples of long-chain PFAS, and PFBS and PFHxA are examples of short-chain PFAS. Furthermore, information on the toxicity and environmental fate and transport of alternative PFAS chemistries is limited (Sun et al., 2016; Wang et al., 2014).

For the 2020 CDR reporting cycle, 52 sites reported a total domestic manufacture and import production volume of 339,000 tons for PFAS in 2019. This represents the aggregate production volume for PFAS produced and imported into the United States across all industries. Although the aggregate production volume might not include all PFAS sources (for instance, a specific chemical or site may not meet reporting obligations), it provides a proxy quantity of all PFAS domestically produced and imported.

PFAS might be released into the environment at every step in the production process, including during synthesis, polymerization, application, transport, usage, waste stream management, and disposal (3M Company, 2000b).

Table 2-1 lists examples of PFAS uses and the resulting solid, liquid, and gas waste streams for primary and secondary manufacturers of PFAS-containing materials and certain service sectors as indicated by industry, national and global inventories, and research. However, this list is not exhaustive or representative of all current uses, applications, recovery and recycling practices, or treatment technologies that could affect the volume and characteristics of the resulting waste streams. EPA recognizes the need for continued research to better characterize the multi-media PFAS-containing materials targeted for destruction or disposal, as discussed in Section 5.a.

2.a.i Solid phase wastes

Primary manufacturing and secondary industrial use of PFAS can generate solid waste streams with PFAS-containing materials (OECD, 2011, 2015). For example, some PFAS synthesis processes can produce tars and polymers consisting of high-molecular-weight byproducts that are either fully or partially fluorinated. These byproducts can be recycled back into the process or treated as waste (3M Company, 2000a, 2000b). Solid wastes such as fly ash (e.g., thermal treatment) or spent GAC (e.g., drinking water treatment) can also be produced from PFAS incineration and other treatment processes.

Other important solid-phase materials include sludges and biosolids (see Section 2.c) that result from stabilizing or treating process waters and wastewaters, either on site or at a municipal wastewater treatment plant (WWTP)[5] that receives influent from industrial sources (Venkatesan & Halden, 2013). In addition to solids produced via treatment, spent water treatment media (such as activated carbon and ion exchange resins) are part of this waste stream (see Section 2.e). Other direct industrial sources of solid wastes containing PFAS include intentional residuals, such as cuttings and fibers from textile manufacturing (see Section 2.d), and materials unintentionally produced outside of product specification. The concentrations and composition of PFAS in solid wastes generated from primary and secondary industrial sources vary by facility and depend on factors such as facility- or industry-specific production processes and the types and quantities of PFAS produced or used (ITRC, 2020).

The characteristics of these solid phase PFAS-containing wastes vary significantly, depending on several factors. As discussed in Section 3.b, the type of PFAS (e.g., polymeric, soluble, or volatile PFAS) as well as the type of waste with which the PFAS is comingled (e.g., biodegradable or non-biodegradable) will dictate the potential and pathways for environmental release. The type of environmental release will, in turn, inform the most appropriate disposal option(s) for each type of PFAS-containing solid waste.

2.a.ii Liquid phase wastes

Primary manufacturers and secondary industrial users of PFAS can generate liquid phase wastes in the form of: (1) liquid byproducts of PFAS synthesis to be recycled or disposed of (e.g., to a landfill) following stabilization, solidification, or another management method; (2) process wastewater resulting from activities using PFAS as a processing aid (e.g., surfactant, emulsifier, mist suppressant, sizing agent) and production of finished products containing PFAS; (3) spills or unintentional releases of liquid wastes and products containing PFAS; and (4) AFFF/water/foam mixtures from the use of fire-extinguishing agents (see Section 2.b) for emergency response activities and emergency response trainings at industrial facilities.

Table 2-1 provides examples of liquid wastes containing PFAS generated by industrial sources and their uses. This is not an all-inclusive list of industries nor waste streams.

Another liquid phase waste stream is wastewater effluent discharged directly from a primary manufacturer or secondary industrial PFAS user. Effluent from wastewater treatment facilities that receive wastewater from industrial PFAS sources may also contain PFAS. According to several studies, conventional wastewater treatment technologies are generally ineffective at destroying or controlling PFAS (Schultz et al., 2006) and may result in higher measurable PFAAs (e.g., PFOA, perfluorooctane sulfonate [PFOS], and their homologues) when precursor compounds (e.g., fluorotelomers) are degraded during the treatment process (Buck et al., 2011; Dauchy et al., 2017a; Schultz et al., 2006; Sinclair & Kannan, 2006). Less often, primary industrial PFAS manufacturers have opted to transport liquid wastes off site for incineration (North Carolina Department of Environmental Quality, 2017).

2.a.iii Gas phase wastes

Studies suggest that PFAS in air emissions from manufacturing facilities are a source of both localized (i.e., within a short radius of the facility) and long-distance (i.e., global) transport of PFAS within the environment (Davis et al., 2007; Dreyer et al., 2009). Non-volatile forms of PFAS, such as the anionic

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[5] Regulations at 40 CFR part 122.2 and part 403.3 identify these “municipal wastewater treatment plants" as treatment works treating domestic sewage (TWTDS) or publicly owned treatment works (POTW), dependent on their ownership.

PFAAs (e.g., PFOA, PFOS, their homologues), are associated with airborne particulates when emitted as aerosols from stack emissions at primary manufacturing facilities (Barton et al., 2006; Dreyer et al., 2015). Gas phase emissions of volatile and semivolatile PFAS, and the subsequent transformation of precursor compounds into persistent PFAAs, are a potential mechanism for the atmospheric transport of PFAS. For example, volatile fluorotelomer alcohols (FTOHs) and perfluoroalkyl sulfonamides can transform into perfluoroalkyl carboxylic acids (e.g., PFOA and homologues) and perfluoroalkyl sulfonates (e.g., PFOS and homologues); these can be deposited at significant distances from their origin, which may result in soil and groundwater contamination (Dreyer et al., 2009; Ellis et al., 2004; Martin et al., 2006; Schenker et al., 2008).

Table 2-1. Examples of PFAS Waste Streams by Industry Type.
This is not an all-inclusive list of industries or waste streams.

Industry Type	Uses	Examples of Waste Streams[a]			Source
		Solid	Liquid	Gas	
Primary chemical manufacturing	PFAS synthesis, feedstocks for primary products, feedstocks for secondary users, processing aids (fluoropolymers)	Process byproducts (tars), sludges/residuals, off-spec materials[b], treatment residuals (GAC/anion exchange resins), spill residues (replacement and legacy), *particulate emissions*	Degraded/stabilized process wastes, wastewater effluent, *stack emissions condensate*	Stack emissions, fugitive volatiles	3M Company (1999, 2000b)
Secondary Manufacturing (Industry Users of PFAS-Containing Materials)[c]					
Adhesives manufacturing	Component of solvent- and water-based adhesives, rubber to allow bonding to steel, and urea-formaldehyde adhesive resins for wood particleboard bonding	*Used filter media and filter residues, residues of cured adhesives, empty containers, used shop rags (from cleaning), contaminated soil (from spill cleanup residues)*	*Residues of liquid adhesives, off-spec products[b], contaminated wastewater (from spill cleanup residues)* *For cleaning: equipment startup, cleaning, and flushing wastes; spent cleaning solvents; and contaminated wastewater*	Stack emissions, fugitive volatiles	ASC (n.d.), RadTech International North America (2010)
Cleaning product manufacturing	Component of household cleaners; car wash products; automobile waxes; wiper fluid; strongly acidic or basic cleaners for concrete, masonry, airplanes		*Off-spec products[b], liquid residues from empty containers, and spills*	Stack emissions, fugitive volatiles	3M Company (1999)

Industry Type	Uses	Examples of Waste Streams[a]			Source
		Solid	Liquid	Gas	
	Used for removal of adhesives, dry cleaning of textiles or metal surfaces, machine parts cleaning after nickel plating, and removal of calcium sulfate from reverse osmosis membranes				
Computers/ electronics manufacturing	Component of sealant for electric circuits, zinc battery electrolyte, wetting agents in solders, polar solvents used before welding Used for removal of cured epoxy resins from integrated circuit modules, treatment of insulated wire, alkaline manganese battery MnO_2 cathode treatment, production of polymer electrolyte membrane for fuel cells, cleaning of electronic components, and coating of the surface of magnetic recording devices	*Collected airborne particulates for cleaning/surface preparation*	*Spent acid solution for cleaning/surface preparation, liquid residues from empty containers, and spills*	Stack emissions, fugitive volatiles	U.S. EPA (1990)
Film/ lithography manufacturing	Used in coatings for surface tension, static discharge, and adhesion control for films, papers, and printing plates, and as a surfactant in mixtures used to process imaging films	Rags and wipes discarded by applicator; solids coated with PFAS from processing, sampling, quality assurance; off-spec products[b]	Solvent waste, liquid residues from empty containers, spills, and unused application mix	Coating application exhaust	3M Company (1999), Bowden et al. (2002)
Metal plating/ fabrication	Used as a surfactant, wetting agent, and mist suppressing agent; as a wetting agent fume suppressant for chromium plating and chromium anodizing; as a dispersion product used to coat metals; as a	*Off-spec products[b]*	Spent plating or etching baths, rinse water effluent, liquid residues from empty containers, and spills	Stack emissions, fugitive volatiles	3M Company (1999), U.S. EPA (2009b)

Industry Type	Uses	Examples of Waste Streams[a]			Source
		Solid	Liquid	Gas	
	blocking agent for aluminum foil; in plating baths; and to treat metal surfaces				
Oil and gas drilling/ extraction/ refinery/ support	Component of chemical barrier used for containing oil spills Used as a surfactant for recovery in oil/gas recovery wells, a jet fuel/hydrocarbon solvent, and in hydraulic oils Used as a gasoline/petroleum product evaporation inhibitor in storage tanks in the following forms: a floating layer of cereal grains treated with PFAS, an aqueous layer containing PFAS		Applied product[d] (oil spills, oil and gas recovery wells), *liquid residues from empty containers, and spills*		UNEP (2011), Kissa (2001)
Paint/coating manufacturing	Component of coatings, paints, varnishes, dyes, ink jet printer inks, and ski waxes	*Pigment dust*	*Unused paint products, off-spec products[b], liquid residues from empty containers, and spills*	*Fugitive volatiles, atomized paint*	Waste Management and Research Center (1992)
Paper products/ packaging manufacturing	Waterproofing and grease proofing for products including food contact paper (e.g., plates, popcorn bags, pizza box liners, food containers, wraps), non-food contact applications (e.g., folding cartons, carbonless forms, masking papers)	Dusts; *solids coated with PFAS from processing, sampling, quality assurance; off-spec products[b]*	Spillage, cleanup, and releases during opening, rinsing, and cleaning of PFAS totes	Fugitive volatiles	U.S. EPA (2009a)
Pesticide/ fertilizer/ other agriculture chemical manufacturing	Pesticide and herbicide ingredients and packaging	*Particulate emissions*	*Liquid residues from empty containers, spills, off-spec products[b], cleaning of equipment, and process wastewaters*	*Fugitive volatiles*	U.S. EPA (2023b)

Industry Type	Uses	Examples of Waste Streams[a]			Source
		Solid	Liquid	Gas	
Plastic materials/ resins/ rubber product manufacturing	Used to make membranes used in fuel cells; chlor-alkali cells; water, caustic soda, and caustic potash electrolyzers; silicone rubber sealants; composite resins; polytetrafluoroethylene (PTFE, or Teflon); and polyvinylidene fluoride or polyvinylidene difluoride (PVDF) Processing aid for PVDF manufacture, mold-release agent in foam molding, and antiblocking agent for rubbers	Dusts, spillage (micropowders and resins), cuttings, scrap, debris, and off-spec products[b]; particulate emissions	*Liquid residues from empty containers, spills, and unused application mix*	Fumes from PTFE heating (volatile)	Ebnesajjad (2015)
Textiles/ apparel/ leather/ carpets/ fiber manufacturing	Dispersion products that coat fabrics: jackets, shoes, umbrellas, carpets, upholstery, leather, tents, sails	Solids coated with PFAS from cutting, shearing, packaging, lab and color sampling, quality assurance; flakes or dust containing PFAS; *off-spec products*[b]	Spills, wastewater effluent from product adhering to inside of drum, and unused application mix	Releases of vapors and aerosols during application of surface treatment and mechanical finishing	U.S. EPA (2009a)
Aerospace component manufacturing	Mechanical components such as tubing, hoses, and seals; brake and hydraulic fluid additive; wire and cable insulation; used in coating and paint		*Wastewater effluent*		FluoroCouncil (2019)
Automotive component manufacturing	Mechanical components such as tubing, hoses, and seals; brake and hydraulic fluid additive; anti-mist film on windshields; used in coating/paint; used in coatings or surface treatments of textiles and upholstery		*Wastewater effluent*		FluoroCouncil (2019), ITRC (2020)

Industry Type	Uses	Examples of Waste Streams[a]			Source
		Solid	Liquid	Gas	
Semiconductor manufacturing	Etching solutions for photolithography, glass etching, plastics etching, fused silica, aluminum; liquid etchant in photo mask rendering		Spent plating or etching baths, PFOA residues from photoresist developers associated with semiconductor liquid waste streams, liquid residues from empty containers, and spills Photoresists and antireflective coatings stripped off from semiconductor devices before shipment are present in waste solvent streams	Photoresists and antireflective coatings stripped off from semiconductor devices before shipment are present in waste gas streams	Bowden et al. (2002), Tremblay (2015)
Building and construction materials manufacturing	Component of cement and primers used to coat cement mortar; used in wire and cable insulation and coatings for wood particleboards	*Cuttings and debris, off-spec materials[b]*	*Wastewater effluent*		Buck et al. (2012), FluoroCouncil (2019), U.S. EPA (2009a)
Mining industry	Surfactant for recovery of metals from ores; used in ore flotation to separate metal salts from soil, electrowinning of metals, and nitrogen flotation to recover uranium	Contaminated rock from applied product[d]	Applied product[d]		ITRC (2020)
Medical uses	Video endoscopes; catheters; saline solutions for in vitro diagnostics; treatment/coatings for textiles such as hospital gowns, curtains, drapes; dialysis machines	*Laboratory/medical solid wastes (e.g., tubing, filters, films)*			FluoroCouncil (2019), Posner (2012)
Cosmetics and personal care product manufacturing	Used in cosmetics, hair conditioning formulations, hair creams, and toothpaste	*Off-spec materials[b]*	*Wastewater effluent*		Danish EPA (2018), FluoroCouncil (2019), Schultes et al. (2018)

Industry Type	Uses	Examples of Waste Streams[a]			Source
		Solid	Liquid	Gas	
Fire suppression systems[e]	AFFF and certain dry fire-extinguishing agents	Contaminated soil and debris from applied product[d]	Applied product[d]		See Section 2.b for more information. Gaines (2023)

[a] The italicized waste streams may contain PFAS, given what wastes the relevant industry sectors are known to generate and given applications of PFAS. The presence and concentration of PFAS have not been quantified.

[b] "Off-spec materials" or "off-spec products" are materials or products that do not meet specified standards or requirements and are discarded rather than sold or used.

[c] Some industries listed under "Secondary Manufacturing" may also include primary manufacturing of PFAS. The waste streams resulting from manufacture of PFAS in these industry sectors are addressed in the first row, "Primary chemical manufacturing."

[d] "Applied product" refers to the intentional application of a PFAS-containing product to the environment.

[e] Fire suppression systems, which include AFFF, are commonly found in manufacturing, storage, extraction and refining, and national defense facilities, as well as airports, fire departments, and other federal facilities (e.g., facilities operated by the National Aeronautics and Space Administration [NASA] and the U.S. Department of Energy [DOE]). Although this is an industrial use, not an industrial sector, the prevalence of AFFF in the above primary and secondary PFAS manufacturing industries warrants highlighting these waste streams.

2.b Aqueous film-forming foam

AFFFs are a group of PFAS-containing fire extinguishing agents for low-flashpoint hydrocarbon fuel fires (Tuve et al., 1964). AFFFs are intended for use where a significant flammable liquid fire hazard exists (FFFC, 2016).

AFFFs are based on synthetic fluorosurfactants that provide unique low surface tension and positive spreading coefficient characteristics. When mixed with water and applied, AFFFs form an aqueous film and a foam solution to coat the liquid fuel, seal fuel vapor, and reduce oxygen availability, extinguishing the fire and preventing burnback (FFFC, 2016; SERDP, 2020; Sheinson et al., 2002).

Until application, AFFF is managed as a concentrated product containing less than 2 percent PFAS fluorosurfactants by weight for a typical 3 percent AFFF concentrate (ITRC, 2020). AFFF is stored in either fixed, structural dispensing systems, such as those in hangars and aboard vessels, or in mobile, vehicle-based systems (i.e., aircraft rescue firefighting [ARFF] vehicles) (Field et al., 2017). Reserve AFFF concentrate inventory may be stored in hangars or warehouses. The amount of AFFF concentrate in the finished foam varies by manufacturer and application circumstances, but is usually between 1 and 6 percent, meaning the fluorosurfactants are diluted to less than a fraction of a percent (FFFC, 2016; ITRC, 2020).

A 2004 inventory estimated that there were 4.6 million gallons of legacy PFOS-containing AFFF in the United States (Darwin, 2011). AFFF inventory depletion is determined by frequency of use for firefighting, training, or testing; transfers between locations; and other factors. However, AFFF's characteristically long shelf life means there should be little disposal due to expiration (FFFC, 2016), increasing the possibility that legacy PFOS-containing AFFF concentrate remains in service or reserve inventories. Fluorotelomer-based AFFF surfactant products became available in the 1970s (Prevedouros, 2006). In 2017, the U.S. Department of Defense (DoD) edited its military specification for AFFF to include no more than 800 parts per billion (ppb) of PFOA and PFOS in the concentrate, the limit set by DoD Quality Systems Manual 5.1. However, this specification does not mention any other PFAS standards.

In the United States, AFFF and associated systems are or have been in service at federal facilities, civil airports, and oil refineries. Civilian fire departments also use or have used AFFF. The federal government is working to identify areas of current and former federal properties where PFOS- or PFOA-containing AFFFs have been used (Darwin, 2011; DoD, 2020). Through September 30, 2023, DoD has determined that 715 active military installations, base realignment and closure (BRAC) locations, National Guard facilities, and formerly used defense sites (FUDS) properties require an assessment of PFAS use or potential release. DoD is performing preliminary assessments/site inspections (PA/SIs) at these installations, which is the first phase of the cleanup process and may take two to three years to complete. As of September 30, 2023, DoD has completed the PA/SI phase at 570 installations. The Department has determined that no further action is required at 118 of these installations, while 452 are proceeding to the next step in the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) process (DoD, 2024).

The FY 2020 NDAA prohibits AFFF use at any military installation, effective October 1, 2024 or up to two years later if the Secretary of Defense deems a waiver necessary. DoD issued a policy in January 2016 to discontinue land-based AFFF training and testing activities, unless the AFFF could be captured and properly disposed. Since then, DoD has managed any mission-critical AFFF use in response to an emergency event as a spill response to mitigate impacts to the environment (DoD, 2019, 2020). DoD, among other entities, is also investing in research and development for fluorine-free AFFF alternatives (SERDP, 2020). In 2023, DoD issued a new military specification for fluorine-free foam (MIL-PRF-32725) (DoD, 2023).

Examples of AFFF users and locations in the United States are listed in Table 2-2. Note that the list of sources in the table is non-exhaustive.

Table 2-2. Examples of AFFF Users and Locations in the United States

AFFF User	Locations	Comments
DoD	• Hundreds of military installations (active and former) with AFFF use • In-service systems installed at an estimated 1,350 locations in aircraft hangars and on an estimated 3,000 ARFF vehicles	• In 2016, DoD discontinued AFFF use for military installation training and testing unless the AFFF could be captured and properly disposed and is investing in fluorine-free alternatives • The FY 2020 NDAA requires DoD to end any land-based AFFF by October 1, 2024 (with extensions possible until 2026) • Sources: DoD (2017, 2019, 2020); CBO (2019); SERDP (2020)
NASA	• Multiple properties with ARFF apparatus • Five properties with fixed systems	• In late 2018, NASA issued order to cease training with AFFF • Source: SERDP (2020)

AFFF User	Locations	Comments
DOE	• Lawrence Berkeley National Laboratory • Brookhaven National Laboratory • Los Alamos National Laboratory • Strategic Petroleum Reserve facilities (Gulf of Mexico)	• Sources: DOE (2021 *DOE Initial Assessment of PFAS at DOE Sites*); Darwin (2011)
Airports and supporting facilities	• 523 Federal Aviation Administration (FAA)-certified civilian airports in 2018 • Since 2006, an annual average of nearly 600 certified airports • Supporting facilities include firefighting training sites, such as the FAA Technical Center's Fire Training Area	• Starting in 1972, FAA regulations allowed for the use of AFFF among a number of different extinguishing agents at civilian airports; since 2006, FAA guidance (FAA, 2006) provides that AFFF must conform to the applicable military specification (AC 150/5210-6 and MIL-F-23485F) at most civilian airports [a] • 80 percent of respondents to a 2017 survey of U.S. and Canadian airports reported training-related discharge directly to the ground; two-thirds reported testing-related discharge directly to the ground • FAA has taken steps to reduce AFFF discharges during testing since 2019 • Sources: DOT (2019); FAA (2006, 2019); Thalheimer et al., (2017); U.S. EPA (2020c)
Fire departments	• At civilian fire departments throughout the United States	• AFFF may be present at fire departments • Some states have begun inventorying and reporting having AFFF (e.g., Michigan, New Hampshire) • Some states have take-back programs to help local fire departments identify AFFF in inventory and assist with removal and disposal (e.g., Vermont) • Sources: Michigan PFAS Action Response Team (n.d.); New Hampshire Department of Environmental Services (2020); Vermont Agency of Natural Resources (n.d.)
Oil refineries and processing facilities	• Oil refineries and related facilities (e.g., storage facilities)	• Little information is available about AFFF in this sector, though published industry guidelines recommend AFFF for pipeline emergencies • Survey-based data suggest this sector is the second largest consumer of AFFF after federal agencies • Sources: Darwin (2011); Noll & Hildebrand (2016)

AFFF User	Locations	Comments
Ships and other vessels	- Ships and other marine vessels, including the U.S. Coast Guard	- Little information is available about AFFF quantities on ships - There has been a shift toward non-fluorinated AFFF for some uses including testing and training, though the Coast Guard has indicated that certain uses (e.g., required inspections) must continue to use fluorinated AFFF - Source: U.S. EPA (2020e)

[a] An exception exists for airports with low departure traffic and serving aircraft less than 90 feet in length. See 14 Code of Federal Regulations (CFR) 139.317 for more information (FAA, 2006).

Fate and transport of PFAS in AFFF after use depends on the release circumstances and chemical-specific properties. Though sometimes classified as incidental releases (Thalheimer et al., 2017), equipment failure, accidental releases, or operator error can result in substantial leaks (Anderson et al., 2016; Resolution Consultants, 2016; Leidos, 2016).

Engineering controls (such as dikes, barriers, or basins) may be installed at facilities with significant flammable liquid hazards to contain foam solution and runoff for later disposal (FFFC, 2016). Where such hazards do not significantly exist, or installed engineering controls are otherwise not practicable, firefighting personnel may block sewer drains or deploy portable dikes as containment measures (FFFC, 2016). Runoff can then be pumped out, and the affected environmental media can be removed for disposal (ITRC, 2020). C&D debris originating from facilities where AFFF was historically released may also be a source of PFAS in landfills and groundwater (Solo-Gabriele et al., 2020).

Though subject to site-specific characteristics and conditions, studies demonstrate AFFF use at airports is a source of PFAS in soil and groundwater (Ahrens et al., 2015; Dauchy et al., 2017b; Høisæter et al., 2019). Further, PFAA precursors from original AFFF concentrate products may transform in the environment to more mobile products over time (Houtz et al., 2013), expanding plumes long after AFFF use is discontinued.

In the 2022 EPA-issued memoranda that recommended NPDES and POTW pretreatment program permitting conditions (U.S. EPA, 2022a, 2022d), EPA recommended that NPDES stormwater permits contain best management practices (BMPs) to address and minimize AFFF discharges, as included in the Naval Air Station Whidbey Island MS4 NPDES Permit[6]:

- Prohibiting the use of AFFFs other than for actual firefighting.
- Eliminating PFOS- and PFOA-containing AFFFs.
- Requiring immediate clean-up in all situations where AFFFs have been used, including diversions and other measures that prevent discharges via storm sewer systems.

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[6] Naval Air Station Whidbey Island MS4 NPDES permit: *https://www.epa.gov/sites/default/files/2020-12/documents/r10-npdes-naval-air-station-whidbey-ms4-was026611-final-permit-2020.pdf*

2.c Soils and biosolids

As required by Clean Water Act Section 405(d), EPA established requirements for the final use or disposal of sewage sludge when it is: (1) applied to land as a fertilizer or soil amendment; (2) placed in a surface disposal site, including sewage sludge-only landfills; or (3) incinerated. The regulation at 40 CFR part 503 defines sewage sludge as "solid, semi-solid, or liquid residue generated during the treatment of domestic sewage in a treatment works." Sewage sludge includes scum or solids removed in primary, secondary, or advanced wastewater treatment processes and any material derived from sewage sludge (e.g., a blended sewage sludge/fertilizer product) but does not include grit and screenings or ash generated by the incineration of sewage. Sewage sludge treated for land application is commonly referred to as "biosolids."

Part 503 considers domestic septage as sewage sludge and sets separate requirements for domestic septage applied to agricultural land, forests, or reclamation sites. Influent containing PFAS that enters wastewater treatment facilities may result in the presence of PFAS in biosolids (Thompson et al., 2023; Helmer et al., 2022; Venkatesan & Halden, 2013). When biosolids are applied to land, there is the potential for leaching or runoff (Schaefer et al., 2022; Navarro et al., 2016; Lindstrom et al., 2011; Sepulvado et al., 2011; Washington et al., 2010). If landfilled instead, there are concerns with PFAS releases via landfill leachate and gaseous emissions, especially for highly organic material (Section 3.b). Alternatively, if incinerated, there is the potential for PFAS to be emitted as PICs (Section 3.a). Research needs and resources on possible emerging PFAS destruction technologies for biosolids (e.g., pyrolysis/gasification coupled with a thermal oxidizer and supercritical water oxidation) are shared in Sections 5 and 6. Across disposal and destruction technologies, there are capacity concerns and more research is needed on potential PICs/products of incomplete destruction (PIDs).

Total nationwide biosolids generation and management statistics in 2021 (reported to EPA's biosolids program as of January 9, 2023) and data considerations are presented in Table 2-3. EPA does not have data on the volume of biosolids that contain PFAS; however, EPA is planning for the next National Sewage Sludge Survey in collaboration with the Effluent Guidelines Program Plan 15's upcoming POTW influent study, both of which will focus on monitoring for PFAS. Additionally, EPA is currently conducting a refined risk assessment for PFOA and PFOS in biosolids (U.S. EPA, 2024), as well as recommending exposure mitigation measures and pretreatment program BMPs to reduce PFAS sources (U.S. EPA, 2022d).

Table 2-3. Biosolids Generation and Management in 2021

NPDES permits issued to POTW[a]	**16,109**
Biosolids NPDES permits[b]	9,834
Biosolids annual reporters (2021)[c]	2,519
Sewage Sludge generated in 2021 (metric tons)[d]	**4,548,462**
Land application of biosolids (metric tons)	1,955,684
Land disposal (also called surface disposal) (metric tons)	94,522
Incineration (metric tons)	633,054
Landfilling (metric tons)	1,808,136
Other management (metric tons)	57,066

[a] An NPDES permit is typically a license for a facility to discharge a specified amount of a pollutant into a receiving water under certain conditions. A POTW is a treatment works treating domestic sewage (TWTDS) that is owned, and usually operated, by a local or regional government agency (U.S. EPA, 2020b).

[b] Biosolids permits apply to any person or treatment works that prepares sewage sludge, applies sewage sludge to the land, or fires sewage sludge in a sewage sludge incinerator and to the owners/operators of surface disposal sites, as well as the exit gas from sewage sludge incinerator stacks (U.S. EPA, 2019).

[c] Annual reports are required for POTW that have a design flow rate equal to or more than one million gallons per day, that can serve populations of 10,000 or more, that are required to have approved pretreatment programs (Class I Sludge Management Facilities), or that are otherwise required to report (U.S. EPA, 2023a).

[d] Total amount of biosolids generated is reported by each TWTDS and may not equal the sum of component amounts reported for each management method. Biosolids totals do not represent PFAS presence in biosolids.

In addition to land application of PFAS-containing materials (e.g., biosolids), there are other scenarios where PFAS release to soils can occur. Direct release of PFAS into soils can occur through use of PFAS-containing products such as AFFF, and the discharge or application of treated industrial or municipal wastewater containing PFAS. PFAS can also be in soils due to unintentional contact, such as spills or leaks during the production, handling, transport, or use of PFAS-containing materials (see Table 2-4 for examples of industrial sources of PFAS in soils). Both direct and indirect soil impacts might also occur via the atmospheric deposition of PFAS released from stack emissions and atmospheric transformation products of volatile precursors, respectively (Davis et al., 2007; Dreyer et al., 2009; Schenker et al., 2008). Remediation wastes such as soils excavated during the cleanup of sites or during decommissioning of facilities where PFAS was manufactured, used, or applied may contain diverse mixtures of PFAS in elevated concentrations.

2.d Textiles, other than consumer goods, treated with PFAS

Because PFAS can repel oil, water, and stains, the textile industry uses these chemicals in a broad range of textile products other than consumer goods (apparel or household textiles). For example:

- PFAS can be used to treat outdoor equipment such as tents and sails (UNEP, 2011).
- Technical or occupational textiles, such as protective clothing for firefighters, can be treated with PFAS or woven from fluoropolymers (OECD, 2013).
- Medical garments can be treated with fluorinated polymers (OECD, 2013).
- Fluoropolymers can be spun into fibers and used to make sailcloth and fabric for fire suppression needs (Tokarsky & Uy, 2003).

- PTFE can be woven to make architectural fabrics, such as roofs, and can also be used to coat fiberglass for tensile structures or long-life structures (Fabric Architect, 2020).
- Textiles made from fiberglass coated with or saturated with PFAS are used for high-temperature or corrosive industrial environments. Kevlar and perfluoroplastic composite textiles are used for similar industrial environments (Robco, 2020).

Examples of typical PFAS-containing waste streams generated from textiles include discarded industrial or commercial textiles such as apparel, carpets, or personal protective equipment; solids coated with PFAS from cuttings and shearings; and fugitive volatiles from spray applications of textile surface treatments. The destruction and disposal technologies used for these waste streams include landfill disposal and thermal treatment. For examples of industrial waste streams from the textiles/apparel manufacturing industry, see Table 2-4.

2.e Spent water treatment materials

EPA published a proposed rule for a PFAS National Primary Drinking Water Regulation (NPDWR) for six PFAS in March 2023 that will require compliance once finalized and promulgated. Additionally, 12 states have established standards limiting the presence of various PFAS in drinking water. EPA has identified "best available technologies" for treating drinking water for PFAS to comply with the NPDWR; these technologies will produce residuals. The 2024 Interim Guidance details options available to manage these residuals while minimizing environmental exposures and releases to the extent feasible. EPA notes that there are multiple approaches for water systems to manage PFAS residuals that are generated when PFAS is removed from drinking water. As of December 2023, there are no specific federal regulatory requirements for PFAS disposal, and the presence of PFAS in treatment residuals does not impose any additional federal requirements, but they may be subject to regulatory attention. EPA's PFAS actions are updated on the webpage *Key EPA Actions to Address PFAS*.

Conventional drinking water treatment methods such as coagulation, flocculation, sedimentation, disinfection, and filtration do not remove PFAS found in source water. Therefore, residues from these processes such as grit, spent membrane, or filter elements are unlikely to contain significant PFAS.

Although novel technologies for removing PFAS from drinking water sources and groundwater are being developed, current processes known to be effective are activated carbon, anion exchange resins, and high-pressure membranes (reverse osmosis [RO] and nanofiltration [NF]) (U.S. EPA, 2022b, 2022c). This section discusses the residual streams of these three processes (see Section 3 for discussions on treatment and disposal considerations, Appendix D for costs and considerations, and Section 5 for discussions of research needs for more novel treatments) and regulatory considerations for drinking water treatment utilities.

Table 2-4 shows the primary residual streams by water treatment process, with options for reactivation, destruction, and disposal of these residuals.

Table 2-4. Destruction and Disposal Options for Drinking Water Treatment Residuals Containing PFAS

Treatment Process	Residuals Generated	Destruction/ Disposal Option(s)	Section Reference
Granular Activated Carbon	Spent GAC	Reactivation	3.a.i.2
		Incineration	3.a
		Landfill	3.b

Treatment Process	Residuals Generated	Destruction/ Disposal Option(s)	Section Reference
Powdered Activated Carbon (PAC)	Spent PAC	Incineration	3.a
		Landfill	3.b
Ion Exchange	Single use and regenerative ion exchange: spent resins	Incineration	3.a
		Landfill	3.b
	Regeneration brine	Interim Storage	1.c
		Underground Injection	3.c.ii.1
RO/NF	Brine	Permitted discharge, direct or indirect	Not described in this guidance. Refer to permit.

The practices discussed in Table 2-4 are not in a hierarchical order and should be chosen based on site-specific conditions. The practices presented in Table 2-4 may help reduce and/or control unintentional PFAS releases to the environment.

2.e.i Activated carbon

Activated carbon (discussed more extensively in Section 3.a.i) is manufactured from carbon-rich sources such as coal or wood and is used to treat water or air by filtering contaminants (U.S. EPA, 2012). Activated carbon can be used in either PAC or GAC form. In PAC treatments, carbon is powdered and added to the water for treatment, and then chemicals in the water stick to the powdered carbon as the water passes through and typically carries the PAC along. In GAC treatments, chemicals in water adsorb to small pieces of carbon in a bed as the water passes through.

PAC application will remove some PFAS from water, though the amount of PFAS removed depends on many factors. For PAC treatment, the residual stream is the sedimentation sludge or the filter backwash solids that contain the PFAS-laden PAC along with the coagulant, coagulant aids (if used), natural particulates, and enmeshed organic carbon (Dudley et al., 2015). Given the many conditions that affect treatment, the weight percent PFAS in a PAC residual stream varies by many orders of magnitude but will be in the same range as GAC treatment, as discussed below.

For GAC, the range of PFAS concentrations on spent media can be estimated in several ways (Crone et al., 2019; U.S. EPA, 2020d). A conservatively high loading of PFAS onto GAC (using a GAC with a high-capacity potential, a strongly adsorbing PFAS, few competitive contaminants, low organic levels, and a high concentration in the feed water treated) is on the order of 2 percent by weight (e.g., 0.02 grams PFOS per gram of GAC). For more common treatment scenarios, or for PFAS that are adsorbed more weakly (i.e., the shorter-chain PFAS), the weight percent of PFAS will be orders of magnitude lower—about 1×10^{-5} grams PFAS per gram of GAC.

GAC can be reactivated and reused (see Section 3.a.i.2 for a discussion on GAC reactivation). If the GAC is landfilled, PFAS can desorb off the carbon into the landfill leachate if the GAC comes into contact with a water stream that contains low PFAS, or more strongly adsorbing constituents that can displace the PFAS. Desorption will occur due to the disequilibrium between the liquid and solid phases and will accelerate at higher temperatures where adsorption becomes less favorable for exothermically adsorbed contaminants. Therefore, the resulting PFAS concentrations in the landfill leachate impacted by spent GAC will vary greatly depending on conditions.

Typically, deep bed GAC does not require backwashing because the particulates in the water are removed ahead of the GAC system, but there may be instances where particulates deposit onto the GAC, creating a headloss that will require backwashing. If so, the backwash water may contain a PFAS foam that may need to be managed.

2.e.ii Ion exchange resins

Like GAC, ion exchange resins are utilized in fixed bed adsorbers. Similarly, the residual stream from this treatment is the spent media; however, ion exchange resins cannot be thermally reactivated like GAC. Due to several factors, the market is moving toward single-use media for anion exchange resin treatment with incineration as the final disposal point for the spent resin.

Final PFAS loadings onto resins vary widely. A conservatively high estimate of loading (calculated using a PFAS-selective resin, few competitive constituents, and a PFAS known to adsorb well) is expected to be in the order of 10 percent by weight for the strongly adsorbing PFAS (e.g., 0.1 gram PFOS per gram of resin). The higher percent weight than GAC is due to the high capacities of PFAS-selective resins. Like GAC, more typical scenarios for PFAS that are more weakly adsorbed (i.e., shorter-chain PFAS) will yield much lower average weight percent of PFAS.

Assuming single-use resin is used, landfilling may be less expensive than incineration. If the resin is landfilled, PFAS can desorb off the resin if the resin comes into contact with a water stream whose counter ions can displace the PFAS. The resulting leachate concentrations will vary tremendously depending on conditions. In rare cases, it may be appropriate to regenerate the ion exchange resins with a brine; however, the latest science has shown that an alcohol solvent is needed along with the inorganic exchange ion. This is difficult to do at the full-scale, especially for drinking water facilities. If done in this manner, the resulting regenerate brine will have high concentrations of PFAS and will need to be disposed of appropriately. In rare cases, there may not be a ready disposal option for this regenerate brine and interim storage may be appropriate. Before selecting interim storage, it is important to consider that generating routine volumes of waste can pile up over time, which may become more difficult to store than originally envisioned. Otherwise, regenerate brine may be handled like other high-strength liquid PFAS wastes.

Typically, resin beds do not require backwashing because the particulates in the water are removed ahead of the resin system, but there may be instances where particulates deposit onto the resin, creating a headloss that will require backwashing. If so, the backwash water may contain a PFAS foam that may need to be managed.

2.e.iii High-pressure membranes (reverse osmosis and nanofiltration)

High-pressure membranes are extremely effective for removing many PFAS from water to a high degree (Crone et al., 2019; U.S. EPA, 2020d). As high-pressure membranes are based on a rejection phenomenon, water treatment with them creates a waste stream with potentially high concentrations of PFAS as well as treated water with a lower pH. These waste streams also have high concentrations of salts, other contaminants, and dissolved organic matter.

Treatment of the concentrated residual stream can be challenging, and the cost will likely be high, similar to those for landfill leachates, ion exchange spent regenerates, and waters from highly contaminated sites. Many variables could affect the cost of treating these waste streams. At this time, there is no obvious treatment technology choice, especially given that the concentrated retentate stream is typically 20 percent of flow (Baruth, 2005). This represents a sizeable flow, especially for large membrane treatment systems, such as those used by large municipalities (e.g., treating 20 million gallons per day [4 million gallons per day] concentrate flow). This large-volume flow would prevent the use of batch treatment processes, which have higher efficiencies because they can process the water multiple times before discharge.

When the membranes lose their integrity or foul to the point of needed replacement, they will need to be disposed of. It is not anticipated that the materials of construction are high in PFAS unless they are specifically made of PFAS materials (e.g., Nafion). Also, the mass of PFAS remaining on the membrane surfaces should be minimal except in extreme cases.

2.e.iv Regulatory considerations for drinking water treatment utilities

Sites with reject water from high pressure membranes typically have a NPDES permit[7]. To meet NPDES statutory requirements to obtain an NPDES permit prior to discharging any pollutant to waters of the United States, drinking water treatment utilities using membranes must follow all NPDES permit or pretreatment program requirements, including those applicable to PFAS, for any permit or control mechanism issued for their facility. For more information, see EPA 2022d.

2.f Landfill leachate containing PFAS

Landfill leachate (discussed in more detail in Section 3.b.iii.5) is the liquid formed by rainwater percolating through waste in landfills. Leachate generation may continue even after a landfill's closure period, as a result of liquids inherent in the waste or if the cap system fails. There are different types of solid waste landfills characterized by the wastes managed, which also dictate the environmental controls employed. Hazardous waste landfills and most MSW landfills are required to collect the liquid leachate captured within the landfill liner and subsequently manage or treat the leachate. While PFAS concentrations in different landfill leachates have been documented (see Table 3-3 in Section 3.b), there are no monitoring or reporting requirements at the federal level for PFAS in landfill waste or leachate. Thus, existing treatment methods are being used to process leachate irrespective of PFAS concentrations.

Landfill leachate can be treated on site or off site. The most prevalent off-site management approach is to export leachate to a TWTDS, where it is mixed with wastewater and treated. However, as noted in Section 2.a.ii, conventional wastewater treatment technologies are generally unable to treat or control PFAS (Schultz et al., 2006). Other off-site treatment methods include incineration and underground injection (see Sections 3.a and 3.c, respectively). The on-site leachate treatment technologies employed at landfills are explored in Table 3-2 in Section 3.b.i. Some management approaches and treatment technologies represent significant pathways for PFAS release. Unlined impoundments, release to constructed wetlands, and land applications can release PFAS and potentially contaminate groundwater. Additional research is needed to determine the efficacy of landfill leachate treatments for PFAS (see Section 5).

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[7] The Clean Water Act prohibits anybody from discharging "pollutants from a point source" to a "Water of the United States" unless they have an NPDES permit. See 40 CFR Section 122.1(b)(1). The terms "pollutant," "point source," and "Waters of the United States" are defined at 40 CFR Section 122.2.

2.g Summary

PFAS are synthetic chemicals that are used in a diverse array of industrial and commercial applications. Industrial waste streams containing PFAS stem from two main sources: (1) primary manufacturing facilities of PFAS chemistry and (2) secondary industries that use PFAS-containing products and/or manufacture finished products containing PFAS. Ultimately, PFAS end up in solid, liquid, or gas waste streams from industrial facilities that manufacture or use PFAS and PFAS-containing products. Other common PFAS-containing waste streams include AFFF, biosolids, textiles, spent water treatment materials, and landfill leachate. Any of these waste streams can contribute to PFAS entering environmental media, including soil and groundwater. The potential destruction, disposal, and treatment technologies for processing these streams are discussed in Section 3.

2.h References

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3. Technologies for the Destruction and Disposal of PFAS and PFAS-Containing Materials

EPA has identified three existing and potentially available destruction and disposal technologies that may be effective for managing PFAS and PFAS-containing materials. These technologies are thermal treatment (Section 3.a), landfilling (Section 3.b), and underground injection (Section 3.c). Each subsection describes various considerations for these technological solutions, including types of treatment, potential for releasing PFAS into the environment, and uncertainties. References appear at the end of each subsection. See Section 6 for a discussion of emerging technologies that show potential to destroy PFAS. See Appendix D for a summary of costs and considerations.

3.a Thermal treatment

As the industrial and commercial use of fluorinated organic compounds grew over the past 70 years, their presence, compositions, and concentrations in materials that need to be destroyed or disposed of also grew. Organic fluorine-containing materials including flame suppressants, refrigerants, long- and short-chain PFAS, fluoropolymers, and their precursor chemicals have been commonly treated using a variety of thermal treatment technologies. These range from permitted HWCs and industrial furnaces to MWCs, SSIs, and thermal oxidizers designed to treat material and process gas emissions. Compared to landfills and underground injection, thermal treatment offers the potential advantage of mineralizing PFAS and preventing further release into the environment.

The carbon–fluorine bond is much stronger than the carbon–chlorine bond. Breaking the carbon–fluorine bond requires 1.5 times more energy and therefore higher temperatures and reaction times. Based on calculated bond energies, the most difficult fluorinated organic compound to decompose is carbon tetrafluoride (CF_4), which requires temperatures over 1,400°C (2,550°F) (Tsang et al., 1998). This is due to the compound's four carbon–fluorine bonds and symmetrical structure. The presence of carbon–carbon or carbon–hydrogen bonds (as in hexafluoroethane [C_2F_6] or fluoroform [CHF_3]) provides a weak point in the structure and thus significantly lowers temperatures needed for decomposition. Due to their thermal stabilities, short-chain fluorinated carbons such as CF_4, CHF_3, C_2F_6, and octafluoropropane (C_3F_8) may be good indicators of broader PFAS defluorination. In addition, these compounds may be relatively easy to monitor by Fourier transform infrared spectrometry (FTIR) and are targets measured in whole gas samples by Other Test Method 50 (OTM-50)[8], which makes them potential low-risk candidate surrogates for thermal destructibility trials.

Further, fluorinated organic compounds can be destroyed in flames by free radical initiation, propagation, and branching mechanisms. To increase the efficiency of these processes, it is important to provide high concentrations of hydrogen radicals (as in flames) to promote hydrogen fluoride (HF) formation, reducing the strong flame inhibition effects of fluorine radicals. There is concern that carbon–carbon bonds can be broken at moderate temperatures, leaving carbon–fluorine fragments. The fate of these carbon–fluorine radicals will depend on the local temperatures and concentrations of

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[8] EPA recently released the OTM-50 methodology *https://www.epa.gov/emc/emc-other-test-methods*.

other free radicals. If the local energies and free radical concentrations are low, these carbon–fluorine fragments may recombine to form fluorinated PICs. However, if the local temperatures and concentrations of free radicals (particularly hydrogen radicals) are high, as occurs in flames, the carbon–fluorine species is more likely to degrade further to CO_2 and HF.

The stability of perfluorinated radicals and their propensity to recombine present the potential for the creation of PFAS PICs distinctive from the original fluorinated compounds. These reactions are promoted by partial combustion caused by insufficient temperatures, time, and turbulence. Many PFAS are composed of very stable fluorinated carbon chains and relatively weak non-fluorinated functional groups. Often, the functional group is easily removed, allowing the fluorinated chain to react with other radicals and create a variety of compounds, which complicates the determination of destruction and removal efficiencies (DREs) and the identification of PICs (Wang et al., 2015). In addition, the presence of catalytic surfaces, often metals, may promote further reaction and PIC formation in post-combustion regions. PFAS PICs may be smaller in molecular weight than the original species or larger in molecular weight when formed via the recombination of two large radicals.

Thermal treatment units generally use high-temperature combustion and incineration to destroy organic materials and control organic pollutants. Incineration is an effective and permitted method for destroying certain halogenated organic chemicals including chlorinated solvents, polychlorinated biphenyls (PCBs), dioxin-laden wastes, brominated flame retardants, refrigerants, and ozone-depleting substances.

Combustion is described by complex sets of physical processes and elementary reactions involving many intermediate and free radical species. Even combustion of a simple fuel, such as methane, involves several hundred elementary reactions. For more complex fuels, or systems that include additional elements, such as fluorine, elementary reactions can number in the thousands or more. These elementary reactions describe pathways or mechanisms whereby large molecules, such as PFAS, decompose and transform during combustion.

Thermal treatment of halogenated organic compounds generates the corresponding halogen acid when sufficiently high temperatures and long residence times break the carbon–halogen bonds and the resulting halogen atoms react with available hydrogen. Thermodynamically, for a reacting system containing carbon, hydrogen, fluorine, and oxygen, chemical equilibrium favors the formation of carbon dioxide, water, and hydrogen fluoride. Unfortunately, complete decomposition is almost impossible to achieve due to kinetic (temperature) and transport (mixing) limitations. If temperatures are insufficient or mixing is incomplete at any point, complete conversion to hydrogen fluoride and carbon dioxide may not be achieved. This incomplete decomposition is the genesis of many pollutants from hydrocarbon fuels PICs. Certain thermal technologies can potentially achieve the operating conditions necessary to break down fluorinated organic compounds, but not every unit typically operates at these conditions. Data are lacking about the effectiveness of the characteristic operating conditions of the different thermal destruction technologies currently used to treat PFAS materials. Also, more measurement methods are needed to evaluate technology performance for both the destruction of specific chemical species and the formation of byproducts. As a result, EPA does not know if/which thermal treatment technologies are fully able to achieve adequate PFAS destruction and minimization of PFAS PICs.

Hazardous waste combustion technologies (commercial incinerators, cement kilns, and LWAKs) can potentially achieve temperatures and residence times sufficient to break apart the PFAS contained in the waste stream being thermally treated, but not every unit currently operates at those conditions and those that do may not operate at those conditions at all times. To assess the capability of HWCs to destroy the materials fed into them and ensure that harmful emissions into the environment are minimized, it is standard practice to evaluate the DRE of one or several principal organic hazardous constituents (POHCs) during a trial burn. POHCs are selected to be at least as difficult to destroy as the feed material, based on an incinerability index scale of how difficult a compound is to destroy. The existing index does not include PFAS compounds. Permitted hazardous waste facilities have stringent regulatory controls on temperatures and other important operating parameters and require greater than 99.99 percent DRE for each selected POHC. Data from pilot-scale experiments conducted by EPA researchers indicate that "several operating conditions above 1090 °C resulted in high destruction efficiency (DE)[9] and few detectable fluorinated PIC emissions. However, several conditions below 1,000 °C produced DEs greater than 99.99 percent for the quantifiable PFAS and mg/m^3 emission concentrations of several nonpolar PFAS PICs. These results suggest that DE alone may not be the best indication of total PFAS destruction, and additional PIC characterization may be warranted" (Shields et al., 2023).

For HWCs, DRE is calculated based on the amount of a specific chemical entering an incinerator and the amount of that same chemical exiting the incinerator. While the compound selected for DRE determination in an HWC is chosen based on the difficulty of destroying it, DRE does not account for chemical reactions or transformations into new/different organic compounds (i.e., PICs). Because DRE does not account for chemical transformations, it is not an accurate indicator, by itself, of complete PFAS destruction. Recent research (Shields et al., 2023) has demonstrated that PIC formation can occur even when high DRE is measured at temperatures below 1,000 C. It is not accurate to state that a high DRE by itself indicates complete destruction of PFAS and that no other PFAS are being emitted. Complete PFAS destruction, also referred to as "complete mineralization," results in the production of only HF and CO_2 and does not include the production of other PFAS or other fluorinated organic compounds. The term "adequate destruction" is used in this guidance because it is not feasible to measure complete mineralization due to methodological limitations.

To fully evaluate whether adequate destruction is occurring, EPA and other investigators need to sample and analyze for a range of chemicals exiting a thermal treatment device, including the specific PFAS in the feed material, other PFAS beyond those in the feed (i.e., potential PICs), and HF. EPA currently lacks test methods for measuring most PFAS before and after the incineration process, but new methods are being advanced that will provide more information in the coming years. Currently, standard methods are available for HF (e.g., EPA Method 26A) and a targeted list of semivolatile PFAS (i.e., Other Test Method 45 [OTM-45]), and other test methods are in development for a relatively small percentage of PFAS as surrogates for the class of PFAS compounds (see Section 3.a.iv).

For this second version of the interim guidance, PFAS destruction is defined as the severing of all carbon–fluorine bonds in a PFAS molecule and the mineralization of carbon and fluorine to CO_2, HF, and water. HF and some of the other products of combustion can be effectively removed in pollution control devices designed for acid gas removal. The PFAS Thermal Treatment Database (PFASTT) is a growing database that contains more than 2,000 records of 80 sources documenting the treatability of PFAS in different media via various thermal processes.

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[9] Destruction efficiency differs from destruction removal efficiency because samples were collected before any pollution control devices.

EPA currently has limited data characterizing emissions from HWCs that burn PFAS (Focus Environmental, 2021, 2022), along with data from a pilot-scale study at an EPA research facility (Shields et al., 2023). These studies are summarized below and provide some insight on the potential efficacy of thermal treatment in an HWC. To date, the commercial facility studies have focused on the removal of easily transformed PFAS target compounds, which is a significant limitation.

Clean Harbors Environmental Services. Two testing campaigns (2021 and 2022) were conducted at Clean Harbors Environmental Services' hazardous waste incinerator located in Aragonite, Utah (EA Engineering, Science, and Technology, Inc., PBC., 2021, 2022). Each test campaign was divided into three test conditions: Baseline (Condition 1), Augmented PFAS Feed (Condition 2, where known amounts of four or five target PFAS were fed to the incinerator), and Post-spiking Normal Operations (Condition 3, where AFFF was included in the incinerator feed). Clean Harbors used OTM-45 (*Measurement of Selected Per- and Polyfluorinated Alkyl Substances from Stationary Sources*) for stack gas sampling and analysis. The 49 targeted PFAS analytes were measured using liquid chromatography/tandem mass spectrometry (LC/MS/MS) with isotope dilution. Utilizing the available analytical methods at the time, the Clean Harbors study demonstrated a DRE for the legacy PFAS in the 99.9999 percent range at high temperatures. Of the 49 compounds detected by OTM-45, 34 were non-detect in the stack gas and only 15 were detectable at extremely low concentrations. The tests appear to demonstrate that the facility's primary objective was met: "demonstrate the ability of the high-temperature incineration system at Aragonite to effectively destroy PFAS in the waste feed during normal process operating conditions." Although a high DRE was achieved (greater than 99.999 percent) for the five selected PFAS compounds that were subjects of these studies, these tests did not collect data to fully support demonstration of adequate destruction as the production of PICs were not evaluated due to methodology limitations.

EPA PFAS thermal destruction research. Recently, EPA's Office of Research and Development (ORD) completed an initial study examining the relationship between destruction efficiency (DE) and PICs (Shields et al., 2023). This study used a pilot-scale research furnace and atomized a PFOS-dominant AFFF at various furnace locations and combustion environments. These included atomization through the flame with the natural gas auxiliary fuel and at various post-flame locations so that the AFFF would experience decreasing peak temperature exposures ranging from ~1,200°C to ~800°C (2,192°F to 1,472°F). Stack measurements included OTM-45 and a new draft GC/MS-based Other Test Method for non-polar volatile PFAS (OTM-50). Instead of DREs, this study calculated DEs, because samples were collected before any pollution control, i.e., removal equipment, including fabric filter and alkaline wet scrubber. Analysis of the AFFF and OTM-45 samples was used to calculate DEs for 10 PFAS quantified in the AFFF. OTM-50 was used to quantify concentrations of 30 PFAS PICs. Results indicate that several idealized operating conditions with peak temperatures above 1,090°C (1,976°F) resulted in high DEs (greater than 99.99 percent) and low PIC emissions that were near or below the detection limits. However, several conditions below 1,000°C (1,832°F) produced DEs greater than 99.99 percent for the quantifiable PFAS while simultaneously measuring emission concentrations of several volatile PFAS (mg/m^3). The study concluded that DE alone may not be the best indication of PFAS destruction.

Additional studies are necessary to more fully measure and evaluate the ability to approach complete mineralization and eliminate potential formation of PICs. EPA is working to advance new measurement methodologies (see Section 3.a.iv.1) and gather additional information to conclude whether potential PICs are adequately controlled (see Appendix A).

This section focuses on considerations associated with thermal treatment of PFAS and PFAS-containing materials. It discusses:

- The types of thermal treatment units that manage PFAS-containing materials and their ability to effectively destroy PFAS.
- The potential for environmental releases during these thermal treatment operations.
- PFAS monitoring methods relevant to these thermal treatment operations.
- Uncertainties and unknowns associated with thermally treating PFAS-containing materials, including the ability to effectively measure and monitor thermal treatment performance.

3.a.i Types of thermal treatment

The following subsections describe the types of thermal treatment devices that may have directly or indirectly received and treated the PFAS-containing materials identified in Section 2.a. These subsections focus on design and operation parameters that are important for destroying PFAS. Treatment devices described here include HWCs, non-hazardous waste combustors, carbon regeneration and reactivation units, and thermal oxidizers. Waste incinerators are typically classified by the type of waste they combust. Suggested resources for locating such facilities are included in Section 1.c.ii.

3.a.i.1 Hazardous waste combustors

HWCs include hazardous waste incinerators (HWIs), cement kilns, LWAKs, boilers, and hydrochloric acid production furnaces that burn hazardous waste. Two types of HWCs that have treated PFAS waste in the United States are commercial[10] incinerators and LWAKs.

All HWIs, LWAKs, and cement kilns are subject to RCRA and Clean Air Act (CAA) permitting requirements[11,12] that provide additional regulatory oversight and include operating requirements and emission limitations to safely and effectively treat regulated hazardous contaminants that may not be required for non-permitted facilities. These types of HWCs are subject to CAA Title V permitting requirements, and to maximum achievable control technology standards pursuant to Section 112 of the CAA (40 CFR part 63, subpart EEE). These standards include emission limitations for metals, dioxins/furans, particulate matter, hydrogen chloride and chlorine gas, and carbon monoxide or hydrocarbons, as well as limits on minimum organic DRE, but do not include emission limitations for PFAS. Also, under the authority of RCRA's "omnibus" clause (Section 3005(c)(3); see 40 CFR 270.32(b)(2)), RCRA permit writers may impose additional terms and conditions on a site-specific basis as may be necessary to protect human health and the environment. Due to these additional safeguards, permitted HWCs may operate under conditions more conducive to destroying PFAS and controlling related PICs relative to thermal treatment units that do not have both RCRA and CAA permits. Further, incinerators permitted to incinerate liquid PCBs are required to meet even more stringent operating requirements with respect to time and temperature (40 CFR 761.70(a)(1)) when incinerating PCBs, and these operating conditions may be even more conducive to destroying PFAS and controlling PIC

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[10] Commercial thermal treatment units primarily treat waste received from other facilities.

[11] Hazardous waste is regulated pursuant to RCRA authority. See 42 U.S.C. 6903. The regulatory definition is found in 40 CFR 261.3. PFAS is currently not a listed or characteristic hazardous waste, but a PFAS-containing waste may meet the regulatory definition of hazardous waste if PFAS is mixed with a listed hazardous waste or if a PFAS-containing mixture exhibits a hazardous characteristic (e.g., corrosivity or another characteristic stemming from the material that is mixed with PFAS) Waste managers must comply with RCRA hazardous waste requirement for the specific regulated hazardous waste, including storage limits, land disposal restrictions, and utilizing disposal and destruction facilities that are permitted to receive hazardous wastes.

[12] Hazardous-waste-burning cement kilns and LWAKs are a small subset of the total cement kiln and LWAK universe—i.e., most kilns do not burn hazardous waste.

formation. The following sections generally describe design and operational parameters of commercial HWIs, LWAKs, and cement kilns, focusing on parameters that are important for destroying PFAS. Design and operation information summarized for HWCs is generally based on background support documents for previous rulemakings (U.S. EPA, 2005b). Information on the numbers of HWC facilities is based on a 2019 national hazardous waste capacity assessment report (U.S. EPA, 2019). The capacity assessment report also provides a list of these facilities.

3.a.i.1.1 Commercial hazardous waste incinerators

Currently, there are 10 commercial HWI facilities operating in the United States. HWIs are designed to optimize temperatures, residence times, turbulence, and other parameters to ensure compliance with organic DRE requirements.[13] Three of these facilities are also permitted for incineration of PCBs. Most commercial HWIs use rotary kilns as primary combustion chambers to facilitate the thermal treatment of containerized wastes (e.g., in drums) and solid wastes such as contaminated soils. Low-heating-value aqueous wastes may also be pumped into the rotary kiln. The kiln maintains a continuous standing flame fueled by high-heating-value wastes and auxiliary fuels that maintain high temperatures. Typically, solids retention time in the kiln is 0.5 to 1.5 hours, while gas residence time through the kiln is usually around two seconds. Kiln flame/solids temperatures range from 650°C to 1,650°C (1,200°F to 3,000°F) (U.S. EPA, 2005b).

After combustion in the rotary kiln, wastes may be moved to an afterburner or secondary chamber. If an afterburner is used, additional high-heating-value gaseous and liquid wastes and auxiliary fuels may be added. The afterburner/secondary chamber is typically operated at about 1,100°C to 1,370°C (2,000°F to 2,500°F) with a gas residence time from one to three seconds to maximize organic destruction and minimize the formation of PICs (U.S. EPA, 2005b). Depending on the fuels used and waste streams introduced, combustion products might include carbon dioxide, water, nitric oxide, a variety of acid gases (sulfur dioxide, hydrochloric acid, HF, etc.), organic PICs, fly ash, and bottom ash constituents. Bottom ash is removed at the end of the kiln and from the bottom of the afterburner/secondary chamber and is typically disposed of in hazardous waste landfills (see Section 3.b). Fly ash entrained in the gas is removed downstream by fabric filters (FFs) or electrostatic precipitators, and acid gases are removed by gas scrubbers. Some HWIs use activated carbon injection systems to control mercury, dioxin/furan, and other emissions, and some also use TMT-15 (1,3,5-triazine-2,4,6(1H,3H,5H)-trithione, trisodium salt) injection for additional mercury and metals emissions control.

Limited studies of PFAS treatment in HWIs have been conducted (EA Engineering, Science, and Technology, Inc., PBC., 2021, 2022). These studies demonstrated treatment of five selected PFAS compounds in the feed material. However, these test data are not able to demonstrate adequate destruction of PFAS (Taylor, 2022) due to the lack of data concerning PIC formation.

3.a.i.1.2 Hazardous-waste-burning cement kilns

Currently, there are 11 hazardous-waste-burning cement kiln facilities in the United States. A cement kiln is a long, cylindrical, slightly inclined rotating furnace designed to calcine a blend of raw materials such as limestone, shale, clay, or sand to produce a key ingredient of Portland cement. These cement kilns burn hazardous-waste-derived fuels to replace some of the fossil fuels. Most burn liquid waste; some may also burn solids and small containers containing viscous or solid hazardous waste fuels.

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[13] DRE = $[1 - (W_{out} / W_{in})] \times 100\%$, where: W_{in} = mass feed rate of an organic compound into the combustion device and W_{out} = mass emission rate of the same organic compound in exhaust emissions. HWCs are required to achieve at least 99.99% DRE.

Extreme combustion conditions (e.g., temperatures above 1,650°C [3,000°F]), turbulent mixing, and long gas phase residence time (from four to as high as 16 seconds in long kilns) may effectively treat hazardous waste. Cement kilns use either electrostatic precipitators or baghouses to collect particulate and metal emissions, referred to as cement kiln dust (CKD). Portions of the CKD can be fed back into the kiln as a raw material feed or used in other industries as neutralizers or additives, but usually the excess CKD is land-disposed. Add-on acid gas air pollution control devices, such as wet or dry scrubbers, are typically not used. The high alkaline content of the raw material feeds already prevents or minimizes the formation and release of acid gases by providing for in situ absorption of chlorine and other halogens and sulfur.

Several laboratory studies have examined the potential benefit of treating PFAS wastes in cement kilns where the calcium in the raw limestone can react with fluorine to form calcium fluoride (CaF_2). These studies indicate that calcium species react readily with PFAS at relatively low temperatures (400-800°C) to promote mineralization of PFAS waste (Wang et al., 2011, 2013, 2015; Reidel et al., 2021). However, the currently available lab-scale data are insufficient to determine with confidence the effectiveness of treating PFAS waste in commercial cement kiln operations.

3.a.i.1.3 Hazardous-waste-burning lightweight aggregate kilns

There is one LWAK facility operating in the United States that burns hazardous waste. LWAKs thermally process raw material (clay, shale, and slate) in slightly inclined, rotating furnaces to produce a coarse aggregate used in lightweight concrete products. In hazardous-waste-burning LWAKs, liquid wastes are either blended directly with conventional fuels burned in the hot end of the kiln or pumped separately into the hot end flame. High combustion gas flame temperatures (above 1,650°C [3,000°F]) and high kiln gas residence times (over two seconds) are used to destroy hazardous organics. Kiln exhaust gases leave the cold upper end of the kiln at a temperature from 205°C to 540°C (400°F to 1,000°F). LWAKs use FFs to control dust contained in the exhaust gas. The collected dust can be recycled back into the kiln (at the hot or cold end) or mixed into the lightweight aggregate product. Some LWAKs also use wet or dry scrubbing for acid gas emissions control. For example, the currently permitted hazardous-waste-burning LWAK has dry scrubbing for acid gas control. There are insufficient data to determine the effectiveness of commercial LWAK facilities in treating PFAS waste.

3.a.i.2 Carbon regeneration and reactivation units

Carbon reactivation units or “furnaces” use high temperatures to thermally desorb contaminants from GAC, which allows for the carbon to be used again. Over a dozen large-scale companies and utilities in the United States reactivate sizeable quantities of GAC. In all, these entities operate about 17 commercial furnaces (Roskill Information Services Ltd., 2017). Four of these commercial furnaces operate under RCRA permits and applicable air permits. RCRA permits provide additional regulatory oversight and include operating requirements and emission limitations to promote safe and effective treatment of the hazardous contaminants; non-RCRA-permitted carbon reactivation furnaces may not be subject to such requirements. Due to these additional safeguards, RCRA-permitted furnaces may operate under conditions more conducive to destroying PFAS and controlling related PICs, relative to furnaces without RCRA permits. This discussion focuses on RCRA-permitted furnaces because EPA has more design and operational information on these devices as a result of the RCRA permitting process.

Reactivation[14] of spent carbon is generally carried out in multiple-hearth ("multi-hearth") or rotary kiln furnaces, although fluidized bed and infrared furnaces are also options. While the furnace designs vary, they all use high temperatures and residence times designed to eliminate the adsorbed contaminants and return the carbon to a virgin state for reuse.

During reactivation, spent GAC is typically exposed to drying, desorption, pyrolysis, and oxidation as it moves through the furnace. The drying stage eliminates moisture via evaporation and occurs when hot combustion gases at temperatures ranging from 100°C to 110°C (212°F to 230°F) contact the carbon. During the desorption stage, hydrocarbons, hydrogen, and water vapor escape the pores of the granular carbon at temperatures of 315°C to 400°C (600°F to 750°F). The pyrolysis stage eliminates any volatile compounds adsorbed within the carbon porosity, including residual moisture, and thermally decomposes other less-volatile compounds. Pyrolysis occurs when the carbon is exposed to temperatures up to 800°C (1,472°F) under inert conditions (i.e., low oxygen). A residue of carbonized char is formed from the adsorbed compounds during pyrolysis, which occupies some of the carbon porosity and must be removed in the next stage. The oxidative stage involves the controlled gasification of the pyrolyzed carbon at temperatures of around 800°C (1,472°F) in the presence of a mildly oxidizing atmosphere, usually steam or carbon dioxide or a mixture of both. This results in the elimination of most of the charred residue and exposes the original carbon-pore structure (Miguel et al., 2001).

The residence times required for carbon reactivation vary by unit design and the contaminant loads and concentrations. For example, two facilities using multi-hearth furnaces have residence times of at least 38 minutes at one facility and between 90 and 120 minutes at the second facility (Kentucky Energy and Environment Cabinet, 2019; U.S. EPA, 2018).[15]

To ensure adequate destruction and removal of any remaining contaminants, reactivation furnaces are typically equipped with afterburners/thermal oxidizers. To meet regulatory requirements, afterburners are designed to achieve 99.99 percent DRE via thermal oxidation. DRE for a reactivation/regeneration furnace is evaluated on a "total hydrocarbon" basis rather than for a specific most-difficult-to-destroy component, as is done for HWCs. This adds additional uncertainty to the question of whether a thermal treatment unit can effectively destroy PFAS. The temperature required to achieve 99.99 percent DRE depends on residence time as well as on the concentration of contaminants, but minimum temperatures are around 885°C (1,625°F) and maximum temperatures are as high as 1,316°C (2,400°F), with a minimum residence above one second (U.S. EPA, 2018).

Depending on the process or waste streams treated, a reactivation furnace can be equipped with add-on air pollution control equipment to ensure environmental standards are met. Equipment can include venturi scrubbers for particulate matter control, packed-bed scrubbers for acid gas and particulate matter control, and wet electrostatic precipitators or baghouses for additional particulate matter control. As discussed in Section 3.a.ii, carbon reactivation systems can degrade PFAS even at the lower temperatures (150°C–700°C, or 302°F–1,292°F).

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[14] "Reactivation" refers to a regeneration process that requires high temperatures. Regeneration also includes low-temperature processes, such as those using brines, solvents, oxidants, biological treatment, etc. These processes may not be as effective as reactivation for GAC (AWWA, 2018); therefore, they are not considered for this discussion.

[15] These two references are RCRA permits for Evoqua Water Technologies LLC and Calgon Carbon Corporation, two companies that accept spent activated carbon from off-site sources. They each hold RCRA permits, which allow them to treat spent carbon that meets the definition of hazardous waste.

3.a.i.3 Non-hazardous-waste combustion sources

Non-hazardous-waste incineration units in the United States includes SSIs at wastewater treatment facilities and MWCs. Under Section 129 of the CAA, these units are regulated for emissions of particulate matter, carbon monoxide, dioxins/furans, sulfur dioxide, nitrogen oxides, hydrogen chloride, lead, mercury, and cadmium. Because of this, the air pollution control devices employed by these units are intended to control these nine pollutants primarily. The following sections generally describe these incinerator types and their design and operational parameters that may influence the treatment of PFAS-containing waste (U.S. EPA, 1995).

3.a.i.3.1 Sewage sludge incinerators

An SSI unit is a combustion device that is used to burn dewatered sewage sludge. There are currently 170 SSI units operating at 86 facilities in the United States (U.S. EPA, 2016). The main types of SSIs are multi-hearth furnaces and fluidized bed combustors (FBCs).

The basic multi-hearth furnace is a vertical cylinder divided into zones. The sludge is dried at temperatures from 425°C to 760°C (800°F to 1,400°F) (U.S. EPA, 1995). Sludge combustion occurs as the temperature is increased to about 925°C (1,700°F) in successive zones. The gas residence times are typically four or five seconds. Emission controls on multiple hearths can include wet scrubbers, wet electrostatic precipitators, afterburners, and regenerative thermal oxidizers.

An FBC consists of a vertically oriented outer steel shell with nozzles designed to deliver fluidizing air at the base of the furnace within a refractory-lined grid. Air is injected into the furnace to fluidize the sludge and the sand. The combustion of the sludge occurs at temperatures from 750°C to 925°C (1,400°F to 1,700°F). The gas residence times are typically two to five seconds. Emission controls on FBCs can include venturi scrubbers, multicyclones, FFs, activated carbon injection, and carbon bed absorbers. There is substantial variation in the design and operation of SSIs and it has not been shown that standard operating conditions are effective for the treatment of PFAS. The data currently available are insufficient to make a determination about the effectiveness of SSIs in treating PFAS.

3.a.i.3.2 Municipal waste combustors

There are 193 MWC units operating at 75 facilities in the United States (Michaels & Krishnan, 2018). Three main classes of technologies are used to combust MSW: mass burn, refuse-derived fuel (RDF), and modular combustors. Mass burn and RDF combustors are the predominant designs.

With mass burn units, the MSW is combusted without any preprocessing other than removal of items too large to go through the feed system or hazardous materials, such as pressurized containers. In a typical mass burn combustor, refuse is placed on a grate that moves the waste through the combustor. The grates typically have three sections. On the initial grate section, referred to as the drying grate, the moisture content of the waste is reduced before ignition. Evaporated moisture is combined with the moisture from combustion and treated with the resulting combustion gases. The second grate section, referred to as the burning grate, is where most of the active burning takes place. The third grate section, referred to as the burnout or finishing grate, is where remaining combustibles in the waste are burned. Typical combustion temperatures for mass burn units can range from 800°C to 1,100°C (1,500°F to 2,012°F) (Reddy, 2016).

RDF combustors burn waste that has been processed to varying degrees to raise its heating value and provide a more uniform fuel. Most boilers designed to burn RDF use spreader stokers and typically operate at around 680°C (1,250°F). RDF-fired FBCs typically operate at bed temperatures of around 815°C (1,500°F) (Reddy, 2016).

Residence times of gases within MSW combustors vary from unit to unit, depending on design and operational factors such as furnace volume, excess combustion air percentage, whether flue gas recirculation is employed, and combustor operating load parameters (Scavuzzo, 1990; Themelis & Reshadi, 2009). Overall combustion air residence times have been calculated in the 7- to 10-second range for a small sampling of MWC design loads (Themelis & Reshadi, 2009). At temperatures above 980°C (1,800°F), the approximate residence time is about two seconds for full combustor loads (Scavuzzo, 1990).

Emission controls on MWCs can include spray dryer or dry sorbent injection, electrostatic precipitators, FFs, selective or non-selective catalytic reduction, and activated carbon injection.

There is substantial variation in the design and operation of MSW combustors. The data currently available are insufficient to make a determination about the effectiveness of MSW combustors to treat PFAS.

3.a.i.4 Thermal oxidizers

Thermal oxidizers are used to treat process gases and destroy volatile organic compounds (VOCs) and organic hazardous air pollutants (HAPs) at a manufacturing or production facility. Some types of thermal oxidizers are also able to accept liquid feed streams. Each thermal oxidizer is designed to optimize temperatures, residence times, and turbulence to address the composition of the particular feed stream at that unit and meet the requirements of a regulation or permit. Depending on the type of thermal oxidizer (direct-fired, regenerative, recuperative, or flameless), the operating temperature ranges from 760°C to 1,200°C (1,400°F to 2,190°F). The residence time of thermal oxidizers ranges from 0.5 to two seconds, depending on site-specific criteria. Catalytic thermal oxidizers operate at lower temperatures than other types of thermal oxidizers—typically at about 400°C (750°F) with similar residence times to non-catalytic thermal oxidizers—while achieving the same efficiency in destruction of VOCs.

Emission controls for thermal oxidizers vary widely depending on the facility and the composition of the feed streams. Existing thermal oxidizers may have no additional pollution controls or may use prefiltration (to prevent fouling of the oxidizer) and/or wet or dry scrubbers or FFs. Thermal oxidizers have historically not been designed with destruction of PFAS as the primary focus, so most currently installed thermal and catalytic oxidizers may not be optimized for PFAS destruction. However, some thermal oxidizers are being used for halogenated wastes, including PFAS precursors. The operating conditions for some thermal oxidizers have the potential to effectively treat PFAS, but the data currently available are insufficient to make a determination about effectiveness.

Chemours. Two testing campaigns (2020 and 2022) were conducted on a thermal oxidizer installed to control PFAS emissions at a Chemours chemical and plastics manufacturing plant located in Fayetteville, North Carolina. The objective of the tests was to determine how the thermal oxidizer and 4-stage scrubber system would achieve a 99.99 percent emission reduction, including the use of a surrogate for all PFAS, such as the hexafluoropropylene oxide (HFPO). The test reports demonstrate a PFAS destruction efficiency above the 99.99 percent for the specific compounds that were tested. HFPO was tested as a surrogate for all species of PFAS, but there are no concrete data showing the performance of the thermal oxidation process for PFAS species outside of the compounds tested. There was also no discussion of possible products of incomplete combustion in the reports. The thermal oxidizer at the

Chemours facility appropriately destroys the five PFAS compounds of interest that have been identified at the facility. However, removal processes for products of incomplete combustion or of destruction of potential compounds not studied but potentially found in the facility waste streams are still unclear.

3.a.ii Limitations and considerations in thermal treatments

Incinerator designs vary, resulting in differing operational and waste feed approaches (see Section 3.a.i). HWIs typically operate at very high average temperatures and employ auxiliary, primary, and secondary flames. MWCs typically operate at lower temperatures and often do not employ auxiliary primary or secondary flames. SSIs vary in design, often operating as dryers with very low temperatures. Even within the same incinerator, wastes can be introduced at different locations and can experience different time, temperature, and mixing histories. PFAS introduced into a hazardous waste rotary kiln incinerator's main burner, along with auxiliary fuel, may experience very different conditions than the same waste introduced to the kiln as contained charges with solid wastes. These factors are expected to affect PFAS destruction and PIC formation. Limited studies have investigated the influence of various factors on PFAS destruction and PIC formation (see Section 3.a).

Carbon reactivation systems can degrade PFAS even at the lower temperatures (150°C–700°C, or 302°F–1,292°F) seen in bench-scale research studies (Watanabe et al., 2018). Experimental data suggest that thermal destruction of PFAS occurs in two stages: first during reactivation of the GAC, and then when the off-gas is introduced into a high-temperature zone with temperatures as high as 1,000^{O}C (Forrester, 2018; Watanabe et al., 2016, 2018; Xiao et al., 2020). DiStefano et al., (2022) showed >99.99 percent destruction of PFAS at a full-scale commercial reactivation facility with a large percentage of the PFAS destruction occurring in the furnace. The fluoride mass balance was reported to be 61.4 percent. Although this low percentage could be due to a number of factors, it may be due to the formation of PICs that were not quantitated in the analytical method. While carbon reactivations systems, with the concomitant use of off-gas incineration (i.e., afterburners) and gas scrubbing units, have the potential to remove PFAS from the activated carbon and destroy it, there is insufficient data to make a determination on its effectiveness. However, as discussed elsewhere in this interim guidance, more work is needed for confirmation, particularly with regard to full-scale reactor conditions, differing carbons, and PICs.

Thermal oxidizers have historically not been designed with destruction of PFAS as the primary focus, so most currently installed thermal and catalytic oxidizers may not be optimized for PFAS destruction. However, some thermal oxidizers are being used for halogenated wastes, including PFAS precursors. Thermal oxidizers are being employed to destroy PFAS-containing and gaseous streams, but the data are insufficient to allow conclusions on the overall efficiency of thermal oxidizers in PFAS destruction. For example, the two tests at Chemours Company's Fayetteville Works (Focus Environmental, Inc., 2020, 2022) focused on five specific PFAS of interest found in waste gas streams at the facility. While the tests seem to prove a DRE of 99.999 percent for the five PFAS compounds, they do not present data on overall PFAS destruction or potential formation of PICs. (See Appendices B and C for summaries of the Clean Harbors and Chemours test data.) EPA is currently unaware of any catalytic oxidizers being used specifically for the destruction of PFAS, particularly in light of their site-specific design and optimization. Though the efficacy of thermal and catalytic oxidizers in destroying PFAS is currently unknown, a properly optimized thermal oxidizer can readily achieve a DRE of 99.99 percent for VOCs.

In addition to incinerators and thermal oxidizers, cement kilns are also used for the destruction of hazardous wastes. Cement kilns operate at very high temperatures (exceeding 1,800°C [3,270°F]), exhibit very large gas and solid residence times, and have the added advantage of providing a caustic environment for halogen reaction and acid neutralization. A cement kiln in Australia has received an operating permit from the government of Queensland to burn PFAS wastes. The permit requires annual monitoring for 21 PFAS (Department of Environment and Heritage Protection, 2018). Test results from 2017 indicate that a DRE in excess of 99.99 percent is possible for specific PFAS compounds at the conditions of the test (Cement Australia, 2017), but the results do not present data on overall PFAS destruction or potential formation of PICs.

Processes involving calcium and alumina may have benefits and may require lower energies to destroy PFAS. Recent research (Wang et al., 2011, 2013, 2015) has investigated PFAS interactions with calcium oxide (CaO) and calcium hydroxide ($Ca(OH)_2$) at moderate temperatures (200°C to 900°C, or 390°F to 1,650°F) and found that these calcium species exhibit a pseudo-catalytic effect promoting PFAS destruction and fluorine capture at relatively low temperatures. Similar studies by the electronics industry indicate that alumina catalysts can effectively convert residual CF_4 and related species (generated by plasma arc techniques for etching purposes) to CO_2 and HF at comparable temperatures.

It is not well understood how effective high-temperature combustion is at completely destroying PFAS or whether the process can form fluorinated or mixed halogenated organic byproducts. Few experiments have been conducted under oxidative and temperature conditions representative of different field-scale incineration devices used for PFAS destruction. Limited studies on the thermal destructibility of fluorotelomer-based polymers found no detectable levels of PFOA after two seconds of residence time at 1,000°C (1,830°F) (Yamada et al., 2005; Taylor et al., 2014). Two recent field studies examining the performance of a specially designed thermal oxidizer in destroying PFAS in process gases at a Chemours production facility near Fayetteville, North Carolina, indicated greater than 99.999 percent destruction of five PFAS (Focus Environmental, Inc., 2020, 2022). Similarly, the recent Clean Harbors studies appear to show that the facility was able to achieve its primary objective: "demonstrate the ability of the high-temperature incineration system at Aragonite to effectively destroy PFAS in the waste feed during normal process operating conditions." A high DRE was achieved (> 99.999 percent) for the five selected PFAS compounds that were subjects of these studies (EA Engineering, Science, and Technology, Inc., PBC., 2021, 2022). However, these test data did not include characterization of potential PICs due to previous methodology limitations.

The Chemours and Clean Harbor test results indicate the PFAS compounds measured were changed by the treatment, but the tests did not investigate whether the compounds were changed to carbon dioxide and hydrofluoric acid to show adequate destruction of the original PFAS. As discussed previously, emission studies, particularly for potential PICs, are largely incomplete due to lack of validated sampling and measurement methods for the potentially large number of fluorinated and mixed halogenated organic compounds that might be formed. EPA continues to seek information on PFAS that may be present in air pollution control device media (e.g., scrubber water, particulate matter control device media) and the presumed effectiveness of these air emission controls. (See Section 5 for more information about ongoing research and research needs.)

Incineration may be a viable PFAS destruction technology if done under certain conditions. The preliminary research on thermal treatment of PFAS suggests that the minimum conditions for PFAS destruction include well mixed environments with temperatures greater than 1,100°C and optimized combustion stoichiometry (Shields et al., 2023). These conditions may adequately destroy PFAS and minimize PICs. However, evaluations have only been done on liquid fired materials at a pilot scale. No comparative studies that include PIC measurements on solids or containerized wastes have been performed. Liquids should preferably have direct flame contact or at least well mixed exposure to temperatures greater than 1,100°C.

EPA encourages additional tests be performed to demonstrate the ability of a thermal treatment technologies to mineralize PFAS. EPA encourages testing to demonstrate destruction of parent compounds (OTM-45) and volatile PICs (OTM-50) based on suggested procedures in Appendix A) to show efficient thermal destruction of atmospherically transformative PICs of health and environmental concern.

3.a.ii.1 Other thermal treatment byproducts of concern

Thermal treatment systems including HWIs, MWCs, and SSIs are configured such that gas phase effluent from the combustion chambers passes through a series of pollution control devices to remove acid gases, particulate matter, and sometimes mercury and other specific HAPs. The behavior of PFAS and PFAS-related PICs in these unit operations is largely unknown (see Section 5 for more information about research on this topic and others). Additionally, these control devices produce secondary waste streams in the form of fly ash and scrubber blowdown solutions, and PFAS and PFAS-related PICs may be present in these solid and liquid effluents depending on their vapor pressure and solubility.

Thermal oxidation processes used for treating PFAS-containing waste, such as incineration, generate HF as the most stable product from combustion of fluorocarbon compounds. HF is a CAA-listed HAP. Highly corrosive, HF creates significant operation and maintenance issues by damaging thermal system materials such as furnace refractory and metal ductwork. Both wet scrubbing and semi-dry scrubbing processes have been applied to control HF emissions from thermal treatment sources.

A thermal oxidizer with a potential for HF emissions typically uses a wet scrubber integrated with the oxidizer to control HF emissions. Hot flue gas exiting the oxidizer is cooled rapidly in a quenching unit, and HF (which has high water solubility) is removed by the quenching water. The cooled flue gas then flows up through a multistep wet scrubbing tower for further HF removal by scrubbing water. Flue gas is scrubbed by a sodium hydroxide solution to neutralize the residual HF as the final scrubbing step. After exiting the tower, flue gas is emitted through a stack. All effluents, including those from the quenching unit and scrubber tower, are mixed with a $Ca(OH)_2$ solution in a reactor where calcium is combined with fluorine and precipitation of water-insoluble calcium fluoride (CaF_2) occurs. After dewatering, dry CaF_2 is sent to a landfill for disposal (see Section 3.b) or used to produce fluorine gas for new PFAS production, and wastewater is discharged from the plant after it is treated by activated carbon to remove trace fluorinated contaminants. Rapid cooling of hot flue gas is known to be effective in reducing catalytic reformation of chlorinated PICs such as dioxins during cooling of incineration flue gas.

A thermal oxidizer equipped with a quenching unit to treat PFAS-containing wastes may also limit catalytic reformation of fluorinated PICs if they are actually formed in the oxidizer. Those PICs may be subsequently transferred into the liquid phase in the wet scrubber. Those PICs could then be partitioned between solid CaF_2 and water in the precipitator, and most of the PICs retained in water are then removed by activated carbon adsorption. EPA is not aware of peer-reviewed studies that measure levels of fluorinated contaminants remaining in both the treated scrubber water stream and the dry CaF_2 stream. Such measurements could be useful for evaluating the potential environmental impacts of byproducts and residuals generated from thermal oxidation of PFAS-containing wastes.

Spray dryer absorber (SDA) technology has been applied to control emissions of halogen acids (including HCl and HF) from both MWCs and HWIs. This semi-dry scrubbing process is designed to inject an alkaline slurry, typically lime, to control acid and fly ash. Water in the fine slurry droplets is vaporized by heat carried by the flue gas, and the lime carried in the droplets neutralizes the halogen acids simultaneously in this two-phase reaction process. The cooled flue gas carries the dried acid neutralization product downstream to a particle collection device, typically an FF. PAC may also be injected into flue gas upstream of the FF to control emissions of mercury and chlorinated dioxins/furans from both MWCs and HWIs. Fly ash, dried acid neutralization product, and PAC are captured by the FF. The SDA/FF with PAC injection flue gas cleaning train produces no scrubber water. The addition of lime (a calcium compound) into the flue gas is known to be effective for forming CaF_2 through hydro-defluorination of PFOS at a moderate temperature of about 350°C (660°F) (Wang et al., 2015); this suggests that the SDA may provide a potential co-benefit of controlling fluorinated PICs. The injection of PAC upstream of the FF may create another potential co-benefit for capturing fluorinated PICs. Studies evaluating PFAS mitigation via SDA/FF with PAC injection (e.g., see research activities in Section 5) will help develop data on this potentially viable technology option.

3.a.iii Potential for releases for thermal treatment technologies

Thermal treatment devices that may be used to treat PFAS-containing waste (see Section 3.a.i for descriptions of these devices) are located in both rural and more densely populated areas throughout the United States. Two possible sources of PFAS emissions from thermal treatment are stack emissions and the management of scrubber water and bottom ash/fly ash. As previously discussed, emissions from thermal treatment activities may contain PFAS if adequate combustion conditions are not achieved or if adequate acid gas scrubbers or other pollution control devices are not used. EPA is aware of a limited number of peer-reviewed studies that have investigated the extent to which emitted PFAS are transported and deposited to surrounding areas (see Section 2.a.iii). EPA plans to conduct research in this area to better characterize the extent to which PFAS deposition may occur near thermal treatment devices (see Section 5 for more information about these types of research activities).

Volatile PFAS releases from thermal treatment equipment (e.g., fugitive emissions) and waste storage activities are another potential source of PFAS releases, if they are not adequately controlled.[16] There is also the potential for releases from management of thermal treatment process residuals, such as air pollution control device media, incinerator bottom ash, and liquids from acid gas scrubbers. For example, incinerator bottom ash disposed of in a landfill would be a possible source of release if the ash contained PFAS and the landfill lacked adequate controls (see Section 3.b for more information on landfill types and controls, and Section 4 for more information about environmental releases and vulnerable populations). HWIs, however, typically dispose of incinerator bottom ash in hazardous waste landfills, minimizing the potential for the release of PFAS to the environment. Again, information on partitioning of PFAS in control technology residuals is lacking at present.

Potential impacts to communities from releases from HWIs should also be considered. Considerations should include which media may be impacted by releases, and the potential exposure pathways resulting from releases. The characteristics of the potential exposed communities (demographics, socioeconomic status, other environmental exposures, health conditions, etc.) provide information regarding the potential for disproportionate and adverse impacts.

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[16] For example, CAA and RCRA regulations applicable to HWCs require controls to prevent/minimize combustion system leaks (e.g., due to positive pressure events in the combustion chamber), as well as controls to minimize releases from equipment and tanks that store or manage hazardous waste. See 40 CFR 63.1206(c)(5) and 264.1050–1090.

3.a.iv Testing and monitoring

Validated measurement methods are limited and under development for reliably identifying and quantifying if PFAS are released into the air from stationary sources. The limited scope of standardized methods to measure PFAS emissions and the limited availability of data using these methods to evaluate PFAS destruction at a variety of treatment devices introduce uncertainty in the understanding of the efficacy of thermal treatment approaches for destroying PFAS.

Ongoing method development by EPA has a broad focus that will address PFAS in various waste streams, stack and fugitive emissions, and ambient air (see Section 5 and Appendix A). Method development also includes identification of PFAS that are potential PICs, not targeted in established water methods such as Methods 533 and 537.1. Due to the number and complexity of PFAS that have already been identified, a simpler class-specific measurement is also being investigated to determine if measurement of a single compound or a small group of compounds is adequate to characterize the completeness of thermal treatment for both targeted PFAS and potential PICs.

These factors raise important risk communication challenges, particularly for communicating the potential significance of testing and monitoring results to adjacent communities. But the importance of encouraging appropriate information access for and dialogue with communities—and, in particular, with potentially vulnerable communities —cannot be overemphasized. Suggestions for risk communication are included in Section 4.

3.a.iv.1 Semivolatile PFAS sampling and analysis

Initial development of methods to measure PFAS in air or air sources has focused on compounds whose physical state is liquid or solid at room temperature. Methods for air have included the target compounds commonly found in water methods. The basis for measuring PFAS compounds from air or air sources involves evaluating and then modifying existing sampling and analysis methods for semivolatile organic compounds.

Filtered particulate, solid sorbent, and aqueous impinger media sampling procedures have been used to collect and recover a wider range of PFAS and associated byproducts in ambient air and source emissions (Barber et al., 2007; Martin et al., 2002). However, these sampling methods have some limits, including poor retention or chemical conversion of the PFAS during sampling and poor recovery during sample preparation prior to chemical analysis (Arp & Goss, 2008).

Ambient sampling for semivolatile PFAS roughly follows either the high-volume air sampling protocol described in EPA compendium Method TO-13a (U.S. EPA, 1999) or the National Atmospheric Deposition Program (NADP) approaches for wet and dry deposition sampling (NADP, 2020). High-volume air samples collect both water-soluble PFAS acids and salts and water-insoluble telomer alcohols. NADP sampling has focused on condensable and particulate-bound targeted PFAS captured in polypropylene buckets to evaluate deposition due to rain.

Current sampling of stationary source air emissions for semivolatile PFAS targets is based on EPA SW-846 Method 0010. Analysis of samples for a list of PFAS compounds is based on EPA Method 533—modified to include collection of both targeted and non-targeted PFAS in a single sampling system. Sampling includes heated or stack temperature probe extraction of emission gases followed by collection on filters, XAD sorbent media, and aqueous impingers. EPA released OTM-45, *Measurement of Selected Per- and Polyfluorinated Alkyl Substances from Stationary Sources,* in January 2021 (U.S. EPA, 2021). OTM-45 experience indicates the method works well to measure targeted PFAS. OTM-45 is an effective and sensitive method to measure a list of PFAS included in the method. In coordination with the release of this guidance, EPA recently released OTM-50, which enables analysis of non-polar volatile PFAS.

These field sampling procedures collect samples that are subsequently transported to a laboratory for extraction and analysis. Analysis procedures include established water methods for targeted compounds and/or non-targeted analysis (NTA) for unknown PFAS. NTA is a research tool that can be used on OTM-45 and OTM-50 samples to investigate semivolatile polar and nonpolar PFAS recovered by these methods. High-resolution mass spectrometry can be used for both targeted analysis and NTA. Qualitative identification of PFAS by NTA reveals PICs/degradants formed during the thermal treatment of PFAS-contaminated media (Aleksandrov et al., 2019; McCord & Strynar, 2019; Newton et al., 2020). NTA currently relies on mass spectrometry, which generates qualitative information about the molecular formula of unknown PFAS. NTA is an important component in characterizing thermal treatment emissions because it provides an approach for identifying unknown PFAS or PICs.

3.a.iv.2 Gaseous volatile PFAS sampling and analysis

Volatile PFAS targets and thermal treatment byproducts from ducted emissions or in ambient air have been sampled using a variety of whole gas sample collection approaches, such as Tedlar® bags and passified canisters, as well as sorbent traps and cryogenic solvents. EPA and private sector investigators have sampled with specialized commercial sorbent traps and Tedlar® bags in laboratory-scale thermal destruction and ambient volatile PFAS measurement of targeted and non-targeted PFAS (Wang et al., 2013, 2015; Yamada et al., 2005). Issues such as sample reactivity, breakthrough volumes, and quantitative transfer to the analysis instrument complicate these approaches. Direct instrumental methods for measuring volatile PFAS (such as FTIR) can suffer from lack of sensitivity compared with extractive methods that allow concentration prior to analysis.

To develop more sensitive methods for measuring volatile fluorocarbon compounds, EPA has evaluated the use of steel canisters for targeted volatile PFAS as well as PICs during multiple-source emissions tests, including at a thermal treatment facility for AFFF-contaminated soil (U.S. EPA, 2020b). Canisters have been used to sample source emissions and perform targeted measurements for PFAS, including both industrially produced compounds (such as FC-116, HFC-23, HFC-125, E1, and E2) and potential PIDs (such as CF_4 through octadecafluorooctane). EPA is developing the ability to characterize a more diverse set of possible PFAS PICs and measure volatile PFAS at trace concentrations. The ability to measure trace concentrations will allow investigators to introduce a known concentration of a hard-to-destroy fluorocarbon and evaluate the behavior of this compound when exposed to specific thermal environments. EPA's preferred approach is to introduce a known concentration of a hard-to-destroy fluorocarbon and evaluate the behavior of this compound when exposed to specific thermal environments in order to characterize a more diverse set of possible PFAS PICs. Though insufficient by itself, this approach is consistent with the EPA approach for determining the DRE of fluorinated greenhouse gas abatement equipment in electronics manufacturing (U.S. EPA, 2010). NTA has also been performed on the same samples. Simultaneous with the release of this 2024 interim guidance, EPA will release OTM 50, which provides an approach and procedures for using silanized steel canisters to sample and analyze volatile PFAS compounds from thermal treatment sources. This and OTM-45 are examples of monitoring methods that could be considered for addition to a potential PFAS waste management system or permit (e.g., RCRA).

EPA continues to evaluate FTIR as a suitable real-time measurement technique for CF_4 and C_2F_6 in stationary source emissions. In addition, in ambient air, EPA also investigated chemical ionization mass spectrometry (CI/MS) to monitor individual PFAS in real time (Riedel et al., 2019). EPA found the technique sufficiently sensitive for fugitive emissions measurements or leak detection.

The quantitative measurement of total organic fluorine (TOF) is also being evaluated to represent all, or most, of the PFAS class of compounds. This approach represents a simpler and more comprehensive measurement alternative to target list approaches that focus on a limited number of PFAS. Several potential techniques warrant consideration and additional evaluation, including (but not limited to) combustion–ion chromatography (CIC), particle-induced gamma emission spectrometry (PIGE), and X-ray photo-electron spectroscopy (XPS). For air and thermal treatment emissions, TOF must measure highly volatile as well as semivolatile PFAS. TOF analysis is an ongoing research area: data users must recognize the benefits of receiving general screening data for a wide array of potentially present PFAS, while also recognizing the limitations and uncertainties inherent in not knowing which PFAS or class of PFAS is present in the sample, including uncertainties associated with potential health risk. In addition, to minimize the risk of PFAS false positives, techniques within a validated method or methods must be developed that demonstrate effective separation and removal of inorganic fluorine from organic fluorine (Koch et al., 2020). TOF is not specific to PFAS, and any fluorine-containing compounds (e.g., pesticides, pharmaceuticals) that are retained during extraction would be included in the organic fluorine measurement.

HF measurement is included in stationary source measurements to evaluate control efficiency of HF as a HAP at emission outlets. Multiple studies of PFAS thermal decomposition and HF monitoring have demonstrated that EPA compliance methods for HF measurement are adequate to meet this need. However, HF is difficult to transport through treatment and control equipment and is therefore not a candidate for mass balance to evaluate PFAS destruction efficiency.

3.a.v Uncertainties/unknowns

EPA is planning to collect additional information and conduct additional research to better understand PFAS destruction and evaluate emission control efficiency (see Section 5). EPA is working to develop a standardized validated methodology for measuring PFAS gaseous emissions (see Section 3.a.iv); however, EPA also lacks detailed information on the amounts and concentrations of PFAS-containing materials that are generated and managed in thermal treatment devices.

Development of sampling and analytical methodologies must continue (see Section 5 research activities) so that emissions and other media from thermal treatment devices burning PFAS-containing materials can be adequately characterized.

When evaluating thermal treatment options, decision-makers (e.g., permit writers, waste managers) should consider potential impacts to communities, including vulnerable populations. Uncertainties such as the overestimation of destruction and disposal capabilities, or a failure to account for potential PIC production and subsequent transformation back to PFAS of concern could have consequences for all nearby communities, including for vulnerable communities that may be disproportionately impacted. EPA recommends decision-makers screen communities located in the vicinity of potential releases from the destruction, disposal, and storage options (considering fate and transport) in order to consider the potential for adverse and disproportionate impacts (see Section 4), and to consider potential measures to prevent, reduce, or address such impacts. Depending on site-specific circumstances (e.g., PFAS concentrations, impacted media, and potential exposure pathways), the size and shape of this area (vicinity) will vary.

In order to provide more detailed recommendations, further data on the ability of thermal treatment units to completely mineralize PFAS-containing materials is necessary (see Appendix A and Section 5.a). Performance testing of all types of combustors is lacking. EPA recommends that full-scale thermal destruction facilities conduct performance testing prior to accepting and processing PFAS materials. Objectives of the testing should include:

- Demonstrate destruction of parent compounds and potential production of semivolatile and volatile PICs (using OTM-45 and OTM-50) to show efficient thermal destruction.
- Characterize PFAS being fed, including the feed rate, whether it is a concentrated or diluted stream, and how the PFAS are fed (e.g., containerized or direct feed).
- Demonstrate utilization of key combustion parameters (temperature, residence time, other conditions that impact mineralization) and include relevant data in the report.
- Characterize release of PFAS and PICs through the system by sampling at multiple points in the process (e.g., before and after a scrubber) and sampling all relevant effluent streams (e.g., scrubber blowdown and fly ash). Document and report operating parameters for all relevant control devices to EPA and relevant state, territorial, or Tribal authorities.

Further details on suggested parameters and procedures are provided in Table 5-1 and Appendix A.

3.a.vi Treatment commercial availability

Section 3.a.i describes the commercial availability of thermal treatment devices. The United States has about 22 commercial hazardous waste combustion facilities[17] in operation; over a dozen large-scale, commercial carbon reactivation companies with about 17 furnaces; 193 MSW incineration units at 75 facilities; and 170 SSIs at 86 facilities.

3.a.vii Summary

PFAS-containing waste can potentially be treated in several types of thermal treatment devices, including HWCs, MWCs, SSIs, and activated carbon reactivation furnaces, but further research is needed to gain a better understanding of what may be possible in practice. These treatment devices operate differently and handle different types of PFAS-containing media with varying concentrations. Even within the same category of thermal treatment device, designs and operating conditions may vary across sources in a way that could affect PFAS treatment efficiency. As a general approach, EPA encourages decision-makers to prioritize the use of destruction and disposal technologies that have a lower potential for PFAS release to the environment, over destruction and disposal technology options with a greater potential for environmental release of PFAS. EPA acknowledges that each destruction and disposal technology has limitations. However, managing PFAS to minimize environmental releases during destruction and disposal activities supports protection of human health and the environment.

There are limited characterizations of potential PFAS emissions from thermal treatment devices that burn PFAS-containing media, and EPA is not aware of any full emission characterizations that have been conducted at HWCs, MWCs, SSIs, or activated carbon reactivation furnaces or any studies to evaluate

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[17] This includes commercial incinerators, cement kilns, and LWAKs that are permitted to burn hazardous waste.

the formation of products of incomplete combustion resulting from inefficiency to convert PFAS into carbon dioxide and hydrogen fluoride. EPA is also not aware of any studies that have been conducted on the extent to which PFAS contaminants partition to air pollution control device residuals or bottom ash, which may also be a concern. This is likely (in part) because PFAS emission measurement methods and other analytical methodologies (e.g., for fly ash, other waste streams) are limited and continue to be developed. Data, however, from pilot-scale experiments conducted by EPA researchers suggest that temperatures above approximately 1,100°C / 2,012°F may result in high destruction efficiencies and few detectable fluorinated products of incomplete combustion.

Given the unique characteristics of fluorine combustion chemistry (particularly the strength of the carbon–fluorine bond), adequate thermal destruction of PFAS requires high temperatures and long residence times and likely benefits from direct flame contact. Hazardous waste combustion technologies (commercial incinerators, cement kilns, and LWAKs) can potentially achieve temperatures (~1,100°C / 2012°F) and residence times sufficient to break all carbon-fluorine bonds in the PFAS contained in the waste stream being thermally treated. Permitted hazardous waste facilities have stringent regulatory controls on temperatures and other important operating parameters to achieve a 99.99 percent destruction efficiency for other (non-PFAS) organic chemicals. Such facilities also have air pollution control devices to prevent certain gaseous and particulate pollutants from entering the atmosphere. However, information on the efficacy of PFAS destruction in these facilities is currently limited. EPA has limited data characterizing PIC emissions from these sources when they burn PFAS. EPA is working to develop measurement methodologies as well as to gather information to conclude whether PICs are adequately controlled. EPA recognizes that PICs are inevitable (even for nonfluorinated compounds); however, based on the unique characteristics of fluorine combustion chemistry, EPA believes it is important to determine whether thermal treatment devices and their associated post-combustion control devices are adequately controlling PICs, especially fluorinated PICs. Given all these factors, there is a current need to continue research activities investigating incineration of PFAS. After sufficient research has been completed to address the related knowledge and data gaps, EPA can make a more informed recommendation on disposal of PFAS compounds and PFAS-containing substances using incineration.

Similar conclusions can be drawn for activated carbon reactivation furnaces as for HWCs. Experimental data suggest that thermal destruction of PFAS will occur in two stages: first during reactivation of the GAC, and then when the off-gas is introduced into a high-temperature zone with temperatures as high as 1,000°C/1,832°F. As referenced in Section 3.a.i, an activated carbon reactivation furnace can be equipped with an afterburner to treat off-gases at high temperatures to achieve 99.99 percent DRE of VOC. In addition, scrubbers can be installed to remove acid gases. This is a promising treatment method, but more information is needed, including confirmation that PICs are controlled based on actual operations, establishment of standard operating conditions for activated carbon reactivation furnaces of various designs to ensure optimal destruction of PFAS, and an understanding of how thermal treatment influences the physical and chemical properties of GAC (in ways that can affect GAC's adsorption behavior and sorption capacity for PFAS).

Research and testing of PFAS destruction performance within MWCs is extremely limited, primarily comprising laboratory and pilot-scale studies (Aleksandrov et al., 2019; Taylor et al., 2014). For example, to assess whether PTFE is destroyed or reformed as PFAS, the Aleksandrov et al. study uses a pilot-scale rotary MWC with afterburner chamber combusting PTFE granules added to wood pellets (also firing natural gas). This study looked at a half-load scenario of 870°C (1,600°F) with a four-second residence time and a full-load scenario of 1,020°C (1,870°F) with a 2.7-second residence time. The study analyzed for 31 PFAS compounds within the flue gas samples collected, assumed to represent a broad range of PFAS. While the laboratory and pilot-scale studies conclude that MSW incineration of PTFE is not a significant source of PFAS, the laboratory thermal reactor and the pilot incinerator used in these studies may not be representative of the design of MWC units currently operating in the United States. For example, the pilot-scale unit in the Aleksandrov et al. study is a rotary combustion chamber followed by an upflow afterburner. No MWC units operating in the United States have a similar configuration. In addition, while several PFAS species were analyzed for in these studies, it is important to note that there are far more PIC species possible, and no studies have thoroughly evaluated the types and quantities of PICs.

As noted earlier in this section, Wang et al. (2013) investigated PFAS interactions with CaO and $Ca(OH)_2$ at moderate temperatures (200°C to 900°C, or 390°F to 1,650°F), both with and without sewage sludge. These experiments were conducted in a laboratory (i.e., combustion in a crucible within a muffle furnace) and found that these calcium species exhibit a pseudo-catalytic effect promoting PFAS destruction and fluorine capture at relatively low temperatures. The study did not investigate the evolution of PICs during the thermal treatment process. While this study shows promising results for the use of catalysts to achieve PFAS destruction and fluorine capture at low temperatures, along with the potential for full-scale application (since lime is occasionally added to sewage sludge to control odor at SSIs), it is important to note that, as with the MWC studies described above, there are caveats for applying these results to the real-world design and operation of SSIs.

More research is needed to address these issues and develop reliable measurement techniques. Section 5 summarizes EPA's continuing PFAS research, as well as a general proposal to collaborate with stakeholders to address these uncertainties promptly. See Appendix D for a summary of costs and considerations.

3.a.viii References for Section 3.a

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3.b Landfills

RCRA regulations define a landfill as “an area of land or an excavation in which wastes are placed for permanent disposal, and that is not a land application unit, surface impoundment, injection well, or waste pile” (40 CFR 257.2). The goal of solid waste landfills is to contain waste, and thereby restrict the release of contaminants within the landfill from entering the environment. Landfills are commonly classified by the types of waste they accept and/or by their ownership status.

Landfills control waste and corresponding pollutants through containment. Because of their many and varied uses, PFAS enter landfills as part of the general MSW stream, in industrial waste, or in other PFAS-containing wastes, with a range of concentrations. Types of landfills include MSW, hazardous waste, industrial, C&D, and ash monofill.

Some landfills are available disposal options for PFAS and PFAS-containing materials. MSW landfills commonly receive large quantities of wastes with significant PFAS concentrations (e.g., biosolids, septage, soils excavated during remediation, and phytoremediation biomass). These concentrations are much lower than some industrial or commercial wastes with high PFAS content (e.g., leather tannery, chrome plating, and textile industry waste). Permitted hazardous waste landfills employ the most extensive set of environmental controls (e.g., double liner systems with leachate collection/treatment and leak detection) and practices (e.g., comprehensive record keeping) that are currently available for the containment of commercial and industrial wastes with high PFAS content (see Table 3-2). Hazardous waste landfills are more effective at minimizing PFAS release into the environment than other landfill types. Note that hazardous wastes that are typically landfilled are not usually biodegradable; thus, most hazardous waste landfills do not have LFG collection and control systems installed and, therefore, may not be appropriate for disposal of biodegradable PFAS-containing wastes (e.g., biosolids) that may produce LFG. It is also noted that leachate from hazardous waste landfills is designated as hazardous waste (F039) and must be managed in accordance with specific requirements for hazardous waste. Nonetheless, some leachate treatment or management options cannot treat or destroy PFAS.

Most modern MSW landfills, when constructed and operated with appropriate controls (e.g., a flexible membrane liner system and leachate and LFG collection and management systems), can also help contain PFAS. However, research indicates that although MSW landfills contain PFAS for the most part, up to 5 percent of the PFAS may be released in the LFG and, 11 percent may be released in the leachate annually (Tolaymat et al., 2023). Also, leachate may be sent to wastewater treatment plants that are not capable of treating or destroying PFAS; thus, the PFAS in landfill leachate may be released to the environment. While some information is available, EPA plans to research to understand the effects of PFAS on liner integrity, gaseous emissions from landfills, the effectiveness of leachate treatment for PFAS removal, and the levels and types of PFAS in landfill leachate (see Section 5). MSW landfills present a good opportunity to dispose of lower-concentration PFAS waste such as biosolids, phytoremediation biomass, and contaminated soils. However, there is a need for additional research regarding the effectiveness of emerging PFAS separation and destruction treatment technologies on a landfill leachate matrix.

While landfills might serve as long-term containment sites, it is unclear how effective they are at containing PFAS. Because landfills are a containment method and do not destroy PFAS, PFAS are anticipated to persist in landfills for the life of the compounds, which could be many centuries.

Section 3.b focuses on the viability of landfilling as a means of containing PFAS and PFAS-containing material, discussing:

- The types of PFAS waste and how they behave within a landfill.
- The types of landfills that receive PFAS-containing waste.
- The types of environmental controls and their ability to effectively contain PFAS.
- The potential for environmental releases of PFAS associated with landfilling.
- Recommendations for deciding what type of landfill is most appropriate for PFAS-containing waste.

- Methods to monitor PFAS at landfills and corresponding waste streams.
- Uncertainties and unknowns associated with landfilling PFAS-containing waste.

3.b.i Types of PFAS and PFAS-Containing Materials

As described in the Executive Summary and Section 2, "PFAS" is a broad term for a wide range of chemicals. When disposed of in a landfill, these different types of PFAS may behave differently from one another and may require different considerations for the types of landfills that are suitable for disposal and the types of emissions controls that are necessary to contain the PFAS over time.

PFAS Polymers. Stable polymeric PFAS (e.g., PTFE or Teflon™) are less likely to migrate within a landfill by partitioning to the gas or liquid (i.e., leachate) phase and should not be confused with side-chain fluorinated polymers; however, physical abrasion of the polymers can lead to the creation of fluorinated microplastics that may be suspended in landfill leachate. Although fluoropolymers are highly resistant to chemical and thermal degradation, they can still break down into microplastics due to weathering and physical strain, increasing their dispersion and bioavailability (Lohmann et al., 2020). Fluorinated microplastics have been measured in landfill leachate (He et al., 2019), but because UV radiation and resulting oxidation in landfills is negligible and the physical processes that lead to abrasion and the development of microplastics are expected to be relatively low in landfills, the bulk of stable fluoropolymers would be expected to be retained for a long duration. Additionally, soluble, and volatile PFAS have been identified in fluoropolymers as synthesis byproducts or incomplete reactants.

Soluble PFAS. Soluble PFAS and precursors that degrade to soluble PFAS, including perfluoroalkyl acids such as PFOA, PFOS, PFHxS, PFBA, and HFPO-DA, can more easily and quickly partition to the liquid phase and become part of the landfill leachate. Landfills without an appropriate liner and leachate collection system (see Section 3.b.iii) will allow this PFAS-containing leachate to enter and contaminate the groundwater. It is therefore not appropriate to dispose of soluble PFAS in landfills without these controls in place.

Volatile PFAS. Volatile PFAS and forms that degrade to volatile PFAS, including alcohols such as FTOHs and FASEs may be released through uncollected or fugitive LFG. These types of PFAS should therefore be disposed of in landfills with LFG collection and control systems in place and operating (see Section 3.b.iii.6). Furthermore, PFAS waste that is comingled or combined with other waste determined to be biodegradable (see EPA Test Methods for biodegradability listed at *https://www.epa.gov/pesticide-labels/criteria-biodegradability-claims-products-registered-under-fifra*) should only be disposed of at landfills with gas collection and control systems in place. The biodegradable fraction of waste will produce LFG as it decomposes and allow volatile PFAS to be emitted from the landfill.

Mechanisms and rates of transformation. The behavior of PFAS in landfills is a complex and multifaceted process that involves various abiotic and biotic transformation and partitioning pathways. While the exact mechanisms and rates of transformation are not fully understood, both aerobic and anaerobic conditions in landfills can facilitate the conversion of PFAS precursors into more persistent terminal PFAS species. It is notable that fluorotelomer carboxylic acids (FTCAs) are a soluble intermediate transformation product which has been well documented to accumulate in biologically active anaerobic landfill conditions and is considered a landfill leachate contamination indicator. FTCAs result from the degradation of volatile PFAS such as FTOHs. For this reason, volatile PFAS-containing wastes should be managed in landfill environments which are also appropriate for soluble PFAS disposal.

As the environmental and health risks of PFAS are the subject of emerging research, it is essential to consider PFAS management strategies that minimize their release and transformation in landfills and other waste management systems. For example, Subtitle D MSW landfills are required to install a low-permeability liner prior to waste disposal, and LFG at MSW landfills is collected and managed based on the landfill's permitted design capacity and emission rate of nonmethane organic compounds (NMOCs) (40 CFR part 60, subparts Cf and XXX). Given these potential pathways for release, Table 3-1 summarizes the recommended engineering controls at landfills accepting PFAS waste based on the type of PFAS and the biodegradability of any comingled waste. It is noted that non-biodegradable stable polymeric PFAS do not require specific controls since they are not anticipated to migrate or biodegrade and generate gas, can be disposed in any type of landfill, and are not included in the table below. Also note that soluble PFAS comingled with biodegradable waste should have both control liquid and gaseous emissions, i.e., leachate collection and gas control.

Table 3-1. Recommendations for Disposal of PFAS-Containing Waste by Type of PFAS, Waste Biodegradability, and Type of Landfill Controls to Reduce Risks of Environmental Release

Type of PFAS	Degradability of comingled waste	Landfills with liner and leachate collection	Landfills with gas collection and control
Stable polymeric PFAS (e.g., PTFE)	Biodegradable		✓
Soluble PFAS (e.g., PFAAs)	Biodegradable	✓	✓
	Non-biodegradable	✓	
Volatile PFAS (e.g., FTOHs)	Biodegradable		✓
	Non-biodegradable		✓

3.b.ii Types of landfills

Landfills are typically classified by ownership status and by the type of solid waste they are permitted to receive, which determines the types of environmental controls they must employ. Depending on the type of waste disposed, a landfill could be subject to regulation and permitting under RCRA, the CAA, and/or TSCA. RCRA regulates two types of landfills: Subtitle C facilities receive hazardous waste, while Subtitle D landfills are primarily intended for the management of non-hazardous waste and can include MSW landfills, industrial non-hazardous waste landfills, C&D waste landfills, and coal combustion residual landfills. The CAA regulates air emissions from MSW landfills and TSCA regulates landfills that contain PCBs. The requirements under RCRA, CAA, and TSCA determine how the landfill must be constructed, operated, controlled, maintained, monitored, and closed when it reaches its final capacity.

Although categories and environmental controls vary from state to state, the following categories of landfills exist in most states. Landfills tend to have similar environmental controls within each category:

- Hazardous waste
- MSW
- Ash monofill
- Industrial
- C&D debris

Table 3-2 identifies the environmental controls required by RCRA and CAA for the landfill types defined in RCRA Subtitles C and D. RCRA regulations focus on the containment of solid waste and leachate, while CAA regulations focus on collection and control (destruction) of components in LFG. The landfill types differ in how they are constructed, operated, controlled, monitored, and closed, reflecting the different types of waste they are allowed to receive.

- Subtitle C hazardous waste landfills that meet the minimum technology requirements set under RCRA.
- Subtitle C hazardous waste landfills are permitted to receive hazardous wastes that pose potential risk to humans and the environment and therefore have the most stringent environmental controls.
- Subtitle D MSW landfills are allowed to receive MSW and are required to install a low-permeability liner prior to waste disposal. LFG must be collected and managed according to the requirements of the CAA standards of performance (40 CFR part 60 subparts Cc, Cf, WWW, and XXX) at landfills that meet the regulatory requirement thresholds. Subtitle D landfills that receive non-hazardous and non-biodegradable waste tend to have environmental controls commensurate with the waste they receive. These controls can vary from state to state; for example, bottom liner, leachate collection, and final cover requirements for industrial and C&D landfills range from none to rigorous.

Note that certain small MSW landfills in arid or remote locations are exempt from RCRA design and groundwater monitoring requirements (40 CFR 258.1). See Sections 3.b.ii.2 and 3.b.iii.1 for additional exemptions.

Table 3-2. Required Environmental Controls by Landfill Type

Landfill Type	Federally Regulated Under	Bottom Liner and Leachate Collection System	Gas Collection and Control System	Final Cover
Hazardous waste	RCRA Subtitle C	Yes (double liner or better)	No	Flexible membrane liner (FML) cap
MSW	RCRA Subtitle D 40 CFR part 258 CAA 40 CFR parts 60 and 63	Yes (composite liner or better)	Yes (when regulatory size and emission thresholds are triggered)	FML cap
Ash monofills	RCRA Subtitle D 40 CFR part 257	Yes (composite liner or better)	No	Clay cap
Industrial	RCRA Subtitle D 40 CFR part 257	Varies by state, from no liner requirement to composite liner	No	Varies by state, from no requirements to FML cap
C&D debris	RCRA Subtitle D 40 CFR part 257	Varies by state, from no liner requirement to composite liner	No	Varies by state, from no requirements to FML cap

PFAS concentrations have been detected in landfill leachates from various types of landfills, as shown in Table 3-3.

Table 3-3. Range of PFAS Concentrations in Different Types of Landfill Leachate Reported in Published Studies. Adapted from Tolaymat et al., 2023.

Matrix	Number of PFAS Detected (Number of PFAS in Method)	∑PFAS Range (ng L^{-1})	Country	Reference
MSW Leachate	25 (26)	300 – 58,000	USA	Chen et al. (2023)
	10 (11)	15,000 – 18,000	USA	Solo-Gabriele et al. (2020)
	24 (24)	2,700 – 7,400	USA	Huset et al. (2011)
	30 (70)	2,000 – 29,000	USA	Lang et al. (2017)
	2 (2)	330 – 2,600	USA	Clarke et al. (2015)
	28 (28)	230 – 29,000	USA	Helmer et al. (2022)
	2 (2)	47 – 3,400	USA	Michigan Waste & Recycling Association (2019)
	22 (25)	1,400 – 125,000	USA	NWRA (2020)
	31 (40)	BDL – 104,000	USA	California Water Boards (2023)
	14 (14)	33 – 15,000	Australia	Gallen et al. (2016)
	9 (9)	210 – 46,000	Australia	Gallen et al. (2017)
	15 (15)	Not reported	Australia	Simmons (2019)
	25 (43)	31 – 13,000	Germany	Busch et al. (2010)
	24 (24)	2,500 – 36,000	Canada	Benskin et al. (2012)
	16 (18)	700 – 6,400	Canada	Li (2009)
	2 (2)	50 – 2,300	Canada	Gewurtz et al.
	16 (27)	2,200 – 6,100	Norway	Eggen et al. (2010)
	7 (8)	200 – 1,500	Norway	Kallenborn et al. (2004)
	4 (4)	210 – 610	Finland	Perkola & Sainio (2013)
	7 (10)	14 – 17,500	Ireland	Harrad et al. (2019)
	8 (16)	640 – 1,400	Spain	Fuertes et al. (2017)
	28 (30)	320 – 11,000	Norway	Knutsen et al. (2019)
	17 (26)	0.3 – 1,300	Sweden	Gobelius et al. (2018)
	11 (14)	7,300 – 290,000	China	Yan et al. (2015)
	33 (57)	3,040 – 109,000	China	Liu et al. (2022)
	17 (17)	1,800 – 43,300	China	Huang et al. (2022)
	18 (18)	1,270 – 7,660	Singapore	Yin et al.
C&D Leachate	8 (9)	4,200 – 11,000	Australia	Gallen et al. (2017)
	24 (26)	270 – 30,500	USA	Chen et al. (2023)
	11 (11)	14,000 – 16,000	USA	Solo-Gabriele et al. (2020)
MSW Incineration Ash Leachate	9 (11)	2,800 – 3,400	USA	Solo-Gabriele et al. (2020)
	26 (26)	39 – 54,500	USA	Chen et al. (2023)
MSW Gas Condensate	26 (26)	199 – 80,900	USA	Chen et al. (2023)
	53 (92)	3,000 – 50,000	USA	Smallwood et al. (2023)
Hazardous Waste Leachate (Primary)	17 (28)	570 – 377,000	USA	California Water Boards (2023)
Hazardous Waste Leachate (Secondary)	13 (24)	25 – 3,700	USA	California Water Boards (2023)

BDL = below detection limit; ng/L = nanograms per liter

The following subsections describe the types of landfills that are expected to contain PFAS or PFAS-containing materials, with a focus on design and operation aspects that EPA considers important from a PFAS-treatment perspective.

3.b.ii.1 Hazardous waste landfills

Hazardous waste landfills are permitted to receive waste that is defined as "hazardous" under RCRA. This waste either is explicitly listed as hazardous or demonstrates at least one of certain characteristics (i.e., toxicity, corrosivity, reactivity, or ignitability) as defined by RCRA. Hazardous waste landfills (Subtitle C landfills) can be used for non-hazardous wastes like PFAS, when the generator wants a more controlled solution. These landfills are required to have a double liner system and must adhere to waste pretreatment standards prior to disposal (40 CFR part 268). Most hazardous waste landfills are not biologically active because they do not typically accept biodegradable wastes (e.g., biosolids, food scraps), and therefore typically do not need to have gas collection systems. Conditions in these landfills are not expected to cause PFAS emissions via LFG, although gas collection systems could be installed if a problem arises related to gas release or gas emissions. Any hazardous waste landfill must have a final cover consisting of an FML covered by soil.

PFAS-containing waste is and will likely continue to consume a fraction of hazardous waste landfill capacity. Although waste containing PFAS is currently not defined as hazardous waste under RCRA (unless the particular wastes are hazardous for some other reason), some waste generators, perhaps to be cautious, have been sending PFAS-containing wastes to hazardous waste facilities. To ensure hazardous waste landfill capacity is available in the future, EPA reviews and analyzes the Biennial Hazardous Waste Report and other data to develop and then publish an assessment of national capacity for hazardous waste management.[18] The last such capacity assessment indicated that there is adequate capacity nationwide through 2044, and a new assessment is currently underway to incorporate new information and extend the time horizon. EPA will continue to work with states to monitor hazardous waste treatment and disposal capacity and report on the status.

3.b.ii.2 Municipal solid waste landfills

EPA estimates that in 2018, 50 percent of the MSW generated was landfilled (U.S. EPA, 2024). An MSW landfill typically has at least a bottom liner and an extensive gas collection and control system (GCCS) to collect the LFG generated when the biodegradable waste, such as food scraps and office paper, degrades over time. Some MSW landfills have accepted PFAS manufacturing wastes, and as a result, PFAS concentrations in leachate are higher than the typical range expected in MSW landfill leachate (NWRA, 2020). Under RCRA, MSW landfills that received waste after 1993 must install a composite liner that meets design criteria (40 CFR 258.40). However, MSW landfills that receive less than 20 tons of waste per day are exempt from installing a liner and monitoring groundwater under RCRA (40 CFR 258.1(f)(1)). MSW landfills that exceed specified thresholds for size (based on design capacity) and emissions of NMOCs are required to install gas collection systems under current CAA regulations. Therefore, a subset of MSW landfills do not have gas collection and could release fugitive PFAS associated with LFG emissions. Similarly, unlined MSW landfills are ineffective at managing the release of mobile PFAS to groundwater (see Section 4.a). The fate of PFAS in LFG that is managed through on-site GCCS including on-site flares, engines, boilers, and turbines is unknown.

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[18] Background information and links to related documents are available at *https://www.epa.gov/hwpermitting/assessment-national-capacity-hazardous-waste-management*.

3.b.ii.3 Ash monofills

Ash monofills are a subtype of MSW landfills that mainly receive ash from MSW incinerators but could also receive smaller volumes of other waste streams such as biosolids from WWTPs. Ash monofills typically have a bottom liner and final cover requirements similar to other MSW landfills. However, they usually do not require GCCSs due to the use of incineration to remove biodegradable waste. Solo-Gabriele et al., (2020) found that ash monofills have lower PFAS concentrations than other landfill types, as shown in Table 3-3. While some landfill wastes, such as MSW ash, may be low in PFAS concentrations, the inclusion of higher-PFAS waste types may increase PFAS release. With limited data on the presence of PFAS in ash monofill leachate, more research is needed to determine the PFAS and precursor content of ash from different incineration technologies and air pollution control systems.

3.b.ii.4 Industrial landfills

Industrial landfills receive solid waste from industrial operations (non-municipal). Industrial landfills are often designed to manage specific waste streams (e.g., furnace slag, fly ash, plastics). The designs of industrial landfills vary widely based on the characteristics of the waste they receive. Requirements for environmental controls at these landfills also vary from state to state. Depending on the waste types and size of the landfill, some states do not require a liner. If a liner is required, a membrane cap is often also required. Due to the variability in control technologies, industrial landfills may not be an effective disposal option for managing uncontrolled releases of PFAS. Landfills that have historically received PFAS manufacturing wastes have been associated with high concentrations in leachate and reported environmental impacts (MPART, 2020). Some waste types received at industrial landfills, including plastics and materials with polishes or coatings, are associated with high concentrations of PFAS (OECD, 2013). Specifically, industrial processes such as the leather tanning, chrome plating, and textile industries contribute to PFAS in the industrial solid waste stream (ITRC, 2022).

3.b.ii.5 Construction and demolition landfills

C&D landfills receive waste from construction, renovation, and demolition projects, as well as other inert materials that are generated in high volumes, comparable to MSW waste (U.S. EPA, 2020b). The requirements for environmental controls at these landfills vary widely from state to state, ranging from no liner to a required composite liner. C&D landfills receive some PFAS-containing wastes (e.g., building materials and carpeting with PFAS treatments) (OECD, 2013; Solo-Gabriele et al., 2020; Bečanová et al., 2016), and soluble PFAS have been noted in the leachate from C&D landfills that are equipped with a leachate collection system (Chen et al., 2023). If a liner is required, a membrane cap could also be required. GCCSs are not required in C&D landfills due to low levels of biodegradable waste received compared to MSW landfills. Therefore, PFAS-containing wastes that also include a biodegradable component should not be disposed of at C&D landfills. A GCCS may sometimes be necessary to remediate a specific issue, such as gases generated from the decay of drywall. C&D landfills that lack composite liners and leachate management systems are unlikely to prevent the migration of soluble or volatile PFAS types; however, longer chain PFAS polymers such as PTFE, which is found in some electrical insulation and membrane material, are less mobile and likely to be contained in the waste mass for a longer period.

3.b.iii Ability of engineered landfill components to contain PFAS

PFAS are emitted from landfills via two possible routes: landfill leachate and LFG. Landfill leachate is liquid that has come into contact with solid waste and is either derived from the waste itself or the result of rainwater intrusion into the waste mass. Leachate contains soluble, suspended, or miscible materials removed from such waste. LFG is the result of the natural decomposition of organic material in landfills under anaerobic conditions. LFG is composed of roughly 50 percent methane, 50 percent carbon dioxide, and a small amount of NMOCs.

Existing efforts to manage contaminants in landfills focus on controlling leachate and LFG emissions. As shown in Figure 3-1, landfills constructed with environmental controls (e.g., bottom liner, leachate collection system, gas collection system, final cover system) manage the release of contaminants into the environment.

The uses of the engineered landfill controls shown in Figure 3-1 vary by landfill type due to differences in types of waste accepted, operating practices, site conditions, and federal and state regulations.

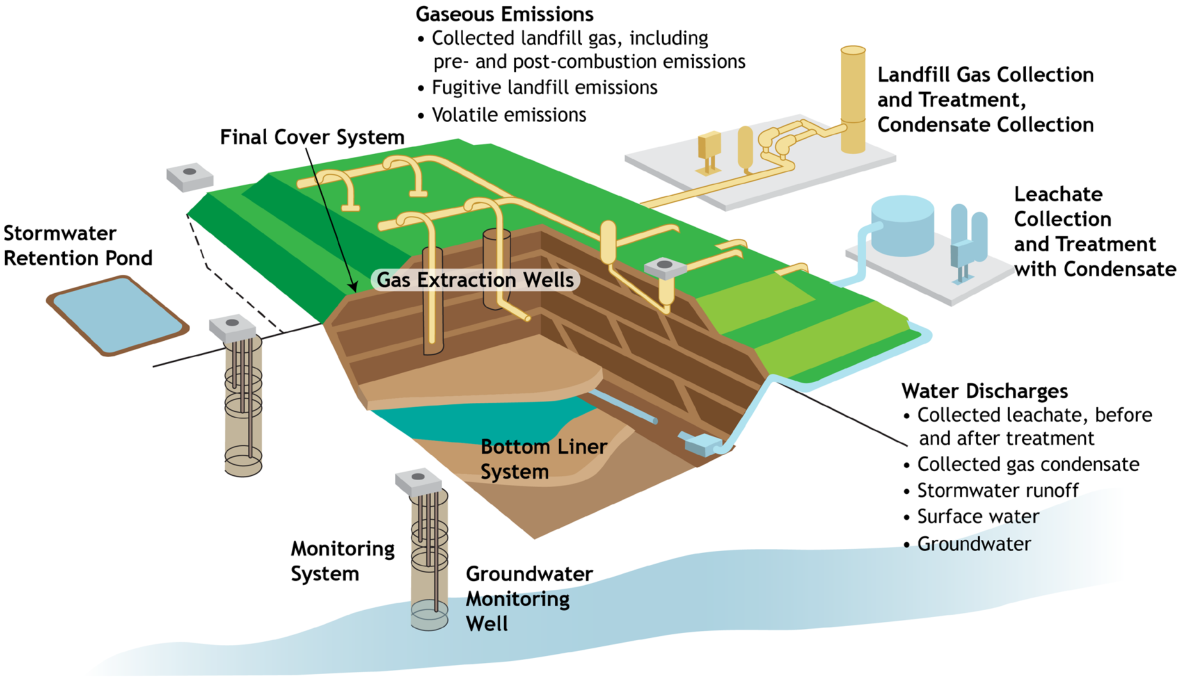

Figure 3-1. Engineered landfill components and potential PFAS release pathways.

3.b.iii.1 Bottom liner and leachate collection system

Under RCRA, any new landfill, replacement of an existing landfill, or lateral expansion of an existing landfill must have a double liner and leachate collection system if it receives hazardous waste, to limit leachate and gas release through the bottom of the landfill. Most landfills that are regulated under RCRA to receive non-hazardous waste are subject to design criteria that require either a composite liner and leachate collections system or a state-approved design that satisfies performance standards to ensure that regulated chemicals do not migrate beyond a specified distance from the landfill.

The variation of landfill requirements among states poses a challenge for the long-term containment of existing PFAS in landfills: state requirements for landfill liners vary, and some landfills are not required to have liners because of waste type or quantities received. Additionally, RCRA-exempt sites do not require liners (40 CFR 258.1). Because PFAS are not a RCRA-regulated hazardous waste, existing unlined landfills could include PFAS-containing wastes that are easily emitted into the environment. Depending on their mobility, PFAS compounds could impact groundwater if disposed of in an unlined landfill.

A liner is built of layers of clay and/or polymers (i.e., FMLs) designed to withstand the weight of waste and soil. Leachate will collect on top of the liner, so its design must include a leachate collection system contoured to collect leachate through a network of pipes leading to a low point called a sump. The collected leachate is pumped from the landfill and managed as liquid wastes (see Section 2.f and Section 3.b.iii.5). Uncontrolled leachate could result in the release of PFAS into the environment.

PFAS interactions with landfill liner materials have been the subject of limited studies. The most common types of FML are made with polyethylene geomembranes. PFAS diffusion through linear low-density polyethylene is reported below detection diffusion rates (Di Battista et al., 2020), and diffusion through high-density polyethylene may be even slower due to differences in material structure. PFAS likely pass through clay liners, which are required to have low hydraulic conductivity on the order of 10^{-6} centimeters per second, at the same rate as leachate and other constituents such as chloride (Li et al., 2015).

The most significant pathway for leachate (and PFAS) transmission through geomembrane or composite liners is via imperfections (e.g., flaws or holes) (Di Battista et al., 2020). A review of landfill liner performance reported median leakage rates of 44 and 33 liters per hectare per day for geomembrane and composite liners, respectively, and overall liner collection efficiency of 98 percent (Jain et al., 2023). Even if liners successfully prevent leachate from reaching groundwater, very few data exist on whether concentrated PFAS waste interacts with the different types of geotextiles used for landfill liners, thus affecting the performance of the liner. While the performance of clay liners may not be affected drastically, there is currently no research on the long-term stability of FML in the presence of PFAS.

3.b.iii.2 Landfill gas collection system

Landfills use GCCSs to manage gas generated from decomposing organic waste. A GCCS consists of a network of perforated pipes sunken into the waste mass. These “gas wells” are connected to a central blower that pulls gas from the wells. Despite the presence of a GCCS, gas can still migrate both through the surface of the landfill and underground through the bottom of the landfill. The gas produced by MSW landfills contains about 50 percent methane that is usually collected and burned off at the site via flares or for energy recovery; however, those systems for LFG destruction (e.g., flares, engines, boilers) typically operate up to 85°C—below the minimum temperature required to destroy PFAS.

Not all landfills are required to collect LFG (see Table 3-2). As noted in Section 3.b.iii.6.1, research has found that soluble PFAS with relatively high vapor pressures can be emitted into the atmosphere via the gas generated at landfills (Ahrens et al., 2011; Hamid et al., 2018; Wang et al., 2020; Weinberg et al., 2011). Direct sampling methodologies are currently under development, and published concentrations of PFAS measured *in situ* are limited to a single study (Titaley et al., 2023). LFG flares generally operate at approximately 650°C to 850°C, lower than the 1,100°C necessary to achieve the mineralization of PFAS. Nonetheless, LFG collection and management systems (e.g., flares) must be present at MSW landfill sites that accept PFAS waste, especially biodegradable PFAS waste such as biosolids. The use of GAC prior to the flaring of LFG can be effective in removing PFAS from LFG. Data on the fate of PFAS in LFG that is managed through on-site GCCS, including on-site flares, are still limited. See Section 5 for potential research needs.

3.b.iii.3 Final cover system

After a regulated landfill has reached its designed and permitted capacity, it must be capped with a cover system. This system typically consists of some combination of soil and membrane materials and is primarily intended to reduce infiltration of rainwater into the landfill to minimize leachate generation. The cover system also helps increase the efficiency of the GCCS and reduce uncontrolled gas emissions. Synthetic membranes and caps are more effective at controlling release of PFAS than earthen covers. Earthen covers are more subject to wet/dry cycles and cracking and are more likely to result in uncontrolled LFG emissions, which could contain PFAS (Ahrens et al., 2011; Tian et al., 2018; Wang et al., 2020; Weinberg et al., 2011).

3.b.iii.4 Other environmental controls and monitoring systems

In addition to the major infrastructure discussed above, solid waste landfills implement other practices and systems to protect human health and the environment. In active landfill cells, daily application of a material such as soil or inert waste covers exposed solid waste. Daily cover reduces leachate generation, gas emissions, and direct exposure to humans and wildlife. Access control for a landfill site, such as a fence, is typically also required to reduce direct human and ecological exposure to waste. Extensive monitoring networks are generally required to measure the landfill’s impact on surface water and groundwater. RCRA Subtitle C requires all hazardous waste landfills to install groundwater monitoring wells. See Section 3.b.iii.5 for leachate discharge controls.

3.b.iii.5 Leachate discharge controls

3.b.iii.5.1 Leachate characteristics

Leachate is the liquid effluent from landfills primarily generated through the percolation or infiltration of rainwater through waste. Leachate often contains high concentrations of biodegradable and non-biodegradable organic matter, dissolved and suspended solids, heavy metals, ammonia, and sulfur compounds (Mukherjee et al., 2015; Renou et al., 2008). The waste type, age, climate at the landfill site, and methods of landfill operation dictate the characteristics of leachate. These factors result in highly variable leachate characteristics across landfill types and on a site-by-site basis. Subtitle C hazardous waste landfills are required to use a leachate collection system during their active and post-closure care periods to mitigate adverse impact to human health and the environment. Since the use of leachate collection systems at Subtitle D landfills varies by landfill type and state requirements, the efficacy of leachate management and PFAS emissions depends on the controls implemented. PFAS containment in landfills is expected to be indefinite, so the generation of PFAS-containing leachates remains a probability during the post-closure care period and beyond. Ideally, the leachate collected at the bottom of the landfill is removed and managed to minimize impacts to human health and the environment. See Section 2 for more a more detailed discussion on the types of leachates and associated PFAS.

3.b.iii.5.2 Off- and on-site management of leachate

The most common method for non-hazardous waste leachate disposal is off-site treatment at municipal WWTPs, where leachate is mixed with wastewater and treated. The dynamic nature of leachate characteristics, the presence of non-biodegradable compounds and ammonia, and the presence of emerging contaminants (such as PFAS) in the leachate may make it difficult for WWTPs to effectively treat the influent water. In some cases, a landfill with elevated PFAS in its leachate may burden a WWTP’s ability to treat, remove, or destroy these compounds before discharge to the environment (Masoner et al., 2020). As mentioned in Section 2.a.ii, wastewater treatment technologies used at most municipal WWTPs are generally ineffective at destroying or controlling PFAS (Schultz et al., 2006) and as a result may also be ineffective at treating PFAS-containing landfill leachate. Furthermore, in WWTPs, PFAS may bind to and accumulate in biosolids, which can lead to PFAS entering the environment if biosolids are applied to land. Some WWTPs have requested that landfill operators pretreat leachate on site, and some WWTPs are implementing surcharges based on the leachate quality. Underground injection is another form of off-site leachate management (see Section 3.c).

Table 3-4 presents leachate management or treatment methods for the on-site management of leachate. Multiple technologies in the table are marked for further research—technologies for which, at the time of publishing, data do not exist to support or reject application for PFAS treatment. "Secondary treatment required" refers to the remaining concentrated or captured PFAS, which must be disposed of after treatment. "Potential secondary release" refers to the potential for PFAS release or breakthrough during the treatment process. Leachate characteristics are site-specific; therefore, the effectiveness of leachate technology should be evaluated on a site-by-site basis. (Table 3-3 lists average PFAS concentrations observed in different types of landfill leachate.)

Landfill operators should identify management or treatment methods that: (1) are suitable for the leachate at their specific sites and (2) meet the leachate discharge standards for chemicals and characteristics of leachate that are regulated. For example, leachate from a hazardous waste landfill is regulated as a listed hazardous waste (F039) and must be managed appropriately but that does not necessarily mean that treatment technologies are adequate to control the release of PFAS into the environment. For example, evaporation is often used to reduce leachate volume, which could lead to uncontrolled PFAS volatilization. The geography of the landfill site and cost-effectiveness of the methods play a crucial role in the identification of viable approaches to leachate management or treatment. For example, evaporation is often used to reduce dependence on treating a specific chemical(s) (e.g., ammonia). The use and effectiveness of leachate management strategies in removing or destroying PFAS during treatment varies (and, as noted in Section 3.b.vi, methods to quantify effectiveness are still under development). Leachate treatment technologies can be largely categorized into physiochemical processes, physical processes, biological processes, natural processes, and other management methods, as grouped in Table 3-4. Considering that leachate contains a variety of chemicals, a combination of physiochemical treatment processes can be used to narrowly target specific parameters for pre-treatment, or as part of a multi-step treatment strategy.

Table 3-4. Existing Landfill Leachate Treatment Technologies for PFAS Removal or Destruction

Treatment Technology	Treatment Mechanism	Pros for PFAS Treatment	Cons for PFAS Treatment	References
Physiochemical Processes				
GAC	Adsorption	• Familiar technology • Effective for long-chain PFAS	• Secondary treatment required • Short-chain PFAS breakthrough • Potential secondary release • Cost	• McCleaf et al. (2017), Pan et al. (2016), Ross et al. (2018)
PAC with coagulation	Adsorption	• Effective for long-chain PFAS	• Secondary treatment required • Costly for high-volume leachate • Potential secondary release	• Bao (2014), Pan et al. (2016)
Polymeric adsorption	Adsorption	• Tailored for specific compounds	• Secondary treatment required • Potential secondary release	• Liu (2017)
Ion exchange resin	Ion exchange adsorption	• Specified for certain compounds • More effective than GAC for long-chain compounds	• Secondary treatment required • Less effective for short-chain PFAS • Potential secondary release	Dickenson & Higgins (2016), McCleaf et al. (2017), Ross et al. (2018)
Zeolite	Ion exchange adsorption	• Inexpensive	• Secondary treatment required • Low surface area compared to GAC • Unknown reaction with short-chain PFAS	Chiang et al. (2017), Ochoa-Herrera & Sierra-Alvarez (2008)
Ozonation/ ozofraction-ation	Oxidation	• Potentially effective multi-contaminant removal	• Potential side-product formation • Secondary treatment required	Franke et al. (2019), Lin et al. (2012), Rahman et al. (2014), Ross et al. (2018)
Fenton oxidation	Oxidation	*Limited data available*	*Limited data available*	*None identified*

Treatment Technology	Treatment Mechanism	Pros for PFAS Treatment	Cons for PFAS Treatment	References
Photocatalytic advance oxidation process	Oxidation	• Permanent degradation	• Potential side-product formation • Cost	Lockwood (2018), Ross et al. (2018)
Coagulation-flocculation	Precipitation	*Limited data available*	*Limited data available*	Bao (2014), Dickenson & Higgins (2016), ITRC (2018), Rahman et al. (2014)
Chemical precipitation	Precipitation	*Limited data available*	*Limited data available*	*None identified*
Air stripping	Volatilization	*More research needed*	• Potential secondary emissions	*None identified*
Physical Processes				
RO	Physical separation	• Commonly used • Effective for short and long-chain PFAS	• Secondary treatment required for high-volume concentrate • Membrane fouling	Dickenson & Higgins (2016), Ross et al. (2018)
NF	Physical separation	• Uses less energy than RO • Effective for short- and long-chain PFAS	• Secondary treatment required	Boo et al. (2018), Dickenson & Higgins (2016)
Ultrafiltration (UF); microfiltration (MF)	Physical separation	*N/A*	• Not effective for PFAS	U.S. EPA (2020a)
Climatic evaporation; thermal evaporation; mist evaporation	Volume reduction	*Limited data available*	• Potential secondary emissions	*None identified*
Other On-Site Management Methods				
Recirculation	Containment	• Co-location with landfill	• Oversaturation • Potential surface water contamination • Dependent on climate	*None identified*
Underground injection	Containment	• Potential solution for PFAS concentrate	• Dependent on site geology • Regulatory approval	ITRC (2018)

Treatment Technology	Treatment Mechanism	Pros for PFAS Treatment	Cons for PFAS Treatment	References
Incineration	Thermal destruction	- PFAS destruction	- Potential secondary emissions (incomplete destruction) - Regulatory approval	ITRC (2017), Yamada et al. (2005)
Solidification	Containment	- Co-location with landfill - Reduces PFAS mobility	- Consumes air space in landfill - Unrealistic for large leachate volume	*None identified*
Biological Processes				
Activated sludge process sequencing; batch reactor; anaerobic; digestor; membrane bioreactor	Biological processes	*Limited data available*	- Limited by high concentrations of non-biodegradable organic matter	Ross et al. (2018), Saez et al. (2008), U.S. EPA (2020a)
Natural Processes				
Constructed wetlands; aerated ponds; phytoremediation; land application	Environmental release	*N/A*	- Direct release of PFAS	U.S. EPA (2020a)

3.b.iii.5.3 Leachate management and treatment technologies

Membrane treatments separate compounds from the leachate using mechanical filtration and pressure. Leachate passes through selective membranes (such as RO, NF, UF, and MF membranes) that divide it into two parts: permeate (which has passed through the membrane) and concentrate (which has not). The permeate and concentrate can then be treated as independent streams. The primary difference between these membranes is the pore size, which in turn affects the operating pressure and removal efficiency for different types of contaminants. RO is the most commonly used type of membrane for leachate treatment, while NF, UF, and MF are generally used in combination with other treatment technologies including RO. RO and NF are known to be effective in concentrating some PFAS, but UF and MF have pores that are too large to limit the release of most water-bound PFAS across the filtration membrane. Membrane fouling and a large amount of concentrate generation are two of the major drawbacks observed in implementing the membrane treatment system for landfill leachate and may be further complicated by high concentrations of PFAS (Dickenson & Higgins, 2016; ITRC, 2018; Ross et al., 2018).

An adsorption process with activated carbon is used for targeted removal of organic matter at some landfill sites. Activated carbon is known to be effective at trapping some PFAS, but it may need to be combined with other treatment methods to manage the range of PFAS found in landfill leachate. Activated carbon is a non-selective treatment method, expected to be less effective in nutrient-laden effluents such as landfill leachate. Once saturated, activated carbon needs to be reactivated through a pyrolysis process (see Section 3.a), collected as solid waste, or otherwise treated, with consideration of the PFAS concentration (ITRC, 2017). Leachate also may need to be pretreated before activated carbon treatment to avoid rapidly saturating the carbon.

Methods using ozone, Fenton's reagent, or advanced photocatalytic technologies are used to oxidize organic matter in the leachate. Ozone and hydrogen peroxide (H_2O_2) in combination with other oxidizing agents have been observed to remove up to 99 percent of chemical oxygen demand from leachate at different operating conditions (Renou et al., 2008) and may be effective at reducing or modifying certain PFAS in leachate (Ross et al., 2018). Ultraviolet photocatalytic advanced oxidation is known to be capable of destroying PFAS, but additional studies are needed to understand the subsequent products associated with this method (Lockwood, 2018; Ross et al., 2018).

Ion exchange processes using zeolite and magnetic ion exchange resin remove ammonia and organic matter, respectively. Ion exchange can be flexibly designed to address different compounds and may be effective at reducing PFAS in leachate; however, performance data are not currently available. As with activated carbon, the leachate may require pretreatment and the spent media would need to be handled as solid waste or otherwise treated.

Air stripping of landfill leachate is used for ammonia removal. While air stripping could be effective in the treatment pathway of leachate, if it is used before the removal of PFAS, it would likely lead to emissions of more volatile PFAS to air.

Recirculation of leachate within a landfill—a management strategy unique to MSW landfills—keeps the leachate within the landfill. Although recirculation can filter heavy metals and improve leachate quality, it is primarily used as a management option that may also help accelerate biodegradable waste decomposition. The recirculation of leachate in the landfill would return any PFAS to containment within the landfill. However, recirculation tends to increase the PFAS content of the landfill and may result in larger fugitive PFAS emissions since the waste will degrade faster.

Underground injection, specifically Class I injection, has also been used to manage landfill leachate in the United States (see Section 3.c).

Natural processes (such as constructed wetlands and phytoremediation) and biological processes (degradation, nitrification, and denitrification) are expected to be ineffective at treating and preventing release of many PFAS into the environment. Current biological treatment processes such as the activated sludge process and sequencing batch reactor have not been shown to be effective at treating many PFAS, but future research may show biological treatment can play a role in controlling some PFAS or converting them into other types of PFAS. Note that biological treatment does not necessarily result in PFAS releases directly into the environment, because the outputs can be further treated.

Leachate treatment through evaporation reduces the volume of leachate. Open-air evaporation methods may be effective at concentrating leachate but are a pathway for secondary PFAS releases to air. Commercial evaporators operated through the heat generated by the LFG combustion or other fuel sources are sometimes used at landfills. Exhaust gases emitted from the evaporators may be exposed to high temperatures, but those temperatures may not be high enough or last long enough to destroy PFAS (see Section 3.a).

3.b.iii.6 Landfill gas emission controls

3.b.iii.6.1 Landfill gas characteristics

Under the anaerobic conditions that dominate landfill environments, organic waste (e.g., food waste, paper, cardboard) decomposes and generates LFG. LFG in MSW landfills consists mostly of methane and carbon dioxide. In most landfills where gas is collected, it is burned for energy or to destroy the methane and other organic chemicals it contains. Even at sites that actively collect LFG, a fraction (EPA estimates 25% on average; AP-42, Vol.1, Chapter 2.4) of the LFG is emitted directly to the environment through the landfill surface and other routes. These uncontrolled emissions are referred to as fugitive losses.

Research has found that soluble PFAS with relatively high vapor pressures can be emitted into the atmosphere via the gas generated at landfills (Ahrens et al., 2011; Hamid et al., 2018; Wang et al., 2020; Weinberg et al., 2011). EPA is currently researching direct LFG sample evaluation for PFAS concentrations (see Section 5).

Unlike waste in MSW landfills, the C&D landfill waste that contributes most to LFG production is generally dominated by gypsum drywall (Yang et al., 2006). Gypsum drywall results in C&D LFG largely consisting of hydrogen sulfide, a highly pungent gas, with a smaller fraction of methane. Because C&D landfills generate a lower volume of gas than MSW landfills, LFG from C&D landfills is typically not collected and is often emitted to the environment without treatment.

3.b.iii.6.2 On- and off-site management of landfill gas

LFG collection and management are regulated under the CAA through New Source Performance Standards (NSPS) and National Emission Standards for Hazardous Air Pollutants (NESHAP) programs. After collection, LFG can be managed on site and burned using a flare. There are two basic types of flares common at MSW sites: open (candlestick) and enclosed flares. LFG can also be managed off site, where it is usually piped from the landfill to a nearby gas-fired system to generate heat or power.

On-site open flares must operate in accordance with key parameters for exit velocity and flare diameter for non-assisted flares (in 40 CFR 60.18). Additionally, a heat-sensing device must be installed to indicate continuous flame presence (but no specific temperature level). A landfill with an enclosed flare must reduce NMOCs to 20 parts per million by volume (ppmv) outlet or reduce NMOCs by 98 percent via a one-time performance test. Lowered emissions are maintained by following operating parameters set during the test for the requisite flare temperature and flow rate.

Combustion temperatures and duration may prove to be critical factors for destruction of PFAS in LFG. While on-site flare systems average 850°C (1,550°F) (U.S. EPA, 2008), power-generating engines and boiler systems may run even cooler. None of these systems operate at the temperature necessary to mineralize PFAS (1,100°C) as indicated in the EPA boiler database. See Section 3.a for a more complete discussion on conditions required for PFAS destruction.

Under the CAA regulations, if LFG is treated for sale or use in a beneficial energy recovery device, it must be treated according to a site-specific treatment plan before being utilized on site or piped out of the facility. Like liners, active gas collection systems are not required at all landfills, depending on the landfill size and level of NMOCs in the LFG. The presence or absence of an LFG collection and control system directly affects the release of PFAS via fugitive emissions.

LFG condensate is a liquid that has condensed in the LFG collection system piping during the extraction of gas from the landfill. In gases with high concentrations of water vapor, water condenses in pipes and accumulates in traps staged throughout the collection network. This condensate contains volatile compounds and accounts for a relatively small percentage of flow from a landfill. Gas condensate is commonly collected and managed with the landfill leachate as liquid waste. PFAS has been detected in LFG condensate, with perfluorobutanesulfonic acid (PFBS) being the dominant species at a concentration of 1,000 nanograms per liter (Li, 2011).

3.b.iv Potential for releases during landfilling

Over time, there is increased potential for certain pollutants to move into the environment beyond the footprint of the landfill. As water passes through the landfill, it may leach pollutants from the waste—including PFAS—and move them deeper into the waste mass. Controlled landfill leachate is collected and either reinjected, treated on site, or sent for off-site treatment. In the absence of leachate management systems, uncontrolled leachate releases occur when water travels through the waste, out of the landfill, and into groundwater or surface water (see Section 4).

LFG can also contain heavy metals, organic chemicals, and greenhouse gases and can produce explosive mixtures of gas in the vicinity of the landfill if not properly controlled (ATSDR, 2001). LFG capture technologies are widely used to control gaseous emissions from landfills. However, not all landfills are required to install and operate LFG collection systems. Even when an LFG collection system is installed and properly operated, a fraction of LFG is emitted directly to the environment through the landfill surface and other routes as fugitive losses (see Section 4).

Even years after landfill closure, direct human and ecological exposure to PFAS is possible if PFAS are emitted through the air, groundwater, or surface water, or if remaining waste is disturbed. Because landfills contain PFAS but are not designed to destroy these compounds, they represent a potential source of PFAS release well beyond the period in which landfills receive waste and the post-closure care period. Ideally, landfill areas could be used for other beneficial purposes after closure; this requires that potential risks from landfill contents be adequately managed. Direct exposure to PFAS from landfilled waste is possible for people living or working near landfills due to gaseous or water releases from the site. Additionally, if PFAS-containing wastes are present on trucks and moved to active cells, landfill employees are directly exposed to PFAS on the job. Research has shown elevated concentrations of PFAS in landfill ambient air (Hamid et al., 2018).

Potential impacts to vulnerable communities from landfill releases should also be considered. Considerations should include which media may be affected by releases, and the potential exposure pathways resulting from releases. The characteristics of the potential exposed communities (e.g., demographics, socioeconomic status, ambient exposures, health conditions) provide information regarding the potential for disproportionate and adverse health and environmental effects, including cumulative impacts.

Ensuring that PFAS-containing waste is properly disposed of in lined landfills with active gas collection systems where volatile PFAS or MSW landfills are involved, is instrumental in reducing PFAS emissions into the environment. Care must also be taken to avoid disposal of PFAS wastes in landfills that are known to have compromised liners, as PFAS in the leachate will migrate and contaminate groundwater.

3.b.v Testing and monitoring

Currently, there are no federal requirements for the monitoring of PFAS in landfill waste, leachate, condensate, or LFG. EPA and others are conducting studies to evaluate the effectiveness of landfills in containing or managing PFAS (see Section 3.b). Landfill operators should include PFAS measurements with the regular leachate monitoring parameters that are regularly evaluated. Analyzing PFAS in LFG may pose a challenge for comparison across destruction/control technologies and types of landfills due to the lack of standardized testing (see Section 5.a for research needs).

The lack of testing and monitoring limit effective risk communication and dialogue with adjacent communities. The importance of dialogue with communities, and in particular, communities that may be more vulnerable to PFAS exposures, cannot be overemphasized. Suggestions for risk communication are included in Section 4.

3.b.vi Uncertainties/unknowns

EPA plans to conduct further research on PFAS within landfills, including the potential for PFAS to migrate to leachate or LFG without adequate controls. As with thermal treatment, EPA lacks detailed information on the amounts and concentrations of PFAS and precursor compounds in wastes that are landfilled. Limited research is providing some information on PFAS fate and portioning in landfill environments. The bulk of PFAS seems to remain with the solid waste mass, with quantifiable percentages in LFG and leachate. However, there is high level of uncertainty in these values and more data are needed. Sampling and analytical methodologies must be developed to quantify potential PFAS flows out of landfills—an effort that may be complicated by the long lifespan of some PFAS. Additionally, the efficacy of treatment options for PFAS captured by leachate and LFG systems is not well understood and is in some cases intrinsically entwined with TWTDS and thermal treatment options. EPA continues to research these complex and important issues. Refer to Section 5 for a summary of EPA and DoD's planned research activities specific to landfill containment, wastewater treatment, and thermal treatment of PFAS.

When evaluating landfill options, decision-makers (e.g., managers of PFAS-containing materials and PFAS-containing waste) should consider potential impacts to communities, including vulnerable populations. Uncertainties regarding LFG or leachate releases could have consequences for communities that are exposed and may be disproportionately affected. EPA recommends that decision-makers screen communities located in the vicinity of potential releases from the destruction, disposal, and storage options (considering fate and transport) in order to consider the potential for adverse or disproportionate impacts (see Section 4) and to consider potential measures to prevent, reduce, or address such impacts. Depending on site-specific circumstances (e.g., PFAS concentrations, impacted media, and potential exposure pathways), the size and shape of this area (vicinity) will vary.

3.b.vii Summary

Due to widespread use and disposal through typical waste management pathways, many PFAS-containing wastes are currently managed through containment in landfills. Though landfills (except C&D landfills) are designed for permanent waste containment and management of liquid and gas production, it is currently unclear if all landfills used to dispose of PFAS and PFAS-containing materials have controls that are effective for managing PFAS discharges and emissions from waste streams. PFAS-containing waste should not be placed in C&D landfills that don't have liner systems to intercept and manage the leachate. Even with appropriate liners, research has shown an overall collection efficiency of 98 percent for geomembrane and composite liners. Given the chemical makeup of PFAS, some compounds are expected to persist in landfills for years.

As leachate passes through landfills, PFAS are released from degrading wastes. PFAS have been detected in the leachate for all types of landfills, and improper management of landfill leachate would result in PFAS releases. To date, research on the efficacy of wastewater treatment technologies in capturing or destroying PFAS in leachate is limited, as most landfills are not currently required to treat leachate for PFAS. The existing data suggest that adsorption and separation treatment mechanisms have been shown to concentrate or capture PFAS from landfill leachate (see Table 3-4). Other leachate management options can control the release of PFAS, including recirculation and solidification, which return PFAS to the landfill. Leachate treatment through natural processes such as constructed wetlands, land application, or ponds is ineffective for preventing the release of PFAS into the environment. More data are needed on the volatilization of PFAS during leachate handling and treatment.

PFAS can be emitted with LFG that is generated as waste decomposes over time. On-site and off-site LFG management commonly uses flares, engines, or boilers to combust LFG. Combustion temperature and duration could be critical factors for the destruction of PFAS in LFG, as discussed in Section 3.a.

To varying degrees, hazardous waste or MSW landfills are feasible and effective disposal options for PFAS and PFAS-containing materials. Permitted hazardous waste landfills employ the most extensive set of environmental controls (e.g., double liner systems with leachate collection and leak detection) and practices (e.g., extensive record keeping) that are currently available for the containment of PFAS waste (see Table 3-2) and as a result would be more effective at minimizing PFAS release into the environment than other landfill types. Hazardous waste landfills do not typically accept biodegradable wastes (e.g., biosolids, food scraps), and are not biologically active, so they do not generally need to be equipped to collect and control LFG. Therefore, hazardous waste landfills are not expected to cause PFAS emissions via LFG and may not be appropriate options for biodegradable PFAS-containing wastes.

Modern MSW landfills, when constructed with appropriate controls (e.g., liner system and leachate and gas collection and management systems), can also control the release of PFAS into the environment. Even with these controls in place, the proper management of landfill gaseous and liquid releases needs to be applied to minimize PFAS release into the environment (see Section 3.b.iii). Care must be taken to apply the leachate control technologies that are effective at containing (e.g., solidification, recirculation) or destroying PFAS (see Table 3-4).

Given the high level of uncertainty associated with PFAS behavior in landfills, the effects of PFAS on liner integrity, gaseous emissions from landfills, the effectiveness of leachate treatment for PFAS removal, and the levels and types of PFAS in landfill leachate—additional research will help to further evaluate this disposal method for PFAS and PFAS-containing wastes. See Figure 3-2 for a summary of recommended landfill disposal options for PFAS-containing wastes and considerations for selecting the best option based on both waste and landfill characteristics.

See Appendix D for a summary of costs and considerations.

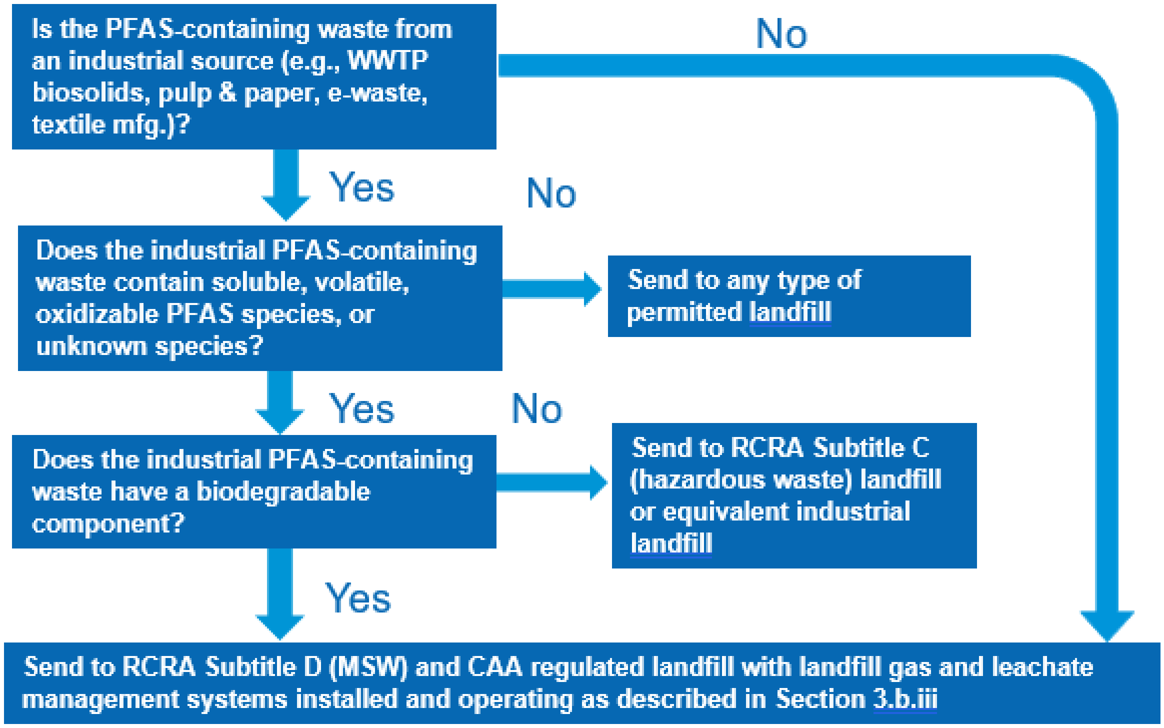

Figure 3-2. Decision framework for selection of landfill disposal for PFAS-containing industrial waste.

3.b.viii References for Section 3.b

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3.c Underground injection

Underground injection wells are a feasible and effective disposal option that normally should minimize release of PFAS into the environment. However, underground injection wells are only suited for the disposal of liquids and are restricted to locations with suitable geology. PFAS-containing fluids are currently handled similarly to other waste streams that are injected deep into geologic formations in Class I non-hazardous industrial and hazardous waste disposal wells, while shallow Class V wells would be inappropriate for the management of PFAS-containing fluids. The limited number of wells currently receiving PFAS and accepting off-site waste, well location, and waste transportation logistics may significantly limit the type and quantity of PFAS-containing fluids appropriate for underground injection.

3.c.i Types of wells

Underground injection is generally defined as the subsurface emplacement of fluids through a well. Under the Safe Drinking Water Act (SDWA), EPA is authorized to regulate the permitting of injection wells—including construction, operation, monitoring, and proper closure—for the purpose of protecting USDWs. Underground injection control (UIC) regulations are found in 40 CFR parts 144 to 148.

EPA's UIC program shares information for owners and operators of injection wells, regulators, and the public about safe injection well operations to prevent the contamination of USDWs. Under the UIC program, EPA regulates the permitting of the following well types:

- **Class I** wells inject into geologic formations below the lowermost USDW and are further subdivided into four categories: municipal wastewater, radioactive waste, hazardous waste, and non-hazardous industrial waste disposal wells (see Figure 3-3).
- **Class II** wells are used for injection activities associated with oil and gas production and hydrocarbon storage.
- **Class III** wells are solution mining wells used to inject fluids for the purposes of dissolving and extracting minerals.
- **Class IV** wells, with limited exceptions, have been banned by EPA since 1984 and were used to inject hazardous or radioactive waste into or above geologic formations containing USDWs.
- **Class V** wells include injection wells that are not included in Classes I, II, III, IV, or VI. EPA has identified multiple subtypes, including stormwater drainage wells, septic system leach fields, and agricultural drainage wells.
- **Class VI** wells are used to inject and geologically sequester carbon dioxide.

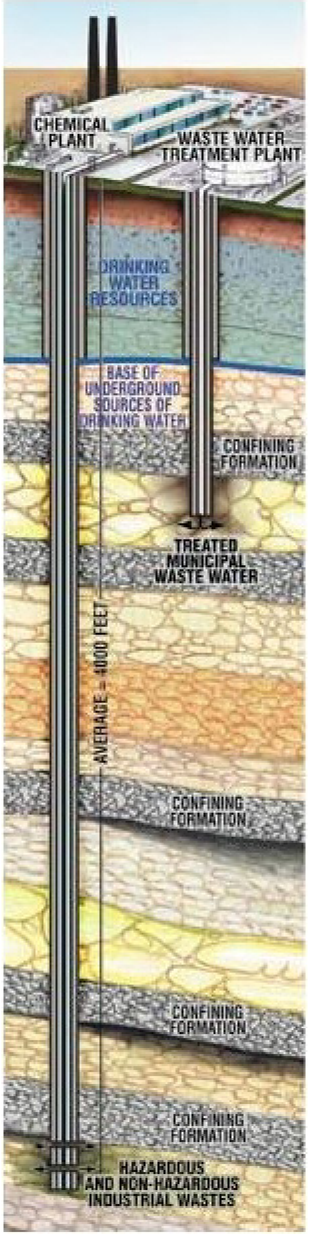

Figure 3-3. Class I wells.

EPA has determined the use of Class I non-hazardous industrial waste and hazardous waste wells for high concentration liquid PFAS waste has a lower potential for environmental release when compared to other PFAS destruction and disposal options, and there is relatively low uncertainty in this determination. Class I underground injection wells are designed to dispose of and isolate liquid waste below the land surface and beneath USDWs. The standards associated with the construction, operation, and monitoring of Class I wells are designed to ensure protection of USDWs. These standards include at least one confining layer between the zone in which the fluid will be emplaced and the lowest USDW. While Class I wells are an option for managing PFAS-containing fluids, this technology may not be appropriate everywhere. The suitability of a site for injection is dependent on the geologic formations in the area. Sites need to be evaluated to ensure that there is an appropriate confining zone and that there are geologic formations that can receive fluids.

Class I wells for non-hazardous industrial and hazardous waste are currently being used for disposal of PFAS-containing fluids and are the focus of this section. PFAS-containing fluids that may be disposed of via Class I underground injection wells may originate from industrial activities such as chemical production (e.g., products and byproducts) and waste management operation (e.g., landfill leachate).

The standards associated with the permitting, construction, operation, and monitoring of Class I hazardous waste wells, which are regulated under RCRA and SDWA, are more stringent than for non-hazardous industrial waste disposal wells.

3.c.ii Siting, engineering, and operational controls

3.c.ii.1 Overview of the regulatory framework

Underground injection through Class I non-hazardous industrial and hazardous waste wells is a long-standing, well-regulated disposal technology. Underground injection has been used as a waste disposal practice in the United States since the 1930s, beginning with disposal of brines from oil production activities. Underground injection of wastewater from industrial facilities has been in practice since the 1950s. In response to concerns around underground injection activities and incidents of well failure, in 1974 (the same year the SDWA was enacted), "EPA issued a policy statement in which it opposed underground injection without strict control and clear demonstration that the wastes will not adversely affect groundwater supplies" (U.S. EPA, 2001).

Final UIC regulations were published six years later, and federal and state regulation and oversight has been informed by extensive reviews of injection practices and the associated risks over time (U.S. EPA, 2001). The requirements for Class I wells under 40 CFR part 146 and 40 CFR part 148 (which applies to hazardous waste wells only) are designed to ensure that injected fluids cannot migrate into USDWs through either of two potential pathways: loss of waste confinement or "improperly plugged or completed wells or other pathways near the well" (U.S. EPA, 2001). These requirements include, but are not limited to:

- Proper siting.
- Conducting geologic and hydrogeologic studies that demonstrate that injected fluids will not endanger USDWs.
- Implementing specific design, construction, and operation requirements.
- Implementing continuous monitoring and periodic monitoring and testing requirements.
- Performing appropriate well closure and plugging.

Specific components of these requirements are discussed further below.

3.c.ii.2 Class I non-hazardous industrial and hazardous waste wells

Underground injection to Class I non-hazardous industrial and hazardous waste wells reduces the potential risks of human exposure to injected materials, avoiding discharge to surface and shallow groundwater and generating little or no air emissions. When injected into non-hazardous industrial or hazardous waste Class I wells, fluids are placed below the lowermost USDW. The area into which waste is injected is referred to as the injection zone. Injection zones of Class I wells typically range from 1,700 to more than 10,000 feet in depth (U.S. EPA, 2001). Injection zones are porous and permeable geologic formations. They are separated from USDWs by one or more confining layers of impermeable rock. The confining layers prevent injected fluids from migrating vertically into a USDW.

Class I wells are sited in geological areas that are conducive to injection operations. Siting considerations include ensuring that injected fluids will not migrate through natural fractures and faults from the injection zone into USDWs. Likewise, well operators are required to demonstrate the absence of non-natural pathways (e.g., abandoned wells) or other nearby active wells that could allow for movement of injected fluids into USDWs, within a prescribed area surrounding the well (known as the area of review). In addition to the safeguards offered by siting, engineering, and operating requirements, well design and construction requirements incorporate redundant safety features, and construction materials are "corrosion-resistant and compatible with the wastewater and the formation rocks and fluids into which they come in contact" (U.S. EPA, 2001). Class I wells might also use multiple strings of well casing, inject through tubing set on a packer, and be constructed with adequate cement alongside the entire well string to protect any USDWs.

3.c.ii.3 Potential for Release

Permitted underground injection of fluids through Class I non-hazardous industrial and hazardous waste wells ensures that injected fluids are confined and cannot enter USDWs—the pathway of concern for this waste disposal technology. In its 2001 study of risks associated with Class I wells, EPA stated that the "probability of Class I well failures, both non-hazardous and hazardous, has been demonstrated to be low. In the unlikely event that a well would fail, the geology of the injection and confining zones serves as a final safety net against movement of wastewaters to USDWs" (U.S. EPA, 2001).

Injection well operators invest millions of dollars in the permitting, construction, and operation of wells. Development of Class I non-hazardous industrial and hazardous waste wells is a resource-intensive process, with the geologic limitations noted previously. In addition, siting requirements limit the areas in the country where Class I wells can be located (see Section 3.c.iii). Routine operation and maintenance include addressing requirements for extensive mechanical integrity testing, monitoring, and periodic submission of permit/no-migration petitions.

3.c.ii.4 Additional requirements for Class I hazardous waste wells

Class I hazardous waste wells are highly protective of USDWs and include additional requirements beyond those of Class I non-hazardous industrial waste wells. The 1984 Hazardous and Solid Waste Amendments to RCRA prohibited land disposal of hazardous waste, including via underground injection, with limited exceptions. Hazardous waste disposal via Class I injection wells is permitted if the operator can demonstrate that the waste will remain where it has been injected for as long as it remains hazardous (defined under regulation as a period of up to 10,000 years). To demonstrate this, Class I hazardous waste well operators must receive approvals of "no-migration petitions" from EPA.

No-migration petitions present information and modeling results using data on local and regional geology, waste characteristics, geochemical conditions of the well site, injection history, and many other factors, which EPA reviews to determine whether the petitioner has adequately demonstrated that the waste will not migrate from the disposal site for as long as it remains hazardous. Furthermore, Class I hazardous waste well facilities are subject to inspections and well operators must conduct annual testing and analysis to demonstrate they are meeting the conditions of the permit and that all assumptions, projections, and models are still appropriate and valid. This includes performing mechanical pressure tests and geophysical logging tests to assess well integrity both internally and externally to ensure injected fluids are being emplaced and are remaining within the injection zone.

3.c.iii Availability

The United States currently has 925 Class I wells. Slightly less than half (48 percent) are permitted for non-hazardous industrial waste injection. Approximately 15 percent are permitted for hazardous waste disposal. The remainder are permitted for municipal wastewater disposal. Currently, EPA is aware of both non-hazardous industrial and hazardous Class I wells that manage PFAS-containing fluids. However, there is currently no national information on the number and location of Class I non-hazardous industrial or hazardous waste wells that could accept PFAS-containing waste or are willing to accept waste not generated on site.

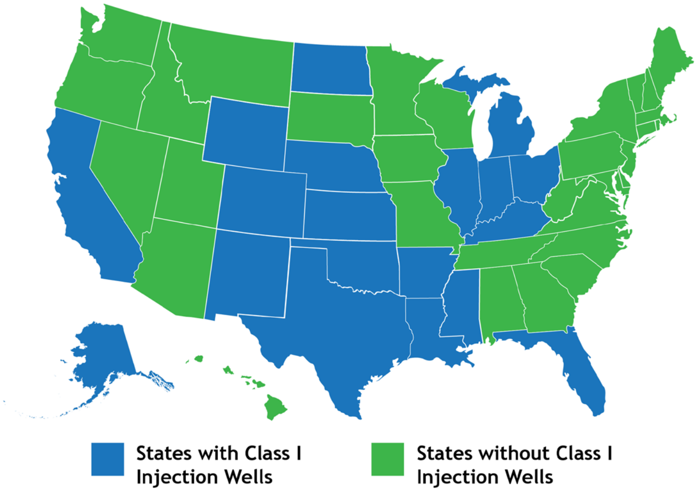

Figure 3-4. States with Class I non-hazardous or hazardous waste injection wells.

Figure 3-4 shows all states that currently have at least one permitted Class I injection well of any type (non-hazardous industrial or hazardous waste). Table 3-5 provides a more detailed breakdown of the number of permitted Class I wells by state.

Table 3-5. Inventory of Permitted Class I Non-Hazardous and Hazardous Waste Wells in the United States (FY 2022; Source: EPA)[19]

Location of Wells		Number of Wells	
EPA Region	State/Tribe	Class I Non-Hazardous	Class I Hazardous
4	Florida	343	-
	Kentucky	2	-
	Seminole Tribe	3	-
	Mississippi	4	4
5	Illinois	5	2

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[19] EPA's inventory of Class I non-hazardous waste wells consists of all non-hazardous waste wells, including municipal and industrial waste wells. However, the municipal waste wells are located exclusively in Florida.

Location of Wells		Number of Wells	
EPA Region	State/Tribe	Class I Non-Hazardous	Class I Hazardous
	Indiana	13	4
	Michigan	40	7
	Ohio	7	11
6	Arkansas	8	4
	Louisiana	16	19
	New Mexico	6	-
	Oklahoma	6	-
	Osage Nation	1	-
	Texas	83	77
7	Kansas	58	7
	Nebraska	11	-
8	Colorado	13	-
	North Dakota	8	-
	Wyoming	89	-
9	California	52	-
10	Alaska	22	-
Total		**790**	**135**

3.c.iii.1 Class I non-hazardous industrial waste wells

Non-hazardous industrial waste wells are currently located across 19 states and two Tribal lands, though the majority are in five states—Wyoming, Texas, Kansas, California, and Michigan. There are currently no Class I non-hazardous waste wells in EPA Regions 1, 2, or 3 (see Figure 3-4 and Table 3-5). Disposal to this type of well requires well operators to apply for and receive a permit for specific waste streams. Current Class I injection wells may have limited capacity for PFAS-containing fluids, as many of them are used for specific purposes and disposal of waste generated on site. To begin accepting PFAS-containing fluids, existing well permits would have to be modified to recognize that the facility is accepting waste from other entities and authorize the facility to inject modified waste streams.

Well operators must also consider the well's capacity to accept additional volumes of waste and the compatibility of the PFAS-containing fluids with the well material, the geochemistry of the injection formation and formation fluids, and the properties of other injected wastes.

3.c.iii.2 Class I hazardous waste wells

Hazardous waste wells are currently located in nine states; however, most are in Texas and Louisiana. Most Class I hazardous waste wells are sited at industrial facilities and dispose of waste generated on site (U.S. EPA, 2016). There are currently no Class I hazardous waste wells in EPA Regions 1, 2, 3, 8, 9, or 10 (see Figure 3-4 and Table 3-5). Because of this geographic concentration of Class I hazardous waste wells, waste producers may face transportation and logistical challenges.

EPA anticipates the number of current Class I hazardous waste wells that would begin accepting new sources of PFAS-containing waste to be very limited due to the necessary modifications of Class I hazardous waste well permits (e.g., increased injection volumes, changes to waste streams, no-migration petitions). These modifications would involve engineering and scientific evaluations, modeling, and public hearings.

3.c.iv Testing and monitoring

Class I non-hazardous industrial waste and hazardous waste disposal wells are subject to extensive testing and monitoring requirements established under federal regulations. Requirements for hazardous waste wells are more stringent than for non-hazardous industrial waste wells. Additionally, by law, states with primary enforcement authority for Class I wells may have more stringent testing and monitoring requirements. Class I monitoring and testing requirements are designed to ensure there are no leaks within or out of the well and that all injected fluid is contained in the injection zone.

Broadly, Class I well operators must:

- Analyze characteristics of injected fluids at a frequency that results in representative data.
- Continuously monitor and record injection pressure, annulus pressure, flow rate, and volume.
- Conduct internal and external mechanical integrity testing.
- Monitor for fluid release into the USDW within the area of review.

Class I hazardous waste wells operators must conduct mechanical integrity testing more often than Class I non-hazardous industrial waste operators. They also must establish and follow additional procedures for reporting and correcting mechanical integrity problems. Class I hazardous waste well operators must also develop and follow a waste analysis plan and conduct annual tests of cement at the base of the well (U.S. EPA, 2015).

Additional information on testing and monitoring requirements, including mechanical integrity testing, is included in EPA's summary document *Requirements for All Class I Wells and Class I Hazardous Waste Wells* and EPA Region 5's guidance on *Determination of the Mechanical Integrity of Injection Wells*.

Communicating testing and monitoring results with adjacent communities is particularly important given the high-profile nature of PFAS. The importance of encouraging dialogue with communities, especially communities that may be more vulnerable to PFAS exposures, cannot be overemphasized. Suggestions for risk communication are included in Section 4.

3.c.v Uncertainties/unknowns

The fate and transport of PFAS in the subsurface depends on the chemical and physical properties of specific PFAS and the geochemical properties of the injection zone. Understanding of the long-term fate and transport properties of PFAS (including precursors) in the injection zone is currently limited. Studies have shown wide ranges in PFAS properties, and these can be altered by mixture effects and interactions with co-contaminants. This creates uncertainty in predictions of PFAS contaminant release and longevity in the injection zone. These uncertainties need to be considered in the development of the required no-migration petition for disposal of PFAS in Class I hazardous waste wells.

When evaluating underground injection options, decision-makers (managers of PFAS-containing materials and PFAS-containing waste) should consider potential impacts to communities, including vulnerable populations. Underestimation of the potential for PFAS release could increase impacts to local communities, including vulnerable communities. EPA recommends that decision-makers screen communities located in the vicinity of the destruction, disposal, and storage options in order to consider the potential for adverse, including any cumulative and disproportionate impacts (see Section 4), and potential measures to prevent, reduce, or address such impacts. Depending on site-specific circumstances (e.g., PFAS concentrations, impacted media, and potential exposure pathways), the size and shape of this area (vicinity) will vary.

3.c.vi Summary

EPA has determined the use of Class I non-hazardous industrial waste and hazardous waste wells for high concentration liquid PFAS waste has a lower potential for environmental release when compared to other PFAS destruction and disposal options. Permitted underground injection of fluids through Class I non-hazardous industrial and hazardous waste wells ensures that injected fluids are confined and cannot enter USDWs—the pathway of concern for this waste disposal technology. Additional Class I wells may need to be constructed where geologically suitable, and existing well permits may need to be modified to meet the capacity needs for PFAS disposal. Research on the long-term fate and transport of PFAS (including precursors) to predict release potential in the injection zone could support future permits.

See Appendix D for a summary of costs and considerations.

3.c.vii References for Section 3.c

U.S. EPA (Environmental Protection Agency). (2001). *Class I Underground Injection Control Program: Study of risks associated with Class I underground injection wells* (EPA 816-R-01-007). *https://www.epa.gov/sites/production/files/2015-07/documents/study_uic-class1_study_risks_class1.pdf*

U.S. EPA (Environmental Protection Agency). (2015). *Requirements for all Class I wells and Class I hazardous waste wells. https://www.epa.gov/sites/production/files/2015-10/documents/page_uic-class1_summary_class1_reqs_508c.pdf*

U.S. EPA (Environmental Protection Agency). (2016). *Class I industrial and municipal waste disposal wells. https://www.epa.gov/uic/class-i-industrial-and-municipal-waste-disposal-wells*

U.S. EPA (Environmental Protection Agency). (2021). *UIC Injection Well Inventory. https://www.epa.gov/uic/uic-injection-well-inventory*

4. Considerations for Potentially Vulnerable Populations Living Near Likely Destruction or Disposal Sites

The FY 2020 NDAA specifies that the interim guidance consider:

- The potential for releases of PFAS during destruction or disposal, including through volatilization, air dispersion, or leachate.
- Potentially vulnerable populations living near likely destruction or disposal sites.

This section describes potential releases and exposure pathways associated with destruction or disposal sites. It defines potentially vulnerable populations and provides interim guidance on considering vulnerable populations when assessing the potential impacts of releases. This section also suggests approaches to screen communities located in the vicinity of the destruction, disposal, and storage options in order to consider the potential for adverse and disproportionate impacts and potential measures to prevent, reduce, or address such impacts. Depending on site-specific circumstances (e.g., PFAS concentrations, impacted media, and potential exposure pathways), the size and shape of this area (vicinity) will vary. Focusing on impacts and potential ways to protect vulnerable populations, such as children who are more vulnerable than adults to chemicals like PFAS, will ensure the protection of all populations exposed to PFAS.

The primary audience of this interim guidance is managers of PFAS-containing materials and PFAS-containing waste and regulators who need to identify the most effective means for destroying or disposing of these materials, including to protect nearby communities who may include vulnerable populations. Others may find it useful, such as communities themselves, as well as community relations personnel, and the public. This section is not a primer on risk assessment and risk communication; rather, it contains pointers and references to existing information.

4.a Potential releases from destruction and disposal facilities

EPA develops regulations, guidance, and policies that ensure the safe management and cleanup of waste.[20] Nonetheless, it is possible for destruction or disposal activities to release PFAS.

For example, as described in Section 3.a.iii, thermal treatment activities could potentially release PFAS to the environment via stack emissions if adequate combustion conditions are not achieved or adequate pollution control devices are not used. Releases can also occur from the management of thermal treatment process residuals such as liquid discharges from acid gas scrubbers, air pollution control device media, and incinerator bottom ash. In addition, if uncontrolled, leachate can travel out of landfills (see Section 3.b.i) and into groundwater or surface water. Disposal of PFAS could also result in potential

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[20] Information on EPA's actions to address PFAS can be found here: *https://www.epa.gov/pfas/key-epa-actions-address-pfas*

releases from increased transport, management, and handling of waste associated with all the available technologies.

Risk assessment and communication are important tools to help protect communities and the environment from potential releases of harmful substances. Tools from the risk assessment process may also be useful when considering whether a facility is an appropriate option for receiving PFAS-containing waste. Risk communication and community engagement are important for building trust and addressing concerns about potential releases. EPA has developed resources for assessing, managing, and communicating environmental risks, including interim guidance and tools available to stakeholders and the public. These resources are summarized in Section 4.c.

4.b Potentially vulnerable populations

To consider potential impacts on vulnerable populations in PFAS destruction and disposal decision contexts, it is helpful to keep in mind that vulnerability refers to characteristics of individuals or populations that place them at increased risk of an adverse health effect. Vulnerability includes the cumulative impacts of economic, demographic, social, cultural, psychological, and physical states of the individual person or population that influence patterns of exposure to environmental contaminants and alter the relationship between the exposure of the environmental agent and adverse health outcomes (U.S. EPA, 2019a).

Described in Guidelines for Exposure Assessment (U.S. EPA, 2019), EPA's Framework for Cumulative Risk Assessment (U.S. EPA 2003) describes four properties of vulnerability:

- Differential susceptibility: An increased likelihood of sustaining an adverse effect from exposure to an agent. For example, an individual, group, or population might be more likely to show a response to an agent at a lower dose than the general population because of a preexisting health condition (e.g., asthma, cardiovascular disease, disability), genetic variation, prior damage from exposure, concurrent exposures to other stressors or developmental or life stage [e.g., children, older adults, pregnant women]).
- Differential exposure: Differences in exposure (e.g., magnitude, duration, frequency, pathway, route) from a variety of factors, including life stage, socioeconomic status, and cultural characteristics. For example:
    - Children might have a higher exposure and proportionally higher body burden of pesticides than adults because of their behavior patterns or food consumption (Moya et al., 2004; NRC, 1993).
    - When neighborhoods are racially or economically segregated, low-income people and people of color might live in neighborhoods or conditions where pollution sources are more likely to be sited and concentrated and thus where they experience higher exposures to air pollution (Lopez, 2003).
    - Studies on fish consumption and subsistence fishing patterns have documented racial/ethnic differences that can increase exposure from persistent, bioaccumulative chemicals in fish or wildlife (Burger, 2000; Burger 2002a; Burger 2002b; Burger et al. 2001; Burger et al., 1999a; Burger et al., 1998; Burger et al., 1993; Burger et al., 1999b; Corburn, 2002).
    - Tribal Nations and Native Americans can be exposed differentially to toxicants when dietary patterns involve consumption of locally caught fish or game for traditional or religious reasons (Fitzgerald et al., 1999; Fitzgerald et al., 1995; Fitzgerald et al., 1998; Fitzgerald et al., 2001; Harper et al., 2002; Schell et al., 2003).

- Differential preparedness: The regulatory frameworks, protections, and resources that an individual, community or population uses or can access to withstand the insult of agents.
- Differential ability to recover: Refers to resources, repair, and resilience systems, such as income level, ability to move from an affected area or access to health care, which can affect recovery from the effects of an agent.

Communities with environmental justice concerns experience disproportionate and adverse human health or environmental burdens. These burdens may arise from a number of causes, including the cumulative impacts of inequitable access to clean water, clean air, natural places, and resources for other basic human health and environmental needs; the concentration of pollution, hazardous waste, and toxic exposures; and underinvestment in affordable housing that is safe and healthy and in basic infrastructure and services to support such housing, including safe drinking water and effective sewage management. The cumulative impacts of exposure to those types of burdens and other stressors, including those related to climate change and the environment, are also important indicators of vulnerable populations (Executive Order 14096, 2023).

The following sections provide examples of tools, methods, and approaches to identifying and considering vulnerable populations.

4.c Considering vulnerability

Executive Order (EO) 14096, "Revitalizing Our Nation's Commitment to Environmental Justice for All"" (issued by President Biden in April 2023), directs federal agencies to advance the goal of environmental justice, which means: "the just treatment and meaningful involvement of all people, regardless of income, race, color, national origin, Tribal affiliation, or disability, in agency decision-making and other federal activities that affect human health and the environment so that people:

(i) are fully protected from disproportionate and adverse human health and environmental effects (including risks) and hazards, including those related to climate change, the cumulative impacts of environmental and other burdens, and the legacy of racism or other structural or systemic barriers; and

(ii) have equitable access to a healthy, sustainable, and resilient environment in which to live, play, work, learn, grow, worship, and engage in cultural and subsistence practices."[21]

To advance this charge and apply the best available science on the vulnerability of exposure to children and other sensitive populations, EPA has developed tools, methods, and approaches to identify and assess the potential for adverse and disproportionate impacts, including risks, to potentially vulnerable populations, including communities with environmental justice concerns.

4.c.i Identifying potentially vulnerable populations

The consideration of potentially vulnerable populations living near likely PFAS destruction or disposal sites starts with identifying and characterizing adjacent and potentially exposed communities and populations. EPA provides the following tools to assist with this task:

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[21] See EO 14096, section 2(b), 88 FR 25251 (2023). EO 14096 builds on and complements EO 12898, "Federal Actions to Address Environmental Justice in Minority Populations and Low-Income Populations" (1994).

4.c.i.1 EJScreen

This EJ mapping and screening tool (*EJScreen*) provides EPA with a nationally consistent dataset and approach for combining environmental and demographic indicators. EJScreen users choose a geographic area; the tool then provides demographic and environmental information for that area. This is EPA's principal tool for characterizing and identifying communities with environmental justice concerns next to facilities or sources of pollution. Users can download reports that summarize search findings.

To summarize how environmental indicators and demographics come together in the same location, EJScreen 2.2 uses 13 EJ indexes that reflect the following 13 environmental indicators:

- Particulate Matter 2.5
- Ozone
- Diesel Particulate Matter
- Toxic Releases to Air
- Air Toxics Cancer Risk
- Air Toxics Respiratory Hazard Index
- Traffic Proximity
- Lead Paint
- Risk Management Plan (RMP) Facility Proximity
- Hazardous Waste Proximity
- Superfund Proximity
- Underground Storage Tanks (USTs) and Leaking USTs
- Wastewater Discharge.

The EJ Indexes combine environmental indicators with information about the low-income and racial and ethnic minority population in a Census block group. A Census block group is the smallest level of geography you can get basic demographic data for, such as total population by age, sex, and race (U.S. Census Bureau, 2023). For example, the EJ Index for traffic combines:

- The traffic indicator
- The low-income population
- The racial and ethnic minority population.

EJScreen presents results in terms of percentiles, allowing community comparisons with the rest of the state, an EPA Region, or the nation, and to identify which communities may be potentially overburdened.

Users can also look at the factors independently. In addition, EJScreen offers several more maps based on demographic and socioeconomic data (e.g., U.S. Census data, the American Community Survey) as well as indicators of other stressors, such as certain health indicators and access to medical care.

4.c.i.2 Enforcement Compliance and History Online (ECHO)

EPA's *Enforcement Compliance and History Online* (ECHO) website lets users search for facilities in communities to assess their compliance with environmental regulations that protect human health and the environment. Permit writers may consider susceptibility and vulnerability when establishing permit conditions and ECHO is a useful tool to evaluate whether a facility is currently violating or has previously violated those conditions. Users access ECHO to:

- Search for facilities
- Investigate pollution sources
- Search for EPA enforcement cases
- Examine and create enforcement-related maps
- Analyze trends in compliance and enforcement data
- Identify facilities on or near Tribal land.

4.c.i.3 PFAS Analytic Tools

The *PFAS Analytic Tools* integrate 11 national datasets on PFAS into an interactive, web-based software. Users can filter data by PFAS or geographic area, explore application charts and maps, or download and open the files to explore the data. Information includes Clean Water Act discharges from permitted sources, a log of spills reported containing PFAS constituents, lists of facilities historically manufacturing and importing PFAS, federally owned locations where PFAS is being investigated, a history of known transfers of PFAS waste and how those wastes were managed, facilities that EPA knows is processing, releasing, and disposing of PFAS, PFAS detections in the environment (e.g., surface water, fish tissue), and drinking water testing results.[22] The data cover a broad list of PFAS and represent EPA's ongoing efforts to consolidate the growing amount of testing information that is available.

4.c.i.4 Private Domestic Well Map

EPA's *Private Drinking Water Wells* collection of online mapping tools provides information on drinking water providers and sources (including domestic [i.e., private] water wells and how many people are using them), as well as potential sources of contamination. Understanding the density and geospatial location of private domestic wells, and housing units relying on them, improves detection and response efforts.

EPA developed this mapping tool based on two methods to estimate private well density. The maps combine reported wells in 20 states with available well log data and the net housing unit method that is available in all 50 states for estimating domestic well use. The maps do not display the location of private wells. The value of this mapping tool is its identification of areas that may be vulnerable to groundwater contamination based on the presence of private wells. The source of contamination widget allows users to identify potential sources of contamination in defined locations.

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[22] Information available to EPA is limited. For example, EPA has identified transfers of PFAS waste in EPA's eManifest system that tracks shipments of federally regulated hazardous wastes. Since PFAS are not currently regulated as hazardous waste, though, this information is not required to be reported – resulting in an incomplete picture of PFAS waste transfers in the United States. What EPA is able to identify in eManifest comes from querying free text fields where PFAS information is mentioned or a state has regulated PFAS as a hazardous waste and has established a state hazardous waste code. The tool also shows release and transfer information from the Toxics Release Inventory. However, the number of reporting facilities and reported recipient facilities has been low since the beginning of PFAS TRI Reporting likely associated with the *de minimis* exemption. A final rulemaking removing that exemption may lead to a more complete picture of releases and transfers in future reporting years. The PFAS Analytic Tools does not currently include information on PFAS destruction efficiency or efficacy.

4.c.i.5 Risk-Screening Environmental Indicators (RSEI) model

EPA's Risk-Screening Environmental Indicators (RSEI) model helps policy makers, researchers, and communities explore data on toxic chemicals being managed by industrial and federal facilities. RSEI incorporates information from the TRI on the amount of toxic chemicals released or transferred from facilities, together with factors such as the chemicals' fate and transport through the environment, each chemical's relative toxicity, and potential human exposure. RSEI model results can be used to help establish priorities for further investigation and to look at changes in potential human health impacts over time.

4.c.i.6 EnviroAtlas

EPA's *EnviroAtlas* highlights relationships between communities, land use, and environmental quality. The mapping functions help identify potential uses of land that, when combined with site contaminant data, may indicate possible exposure pathways. Such land use relationships can trigger deeper investigation into land uses to characterize relevant activities to inform site sampling and risk assessment.

The information gathered can be used to modify Conceptual Site Models, which are descriptions of contaminant sources, releases, transport pathways, and potential receptors. Accurate and complete Conceptual Site Models are required to ensure that samples are collected in appropriate media and risks are analyzed for potentially exposed receptors.

4.c.i.7 Climate and Economic Justice Screening Tool (CEJST)

The *Climate and Economic Justice Screening Tool* (CEJST) helps federal agencies identify disadvantaged communities geographically that are marginalized by underinvestment and overburdened by pollution. These communities are in Census tracts that are at or above the thresholds in one or more of eight categories of burden or are located on the lands of Federally Recognized Tribes.[23] The current version of the tool evaluates the likelihood of disadvantage based on climate change, energy, health, housing, legacy pollution, transportation, water and wastewater, and workforce development factors, each in combination with low-income or educational attainment status. The current version of the tool will be updated based on more feedback and research.

4.c.i.8 Centers for Disease Control and Prevention (CDC) EJ Dashboard

The Centers for Disease Control and Prevention (CDC) *EJ Dashboard* provides contextual public health messaging and allows users to evaluate several screening factors, including community characteristics, environmental exposures, health burden, and indexes. Assessments are based on ZIP code. and census tracts. Evaluating communities surrounding a facility may require the use of multiple ZIP codes and census tracts. Community characteristics include demographics and social vulnerability, as well as a description of the built environment (e.g., access to parks). The Social Vulnerability Index (SVI) uses U.S. Census data to determine the social vulnerability of every Census tract. Census tracts are subdivisions of counties for which the U.S. Census collects statistical data. The SVI ranks each tract across 15 social factors, including poverty, lack of vehicle access, and crowded housing, and groups them by theme.

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[23] Census tracts that are completely surrounded by disadvantaged communities and are at or above the 50th percentile for low-income are also identified as disadvantaged by the CEJST.

The EJ Dashboard also provides more information on environmental exposures, health, and EJ. The Environmental Exposures tab presents environmental pollutant data, including water quality, age of housing, $PM_{2.5}$, and ozone. It also includes factors related to climate change. Under "Health Burden," users can examine health vulnerability based on access to health insurance and hospitals, disability, infant mortality, and percentage of low birth weights. The *Environmental Justice Index* is a calculation based on the combined rankings of the Environmental Burden Module, the Social Vulnerability Module, and the Health Vulnerability Module. The Environmental Justice Index represents a measure of cumulative impacts on human health and well-being.

4.c.i.9 Tribal Data

Tribal populations and Tribal lands can be identified in EJScreen, ECHO, PFAS Analytic Tools (U.S. EPA, 2023) or with Housing and Urban Development's Tribal Directory Assessment Tool (TDAT): *https://egis.hud.gov/TDAT/*. For example, by clicking on a specific state, the user can identify Bureau of Indian Affairs Tribal boundaries and areas along with environmental media and other mapping layers. In addition, all lands of Federally Recognized Tribes, including Alaska Native Villages, are identified as disadvantaged in the CEJST.

4.c.ii Screening strategies to support waste management decision-making

There is no singularly definitive way to screen for the presence of and potential impacts on vulnerable populations. However, combining information on demographics and environmental and enforcement compliance could provide a useful indication of already overburdened or potentially vulnerable populations to inform decision-making. To avoid adding to disproportionate impacts, decisions regarding the destruction/disposal of PFAS-containing materials should be supported by an evaluation of communities located in the vicinity of the destruction, disposal, and storage. For example, less preferred facilities could be identified as those having a history of significant Clean Air Act violations, in an area with poor air quality and a high percentage of low-income people with significant air-related health disparities (e.g., asthma, lung cancer). This type of screening-level analysis can be used to identify communities with adverse and disproportionate impacts so as not to further exacerbate those impacts. It could also trigger questions for the facility regarding additional strategies to reduce the release of PFAS and other substances into the surrounding environment, including additional health protection, mitigation, and monitoring measures needed.

EPA has created several tools that can be helpful for screening, as have other federal and state agencies. Each tool has strengths and limitations. Options and suggestions for conducting a screening are described below.

EPA's EJScreen (see Section 4.c.i.1) collates social and demographic information that provides insights into non-chemical stressors to communities and subpopulations. EJ screen provides data from the U.S. Census and the American Community Survey[24], as well as information on race, income, unemployment rate, limited English proficiency, educational attainment, and age. Users can drill down to learn about additional demographics, including population, health, housing, and other points of interest. These data can be particularly useful to characterize community resilience and limitations. EJScreen also has

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[24] The American Community Survey is an ongoing survey that annually provides vital demographic information about the United States (*https://www.census.gov/programs-surveys/acs*). Data are updated annually as a 5-year average. EJScreen version 2.2 includes 2017-2021 ACS 5-year summary file data.

important information about existing pollution levels. It is most robust for air quality, although it does have information on releases to water via NPDES permits and other sources.

EJScreen also provides indicators on health disparities. Other sources on health include CDC's EJ Dashboard. States may include additional details on health burdens on their websites and tools.

Collectively, this information can provide a snapshot of communities with environmental justice concerns, allowing the user to flag communities that may be stressed due to multiple factors. Decision-makers, for example, could perform additional screening at destruction and disposal facilities with communities exceeding the 80th (or another appropriate) percentile in multiple indicators, and therefore more closely consider vulnerable populations while evaluating destruction, disposal, and storage facilities.

Regarding compliance, EPA's ECHO tool (see Section 4.c.i.1) provides enforcement and compliance data on specific candidate destruction and disposal facilities. A history of Clean Water Act violations, for example, could be an indicator of disproportionately exposed communities, particularly if they rely on public water systems using a surface water source downstream of the facility. In addition, communities using private wells for drinking water may be more vulnerable to releases regulated under RCRA or CERCLA.

4.c.iii Considerations for community engagement

In certain cases, community engagement is required under law. For example, facilities must hold public meetings before submitting part B RCRA permit applications (U.S. EPA, 2013a) and public hearings if they are requested for CAA Title V permits. In some cases, EPA's policy is to consult and coordinate with Tribes (U.S. EPA, 2013b). Community engagement is not merely a matter of meeting requirements, though. Advancing the goal of meaningful involvement is a core principle under EO 14096[25]. Reaching out to the community and providing notification, information, and an opportunity for input before deciding whether to accept PFAS-containing waste for destruction or disposal will help build trust and support for operations and can reduce the likelihood of negative reactions stemming from unresolved concerns.

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Meaningful community engagement typically includes two key elements:

- Public outreach to disseminate relevant and accessible information to the community, including in relevant local languages and in formats accessible to individuals with disabilities.

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[25] *https://www.federalregister.gov/documents/2023/04/26/2023-08955/revitalizing-our-nations-commitment-to-environmental-justice-for-all*

- Public participation, which generally entails a dialogue with the community to ascertain information and viewpoints. This dialogue is particularly important because the community can provide local knowledge of health and existing conditions, identify concerns and issues that may not be readily apparent outside the community, and offer contextual/cultural perceptions and experience (U.S. EPA, 2016).

Although presenting highly technical information is always a challenge, involving vulnerable populations in a meaningful way may present different challenges and opportunities from those in a general public involvement effort. To foster meaningful participation of all community members, it may be important to address issues that could hinder a community's participation in the decision-making process. These may include time and resource constraints, language barriers, and lack of trust (U.S. EPA, 2016).

Examples of effective practices to engage diverse and vulnerable populations include the following:

- Early notification so that community input can be offered in a timely way and incorporated into the decision-making process.
- Conveying issues in ways that are tailored (for example, translation, timing, location) to each specific population.
- Bridging cultural and economic differences that affect participation.
- Developing trust between the government and potentially affected populations.
- Working closely with state and local partners, as well as other federal agencies, to present a unified, consistent message to communities.
- Developing stakeholder capacity or providing technical assistance to effectively participate in future decision-making processes (U.S. EPA, 2015b).

EPA has developed tools to assist the federal government, states, and private entities with community engagement and outreach. For example, the *Superfund Community Involvement Toolkit* (CI Toolkit, available at *https://www.epa.gov/superfund/superfund-community-involvement-tools-and-resources*) provides practical information to design and enhance community involvement activities. While the CI Toolkit is designed for users to quickly review and adapt a variety of community involvement tools to engage the community during all stages of the Superfund processes, the same tools can be adapted to engage communities adjacent to destruction and disposal facilities.

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5. Research Needs and Data Gaps for Destruction and Disposal Technologies

EPA has a better understanding of potential PFAS releases from destruction and disposal activities, but more research is needed. This section identifies high-priority data gaps that researchers across government, academia, and industry need to address. This additional research will help EPA provide meaningful updates on effective destruction and disposal practices for PFAS-containing materials. This section outlines research needs to inform future guidance updates and provides an overview of existing research activities to address these gaps.

5.a Research needs to inform future guidance updates

EPA has translated the uncertainties and data gaps identified throughout this interim guidance into research needs, which are presented in Table 5-1. The research needs in Table 5-1 are focused on advancing our ability to measure PFAS in materials and releases from destruction and disposal technologies and improving our understanding of the performance of thermal treatment units, landfills, and underground injection control wells for managing PFAS and PFAS-containing materials. Individually, the research needs identified in Table 5-1 are complex to address because of the variety of substances within the PFAS class, the range of PFAS-containing materials discussed in this interim guidance, and the variability in technology designs and operating conditions. Collectively, some of the research needs identified in Table 5-1 are dependent upon other research needs. For example, fully characterizing PFAS releases from destruction and disposal technologies is dependent upon the availability of methods for measuring PFAS in air, water, and solids. Because of the complexity of the research needs and their inter-dependence, EPA anticipates that it will take years for the research needs in Table 5-1 to be fully addressed.

EPA has also prioritized the research needs in Table 5-1 as either *high*, *medium*, or *low* based on the potential for results to inform future versions of this interim guidance. In general, EPA designated the characterization of potential PFAS releases from thermal treatment units and landfills as high priority because these data are essential for determining whether communities surrounding destruction and disposal sites could be exposed to PFAS at unreasonable levels from these facilities. At the same time, EPA recognizes that current laboratory methods for identifying and quantifying PFAS are limited. As such, the agency also considers the development of sampling and analytical methods to be medium or high priority. The agency does not, however, believe that characterizing PFAS releases from PFAS destruction and disposal activities should wait until additional analytical methods are available. Rather, EPA expects that characterizing PFAS releases from destruction and disposal sites will be iterative and that future characterization efforts will be informed and guided by past efforts and the availability of new methods.

EPA designated research needs associated with the underground injection of PFAS and PFAS-containing materials via Class I wells as a low priority. The agency made this designation because the standards associated with the construction, operation, and monitoring of Class I wells are designed to isolate liquid wastes deep below the land surface and ensure protection of underground sources of drinking water. As a result, EPA expects underground injection of liquid PFAS waste to have a lower potential for environmental release, compared to managing PFAS wastes through thermal treatment or landfills. While data on the performance of Class I wells for managing liquid PFAS waste would be informative, the highest priority continues to be on characterizing PFAS releases from thermal treatment units and landfills.

Table 5-1. Prioritized Research Needs to Inform Future Guidance Updates

Topic	Research Needs	Priority
Thermal Treatment	Methods for sampling and analyzing PFAS in solid-, liquid-, and gas-phase waste streams from thermal treatment units[a]	
	• Organic fluorine (all phases)	Low
	• Targeted methods for non-polar, semivolatile PFAS (gas-phase)	High
	• Targeted methods for cationic, zwitterionic, and nonionic PFAS (solid- and liquid-phase)	High
	Full-scale performance testing—following the parameters and procedures suggested in Appendix A—at:	
	• Hazardous waste combustors	
	○ Commercial hazardous waste incinerators	High
	○ Hazardous-waste-burning cement kilns	High
	○ Hazardous-waste-burning lightweight aggregate kilns	Medium
	• Carbon regeneration and reactivation units	High
	• Non-hazardous waste combustors	
	○ Sewage sludge incinerators	High
	○ Municipal waste combustors	Medium
	○ Thermal oxidizers	High
	Characterization of PFAS releases from thermal treatment units operating under different conditions/designs	
	• Gas-phase emissions	High
	• Air pollution control device discharges	High
	• Bottom ash	High
	• Biosolids, remediated soil	High
Landfills	Methods for sampling and analyzing PFAS in liquid- and gas-phase waste streams from landfills[a]	
	• Organic fluorine (liquid- and gas-phase)	Medium
	• Targeted methods for non-polar, semivolatile PFAS (gas-phase)	Medium
	• Targeted methods for cationic, zwitterionic, and nonionic PFAS (liquid-phase)	Medium
	Approaches for solidification and stabilization of PFAS-containing materials[a]	
	• Effectiveness of differing solidification and stabilization techniques	High
	• Performance over time	High
	Fate and transport of PFAS—including partitioning between solid-, liquid-, and gas-phases in:[a]	
	• Hazardous waste landfills	High
	• Industrial waste landfills	High
	• Municipal solid waste landfills	Medium
	• Ash monofill landfills	Low
	• Construction and demolition (C&D) landfills	High
	Full-scale performance testing of engineered landfill components	
	• Bottom liner and leachate collection systems	Low
	• On-site leachate treatment systems	High

Topic	Research Needs	Priority
	- Gas collection and emission control systems	Low
	- Final cover systems	Low
	Characterization of PFAS releases from landfills operating in different regions and under different conditions/designs	
	- Gas-phase emissions	High
	- Groundwater contamination	High
	Efficacy and potential release of PFAS during leachate treatment	
	- Conventional technologies	High
	- Emerging technologies	Medium
Underground Injection (Class I)	Performance data from wells that are managing PFAS-containing liquids	
	- Surface management practices	Low
	- Effects on injection performance	Low
	Long-term fate and transport of PFAS in injection zones of Class I wells	
	- Mixture effects	Low
	- Interactions with co-contaminants	Low
	Data on the presence/absence of PFAS in groundwater near Class I wells receiving PFAS-containing liquids	Low
Emerging Technologies	Lab-, pilot-, and field-scale research to test the effectiveness of emerging technologies for different PFAS-containing materials and to characterize the outputs for PFAS and non-PFAS constituents.[b]	High

[a.] EPA is conducting research on this topic. See Section 5.b.ii for a high-level overview of EPA research and development activities.

[b.] See Section 6 for discussion of emerging technologies. EPA encourages technology developers to generate and publicly release data that can be used to answer the questions presented in the technology evaluation framework in Section 6.b.

5.b Current research efforts to address data gaps

Addressing the research and data gaps described in Section 5.a will require effort across government, academic, and private institutions. This section provides an overview of resources to learn more about prospective research activities.

5.b.i Governmental, academic, and industry research activities

In late 2021, the Office of Science and Technology Policy (OSTP) formed the interagency PFAS Strategy Team as part of the National Science and Technology Council (NSTC) to coordinate federal research on PFAS. In early 2023, the PFAS Strategy Team published the *Per- and Polyfluoroalkyl Substances (PFAS) Report*. The PFAS Report provides a high-level overview of research on PFAS as a chemical class by addressing the following strategic areas: removal and destruction; safer alternatives; sources and pathways of exposure; and toxicity. This document is a state-of-the-science report that includes research activities, gaps, and opportunities for the federal government. Following release of the PFAS Report, the PFAS Strategy Team initiated efforts to develop a federal strategic plan to address data gaps identified in the PFAS Report. Additional information is available in the *CEQ Report on Biden-Harris Administration Progress on PFAS* (March 2023).[26]

While this interim guidance only outlines major DoD and EPA research programs, a more complete overview of existing federal PFAS research activities is available in Appendix A of the OSTP PFAS Report. DoD is the largest federal funder of PFAS destruction and disposal research (see Appendix B, OSTP 2023). Within DoD, the Strategic Environmental Research and Development Program (SERDP) and the Environmental Security Technology Certification Program (ESTCP) have been funding significant research on PFAS for several years. The goals of this research include:

- Improving PFAS analytical methods and AFFF site characterization
- Understanding PFAS ecotoxicological effects
- Developing tools for assessing the fate of PFAS in the subsurface
- Developing and validating *in situ* and *ex situ* PFAS treatment technologies.

In 2022, SERDP-ESTCP published its *Summary Report: Strategic Workshop on Management of PFAS in the Environment* (Leeson et al., 2022). This report identifies research and demonstration needs for PFAS fate and transport, sampling and analysis, thermal treatment, non-thermal destructive treatments, and concentration technologies. To address these research needs, SERDP and ESTCP fund a robust research program, with annual solicitations for proposals. Descriptions of all PFAS-related statements of need and funded projects are available at *https://www.serdp-estcp.org/Featured-Initiatives/Per-and-Polyfluoroalkyl-Substances-PFASs*. Additionally, the Defense Innovation Unit has begun funding opportunities for innovation in technology for on-site PFAS destruction and remediation.

Colleges and universities, who are the recipients of federal and other research funding, conduct cutting-edge research related to PFAS destruction and disposal. These institutions often bring together expertise from many disciplines. This collaboration enables the rapid development of innovative approaches for managing the constantly changing set of PFAS-containing materials.

Other entities have strong interests in expanding into the area of PFAS material management, disposal, and destruction. These entities include private industry, the operators of waste disposal operations and technologies, municipal water utilities, professional and trade associations, and relevant research foundations. These organizations are actively developing and marketing solutions to address PFAS material management, disposal, and destruction.

5.b.ii EPA research and development activities

As described above, EPA is one of many governmental, academic, and industry organizations engaged in research on PFAS destruction and disposal. EPA's ORD is conducting research to characterize PFAS-contaminated sites and sources, and to understand the fundamental mechanisms of PFAS destruction and disposal technologies. This section provides a high-level overview of EPA research activities to support future interim guidance updates.

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[26] See also White House Fact Sheet: Biden-Harris Administration Takes New Action to Protect Communities from PFAS Pollution (March 14, 2023): *https://www.whitehouse.gov/briefing-room/statements-releases/2023/03/14/fact-sheet-biden-harris-administration-takes-new-action-to-protect-communities-from-pfas-pollution/*.

5.b.ii.1 Characterizing PFAS-containing materials

EPA is developing laboratory-based aqueous leaching methods to characterize leaching behavior of organic constituents, including PFAS, from solid materials under a range of environmental conditions. These methods are based on existing SW-846 Leaching Environmental Assessment Framework (LEAF) methods for inorganic constituents of potential concern (U.S. EPA, 2023). The methods produce a source term (i.e., as a concentration or release rate). Screening-level or scenario-based assessments can use the source term to model transport in the subsurface environment (Garrabrants et al., 2021a and 2021b).

EPA is continuing to develop and validate robust analytical methods to detect and measure PFAS in aqueous, solid, and gas-phase samples. The data generated with these methods will improve the understanding of environmental fate and transport and help evaluate the effectiveness of various treatment and destruction technologies. The suite of PFAS analytical methods under development includes targeted methods validated for quantitative analysis of a wider range of target analytes and matrices; class-specific methods, such as adsorbable organic fluorine and the total oxidizable precursors assay, to screen for a range of PFAS precursors; and non-targeted analytical methods to identify novel PFAS.

5.b.ii.2 Thermal treatment

ORD continues to conduct laboratory- and pilot-scale research to describe the behavior and mechanisms of destruction of PFAS-containing materials subject to thermal treatment. ORD facilities include a pilot-scale incinerator (the Rainbow furnace) and rotary kiln. This research examines the operating conditions (time, temperature, and combustion parameters) necessary for adequate PFAS destruction using conventional thermal treatment. The goal is to develop a mechanistic understanding of PFAS behavior and help interpret results from full-scale field studies. This includes measuring the generation of PICs from different PFAS-containing materials.

ORD is also interested in opportunities to partner with thermal treatment facilities to conduct field-scale research. The goals of this research would be to determine if representative field conditions achieve adequate thermal destruction of PFAS as defined by conventional DRE; characterize potential PICs/PIDs; and evaluate alternate indicators of destruction performance. The details of this thermal treatment field testing request are outlined in Appendix A.

5.b.ii.3 Landfills

ORD is developing novel sampling and modeling approaches to identify PFAS and characterize fate and transport at contaminated sites and source areas, including landfills. ORD is also evaluating solidification and stabilization techniques for PFAS applications.

5.b.ii.4 Extramural research

EPA supports PFAS research and development efforts through several programs, including *research grants*, the *Small Business Innovation Research* (SBIR) program, *challenges and prizes*, and the *P3 Student Design Competition*. For example:

- In 2019, *EPA awarded grants to eight universities* to better understand the environmental risks posed by PFAS and identify practical approaches to manage their potential environmental impacts. Several of these projects focused on characterizing PFAS in landfills. Research completed under these projects contributed to EPA's improved understanding of the potential for PFAS releases from landfills.

- In 2021, EPA collaborated on the *Innovative Ways to Destroy PFAS* challenge. Partners included SERDP-ESTCP, the Environmental Council of States and the Environmental Research Institute of the States, the Michigan Department of Environment, Great Lakes and Energy, and the Colorado Department of Public Health and Environment. The goal of the challenge was to discover new technologies and approaches that have the potential to remove at least 99 percent of PFAS in unused AFFF without creating harmful byproducts.
- EPA's SBIR program has funded many small businesses developing technologies to test and treat PFAS. These companies are developing products to destroy PFAS in a range of media using technologies including plasma reactors, electrochemical processes, sonolysis, hydrothermal alkaline treatment, and mechanochemical destruction. More information on technologies funded by EPA's SBIR program is available at *https://www.epa.gov/sbir/test-and-treat-pfas-epa-sbir-technologies*.

Information on open solicitations can be found at *https://www.epa.gov/chemical-research/pfas-research-extramural-funding-opportunities*.

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6. Emerging Technologies for PFAS Destruction and Disposal

Because of their chemical and physical properties, PFAS can pose unique challenges for end-of-life materials management. As described in Section 3, the effectiveness of commonly used waste management technologies, such as thermal treatment and landfilling, for managing PFAS materials can vary. Additionally, current operating conditions and practices at some thermal treatment facilities and landfills may not be sufficient to limit releases of PFAS to the environment. Concerns about the effectiveness of widely-used technologies for managing PFAS materials call for innovative research and development efforts toward new technologies as well as testing to apply existing destruction technologies to PFAS materials at various scales.

Development of new technologies typically involves the following stages:

- Technology development often starts with exploring and understanding the basic scientific principles and processes that underly the technology. At this early stage, the primary focus is on gaining understanding of how the process works and whether it shows promise for a particular application (e.g., PFAS destruction).
- If the results of early experiments are promising, the next stage of development generally focuses on prototype development and testing under controlled conditions. This stage allows developers to better understand whether the technology is practical and effective at larger scales. It also provides developers with information on costs associated with manufacturing and operating units.
- In the final stage of development, the focus generally shifts to field-scale demonstration projects that allow developers to test the technology under a variety of real-world conditions and to develop better cost estimates for operation and maintenance. During this stage, developers may identify the best and most cost-effective applications for the technology. In the case of PFAS destruction and disposal, this may include identifying the types of PFAS materials best suited for the technology (e.g., high-concentration materials versus low-concentration materials).

At any stage of the technology development process, developers may determine that the technology is not effective or is not cost-effective and development can stop.

In addition to research to identify and develop novel technologies for PFAS destruction, efforts are underway to apply existing destruction technologies—other than thermal treatment—to PFAS materials. In these cases, the scientific principles and processes underlying a technology may already be understood, but there is little information on the performance of the technology or its applicability to PFAS materials. Evaluating the applicability of an existing technology to PFAS materials can occur at various scales:

- Laboratory- and pilot-scale studies can be used to evaluate the effectiveness of a technology under controlled conditions and to determine how operating conditions can affect PFAS destruction.
- Field studies can provide data from full-scale applications that reflect the variability seen in real-world operating conditions.

The purpose of this section is to: (1) provide an overview of EPA efforts to identify and test emerging technologies for PFAS destruction and (2) introduce the PFAS destruction and disposal technology evaluation framework, which provides a transparent, consistent approach for evaluating destruction and disposal technologies for PFAS materials.

6.a EPA's PFAS Innovative Treatment Team

EPA's ORD initiated the PFAS Innovative Treatment Team (PITT) in 2020 as a short-term dedicated, cross-ORD effort to identify, review, and conduct preliminary research on potential treatment technologies. Over six months, the multi-disciplinary PITT conducted state-of-the-science reviews of proposed methods to dispose and/or destroy PFAS in media and wastes. Several criteria were evaluated using available information, including efficacy, feasibility, performance, and costs of each technology, while also considering potentially hazardous byproducts. The PITT partnered with other agencies and states to conduct a competitive public challenge to identify solutions that were potentially overlooked and to encourage further development of potential technologies.

The PITT chose to focus on the effectiveness of four technologies: mechanochemical degradation, electrochemical oxidation, gasification and pyrolysis, and supercritical water oxidation. These four technologies showed theoretical promise of the sought-after criteria: PFAS destruction, production of few to no hazardous residuals or byproducts, commercial availability, and cost effectiveness for various contaminated matrices. References and discussion for each technology can be accessed in the four PITT Research Briefs at *https://www.epa.gov/chemical-research/pfas-innovative-treatment-team-pitt*. The identification of these technologies by the PITT should not be considered an endorsement or recommendation to use these technologies to destroy PFAS. Additional research is needed to determine the effectiveness of these technologies for different PFAS-containing materials and to characterize the outputs for PFAS and non-PFAS constituents. An overview of the results of the PITT's state-of-the-science review is available in *Developing Innovative Treatment Technologies for PFAS-Containing Wastes* (Berg et al., 2022).

After the state-of-the-science reviews, the PITT selectively encouraged development and research of the four chosen emerging technologies to better understand the potential for PFAS destruction, specifically AFFF, while avoiding creation or transfer of environmental hazards. The PITT conducted in-house research and partnered with industry and academia to collect data regarding the technologies and to encourage further development of the methods required to ascertain PFAS destruction efficacy. Although the effort originally was allotted six months in 2020, the work initiated by the PITT continues to produce journal articles, fact sheets, and ORD reports. Some of the results from this continued work are described below. When considering the timeline, including navigating the initial restrictions of the COVID-19 pandemic, note there were several hurdles to obtaining necessary data, such as limited access to industrial field sites, analytical methods development, and availability of standards.

6.a.i PITT-investigated emerging technologies

To further explore the potential for mechanochemical degradation of PFAS, ORD scientists partnered with colleagues in New Zealand to conduct a proof-of-concept case study using a benchtop ball mill, AFFF-spiked sand, and AFFF-contaminated soil (Gobindlal et al., 2023). Mechanochemical degradation is a treatment technology that uses a high-energy ball-milling device, with the option of co-milling reagents, to produce highly reactive conditions to degrade contaminants. Results from the AFFF-spiked sand showed a 99.99% destruction efficiency for the sum of all measured PFAS after 960 min, and results from the AFFF-contaminated soil indicate that measured PFAS were below the limit of detection after 1440 min. Because this study analyzed PFAS in the solids after different ball milling time intervals, from 15 to 1440 min, the authors were able to identify increases in some of the measured PFAS before concentrations decreased, indicating the transformation of PFAS during ball milling. An extractable organic fluorine method was also used to determine the reduction in extractable organic fluorine before and after ball milling. In the case of AFFF-spiked sand, the authors reported a 97.8% reduction in extractable organic fluorine. A 53.5% reduction in extractable organic fluorine was reported for the AFFF-contaminated soil. Air emissions were not characterized as part of this study, making it difficult to determine whether full mineralization of PFAS occurred, and more work is needed in this regard. As noted in the study, more work is also needed to identify primary operations variables to inform the scalability of mechanochemical destruction.

In 2020, ORD conducted a pilot study at a biosolids treatment facility using pyrolysis (Thoma et al., 2022). Pyrolysis is a process that decomposes materials at moderately elevated temperatures in an oxygen-free environment. During the pilot study, dried biosolids were fed into a pyrolysis reactor operating at approximately 610 °C (1100 °F). The dried biosolids and the resulting biochar were sampled and tested for 41 target PFAS. Twenty-one PFAS were detected in the input biosolids, and none of the target PFAS were detected in the biochar. The pyrolysis system's target PFAS removal efficiencies were estimated to range between >81.3% and >99.9%. The pilot study also included limited analysis of the air emission control system, with results indicating no transmission of the target PFAS to the air. It is important to note, however, that full characterization of PFAS using non-targeted methods was not performed and more work is needed to understand PFAS transformations and the destruction mechanism.

ORD also completed studies to test the effectiveness of supercritical water oxidation for treating AFFF. Supercritical water oxidation is a process that occurs when the temperature and pressure of water is above the critical point. Above the critical point, oxidation processes are accelerated. In 2020, ORD partnered with three companies to perform tests of their supercritical water oxidation systems on dilute, PFOS-based AFFF (Krause et al., 2022). During these demonstration tests, each system was operated under slightly different conditions, and each company analyzed samples of the influent and effluent for slightly different PFAS. The overall destruction efficiency of each system was calculated by summing the concentrations of the measured PFAS. In all cases, the overall destruction efficiency was found to be greater than 99%. It is important to note, however, that the sum of the PFAS measured in the effluent ranged from approximately 10 µg/L to approximately 100 µg/L, suggesting that further treatment of the effluent may be necessary. It is also important to note that non-targeted analysis was not conducted, and air emissions were not monitored, making it difficult to assess the fate, transport, and transformation of PFAS with the supercritical water oxidation systems.

In 2021, ORD also partnered with a company to conduct tests at industrial-scale supercritical water oxidation system (Sahle-Demessie et al., 2022). These tests used a dilute, FTS-based AFFF. The primary objective of the study was to determine the destruction efficacy of the PFAS in the input material. Analysis of the influent and effluent showed greater than 99.99% destruction of the total PFAS measured. The average sum of the targeted PFAS in the effluent was approximately 55 µg/L, but PFAS were also detected in the tap water used as quench water in the system, suggesting that the overall performance of the system could be improved if non-PFAS-contaminated water is used in the system. A secondary objective of the study was to characterize degradation products using non-targeted analysis and stack sampling for nonpolar volatile fluorochemicals. Non-targeted analysis of the effluent tentatively identified the shorter carbon-chain PFAS on the targeted analyte list, and the authors suggested a possible destruction mechanism that would be consistent with this result. Further work is needed, however, to determine the destruction mechanism. The air emissions sampling did not detect targeted PFAS in air emissions above background levels; further work with finalized air methods (e.g., OTM-50) is needed to confirm this observation.

At this time, EPA is neither recommending nor discouraging the use of any emerging technology for managing PFAS-containing materials, including the technologies studied by the PITT. While the PITT studies generally indicated potential for PFAS destruction, further work using newly available methods is needed to more fully characterize the outputs of these processes and to evaluate their performance for PFAS-containing materials beyond AFFF.

6.b PFAS Destruction and Disposal Technology Evaluation Framework

Researchers have proposed metrics to evaluate PFAS destruction and disposal technologies, including technology readiness levels (TRLs), DREs of target compounds, fluorine mass balances, and others (Berg et al., 2022; Deeb et al., 2021; Krause et al., 2021). When used individually, these metrics fail to accurately capture comprehensive system performance or provide a holistic understanding of the potential for PFAS transformation and/or release to the environment. For example, DREs calculated from targeted analyses using existing methods can result in underreporting of the total PFAS in a matrix. That's because existing detection methods can only measure a fraction (e.g., 50 individual PFAS) of the total number of PFAS that may be present in a sample. Similarly, TRLs can fail to accurately reflect the current development stage of a technology by either reporting the overall maturity of the technology, instead of its specific PFAS application, or by relying on a niche, small-scale application to justify an overall readiness level.

These limitations drive the need for a comprehensive framework for evaluating a technology and its application to PFAS-containing materials. Such a framework should allow the user to prioritize the collection and consideration of different pieces of information. To meet these needs, EPA developed a technology evaluation framework that uses a multiple-lines-of-evidence approach to evaluate a technology/PFAS material combination (Table 6-1). The framework provides a transparent approach for evaluating and selecting a technology to reduce PFAS releases to the environment. EPA recommends that managers of PFAS-containing materials use the framework to evaluate emerging technologies, and to inform decisions about destruction and disposal of PFAS-containing materials.[27] EPA encourages technology developers to generate and publicly release data that can be used to answer the questions presented in the framework.

The framework comprises several topical sections: technology, material, analytical methods, disposal/destruction efficacy, community considerations, and regulatory requirements. Within each section, EPA has identified key pieces of information that, when considered together, provide a more holistic evaluation of a technology and its applicability for PFAS disposal or destruction.

The framework is intended to be an adaptable tool that allows the user to prioritize the collection and consideration of different pieces of information via the "priority" column in the framework. This allows users to address questions most relevant to their priorities and with respect to their decision context

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[27] The Interstate Technology and Regulatory Council (ITRC) PFAS Team developed the *PFAS Technical and Regulatory Guidance Document* (available at *https://pfas-1.itrcweb.org/*) to support state and federal environmental staff, as well as others, to gain a working knowledge of the current state of PFAS science and practice. Section 12 of the guidance document provides information on PFAS treatment technologies, including destruction technologies. Managers of PFAS-containing materials may find this resource useful when completing Table 6-1.

and the shared priority of preventing and mitigating potential exposures. Users may also choose to prioritize addressing certain questions before others to streamline their evaluation of a technology/material combination. For example, if a user of the framework is evaluating the performance of a destruction technology, the questions in the disposal efficacy section of the framework are not needed and could be removed from the framework. As another example, a user may prioritize answering questions under the technology section of the framework because those questions may rule out a technology for a particular application (e.g., the technology may not be available at the scale needed to process the PFAS-containing material of interest). Once a technology is ruled out, lower priority questions do not need to be answered. While users may not answer each question in the framework, EPA encourages users of the framework to address as many questions as possible to inform decisions about the destruction and disposal of PFAS materials.

Table 6-1 shows this approach, followed by more detailed considerations for each question.

Table 6-1. Framework for Evaluating Destruction/Disposal Technologies for PFAS-Containing Materials

Category of Information	Key Questions	Priority[a]	Answer and Associated Notes for PFAS Material of Interest
Technology	If available, what is the TRL for the technology/material combination of interest?		
	Is the technology of interest available at the scale needed for the intended application?		
	Is a treatment train needed to meet the desired destruction/disposal outcome?		
Material	What PFAS-containing material has been used to test the performance of the technology of interest?		
	Which PFAS, and at what concentration ranges, have been used to test the performance of the technology of interest?		
	Are there non-PFAS constituents that may affect the performance of the technology of interest?		
Analytical methods	Which targeted analytical methods (e.g., OTM-45, draft EPA Method 1633) have been used to characterize the performance of the technology for the PFAS-containing material of interest?		
	Has non-targeted analysis been performed on waste streams from the technology/material combination of interest?		
	What other analytical approaches have been used to characterize the performance of the technology for the PFAS-containing material of interest?		
	Are there any concerns about the quality of the data generated during testing?		
Disposal efficacy	Have potential transformation and partitioning pathways been identified for the technology/material combination of interest?		
	Are engineering controls in place to prevent or minimize release of PFAS to the environment?		

Category of Information	Key Questions	Priority[a]	Answer and Associated Notes for PFAS Material of Interest
	What is the longevity of the disposal technology?		
	Has a fluorine mass balance been determined?		
	Does the disposal of the PFAS-containing material contribute to the potential release of non-PFAS constituents to the environment?		
Destruction efficacy	Is the mechanism of destruction understood?		
	Have all process inputs and outputs been characterized for PFAS to the extent possible given current analytical methods?		
	What is the reported destruction efficiency?		
	Have any PIDs (or PICs) been identified?		
	Has a fluorine mass balance been determined?		
	Have all process outputs been characterized for non-PFAS constituents to the extent possible given current analytical methods and understanding of the composition of the PFAS-containing material?		
Community Considerations	Has a destruction/disposal site been identified?		
	What are the characteristics of the surrounding community?		
	Are there potentially vulnerable populations and/or communities with environmental justice concerns near the destruction/disposal site?		
	Are PFAS releases anticipated from the technology/material combination of interest?		
	Has the surrounding community been engaged?		
Regulatory requirements	Are there state or federal regulations that control emissions or releases from the technology of interest?		
	Are there state or federal regulations that control the management of the PFAS-containing material of interest?		

[a] Users can prioritize key questions using a variety of approaches. For example, questions could be labeled as *high, medium*, or *low* priority. Questions could also be prioritized using a numerical ranking system with *1* indicating questions to answer first, *2* indicating questions to answer second, and so forth.

6.b.i Key questions for technology evaluation

Technology. The first section of the framework focuses on the destruction or disposal technology. Key questions include:

- **If available, what is the TRL for the technology/material combination of interest?** TRLs are generally assigned using the *definitions developed by NASA* (NASA, 2023), where level 1 corresponds to observation and reporting of basic scientific principles and level 9 corresponds to “flight proven” technologies. TRLs may be determined through technology readiness assessments[28] or through other methods. If other methods are used, EPA recommends that users consider the source of reported TRLs and prioritize TRLs determined through independent technology readiness assessments over TRLs assigned by the technology developer.

- **Is the technology of interest available at the scale needed for the intended application?** Some technologies may only be available as mobile, containerized units, which may be suitable for high-concentration, low-volume materials. These units may not be appropriate for low-concentration, high-volume materials, where a stationary treatment unit with greater capacity is needed.
- **Is a treatment train needed to meet the desired destruction/disposal outcome?** If so, describe the treatment train needed and whether such technology is available and effective for the PFAS-containing material of interest. Note that treatment trains may produce materials that will also need to be managed.

Material. The second section of the framework focuses on the PFAS-containing material of interest. The six types of PFAS-containing materials discussed in this interim guidance vary in terms of phase, PFAS composition and concentration, physical and chemical properties, and overall composition (e.g., the presence of other contaminants). To understand technology performance across the range of variations within a material type, the following key questions should be addressed:

- **What PFAS-containing material has been used to test the performance of the technology of interest?** The description of the material(s) used to evaluate a technology/material combination of interest should be specific as possible (e.g., 3M Litewater versus AFFF), and links to bulk material characterization should be included, if available. If the material evaluated is artificially contaminated (i.e., spiked with known PFAS species and concentrations), that should be noted because it may not be an accurate reflection of the performance of the technology for the intended application.
- **Which PFAS, and at what concentration ranges, have been used to test the performance of the technology of interest?** To best inform decisions on technology use, the PFAS and concentrations used to test performance should match the PFAS and concentrations anticipated for the intended application.
- **Are there non-PFAS constituents that may affect the performance of the technology of interest?** This could include constituents in the original PFAS-containing material or constituents that may be created or introduced during a treatment train or by the technology of interest. For example, hydrofluoric gas may be produced during PFAS destruction, which may cause corrosion.

Analytical methods. PFAS are a large class of substances, and there are various approaches for identifying and measuring PFAS. EPA recommends using a combination of approaches to characterize the performance of the technology for the PFAS-containing material of interest, including characterizing PFAS in emissions or waste streams from the technology of interest. Key questions to be addressed under this section include:

- **Which targeted analytical methods (e.g., OTM-45, draft EPA Method 1633) have been used to characterize the performance of the technology for the PFAS-containing material of interest?** When possible, EPA recommends the use of standardized and/or validated methods for targeted analysis.

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[28] See the *Technology Readiness Assessment Guide* for best practices for evaluating technology maturity. The guide, developed by the U.S. Government Accountability Office and released in January 2020, provides organizations with best practices for conducting high-quality assessments.

- **Has non-targeted analysis been performed on waste streams from the technology/material combination of interest?** Non-targeted analysis is an important tool for characterizing PFAS in waste streams from a technology/material combination of interest because it can tentatively identify PFAS beyond those measured using targeted analytical methods.
- **What other analytical approaches have been used to characterize the performance of the technology for the PFAS-containing material of interest**? Other analytical approaches may include total oxidizable precursor assays and total organic fluorine assays for PFAS and analysis for non-PFAS substances (e.g., HF).
- **Are there any concerns about the quality of the data generated during testing?** For any quality assurance/quality control concerns identified, the user should determine whether it is appropriate to use the data to inform decisions about destruction and disposal of the PFAS-containing material using the technology of interest.

Disposal efficacy. Effective disposal of PFAS-containing material should prevent or minimize re-release of PFAS to the environment. Key questions for evaluating the efficacy of a disposal technology for a particular PFAS-containing material are identified below. (Questions included in this section should be skipped when evaluating a destruction technology.)

- **Have potential transformation and partitioning pathways been identified for the technology/material combination of interest?** The fate and transport of PFAS within a disposal technology can help inform whether PFAS may be present in emissions or releases from the technology. Note that the characteristics of the receiving matrix and environmental conditions can affect the fate and transport of PFAS within the disposal technology.
- **Are engineering controls in place to prevent or minimize release of PFAS to the environment?** If so, is there a monitoring system in place to evaluate the performance of the engineering controls?
- **What is the longevity of the disposal technology?** Because of the strength of the carbon–fluorine bond, PFAS are persistent. It is important to consider the lifespan of the disposal technology and whether the technology can contain PFAS for very long timeframes.
- **Has a fluorine mass balance been determined?** If so, is there fluorine that is unaccounted for? In some cases, a fluorine mass balance may be impractical.
- **Does the disposal of the PFAS-containing material contribute to the potential release of non-PFAS constituents to the environment?** PFAS-containing materials may contain other, non-PFAS constituents. Depending on the identity of those constituents and their potential to impact human health and the environment, it may be important to consider whether appropriate engineering controls are in place to prevent or minimize releases of non-PFAS constituents.

Destruction efficacy. Destruction of PFAS should result in breaking carbon–fluorine bonds without reforming PFAS and releasing PFAS to the environment. Key questions for evaluating the efficacy of a destruction technology for a particular PFAS-containing material are identified below. (Questions included in this section should be skipped when evaluating a disposal technology.)

- **Is the mechanism of destruction understood?** If so, has the destruction mechanism been experimentally confirmed, and have the reaction kinetics been determined? A thorough understanding of the destruction mechanism improves confidence in the performance of the technology of interest for PFAS.
- **Have all process inputs and outputs been characterized for PFAS to the extent possible given current analytical methods?** If not, what information may be missing?

- **What is the reported destruction efficiency?** Destruction efficiencies (or DREs) are generally reported for specific PFAS (e.g., PFOA or PFOS). As noted earlier, the destruction efficiency (or DRE) alone may not be sufficient to determine whether other PFAS may be released to the environment during the destruction process.
- **Have any PIDs (or PICs) been identified?** If so, which PFAS have been detected and at what levels? Are those PFAS regularly detected for this technology/material combination?
- **Has a fluorine mass balance been determined?** If so, is there fluorine that is unaccounted for? In some cases, a fluorine mass balance may be impractical.
- **Have all process outputs been characterized for non-PFAS constituents to the extent possible given current analytical methods and understanding of the composition of the PFAS-containing material?** The destruction of PFAS-containing materials is expected to result in non-PFAS products. Depending on the identify of those products and their potential to impact human health and the environment, it may be important to consider whether appropriate engineering controls are in place to prevent or minimize releases of non-PFAS constituents. It may also be important to consider the effect of the destruction process on non-PFAS constituents in the PFAS-containing material of interest. Thus, the characterization of process outputs beyond PFAS may be warranted.

<u>Community considerations</u>. When considering destruction and disposal technologies for PFAS-containing materials, it is important to consider the community that may be exposed to potential PFAS releases from intended destruction or disposal. Key questions to consider include:

- **Has a destruction/disposal site been identified?** Some destruction technologies may be mobile, allowing for treatment and destruction to occur where PFAS-containing materials are generated or stored. Other technologies may be stationary, requiring PFAS-containing materials to be brought on site for destruction or disposal. The use of mobile technologies may allow flexibility in siting to avoid potential impacts on overburdened and vulnerable communities.
- **What are the characteristics of the surrounding community?** See Section 4 for tools and approaches that can be used to characterize communities near destruction/disposal sites.
- **Are there potentially vulnerable populations and/or communities with environmental justice concerns near the destruction/disposal site?** Section 4 provides tools and strategies that can be used to screen nearby communities for potentially vulnerable populations and/or communities with environmental justice concerns.
- **Are PFAS releases anticipated from the technology/material combination of interest?** If so, describe which PFAS may be released, at what levels, and over what duration. Also note whether there are controls in place to mitigate releases to avoid or minimize potential impacts to surrounding communities.
- **Has the surrounding community been engaged**? Sharing information about planned destruction or disposal activities for PFAS-containing materials allows community members to take actions to address concerns they may have. There may also be opportunities to collaborate with government officials and community leaders on engagement activities.

<u>Regulatory requirements</u>. Regulations regarding the destruction or disposal of PFAS-containing materials can vary from state to state and may change over time. Key questions to consider include:

- **Are there state or federal regulations that control emissions or releases from the technology of interest?** Such regulations may limit the amount of PFAS or other pollutants present in emissions or other releases from the technology of interest.

- **Are there state or federal regulations that control the management of the PFAS-containing material of interest?** Such regulations may specify or restrict destruction or disposal options for the PFAS-containing material of interest.

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Appendix A: EPA Guidance to Conduct PFAS Emissions Field Testing at Commercial Thermal Destruction Sources

The U.S. Environmental Protection Agency (EPA) Office of Research and Development (ORD) is conducting per- and polyfluoroalkyl substances (PFAS) thermal destruction emissions characterization research to improve our understanding of the behavior of PFAS with respect to waste incineration and thermal treatment (e.g., hazardous waste, municipal waste, biosolids, contaminated soils, spent media), as well as thermal destruction emissions control technologies (e.g., afterburners, thermal and catalytic oxidizers). The following could be used by any entity evaluating thermal treatment of PFAS-containing materials either in collaboration with EPA or independently.

Overall goals of the field tests are as follows:

- Confirm/determine if thermal treatment under representative conditions can result in the thermal destruction of PFAS as defined by conventional destruction and removal efficiency (DRE) testing.
- Characterize potential products of incomplete combustion/destruction (PICs/PIDs) and evaluate the use of alternative, or surrogate, indicators of destruction performance.
- Confirm that any treated material is appropriate for any defined beneficial reuse (e.g., treated soils, reactivated granular activated carbon [GAC]).

To achieve these goals, it is necessary to determine the specific conditions (i.e., temperature, residence time, and turbulence) required to maximize DREs, minimize PICs/PIDs, and produce products appropriate for beneficial reuse. Because bench- and pilot-scale testing have limitations, actual test data under full-scale thermal treatment conditions are necessary to determine if representative thermal processes can achieve these goals.

Sampling Details:

- EPA is available to collaborate with the host site on the technical approach for source characterization. Testing would require multiple stack sampling methodologies performed by commercial stack testing companies, including the host's regular tester, if desired.
- All gas, liquid, and solid inlet, intermediate, and outlet streams will be collected and analyzed. This includes:
    - At the stack and including all air pollution control devices
    - All inlet streams (e.g., materials to be thermally treated, air or gas inlets, water), including samples taken prior to testing
    - All residual streams (e.g., bottom ash, baghouse and/or electrostatic precipitator [ESP] solids, scrubber effluent, spent carbon)
    - All final products (e.g., reactivated carbon, soils), if applicable
- A comprehensive air emission source characterization would include:
    - Samples would be collected for semivolatile PFAS, using method OTM-45 targeted (known) PFAS, or method established endpoints. Samples would be collected for volatile PFAS, using

method OTM-50 and analyzed for targeted (known) PFAS in samples and the identification of unknown detected volatile fluorocarbons.

- Samples would be collected for other semivolatile target compounds using methods SW-846 Methods 0010/3542/8270 with the inclusion of the Method 8270 procedure. Carbon hexafluoride (C_2F_6) and carbon tetrafluoride (CF_4) could be injected during one day of testing, ideally at multiple injection locations, as a surrogate measure of destruction efficiency if appropriate and permitted.
- Tests would be run under a single, representative process condition including the waste that would be co-fired with PFAS-containing material.
- EPA would prioritize testing several PFAS-containing materials, including but not limited to:
  - Incineration of aqueous film forming foam (AFFF)
  - Incineration of wastewater sludges
  - Reactivation of spent GAC
  - Incineration of municipal solid waste
  - Thermal treatment of contaminated soils
  - Incineration of spent ion exchange resins and their concentrated regeneration liquids
- All sampling would be conducted under a well-documented sampling plan and quality assurance project plan.
- Final testing results would be shared with EPA, the state, local authorities.

Additional details on emission testing approaches can be provided upon request.

Supplemental Information

Emissions sampling and analytical approaches

Emissions samples would be collected for polar, nonpolar, volatile, and semivolatile PFAS compounds and analyzed for both targeted PFAS compounds, or method established endpoints.

EPA's Other Test Method (OTM) 45 would be used to collect polar semivolatile PFAS compounds for targeted analyses. The method's 50+ target analytes include many PFAS commonly found in AFFF. This would enable DE, DRE, and emission rate determination as appropriate for a known list of PFAS compounds.

The OTM-50 canister sampling method would be used to collect nonpolar volatile PFAS compounds for targeted compound analyses. The current target list includes CF_4 and C_2F_6 as well as a procedure to identify unknown volatile fluorocarbons.

Non-targeted analyses (NTA) are an optional analytical tool for characterizing potential PICs/PIDs. High resolution mass spectrometry, combined with chromatographic separation and multiple ionization techniques, is used to identify compounds present in collected samples. NTA could be performed on the OTM-45 samples. Fourier transform infrared (FTIR) spectroscopy would be used to measure a variety of compounds in real-time, including hydrogen fluoride (HF), hydrogen chloride (HCl), carbon dioxide (CO_2), and water (H_2O); multiple PFAS compounds including CF_4 and C_2F_6; and multiple known PICs/PIDs. FTIR is particularly critical during surrogate compound injection.

ORD research questions

The following section describes detailed elements of the research questions ORD will ultimately work to address, as well as the input information that is needed. ORD requests an opportunity to engage with the host facility prior to developing a full sampling plan to better understand site-specific conditions and requirements.

Process characterization

- **Question:** How does the process operation affect the destruction and removal of PFAS compounds present in the feed?
- Requires understanding of waste feed (e.g., description, concentration, other wastes), feed approach (e.g., liquid feed, through flame/post flame, co-fired, drums/barrels), and feed rate
- Requires data on combustor operations and conditions (e.g., temperature, residence time)
- Requires data on air pollution control devices (APCD) and their operation: configuration, operating parameters (e.g., temperature, pressure drop, pH of scrubbing liquid, gas velocity)

Determine destruction removal efficiency (DRE)

- **Question:** What is the DRE and emission rate of the specific PFAS compounds present in the feed
- material?
- Requires known input rate and identity of PFAS compounds in waste
- Requires evaluation of the extent of PFAS mineralization
- Requires replicate sampling of polar semivolatile organics using OTM-45 for quantitative measurement of feed PFAS compounds in emissions
- Requires sampling across multiple APCDs

Products of incomplete combustion/destruction (PICs/PIDs)

- **Question:** Are PICs/PIDs formed during thermal treatment and measured by OTM-50 present and, if so, what are they and under what conditions?
- Requires replicate sampling of polar, nonpolar, semivolatile, and volatile organics for non-targeted compound analysis using OTM-45, OTM-50, and Method 0010/3542/8270.
- Requires sampling across multiple APCDs.

Alternative indicators of destruction performance

Question: Can easily measured/monitored, hard-to-destroy compounds serve as alternative (surrogate) indicators for PFAS adequate destruction or PID formation conditions?

- Requires injection of known concentrations/masses of CF_4 and C_2F_6 (approximately 10 parts per million by volume stack equivalent) at representative waste injection locations
- Requires on-line FTIR monitoring of the surrogate compounds, PICs, HF, HCl and other compounds of interest
- Requires initial background signal baseline
- Requires concurrent OTM-50 sampling for analysis of surrogate compounds and known PICs

Analysis of fluorinated compounds in waste/effluent streams

Question: What is the fate of PFAS compounds across the entire thermal system?

- Requires sampling of all discharges
    - Gas sampling prior to and after APCDs (e.g., spray dryer, wet scrubber, baghouse and/or ESP, activated carbon injection)
    - Sampling of solid and liquid discharges (e.g., bottom ash, baghouse and/or ESP solids, scrubber effluent analyses, spent carbon)

Analysis of fluorinated compounds in final product materials, where applicable

Question: What PFAS remain in the material treated, such as reactivated GAC or treated soils?

- Requires sampling of influent and effluent materials
    - Sampling of solid materials (e.g., spent carbon and reactivated carbon)

Appendix B: Summary of the Clean Harbors Test Data

Summary

Two testing campaigns were conducted at Clean Harbors Environmental Services' (CHES) hazardous waste combustor (HWC) located in Aragonite, Utah. Each test campaign was divided into three test conditions: Baseline (Condition 1), Augmented Per- and Polyfluorinated Alkyl Substances (PFAS) Feed (Condition 2, where known amounts of four or five target PFAS were fed to the incinerator), and Post-spiking Normal Operations (Condition 3, where aqueous film-forming foam [AFFF] was included in the incinerator feed). Clean Harbors used the *Other Test Method 45 (OTM-45) Measurement of Selected Per- and Polyfluorinated Alkyl Substances from Stationary Sources* (OTM-45) from the U.S. Environmental Protection Agency (EPA) for stack gas samples. The 49 targeted PFAS analytes were measured using liquid chromatography/tandem mass spectrometry (LC/MS/MS) with isotope dilution. The tests appear to demonstrate that the primary objective, "demonstrate the ability of the high-temperature incineration system at Aragonite to effectively destroy PFAS in the waste feed during normal process operating conditions," was achieved for the five selected PFAS compounds that were subjects of these studies.

Introduction

Two testing campaigns were conducted at CHES HWC located in Aragonite, Utah. The first campaign was conducted from June 17 to 19, 2021 and the second from February 17 to 19, 2022. The stated principal objective of the test program was to "demonstrate the ability of the high-temperature incineration system at Aragonite to effectively destroy PFAS in the waste feed during normal process operating conditions," with secondary objectives of conducting a mass balance of PFAS in process streams and demonstrating high hydrogen fluoride (HF) removal that would be associated with a high destruction of PFAS. This summary presents a high-level overview of both tests' final reports as well as a third-party review of the 2021 test performed by Dr. Philip H. Taylor under contract to CHES.

General Process Description

The HWC tested at the CHES Aragonite facility consists of a slagging rotary kiln incinerator followed by an afterburner. Solid waste materials and containerized wastes are fed into the rotary kiln only, while liquid wastes may be injected directly into rotary kiln or the afterburner. The air pollution control device system for the HWC as tested[29] consists of a spray dryer/gas quench followed by activated carbon injection upstream of a fabric filter (particulates, dioxin/furan, and mercury control). Following the fabric filter, the gases enter a sodium carbonate saturator/wet scrubber (sulfur dioxide, hydrogen chloride, hydrogen fluoride control) with TMT-15 (1,3,5-triazine2,4,6(1H,3H, 5H)-trithione, trisodium salt, used for additional mercury and metals control) injection.

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[29] Stack testing was performed in the exit duct of the wet scrubber instead of at the stack for safety reasons (lack of scaffolding). The HWC is equipped with a wet electrostatic precipitator (WESP) following the wet scrubber and prior to the stack, but the WESP was decommissioned in 2004 and is no longer in service. These tests were conducted prior to the WESP although the stack location is the test location for compliance performance tests for the facility.

Operational Description

Each test campaign was divided into three test conditions: Baseline (Condition 1), Augmented PFAS Feed (Condition 2), and Post-Spiking Normal Operations (Condition 3), with the waste feed matrices as follows:

1. **Condition 1.** Normal matrix of liquid materials (waste fuel, clean fuels, sludge, and aqueous wastes) and solid waste materials (containerized, shredded, and bulk solid wastes). Following the 2021 testing, CHES determined that the sludge stream fed to the kiln during all three conditions included some AFFF. During the first run of the 2022 testing, CHES determined that AFFF was inadvertently being fed into the afterburner's corrosive waste port. The testing team decided to be consistent with runs 2 and 3 and continued the AFFF feed, with additional amounts of AFFF fed during Condition 3.
2. **Condition 2.** Normal matrix of liquid materials (waste fuel, clean fuels, sludge, and aqueous wastes) and solid waste materials (containerized, shredded, and bulk solid wastes), with the addition of augmented feed rates of four (2021) or five (2022) PFAS (or their equivalent potassium salt). The four PFAS are perfluorooctanioic acid (PFOA), perfluorooctanesulfonic acid (PFOS), perfluorohexane sulfonic acid (PFHxS), hexafluoropropylene oxide-dimer acid (HFPO-DA), and perfluorobutanesulfonate (PFBS, 2022 only). The target feed rate of the PFAS was estimated to be .0816 pounds per hour or around 0.62 grams per minute in 2021 for all PFAS, but the HFPO-DA was increased to 2.2 grams per minute in the 2022 testing to reflect the method detection limit (MDL) for this compound. Similarly, the PFBS feed rate was 0.33 grams per minute on average to reflect the MDL. The PFAS were fed in boxed containers to the rotary kiln via the containerized solid conveyor in approximately 10-gram portions of all PFAS compounds about every 15 minutes for the duration of these test runs.
3. **Condition 3.** Normal matrix of liquid materials (waste fuel, clean fuels, sludge, and aqueous wastes) and solid waste materials (containerized, shredded, and bulk solid wastes). For this test condition, AFFF concentrate was also fed (neat) to the incinerator. In 2021, the AFFF was fed via the containerized solids in 5-gallon buckets at a rate of 176 pounds per hour. During the 2022 test, the AFFF was labeled as Ansulite and was fed through the educt station to the incinerator directly from a tote bin. The AFFF feed rate ranged from 244 pounds per hour to 316 pounds per hour during the 2022 test.

The rotary kiln was operated in a temperature range between 1,893°F and 2,008°F for the 2021 test and between 1,949°F and 2,219°F for the 2022 test. The afterburner was operated at 2,052°F to 2,110°F during the 2021 test and 2,048°F to 2,075°F for the 2022 test. Oxygen content (free) at the exit of the rotary kiln was maintained between 8 and 9 percent and ranged from 6 to 7 percent for the afterburner for both test campaigns. Clean Harbors estimates that non-combustible material is retained within the combustor for approximately 60 minutes, while combustion gas residence time of the afterburner is two to three seconds.

For the 2021 test, the temperature of the flue gas was reported as 369°F following the spray dryer; between 341°F and 343°F following the fabric filter; between 166°F and 168°F at the saturator outlet; and 148°F to 153°F at the stack. For the 2022 test, the temperature of the flue gas was reported as

363°F following the spray dryer; between 168°F and 170°F at the saturator outlet; and 149°F to 151°F at the stack. The following table lists the air pollution control device operational parameters during the testing campaigns.

Test Campaign	Stack gas flow (acfm)	Soda Ash Flow (lb/hr)	Carbon Injection (lb/hr)	TMT-15 flow (lb/hr)
2021	66,793–68,761	452–789	30.7–31.5	3.95–4.48
2022	51,053–53,640	411–1594	32.0–33.1	3.64–4.00

Test Methods and Scope

During both testing campaigns, samples of stack gases and process streams were taken and analyzed for 49 PFAS target compounds, with the stated analytical objective of characterizing as many streams entering or exiting the incinerator as possible. Clean Harbors used EPA's *Other Test Method 45 (OTM-45) Measurement of Selected Per- and Polyfluorinated Alkyl Substances from Stationary Sources* (OTM-45) for stack gas samples (as noted earlier, "stack" meaning the exit duct from the wet scrubber as it was deemed a safer sampling location than the actual stack). The 49 targeted PFAS analytes were measured using LC/MS/MS with isotope dilution.

Process streams sampled included waste fuels, clean fuels, waste feeds, process water, and residuals (liquids and solids), as well as some pre-test sampling of chemical inputs (soda ash, activated carbon, and TMT-15). As noted in the reports, however, due to the waste profile matrix complexity and the heterogeneity of all three solid waste streams, they were not sampled or analyzed for PFAS. Absent any final EPA-approved methods for media other than drinking water, LC/MS/MS analysis was used to measure the targeted PFAS compounds in the process stream samples.

In addition to the 49 PFAS analytes, the HF concentration of the stack gas was measured using EPA Method 26A. The AFFF-containing material in the 2021 test was analyzed for the 49 target PFAS both before and after chemical oxidation as a total oxidizable precursor assay (TOPA). During the 2022 test, a TOPA and total organic fluorine assay (TOFA) were performed on the AFFF-containing materials. The objective of performing TOPA is to "reveal perfluoroalkyl acid (PFAA), fluorotelomer sulfonate (FTS), and possibly other precursors that are not identifiable with targeted PFAS analysis." TOFA, as described in the 2022 report, identifies the mass of compounds containing carbon-fluorine bonds, and was performed using combustion ion chromatography following a laboratory in-house method for adsorbable organo-fluorine.

Results

According to the tests and third-party review, the primary objective of determining the destruction and removal efficiency (DRE) of each of the spiked (augmented feed) PFAS was achieved. The calculated DREs for each test (triplicate test run average) and PFAS compound are as follows:

PFAS Analyte	2021 Test DRE (%)	2022 Test DRE (%)
PFOA (perfluorooctanioic acid)	99.999943	99.99990
PFOS (perfluorooctanesulfonic acid)	99.999955	99.99977
PFHxS (perfluorohexane sulfonic acid)	99.999977	99.99996

PFAS Analyte	2021 Test DRE (%)	2022 Test DRE (%)
HFPO-DA (hexafluoropropylene oxide – dimer acid)	99.999979	99.99984
PFBS (perfluorobutanesulfonate)	n/a	99.99997

Both test reports note that none of the other 45 (or 44, in 2022) target PFAS analytes were present in the waste feed at levels sufficient to demonstrate 99.9999 percent DRE without spiking, even though stack gas concentrations were non-detect in many cases. According to the 2021 test report, there were three other PFAS present in the waste feeds at sufficient mass feed rate without spiking to enable demonstration of DRE's exceeding 99.999 percent. These included PFOS, 6:2 fluorotelomer sulfonate (FTS), and 8:2 FTS. The report stated that the FTS compounds are common constituents of AFFF.

As for the secondary objectives, the results are, as stated within Dr. Taylor's review, "most difficult to achieve as they involve more complex sampling issues." With respect to the secondary PFAS mass balance objective, he notes that "some of the PFAS that was not sampled in the infeed may have been more difficult to gasify and burn, contrary to the assertions in the report." However, he concurs with the report statements that the PFAS DRE is conservative.

The objective of demonstrating high HF removal that would be associated with a high destruction of PFAS is seemingly not achieved, primarily due to analytical issues. Dr. Taylor notes that "the sinks for the HF were not accurately quantitated leading to some questions about the validity of the reported HF removal efficiency." He further notes that the assumption that all fluorine in the waste if converted to HF is challenged, as there would be a required excess of hydrogen to fluorine to be demonstrated, which are not provided for in the data. Further, he stated that there is a "demonstrated lack of accuracy of the analytical methods for measuring inorganic fluorine in the residual streams," which, in his opinion, leads to a very low fluorine recovery mass balance. He concluded that, "development of better analysis methods for complex matrices for inorganic fluorine are needed before high HF removal efficiencies can be reported at full scale."[30]

Conclusions

The tests appear to demonstrate that the primary objective, "demonstrate the ability of the high-temperature incineration system at Aragonite to effectively destroy PFAS in the waste feed during normal process operating conditions," was achieved for the five selected PFAS compounds that were subjects of these studies. However, this study did not evaluate the formation of PICs due to previous methodology limitations. Dr. Taylor, in his review, noted this as well: "The larger question from an environmental viewpoint is the complete mineralization of these substances and prevention of emission of highly stable C_1-C_2 PFCs (perfluorocarbons)." Dr. Taylor concludes that CHES' results, although promising and demonstrating reduction of a limited selection of specific PFAS compounds, did not support complete mineralization.

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[30] Although not stated by Dr. Taylor in his assessment, another possible reason for lack of finding inorganic fluorine (HF) could be the result of poor conversion (mineralization) of PFAS to HF and carbon dioxide during incineration. This may indicate formation of non-target PFAS or fluorocarbons during incineration that were not measured by these tests.

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Appendix C: Summary of the Chemours Thermal Oxidizer Test Data

Introduction

Chemours Company FC, LLC, located in Fayetteville, North Carolina, manufactures chemicals and plastics and was required to install a thermal oxidizer to control per- and polyfluoroalkyl substances (PFAS) emissions. After installation, tests were conducted in March 2020 and March 2022. The objective of the tests was to determine how "the Thermal Oxidizer and 4-stage Scrubber system will achieve the emission reduction [of 99.99 percent], including the use of a surrogate for all PFAS, such as the hexafluoropropylene oxide (HFPO)." This summary presents a high-level review of the two tests' reports.

General Process Description

The thermal oxidizer at the Chemours facility operates at 10 million BTU per hour and is powered by natural gas. Waste gases from both polymer and monomer production are fed into the thermal oxidizer. The gaseous emissions from the oxidizer are fed into the four-stage packed bed column scrubber system while solid calcium fluoride (CaF_2) is removed, collected, and disposed of off site. The scrubber has three water scrubbing stages and one caustic scrubbing stage and is stated to remove hydrogen fluoride (HF) from the final emissions.

Operational Description

Test Methods and Scope

The tests done on the thermal oxidizer focused on five PFAS compounds: HFPO, hexafluoropropylene oxide-dimer acid (HFPO-DA), hexafluoropropylene oxide-dimer acid fluoride (HFPO-DAF), carbonyl fluoride (COF_2), and Flouroether E-1. These compounds were selected because they are handled and used at the Fayetteville facility. The compounds have varying chemical structures and properties and cannot be sampled and tested in the same manner.

Modified Method 18 sampling was used to capture the PFAS compounds. This method uses six fluoropolymer (PFA) impingers charged with methanol arranged in a series and captures the compounds using chemical reactions and condensation.

To capture HFPO-A, the facility needed to use Modified Method 0010 sampling with a heated probe and filter, condenser coil, XAD-2 resin cartridge, deionized water impingers, and a silica gel impinger. In addition to the normal Modified Method 0010 sampling process, this test added a second XAD-2 resin cartridge to monitor possible target PFAS breakthrough.

Three compounds, HFPO, HFPO-DAF, and COF_2, were sampled by chemical reaction. The sample gas was passed through the series of methanol impingers. The compounds then react with the methanol, and each produced unique ether compounds that were monitored and quantified using SW-846 Method 8260.

The strategy used to sample Flouroether E-1 and HFPO-DA was to capture the compounds through condensation and dissolution in the same methanol impingers. The volatile organic compound, Flouroether E-1, was then quantified using SW-846 Method 8260b. The semivolatile organic compound HFPO-DA was quantified using EPA Method 537.

Samples were taken from the polymer production and monomer production waste gas feed lines to capture the initial amount of PFAS compounds they produced as waste and give them a baseline concentration of PFAS prior to treatment. The other sampling location was in the stack after the thermal oxidizer and scrubber. This location would show how effective the process was at destroying the targeted PFAS compounds.

The sampling and testing methods only captured the amount of target PFAS compounds and did not measure any products of incomplete combustion. Although the four-stage scrubber was implemented to reduce the possible HF emissions, there was no sampling of HF.

Results

The results of the tests showed that the mandated 99.99 percent destruction of targeted PFAS compounds had been achieved. In both testing years the average PFAS destruction and removal efficiency (DRE) was above 99.999 percent with no result being lower than 99.999 percent. Results were presented as a total FPAS destruction efficiency with the amount of each target PFAS compound summed to provide an aggregate destruction efficiency. The total PFAS destruction efficiencies for each run and each year are as follows:

Year	Run 1	Run 2	Run 3	Average
2020	99.99982%	99.99974%	99.99986%	99.99981%
2022	99.99951%	99.99966%	99.99967%	99.99961%

Conclusions

The test reports demonstrate a PFAS destruction efficiency above the 99.99 percent that was mandated for the Chemours facility for the specific compounds that were tested. HFPO was tested as a surrogate for all species of PFAS, but there are no concrete data showing the performance of the thermal oxidation process for PFAS species outside of the compounds tested. There was also no discussion of possible products of incomplete combustion in the reports. The thermal oxidizer at the Chemours facility appropriately destroys the five PFAS compounds of interest that have been identified at the facility. However, removal processes for products of incomplete combustion or of destruction of potential compounds not studied but potentially found in the facility waste streams are still unclear.

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Appendix D: Summary of Costs and Considerations

Introduction

This appendix provides some estimates of costs for end-of-life management of PFAS materials by thermal treatment, landfilling, and underground injection. The cost information presented is relatively uncertain and is likely most suitable to make comparisons between techniques.

The cost to destroy or dispose of PFAS and PFAS-containing materials can vary widely depending on a number of factors. Waste management infrastructure's design and operation are heavily influenced by regional needs, local regulations, and technological advancements. As a result, waste management costs vary by the type of waste and by location due to factors like availability and transportation costs. In addition, technological advances specifically related to the management of PFAS-containing materials can potentially reshape the cost and operational landscape of PFAS waste management. Multiple factors and limitations contribute to cost variability and should be considered when using the information in this appendix.

- Market conditions such as supply and demand change frequently, resulting in fluctuating costs.
- Location of destruction and disposal options and transportation affect case-by-case costs.
- Transportation costs or any special handling costs for accepting and treating additional waste are not included in most of these estimates, thus actual costs would vary.
- Costs may be affected by permitting and state/local requirements for PFAS-containing material, which vary depending on state/local regulations, public input, and possible environmental justice concerns.

Thermal Treatment

Hazardous Waste

Table D-1. Estimated Costs to Incinerate Different Types of Hazardous Waste (U.S. EPA, 2005a)

Hazardous Waste Type	Estimated Thermal Treatment Cost per Ton[a]
Liquids, sludges, solids (halogenated)	$1,300-1,889
Liquids, sludges, solids (non-halogenated)	$381-1,040
Lab packs	$6,448
Containerized gases	$3,120

[a] Costs were developed for hazardous waste combustors (HWC) that were affected by the 2005 National Emission Standards for Hazardous Waste Pollutants (NESHAP), including commercial incinerators, cement kilns, lightweight aggregate kilns, and commercial HWC boilers. Cost per ton is assumed to be in 2002 dollars, the same year basis of the regulation from which these costs were obtained. Costs were normalized to 2022 dollars using the Bureau of Economic Analysis (BEA) gross domestic product (GDP) deflator (U.S. BEA, 2023) for waste management and remediation services using a base year of 2002. Note that applying the BEA GDP deflator for this industry sector is a top-down approach. For regulatory purposes, a bottom-up approach is typically used, accounting for capital costs based on the Chemical Engineering Plant Cost Index (CEPCI), energy prices typically based on Energy Information Administration (EIA) data, and labor costs based on industry segment indices using Bureau of Labor Statistics (BLS) data.

Limitations:

- Costs have likely changed since 2002 and scaling costs to 2022 contributes to uncertainty.

- Does not include transportation costs or any special handling costs for accepting and treating additional waste materials.

Table D-2. Example Disposal/Reactivation Costs for Drinking Water Treatment Residuals (Derived from U.S. EPA, 2020a)

Method	Cost of Disposal ($/Pound of Media)	Estimated Cost of Disposal Plus Replacement Media ($/Pound of Media)[a]
Reactivated GAC—off-site	$0	$1.74
Disposal via landfill	$0.06	$2.35
Disposal via incineration	$1.28	$3.57

[a] Cost per pound is in 2022 dollars. For GAC, on-site reactivation is possible. However, the utility or site would have to have ample workforce, managerial, and financial (both capital and operating) resources to justify this choice. It is likely to be cost-effective only for very large facilities and would require consideration of other factors including availability of land and public opinion. Due to the complex analysis needed, a full comparison of off-site versus on-site is beyond the scope of this document.

Limitations:

- These estimates are presented as examples only to illustrate tradeoffs. Actual site-specific costs are likely to vary based on factors including the quantity of media reactivated/replaced and are used as an example of trends. In this example, reactivated GAC and virgin GAC are estimated to be $1.51 per pound and $2.29 per pound, respectively.
- All unit costs assume 21,000 lbs of GAC reactivated/replaced.
- For reactivated GAC, it is assumed that the “replacement” media is the returned reactivated media plus the replacement of the GAC lost during reactivation with virgin media (assumed to be 30 percent).
- Disposal costs assume 10 miles of transportation to the disposal site.
- Incineration costs are based on incineration of ion exchange resin media. Incineration of GAC could be somewhat more expensive, depending on the relative fuel value of the spent media.
- The analysis is complex and a number of issues need to be considered at the site level, such as those that affect costs (cost of energy, shipping, labor, construction, operation, sampling, etc.) and those that affect other matters (practicality, public versus private ownership, contract availability, regional reactivation availability, off-gas permitting, public opinion, etc.).

Landfills

Hazardous Waste

Table D-3. Estimated Costs to Landfill A Representative Type of PFAS-Containing Material (Defense Logistics Agency (DLA), 2023)

Hazardous Waste Type	Estimated Cost per Ton of Waste
Aqueous film-forming foam (AFFF) and other PFAS-related Solids/Debris, Non-RCRA, >100 parts per million (ppm) PFAS for direct disposal at RCRA Subtitle C Landfill (e.g., contract line item number (CLIN) PF06B1)	$1,300-3,660 ($0.65-1.83/lb)

Limitations:

- The DLA pricing web application lists current government contracts for destroying or disposing of PFAS and PFAS-containing materials and may be useful for government contract cost estimates. The estimated cost per ton (lb) of waste appears to include removal and disposal costs for the specific material, although a portion of transportation costs may be attributed to other cost elements of each respective contract so transportation costs may not be fully reflected in these costs.
    - **Per the DLA pricing web application:** The prices contained in the database were awarded based on contract terms and conditions, procurement requirements, and price reasonableness determinations. The prices and contract information in the database are for reference only.
- Users of the DLA pricing web application are cautioned that each contract was developed to meet specific requirements for specific geographic regions or military installations. There are multiple variables that result in the pricing information displayed, including contract award date, proximity to DLA-approved disposal facilities, quantities of PFAS material estimated for disposal in proportion to other hazardous waste on the contract, reoccurring need for services, and whether PFAS material is disposed of in bulk or whether there are small, limited quantities.
- Pricing is provided here for bulk waste types for disposal at a Subtitle C hazardous waste landfill. Pricing for multiple waste types with varying units of measure for disposal, including containerized waste, are provided in the DLA pricing web application.
- The estimated cost in the table above only applies to a limited number of military installations and only considered bulk pricing from 600 to 10,000 pounds. Pricing is expected to differ for containerized waste and should be considered when relying on this cost information.

Non-Hazardous Waste (Municipal solid waste)

Table D-4. Average MSW Tipping Fees per Ton by U.S. Region (EREF, 2022)

Region	Average Tipping Fee ($/ton)	Region	Average Tipping Fee ($/ton)
Pacific	$69.02	Northeast	$75.92
Mountains/Plains	$50.84	Southeast	$48.70
Midwest	$62.02	South Central	$50.84

Limitations:

- MSW landfills emit more PFAS than originally thought in 2020. Research estimates that although MSW landfills contain PFAS for the most part, up to 5 percent of the PFAS may be released in the landfill gas and, 11 percent may be released in the leachate annually (Tolaymat et al., 2023). Thus, MSW tipping fees may not be a relevant data point for cost information.

Non-Hazardous Waste (Liquid waste)

Table D-6. Average Industrial Wastewater Rates by U.S. Region (Unger et al, 2023)

Region	$/1,000 gal	Region	$/1,000 gal
Pacific	$11.90	Northeast	$4.82
Mountains/Plains	$3.12	Southeast	$7.52
Midwest	$4.77	South Central	$5.20

[a] Data are consolidated by region and the number of data points varies by region. American Water Works Association (AWWA) water and wastewater rate surveys were used to gather historical rate data for water and wastewater utilities in the United States. These datasets were compiled and assessed to produce a single dataset of time series rate data for 112 water utilities and 76 wastewater utilities located throughout the United States.

Limitations:

- These data do not include specific information on extra treatment considerations that may be required by a national pollutant discharge elimination system (NPDES) permit or by an industrial user permit for a discharge into a POTW to control the release of PFAS. Thus, similar to the cost of landfilling PFAS waste, the associated treatment of PFAS-laden leachate is difficult to assess because the available data are associated with typical industrial wastewater generators or typical landfill leachate.

One report examined the cost to manage landfill leachate on site (U.S. EPA, 2021). Based on review of the treatment processes and technologies that are currently used, the cost to manage leachate on site is highly variable but generally ranges between $10-150/1,000 gallons ($0.01-0.15/gallon). Capital costs can range from hundreds of thousands to several million dollars. The report lists multiple limitations and does not specify treatment for PFAS.

Underground Injection

Class I Wells (hazardous and non-hazardous)

Table D-6. Estimated Costs for Class I Wells

Underground Injection Cost	$/1,000 gal[a]
Deep well injection	$190-270 (deSilva, 2019)
Trucking costs	$10-200 (McCurdy, 2011)

[a] Costs were normalized to 2022 dollars using the Bureau of Economic Analysis (BEA) gross domestic product (GDP) deflator (U.S. BEA, 2023) for waste management and remediation services using a base year of 2019.

Limitations:

- Class I well capacity is limited, which may affect the costs associated with deep well injection.
- Because of the geographic concentration of Class I wells, waste producers may face transportation and logistical challenges.
- To begin accepting PFAS-containing fluids, owners/operators of existing Class I wells may incur costs to modify their permits to accept waste from other entities or new waste streams.

Another source includes costs for disposing of sewage sludge and indicates that among fees and transportation costs is a $37.50 per ton fee for deep well injection of leachate (CDM Smith, 2020).

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Additional Resources

Barr Engineering Co. and Hazen and Sawyer. (2023). Evaluation of Current Alternatives and Estimated Cost Curves for PFAS Removal and Destruction from Municipal Wastewater, Biosolids, Landfill Leachate, and Compost Contact Water - Prepared for Minnesota Air Pollution Control Agency. *https://www.pca.state.mn.us/sites/default/files/c-pfc1-26.pdf*

> **Per the report:** *This study develops alternatives to remove and destroy per- and polyfluoroalkyl substances (PFAS) from water resource recovery facility (WRRF) effluent, biosolids, mixed municipal solid waste (MSW) landfill leachate, and compost contact water (waste streams) using currently feasible technologies (i.e., could be built today). The report summarizes estimated capital and operations and maintenance (O&M) cost ranges for the two highest-ranking PFAS management alternatives for four waste streams (WRRF effluent, WRFF biosolids, landfill leachate, and compost contact water).*

Defense Logistics Agency (DLA) (2023). DLA Disposition Services hazardous waste disposal pricing web application. *https://public.logisticsinformationservice.dla.mil/hazmat/SearchParameters.aspx* (accessed March 2024).

> The pricing web application lists Department of Defense contracts for destroying or disposing of PFAS and PFAS-containing materials. The website is organized by multiple contract line item numbers (CLIN), each of which describes a material or waste stream and a unit of measure for disposal. The site lists contracts and pricing for available disposal technologies, including solidification and disposal in a hazardous waste landfill. The application may be useful as a reference for historical pricing for Department of Defense contracts.
>
> **Per the DLA pricing web application:** *The prices contained in the database were awarded based on contract terms and conditions, procurement requirements, and price reasonableness determinations. The prices and contract information in the database apply to Department of Defense requirements and are for reference only.*

Exhibit D

2024 Interim Guidance on the Destruction and Disposal of PFAS

Fact Sheet

U.S. Environmental Protection Agency has issued an update to the *Interim Guidance on the Destruction and Disposal of Perfluoroalkyl and Polyfluoroalkyl Substances and Materials Containing Perfluoroalkyl and Polyfluoroalkyl Substances (PFAS)*.

The interim guidance provides recommendations for managers of PFAS and PFAS-containing materials to protect human health and the environment. It also contains a new technology evaluation framework to help analyze the safety and effectiveness of new destruction and disposal (D&D) technologies. The interim guidance does not establish requirements for destruction or disposal of PFAS materials. For more info about PFAS and what EPA is doing to address PFAS, visit EPA's *PFAS website*.

The interim guidance summarizes scientific information on current understanding of PFAS and focuses on three currently used D&D technologies:[1]

1) Underground injection (UIC)
2) Landfills
3) Thermal treatment under certain conditions, which includes incineration.

The interim guidance summarizes research needs and data gaps and calls for increased collaboration with EPA to collect data and enhance decision-making. The document also describes new EPA test methods and improved screening tools to identify and prioritize safeguards for communities located near D&D facilities that are already overburdened by pollution. Decisions regarding the management of PFAS and PFAS-containing materials are specific to each type of material and D&D option.

Key Findings in 2024:

Updated information on destruction and disposal technologies

As a general approach, EPA encourages managers of PFAS and PFAS-containing materials to use D&D options that have a lower potential for releasing PFAS to the environment as described in Section 1 of the interim guidance. In general, the following technologies (in no particular order) have a lower potential for environmental release of PFAS compared to other technologies within the categories of storage, underground injection, landfilling, and thermal treatment:

- **Interim storage with controls**: Storage is not a D&D technology but may be a short-term option. Storage may be more fitting for some PFAS materials than others. For example, EPA recommends interim storage of containerized or high PFAS-content materials. In contrast, some materials may be less fit for storage because they are continuously generated or have high-volume and low-PFAS content. With proper controls in place, interim storage can control PFAS migration (Section 1).
- **UIC–Permitted Class I non-hazardous industrial or hazardous waste injection wells:** The standards associated with the construction, operation, and monitoring of these Class I wells are designed to isolate liquid wastes deep below the land surface and ensure protection of underground sources of drinking water.

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[1] For general information on these technologies, see EPA websites for *incineration* , *landfills*, and *underground injection*. Please note that these resources are not specific to PFAS and not all information in them may be relevant to PFAS D&D.

While Class I wells are an option for managing PFAS-containing fluids, this technology may not be appropriate or available everywhere (Section 3).

- **Landfills–Permitted hazardous waste landfills:** When landfill disposal is selected and PFAS concentration of the waste is relatively high, EPA recommends using a hazardous waste landfill. However, for all landfill types, new information shows landfills release more PFAS to the environment than previously thought in 2020. Hazardous waste landfills have leachate emission protections that help control environmental releases of PFAS. These controls are especially important for certain types of PFAS-containing materials that break down more easily in landfill conditions (Section 3).
- **Thermal treatment–Permitted hazardous waste combustors that operate under certain conditions:** New research since 2020 indicates that thermal treatment units operating under certain conditions are more effective at destroying PFAS and minimizing releases or exposures (Section 3). Certain hazardous waste combustors and certain granular activated carbon (GAC) reactivation units may operate under these conditions, but uncertainties remain. For example, more information is needed to determine whether harmful products of incomplete combustion or PFAS air emissions are formed by units operating at lower temperatures (e.g., municipal waste combustors).

EPA has released a new analytical test method, *OTM-50*, that will help collect more data and answer some of these questions—such as those concerning products of incomplete combustion. The updated interim guidance encourages testing with a range of methods at thermal treatment facilities before accepting large quantities of PFAS-containing materials (Section 3).

These and other technologies are discussed in the interim guidance, plus testing and research needs to improve technology performance, improve understanding of PFAS behavior, and reduce uncertainties (Section 5).

Emerging Destruction and Disposal Technologies

Many companies and researchers are developing and testing new PFAS D&D technologies. The guidance provides a technology evaluation framework to help analyze the safety and effectiveness of new D&D technologies, and notes the need for innovation, research, and validation (Section 6).

Impact on Vulnerable Communities

EPA has also shared updated tools, methods, and approaches[2] for considering the impacts of potential releases and exposure on communities located near D&D facilities (Section 4). EPA recommends using these tools to identify and consider potential impacts and ways to protect the health of nearby vulnerable populations, engage the community, and inform decision-making.

Public Input and Participation

EPA has opened a *docket* to take input on the *Interim Guidance on the Destruction and Disposal of Perfluoroalkyl and Polyfluoroalkyl Substances and Materials Containing Perfluoroalkyl and Polyfluoroalkyl Substances*—Version 2 (2024). We welcome public input on how to improve the interim guidance and this fact sheet.

What's Next?

EPA and other government, academic, and private institutions will continue research to better understand PFAS D&D. EPA will review public comments, advances in research, and new science to revise the interim guidance again within three years as required in the National Defense Authorization Act (NDAA).

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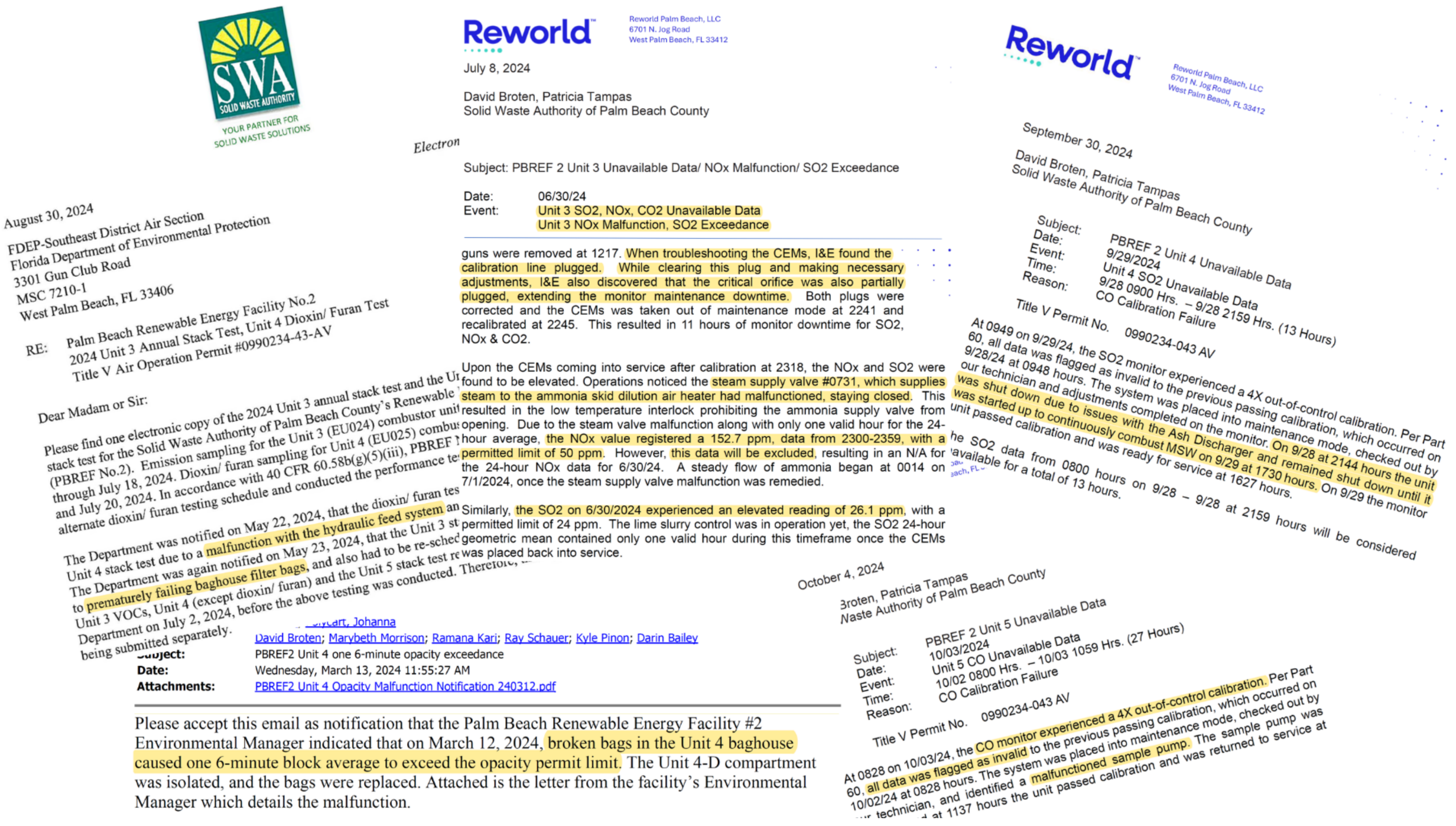
[2] https://www.epa.gov/ejscreen

Exhibit E

Operating Track Record of the “Cleanest and Greenest” Trash Incinerator in the United States

A Critical Review of Select Air Permitting files from Florida Department of Environmental Protection on Palm Beach Renewable Energy Facility 2 in West Palm Beach, Florida

January 24, 2025



SWA SOLID WASTE AUTHORITY
YOUR PARTNER FOR SOLID WASTE SOLUTIONS

August 30, 2024

FDEP-Southeast District Air Section
Florida Department of Environmental Protection
3301 Gun Club Road
MSC 7210-1
West Palm Beach, FL 33406

RE: Palm Beach Renewable Energy Facility No.2
2024 Unit 3 Annual Stack Test, Unit 4 Dioxin/ Furan Test
Title V Air Operation Permit #0990234-43-AV

Dear Madam or Sir:

Please find one electronic copy of the 2024 Unit 3 annual stack test and the Ur
stack test for the Solid Waste Authority of Palm Beach County’s Renewable
(PBREF No.2). Emission sampling for the Unit 3 (EU024) combustor unit
through July 18, 2024. Dioxin/ furan sampling for Unit 4 (EU025) combus
and July 20, 2024. In accordance with 40 CFR 60.58b(g)(5)(iii), PBREF
alternate dioxin/ furan testing schedule and conducted the performance te

The Department was notified on May 22, 2024, that the dioxin/ furan tes
Unit 4 stack test due to a malfunction with the hydraulic feed system an
The Department was again notified on May 23, 2024, that the Unit 3 st
to prematurely failing baghouse filter bags, and also had to be re-sched
Unit 3 VOCs, Unit 4 (except dioxin/ furan) and the Unit 5 stack test re
Department on July 2, 2024, before the above testing was conducted. Therefore,
being submitted separately.

Electron

Reworld Palm Beach, LLC
6701 N. Jog Road
West Palm Beach, FL 33412

July 8, 2024

David Broten, Patricia Tampas
Solid Waste Authority of Palm Beach County

Subject: PBREF 2 Unit 3 Unavailable Data/ NOx Malfunction/ SO2 Exceedance

Date: 06/30/24
Event: Unit 3 SO2, NOx, CO2 Unavailable Data
Unit 3 NOx Malfunction, SO2 Exceedance

guns were removed at 1217. When troubleshooting the CEMs, I&E found the calibration line plugged. While clearing this plug and making necessary adjustments, I&E also discovered that the critical orifice was also partially plugged, extending the monitor maintenance downtime. Both plugs were corrected and the CEMs was taken out of maintenance mode at 2241 and recalibrated at 2245. This resulted in 11 hours of monitor downtime for SO2, NOx & CO2.

Upon the CEMs coming into service after calibration at 2318, the NOx and SO2 were found to be elevated. Operations noticed the steam supply valve #0731, which supplies steam to the ammonia skid dilution air heater had malfunctioned, staying closed. This resulted in the low temperature interlock prohibiting the ammonia supply valve from opening. Due to the steam valve malfunction along with only one valid hour for the 24-hour average, the NOx value registered a 152.7 ppm, data from 2300-2359, with a permitted limit of 50 ppm. However, this data will be excluded, resulting in an N/A for the 24-hour NOx data for 6/30/24. A steady flow of ammonia began at 0014 on 7/1/2024, once the steam supply valve malfunction was remedied.

Similarly, the SO2 on 6/30/2024 experienced an elevated reading of 26.1 ppm, with a permitted limit of 24 ppm. The lime slurry control was in operation yet, the SO2 24-hour geometric mean contained only one valid hour during this timeframe once the CEMs was placed back into service.

Reworld Palm Beach, LLC
6701 N. Jog Road
West Palm Beach, FL 33412

September 30, 2024

David Broten, Patricia Tampas
Solid Waste Authority of Palm Beach County

Subject: PBREF 2 Unit 4 Unavailable Data
Date: 9/29/2024
Event: Unit 4 SO2 Unavailable Data
Time: 9/28 0900 Hrs. – 9/28 2159 Hrs. (13 Hours)
Reason: CO Calibration Failure

Title V Permit No. 0990234-043 AV

At 0949 on 9/29/24, the SO2 monitor experienced a 4X out-of-control calibration. Per Part 60, all data was flagged as invalid to the previous passing calibration, which occurred on 9/28/24 at 0948 hours. The system was placed into maintenance mode, checked out by our technician and adjustments completed on the monitor. On 9/28 at 2144 hours the unit was shut down due to issues with the Ash Discharger and remained shut down until it was started up to continuously combust MSW on 9/29 at 1730 hours. On 9/29 the monitor unit passed calibration and was ready for service at 1627 hours.

he SO2 data from 0800 hours on 9/28 – 9/28 at 2159 hours will be considered
available for a total of 13 hours.

David Broten; Marybeth Morrison; Ramana Kari; Ray Schauer; Kyle Pinon; Darin Bailey

Subject: PBREF2 Unit 4 one 6-minute opacity exceedance
Date: Wednesday, March 13, 2024 11:55:27 AM
Attachments: PBREF2 Unit 4 Opacity Malfunction Notification 240312.pdf

Please accept this email as notification that the Palm Beach Renewable Energy Facility #2 Environmental Manager indicated that on March 12, 2024, broken bags in the Unit 4 baghouse caused one 6-minute block average to exceed the opacity permit limit. The Unit 4-D compartment was isolated, and the bags were replaced. Attached is the letter from the facility’s Environmental Manager which details the malfunction.

October 4, 2024

Broten, Patricia Tampas
Naste Authority of Palm Beach County

Subject: PBREF 2 Unit 5 Unavailable Data
Date: 10/03/2024
Event: Unit 5 CO Unavailable Data
Time: 10/02 0800 Hrs. – 10/03 1059 Hrs. (27 Hours)
Reason: CO Calibration Failure

Title V Permit No. 0990234-043 AV

At 0828 on 10/03/24, the CO monitor experienced a 4X out-of-control calibration. Per Part 60, all data was flagged as invalid to the previous passing calibration, which occurred on 10/02/24 at 0828 hours. The system was placed into maintenance mode, checked out by technician, and identified a malfunctioned sample pump. The sample pump was at 1137 hours the unit passed calibration and was returned to service at

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Commissioned by The Goldstein Environmental Law Firm, P.A.,
on behalf of the City of Miramar, Florida

Principal Author:

Mike Ewall
Energy Justice Network
215-436-9511
mike@energyjustice.net
www.energyjustice.net

An online copy of this report is available at:
www.energyjustice.net/fl/wpb2history.pdf

SUMMARY

From December 1981 until February 12, 2023, Miami-Dade County relied on burning much of its municipal solid waste (MSW) at the trash incinerator known as the Miami-Dade County Resource Recovery Facility or “Covanta Dade” in the City of Doral. The incinerator was owned by the county but privately operated by Covanta (now renamed “Reworld” as of April 2024), just as the Palm Beach County incinerators are county-owned and now privately operated by Covanta (now Reworld).

On February 12, 2023, a fire broke out and burned for three weeks, causing the permanent closure of the incinerator. Miami-Dade County has since been planning to build a new $1.6 Billion incinerator capable of burning 4,000 tons/day, which would be the largest in the U.S. and one of the largest in the world. Currently, the largest incinerator in the U.S. burns up to 3,500 tons/day.

Maimi-Dade County claims new incinerators are clean and safe. The newest trash incinerator built in the U.S., which came online in early 2015, is Palm Beach Renewable Energy Facility 2 built adjacent to Palm Beach County’s older incinerator. **Miami-Dade County’s website boasts that “[t]he Renewable Energy Facility in West Palm Beach is a $672,000,000, state-of-the-art waste-to-energy facility – the most advanced, efficient, cleanest and greenest waste-to-energy power plant in the world.”**[1]

This impression of the county apparently comes from its chosen solid waste consultant, Arcadis, a firm with a strong pro-incineration bias reflected in their work across the United States. On September 19, 2023, Arcadis testified before the County Commission that the new trash incinerator in Palm Beach County used state-of-the-art incinerator technology. The facility began commercial operations on July 18, 2015, and, according to Arcadis, has “worked fabulously for them ever since.”[2]

This report reviews the actual operating history of this trash incinerator, the newest in the U.S. and the only one operating under what were modern emissions standards at the time it was permitted.[3] **This report catalogues 176 incidents (emissions limit exceedances, emissions equipment malfunctions, lost emissions data, and more) at the Palm Beach Renewable Energy Facility 2 trash incinerator in its first decade of operation.**

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[1] Miami-Dade County, “The Future of Solid Waste in Miami-Dade.” See section under “Waste-to-energy around the world.” https://www.miamidade.gov/global/solidwaste/sustainable-solid-waste/wte-home.page

[2] Christopher Tillman testified for Arcadis, stating: “We, you know, we work with other municipalities on this and other types of solid waste issues. And one thing, one very, very important point I'd like to make is that a modern waste-to-energy, a modern mass burn waste-to-energy plant is not an incinerator of the 1970s. Your [resource recovery] facility was an old what we call an RDF [refuse-derived fuel] facility. It was built in 1978, and it was an older, outmoded technology. That is not what we're talking about here. A modern mass burn waste-to-energy facility would be something like what they have in West Palm Beach, which they just built in 2015. It went operational in 2015, and it has worked fabulously for them ever since.” Find the full conversation on pages 7-9 of the transcript here: https://drive.google.com/file/d/1cgO1vzcVWN-Bdc_VzZ9wfKuhY4WZdmiE

[3] Note that the U.S. Environmental Protection Agency (EPA) is required to update emissions standards for Large Municipal Waste Combustors and other industries every five years. However, EPA had not done so since 2006 and had to be sued in federal court to enforce the requirements of the Clean Air Act. EPA finally proposed a new rule in January 2024, and this rule was court ordered to be completed by the end of 2024. However, on December 17, 2024, the environmental organizations that sued to enforce the law agreed to delay the new rules until December 22, 2025. See: https://www.wastedive.com/news/epa-large-municipal-waste-combustor-rule-deadline-incinerator-regulations/736160/ It is unclear whether the rule will be finalized and enforced under a Trump administration. Even if it were finalized in December 2024, the new regulations would not be in effect until around 2028. Large Municipal Waste Combustors are trash incinerators where each burner can burn more than 250 tons/day – a size which pertains to all of the incinerators discussed here. See: https://www.epa.gov/stationary-sources-air-pollution/large-municipal-waste-combustors-lmwc-new-source-performance

OFF TO A BAD START

Palm Beach Renewable Energy Facility 2 (PBREF#2) has three burners, each capable of burning 1,000 tons of waste per day. Since the older incinerator at the adjacent Palm Beach Renewable Energy Facility 1 plant has two burners, the three burners at PBREF#2 are referred to as Units 3, 4, and 5. Unit 3 started burning waste in January 2015 and Units 4 & 5 started in March 2015, and the plant is considered to have started commercial operation in July 2015.

PBREF#2 began experiencing emissions control problems within a week of commercial operations in July of 2015. First, Unit 3 malfunctioned when a draft fan tripped due to above-standard flue gas temperatures, resulting in a 4-hour exceedance of carbon monoxide above PBREF#2's air emissions permit limit. Two days later, Unit 4's inlet probe (which monitors carbon dioxide (CO_2), carbon monoxide, sulfur dioxide, and nitrogen oxides) malfunctioned, resulting in a loss of inlet sample readings. One day after Unit 4's inlet probe malfunctioned, Unit 4's nitrogen oxides outlet probe – as well as Unit 3's carbon monoxide inlet and nitrogen oxides outlet probes – also malfunctioned, resulting in lost sample readings. Before the end of the year, there would be two more probe malfunctions for Units 3 and 5, as well as an opacity exceedance at Unit 4.

NOT JUST A LEARNING CURVE: EXCESS EMISSIONS AND MISSING DATA CONTINUE

This wasn't just a matter of a learning curve, where operations smooth out after a rocky start. Quarterly operating reports show excessive exceedances of opacity (darkness of air emissions, a proxy for particulate matter pollution) in late 2019, and excessive exceedances of nitrogen oxides and sulfur dioxide emissions in mid-2022.

There was also an unusual amount of down time in 2022 and 2023 for the plant's continuous emissions monitors for sulfur dioxide, nitrogen oxides and opacity.

While the incinerator was required to operate continuous monitors for mercury from 2016 through 2018, they had an extraordinary amount of down time, with three quarters reporting that these monitors were not operating 32 to 43% of the time, even though two quarters managed to report down time as low as 0.09% and 0.11%. By way of comparison, down time for continuous monitoring of carbon monoxide, sulfur dioxide, nitrogen oxides and opacity averaged 0.61% and no quarterly report evaluated for this report exceeded 8.73% during a quarter.

VIOLATIONS IDENTIFIED

The following violations were found in the review of documents for this report:

2016: according to a letter regarding the 2017 annual compliance test failure, the particulate matter and ammonia test had also failed in 2016 and were granted a retest. However, no indication of a formal violation was identified in the document search.

2017: the annual compliance test in April 2017 failed for particulate matter (PM) and ammonia was determined to be a **High Priority Violation, for which the Solid Waste Authority was fined $850** for the following reasons:

1) The facility operated for 1 month (714.2 total hours) from the Violation Date (3/24/2017) to the scheduled outage on 4/24/2017.
2) Because the PM limit established in Specific Condition B.13 of Permit No. 0990234-037-AV is based on a BACT Determination pursuant to CAA Title I, Part C (Prevention of Significant Deterioration, PSD), this meets Criterion 2 of the High Priority Violation Determination.

2017: permit limit exceedance: on 8/30/2017, "the ID fan tripped which caused a master fuel trip on Unit 5 resulting in one (1) 4- hour block average of CO to exceed the 100 ppmvd permit limit." No indication of a formal violation was identified in the document search.

2019: Procedural violations determined in Jan 2020:

1) 3-year opacity zero alignment check on COMS may not have been performed in 2018, as required by 40 CFR 60, Appendix F, Procedure 3, Section 10.3 (3).
2) The SO_2 outlet monitor for PBREF-2 span range was 0-500ppm instead of the 50% of the maximum uncontrolled SO_2 emissions of 400ppm [40 CFR 60.58b(e)(12)(ii)]
3) PBREF-1 and PBREF-2 data were not properly invalidated and reported, as defined in 40 CFR 60 Appendix F, Procedure 1, Section 4.3.1.

2022: letter dated 9/22/2022: "Based on the information the facility has provided during the site visit on August 4, 2022 and the action plan submitted on September 20, 2022 the facility was determined to have returned to compliance with all permit conditions." We were unable to identify the document referred to in this letter.[4]

2022: from an incident/malfunction report dated 10/24/2022:

> "On 7/6/2022, Unit 4 recorded an SO_2 24-hour geometric average from 0000-2359 hours of 26.3 ppm, corrected to 7% 02. Unit 4 was experiencing unusually elevated inlet SO_2 levels in the flue gas beginning at approximately 1415 hours. Facility personnel responded to the rapidly increasing inlet SO_2 by reducing the boiler load, initiating, and then increasing gas flow to the boiler and placing the lime system in manual control in order to override system interlocks. Inlet and outlet SO_2 values continued to increase until approximately 1640 hours at which time the boiler's **CEMS analyzers reached an over-range condition and flagged the SO_2 data as "Data Error" thereby excluding the data from compliance averaging periods**. Inlet and outlet SO_2 levels continued at CEMS over-range levels until approximately 1820 hours when the **unusually high SO_2-containing fuel** had been processed through the boiler." [Emphasis added.]

It is concerning that where emissions are actually found to be excessive, and attributed to burning waste that is high in sulfur, the high emissions are discounted as a data error and are excluded from

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[4] https://depedms.dep.state.fl.us:443/Oculus/servlet/shell?command=view&[guid=75.283993.1]&[profile=profile]

the data used to determine compliance. Continuous emissions monitors (CEMS) should not be programmed to flag high emissions as errors. This situation means that there is no way to know how high emissions actually get when there are unusually high emissions.

When CEMS data is unavailable, Reworld's incident notifications to the DEP almost routinely state that the boiler was operating normally and all air pollution control equipment was functioning properly and no excess emissions were expected.

DATA GAPS AND LIMITATIONS

All data presented in this report is from the Florida DEP, and primarily represents information provided to DEP by Palm Beach County's Solid Waste Authority (the incinerator's owner) or Reworld (formerly Covanta), the company hired to operate the incinerator.

DEP makes their files available online through a system called OCULUS.[5] While this transparency is much better than most states, it is very challenging for the general public to navigate and learn how to find a given facility's files. Moreover, the files are poorly categorized, making it difficult to focus in on any given set of files.

Making it far more challenging to navigate is that the Palm Beach County Solid Waste Authority's waste complex operates several waste facilities under a single permit number. In most states, each facility would have its own permit number, but in DEP's file system, the Palm Beach Renewable Energy Park ("PBREP") has two trash-burning incinerators with five burners between them, two landfills, a sewage sludge ("biosolids") processing facility, and a couple dozen ancillary pollution point sources. Naming conventions in DEP's OCULUS are inadequate to distinguish between facilities in many cases, requiring one to go in and out of thousands of documents to figure out which ones pertain to which facility. There is not even a consistent naming convention to easily identify the periodic quarterly and annual reports. Due to these challenges, we could only identify 27 of the 37 (73%) of the quarterly reports that should be available, and cannot provide comprehensive trendlines for compliance data we compiled.[6]

Due to the challenges and limitations of this data review, this report should not be considered comprehensive, but just a sampling of the available data available on emissions monitoring gaps, violations, and other mishaps at this incineration facility.

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[5] https://depedms.dep.state.fl.us/Oculus/

[6] The air-related files for the "Palm Beach Renewable Energy Park" can be found here: https://prodenv.dep.state.fl.us/DepNexus/public/electronic-documents/AIR_0990234/facility!search Note that this does not include other file types like regulation under DEP' waste program, and that the 2,842 files available in this search (as of this writing) pertain to all facilities at the complex, including the older incinerator (REF 1), the landfills, and the sewage sludge "biosolids processing facility" (BPF). One must be careful to distinguish when using these files to examine any one of the facilities at that site.

ACRONYMS

BACT	Best Available Control Technology
CEM / CEMS	Continuous Emissions Monitor / Continuous Emissions Monitoring System
CGA	Cylinder Gas Audit
CO	Carbon monoxide
CO_2	Carbon dioxide
DEP	Florida Department of Environmental Protection
Hg	Mercury
MCC	Motor Control Center
MSW	Municipal Solid Waste
NH_3	Ammonia
NOx	Nitrogen oxides
NSPS	New Source Performance Standards
ppm	parts per million
ppmdv/ppmvd	parts per million dry volume
REF1 / PBREF1	Palm Beach Renewable Energy Facility #1 (the old trash incinerator)
REF2 / PBREF2	Palm Beach Renewable Energy Facility #2 (the new trash incinerator)
SCR	Selective Catalytic Reduction (a modern control system for NOx emissions that involves spraying ammonia or urea into the exhaust, then using vanadium pentoxide catalyst to further reduce NOx emissions; PBREF 2 is the only trash incinerator to use this in the U.S.)
SDA	Spray Dryer Absorber ("scrubbers" that spray lime into incinerator exhaust to reduce SO_2 emissions)
SO_2	Sulfur dioxide

VIEWING SOURCE DOCUMENTS

Source documents for incidents in the chart below are available by clicking on the date for each incident. When visiting each link, click through the "Public OCULUS Login" button to download and view the source document. If the document does not automatically download or display, click the Adobe symbol under "File Type." If seeing a prompt to get Adobe Reader (and if you already have it installed), you may need to then click on the document name in the left sidebar.

2015-2024 Incidents, Exceedances, Violations, and other Mishaps at Palm Beach Renewable Energy Facility #2 Trash Incinerator

Date (link)	Summary	Unit	Pollutant	Notes
7/7/2015	On July 6, 2015, there was a malfunction with Unit 4 inlet analyzers for CO2, NOx, and SO_2 which resulted in 4 hours of invalid data. A piece of debris was found lodged in the solenoid valve allowing calibration gas to leak by the valve.	Unit 4	CO2, NOx, SO_2	solenoid valve blocked by debris caused calibration gas leak
7/20/2015	Unit 3 ID Fan drive tripped due to high temperature. The air conditioner tripped in the MCC room causing the variable frequency drive to overheat, thus causing the fan to trip.	Unit 3		ID fan trip
7/28/2015	On 7/22/2015 at approximately 09:55 Unit 4 inlet probe malfunctioned, resulting in a loss of inlet sample readings for CO2, CO, SO_2 and NOx. Troubleshooting the cause of the loss of sample readings followed. Several maintenance components within the probe were cleaned such as the fast loop eductor and dilution manifold. The probe filter and critical orifice were also replaced. Several leak checks were performed on the umbilical also. These troubleshooting strategies had no positive results in restoring the probes functionality.	Unit 4	CO2, CO, SO_2, NOx	probe inlet
9/14/2015	On September 11, 2015, the Unit 3 outlet SO_2 and NOx probe malfunctioned as indicated by a slow response to calibration and low calibration gas flow. Through troubleshooting, it was discovered that there was a small crack in a piece of stainless steel tubing associated with the probe. This probe malfunction resulted in four hours of unavailable NOx and SO_2 data.	Unit 3	SO_2, NOx	probe outlet
10/22/2015	On October 21, 2015 the Unit 5 CO probe plugged. This malfunction resulted in 2 hour of unavailable data.	Unit 5	CO	probe plugged
10/23/2015	On October 22, 2015 there was a sudden increase in furnace pressure, ID fan speed, and baghouse differential pressure on Unit 4. It is believed that there may have been a small explosion in the furnace which resulted in a one (1) 6-minute opacity exceedances of 10.5%.	Unit 4		furnace pressure increase, maybe from small explosion
1/28/2016	January 27, 2016, there was a loss of air pressure at PBREF#2 which tripped Units 3, 4 and 5. There were no excess air emissions above the Title V air permit limits as a result of the malfunction on Units 3 and 5, however, Unit 4 experienced a CO 4-hour block average of 109.3 ppmvd. The permit limit is 100 ppmvd 4-hour block for CO.	Units 3, 4, and 5	CO	air pressure loss
1/28/2016	The mercury analyzer (SICK MERCEM) was experiencing erratic readings, elevated at times, but reading values with an error message for heater trouble. After some troubleshooting a thermocouple was replaced to erase the heater error. Additional adjustments were made to lamp intensity as well as some cleaning to the reflector lens.	Unit 5	Hg	thermocouple

Date (link)	Summary	Unit	Pollutant	Notes
2/1/2016	Units 3, 4, and 5 tripped due to loss of instrument air when a solenoid valve failed to open. Boiler 3 and 5 were brought back online and Boiler 4 tripped. FD fan tripped on Boiler 4 due to high temperature. I&E troubleshooting fan high temperature issue. The malfunction caused the 4 hour block for CO to exceed the permit limit of 100 ppmvd.	Units 3, 4, and 5	CO	solenoid valve
2/18/2016	On February 17, 2016 the Unit 5 CO2 probe malfunctioned which caused elevated CO values. The system was put into maintenance mode and the CO2 probe was cleaned and calibrated. This malfunction resulted in 3 hour of unavailable data.	Unit 5	CO	probe
3/18/2016	The wet scrubber, known as the Wet Whirl, in the ash management building is currently out of service. On January 30, 2016 the conduit feeding the electric to the motor was struck and damaged by a vehicle operating within the building. There was significant damage to electrical system and parts had to be ordered.	Wet Whirl Scrubber	ash	electric conduit
3/23/2016	On March 22nd, 2016, the Unit 4 Opacity Monitor had two (2) six minute periods of unavailable data.	Unit 4	opacity	
4/18/2016	On April 15, 2016 there was an opacity exceedance on Unit 3.	Unit 3	opacity	
4/18/2016	On April 16, 2016, there was a loss of hydraulic pressure to the grates and charging hoppers on Unit 3. This malfunction resulted in 1 4-hour block to exceed the permit limit of 100 ppmvd.	Unit 3	CO	
4/21/2016	On April 20, 2016, the PBREF No. 2 Hg CEMS began malfunctioning giving erratic readings. The facility operator has tried troubleshooting the problem, but has not been successful in resolving the erratic readings.	Unit 5	Hg	
5/2/2016	On April 30, 2016, at the PBREF#2, an air heater tube ruptured on Boiler 4 releasing steam into the gas stream downstream of the baghouses. This steam release caused two invalid periods of opacity data before the unit was shut down.	Unit 4	opacity	air heater tube rupture
5/5/2016	On May 4, 2016 there was a malfunction of the level control to the steam coil gas heater resulting in the ammonia injection system to shut down. This malfunction resulted in the NOx 24-hour average to exceed the permit limit of 50 ppmvd.	Unit 4	NOx	ammonia injection system
5/6/2016	As Unit 4 was returning to service the atomizer tripped stopping slurry and water control, thus increasing SDA outlet temperature. Several attempts were made to restart the atomizer with each failing. The in-service atomizer was replaced with the spare atomizer. After resetting the power to the atomizer at the main breaker, the atomizer was successfully restarted.	Unit 4		atomizer failure
5/16/2016	On May 14, 2016, at the PBREF#2, within the Boiler 5 Finishing Heat Exchanger, air heater tube(s) ruptured, releasing steam into the gas stream. This steam release caused four invalid periods of opacity data. The unit was shut down and the Finishing Heat Exchanger was bypassed.	Unit 5	opacity	air heater tube rupture
5/20/2016	PBREF#2 invalid SO_2 lb/hr data on Unit 3	Unit 3	SO_2	

Date (link)	Summary	Unit	Pollutant	Notes
5/23/2016	The mercury analyzer (SICK MERCEM) was experiencing communication failure, erratic readings and was unable to calibrate. KVB Enertec on site troubleshooting the communication failure between the Hg analyzer and the DAHS.	Unit 5	Hg	analyzer communication failure
5/31/2016	Unit 4 NH_3 (CTM 027- Ammonia slip) compliance test result, conducted on April 22, 2016, was 3.34 lb/hr, exceeding the 2.76 lb/hr permit limit. Please note however, that the NH_3 ppmvd value on this unit was 9.35 ppmvd, which is below the 10 ppmvd permit limit.	Unit 4	NH_3	**permit limit exceedance**
7/5/2016	On July 4, 2016, the Unit 5 ash discharger plugged. This malfunction resulted in one 4-hour block to exceed the permit limit of 100 ppmvd.	Unit 5	CO	ash discharger plug
7/5/2016	On July 3, 2016 there was a loss of NOx outlet data on Unit 4. This malfunction resulted in nine hours of unavailable data. During this time the boiler was operating normally as indicated by CO and outlet SO_2 values. Therefore, it is not anticipated that the 24-hour average NOX value would be above the permit limit.	Unit 4	NOx	probe outlet malfuntion
7/18/2016	On July 16, 2016, the Unit 5 ID Fan tripped. This malfunction resulted in two (2), 4-hour block averages of CO to exceed the permit limit of 100 ppmvd.	Unit 5	CO	ID fan trip
10/21/2016	Palm Beach Renewable Energy Facility No. 2 (PBREF #2) indicated that on October 20, 2016, two 4- hour blocks of CO exceedances occurred on Boiler #4. The first 4- hour block was due to an ID fan trip.	Unit 4	CO	ID fan trip
1/23/2017	Notice of lost opacity data EU 025	Unit 4	opacity	
2/20/2017	The daily calibration check failed on the Unit 4 inlet probe. During the attempt to manually calibrate the analyzers it was noted that there was no response to calibration gas. Typical plan of action would be to clean or replace the sample probe filter and flow calibration gas again. This was done without success. The probe critical orifice and fast loop eductor were cleaned and replaced. Unfortunately, this did not solve the problem either. Troubleshooting continued for several hours in an attempt to understand why the probe was not responding to calibration gas. During another line check in the probe enclosure it was found that a piece of stainless tubing had a small crack in the back of a bend. The tubing was removed, a new piece bent and installed.	Unit 4		probe inlet tubing crack
5/11/2017	PBREF-2 Boiler #4 (EU025) failed 2 of 3 runs during the 2017 stack testing for PM and Ammonia. This facility was granted a retest in 2016 for the same unit and same parameters. Based on the BACT determination (0990234-032-AC and PSD-FL-413C), the PM limits are 4.7 lb/hr (12.0 ug/dscm) and the Ammonia limits are 2.76 lb/hr (10 ppm @ 7% O2). Based on Specific Condition B.30 of Permit -038-AV, the facility is required to perform an annual stack test to demonstrate compliance with the emission limits. These units are also subject to 40 CFR 60, Subpart Eb - NSPS for Large Municipal Solid Waste Combustors.	Unit 4	PM, NH_3	**permit limit exceedance**

Date (link)	Summary	Unit	Pollutant	Notes
8/9/2017	On August 5, 2017, there has been a loss of data on the Boiler 5 mercury analyzer which was reporting null or negative readings. The facility attempted to trouble shoot with remote assistance from SICK MERCEM, but was not able to resolve the issue.	Unit 5	Hg	
8/10/2017	On August 9, 2017 there was a 3 hour period of invalid data on Unit 3 SO_2 and NOx monitors due to scheduled quarterly maintenance	Unit 3	SO_2, NOx	
8/21/2017	On 8/5/2017, the mercury analyzer (SICK MERCEM) reported null or negative readings. Upon discovery of the issue, facility staff and remote assistance troubleshooting from SICK was unable to resolve the issue. On 8/9/2017, A SICK MERCEM technician arrived on-site for further troubleshooting. After multiple attempts of powering down and restarting the analyzer, it was discovered that the main motherboard was faulty. Upon ordering and replacement of the motherboard, it was also discovered that the power supply board received alarms related to the heat controller. The alarm prevented the system from collecting sample gas data. The heat controllers were verified to be in working condition, and the analyzer was manually programmed to gather sample data.	Unit 5	Hg	motherboard faulty; heat controller alarms
8/25/2017	On August 24, 2017, there was a loss of 28 hours of data on the Boiler 5 mercury analyzer.	Unit 5	Hg	
8/31/2017	On August 30, 2017, the ID fan tripped which caused a master fuel trip on Unit 5 resulting in one (1) 4- hour block average of CO to exceed the 100 ppmvd permit limit.	Unit 5	CO	**permit limit exceedance**
9/6/2017	The mercury analyzer (SICK MERCEM) was reporting null or negative readings as of 9/5/2017 at approximately 0656. Unit 5 Boiler returned to operation following startup at approximately 0656 on 9/5/2017. The Hg monitor displayed standard alarms upon Unit 5 startup, but was not in operation. Upon restarting the SICK Hg monitor using SICK guidelines and procedure, the monitor did not show any improvement. Following troubleshooting, it was determined that a heat controller component was malfunctioning. The heat controller component was replaced and put back into service. After multiple successful recalibrations, the SICK Hg monitor was put back into service, but the valid data was not being recorded by the CEMS DAHS. Further troubleshooting will be required to correct the CEMS DAHS recording issue.	Unit 5	Hg	heat controller
9/12/2017	On September 8, 2017, there was a loss of data on the Boiler 5 mercury analyzer.	Unit 5	Hg	
9/29/2017	On September 27, 2017, there has been a loss of data on the Boiler 5 mercury analyzer. The operator has tried to troubleshoot the problem, but has not been successful	Unit 5	Hg	
10/2/2017	On September 28, 2017, the daily average for Unit 5 SO_2 lb/hr was invalid due to excessive drift in the flow meter which was confirmed during a flow velocity traverse on the ID fan on October 1, 2017.	Unit 5	SO_2	flow meter excessive drift

Date (link)	Summary	Unit	Pollutant	Notes
10/10/2017	The mercury analyzer experienced a “hardware fault” error resulted in unavailable data from October 8th at 09:29 through October 9th, 2017 at 12:33 for a total of 26 hours	Unit 5	Hg	hardware fault error
10/26/2017	October 25 through October 26, 2017, there was a loss of data on the Boiler 5 mercury analyzer for a total of 16 hours.	Unit 5	Hg	
11/9/2017	PBREF NO. 2 U4 NOX INVALID DATE, EU 025	Unit 4	NOx	
11/13/2017	The Unit 5 Mercury analyzer did not display a fault or warning, however the PBRRC EHS Technical Specialist noticed that the DAHS was not recording the mercury data while he was completing the CEMS CGA on Unit 5 November 8. The facility attempted to trouble shoot the analyzer, but was not able to resolve the issue. A SICK MERCEM technician arrived on site the following day. The analyzer malfunction resulted in 36 hours total down time.	Unit 5	Hg	
11/27/2017	On November 26 through November 27, 2017, there was a loss of data on the Boiler 5 mercury analyzer totaling 30 hours. During this time, the boiler was operating normally and all air pollution control equipment was functioning properly	Unit 5	Hg	
12/18/2017	On December 17 through December 18, 2017, there was a loss of data on the Boiler 5 mercury analyzer totaling 17 hours. During this time, the boiler was operating normally and all air pollution control equipment is functioning properly.	Unit 5	Hg	
12/18/2017	On December 17, 2017, there was a loss of data on the Boiler 4 inlet SO_2 analyzer totaling approximately 4 1/2 hours. The Unit 4 stack SO_2exhaust monitors were fully functioning and showed that there were not any exceedances during this time.	Unit 4	SO_2	
12/29/2017	The original notification of the Hg malfunction resulting in unavailable data was sent to the Department on December 27, 2017. The monitor has been repaired and began collecting data on December 28, 2017 at 11:39am. The total hours of unavailable data for this malfunctions is 133 hours.	Unit 5	Hg	
1/2/2018	On December 30, 2017 through January 01, 2018, there was a loss of data on the Boiler 5 mercury analyzer totaling 52 hours. During this time, the boiler was operating normally and all air pollution control equipment is functioning properly.	Unit 5	Hg	
1/23/2018	Unit 5 Hg analyzer experienced a "hardware fault" error on 1/20/2018. Loss of data for 61 hours.	Unit 5	Hg	hardware fault error
1/29/2018	On January 26, 2018, there was a loss of data on the Boiler 5 mercury analyzer totaling 5 hours.	Unit 5	Hg	
1/29/2018	There was a loss of data on the Boiler 5 mercury analyzer totaling 5 hours.	Unit 5	Hg	
2/8/2018	On February 7, 2018, at 2:29am, there was a loss of data on the Boiler 5 mercury analyzer. The operator restarted the monitor and it is currently working normally. The monitor malfunction resulted in 5 hours of unavailable data.	Unit 5	Hg	
2/9/2018	The Unit 5 Hg analyzer experienced a hardware fault beginning February 8, 2018, at 4:19am, resulting in unavailable data.	Unit 5	Hg	

Date (link)	Summary	Unit	Pollutant	Notes
2/15/2018	The Unit 5 Mercury (Hg) monitor experienced a 'hardware fault' error on 2/8/2018 at 04:19 and was returned to service on 2/14/2018 at 12:00. Details on the event are described below: Upon discovering the 'hardware fault error' on 2/8/2018, PBRRC EHS and I&E attempted to troubleshoot the SICK MERCEM monitor. A lamp ignition failure was noted in the digital log historian. The standard procedure for monitor failures began with restarting the monitor. Fuses for the main motherboard failed upon restart of the monitor, and were replaced. Inconsistently, fuses failed upon restarting the monitor. Unable to determine a root cause for the fuse failures, a SICK MERCEM technician was notified the same day. Due to conflicts in SICK scheduling, the SICK technician was unable to arrive until 2/12/2018. Upon the SICK technician's arrival, multiple attempts to determine the root cause of the initial fuse failures were unsuccessful. Fuses were replaced, and did not fail for the entirety of the SICK technician service call (3 day duration). The original lamp ignition failure was determined to be due to a malfunctioned lamp control module. The lamp control module was replaced on 2/14/2018 and the monitor returned to service at approximately 12:00. The malfunctioned lamp control module was replaced recently and was still under warranty at the time of failure.	Unit 5	Hg	hardware fault
2/26/2018	The Unit 5 Hg analyzer experienced a hardware fault beginning February 23, 2018, at 22:58pm, resulting in unavailable data.	Unit 5	Hg	hardware fault
3/10/2018	The Unit 5 Hg analyzer experienced a hardware fault beginning March 10, 2018, at 14:21pm, resulting in unavailable data	Unit 5	Hg	hardware fault
3/27/2018	The Unit 5 Mercury (Hg) monitor experienced a 'hardware fault' error on 3/26/2018 at approximately 14:10 and has returned to service on 3/27/2018 at approximately 11:27 (21 hours down)	Unit 5	Hg	hardware fault
4/2/2018	On March 30, 2018, Unit 4 CO, NOx and SO_2 lb/hr was invalid for a total of 45 hours due to a malfunction of the stack exhaust pressure transducer.	Unit 4	CO, NOx, SO_2	stack exhaust pressure transducer malfunction
4/3/2018	On April 2, 2018, Unit 4 CO, NOx and SO_2 lb/hr was invalid for a total of 8 hours due to a malfunction of the stack exhaust pressure transducer.	Unit 4	CO, NOx, SO_2	stack exhaust pressure transducer malfunction

Date (link)	Summary	Unit	Pollutant	Notes
4/23/2018	PBREF2 Boiler 3 experienced a CO spike on 4/22/2018 at around 03:14. The control room operator informed the crane operator to select a different area of the pit for MSW fuel feed. In addition, standard operating procedures to adjust air flow and initiate gas guns were followed. Typically, the re-balance of air flow is sufficient to decrease CO to acceptable levels. However, Boiler 3 currently has an over-grate air (OGA) fan which has malfunctioned. The lack of an operational OGA fan prevented CO from being meaningfully affected by air adjustments from over-fire air (OFA) alone. During normal gas gun operations, the burners are ignited and gas guns are inserted into the boiler. Multiple internal system safety checks are performed in order to verify the burners and ignitors are in good operating condition before guns are allowed to be inserted. On 4/22/18, at around 13:24, the attempt to insert gas guns failed due to trips from the internal system safety checks. Multiple attempts to restart the gas gun insertion finally resulted in successful gas gun firing at approximately 13:35. The above malfunctions caused one, 4-hr average CO period to exceed the permit limit of 100ppmvd. The new maximum 4 hour block average is 68.7 ppmvd.	Unit 3	CO	OGA fan malfunction
5/21/2018	On 5/19/2018, Boiler 4 experienced above normal NOX from the period of 0:00 to 13:20. The control room operator informed the crane operator to select a different area of the pit for MSW fuel feed. In addition, standard operating procedures to adjust air flow and initiate gas guns were followed. The air pollution systems at PBREF2 have minimum operating temperatures. Specifically, the spray dry absorber (SDA – for SO_2 control) has a minimum operating temperature of 312F at the inlet and 270°F at the outlet. An additional restriction, is the ammonia injection (SCR system - for NOX control) will not operate unless SO_2 is below a certain concentration (45 ppm) to protect against fouling of the catalyst. Typically, the flue gas in the boilers are high in temperature and allow the SDA, and in turn the SCR, to operate normally.	Unit 4	NOx	fuel content
6/6/2018	There was a probe malfunction on Unit 5 due to particulate plugging. This resulted in 3 hours of unavailable NOx and SO_2 data while the probe was repaired. It is not expected that there were any pollutant exceedances during this time as boiler operations and pollution control systems were functioning normally.	Unit 5	NOx, SO_2	probe particulate plugging
6/26/2018	The Unit 5 Hg analyzer experienced a hardware fault beginning June 25, 2018, at 07:06, resulting approximately 6 hours of unavailable data	Unit 5	Hg	hardware fault
8/30/2018	PBREF No. 2 Unit 5 Hg analyzer experienced a hardware fault malfunction beginning August 29, 2018, at 09:47, resulting approximately 6 hours of unavailable data.	Unit 5	Hg	hardware fault malfunction

Date (link)	Summary	Unit	Pollutant	Notes
9/17/2018	On September 16, 2018, there was a loss of NOx and SO_2 data on the Boiler 3. While the malfunctioning opacity monitor was being replaced (notification is attached), the system was in maintenance mode. During this time, the boiler was operating normally and all air pollution control equipment were functioning properly.	Unit 3	NOx, SO_2	malfunctioning opacity monitor was being replaced
9/17/2018	On Saturday, 9/15/2018 at 22:18 through 9/16/2018 at 13:42, the Unit 3 opacity data was invalid due a malfunctioning opacity monitor. The malfunctioning monitor, serial number 1235735, was replaced on 9/16/18 with a spare monitor, serial number 1244635.	Unit 3	opacity	opacity monitor malfunction
9/27/2018	There was a 4-hour block CO exceedance on Unit 3 on September 27, 2018, due to a malfunctioning variable frequency drive (VFD) of the ID fan causing the boiler to experience a major fault trip (MFT). The gas guns could not ignited due to the loss of air. In addition, the MFT shut down the PAC flow. Because the gas guns were unavailable for shutdown, combustion continued for 83 minutes after the MFT, even though no additional fuel was introduced. The CEMS continued to record the PAC flow averages after the MFT until steam flow ceased. PBRRC staff was able to show compliance with the PAC flow 8-hour block by recalculated the PAC flow using hourly averages, eliminating the time after the MFT and combustion ceased.	Unit 3	CO	ID fan trip
10/1/2018	On Thursday, 9/27/2018 through Monday, 10/1/2018, the Unit 5 opacity monitor experienced 66 six-minute periods of unavailable data due a malfunctioning opacity monitor evaluation unit (Durag DR-290 AW). The opacity monitor normally experiences self-initiating calibrations one time per day at a fixed time. However, on Thursday 9/27/2018, after Boiler 5 came back on line after an outage, the opacity monitor self- initiated calibrations for a total of 5 times, each calibration resulted in a six-minute period of loss of data. Friday through Monday, the self- initiating calibrations increased in frequency. Initially, there was no error code indicating a malfunction, until Monday, 10/1/2018 at approximately 3:30am, error codes were discovered that indicated communication failures and display module error. The evaluation unit for the opacity monitor was replaced and the opacity monitor passed calibrations at approximately 10:48 on 10/1/2018	Unit 5	opacity	opacity monitor malfunction

Date (link)	Summary	Unit	Pollutant	Notes
10/16/2018	There was a 4-hour block CO exceedance on Unit 5 on October 15, 2018, due to a malfunctioning variable frequency drive (VFD) of the ID fan causing the boiler to experience a master fuel trip (MFT). The gas guns could not ignited due to the loss of air. Please note that Condition B.17.c., Special Provisions for CO, of permit 0990234-037-AV allows for a CO exceedance of 15 hours for induced draft fan malfunction. In addition, the MFT shut down the PAC flow through a safety interlock. Because the gas guns were unavailable for shutdown, combustion continued for 317 minutes after the MFT, even though no additional fuel was introduced. The CEMS continued to record the PAC flow averages after the MFT. PBRRC staff was able to show compliance with the PAC flow 8-hour block by recalculated the PAC flow using hourly averages, eliminating the time after the MFT.	Unit 5	CO	ID fan trip
1/16/2019	The Unit 3 Inlet probe experienced plugging of the probe filter, which caused invalid data for the periods of 1/15/2019 16:00 to 1/15/2019 18:00. The probe was unplugged, the filter and O-ring was replaced, and the U3 inlet system returned to service on 1/15/2019 at 18:25. Due to the Unit 3 Inlet probe malfunction, Unit 3 Inlet CO, NOx, and SO_2 data was unavailable from 1/15/2019 at 16:00 through 1/15/2019 at 18:00. Total downtime for this event is 2 hours.	Unit 3	CO, NOx, SO_2	probe inlet filter plugged
2/21/2019	The variable frequency drive (VFD) of the ID (induced draft) fan tripped (Malfunctioned) at approximately 1740 on 2/20/2019. This malfunction caused Boiler 3 to experience a master fuel trip (MFT). Due to the loss of air, gas guns could not be ignited during the boiler shutdown. The control room operator successfully attempted to restart the ID fan approximately 15 minutes after the initial fan trip (fan requires a 15 minute wait time prior to start). The startup of the ID fan allowed combustion to resume and stable combustion was achieved approximately at 2000. The above boiler malfunction caused one, four hour block average CO period to exceed the permit limit of 100 ppmvd. The CO daily average for 23 operating hours is 20.8 ppmvd, and the highest 4 hour block during operation is 36.6 ppmvd. The above exceedance is within the allowed fifteen hours per occurrence for CO as per section B.17 of the Title V Air Operating Permit, number 0990234-040-AV.	Unit 3	CO	ID fan trip
3/12/2019	There was a 4-hour block CO exceedance on Unit 5 on March 11, 2019, due to a malfunctioning variable frequency drive (VFD) of the ID fan causing the boiler to experience a master fuel trip (MFT). The gas guns could not ignited due to the loss of air. Please note that Condition B.17.c., Special Provisions for CO, of permit 0990234-040-AV allows for a CO exceedance of 15 hours for induced draft fan malfunction.	Unit 5	CO	ID fan trip

Date (link)	Summary	Unit	Pollutant	Notes
4/15/2019	The SDA atomizer experienced loss of power on 4/14/2019 at approximately 2021. I&E was called on-site to attempt repairs. During troubleshooting, it was discovered that the main power plug to the atomizer had melted and needed to be replaced. Boiler 5 was forced to shutdown for atomizer repairs. Following atomizer repairs, an electrical relay was also discovered to have malfunctioned, and was replaced. Boiler 5 startup occurred the following day with no additional issues to the SDA atomizer.	Unit 5	SO_2	atomizer; electric relay malfunction
5/3/2019	There was a 4-hour block CO exceedance on Unit 3 on May 2, 2019. A high inverter temperature alarm of the induced draft (ID) fan caused the boiler to experience a master fuel trip (MFT). The gas guns could not ignited due to the loss of air. This malfunction is within the 15 hours allowed for a CO exceedance caused by an ID fan malfunction as specified in Condition B.17.c., Special Provisions for CO, of permit 0990234- 040-AV. Please note that this malfunction occurred during the second run of EPA Method 5/29 of the 2019 annual stack test. The run was paused during the malfunction. The control room operator was able to restart the ID fan and the run was completed before an identical high inverter temperature alarm of the ID fan caused a MFT again. The third Method 5/29 test run is anticipated to be completed this afternoon (5/3/2019). The high inverter temperature alarm is currently under investigation.	Unit 3	CO	ID fan trip
5/31/2019	On May 30, 2019, there was a malfunction with the Boiler 5 opacity monitor that resulted in 42 six- minute periods of unavailable data.	Unit 5	opacity	monitor malfunction
6/24/2019	The Unit 3 stack probe experienced a worn critical orifice, which was replaced after all readings at the Unit 3 stack location were noticed to be elevated. A successful calibration was completed after the orifice replacement, and normal operating emissions values were observed for all Unit 3 stack readings. Due to troubleshooting and re-calibration, Unit 3 Stack NOx, and SO_2 data was unavailable from 6/21/2019 at 10:31 through 6/21/2019 at 12:39. Total downtime for this event is 1 hour.	Unit 3	NOx, SO_2	probe stack orifice worn

Date (link)	Summary	Unit	Pollutant	Notes
6/28/2019	The Boiler 5 ID (induced draft) fan tripped (Malfunctioned) at approximately 20:51 on 6/27/2019. This malfunction caused Boiler 5 to experience a master fuel trip (MFT). Due to the loss of air, gas guns could not be ignited during the boiler shutdown. The control room operator successfully attempted to restart the ID fan approximately 25 minutes after the initial fan trip (fan requires a 15 minute wait time prior to start) at 21:16. Combustion never reached stability, as the ID fan once again tripped at approximately 22:56. After investigation, it was determined that the ID fan tripped due to high inverter temperature. The air conditioning in the motor control center (MCC) room had malfunctioned, which caused the high inverter temperature. The ID fan speed / load has been restricted to 80% to prevent overheating of the inverter, and repairs to the air conditioning in the MCC is currently underway. On 6/28/2019 at approximately 04:00 a successful startup was conducted on Boiler 5, which continues to be in stable operation as of 10:00.	Unit 5	CO	ID fan trip
8/21/2019	The intent of this letter is to correct the duration and times of the unavailable data, and to add CO as a parameter affected for the notification letter sent on 8/12/19: The daily average for Unit 3 CO, NOX, SO_2 lb/hr was invalid due to excessive drift in the flow monitor. Unit 3 came online from a forced outage on 8/10/19 at approximately 09:57. The flow monitor showed elevated stack gas velocities for the duration of 8/10/19 and 8/11/19. The flow monitor was recalibrated on 8/12/2019 at approximately 08:24, and currently measures normal exhaust flue gas velocities. The attached data shows normal CO, NOx, and SO_2 ppmvd hourly averages, and the invalidated abnormal lb/hr hourly data.	Unit 3	CO, NOx, SO_2	flow monitor drift
10/15/2019	Unit 4 has just completed the fall boiler outage. Upon startup, it was noted that the opacity monitor read negative values. On 10/14/19, during the periods of 9:30-10:24 (9 periods) and 14:06-14:24 (3 periods), the opacity monitor was cleaned and recalibrated. Currently, the monitor reads positive values in general agreement with opacity values based on process knowledge. There were a total of 12 six-minute periods of unavailable data	Unit 4	opacity	opacity monitor dirty
11/22/2019	On 11/20/19, the Unit 5 inlet probe showed fluctuating values during the fourth quarter scheduled maintenance event. At approximately 16:13, CEM Solutions began troubleshooting for approximately 5 hours, when it was determined that the vacuum line was intermittently leaking. The line was bypassed and successful calibrations completed at approximately 21:30	Unit 5		vacuum line leak

Date (link)	Summary	Unit	Pollutant	Notes
12/5/2019	The CEMS calibration gas for the daily calibration of the CEMS system was replaced on 12/3/2019 after the morning calibrations. On 12/4/2019, all units experienced CEMS daily calibration failures at approximately 0600-0700 due to the installation of incorrect calibration gas bottles. The correct calibration bottles were installed and recalibrations were conducted. No additional CEMS system hardware changes or repairs were necessary for a successful recalibration. The calibration failure on 12/4/19 was due to the use of wrong calibration gas cylinders. Data was invalidated from 12/4/19 starting at the beginning of the failed daily calibration, until the completion of the respective successful calibration. Training is currently in progress for CEMS technicians to minimize unavailability incidents. Unit 3 is currently in an outage so no unavailable data is reported.	Units 4 and 5		calibration gas bottles wrong type
12/16/2019	2019 INTERNAL ENVIRONMENTAL AUDIT FINDINGS FOR PBREF-1 AND PBREF-2: 1. Three year opacity Zero Alignment check on Continuous Monitor Systems (COMS) may not have been performed in 2018 2. Reset PBREF No. 2 SO_2 stack monitor range from 500 ppm to 200 ppm. 3. ... after a 4x Out of Control (OOC) daily calibration drift (CD) check failure, a portion of the data was not properly invalidated and reported.	Units 3, 4, and 5		VIOLATION, see NONCOMPLIANCE 1/3/2020
12/20/2019	Unit 3 experienced two significant CO spikes on 12/19/2019 at 01:01 and 02:03. During both events, the CRO made several air adjustments, followed by firing gas guns. The actions taken by the CRO was able to reduce CO to normal operating levels following both spike events. Upon further investigation, it was discovered that the 3A sifting conveyor water level transmitter was providing a false-high reading, indicating that the water level in the quench tank was full. During this period, the sifting conveyor water level was actually below normal operating levels, causing the water-air seal of the boiler to be intermittently lost. This loss of water-air seal caused cold tramp air to enter the boiler, producing cold furnace CO spikes.	Unit 3	CO	sifting conveyor water level transmitter
12/30/2019	Unit 3 experienced significant CO spikes on 12/27/2019 beginning at approximately 17:18. During this event, the CRO made several air adjustments, followed by firing gas guns. The actions taken by the CRO was able to reduce CO to normal operating levels following the spike event. Upon further investigation, it was discovered that the recently installed settling basin above-ground plumbing (which serves to replace the previous underground plumbing) had become plugged with material. This plug stopped the flow of supply water to the discharger quench tank, causing the water-air seal of the boiler to be intermittently lost. This loss of water-air seal caused cold tramp air to enter the boiler, producing cold furnace CO spikes.	Unit 3	CO	settling basin plugged

Date (link)	Summary	Unit	Pollutant	Notes
1/3/2020	Federally Reportable Violation - Procedural Violation - Continuous parameter monitoring report - Facility has already completed necessary corrective actions to return to compliance. - The violation does not meet one of the six criteria for HPV [High Priority Violation] classification.			
1/3/2020	On 1/2/20 starting at approximately 11:59, the Unit 4 CO monitor began reading negative values. The alarms on the monitor indicated “High AGC Intensity” and “Low Motor Speed”. This combination of alarms was not experienced previously at the facility. A third party vendor was consulted to assist facility staff in troubleshooting the unusual alarms. After several hours of unsuccessful troubleshooting, it was finally discovered that the chopper motor had seized, and needed replacement. The chopper motor, IR source, preamp board, were all replaced during the troubleshooting process. At approximately 19:05, a successful calibration was completed. Since no parts are considered major components of the CO monitor, a RATA is not necessary to be conducted. No alarms are presently active and no additional repairs are expected from this event.	Unit 4	CO	chopper motor failure
1/23/2020	The after a successful daily calibration on 1/22, the Unit 5 flue probe experienced a sudden drop in all measured values at approximately 07:00. Facility I&E staff began troubleshooting, and determined that the root cause was a faulty check valve in the calibration line which caused ambient air to leak into the probe. The check valve was replaced and all values returned to normal. All calibrations passed as of 1/23/2020.	Unit 5	CO	calibration line valve fault
1/24/2020	After a successful daily calibration on 1/23, the Unit 5 flue probe experienced a sudden drop in all measured values at approximately 07:00. Facility I&E staff began troubleshooting, and determined that the root cause was a faulty check valve in the calibration line which caused ambient air to leak into the probe. The check valve was replaced and all values returned to normal. All calibrations passed as of 1/24/2020.	Unit 5	CO	calibration line valve fault
1/27/2020	After a successful daily calibration on 1/25, the Unit 5 flue probe experienced a sudden drop in all measured values at approximately 07:00. This caused the CO corrected to 7%O2 value to drastically increase due to near-zero CO2%V values. Zero percent CO2 values is not realistic due to normal boiler operating conditions. Facility I&E staff conducted a calibration gas cycle, which restored normal flue gas flow. All values returned to normal.	Unit 5	CO	calibration line valve fault
2/3/2020	On 2/1/2020 at approximately 0700, the Unit 3 CO monitor has experienced a 4X calibration failure. In this event, the last known good data prior to successful calibration on 1/31/20 at approximately 0800. Data was invalidated from 1/31/20 at 0800 until the completion of a successful calibration on 2/1/20 at approximately 1000.	Unit 3	CO	calibration failure

Date (link)	Summary	Unit	Pollutant	Notes
2/10/2020	On 2/9/2020 at approximately 0700, the Unit 3 CO monitor has experienced a 4X calibration failure. In this event, the last known good data was prior to successful calibration on 2/8/20 at approximately 0700. Data was invalidated from 2/8/20 at 0700 until the completion of a successful calibration on 2/9/20 at approximately 0800. In addition, on 2/9/2020 at approximately 1600, the CO monitor experienced an IR source failure. The IR source was replaced, and a successful calibration was completed at approximately 1900.	Unit 3	CO	calibration failure
2/11/2020	On 2/10/2020 at approximately 08:00, the Unit 3 stack CO2 monitor experienced a sudden drop from normal operating values. During troubleshooting, it was discovered that this was due to an IR source failure. The IR source was replaced, and a successful calibration was completed at around 10:14. This event caused approximately 2 hours of unavailable data. Since the NOX and SO_2 ppmvd@7%O2 require CO2 data for the diluent correction calculation, the CO2 unavailable data also caused unavailable data for NOX and SO_2. It is not expected that any exceedances occurred for NOX or SO_2, since all uncorrected data indicated normal operating concentrations. In addition, on 2/10/2020 at approximately 11:00, the CO monitor temperature alarm was active. The temperature monitor was faulty and was replaced. Successful calibration was completed by 12:42. This event caused approximately 1 hour of unavailable data.	Unit 3	CO, NOx, SO_2	IR source failure, termperature alarm fault
2/11/2020	Successful calibrations after the 1st day of quarterly maintenance was completed at approximately 17:50. Shortly after, Unit 5 flue probe experienced unrealistic readings in all measured values at approximately 17:53. Upon troubleshooting, it was discovered that the flue probe contained a minor leak at the filter connection. The leak was repaired and a successful calibration was conducted at 21:08. The NOX and SO_2 inlet values were also affected by this event, however, these parameters are only used for process control purposes. All NOX and SO_2 limits were still within normal operating limits.	Unit 5	CO?, NOx SO_2	probe flue leak at filter connection
2/19/2020	At around 08:40, Unit 3 CO increased to abnormal levels despite the use of gas guns and air changes. It was suspected that the probe was unresponsive. Upon further troubleshooting, it was suspected that a calibration solenoid which controls the flow of calibration gas had malfunctioned, and the solenoid allowed leaking of calibration gas. The solenoid was replaced by 10:28 and the probe was returned to service. The NOX and SO_2 inlet values were also affected by this event, however, these parameters are only used for process control purposes. All NOX and SO_2 limits were still within normal operating limits.	Unit 3	CO, NOx, SO_2	calibration solenoid malfunction

Date (link)	Summary	Unit	Pollutant	Notes
2/24/2020	On 2/19/20 starting at approximately 06:35, the Unit 4 CO monitor began reading elevated CO values. All combustion controls and indicators showed normal operation. The use of gas guns and air adjustments had no affect on the elevated CO. Upon investigation, it was discovered that a pattern of high CO spikes were consistent with durations of Unit 4 stack daily calibrations. This is suspected to be caused by plugged fast loop orifices, causing calibration gas to remain in the lines for an extended period of time, affecting actual measured values. Replacement of the fastloop on 2/20/20 caused a significant decrease in spikes observed, and was confirmed as a root cause solution of the CO spike.	Unit 4	CO	fast loop orifices plugged
4/21/2020	The opacity values for Unit 4 on 4/20/2020 was unavailable during window cleaning for the period of 08:00–08:06 and monitor troubleshooting during the period of 09:42- 11:00. The opacity data was also invalid during the period of 09:00 to 09:42 due to the finishing heat exchanger leak. The leak contributed to higher than normal levels of condensation in the process stream, creating condensate on the optical lenses of the Durag opacity monitor. It was previously believed on 4/19 that the heat exchanger was isolated by closing the inlet valve, however, it was discovered during the event on 4/20 that water was being back fed to the heat exchanger through the outlet. The outlet valve was closed at approximately 10:00 to stop the supply of back-fed water. The opacity values decreased to normal operating opacity values and remained within acceptable values for the remainder of the day.	Unit 4	opacity	finishing heat exchanger leak
6/15/2020	On June 12, 2020, there was a SO_2 exceedance on Boiler 3. It was determined that the lime supply valve tripped on low SDA outlet temperature and stopped the lime slurry delivery to the SDA, resulting in the SO_2 24- hour geometric mean to exceed the permit limit. The SDA outlet temperature was low because the boiler started up at 11:00 am from a planned outage.	Unit 3	SO_2	lime supply valve tripped due to low temps
6/15/2020	On June 13, 2020, there was a SO_2 exceedance on Boiler 5. It was determined that the lime supply valve tripped on low SDA outlet temperature and stopped the lime slurry delivery to the SDA, resulting in the SO_2 24- hour geometric mean to exceed the permit limit. The SDA outlet temperature was low because the boiler started up at 18:00 from a planned outage. Additionally, the low temperatures also caused the SCR ammonia injection system was to trip resulting in high NOx spikes.	Unit 5	SO_2, NOx	lime supply valve & SCR ammonia injection system tripped due to low temps

Date (link)	Summary	Unit	Pollutant	Notes
6/16/2020	On 6/14/20 at approximately 05:30, the Unit 5 SO_2 monitor experienced a 4X calibration failure. In this event, the last known good data was prior to successful calibration on 6/13/20 at approximately 0530. Data was invalidated from 6/13/20 at 0530 until the completion of a successful calibration on 6/14/20 at approximately 0740. Note that the boiler was not in operation prior to 6/13/20 at 18:00. An exceedance was noted during this time on 6/13/20. Please refer to the submitted notification for additional details. No exceedance is expected for the time period on 6/14/20, as all combustion and air pollution control systems were in stable and good operating condition during this time.	Unit 5	SO_2	calibration failure
6/17/2020	As a follow-up to the U5 CO invalid data on 6/14/2020, it was discovered that the recently installed fast-loop orifice contained a manufacturing defect. The fast-loop orifice is a component which cycles fresh stack gas through the CEMS probe quickly in order to achieve fast response times. The defective fast-loop orifice was replaced, and symptoms experienced on 6/14/2020 were resolved. The system was in maintenance mode on 6/16/2020 from around 0800 to 1200 during troubleshooting and replacement of the fast loop orifice, causing 4 hours of unavailable data.	Unit 5	CO	fast -loop orifice defective
6/26/2020	On 6/25/2019 at approximately 17:43, Unit 4 SDA lime supply valve tripped on low outlet temperature which stopped the lime slurry delivery to the SDA. The trip caused SO_2 to become elevated, and caused the SO_2 ppm average to be 60.5 ppm for the 17:00 hour. In order to restore lime slurry flow, the slurry supply valve was forced open by the control room operator by putting the valve in manual control mode and opening the valve. Upon further review, it was determined that the root cause of the trip was due to the low SDA temperatures (<270 degrees F) caused by the continuously high slurry flow in response to prolonged high inlet SO_2 conditions (normally <150 ppm, recently >300ppm). High slurry flows caused the SDA temperature to decrease steadily over time, which caused the slurry supply valve to trip due to low outlet SDA temperature. One of the main reasons for the low temperature supply valve trip is due to equipment safety to protect the downstream air pollution control equipment. When the slurry valve was locked out, the SDA temperature returned to safe operating temperatures, which enabled the control room operator to allow the supply valve to openly flow slurry again.	Unit 4	SO_2	lime supply valve tripped due to high and low temps
8/3/2020	On 8/1/20 at approximately 07:21, the Unit 4 CO monitor experienced a 4X calibration failure. In this event, the last known good data was prior to successful calibration on 7/31/20 at approximately 0721. Data was invalidated from 7/31/20 at 0800 until the completion of a successful calibration on 8/1/20 at approximately 0744.	Unit 4	CO	calibration failure

Date (link)	Summary	Unit	Pollutant	Notes
9/21/2020	On 9/20/2020, Boiler 4 experienced higher than normal CO values around 05:18. The cause was determined to be tramp air entering the feed chute as a result of blockage above the feed pushers. To clear the blockage, a feed pusher was activated. However, a failed feed pusher cylinder caused the feed pusher to be stuck in place, exacerbating the tramp air. The tramp air caused a decrease in furnace temperatures (1425F to 1246F), which led to high CO values (311.5 hourly average for the 0500 hour).	Unit 4	CO	feed pusher cylinder failure
12/14/2020	The Boiler 3B gas gun tripped (Malfunctioned) during high CO spikes (~2800ppm) on 12/12/2020. Gas guns are often used to assist furnace combustion during unstable or high CO incidents. The unavailability of 3B gas gun caused the 4-hr block to be in exceedance during this 4-hr block period. When gas guns were restored at 05:37, CO levels returned to normal but the 4-hr block average was already unrecoverable. The above boiler malfunction caused one, four hour block average CO period to exceed the permit limit of 100 ppmvd. The CO highest 4 hour block during operation is 95.1 ppmvd	Unit 3	CO	gas gun trip
12/28/2020	Unit 3 experienced significant CO spikes on 12/25/2020 beginning at approximately 04:23. During this event, the CRO made several air adjustments, steam flow reduction, followed by attempts to fire gas guns. Multiple attempts of lighting gas guns were not successful. The actions taken by the CRO was able to reduce CO to normal operating levels following the spike events at approximately 05:37. Upon further investigation, it was discovered that the 3A and 3B gas gun airflow dampers had malfunctioned during this CO event. Specifically, under automatic control, the gas gun dampers failed closed, and had to be forced from the closed position in automatic mode to the open position in manual mode. Since use of the gas guns require the damper controls to be in automatic mode and in open position, this also prevented the use of gas guns to control the CO event. The loss of supplemental airflow to the upper portion of the furnace in combination with the inability to operate the gas guns caused high CO during this period.	Unit 3	CO	gas gun trip - dampers failed in closed position

Date (link)	Summary	Unit	Pollutant	Notes
1/4/2021	The daily average for Unit 4 CO, NOX, SO_2 lb/hr was invalid due to excessive drift in the flow monitor. Unit 4 flow monitor shows stack gas velocities which are unreasonable given ID fan speeds and operational knowledge. The induced draft (ID) fan of each unit controls the overall draft and velocity of flue gas through the ductwork. During the period of 1/02/21 through 1/04/21, the stack gas velocity is shown to be approximately 10 feet per second (15%) higher than either Units 3 or 5. This is unlikely, given that the steam load, combustion, and operation of the ID fan is similar across all units. Although the flow monitor successfully passed a daily automatic calibration every day throughout the event, this calibration is an 'electronic' calibration that only verifies the electrical response/health of the unit. The flow monitor was cleaned and manually recalibrated on 1/4/2021 at approximately 10:30. Currently, Unit 4 flow measures normal exhaust flue gas velocities (<50 feet per second). The attached data shows normal CO, NOx, and SO_2 ppmvd hourly averages, and the invalidated abnormal lb/hr hourly data. It is not expected that there was any exceedance during this time since the CO, NOX and SO_2 ppmvd averages were within permit limits and not affected by stack gas flow calculations.	Unit 4	CO, NOx, SO_2	flow monitor dirty
1/11/2021	The Boiler 3 ID (induced draft) fan tripped (Malfunctioned) due to variable frequency drive (VFD) fault alarm at approximately 04:58 on 1/9/2021. This malfunction caused Boiler 3 to experience a master fuel trip (MFT). Due to the loss of air, gas guns could not be ignited during the boiler shutdown. The control room operator successfully restarted the ID fan approximately 15 minutes after the initial fan trip (fan requires a 15 minute wait time prior to start). Combustion was stable at 05:24. The above boiler malfunction caused one, four hour block average CO period to exceed the permit limit of 100 ppmvd. The above exceedance is within the allowed fifteen hours per occurrence for CO as per section B.17 of the Title V Air Operating Permit, number 0990234-041-AV.	Unit 3	CO	ID fan trip
1/20/2021	Upon experiencing vibration issues with the PBREF2 Ash Management Building Wet Whirl Blower, a vibration test was conducted today on 1/19/21. Following the results of vibration testing at around 11:00am, it has been determined that the blower must be removed from service to prevent further equipment damage until repairs can be made. At this time, it is currently anticipated that repairs will be completed in 1-2 days upon receipt of replacement parts (currently, delivery is expected 1/20/21). Based on current information, normal operation of the wet whirl blower should resume by 1/22/21.	Wet Whirl Blower	ash	blower vibration

Date (link)	Summary	Unit	Pollutant	Notes
1/25/2021	The Boiler 3 primary air damper on Zone 4 right experienced an unknown control logic error which caused the damper to close shut. This malfunction caused Boiler 3 to experience a disturbance to combustion air, which was the leading factor to the CO spike event on 1/22/2021 at around 09:00. Following the CO spike, the CRO made several air adjustments and inserted gas guns to stabilize combustion. After combustion stabilized the unit was brought offline for additional investigations, but did not yield any immediate findings. The boiler was brought back to operation, and no exceedance has occurred since, as of 1/25/2021. Further investigation is on-going to correct the Zone 4 primary air damper control logic, including instrumentation checks which could affect process controls. The above air damper malfunction caused one, four hour block average CO period to exceed the permit limit of 100 ppmvd. The above exceedance is within the allowed three hours per occurrence as per section B.17 of the Title V Air Operating Permit, number 0990234-041-AV.	Unit 3	CO	air damper control logic error
1/26/2021	The Boiler 5 ID (induced draft) fan tripped (Malfunctioned) due to variable frequency drive (VFD) temperature alarm at approximately 21:33 on 1/25/2021. This malfunction caused Boiler 5 to experience a master fuel trip (MFT). Due to the loss of air, gas guns could not be ignited during the boiler shutdown. The control room operator successfully restarted the ID fan approximately 15 minutes after the initial fan trip (fan requires a 15 minute wait time prior to start). Combustion was stable at 22:03. The above ID fan malfunction caused one, four hour block average CO period to exceed the permit limit of 100 ppmvd. The above exceedance is within the allowed three hours per occurrence as per section B.17 of the Title V Air Operating Permit, number 0990234-041-AV.	Unit 5	CO	ID fan trip - temp alarm

Date (link)	Summary	Unit	Pollutant	Notes
1/29/2021	On 1/27/2021, Boiler 3 experienced several CO spike events beginning at 16:09. Upon experiencing the CO spikes, the CRO made several air adjustments and increased the O2 setpoint. Gas guns were inserted to stabilize combustion at 18:16. Stable combustion was achieved at 19:58. Upon investigation, it is believed that a hole in the 3B discharger was allowing tramp air to enter the boiler. The tramp air is believed to be the primary root cause of the CO exceedance. A temporary increased O2 setpoint was maintained to control CO until the boiler was shutdown so repairs could be made on the discharger. The boiler was shutdown on 1/29/2021 at 07:45. Preparation for the discharger repair is currently underway as of 1/29/2021, and is anticipated to be fully completed during this outage. The above discharger malfunction caused one, four hour block average CO period to exceed the permit limit of 100 ppmvd. The above exceedance is within the allowed three hours per occurrence as per section B.17 of the Title V Air Operating Permit, number 0990234-041-AV.	Unit 3	CO	discharger hole
2/11/2021	On 2/10/2021 at approximately 06:40 the Unit 5 CO flue experienced a calibration failure. This invalidated data from the previous successful calibration on 2/9/2021 at 13:05 to the successful recalibration on 2/10/2021 at 09:50. It was discovered that the calibration failure was due to a malfunctioned IR source. The system was in maintenance on 2/10/2021 from around 0800 to 0950 during this time until IR source was replaced and successful calibration was completed. It is not expected that there was any exceedance during this time since all air pollution control equipment was in good operating condition and stable combustion of the boiler was maintained with supplemental natural gas as needed.	Unit 5	CO	IR source failure
2/22/2021	On 2/20/2021 at approximately 07:00 the PBREF2 Unit 5 CO, NOx, and SO_2 experienced unavailable data due to a PLC error which prevented the daily calibration from initiating. At around 0700, the daily calibration did not complete successfully. I&E was notified, and troubleshooting began. At around 11:18, the PLC was identified as the cause of the unsuccessful calibration, and a restart of the PLC was completed. The PLC restart resolved the issue, and allowed the daily calibration to be successfully conducted. After the PLC restart, the daily calibration began 11:22 and was completed at around 12:00. Data was unavailable for the period between the normal start time of the daily calibration (around 0700) and the completion of the successful passing calibration (around 1200). It is not expected that there was any exceedance during this time since all air pollution control equipment was in good operating condition and stable combustion of the boiler was maintained.	Unit 5	CO, NOx, SO_2	PLC error

Date (link)	Summary	Unit	Pollutant	Notes
3/18/2021	On 3/17/2021, Boiler 3 experienced a Zone 2 left side sifting hopper plug around 15:00. The plugged material led to grates piling, causing poor combustion. Operations began to clear the plug immediately. The poor combustion caused a large CO spike event at 15:28. Upon experiencing the CO spike, the CRO made several primary air adjustments and secondary air adjustments. After sufficient air adjustments and the clearing of the hopper plug was completed, stable combustion was achieved around 16:30. The above sifting hopper plug caused one, four hour block average CO period to exceed the permit limit of 100 ppmvd. The above exceedance is within the allowed three hours per occurrence as per section B.17 of the Title V Air Operating Permit, number 0990234-041-AV.	Unit 3	CO	hopper plug
5/21/2021	Notification that the Palm Beach Renewable Energy Facility #2 EHS specialist indicated that there was a 24 hour NOx exceedance on Unit 4 on May 20, 2021 upon start up after an outage. Covanta staff is currently investing the cause of this exceedance. A detailed report will be provided once the investigation is complete. **NOTE: REPORT NOT IN FILE**	Unit 4	NOx	
6/1/2021	On 5/28/2021 the PBREF2 Unit 5 NOx and SO_2 experienced unavailable data from 0900 to 2300. At approximately 08:56, PBREF2 Unit 5 CEMS were placed in maintenance mode due to a high cooler temperature alarm on the NOx analyzer. I&E reached out to tech support and began troubleshooting the issue. Following tech support guidance, I&E replaced the sample cooler but then experienced a high-pressure alarm (~660mmHg). A service call was initiated at approximately 17:00, the technician arrived onsite at 21:45. The NOx pump was swapped with the spare after finding the diaphragm badly worn. Capillary O-rings and tightness of fittings were also checked. Pump pressure dropped to 230mmHg. After confirming parameters and taking the unit off maintenance mode, a successful calibration was performed.	Unit 5	NOx, SO_2	NOx pump diaphragm worn out
6/28/2021	On 6/27/2021 the PBREF2 Unit 5 NOx experienced a 4X out of control calibration at approximately 8:44 am. Per part 60, the data is flagged as invalid to the previous valid calibration on 6/26/2021 at approximately 6:06 am. Following the failed calibration on 6/27/2021, I&E placed the monitor in maintenance mode and began to troubleshoot the issue. I&E found a plugged sample line. Once the line was cleaned, a successful calibration was completed on the monitor, and returned to service. This caused unavailable data for a total of 14 hours for the NOx U5. (Unit 5 Boiler was offline from 6/26/2021 12:00 to 6/27/2021 04:00.)	Unit 5	NOx	sample line plugged

Date (link)	Summary	Unit	Pollutant	Notes
7/21/2021	Upon completion and review of the second quarter 2021 report, it was noted that PBREF2 Unit 5 SO_2 experienced one, 24-hour geometric mean period to exceed the permit limit of 24 ppmvd and 25 lb/hr. Due to the previously reported CEMS Malfunction (sample cooler malfunction) on 5/28/2021 which caused unavailable data for 15 hours, the short averaging period for SO_2 registered a 24.8 ppmvd and 28.8 lb/hr when using the available 9 hours of data. However, on 6/3/2021 the Unit 5 CEMS experienced a 4X out-of-control calibration due to a broken solenoid valve which resulted in an unavailable data notification to FDEP. It is believed that until the solenoid valve was replaced on 6/3/2021, the failing solenoid valve allowed the calibration gas to remain in the line producing artificially high SO_2 values. Therefore, when the data is excluded for the calibration period 0500 to 0700 on 5/28/2021, the new SO_2 24-hour geometric means are 21.5 ppm and 25.0 lb/hr. Unit 5 was operating normally during this time, and the SO_2 setpoint was reduced to 12 ppm for the remainder of the day. Had the CEMS monitor been available for the entire 24-hour averaging period and not operated with a faulty solenoid valve, it is expected the SO_2 would have remained below the permit limit.	Unit 5	SO_2	solenoid valve
8/3/2021	On 07/24/2021 at approximately 01:08, the Control Room received an alarm that the PBREF2 Unit 3 CO flue CEM experienced a hardware failure. Unit 3 Boiler remained operational with good combustion practices. When the I & E technicians arrived for their shift, at approximately 07:00, diagnosis of the CO monitoring system was initiated, and the analyzer was placed in Maintenance Mode at approximately 08:05. After trouble shooting the analyzer, the vacuum pump was repaired, and the correlation motor and the IR source were replaced. The analyzer was removed from Maintenance Mode at approximately 13:24 on 07/24/2021. Due to the monitor equipment failures, the calibration subsequently failed at 06:21 – 06:52 on 7/24/2021, thus data is flagged as invalid to the previous valid calibration at 06:21 – 06:52 on 7/23/2021 per 40 CFR Part 60, Appendix F. A calibration was completed successfully at 14:02 on 7/24/2021. Therefore, the CO data was unavailable for the period between 07/23/2021 (06:00 – 23:00) – 07/24/2021 (00:00 – 13:00) for a total of 32 hours. The repairs do not require a Performance Specification 4 recertification of the analyzer. It is not expected that there was any exceedance during this time since all air pollution control equipment was in good operating condition and stable combustion of the boiler was maintained.	Unit 3	CO	vacuum pump failure

Date (link)	Summary	Unit	Pollutant	Notes
9/20/2021	Notification that the Palm Beach Renewable Energy Facility #2 Regional Environmental Specialists indicated that September 17, 2021, there may have been a temperature exceedance on Unit 3 . The plant operator is currently investing this potential temperature exceedance. A report will be provided once the investigation is complete. **NOTE: REPORT NOT IN FILE**	Unit 3	temperature	
9/22/2021	At 1140 hrs on 9/20/21, Boiler 5 experienced an ash discharger chute plug. The unit was shut down to clear the plug. At 1701 hrs the plug was cleared, and the control room operator (CRO) began feeding MSW to the boiler. Combustion was stable at 1734 hrs. There were not any permit exceedances during the period of Boiler shutdown and startup. At 2020 hrs, Boiler 5 experienced a feed chute plug, which caused loss of fuel feed to the boiler. This malfunction caused unstable combustion resulting in a CO spike. The CRO ignited the auxiliary gas burners at 2045 hrs in an effort to control the CO spike. Personnel cleared the feed chute plug and the CRO began feeding MSW at 2212 hrs. Combustion stabilized and the auxiliary gas burners were removed at 0005 hrs on 9/21/2021. This malfunction caused one 4-hour block average CO period to exceed the permit limit of 100 ppmvd. The above exceedance is within the allowed three hours per occurrence as per section B.17 of the Title V Air Operating Permit, number 0990234-043-AV.	Unit 5	ash, CO	ash discharger chute plug, feed chute plug
10/12/2021	On 10/11/2021, upon completion of the daily calibration at 06:52, the PBREF2 Unit 3 CO CEMS recorded elevated CO values due to calibration gas remaining in the system. The calibration gas did not fully purge the system. Investigation was started around 07:00, and the unit 3 inlet was placed in maintenance mode to trouble shoot the elevated CO at 08:00. The investigation found that the inlet CO monitor probe dilution block (which includes an eductor nozzle and critical orifice) and the dilution bypass block (which includes an eductor nozzle) were plugged. The repairs consisted of replacing both blocks allowing the gas to flow freely to the analyzer. Data was unavailable for the period between the end of the calibration at 06:53 and completion of repairs at 10:37. It is not expected that there was any exceedance during this time since all air pollution control equipment was in good operating condition and stable combustion of the boiler was maintained.	Unit 3	CO	probe inlet blocked

Date (link)	Summary	Unit	Pollutant	Notes
10/25/2021	Upon completion and review of the third quarter report, it was discovered that PBREF2 Unit 5 experienced 2 incidents of unavailable data that were not identified. On September 26 the U5 CO, SO_2, and NOx CEMS experienced a 4X calibration failure invalidating data back to the previous passing calibration on September 25. Upon investigation, I&E discovered there was a PLC output card failure. A new card was installed on September 26 and the monitors returned to service. The CEMS was successfully calibrated at 1400 Hrs (SO_2 and NOx)and 1710 Hrs (CO). On September 30, the U5 CO, SO_2, and NOx CEMS experienced a 4X calibration invalidating data back to the previous passing calibration on September 29. Upon investigation, I&E found the computer PLC had experienced a computer error. The computer was rebooted, and the monitors were returned to service. The CEMS system was successfully calibrated at 0939 Hrs. The Unit 5 was shutdown at 0800 Hrs and startup commenced at 1700 Hrs on September 30 due to a grate cooling water valve failure.	Unit 5	CO, NOx, SO_2	PLC card failure
11/15/2021	On 11/14/2021 at approximately 1831 Hrs, the Unit 3 Inlet monitors experienced an IR source failure, and the monitor was placed into maintenance mode at approximately 2115 Hrs. The IR source was replaced, and the system was removed from maintenance mode at approximately 2134 Hrs. The inlet CO2, CO, NOx and SO_2 data was affected during this time period. This event resulted in 2 hours of unavailable data. It is not expected that there was any exceedance during this time since all air pollution control equipment was in good operating condition and stable combustion of the boiler was maintained with supplemental natural gas as needed.	Unit 3	CO, CO2, NOx, SO_2	IR source failure
12/17/2021	On 12/16/2021, the PBREF2 Unit 3 SO_2 experienced a 4X out-of-control calibration at 05:37 hours. Per part 60, the data is flagged as invalid to the previous valid calibration, which occurred on 12/15/2021 at approximately 10:51 hours. Following the failed calibration on 12/16/2021, I&E placed the monitor in maintenance mode and confirmed that the SO_2 solenoid was plugged and there was a failed SO_2 pump. The repairs were completed, and a successful calibration was performed and the SO_2 monitor was placed back into service at 09:26 on 12/16/2021.This caused invalid data for a total of 22 hours for SO_2 on Unit 3.	Unit 3	SO_2	solenoid plugged, pump failure

Date (link)	Summary	Unit	Pollutant	Notes
1/14/2022	On 1/14/2022, the PBREF2 Unit 3 SO_2 experienced a 4X out-of-control calibration at 0537 hours. Per Part 60, the data is flagged as invalid to the previous valid calibration, which occurred on 1/13/2022 at approximately 0537 hours. Following the failed calibration on 12/14/2022, I&E placed the monitor in maintenance mode for the monthly PM. After confirming parameters and taking the unit off maintenance mode, a successful calibration was performed at 1010 hours. A total of 28 hours of data is excluded.	Unit 3	SO_2	calibration failure
2/3/2022	On 2/2/2022, the PBREF2 Unit 4 CO2 experienced a 4X out of control calibration at approximately 06:38 hours affecting the corrected NOx and Corrected SO_2 values. Per part 60, the data is flagged as invalid to the previous valid calibration, which occurred on 2/1/2022 at approximately 06:38 hours. The source of the calibration failure could not be determined; however, the monitor was re-calibrated successfully at 08:11 hours. This occurrence resulted in 25 hours of unavailable data for NOx and SO_2 on Unit 4. The attached data shows the invalidated NOx, and SO_2 ppmvd hourly averages. It is not expected that there was any exceedance during this time since all air pollution control equipment was in good operating condition and stable combustion of the boiler was maintained.	Unit 4	CO, CO2, NOx, SO_2	calibration failure
2/9/2022	On 2/8/2022 at approximately 07:21 Hrs, the PBREF2 Unit 4 inlet CO monitor experienced a 4X out of control calibration. Per 40 CFR Part 60, the data is flagged as invalid to the previous valid calibration at 06:37 Hrs on 8/7/2022. Following the failed calibration on 2/8/2022, I&E placed the analyzer in Maintenance Mode at 07:30 Hrs, and determined that the analyzer had a vacuum pump failure and an IR source failure. Both components were replaced and the system was removed from Maintenance Mode at approximately 08:49 Hrs. The analyzer was successfully calibrated at 09:12 Hrs. The CO data was unavailable for a total of 26 hours. The repairs do not require a Performance Specification recertification of the analyzer. It is not expected that there was any exceedance during this time since all air pollution control equipment was in good operating condition and stable combustion of the boiler was maintained.	Unit 4	CO	IR source failure, vacuum pump failure

Date (link)	Summary	Unit	Pollutant	Notes
2/11/2022	On 2/9/2022, the PBREF2 Unit 4 NOx experienced a 4X out of control calibration at approximately 06:38 hours affecting the corrected NOx values. Per part 60, the data is flagged as invalid to the previous valid calibration, which occurred on 2/8/2022 at approximately 06:38 hours. The source of the calibration failure could not be determined; however, the monitor was re-calibrated successfully at 07:57 hours. This occurrence resulted in 19 hours of unavailable data for NOx on Unit 4 because the unit experienced a controlled shutdown at 01:39 Hrs on 2/9/2022 to clear a discharger plug. The controlled shutdown did not result in any permit limit exceedances. The attached data shows the invalidated NOx ppmvd hourly averages. It is not expected that there was any exceedance during this time since all air pollution control equipment was in good operating condition and stable combustion of the boiler was maintained during the period of operation.	Unit 4	NOx	calibration failure; discharger plugged
2/14/2022	On 2/13/2022, the PBREF2 Unit 4 CO2 experienced a 4X out of control calibration at approximately 06:38 hours affecting the corrected NOx and corrected SO_2 values. Per part 60, the data is flagged as invalid to the previous valid calibration, which occurred on 2/12/2022 at approximately 06:37 hours. The source of the calibration failure could not be determined; however, the monitor was re-calibrated successfully at 09:48 hours. This occurrence resulted in 26 hours of unavailable data for NOx and SO_2 on Unit 4. Unit 4 was shutdown at 11:09 hrs for an ash conveyor drive chain failure. Repairs were completed, and startup was initiated at 14:22 hours. The attached data shows the invalidated NOx, and SO_2 ppmvd hourly averages. It is not expected that there was any exceedance during this time since all air pollution control equipment was in good operating condition and stable combustion of the boiler was maintained.	Unit 4	CO2, CO, SO_2, NOx; ash	calibration failure; conveyor drive chain failure
2/14/2022	The Unit 4 experienced one 6-minute block average above the permit limit of 10% during the malfunction event. The opacity spike resulted in a 6-minute block average of 32.4%. The malfunction event was caused by failure of one or more bags in the Unit 4-D baghouse compartment. The compartment was isolated, and operational controls were adjusted to prevent recurrence of the elevated opacity reading. The defective bags will be evaluated and replaced, as needed. The above exceedance is within the allowed three hours per occurrence as per section III.B.17 of the Title V Air Operating Permit, number 0990234-043-AV.	Unit 4	opacity	baghouse failure

Date (link)	Summary	Unit	Pollutant	Notes
2/24/2022	On 2/22/2022, the PBREF2 Unit 4 NOx and SO_2 monitors experienced a 4X out of control calibration at approximately 06:38 hours. Per part 60, the data is flagged as invalid to the previous valid calibration, which occurred on 2/21/2022 at approximately 06:37 hours. The source of the calibration failure could not be determined; however, the monitor was re-calibrated successfully on 2/22/2022 at 07:51 Hrs for NOx and 08:22 Hrs for SO_2. This occurrence resulted in 25 hours and 26 hours of unavailable data for NOx and SO_2 on Unit 4, respectively. On 2/22/2022, a conference call was initiated with CEMTEK and corporate CEMS experts to discuss possible causes for the frequent calibration failures. CEMTEK recommended change out of the critical orifice, the turbo-valve (fast loop valve) and also a pressure test of the umbilical. Following the call, I&E cleaned the critical orifice and the turbo-valve. These corrections were put in place to eliminate fluctuations in dilution gas pressures during the daily calibrations. On 2/23/2022, the PBREF 2 Unit 4 SO_2 and NOx monitors experienced another 4X out of control calibration at approximately 06:38 hrs. This time the critical orifice and turbo-valve were replaced, and the CO2 scrubbing towers and gas line were cleaned. Following these corrective actions, a successful calibration was completed at 11:31 on 2/23/2022. The data was flagged as invalid from the last failed calibration on 2/23/2022 to the previous calibration which occurred on 2/22/2022 at 07:51 Hrs for NOx and at 08:22 for SO_2. On 2/24/2022, CEMTEK analysts will be on site to perform a full evaluation of the CEMS hardware and software to determine additional potential causes of the frequent calibration errors. These occurrences resulted in 53 hours of unavailable data for NOx and SO_2. The attached data shows the invalidated NOx, and SO_2 ppmvd hourly averages. It is not expected that there was any exceedance during this time since all air pollution control equipment was in good operating condition and stable combustion of the boiler was maintained.	Unit 4	CO2, CO, SO_2, NOx; ash	orifice & turbovalve failure
2/24/2022	On 1/23/2022, the PBREF2 Unit 3 NOx monitor experienced a 4X out-of-control calibration at 0537 hours. Per Part 60, the data is flagged as invalid to the previous valid calibration, which occurred on 1/22/2022 at approximately 0537 hours. Following the failed calibration on 2/23/2022, I&E successfully completed a new calibration at 0814 hrs. A total of 26 hours of data is excluded. CEMTEK, will be onsite on 2/24/2022 to diagnose the CEMS hardware and software systems. It is not expected that there was any exceedance during this time since all air pollution control equipment was in good operating condition and stable combustion of the boiler was maintained.	Unit 3	NOx	calibration failure

Date (link)	Summary	Unit	Pollutant	Notes
2/28/2022	On 2/25/2022 at approximately 06:40 Hrs Unit 5 Inlet NOx monitor experienced a 4X calibration failure. On 2/25/2022 at 07:16 Hrs, the Unit 5 Inlet NOx, SO_2 and CO monitoring system was placed in maintenance mode for troubleshooting. I&E replaced a failed pump, worn permeation tube, fast loop orifice and cleaned the inlet filter. The Unit 5 inlet monitoring system was removed from maintenance mode at 12:30 Hrs on 2/25/2022, resulting in 4 hrs of unavailable data for CO during the maintenance period on 2/25/2022. It is not expected that there was any exceedance during this time since all air pollution control equipment was in good operating condition and stable combustion of the boiler was maintained with supplemental natural gas as needed.	Unit 5	CO, NOx, SO_2	calibration failure; pump failed, permeation tube & fast loop orifice worn out, inlet filter dirty
4/5/2022	On 4/4/2022, the PBREF2 Unit 4 SO_2 monitor experienced a 4X out-of-control calibration at approximately 0609 hours. Per Part 60, the data is flagged as invalid to the previous valid calibration, which occurred on 4/3/2022 at approximately 0609 hours. The source of the calibration failure could not be determined; however, the monitor was recalibrated successfully on 4/4/2022 at 0736 hours. This occurrence resulted in 24 hours of unavailable data. On 4/5/2022, the PBREF2 Unit 4 SO_2 monitor experienced another 4X out-of-control calibration at 0609 hours. Per Part 60, the data is flagged as invalid to the previous valid calibration, which occurred on 4/4/2022 at approximately 0736 hours. Following the failed calibration on 4/5/2022, I&E completed a successful calibration at 0749 hours, and then at 0912 hours, I&E placed the SO_2 monitor into maintenance mode for troubleshooting and a second calibration. The technicians determined that several solenoid valves associated with the SO_2 monitor were not functioning as required, which would have contributed to the failed calibration. The valves were repaired, and a successful calibration was completed at 1029 hours. This occurrence resulted in 25 hours of unavailable data. These two occurrences resulted in 49 hours. The attached data shows the invalidated SO_2 ppmvd hourly averages.It is not expected that there was any exceedance during this time since all air pollution control equipment was in good operating condition and stable combustion of the boiler was maintained.	Unit 4	SO_2	solenoid valves

Date (link)	Summary	Unit	Pollutant	Notes
4/15/2022	On 4/14/2022, the PBREF2 Unit 4 CO2, NOx, and SO_2 monitors experienced a 4X out-of-control calibration at approximately 0745 hours. Per Part 60, the data is flagged as invalid to the previous valid calibration, which occurred on 4/13/2022 at approximately 0745 hours. The monitors were placed into maintenance mode at 08:33 hours to troubleshoot the calibration issues. The fast loop and critical orifice on each monitor were changed out, and the monitors were removed from maintenance mode at approximately 11:27 hours. The Unit 4 monitors were successfully recalibrated at approximately 12:06 hours. This occurrence resulted in 28 hours of unavailable data. The attached data shows the invalidated NOx and SO_2 ppmvd hourly averages. It is not expected that there was any exceedance during this time since all air pollution control equipment was in good operating condition and stable combustion of the boiler was maintained	Unit 4	CO, CO2, NOx, SO_2	fast loop & critical orifices failed
4/19/2022	On 4/18/2022 at approximately 08:30 Hrs Unit 5 Inlet CO monitor experienced a 4X calibration failure. On 4/18/2022 at approximately 08:30Hrs, the Unit 5 Inlet CO monitoring system was placed in maintenance mode for troubleshooting. I&E replaced a failed I/O expander board. The Unit 5 inlet CO monitoring system was removed from maintenance mode at approximately 10:30 Hrs on 4/18/2022, and a successful calibration was completed at 10:59. Per Part 60, the data is flagged as invalid to the previous valid calibration, which occurred on 4/17/2022 at approximately 08:30 Hrs; however, the Unit 5 was shutdown through 19:04 Hrs in association with the turbine generator trip on 4/16/2022. This event resulted in 16 hrs of unavailable data for CO. The I/O expander board replacement will not require a Performance Specification 4 recertification of the analyzer. It is not expected that there was any exceedance during this time since all air pollution control equipment was in good operating condition and stable combustion of the boiler was maintained with supplemental natural gas as needed.	Unit 5	CO	I/O expander board failed
4/20/2022	On 4/19/2022, the PBREF2 Unit 3 NOx monitor experienced a 4X out-of-control calibration at approximately 0745 hours. Per Part 60, the data is flagged as invalid to the previous valid calibration, which occurred on 4/18/2022 at approximately 0730 hours for Unit 3. The Unit 3 monitor was placed into maintenance mode at 0917 hours. The Unit 3 monitor was removed from maintenance mode at 0930 hours and recalibrated. The Unit 3 NOx monitor was successfully calibrated at 0956 hours. The Unit 3 NOx occurrence resulted in 26 hours of unavailable data. The cause for the calibration failures was not determined. CEMTEK, CEMS contractor, was onsite at the time of the calibration failures and they are continuing to troubleshoot the CEMS system.	Unit 3	NOx	calibration failure

Date (link)	Summary	Unit	Pollutant	Notes
4/20/2022	On 4/19/2022, the PBREF2 Unit 4 NOx and SO_2 monitors experienced a 4X out-of-control calibration at approximately 0800 hours. Per Part 60, the data is flagged as invalid to the previous valid calibration, which occurred on 4/18/2022 at approximately 0800 hours for Unit 4. The Unit 4 monitors were placed into maintenance mode at 0917 hours. The Unit 4 monitors were removed from maintenance mode at 0930 hours and recalibrated. The Unit 4 SO_2 calibration was successfully calibrated at 1025 hours, but the Unit 4 NOx calibration failed again. The Unit 4 NOx monitor was placed back into maintenance mode at 1020 hours to troubleshoot the NOx calibration issues. The Unit 4 NOx monitor was removed from maintenance mode at 1046 hours and successfully recalibrated at approximately 1105 hours. The Unit 4 NOx occurrence resulted in 27 hours of unavailable data. The Unit 4 SO_2 occurrence resulted in 26 hours of unavailable data. The cause for the calibration failures was not determined. CEMTEK, CEMS contractor, was onsite at the time of the calibration failures and they are continuing to troubleshoot the CEMS system.	Unit 4	NOx, SO_2	calibration failure
4/21/2022	On 4/18/2022 and 4/19/2022, the PBREF2 Unit 5 CO monitor experienced a period of unavailable data during which the CO monitor was recording values of 0.0 ppmvd. Per Part 60, the data is flagged as invalid during this period of time. The Unit 5 CO monitor was successfully recalibrated on 4/19/2022 at 1718 hours. Following the recalibration, the Unit 5 CO values began recording at the expected values. The Unit 5 CO occurrence resulted in 21 hours of unavailable data. The cause for the unavailable data was not determined. CEMTEK, CEMS contractor, was onsite during this time period, and they are continuing to troubleshoot the CEMS system. The attached data shows the invalidated CO ppmvd hourly averages. It is not expected that there was any exceedance during this time since all air pollution control equipment was in good operating condition and stable combustion of the boiler was maintained.	Unit 5	CO	calibration failure
5/4/2022	On 5/3/2022 at 09:00 Unit 4 SO_2 and NOx monitors experienced a 4X out of control (OOC) calibration failure. The monitors completed a successful recalibrated at 10:57 on 5/3/2022. However, due to the 4X OOC, per 40 CFR, Appendix F, data is flagged as invalid back to the previous valid calibration at 09:28 on 5/2/2022. Therefore, making a total of 25 hours of SO_2 and NOx data unavailable for 5/2/2022 through 5/3/2022. I&E continues to investigate the cause of the frequent monitor OOC episodes along with the facility CEM contractor, CEMTek. CEMTek is currently onsite troubleshooting the monitors and performing quarterly preventative maintenance activities. It is not expected that there was any exceedance during this time since all air pollution control equipment was in good operating condition and stable combustion of the boiler was maintained.	Unit 4	NOx, SO_2	calibration failure

Date (link)	Summary	Unit	Pollutant	Notes
5/16/2022	Unit 3 SO_2 lb/hr on 5/13/2022 was elevated above the permitted limit of 25.0 lb/hr @ 7% O2 for the for the 00:00 – 23:00 timeframe. The data is showing a SO_2 24-hour geometric mean of 27.7 lb/hr @ 7% O2. The facility is investigating the cause of the high 24-hour SO_2 geometric mean lb/hr daily reading since the SO_2 daily ppm @ 7% O2 remained beneath its permitted limit at 20 ppm @ 7% O2. Upon comparing SO_2 values for Unit 5 on the same day, the Unit 3 SO_2 lb/hr data appears suspect. The facility is reviewing the lb/hr calculations and placed a request with CEMTek (the NetDAHS contractor) to do the same. At the same time, the facility is investigating other potential causes.	Unit 3	SO_2	
5/18/2022	Units 3 and 5 experienced elevated NOx and SO_2 readings throughout the day on 4/15/2022 and 4/16/2022. This caused the 24-hour arithmetic average for NOx to go above the permitted limit of 50 ppmvd @ 7% O2 for the 00:00-23:00 timeframe for Unit 3 on 4/16/2022 and Unit 5 on 4/15/2022 & 4/16/2022. In addition, the 24-hour geometric mean for SO_2 went above the permitted limit of 24 ppmvd @ 7% O2 for the 00:00-23:00 timeframe for Unit 3 on 4/16/2022 and Unit 5 on 4/15/2022. The 4/16/2022 emission spikes on U3 resulted in a NOx 24-hour average of 65 ppmvd @ 7% O2 and SO_2 24-hour geometric mean of 30 ppmvd @ 7% O2. The 4/15/2022 emission spikes on U5 resulted in a NOx 24-hour average of 53 ppmvd @ 7% O2 and SO_2 24-hour geometric mean of 25 ppmvd @7% O2, and on 4/16/2022 a NOx 24-hour average of 90 ppmvd @ 7% O2.	Unit 3, 5	NOx, SO_2	steam pressure control valve malfunction
5/18/2022	Unit 5 Inlet CO was unavailable for 4 hours on 4/15/2022 from 20:00 – 23:00 due to the monitor reading approximate zero values during this timeframe. A faulty I/O expander board can cause erratic Inlet CO readings, such as this, and reset to normal readings, as seen on 4/15/2022. As previously reported on 5/18/2022 Unit 5 Inlet CO monitor I/O expander board was replaced, which includes the optics and collection chamber that performs the analyzing. The I/O expander board replacement does not require a Performance Specification 4 recertification of the analyzer. It is not expected that there was any exceedance during this time since all air pollution control equipment was in good operating condition and stable combustion of the boiler was maintained.	Unit 5	CO	I/O expander board fault

5/18/2022	Units 3 and 5 experienced elevated NOx and SO_2 readings throughout the day on 4/15/2022 and 4/16/2022. This caused the 24-hour arithmetic average for NOx to go above the permitted limit of 50 ppmvd @ 7% O2 for the 00:00-23:00 timeframe for Unit 3 on 4/16/2022 and Unit 5 on 4/15/2022 & 4/16/2022. In addition, the 24-hour geometric mean for SO_2 went above the permitted limit of 24 ppmvd @ 7% O2 for the 00:00-23:00 timeframe for Unit 3 on 4/16/2022 and Unit 5 on 4/15/2022. The 4/16/2022 emission spikes on U3 resulted in a NOx 24-hour average of 65 ppmvd @ 7% O2 and SO_2 24-hour geometric mean of 30 ppmvd @ 7% O2. The 4/15/2022 emission spikes on U5 resulted in a NOx 24-hour average of 53 ppmvd @ 7% O2 and SO_2 24-hour geometric mean of 25 ppmvd @7% O2, and on 4/16/2022 a NOx 24-hour average of 90 ppmvd @ 7% O2. In reviewing the operational data, trends of the 150-pound steam pressure control valve can be seen fluctuating throughout the day on 4/15/2022 and progressively worsening. Two significant pressure drops occurred at 05:30 and 11:00 on 4/15/2022. This malfunctioning steam valve impacted the steam supply to the heat exchanger which allows the aqueous ammonia mixture to maintain a temperature above the dew point. As the ammonia vaporizer temperature fluctuates, this in turn upsets the NOx control delivery. This pressure steam valve continued to fluctuate the morning of 4/16/2022, with the start of the malfunction at 01:39 and ultimate catastrophic failure of the steam valve which tripped the turbine at 04:40. At 04:50, the control room operator manually tripped all 3 units to cease operation and prevent damage to the boilers and facility. For Unit 5 4/15/2022, upon excluding 2 hours of NOx data during the first significant steam pressure control valve malfunction from 05:00 – 06:00 and an additional 2 hours during the second major malfunction occurrence from 11:00 – 12:00, the new U5 NOx arithmetic average is below the permit limit, with a value of 30 ppmvd @ 7% O2. In an attempt to mitigate emissions, operations reduced the steam flow and SO_2 setpoints, placed the gas burners in service and changed the fuel feed location. The facility adhered to best management practices to minimize emissions during this facility-wide upset. The lime delivery systems for all units were operating normal at the time of this incident. As a corrective action to the elevated SO_2, the facility high SO_2 response procedures are under review. No exclusion applies to the SO_2, therefore the U5 SO_2 geometric mean remains at 25 ppmvd @ 7% O2. For Unit 3 and Unit 5 on 4/16/2022, upon excluding 3 hours of NOx data during the initial significant pressure valve malfunction on 4/16/2022 from 03:00 – 04:00 the new NOx arithmetic average for U3 with a value of 30 ppmvd @ 7% O2 and the new NOx arithmetic average for U5 calculates to a value of 25 ppmvd @ 7% O2, both are below the limit. U3 began operation at 20:59. When excluding the 21:00, 1-hour of data during the start-up period for U3 SO_2,	Units 3 and 5	NOx, SO_2	steam pressure control valve failure tripped turbine

Date (link)	Summary	Unit	Pollutant	Notes
	the U3 SO_2 geometric mean has a value of 32 ppmvd @ 7% O2. With U3 and U5 not operating for most of 4/16/2022 due to the valve malfunction, this allowed only 8 operating hours for U3 and 5 operating hours for U5 to make up the above 24-hour averages. The 4/15/2022 U5 and 4/16/2022 U3 NOx exceedances are within the allowed 3 hours per occurrence per section B.17 of the Title V Air Operating Permit, number 0990234-043-AV. All U3 and U5 NOx and SO_2 emissions surrounding this event will be reported accordingly per Title V Air Operating Permit, number 0990234-043-AV, Appendix RR.			
5/23/2022	Unit 5 Inlet CO was unavailable for 3 hours on 5/22/2022 from 09:00 – 11:00 as a result of the control room screen and the CEMS Software Program (NetDahs) showing the CEMS experienced a Hardware Failure. I&E placed the Inlet CO CEMS in maintenance mode from 09:20 – 09:38 to troubleshoot the cause of the hardware fault alarm. The CEMS Inlet CO passed calibration at 09:24 – 09:38, however the control room display along with NetDahs continued to display a hardware failure for the CEMS Inlet CO. A manual calibration was initiated at 10:05 – 11:09, which failed. The monitor was placed back into maintenance mode at 11:19 and the analyzer was reset and placed back into service at 12:08. A subsequent calibration passed from 12:21 – 12:43 and the hardware failure was no longer present on the displays. It is not expected that there was any exceedance during this time since all air pollution control equipment was in good operating condition and stable combustion of the boiler was maintained.	Unit 5	CO	hardware failure error

Date (link)	Summary	Unit	Pollutant	Notes
5/24/2022	Unit 3 SO_2 lb/hr on 5/13/2022 was elevated above the permitted limit of 25.0 lb/hr @ 7% O2 for the 00:00 – 23:00 timeframe. The data had shown a SO_2 24-hour geometric mean of 27.7 lb/hr @ 7% O2. However, upon investigation, it was found noted in the CEMS logbook that maintenance was performed on the U3 Corrected Stack Flow monitor on 5/15/2022 because of higher than normal flow values. At 04:34 hours, on 5/15/2022, I&E placed the U3 Corrected Stack Flow monitor in maintenance mode and discovered that the readings were higher due to dirty stack flow monitor mirrors. The mirrors were cleaned, and the flow monitor was placed back into service at 05:20 hours. Flow values returned to the expected range. The U3 flow is used to calculate the SO_2 lb/hr. Therefore, a falsely higher flow from the dirty mirrors would in turn calculate an incorrectly higher SO_2 lb/hr. After reviewing the data, it appears that the U3 stack flow monitor data was elevated starting on 5/11/2022 at 14:00. The values remained elevated until the stack flow mirror was cleaned and placed back in service on 5/15/2022 at 05:20, for a total of 87 hours. This subsequently invalidates the U3 SO_2 lb/hr during this same time. It is not expected that there was any exceedance during this time since the U3 SO_2 ppm 24- hour geometric mean remained beneath the permitted limit with all air pollution control equipment operating normal and stable combustion of the boiler was maintained.	Unit 3	SO_2	stack flow mirror dirty
5/31/2022	Palm Beach Renewable Energy Facility #2 Regional Environmental Manager indicated that Unit 5 experienced 4X calibration failures on May 27, 2022, on the NOx monitor resulting in approximately 26 hours of unavailable data, and on May 28, 2022, and on the SO_2 monitor resulting in approximately 27 hours of unavailable data. The facility is able to review and monitor the CEMs data, but due to technical difficulties resulting from the weekend storms is unable to print reports.	Unit 5	NOx, SO_2	calibration failure
6/9/2022	Unit 5 Stack NOx continuous emissions monitor experienced a 4X Out of Control (OOC) calibration during the daily calibration from 10:30 – 10:57 on 5/27/2022. The monitor completed a successful recalibrated at 13:12 – 13:30 on 5/27/2022. However, due to the 4X OOC, per 40 CFR, Appendix F, data is flagged as invalid back to the previous valid calibration at 10:30 – 10:57 on 5/26/2022. Therefore, making a total of 27 hours of NOx data unavailable for 5/26/2022 through 5/27/2022. Unit 5 Stack SO_2 continuous emissions monitor experienced a 4X OOC calibration during the daily calibration from 10:30 – 10:57 on 5/28/2022. The monitor completed a successful recalibrated at 11:23 – 11:47 on 5/28/2022. However, due to the 4X OOC, per 40 CFR, Appendix F, data is flagged as invalid back to the previous valid calibration at 10:06 – 10:30 on 5/27/2022. Therefore, making a total of 23 hours of SO_2 data unavailable for 5/27/2022 through 5/28/2022.	Unit 5	NOx, SO_2	calibration failure

Date (link)	Summary	Unit	Pollutant	Notes
7/28/2022	Upon compilation of the Title V Quarterly Excess Emission and Semi-Annul reports, it was discovered that Unit 3 experienced elevated NOx reading on 4/16/2022, in addition to the previously reported elevated ppm values during the same timeframe for the Steam Pressure Value Malfunction/Master Fuel Trip. This caused the 24-hour arithmetic average for NOx to go above the permitted limit of 37.4 lb/hr for the 00:00-23:59 timeframe for Unit 3 on 4/16/2022. For Unit 3, on 4/16/2022, upon excluding 3 hours of NOx data during steam pressure valve malfunction and Master Fuel Trip from 02:00 - 04:59 the new NOx arithmetic average has a value of 17.0 lb/hr, which is below the limit. The above U3 NOx emissions are within the 3 hours per occurrence per section B.17 of the Title V Air Operating Permit, number 0990234-043-A V	Unit 3	NOx	steam pressure control valve failure tripped turbine on 4/16/2022
7/28/2022	On the morning of 6/30/2022 the PBREF2 Unit 5 NOx monitor recorded a 4X out-of-control calibration at approximately 0608 hours. Per Part 60, all data is flagged as invalid to the previous passing calibration, which occurred on 6/29/2022 at approximately 0608 hours. After facility I&E personnel placed the analyzer in maintenance mode, flowed calibration gas to the analyzer and made appropriate adjustments, the Unit 5 NOx monitor was successfully recalibrated on 6/30/2022 at 0906 hours. The Unit 5 NOx occurrence resulted in 27 hours of unavailable data. Additionally, facility I&E personnel placed the outlet system in maintenance mode at 1108 hours due to a high flow alarm on the NOx analyzer. Pieces of tubing were found rubbing together and appropriate repairs were made. Calibration gas was flowed to verify analyzer response and a successful calibration initiated. The system was taken out of maintenance mode at 1259 hours. The attached data charts show the invalid NOx and SO_2 averages. It is not expected that there was any exceedance during this time since all air pollution control equipment was in good operating condition and stable combustion of the boiler was maintained.	Unit 5	NOx, SO_2	calibration failure; tubing rubbing together
7/29/2022	2022 Q2 Emissions Report - Time out of compliance for stack 3 SO_2 0.28% (24-hr) and stack 5 SO_2 0.44% (24-hr); excess emissions for stack 3 NOx 0.17% (24-hr and 24-lb/hr), SO_2 0.28% (24-hr); excess emissions for stack 4 SO_2 0.19% (24-hr); excess emissions for stack 5 NOx 0.68% (24-hr), SO_2 0.64% (24-hr)		SO_2, NOx	**out of compliance**

Date (link)	Summary	Unit	Pollutant	Notes
9/22/2022	On 9/20/2022 Unit 4 recorded a CO 4-hour block average from 1600-1959 hours at 116. 7 ppm, corrected to 7% 02. Unit 4 was started up at 1252 hours on 9/20/2022. Facility operations personnel experienced difficulties controlling combustion throughout the afternoon, especially on the right side of the boiler, resulting in periodic elevated oxygen and CO levels. Even though CO hourly averages were controlled below the facility permit limit of 100 ppm, corrected to 7% 02, facility operations personnel continued to troubleshoot the issue. The CO hourly average from 1900-1959 hours has been excluded from the 1600-1959 4-hour block average due to this malfunction event. With this exclusion, the new CO 4-hour block average from 1600-1959 hours is 74.9 ppm, corrected to 7% 02. The elevated CO on Unit 4 caused by the grate cooling water leaks and broken grate linkage are within the allowed 3 hours per occurrence per section B. I 7 of the Title V Air Operating Penn it, number 0990234-043- A V. All information surrounding this event will be reported accordingly per Title V Air Operating Permit, number 0990234-043-A V, Appendix RR.	Unit 4	CO	grate linkage broken, cooling water leaks
10/14/2022	9/26/2022 the PBREF2 Unit 5 inlet CO and CO2 monitors recorded an invalid calibration at approximately 0836 hours. Facility personnel investigated the cause of the invalid calibration and determined a card in the Unit 5 PLC controller had malfunctioned. Facility personnel contacted the facility's CEMS software vendor, CemTek, and installed a new PLC controller card which was reprogrammed online by CemTek. After installation and programming of the PLC card, the PLC controller was placed in service at 12:55 hours and an automatic calibration was initiated and passed at approximately 13:24 hours. Further investigation after the passing calibration revealed that the PLC controller malfunction had caused the Unit 5 opacity monitor to lock in place at 10:24 hours on 9/25/2022 and a slow decrease in recorded CO2 values was also recorded during the same time frame. All opacity and CO data has been marked as invalid from 10:24 hours on 9/25/2022 until 12:55 hours on 9/26/2022 for opacity when the PLC controller returned to service and until 13:24 hours on 9/26/2022 for CO when the analyzer passed calibration.	Unit 5	CO2, CO, opacity	PLC card failure

Date (link)	Summary	Unit	Pollutant	Notes
10/24/2022	On 7/6/2022, Unit 4 recorded an SO_2 24-hour geometric average from 0000-2359 hours of 26.3 ppm, corrected to 7% 02. Unit 4 was experiencing unusually elevated inlet SO_2 levels in the flue gas beginning at approximately 1415 hours. Facility personnel responded to the rapidly increasing inlet SO_2 by reducing the boiler load, initiating, and then increasing gas flow to the boiler and placing the lime system in manual control in order to override system interlocks. Inlet and outlet SO_2 values continued to increase until approximately 1640 hours at which time the boiler's **CEMS analyzers reached an over-range condition and flagged the SO_2 data as "Data Error" thereby excluding the data from compliance averaging periods.** Inlet and outlet SO_2 levels continued at CEMS over-range levels until approximately 1820 hours when the unusually high SO_2-containing fuel had been processed through the boiler.	Unit 4	SO_2	fuel content
10/26/2022	2022 Q3 Emissions Report - no time out of compliance; 0.14% excess emission time for CO (4-hr), 0.05% for SO_2 (24-hr)		CO, SO_2	
11/21/2022	Unit 3 Inlet CO was unavailable for 2 hours on 11/20/2022 from 00:00 – 01:59. The Inlet CO2 monitor purge occurred at 00:01 – 00:05, whereas after the purge, the CO2 readings dropped to <0.5% beginning at 00:06, causing the corrected CO to read artificially high. Several attempts to contact I&E began at 00:20. The control room operator initiated a shutdown at 00:21 due to the data unavailability along with unknown cause, per best operational practices. Unit 3 was shutdown from 01:23 until 11:46.	Unit 3	CO2, CO	
11/23/2022	Unit 3 Inlet CO was unavailable for 2 hours on 11/22/2022 from 18:00 – 19:59. The Inlet CO2 monitor purge occurred at 18:01 – 18:05, whereas after the purge, the CO2 readings dropped to <0.5% beginning at 18:06, causing the corrected CO to read artificially high.	Unit 3	CO	
11/23/2022	Unit 3 CO data unavailable for 1 hour to troubleshoot earlier data unavailable issues	Unit 3	CO	
12/7/2022	Unit 4's CEMS recorded elevated SO_2 emissions on 12/4/2022 due to a malfunction of the "B" slaker rotary feeder. This resulted in the 24-hr geometric average for SO_2 to go above the permitted limit of 24 ppm@ 7% 02 and 25.0 lbs/hr for the 00:00-23:59 timeframe. The above Unit 4 SO_2 emissions are within the allowed 3 hours per occurrence per specific condition B.l 7 of the Title V Air Operating Permit, number 0990234-043-AV.	Unit 4	SO_2	slaker rotary feeder malfunction
12/20/2022	Unit 3 SO_2 and NOx [file not available]	Unit 3	NOx, SO_2	
12/30/2022	Unit 5 CO data unavailable for 27 hours; exceedance not expected	Unit 5	CO	
2/1/2023	Units 3 and 5 CO analyzer on 1/19/23 between 11:34-13:32 (Unit 3) and 11:43-13:30 (Unit 5); Unit 3 values later corrected but Unit 5 values are invalid	Units 3 and 5	CO	

Date (link)	Summary	Unit	Pollutant	Notes
2/16/2023	Fire Notification - Fire in REF2 tipping floor pit – fire department extinguished but fire reignited the following day. No impacts to building, boilers, or pollution control devices.	Tipping Floor		tipping floor fire
3/3/2023	Units 3 and 4 CO	Units 3 and 4	CO	
3/13/2023	March 10, 2023, at 8:10 am the Unit 5 CO monitor experienced a 4X calibration failures during the morning calibration. Data was invalid from the last successful calibration on March 9 at 8:30 am, through the successful calibration on March 10, 2023, at 9:27 am, resulting in a total of 25 hours of invalid data for CO.	Unit 5	CO	
3/17/2023	"Unit 5 experienced one 6-minute block average above the permit limit of 10% during a malfunction event. The opacity spike resulted in a 6-minute block average of 25.6%. The malfunction event was caused by the failure of three (3) bags in the Unit 5-E baghouse compartment. The compartment was isolated, and operational controls were adjusted to prevent recurrence of the elevated opacity reading. The defective bags were replaced. The above exceedance is within the allowed three hours per occurrence as per section III.B.17 of the Title V Air Operating Permit, number 0990234-043-AV."	Unit 5	opacity	baghouse failure
5/3/2023	Unit 3 SO_2 Stack Flow Monitor malfunction; exceedance not expected	Unit 3	SO_2	
10/19/2023	Unit 5 SO_2 exceedance 34.4 ppmvd (limit: 24 ppmvd), 29.9 lb/hr (limit: 25 lb/hr)	Unit 5	SO_2	lower furnace draft ports inadvertently plugged after maintenance work
3/13/2024	On March 12, 2024, broken bags in the Unit 4 baghouse caused one 6-minute block average to exceed the opacity permit limit. The Unit 4-D compartment was isolated, and the bags were replaced.	Unit 4	opacity	baghouse failure

Date (link)	Summary	Unit	Pollutant	Notes
7/8/2024	On 6/30/2024 the Unit 3 Outlet CO2 and Outlet SO_2 monitor experienced a failed 4X out of control (OOC) calibration while the unit was not operating… found the calibration line plugged… critical orifice was also partially plugged. This resulted in 11 hours of monitor downtime for SO_2, NOx & CO2. Upon the CEMs coming into service after calibration at 2318, the NOx and SO_2 were found to be elevated. Operations noticed the steam supply valve #0731, which supplies steam to the ammonia skid dilution air heater had malfunctioned, staying closed. This resulted in the low temperature interlock prohibiting the ammonia supply valve from opening. Due to the steam valve malfunction along with only one valid hour for the 24-hour average, the NOx value registered a 152.7 ppm, data from 2300-2359, with a permitted limit of 50 ppm. However, this data will be excluded, resulting in an N/A for the 24-hour NOx data for 6/30/24.	Unit 3	CO2, NOx, SO_2	steam supply valve malfunction, calibration line plugged, critical orifice partially plugged
8/30/2024	The Department was notified on May 22, 2024, that the dioxin/ furan test was aborted during the Unit 4 stack test due to a malfunction with the hydraulic feed system and had to be rescheduled. The Department was again notified on May 23, 2024, that the Unit 3 stack test was aborted due to prematurely failing baghouse filter bags, and also had to be re-scheduled.	Unit 3, Unit 4	D/F	hydraulic feed system failure, baghouse failure
9/30/2024	At 0949 on 9/29/24, the SO_2 monitor experienced a 4X out-of-control calibration. Per Part 60, all data was flagged as invalid to the previous passing calibration, which occurred on 9/28/24 at 0948 hours. The system was placed into maintenance mode, checked out by our technician and adjustments completed on the monitor. On 9/28 at 2144 hours the unit was shut down due to issues with the Ash Discharger and remained shut down until it was started up to continuously combust MSW on 9/29 at 1730 hours. On 9/29 the monitor unit passed calibration and was ready for service at 1627 hours. The SO_2 data from 0800 hours on 9/28 – 9/28 at 2159 hours will be considered unavailable for a total of 13 hours.	Unit 4	SO_2	calibration error, ash discharger isssues
10/4/2024	At 0828 on 10/03/24, the CO monitor experienced a 4X out-of-control calibration. Per Part 60, all data was flagged as invalid to the previous passing calibration, which occurred on 10/02/24 at 0828 hours. The system was placed into maintenance mode, checked out by our technician, and identified a malfunctioned sample pump. The sample pump was replaced and at 1137 hours the unit passed calibration and was returned to service at 1138 hours.	Unit 5	CO	sample pump malfunction
11/13/2024	On 11/12/2024, the CO monitor experienced a 4X out-of-control calibration.	Unit 5	CO	calibration failure

Date (link)	Summary	Unit	Pollutant	Notes
11/27/2024	On 11/25/24 at 08:10 – 08:35 hrs. the PBREF2 Unit 5 CO inlet analyzer registered a 2X calibration warning. In response to this warning, I&E unsuccessfully conducted a manual calibration on the monitor at 10:44 – 11:05, resulting in a 4X calibration failure. I&E performed a subsequent passing manual calibration from 11:06 – 11:22. Then on 11/26/2024, Unit 5 CO inlet analyzer failed calibration from 08:10 – 08:35. Upon investigation, I&E found the analyzer had been locked up ("froze") since 08:08. The monitor check valve failed to open position and the sample line was also cracked, allowing tramp air to enter. This caused the false high CO reading and 4X calibration. I&E had placed the analyzer into maintenance mode at 08:35 to reset the analyzer, to replace the check valve, and to repair the damaged sample line with new hardware. The monitor was returned to service at 11:36, and a calibration was successfully performed from 11:37 – 11:58 Due to the 4X on 11/26/2026, the data is invalidated back to the last good calibration on 11/25/2024. The unit was proactively shut down from 10:03 – 12:16 until the CO monitor issue was resolved. Therefore, the total Unit 5 CO Inlet unavailable data for 11/25/2024 through 11/26/2024 was 23 hours.	Unit 5	CO	calibration failure, check valve failure, sample line cracked
12/4/2024	On 12/03/24, the SO_2 monitor experienced a 4X out-of-control calibration. Per Part 60, all data was flagged as invalid to the previous passing calibration, which occurred on 12/02/24 at 0948 hours. On 12/02 at 1441 hours the unit was shut down due to begin the start of a scheduled outage. Our technician reviewed the system and could not determine a cause for the failure. During the scheduled outage the monitor will be serviced and available prior to the starting up of the unit 4.	Unit 4	SO_2	calibration failure

Exhibit F

Quantitative Analysis of Projected Emissions from Proposed Miami-Dade County Trash Incinerator

An Evaluation of Miami-Dade County's Claims that a New 4,000 Ton/Day Mass Burn Incinerator will Result in No Unacceptable Pollution Impacts

January 24, 2025

Palm Beach Renewable Energy Facility 2 (REF 2) Trash Incinerator
in West Palm Beach, Florida

**Commissioned by The Goldstein Environmental Law Firm, P.A.,
on behalf of the City of Miramar, Florida**

Principal Author:

Mike Ewall
Energy Justice Network
215-436-9511
mike@energyjustice.net
www.energyjustice.net

An online copy of this report is available at:
www.energyjustice.net/fl/mdcincin.pdf

SUMMARY

From December 1981 until February 12, 2023, Miami-Dade County relied on burning much of its municipal solid waste (MSW) at the trash incinerator known as the Miami-Dade County Resource Recovery Facility or “Covanta Dade” in the City of Doral. The incinerator was owned by the county but privately operated by Covanta (now renamed “Reworld” as of April 2024).

On February 12, 2023, a fire broke out and burned for three weeks, causing the permanent closure of the incinerator. Such fires are increasingly common. Two other trash incinerators operated by Covanta had waste pile fires that burned for as long as two weeks in Montgomery County, Maryland and Fairfax County, Virginia in late 2016 and early 2017, respectively. Many smaller fires at incinerators have required an off-site emergency response and these fires are thought to be increasing due to the prevalence of lithium-ion batteries in the waste stream.

Miami-Dade County is planning to build a new incinerator capable of burning 4,000 tons/day, which would be the largest in the U.S. and one of the largest in the world. Currently, the largest incinerator in the U.S. burns up to 3,500 tons/day.

Maimi-Dade County claims new incinerators are clean and safe. The newest trash incinerator built in the U.S., which came online in July 2015, is Palm Beach Renewable Energy Facility 2 built adjacent to Palm Beach County’s older incinerator. **Miami-Dade County’s website boasts that “[t]he Renewable Energy Facility in West Palm Beach is a $672,000,000, state-of-the-art waste-to-energy facility – the most advanced, efficient, cleanest and greenest waste-to-energy power plant in the world.”**[1]

This report examines the actual reported emissions from the Palm Beach Renewable Energy Facility 2 incinerator, models what a new incinerator would emit if built in Miami-Dade County based on the emission rates of this “cleanest and greenest” incinerator combined with newer regulatory requirements, and compares how such a new incinerator would rank next to existing industrial air polluters in Miami-Dade County.

This study finds that a new 4,000 ton/day trash incinerator in Miami-Dade County would be one of the largest industrial air polluters in the county. It would rank #1 in air emissions of ammonia, cadmium, dioxins/furans, hydrochloric acid, and sulfur dioxide, #3 in greenhouse gases and mercury, #4 in nitrogen oxides, #7 in lead and particulate matter, and #9 in carbon monoxide. Dioxins and furans are the most toxic chemicals known to science. This proposed incinerator would become responsible for 73% of the dioxin and furan emissions from industry in the county.

This is based on a new incinerator emitting these pollutants at the same rate as Palm Beach Renewable Energy Facility 2 except for the two pollutants where pending new federal regulations would require meeting a stricter standard. In these cases (carbon monoxide and sulfur dioxide), this analysis assumes that emissions are further reduced by 40% as needed to meet new federal regulations for Large Municipal Waste Combustors, as proposed by the U.S. Environmental Protection Agency (EPA) in 2024 – which are due to be finalized in December 2025 and in effect by around 2029.

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[1] Miami-Dade County, “The Future of Solid Waste in Miami-Dade.” See section under “Waste-to-energy around the world.” https://www.miamidade.gov/global/solidwaste/sustainable-solid-waste/wte-home.page

This analysis is based solely on air emissions data reported from the incinerators themselves to the Florida Department of Environmental Protection, as well as emissions data from the U.S. Environmental Protection Agency's National Emissions Inventory, Greenhouse Gas Reporting Program and Toxics Release Inventory databases, and the U.S. Energy Information Administration.

BACKGROUND

Municipal solid waste (MSW) is the term for household and commercial trash. The closed Miami-Dade incinerator burned refuse-derived fuel (RDF), which is MSW that undergoes minimal processing to remove metal and glass before burning. Most trash incinerators burn MSW with no processing and are called "mass burn" facilities. The RDF vs. mass burn distinction is significant because RDF incinerators are held to weaker emissions standards for carbon monoxide and nitrogen oxides.

The newer incinerator at West Palm Beach (Palm Beach Renewable Energy Facility 2) is the only one in the U.S. that operates under more protective modern standards for nitrogen oxides (NOx), a pollutant that contributes to ground-level ozone (smog) and triggers asthma attacks. They are the only incinerator in the nation using selective catalytic reduction (SCR) technology for reduction of nitrogen oxides. Any new incinerator would also have to use SCR.

Incinerator Name	Owner	Operator	Opened	Closed	Fuel*	Capacity (tons/day)
Palm Beach Renewable Energy Facility 1	Palm Beach County	Covanta (Reworld)	1989		RDF	2,000
Palm Beach Renewable Energy Facility 2	Palm Beach County	Covanta (Reworld)	2015		MSW	3,000
Miami-Dade County Resource Recovery Facility	Miami-Dade County	Covanta (Reworld)	1981	2023	RDF	2,592
	Miami-Dade County		Projected ~2032		MSW	4,000

* RDF = Refuse Derived Fuel (pulled out the metals and glass, then pelletizing or shredding waste before burning);
MSW = municipal solid waste (burning waste without processing… also called "mass burn")

How are incinerator air emissions regulated?

There are many misconceptions about air pollution regulation. Smokestack industries paint a certain narrative. They claim that they use continuous emissions monitors to measure their pollution and that the state regulatory agencies gets this data. They claim that the state regulatory agencies subject them to strict standards and would fine them heavily if they exceed a permit limit. They also claim that they are continually well within their permit limits and that staying within permit limits mean that they're safe and clean and do not cause any harm to public health or the environment.

In order to have a protective air pollution regulatory system, the following are needed:

- Strong, protective standards
- Continuous emissions monitoring
- Aggressive enforcement

In the U.S., all three links in this chain are broken, making the industry narrative misleading.

Incinerators are not always within permit limits. Violations in the industry are not unusual. Some incinerators are known to provide a check every quarter to the state environmental agency when they submit their quarterly reports of their continuous emissions monitoring data, paying for the violations they've had in that quarter. It is the cost of doing business.

Enforcement is lax and fines are not sufficient to change behavior. State enforcement agencies are notoriously lax and understaffed, and when notices of violation are issued, they're often accompanied by zero fines, or fine amounts are allowed to be negotiated down. Imagine being stopped for speeding and telling the officer that you think you should pay $30, not $150 for a ticket. This actually happens with incinerators and other industrial facilities are issued proposed fines for violations, and agencies allow fines to be negotiated down.

Fines are rarely significant enough to change profitable behaviors. Even "large" fines can amount to just a few days of tipping fee revenue, and are insufficient to cause an incinerator owner to invest in needed boiler upgrades or more protective emissions controls, which are more costly than habitually paying fines. Some states even limit the amount of fines under old laws that ensure that fines are just an annoyance to be budgeted for – a "cost of doing business," rather than a deterrent.[2]

Emissions limits are not strict. State environmental agencies issue air permits with emissions limits for about a dozen select pollutants (not all pollutants). State agencies (and local governments) are empowered by the federal Clean Air Act to adopt more protective standards than the federal minimums.[3] This rarely happens, though, as state agencies and permit limits are typically set to the minimum standards in federal regulations. The federal regulations for large municipal waste combustors[4] were last adopted in 2006 and are required by federal law to be updated every five years. However, EPA had to be sued in federal court to enforce this requirement, and finally proposed a new rule in January 2024. That rule was to be finalized by December 2024, but that deadline was extended to December 2025 with the reopening of a comment period. It is unclear whether the rule will be finalized and implemented under the Trump administration. When EPA first proposed these overdue new regulations, during a presentation in early 2023, the agency suggested low, medium, and high levels of emissions reductions for nine pollutants. When EPA's draft rule came out, it became clear that EPA chose the weakest of the three options for eight of the nine regulated pollutants, and the middle option for nitrogen oxides.

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[2] For example, in October 2020, the Covanta Plymouth Renewable Energy trash incinerator in Montgomery County, Pennsylvania was fined $218,393 for violations relating to operational problems causing loud noise and burning plastic and electrical fire smells in the community that have been recurring for over three years now. That amount was considered to be a large fine, but amounted to about three days of Covanta's tipping fee revenues, and failed to stop the recurring problems that continue to this day.

[3] The Clean Air Act, at 42 U.S.C. § 7416, states: "Retention of State authority – Except as otherwise provided in sections 119(c), (e), and (f) (as in effect before the date of the enactment of the Clean Air Act Amendments of 1977), 209, 211(c)(4), and 233 (preempting certain State regulation of moving sources) **nothing in this Act shall preclude or deny the right of any State or political subdivision thereof** to adopt or enforce (1) any standard or limitation respecting emissions of air pollutants or (2) any requirement respecting control or abatement of air pollution; except that if an emission standard or limitation is in effect under an applicable implementation plan or under section 111 or 112, such State or political subdivision may not adopt or enforce any emission standard or limitation which is less stringent than the standard or limitation under such plan or section."

[4] Large Municipal Waste Combustors are trash incinerators where each burner can burn more than 250 tons/day – a size which pertains to all of the incinerators discussed here. See: https://www.epa.gov/stationary-sources-air-pollution/large-municipal-waste-combustors-lmwc-new-source-performance

Permit limits are not based on health and safety, but are technology-based. Permitted emission limits set by state environmental agencies are not based on health and safety. Arguments that complying with permit limits equates to "no harm to health and the environment" are a fallacy. As some state environmental regulators have admitted, permit limits are technology-based standards, and do not ensure that there will be no harm to public health.[5] Many permit limits also factor in the cost to a facility, allowing companies to choose cheaper control technologies if more protective ones are deemed too expensive.[6]

Bigger plants are permitted to be dirtier because permit limits are concentration-based. Air pollution permits are written in such a way that allowed emissions are in units such as parts *per* million (ppm) or micrograms *per* dry standard cubic meter (µg/dscm). It's always "per" something, representing the concentration of a pollutant in a certain volume of air. This design means that a 1,000 ton/day trash incinerator would be allowed to emit a certain amount of a pollutant, but a 4,000 ton/day trash incinerator is allowed to emit four times as much. Regulations also allow incinerators to comply by showing a certain percentage reduction for certain pollutants as an alternative way to meet a limit. In other words, an incinerator can be violating a concentration-based limit, but if the amount going into the pollution control device is so high that they achieve a certain percentage reduction, then they are still deemed to be in compliance.

Emissions monitoring is not always honest. There's the possibility that Covanta's emissions data is not honest. Both annual stack tests and continuous emissions monitors have been rigged at trash incinerators, by Covanta and others, but are rarely caught. In Connecticut, Covanta was fined $20,000 in a civil action filed by the state Attorney General in response to an employee adjusting a continuous emissions monitoring device to alter a reading in order to pass a continuous emissions monitoring audit.[7] In Tulsa, Oklahoma, Covanta was the target of a criminal investigation by the U.S. Attorney's Office "related to alleged improprieties in the recording and reporting of emissions data" in which

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[5] 8/28/2007 Pennsylvania Department of Environmental Protection public hearing on BioNol's proposed natural gas-powered ethanol biorefinery in Clearfield, Pennsylvania. youtu.be/HQtYjEJq4wI When questioned about why residents were told that the proposed air pollution permit means that the facility would be healthy and safe for the community, while permit limits were six times different at a same-sized second ethanol biorefinery proposed eight miles away in Curwensville (but powered by waste coal, not natural gas). DEP's engineer stated: "The quick answer is that our evaluation is based on technology standards, not health standards… The underlying concept around the country is technology based. What is says essentially is that as older plants and older sources fall apart and become useless and are replaced, they need to be replaced with things that are cleaner. …We don't make evaluations of permits based on health standards in a direct fashion. …For some of the large, very large permits like that one [a waste coal burning power plant], there are direct analysis of health issues. In this case, there is none. Typically, for smaller cases like this one, there isn't any. …Are we looking at the cumulative impacts [of multiple large pollution sources] … the answer is 'no.'"

[6] The federal Clean Air Act has several standards that apply, nearly all of which allow for cost considerations. Sections 108-109 set National Ambient Air Quality Standards (NAAQS) for which states must adopt State Implementation Plans to reduce certain pollutants. In areas considered to be in attainment with NAAQS for criterial air pollutants (nitrogen oxides, sulfur dioxide, carbon monoxide, particulate matter, ozone precursors such as volatile organic compounds, and lead), a facility must meet Reasonably Available Control Technology (RACT) standards, where economic feasibility is a factor, and more expensive technology can be ruled out. This is the standard that was recently applied when MDE set the new limit for nitrogen oxide emissions that required no further action by Covanta. In "non-attainment" (unacceptably polluted) areas, the Lowest Achievable Emissions Rate (LAER) standard is applied for that specific pollutant. LAER *does not* consider cost, but allows for a facility to buy offsets (a right to pollute) from polluters in other areas that have closed or reduced their pollution. Section 111 of the Clean Air Act sets New Source Performance Standards for nine pollutants: particulate matter, carbon monoxide, dioxins/furans, sulfur dioxide, nitrogen oxides, hydrogen chloride, lead, mercury, and cadmium. For these, EPA must look at what is maximally achievable to reduce emissions rates, but must also assess the financial implications and must avoid a mandate that would cause "serious economic disruption in the industry." Section 112 of the Clean Air Act sets National Emissions Standards for Hazardous Air Pollutants (NESHAPS), for which cost is not to be considered.

[7] See page 37 for this 1993 incident reported in this 93-page compilation of Covanta's U.S. violations through September 2006: www.energyjustice.net/files/incineration/covanta/violations2006.pdf

Covanta entered into a non-prosecution agreement to follow applicable laws and regulations and pay a $200,000 "community service payment" to the state environmental agency.[8] At other incinerators, including some run by Covanta, the operator has stockpiled cleaner-burning materials like cardboard to use on its annual stack testing day, to make it seem as if their emissions are cleaner year-round.

There is no safe dose of several chemicals released by incinerators. Some chemicals known to be released by incinerators have no safe dose. This includes dioxins,[9] lead,[10] mercury,[11] and particulate matter.[12]

Only a few chemicals are monitored continuously (none of the toxic ones), and only about ten others are tested at all (typically once per year). Only three pollutants are monitored on a continuous basis at most trash incinerators: nitrogen oxides (NOx), sulfur dioxide (SO_2), and carbon monoxide (CO). Some larger incinerators will also continuously monitor carbon dioxide (CO_2). Some parameters are also continuously monitored, like temperature, oxygen, and opacity (darkness of emissions). In rare cases, additional pollutants are monitored on a continuous basis, such as the six trash incinerators in Pennsylvania having to continuously monitor their hydrochloric acid emissions. Other pollutants, if monitored at all, are typically tested once per year, and sometimes less frequently. These other pollutants that are typically tested once per year in an annual stack test are ammonia, dioxins/furans, hydrochloric acid, particulate matter, mercury, lead, and cadmium.

In the case of dioxins and furans, the most toxic chemicals known to science, federal regulations allow just one burner to be tested each year, so an incinerator with three burners (like Palm Beach Renewable Energy Facility 2 or Wheelabrator South Broward) test each burner once every three years, rotating which burner they test each year.

To illustrate, if speeding motorists were regulated the way most industrial air pollutants are, it would be akin to enforcing a speed limit by allowing drivers to drive all year with no speedometer. Once a year, a speed trap would be set on the highway with signs warning "slow down... speed trap ahead," and the driver's designee would be running the speed trap (companies choose who they pay to conduct the test).

The technology exists to continuously monitor over 50 pollutants from incinerators[13], but this is not required by state or federal regulations, so it is rare than an incinerator monitors any of the toxic chemicals on a continuous basis.

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[8] Covanta Holding Corporation's 2019 10-K Securities and Exchange Commission filing, p. 105. (see "Tulsa Matter" describing the consequences of this 2013 incident) d18rn0p25nwr6d.cloudfront.net/CIK-0000225648/992dfb7f-398d-4b17-8e33-75e956f6f235.pdf

[9] "No evidence of dioxin cancer threshold," *Environmental Health Perspectives* 2003 Jul; 111(9): 1145–1147. www.ncbi.nlm.nih.gov/pmc/articles/PMC1241565/

[10] "Lead in the environment: No safe dose," Harvard University excerpt of *The Lancet* (Sept. 11, 2010). www.hsph.harvard.edu/news/multimedia-article/lead/

[11] "Mercury Exposure and Children's Health," *Current Problems in Pediatric and Adolescent Health Care*, 2010 September; 40(8): 186–215. www.ncbi.nlm.nih.gov/pmc/articles/PMC3096006/

[12] World Health Organization, "Ambient (outdoor) air pollution," May 2, 2018. www.who.int/news-room/fact-sheets/detail/ambient-(outdoor)-air-quality-and-health

[13] "Continuous Emissions Monitors (CEMs)." https://www.ejnet.org/toxics/cems/

Failure to continuously monitor these more dangerous chemicals means that testing is only done during optimal operating conditions, as testing is not allowed to be conducted during startup, shutdown, or malfunction times, when emissions are known to be higher.

Testing emissions just once per year can greatly understate actual emissions. At the nation's largest waste incinerator, Reworld (Covanta) Delaware Valley in the City of Chester, Pennsylvania, continuous emissions monitoring of hydrochloric acid emissions shows that actual emissions are 62% higher than their annual stack tests indicate.

Dioxin and furan emissions are an even more stark example. One study out of Europe documented that using continuous sampling for dioxins at incinerators revealed the actual emissions to be 32-52 times higher than we think they are in the U.S. when requiring incinerators to test each unit just once every one to four years under ideal operating conditions.[14] A more recent study found that failure to use continuous sampling technology is underestimating dioxin emissions by 460 to 1,290 times.[15]

In 2023, the Oregon state legislature passed a law (SB 488) requiring the state's only trash incinerator, also a Covanta plant, to continuously monitor nine toxic metals and to continuously sample dioxins/furans and PCBs.[16] After many delays, legal threats, and winning an exemption from the legally required dioxin/furan and PCB monitoring, Covanta announced that they'll be closing their incinerator by December 31, 2024 – just before they'd have to start continuously monitoring for their toxic metal emissions.[17] The company then changed their mind and filed a legal challenge to the monitoring law while also getting legislation introduced seeking to repeal the law in the 2025 legislative session.

Covanta lobbied against Oregon's SB 488, and against a similar bill in the Hawaii state legislature in 2024.[18] Covanta is the nation's largest waste incineration corporation, and operates 33 of the 63 remaining trash incinerators still operating as of January 1, 2025 (after closing both of their California incinerators in 2024). The aggressive effort to avoid continuous monitoring at their Oregon incinerator raises questions of whether the company is concerned about what results from continuous monitoring at any single facility would reveal about underestimated emissions across their entire fleet.

The chart below shows the frequency of testing required by Palm Beach Renewable Energy Facility 2 under their Title V Operating Permit, the air pollution permit granted by Florida DEP. This arrangement is typical for trash incinerators in the U.S.

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[14] De Fré R, Wevers M. "Underestimation in dioxin emission inventories," Organohalogen Compounds, 36: 17–20. www.ejnet.org/toxics/cems/1998_DeFre_OrgComp98_Underest_Dioxin_Em_Inv_Amesa.pdf

[15] Arkenbout, A, Olie K, Esbensen, KH. "Emission regimes of POPs of a Dutch incinerator: regulated, measured and hidden issues." docs.wixstatic.com/ugd/8b2c54_8842250015574805aeb13a18479226fc.pdf

[16] Oregon SB 488 of 2023. https://olis.oregonlegislature.gov/liz/2023R1/Downloads/MeasureDocument/SB488

[17] Beyond Toxics, "Reworld Waste Incinerator Announces Closure," Oct. 11, 2024. https://www.beyondtoxics.org/wp-content/uploads/2024/10/BeyondToxics_PressRelease_Reworld-announces-closure_10-11-24.pdf

[18] Hawaiʻi SB 2101 SD1 of 2024. https://www.capitol.hawaii.gov/session/measure_indiv.aspx?billtype=SB&billnumber=2101&year=2024

Testing requirements in Title V Operation Permit for Palm Beach Renewable Energy Facility 2 Trash Incinerator

Chemical	Abbreviation	Testing frequency under state permit	Category
Sulfur dioxide	SO_2	Continuous	Criteria air pollutant
Nitrogen Oxides	NOx	Continuous	Criteria air pollutant
Carbon Monoxide	CO	Continuous	Criteria air pollutant
Particulate Matter *	PM / PM10 / PM2.5	Annual	Particulate matter
Carbon dioxide	CO_2	Optional (must monitor CO_2 or oxygen continuously)	Global warming pollutant
Ammonia	NH_4	Annual	
Dioxins/Furans	D/F	One burner per year (each burner once every three years)	Highly toxic organohalogens
Polychlorinated biphenyls	PCBs	Never	Highly toxic organohalogens
Per- and polyfluoroalkyl substances	PFAS	Never	Highly toxic organohalogens
Polycyclic Aromatic Hydrocarbons	PAHs	Never	
Volatile Organic Compounds	VOC	Annual	
Hydrochloric Acid	HCl	Annual	Acid gas
Hydrofluoric acid	HF	Never	Acid gas
Arsenic	As	Never	Toxic metal
Beryllium	Be	Never	Toxic metal
Cadmium	Cd	Annual	Toxic metal
Chromium (VI)	Cr (VI)	Never	Toxic metal
Lead	Pb	Annual	Toxic metal
Manganese	Mn	Never	Toxic metal
Mercury	Hg	Annual	Toxic metal
Nickel	Ni	Never	Toxic metal
Selenium	Se	Never	Toxic metal
Zinc	Zn	Never	Toxic metal

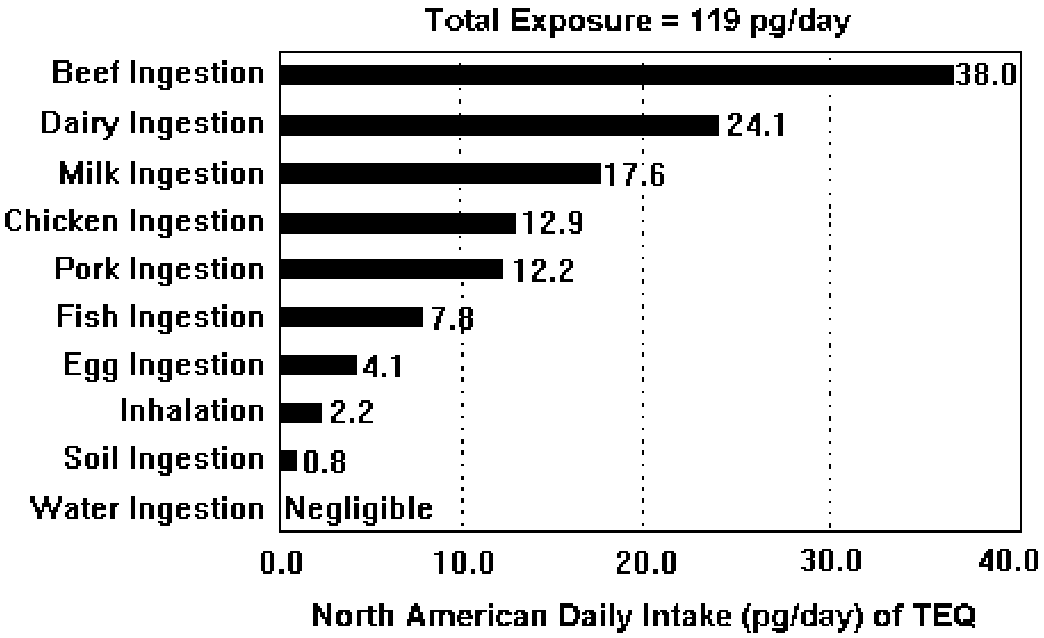
* Opacity (darkness of emissions) is an indirect way of monitoring particulate matter and has to be monitored continuously and in an annual test of visible emissions, but is not a true replacement for actual PM testing.

Dioxin/Furan Emissions

Dioxins and furans are the most toxic class of chemicals known to science. They are largely created in combustion systems like waste incinerators and backyard burn barrels. Dioxins and furans are measured in relation to the most toxic variety of dioxin, which is known as 2,3,7,8 Tetrachlorodibenzo-p-dioxin, or 2,3,7,8-TCDD. Dioxins and furans are chemically similar and are often simply just referred to as dioxins, as the rest of this section does.

Dioxins are a known human carcinogen.[19,20] In addition to causing cancers, exposure to dioxin can also cause severe reproductive and developmental problems at levels 100 times lower than those associated with its cancer causing effects. Dioxin is well-known for its ability to damage the immune system and interfere with hormonal systems. It is associated with causing birth defects, inability to maintain pregnancy, decreased fertility, reduced sperm counts, endometriosis, diabetes, learning disabilities, immune system suppression, lung problems, skin disorders, lowered testosterone levels and much more.[21,22]

EPA has documented that 93% of exposure to dioxins comes through consuming meat and dairy products, since dioxins are fat-soluble and readily bioaccumulate in the food chain. EPA describes dioxins as hydrophobic and lipophilic, meaning that they avoid water but cling to fat. The following chart shows where people consuming a typical North American diet get their dioxin exposure.[23]



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[19] See National Toxicology Program, "Report on Carcinogens, Fifteenth Edition -- 2,3,7,8-Tetrachlorodibenzo-p-dioxin," https://ntp.niehs.nih.gov/sites/default/files/ntp/roc/content/profiles/tetrachlorodibenzodioxin.pdf

[20] International Agency for Research on Cancer, "Polychlorinated Dibenzo-para-Dioxins and Polychlorinated Dibenzofurans," IARC Monographs on the Evaluation of Carcinogenic Risks to Humans – Volume 69, 1997. https://publications.iarc.fr/87

[21] Dioxin Homepage. http://www.ejnet.org/dioxin

[22] Center for Health, Environment & Justice, "The American People's Dioxin Report," p.11, 1999. https://chej.org/wp-content/uploads/American%20Peoples%20Dioxin%20Report.pdf

[23] U.S. Environmental Protection Agency, "Estimating Exposure to Dioxin-Like Compounds, Volume 1: Executive Summary," June 1994, p.36, Figure II-5. "Background TEQ exposures for North America by pathway." https://oaspub.epa.gov/eims/eimscomm.getfile?p_download_id=438673

Once ingested, men do not have a way of ridding their bodies of dioxin, but women have two ways: if pregnant, dioxin will cross the placenta into the growing fetus, and after childbirth, a nursing infant will be exposed via its mother's breast milk. It is estimated that approximately 10-14% of total lifetime dioxin exposure can occur via nursing.[24,25]

EPA classifies dioxins as 140,000 times more toxic than mercury for toxicity via oral exposure.[26] Health impacts are found at levels so small that emissions are measured in nanograms (ng) and exposures are measured in picograms (pg) as the chart above shows. While dioxins are the most toxic chemicals known to be released from incinerators, they are the least monitored. Typically only one burner per year has to be tested, so an incinerator with three burners has each burner tested just once every third year, while all other pollutants have to be tested at each burner annually (if not also continuously).[27]

Truly continuous emissions monitoring technology exists for dioxins but is not commercially available. However, continuous *sampling* technology has been commercially established since at least the late 1990s. Instead of having results immediately available on-site, continuous sampling collects a sample in a cartridge for up to 4-6 weeks, then that cartridge is switched out with a new one and sent to a lab to test for the cumulative amount of dioxins emitted over that span of time. This makes it possible to get the full picture of emissions, capturing data during startup, shutdown and malfunction times when dioxins are known to spike, even though the spikes aren't particularly visible because they're averaged into the sample across several weeks.

The most common continuous sampling method is known as AMESA (Adsorption Method for Sampling of Dioxins and Furans).[28,29] This and other methods were tested and verified by EPA in 2006.[30] Dioxin continuous sampling technology is not used in the U.S., but is used at incinerators in Europe as well as at the only new trash incinerator in Canada, the Durham York Energy Centre in Clarington, Ontario. When that incinerator opened in 2015 (same year as Palm Beach Renewable Energy Facility 2), it failed both of its initial dioxin stack tests, and has continued to experience documented exceedances at times (once with dioxin emissions 13.6 times the permitted limit), though it's hard to know how often there are exceedances because much of the data is not being released.

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[24] Patandin, S., Dagnelie, P.C., Mulder, P.G.H., Op de Coul, E., van der Veen, J.E., Weisglas-Kuperus, N., and Sauer, P.J.J. (1999) "Dietary exposure to polychlorinated biphenyls and dioxins from infancy until adulthood: A comparison between breast-feeding, toddler and long-term exposure." Environmental Health Perspectives 107 (1): 45-51. https://pmc.ncbi.nlm.nih.gov/articles/PMC1566290/

[25] Schecter, A., Papke O., Lis, A., Ball, M., Ryan, J.J., Olson, J.R., Li, L., and Kessler, H. (1996) "Decrease in milk and blood dioxin levels over two years in a mother nursing twins: Estimates of decreased maternal and increased infant dioxin body burden from nursing." Chemosphere 32 (3): 543-549. https://pubmed.ncbi.nlm.nih.gov/8907231/

[26] U.S. Environmental Protection Agency, Risk-Screening Environmental Indicators (RSEI) Model. https://www.epa.gov/rsei

[27] The alternative performance testing schedule for dioxins/furans (D/F) specified in 40 CFR 60.58b(g)(5)(iii) allow testing at just one unit each year so long as they stay under 7 ng/dscm. https://www.ecfr.gov/current/title-40/chapter-I/subchapter-C/part-60/subpart-Eb/section-60.58b

[28] U.S. Environmental Protection Agency, "ETV Joint Verification Statement -- Adsorption Method for Sampling Dioxins and Furans," 2006. https://archive.epa.gov/nrmrl/archive-etv/web/pdf/600etv06047sv2.pdf

[29] Wikipedia, "Adsorption Method for Sampling of Dioxins and Furans," https://en.wikipedia.org/wiki/Adsorption_Method_for_Sampling_of_Dioxins_and_Furans

[30] U.S. Environmental Protection Agency Environmental Technology Verification Program, "Dioxin Emission Monitoring Systems," https://archive.epa.gov/nrmrl/archive-etv/web/html/vt-ams.html#dems

The following section is drawn verbatim from "CAUTIONARY TALES: Examples from across Canada," a report discussing Canada's trash incinerators.[31] Find the footnoted references in the original, as cited.

> ### Issues At The Durham/York Incinerator (DYEC): Stack Exceedances in 2015,2016 for dioxins and furans
>
> Acceptance stack testing in early October 2015, found dioxin/furan emissions in exceedance of the stack limit for both boilers during initial tests.[7] Stack source testing in May 2016 again found a dioxins/furans exceedance. One boiler was emitting 818 pg TEQ/m3 while the emission limit is 60 pg TEQ/m3.[8] For the May 2016 major exceedance there was no indication from the continuous emissions monitors (CEMS) in the control room that there were problems.[9]
>
> Stack source testing is pre-announced and completed only twice a year (once for compliance, once voluntarily). Dioxins/furans stack source testing only provides a snapshot as it covers less than 0.5% of the facility's operating time. Short-term (hourly, daily, weekly) dioxins/furans stack concentrations for the remaining 99.5+% of the year are unknown. This concern applies to most pollutants - including heavy metals, polycyclic aromatic hydrocarbons (PAH)s, and volatile organic compounds (VOCs) which are monitored at the stack less than 1% of operational time.
>
> The duration of the exceedance for dioxins/furans in 2016 is unknown. The Regions are required under the Environmental Compliance Approval (ECA) to conduct long-term sampling of dioxins and furans,[10] however the Regions have withheld the monthly sampling data from 2015 to 2019, despite public requests. Two [Freedom of Information] requests filed in May 2019 remain active. The owners shut down the DYEC for a period after the May 2016 exceedance. An abatement plan followed.[11] Multiple problems were identified.[12] Major repairs, operational changes were made.[13]
>
> ### Ambient air exceedance in 2018 for dioxins/furans
>
> A concerning ambient air exceedance of dioxins/furans occurred in May 2018 at a DYEC ambient air monitoring station.[14] The Regions' consultant's limited review concluded "the DYEC is unlikely to have substantially contributed to the elevated D/F concentration".[15] Members of the public contend that the investigation of this troubling exceedance was inadequate and did not include review of all pertinent data. The source cause of the exceedance remains unexplained.
>
> ### Ongoing concerns with dioxins/furans AMESA reporting
>
> Long-term sampling for dioxins and furans is required under the ECA. Durham uses the AMESA system for sampling. Dioxins/furans are collected in a cartridge over a month and sent for analysis. The public advocated for this monitoring during the Environmental Assessment and ECA phases. The Regions have withheld the monthly data for years 2015 to 2019. From 2020 onward, the Regions have provided some of the data, however, do not provide the underlying lab and other reports. Over time the Regions modified the sampling equipment and developed sampling procedures and protocols, including for data validation. In short, the reporting is neither traceable nor transparent.

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[31] Maxwell, S., Benneian, L., Bracken, W., and Gasser, L., "CAUTIONARY TALES: Examples from across Canada," Dec. 2023. https://drive.google.com/file/d/1-oet-KSfK60A7tTVUR5SaTYPZHVT4qui/view

HOW POLLUTING IS A NEW INCINERATOR?

No incinerator is “clean,” as all involve putting significant amounts of pollutants into the air. The question is a relative one. Yes, newer trash incinerators such as Palm Beach Renewable Energy Facility 2 are less polluting than the older generation. However, Palm Beach Renewable Energy Facility 2 is still a major air polluter. A new incinerator such as Miami-Dade County is pursuing would largely use the same technology that Palm Beach Renewable Energy Facility 2 uses.

A new incinerator may have to meet somewhat stricter EPA regulations for Large Municipal Waste Combustors. However, except for two pollutants, Palm Beach Renewable Energy Facility 2 already meets these pending regulations, which means that the emissions from a new incinerator can be expected to be similar to those from Palm Beach Renewable Energy Facility 2 except for carbon monoxide and sulfur dioxide, where an additional 40% emissions reduction would be needed to meet the proposed new regulations.

As this report documents, a new 4,000 ton/day trash incinerator in Miami-Dade County, permitted under the proposed new federal regulations, would be one of the largest industrial air polluters in the county. It would rank #1 in air emissions of ammonia, cadmium, dioxins/furans, hydrochloric acid, and sulfur dioxide, #3 in greenhouse gases and mercury, #4 in nitrogen oxides, #7 in lead and particulate matter, and #9 in carbon monoxide.

METHODOLOGY

In Florida, emissions data is reported by incinerator owners to the state Department of Environmental Protection (DEP). Documents on these facilities are available online through DEP’s Oculus system. Stack test reports and annual operating reports are among these online files. Stack test reports offer the emissions test data for the once-per-year tests, presenting them in units that line up with the permit limits in the incinerators’ Title V air pollution permit issued by DEP under the Clean Air Act. Annual operating reports offer calculated estimates of the annual amount of each pollutant released. Since few air pollutants are measured continuously, most of this data is based on once per year stack tests under optimal operating conditions. This means that the emissions data can be optimistic since the calculation method assumes that the facility operates under those optimal conditions all year. The lack of continuous emissions monitoring results in annual emissions estimates that are likely lower than reality.

For the Palm Beach County incinerators, 2020 data was not able to be found in DEP’s Oculus system, so values were obtained from EPA’s National Emissions Inventory, which contains the same sort of data as obtained from state agencies like DEP.

The pounds of each pollutant released per year from each facility were totaled and divided into the total amount of tons burned over those same years. Since a 2020 annual operating report could not be found for the Palm Beach County incinerators, data on the tons of waste burned was obtained from the Energy Information Administration’s Form 923 database, which tracks fuel usage by month for all electric generators.

With data from the new Palm Beach Renewable Energy Facility 2 incinerator, the pounds of emissions per ton of waste were used as a model for what a new incinerator in Miami-Dade County would release. This was done in cases where the concentration-based emissions data fell within the limits of the new EPA regulations for Large Municipal Waste Combustors which might be applied if finalized in these final months of the Biden administration and if not repealed by the Trump administration. There are two pollutants for which the Palm Beach Renewable Energy Facility 2 emissions exceed this new standard for new incinerators: carbon monoxide and sulfur dioxide. For each of these, a further 40% reduction in emissions was assumed in order to bring emissions down sufficiently to meet the new standard.

Greenhouse gases are not tracked in the DEP's files, so data for carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) were obtained from EPA's Greenhouse Gas Reporting Program, the most accurate data source for such data. Since EPA intentionally uses outdated global warming potentials for methane, that data was adjusted to bring it in line with the latest global warming potentials for methane and nitrous oxide, looking at it over 20 years instead of 100.[32]

Dioxin data is not tracked in EPA's National Emissions Inventory, so data for other facilities in the county had to be obtained from EPA's Toxics Release Inventory (TRI), which is released annually.[33] Since there were only two data points available for dioxin emissions from Miami-Dade County sources in 2020, an additional 17 data points were available and used to find average dioxin emissions levels for the four facilities reporting dioxin emissions to the TRI database between 2014 and 2023, making the data more robust, and matching the methodology of averaging multiple years of data from Palm Beach Renewable Energy Facility 2 to come up with a more honest comparison. Because waste incinerators have always been exempt from reporting to the TRI database, data on the old Miami-Dade County trash incinerator are not presented here.[34] Also, it should be noted that dioxin emissions levels are likely underestimated for lack of continuous sampling, but that the test methods for one facility vs. another are similar, so the underestimation applies to all facilities and the relative rankings of them are thus the most fair and accurate comparisons that can be made.

Using "pounds of a pollutant released per ton of waste burned" metrics for each air pollutant (see the table below), these metrics were multiplied times 4,000 tons of waste burned per day times 365 days per year times 95% for the capacity factor, which represents the percentage of a year that the facility is operating at full capacity. 95% is what a well-run facility should be capable of, but many older generation incinerators operate closer to 90% capacity.

The following chart shows the pounds per ton burned for the Palm Beach Renewable Energy Facility 2, averaging all available years of their emissions test data, then the calculated amount of emissions from a new 4,000 ton/day incinerator. The final column is the same except that carbon monoxide and sulfur dioxide are reduced by 40% to ensure that the emissions would fall within EPA's proposed new regulations for Large Municipal Waste Combustors.

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[32] A chart showing the evolving science on methane global warming potentials across International Panel on Climate Change reports, with links to the sources, is here: https://www.energyjustice.net/naturalgas#GWP

[33] U.S. Environmental Protection Agency, Toxics Release Inventory database. https://www.epa.gov/toxics-release-inventory-tri-program

[34] In April 2023, Energy Justice Network and the Public Employees for Environmental Responsibility filed a formal rulemaking petition with EPA to require that incinerators start reporting to the TRI database. On December 20, 2024, EPA granted the petition, requiring trash incinerators and certain other incinerators to start reporting to the TRI database. See: http://www.energyjustice.net/tri

Chemical (all in lbs except GHGs)	Palm Beach REF 2 Lbs/Ton burned	Annual lbs from new 4,000 tpd plant	After adjusting for new regs, if adopted
Ammonia	0.0294	40,715	40,715
Cadmium	0.00000585	8	8
Carbon monoxide	0.345	477,945	286,767
Dioxins/Furans	0.0000000036	0.0050	0.0050
Hydrochloric acid	0.0626	86,895	86,895
Lead	0.0000231	32	32
Mercury Compounds	0.00000768	11	11
Nitrogen Oxides	0.659	913,588	913,588
Particulate Matter	0.0318	44,138	44,138
Sulfur Dioxide	0.551	764,375	496,844
Greenhouse Gases (GHGs) [metric tons of CO2 equivalents]	0.879	1,219,103	1,219,103

How would a new incinerator in Miami-Dade County rank among existing industrial air polluters?

The following charts show how a new 4,000 ton/day trash incinerator operating under the pending regulations for new trash incinerators (large municipal waste combustors) compares to existing industrial air polluters in Miami-Dade County. The comparison data is the latest available from EPA's National Emissions Inventory, which is data from 2020. That data is published every third year, and 2023 data is supposed to be available in 2026. The tables below include the old county incinerator that is now closed, for reference. The rankings use the 2020 emissions data, but are ranked as if the theoretical proposed incinerator is operating in place of the old incinerator.

Note that there are no incinerators in the nation using any sort of carbon capture and sequestration technology, which is prohibitively expensive. Nonetheless, Miami-Dade County is looking around the world at incinerators that are experimenting with such technology, and claims to want to implement that in a new plant, even though there are no legal requirements for it, nor are any such regulations on the horizon. The rankings below are based on the current regulatory requirements plus proposed regulations, which do not include CO_2 limits.

Summary of rankings:

Pollutant	New incinerator's rank among Miami-Dade County industrial air polluters	New incinerator's percentage of total emissions from Miami-Dade County industrial air polluters
Ammonia	1	54%
Cadmium	1	56%
Carbon monoxide	9	2%
Dioxins/furans	1	73%
Hydrochloric acid	1	84%
Lead	7	2%
Mercury	3	11%
Nitrogen Oxides	4	1%
Particulate Matter	7	3%
Sulfur Dioxide	1	34%
Greenhouse Gases	3	16%

Emissions rankings of new incinerator vs. 2020 EPA data on existing industrial air polluters in Miami-Dade County:

[Comparison data from EPA's 2020 National Emissions Inventory and,
for dioxins/furans, EPA's Toxics Release Inventory]

Pollutant	Facility	Industry
Greenhouse Gases (CO_2e) [Metric tons]		
2,363,081	Turkey Point Power Plant (Florida Power & Light)	1,224 MW gas fired power plant
1,305,341	Titan Florida LLC	Waste burning cement kiln
1,219,103	**Proposed 4,000 ton/day Trash Incinerator**	Incinerator
792,912	Miami-Dade County - North Dade Landfill	Landfill
752,748	**Covanta Dade 2,592 ton/day Trash Incinerator [closed]**	Incinerator
679,213	Cemex Construction Materials FL. LLC.	Cement kiln
530,473	WM - Medley Landfill	Landfill
263,971	Miami International Airport	Airport
230,057	Miami-Dade County - South Dade Landfill	Landfill
209,469	58th St Landfill (Main County LF)	Landfill
29,605	Homestead Air Reserve Base - Airport	Airport
22,288	Opa Locka Executive Airport	Airport
21,951	Kendall-Tamiami Executive Airport	Airport
10,137	Homestead General Aviation	Airport
557	Miami	Airport
437	CSX Transportation – Hialeah	Rail Yard

Ammonia (lbs)		
40,715	**Proposed 4,000 ton/day Trash Incinerator**	Incinerator
34,356	Turkey Point Power Plant (Florida Power & Light)	1,224 MW gas fired power plant
685	NUSFC, LLC	Iron Foundry
8	CSX Transportation - Hialeah	Rail Yard

Cadmium (lbs)		
8.1	**Proposed 4,000 ton/day Trash Incinerator**	Incinerator
2.8	**Covanta Dade 2,592 ton/day Trash Incinerator [closed]**	Incinerator
1.9	Cemex Construction Materials FL. LLC.	Cement kiln
1.7	Titan Florida LLC	Waste burning cement kiln
1.0	Goodrich Corporation	Aerospace Product and Parts Manufacturing
0.7	NUSFC, LLC	Iron Foundry
0.6	U.S. Dept of Agriculture	Government agency
0.2	Jackson Memorial Hospital	Hospital
0.1	Quikrete Companies, LLC	Concrete Product Manufacturing
0.0	Hometown Foods USA, LLC	Bakery
0.0	Sunlite Sales, Inc.	Metal Coating / Engraving
0.0	Hydro Conduit, LLC (DBA Rinker Materials)	Concrete Block and Brick Manufacturing

Dioxins and furans (lbs)		
0.005	**Proposed 4,000 ton/day Trash Incinerator**	Incinerator
0.00078	Cemex Construction Materials FL. LLC.	Cement kiln
0.00049	Titan Florida LLC	Waste burning cement kiln
0.00028	Turkey Point Power Plant (Florida Power & Light)	1,224 MW gas fired power plant
0.00027	NUSFC, LLC	Iron Foundry

Hydrochloric acid (lbs)		
86,895	**Proposed 4,000 ton/day Trash Incinerator**	Incinerator
89,728	**Covanta Dade 2,592 ton/day Trash Incinerator [closed]**	Incinerator
9,440	WM - Medley Landfill	Landfill
6,433	Cemex Construction Materials FL. LLC.	Cement kiln
774	Titan Florida LLC	Waste burning cement kiln
111	NUSFC, LLC	Iron Foundry
30	U.S. Dept of Agriculture	Government agency

Lead (lbs)		
807	Kendall-Tamiami Executive Airport	Airport
520	Opa Locka Executive Airport	Airport
332	Homestead General Aviation	Airport
183	NUSFC, LLC	Iron Foundry
73	Cemex Construction Materials FL. LLC.	Cement kiln
70	Miami International Airport	Airport

57	**Covanta Dade 2,592 ton/day Trash Incinerator [closed]**	Incinerator
32	**Proposed 4,000 ton/day Trash Incinerator**	Incinerator
7	Titan Florida LLC	Waste burning cement kiln
3	Turkey Point Power Plant (Florida Power & Light)	1,224 MW gas fired power plant
2	Miami	Airport
0	Quikrete Companies, LLC	Concrete Product Manufacturing
0	Goodrich Corporation	Aerospace Product and Parts Manufacturing
0	Jackson Memorial Hospital	Hospital
0	Hometown Foods USA, LLC	Bakery
0	Derby Building Products, LLC	Plastics Pipe & Pipe Fitting
0	Sunlite Sales, Inc.	Metal Coating / Engraving
0	Hydro Conduit, LLC (DBA Rinker Materials)	Concrete Block and Brick Manufacturing

Mercury (lbs)		
51	Titan Florida LLC	Waste burning cement kiln
30	Cemex Construction Materials FL. LLC.	Cement kiln
11	**Proposed 4,000 ton/day Trash Incinerator**	Incinerator
8	NUSFC, LLC	Iron Foundry
5	**Covanta Dade 2,592 ton/day Trash Incinerator [closed]**	Incinerator
0.040795	58th St Landfill (Main County LF)	Landfill
0.016549	South Florida Water Management District	Administration of Water Resources
0.008000	Asahi Refining Florida, Inc. - Miami Gardens	Secondary Smelting / Refining
0.003719	CSX Transportation - Hialeah	Rail Yard
0.001577	Miami-Dade Water and Sewer Department - 3869 Rickenbacker Causeway	Sewage Treatment Plant
0.000505	Turkey Point Power Plant (Florida Power & Light)	1,224 MW gas fired power plant
0.000066	Miami-Dade Water and Sewer Department - 1100 W 2nd Ave	Water Supply and Irrigation Systems
0.000020	WM - Medley Landfill	Landfill
0.000009	Miami-Dade Water and Sewer Department - 6800 SW 87th Ave	Sewage Treatment Plant
0.000002	Flowers Baking Company of Miami, LLC	Bakery
0.000002	Miami-Dade Water and Sewer Department - 2575 NE 156th St	Sewage Treatment Plant

Nitrogen Oxides (lbs)		
3,864,995	Titan Florida LLC	Waste burning cement kiln
3,615,200	Cemex Construction Materials FL. LLC.	Cement kiln
2,454,167	**Covanta Dade 2,592 ton/day Trash Incinerator [closed]**	Incinerator
1,749,041	Miami International Airport	Airport
913,588	**Proposed 4,000 ton/day Trash Incinerator**	Incinerator
551,631	Homestead Air Reserve Base - Airport	Airport
526,927	Miami-Dade Water and Sewer Department - 6800 SW 87th Ave	Sewage Treatment Plant
359,643	Turkey Point Power Plant (Florida Power & Light)	1,224 MW gas fired power plant
163,499	South Florida Water Management District	Administration of Water Resources
138,215	Miami-Dade Water and Sewer Department - 8932 SW 232nd St	Sewage Treatment Plant

125,254	Miami-Dade Water and Sewer Department - 2575 NE 156th St	Sewage Treatment Plant
107,000	WM - Medley Landfill	Landfill
103,162	Miami-Dade Water and Sewer Department - 3869 Rickenbacker Causeway	Sewage Treatment Plant
100,860	Opa Locka Executive Airport	Airport
47,421	Miami-Dade Water and Sewer Department - 1100 W 2nd Ave	Water Supply and Irrigation Systems
23,566	Homestead General Aviation	Airport
18,728	CSX Transportation - Hialeah	Rail Yard
17,896	Kendall-Tamiami Executive Airport	Airport
13,038	Jackson Memorial Hospital	Hospital
11,842	Community Asphalt Corporation	Asphalt Plant
9,795	Miami-Dade County - South Dade Landfill	Landfill
7,683	Homestead Air Reserve Base	Military Base
7,409	Miami-Dade County - North Dade Landfill	Landfill
7,353	Homestead City Utilities	Oil-burning Power Plant
7,100	NUSFC, LLC	Iron Foundry
4,029	Flowers Baking Company of Miami, LLC.	Bakery
1,999	Derby Building Products, LLC	Plastics Pipe & Pipe Fitting
1,240	Hometown Foods USA, LLC	Bakery
1,015	Kingspan Insulation, LLC	Plastics Product Manufacturing
784	Noven Pharmaceuticals, Inc.	Pharmaceutical Preparation Manufacturing
759	AAR Landing Gear Services	Aerospace Product and Parts Manufacturing
676	Miami	Airport
561	U.S. Dept of Agriculture	Government agency
354	Solo Printing, LLC	Printing
320	Bill Ussery Motors Body Shop, Inc	Auto Body / Paint Shop
280	Sunlite Sales, Inc.	Metal Coating / Engraving
210	Delta Apparel, Inc.	Printing
166	Goodrich Corporation	Aerospace Product and Parts Manufacturing
93	Aircraft Electric Motors, Inc.	Motor and Generator Manufacturing
59	Heico Corporation	Aerospace Product and Parts Manufacturing
45	Hydro Conduit, LLC (DBA Rinker Materials)	Concrete Block and Brick Manufacturing
16	Asahi Refining Florida, LLC - Opa-Locka	Secondary Smelting / Refining
12	Asahi Refining Florida, Inc. - Miami Gardens	Secondary Smelting / Refining
12	Aerothrust Holdings, LLC	Aircraft Manufacturing

Sulfur Dioxide (lbs)		
496,844	**Proposed 4,000 ton/day Trash Incinerator**	Incinerator
441,262	WM - Medley Landfill	Landfill
219,230	Miami International Airport	Airport
89,912	**Covanta Dade 2,592 ton/day Trash Incinerator [closed]**	Incinerator

66,040	Cemex Construction Materials FL. LLC.	Cement kiln
52,094	Homestead Air Reserve Base - Airport	Airport
45,951	Miami-Dade Water and Sewer Department - 8932 SW 232nd St	Sewage Treatment Plant
39,831	Titan Florida LLC	Waste burning cement kiln
32,877	NUSFC, LLC	Iron Foundry
27,245	Turkey Point Power Plant (Florida Power & Light)	1,224 MW gas fired power plant
15,481	South Florida Water Management District	Administration of Water Resources
11,359	Opa Locka Executive Airport	Airport
4,553	Miami-Dade County - South Dade Landfill	Landfill
4,082	Miami-Dade County - North Dade Landfill	Landfill
3,575	Kendall-Tamiami Executive Airport	Airport
3,037	Homestead General Aviation	Airport
2,067	Jackson Memorial Hospital	Hospital
1,809	Community Asphalt Corporation	Asphalt Plant
482	Homestead City Utilities	Oil-burning Power Plant
405	Miami-Dade Water and Sewer Department - 6800 SW 87th Ave	Sewage Treatment Plant
342	U.S. Dept of Agriculture	Government agency
337	Homestead Air Reserve Base	Military Base
139	Miami	Airport
132	Miami-Dade Water and Sewer Department - 1100 W 2nd Ave	Water Supply and Irrigation Systems
86	Miami-Dade Water and Sewer Department - 2575 NE 156th St	Sewage Treatment Plant
84	Miami-Dade Water and Sewer Department - 3869 Rickenbacker Causeway	Sewage Treatment Plant
26	Flowers Baking Company of Miami, LLC.	Bakery
20	Goodrich Corporation	Aerospace Product and Parts Manufacturing
20	Bill Ussery Motors Body Shop, Inc	Auto Body / Paint Shop
16	AAR Landing Gear Services	Aerospace Product and Parts Manufacturing
14	Asahi Refining Florida, LLC - Opa-Locka	Secondary Smelting / Refining
12	Derby Building Products, LLC	Plastics Pipe & Pipe Fitting
9	CSX Transportation - Hialeah	Rail Yard
7	Hometown Foods USA, LLC	Bakery
6	Kingspan Insulation, LLC	Plastics Product Manufacturing
5	Noven Pharmaceuticals, Inc.	Pharmaceutical Preparation Manufacturing
2	Solo Printing, LLC	Printing
2	Sunlite Sales, Inc.	Metal Coating / Engraving
1	Delta Apparel, Inc.	Printing
0	Heico Corporation	Aerospace Product and Parts Manufacturing
0	Hydro Conduit, LLC (DBA Rinker Materials)	Concrete Block and Brick Manufacturing
0	Asahi Refining Florida, Inc. - Miami Gardens	Secondary Smelting / Refining

Carbon Monoxide (lbs)		
3,521,121	Miami International Airport	Airport
2,012,665	Titan Florida LLC	Waste burning cement kiln
1,699,948	**Covanta Dade 2,592 ton/day Trash Incinerator [closed]**	Incinerator
990,800	Kendall-Tamiami Executive Airport	Airport
782,623	Opa Locka Executive Airport	Airport
641,255	Homestead Air Reserve Base - Airport	Airport
633,056	Cemex Construction Materials FL. LLC.	Cement kiln
428,536	Homestead General Aviation	Airport
395,782	WM - Medley Landfill	Landfill
286,767	**Proposed 4,000 ton/day Trash Incinerator**	Incinerator
177,362	Miami-Dade County - South Dade Landfill	Landfill
148,641	Miami-Dade Water and Sewer Department - 6800 SW 87th Ave	Sewage Treatment Plant
136,086	Miami-Dade County - North Dade Landfill	Landfill
57,641	NUSFC, LLC	Iron Foundry
56,801	Miami-Dade Water and Sewer Department - 8932 SW 232nd St	Sewage Treatment Plant
52,756	Turkey Point Power Plant (Florida Power & Light)	1,224 MW gas fired power plant
47,674	Miami-Dade Water and Sewer Department - 2575 NE 156th St	Sewage Treatment Plant
46,732	Miami-Dade Water and Sewer Department - 3869 Rickenbacker Causeway	Sewage Treatment Plant
27,991	Community Asphalt Corporation	Asphalt Plant
18,516	Miami-Dade Water and Sewer Department - 1100 W 2nd Ave	Water Supply and Irrigation Systems
6,708	Miami	Airport
4,473	Jackson Memorial Hospital	Hospital
4,129	Homestead Air Reserve Base	Military Base
3,366	Flowers Baking Company of Miami, LLC.	Bakery
2,750	58th St Landfill (Main County LF)	Landfill
2,607	CSX Transportation - Hialeah	Rail Yard
1,679	Derby Building Products, LLC	Plastics Pipe & Pipe Fitting
822	Kingspan Insulation, LLC	Plastics Product Manufacturing
659	Noven Pharmaceuticals, Inc.	Pharmaceutical Preparation Manufacturing
591	South Florida Water Management District	Administration of Water Resources
480	Homestead City Utilities	Oil-burning Power Plant
260	Hometown Foods USA, LLC	Bakery
235	Sunlite Sales, Inc.	Metal Coating / Engraving
177	Delta Apparel, Inc.	Printing
150	Solo Printing, LLC	Printing
96	Goodrich Corporation	Aerospace Product and Parts Manufacturing
80	Bill Ussery Motors Body Shop, Inc	Auto Body / Paint Shop
79	Aircraft Electric Motors, Inc.	Motor and Generator Manufacturing
52	AAR Landing Gear Services	Aerospace Product and Parts Manufacturing
52	Asahi Refining Florida, LLC - Opa-Locka	Secondary Smelting / Refining

50	Heico Corporation	Aerospace Product and Parts Manufacturing
38	Hydro Conduit, LLC (DBA Rinker Materials)	Concrete Block and Brick Manufacturing
20	U.S. Dept of Agriculture	Government agency
10	Asahi Refining Florida, Inc. - Miami Gardens	Secondary Smelting / Refining
2	Aerothrust Holdings, LLC	Aircraft Manufacturing

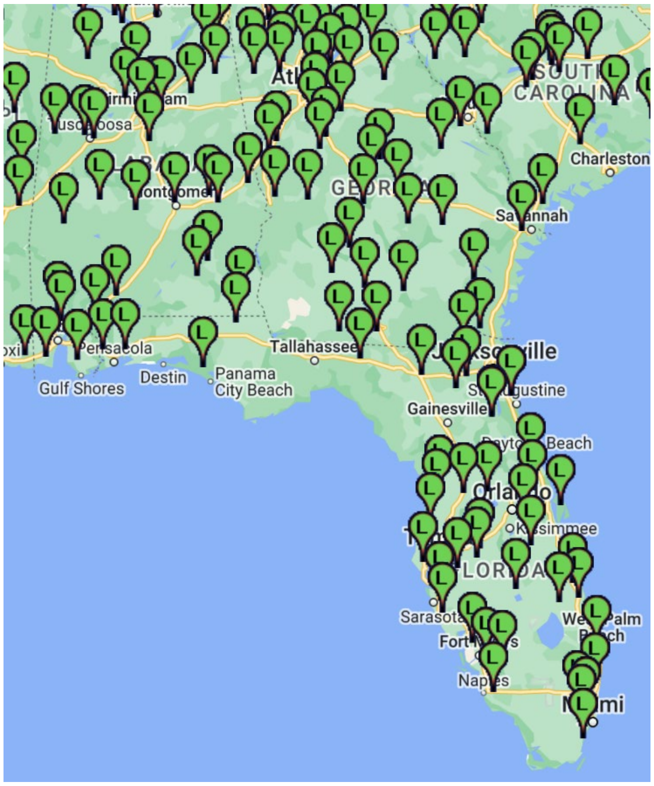
Particulate Matter (lbs)		
300,376	Turkey Point Power Plant (Florida Power & Light)	1,224 MW gas fired power plant
299,801	Miami-Dade Water and Sewer Department - 6800 SW 87th Ave	Sewage Treatment Plant
202,644	Cemex Construction Materials FL. LLC.	Cement kiln
117,838	Titan Florida LLC	Waste burning cement kiln
98,829	**Covanta Dade 2,592 ton/day Trash Incinerator [closed]**	Incinerator
52,397	WM - Medley Landfill	Landfill
51,793	NUSFC, LLC	Iron Foundry
44,138	**Proposed 4,000 ton/day Trash Incinerator**	Incinerator
37,767	Miami International Airport	Airport
34,409	Homestead Air Reserve Base - Airport	Airport
23,095	Opa Locka Executive Airport	Airport
21,992	Kendall-Tamiami Executive Airport	Airport
19,511	Cemex Construction Materials Florida LLC.	Concrete Batch Plant
18,253	South Florida Water Management District	Administration of Water Resources
14,578	Hydro Conduit, LLC (DBA Rinker Materials)	Concrete Block and Brick Manufacturing
10,311	Homestead General Aviation	Airport
10,152	Miami-Dade Water and Sewer Department - 8932 SW 232nd St	Sewage Treatment Plant
7,389	Miami-Dade County - South Dade Landfill	Landfill
7,032	FPT Florida LLC	Recyclable Material Merchant Wholesalers
5,670	Miami-Dade County - North Dade Landfill	Landfill
5,190	Miami-Dade Water and Sewer Department - 1100 W 2nd Ave	Water Supply and Irrigation Systems
4,727	Solo Printing, LLC	Printing
3,671	Miami-Dade Water and Sewer Department - 2575 NE 156th St	Sewage Treatment Plant
3,544	Miami-Dade Water and Sewer Department - 3869 Rickenbacker Causeway	Sewage Treatment Plant
3,182	Quikrete Companies, LLC	Concrete Product Manufacturing
1,739	Central Concrete Supermix, Inc.	Concrete Batch Plant
1,634	Derby Building Products, LLC	Plastics Pipe & Pipe Fitting
1,112	Trademark Metals Recycling LLC	Recyclable Material Merchant Wholesalers
914	Jackson Memorial Hospital	Hospital
719	Community Asphalt Corporation	Asphalt Plant
636	Kingspan Insulation, LLC	Plastics Product Manufacturing
630	Flowers Baking Company of Miami, LLC.	Bakery
620	Homestead Air Reserve Base	Military Base
588	Miami	Airport

558	U.S. Dept of Agriculture	Government agency
491	CSX Transportation - Hialeah	Rail Yard
489	Hector & Hector, Inc.	Institutional Furniture Manufacturing
483	Artco Group, Inc.	Wood Kitchen Cabinet and Countertop Manufacturing
226	Goodrich Corporation	Aerospace Product and Parts Manufacturing
224	Homestead City Utilities	Oil-burning Power Plant
170	Noven Pharmaceuticals, Inc.	Pharmaceutical Preparation Manufacturing
147	Hometown Foods USA, LLC	Bakery
86	Packaging Corporation of America	Corrugated and Solid Fiber Box Manufacturing
40	Delta Apparel, Inc.	Printing
33	Viking Kabinets, Inc.	Nonupholstered Wood Household Furniture Manufacturing
14	Sunlite Sales, Inc.	Metal Coating / Engraving
7	AAR Landing Gear Services	Aerospace Product and Parts Manufacturing
7	Heico Corporation	Aerospace Product and Parts Manufacturing
3	Asahi Refining Florida, Inc. - Miami Gardens	Secondary Smelting / Refining

Exhibit G

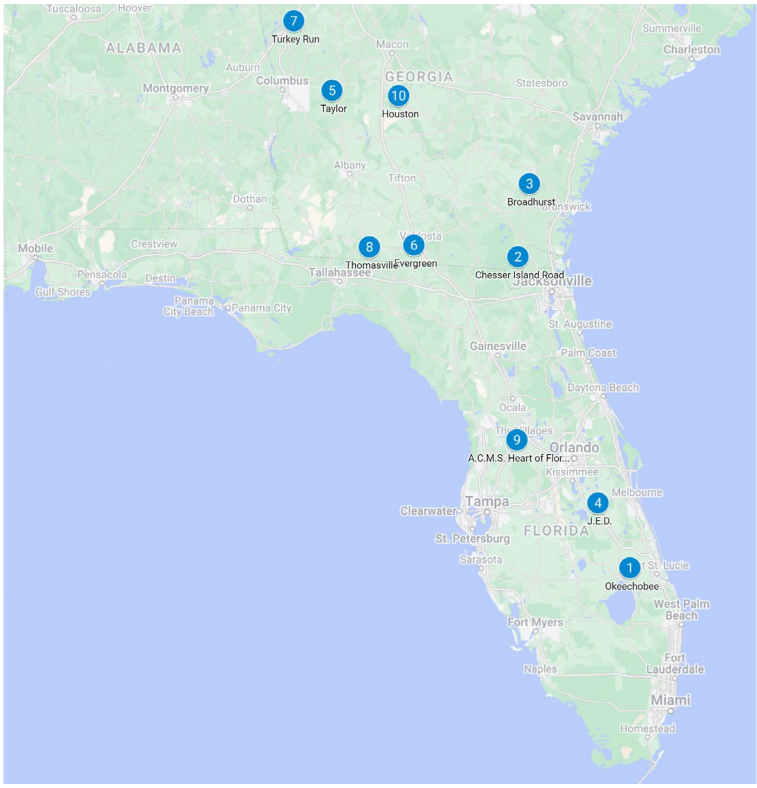
Most Responsible Landfill Options for Miami-Dade County

January 24, 2025



Operating Landfills in the Southeastern U.S. (ejmap.org)

➔



Most responsible landfill options for Miami-Dade County

Analysis of 63 Florida and Georgia landfills examining:

- Transportation Distance / Rail Access / Cost
- Available capacity
- Population impacted
- Environmental justice impacts
- Environmental compliance
- Landfill ownership
- Landfill gas management methods
- Rainfall (affecting landfill gas generation)
- Future availability as incinerators retire
- Acceptance of out-of-county municipal waste

Commissioned by The Goldstein Environmental Law Firm, P.A., on behalf of the City of Miramar, Florida

Principal Author:

Mike Ewall
Energy Justice Network
215-436-9511
mike@energyjustice.net
www.energyjustice.net

An online copy of this report is available at:
www.energyjustice.net/fl/landfills.pdf

BACKGROUND

Currently, and for the foreseeable future, Miami-Dade County's trash is going to landfills. It is important to pick the most responsible landfills while working to reduce landfilling impacts through Zero Waste management practices.

Incinerators (so called "waste-to-energy" or "WTE" facilities) do not replace landfills. For every 100 tons burned in an incinerator, close to 30 tons of toxic ash are produced which are then landfilled. Incineration makes landfills more toxic by making toxic chemicals in waste more available to be inhaled or ingested through air and water releases. This takes place when toxic elements like heavy metals in waste are spread into the air and also concentrated in fine ash that can blow off of trucks and off of the surface of a landfill, and can also impact groundwater more readily. Incineration also creates new toxic chemicals in the combustion process, including acid gases and highly toxic dioxins and furans which are both emitted into the air and concentrated in the ash. Incineration causes landfills to be smaller, but more toxic.

The most comprehensive life cycle assessments of waste systems have shown that incineration (and landfilling ash) is 2-3 times more harmful for human health and the environment (including climate impacts) than landfilling waste directly without burning it first.[1]

Miami-Dade County residents, businesses and tourists produce about 3.5 million tons of municipal solid waste (MSW) per year after recycling about 840,000 tons. Before the county's trash incinerator in Doral burned down in February 2023, it was burning about 520,000 tons per year, and the county was landfilling close to 3 million tons per year. It is unclear from state reporting data whether this landfilling figure includes the 150,000 tons of ash that would result from the 520,000 tons of trash burned at the incinerator.

If the county were to build a 4,000 ton/day trash incinerator, as is proposed, this would be the largest in the nation and would be capable of burning 40% of the annual tonnage currently being generated in Miami-Dade County. As the incinerator would take about 10 years to build, this percentage could change in either direction depending on waste generation trends and waste reduction policies and programs. Currently, the county is responsible for managing 40% of the county's waste generation, while the other 60% is managed by the private sector.

No new trash incinerator has been successfully sited, financed, constructed, and operated at a new site in the U.S. since 1995, despite hundreds of attempts. The building of a second incinerator adjacent to the existing incinerator at West Palm Beach, Florida is a rare exception where a new, expanded, or rebuilt incinerator was developed where an operating incinerator exists.

Even if Miami-Dade County were to succeed in being the first to build a new trash incinerator at a new site in the U.S. in 3-4 decades, the proposed incinerator would handle only 40% of the county's current annual waste generation. The other 60% (about 2 million tons/year currently handled by the

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[1] "Life Cycle Analysis: Incineration vs. Landfilling vs. Zero Waste," Appendix to Zero Waste Plan for Delaware County, Pennsylvania, 2024. https://www.energyjustice.net/files/incineration/DelcoLCA.pdf

private sector) would still need to go to landfills, as would the toxic ash from the waste burned at any new incinerator (about 400,000 tons/year). Even after building the nation's largest incinerator, with operation starting around 2035, the county will need a landfill location for about 2.5 million tons of waste per year for the remaining trash plus the incinerator's ash. Until then, the county will be landfilling the 3.5 million tons/year generated until Zero Waste efforts are implemented to reduce this amount.

While landfills are needed, even with a large incinerator, some existing ones are better than others, and county contracts can also move private landfill operators toward better landfill management practices in line with the Zero Waste Hierarchy.[2]

Building a new landfill in Miami-Dade County is also an option, but is not recommended for a few reasons. First, developing a new landfill is quite costly, though not as costly as a new incinerator. The large public investment required could go much further if invested in waste reduction, reuse, recycling and composting strategies that are higher in the Zero Waste Hierarchy. A new landfill is also a poor choice because it risks contaminating a new location, as opposed to utilizing existing landfills. Finally, Miami-Dade County, especially at the South Dade landfill site, faces flood risks from global warming and related extreme weather events that are becoming more common.

In evaluating the best landfills for Miami-Dade County's use, we examined the 63 landfills in Florida and Georgia, as far north as Atlanta. We looked at 18 metrics and grouped them into ten criteria that we weighted and combined into a single score with which to rank the most preferred landfills. The ten criteria are:

1. Transportation Distance / Rail Access / Cost
2. Available capacity
3. Population impacted
4. Environmental justice impacts
5. Environmental compliance
6. Landfill ownership
7. Landfill gas management methods
8. Rainfall (affecting landfill gas generation)
9. Future availability as incinerators retire
10. Acceptance of out-of-county municipal solid waste

The tenth criteria ruled out many landfills that do not accept municipal solid waste (MSW) from outside of their county or a small group of counties, reducing the 63 potential landfills to 30 that will be summarized in the following review of evaluation metrics.

1. Transportation Distance / Rail Access / Cost

Transportation distance and cost are related factors. A greater transportation distance can be a major factor in total cost of disposal, especially when fuel prices are high. Three options for transportation were evaluated: 1) trucking while avoiding toll roads, 2) trucking using available toll roads, and 3) rail.

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[2] "Zero Waste Hierarchy," Zero Waste International Alliance. http://www.zwia.org/zwh

Miami-Dade County has three trash transfer stations: Northeast, Central, and West.[3] To standardize the transportation distance analysis, the trucking distance to landfills was measured with Google Maps from the Central Transfer Station at 1150 NW 20th St, Miami, FL 33127.

Toll roads were an option for 22 of the 30 eligible landfills. For ten of those, the difference in distance was under ten miles, most of them actually a longer distance than taking non-toll roads. Another 11 landfills had toll road distances that were 24-33 miles shorter than the non-toll route. One landfill (Toombs County, GA) had a toll route that cut out 66 miles compared to the non-toll route. All of these are one-way distances.[4]

Rail transportation cuts transportation emissions by about half, but severely limits available landfill options, as few landfills have rail service. Only one landfill in Florida and Georgia has rail access. This is the Taylor County Landfill in Mauk, GA, which is served by CSX.[5]

Distance was scored using the non-toll routes such that a road miles distance of 400 miles one-way would be a score of zero, and longer distances would produce a negative score up to -1.4. Shorter distances earned closer landfills a positive score up to 2.0.

Data on transportation distance, rail access, and tip fees is available in Table 3.

a) Transportation emissions

Transportation distance to reach out-of-county landfills is not a significant environmental concern, as several life cycle assessment studies have shown, because the emissions from truck or rail transport are tiny relative to the emissions from landfills themselves, and even tinier relative to the much larger emissions from incineration.[6]

For example, the following chart shows an analysis of in-county incineration vs. ten out-of-state landfill options for Montgomery County, Maryland. Transportation climate impacts (by truck in blue or rail in black) are minor compared to incineration (red) or landfilling (yellow). The last bar represents the status quo, where waste is brought 18 miles by rail from the county's transfer station to their incinerator within the county, and then the ash brought much further (112 miles) to a landfill in Virginia, also by rail. All of the rail transportation is included in the black line capping the red bar which shows the greenhouse gas (GHG) emissions from the incinerator itself. The GHG emissions from the ten landfills are all represented in yellow, with all available by truck and a few also available

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[3] Miami-Dade County Solid Waste Management Department, "Regional Transfer Stations." https://www.miamidade.gov/global/service.page?Mduid_service=ser1464808248005568

[4] When evaluating trucking impacts, the return trips are not as fuel-intensive because long-haul trucks are returning empty and weigh much less.

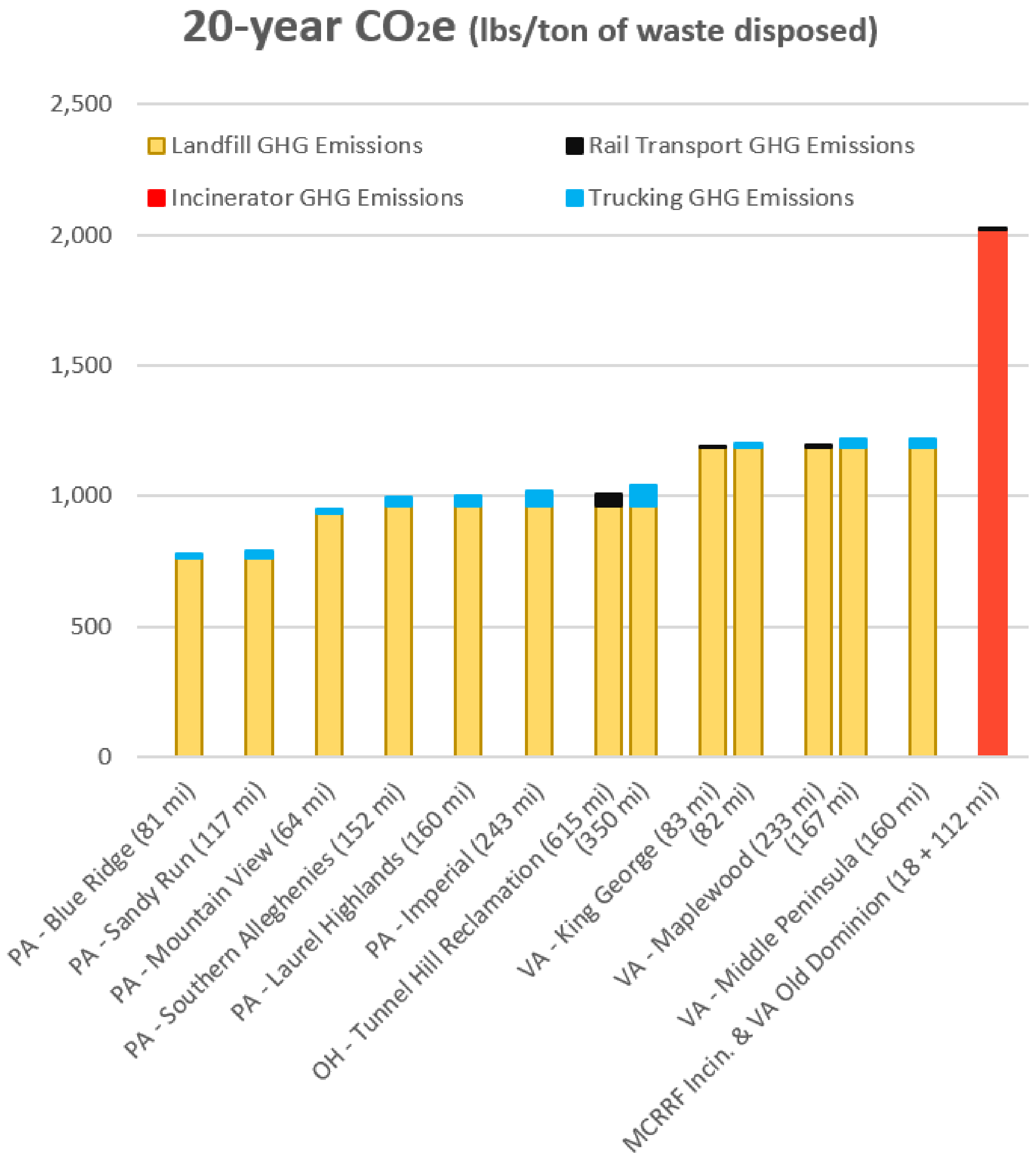
[5] https://www.csx.com/index.cfm/customers/commodities/waste/maps-locations/municipal-and-consumer-waste-map/
Note that only CSX and minor or passenger rail carriers service Miami-Dade County, so other major rail corporations offering waste disposal services are not operating in the region. See rail carrier map here: https://www.arcgis.com/apps/mapviewer/index.html?webmap=96ec03e4fc8546bd8a864e39a2c3fc41

[6] "Beyond Incineration: Best Waste Management Strategies for Montgomery County, Maryland," Zero Waste Montgomery County. See Life Cycle Assessment results in Chapter 7. https://www.energyjustice.net/md/moco

by rail. The round-trip GHG emissions from these diesel truck and train trips are represented with the blue and black lines showing slightly greater emissions for the longer distances.

The striking conclusion from this and similar studies is that no realistic transportation distance can use trucking emissions to justify incinerating in-county over transporting trash to distant landfills, as the gap between incineration and landfilling is so great that a truck would have to drive from the east coast to California and back to come close to closing the emissions gap.

Montgomery County, MD GHG analysis of incineration vs. landfill options



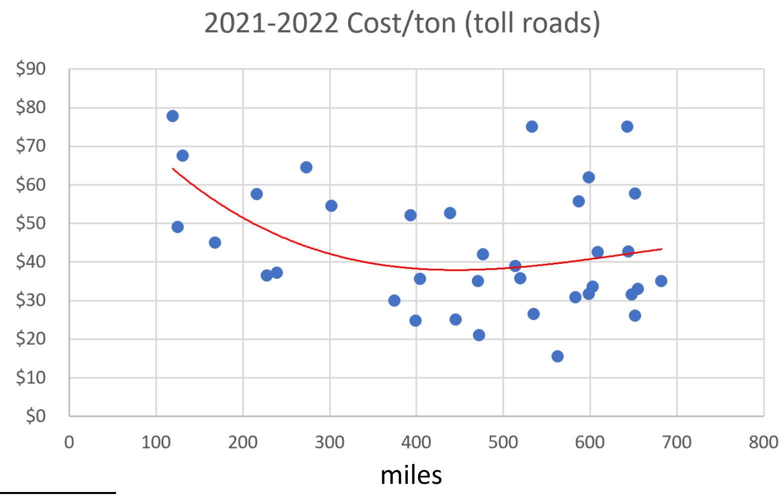
These results showing transportation impacts to be relatively minor have been reproduced in waste studies done for Delaware County, PA[7], Hawaiʻi County, HI[8], and Washington, DC, as well as studies of food systems.[9]

b) Transportation costs

Transportation distance can be a cost concern due to fuel prices, tire and truck wear, and driver turnaround time. The cost of transportation can be about as significant as landfill tipping fees. Landfill tipping fees get cheaper in Georgia compared to Central or North Florida, though, somewhat mitigating the cost of transportation.

Unfortunately, one cannot get reliable tipping fee data for landfills without issuing a Request for Proposals (RFP). Some large waste corporations will not provide quotes and will not respond to a Request for Information, but will provide bids in response to a competitive RFP. Generally, landfills will offer lower tip fees for long-term contracts and larger volumes.

Tip fee data for some landfills can be found through the EREF Tip Fee Report.[10] There is also tip fee data for Georgia's landfills available through the GEOS system via Georgia's Department of Natural Resources, Environmental Protection Division.[11] This data is gate fee prices, which tend to be higher than what can be obtained with a long-term county contract. While the data was not complete enough to be able to assess all 30 landfills, there is a trendline where greater distance from Miami lowers the average tip fee until approaching Atlanta, where prices start to rise a bit. Using 2022 data from EREF's report and filling in gaps with 2021 data from GEOS, the tipping fee trend is apparent and linked to distance from Miami (and Atlanta). The cheapest landfills are in the 375 to 600 miles range.



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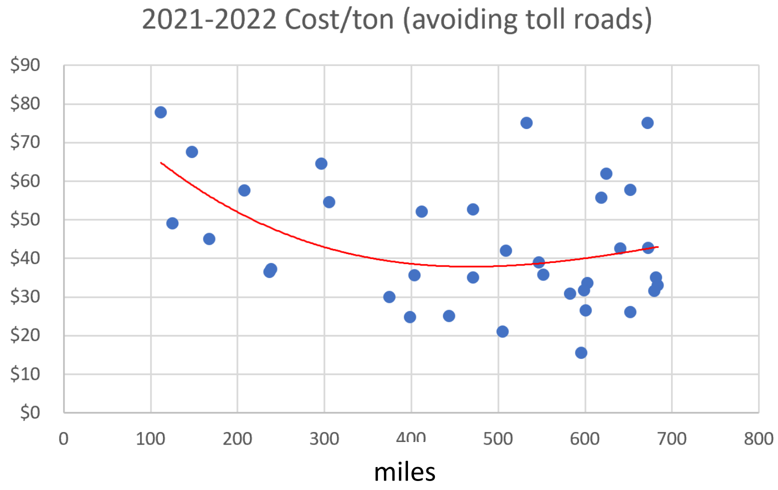
[7] See summary chart in page 7 here: https://www.energyjustice.net/files/incineration/DelcoLCA.pdf

[8] Parametrix, Life Cycle Assessment Technical Memorandum, Feb. 20, 2023. https://drive.google.com/file/d/1tdhufZvfyXM64OnU7Z9Bdfts-_xoptaq/view

[9] Poore, J., & Nemecek, T. (2018). Reducing food's environmental impacts through producers and consumers. Science, 360(6392), 987-992. http://www.researchgate.net/publication/325532198 Chart from: http://www.ourworldindata.org/food-choice-vs-eating-local

[10] "2022 Tip Fee Report," Environmental Research & Education Foundation. https://erefdn.org/analyzing-municipal-solid-waste-landfill-tipping-fees/

[11] "Georgia EPD Online System (GEOS) for Permitting, Compliance and Facility Information." https://geos.epd.georgia.gov/GA/GEOS/Public/GovEnt/Shared/Pages/Main/Login.aspx



2. Available capacity

Available capacity is one of the most important criteria, but is also ever-changing as landfills tend to expand over time. Data from EPA's Landfill Methane Outreach Program[12] provides the design capacity for most landfills, the amount of waste already in place, the amount landfilled in 2022, and the expected year of closure. From these, remaining capacity and remaining years were calculated for all landfills. Landfills earned scores of zero to four based on tons of remaining capacity, on having projected closure years after 2050, and on having more than 30 years of calculated capacity remaining based on current rates of waste acceptance.

Pending landfill expansion data is not readily available and could not be factored in. However, it should be noted that landfills that seem like they are out of capacity soon are likely in the process of obtaining permits for expansion. For example, the landfill that scored 9th highest in this analysis (A.C.M.S. / Heart of Florida Landfill in Sumter County, FL) has a zero score for capacity with only about 11 million tons of capacity remaining, which would last 10-12 years. However, that landfill would score 3rd or 4th once their proposed 60-acre expansion is approved.[13]

Capacity data is available in Table 4.

3. Population impacted

Landfills are not the best neighbors. As with trash incinerators, landfills often bring odors, truck traffic, dust, and pests, lower property values, and release air emissions that can increase risks of cancer and other health impacts among those closeby. Landfills such as those in Broward and Miami-Dade Counties are surrounded by large populations (a few with over 100,000 residents within a 3-mile radius), while more rural landfills have very few neighbors (8 of the 30 have fewer than 250 residents within a 3-mile radius, and one – J.E.D. Landfill – has zero). Landfills with greater populations scored lower in order to avoid impacting many people, with the most urban landfill scoring -4 due to the 3-

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[12] Landfill Methane Outreach Program, U.S. Environmental Protection Agency. https://www.epa.gov/lmop

[13] Heart of Florida Landfill Expansion. https://www.hoflenv.com/expansion

mile population of 158,662 people. 2020 U.S. Census Data was collected using the JusticeMap feature of EJmap.org.[14]

Population data is available in Table 6.

4. Environmental justice impacts

Title VI of the Civil Rights Act requires that a recipient of federal funds, such as Miami-Dade County must not take actions that have a discriminatory effect on racial minorities. This includes not selecting communities of color to receive millions of tons of waste annually. A decision to select such a community for an incinerator or as a landfill destination would be grounds for the filing of a Title VI Civil Rights Act complaint with EPA's Office of Civil Rights.

In order to avoid violating Title VI, and to avoid environmental justice impacts by race or class, 2020 U.S. Census data was collected for a 3-mile radius around landfills using the JusticeMap feature of EJmap.org. Environmental justice scores ranged from -2 to 1.4 based on how far racial composition and median household income varied from the national averages in the 2020 U.S. Census.

Environmental justice data is available in Table 6.

5. Environmental compliance

To measure each landfill's history of compliance with environmental laws, EPA's Enforcement and Compliance History Online (ECHO) database was used to evaluate compliance with the Clean Air Act, Clean Water Act, and the Resource Conservation and Recovery Act, as well as enforcement actions within the past five years, and the number of quarters in non-compliance over the last three years. These measures were combined into a compliance score that reduced the score by as much as 2.1 for non-compliant landfills based on the severity of their non-compliance.

Environmental compliance data is available in Table 5.

6. Landfill ownership

Outside of Miami-Dade County's own public landfills, the use of public landfills owned by other counties is a risk. Most publicly-owned landfills serve only the county that owns it, or a group of counties where it's a regional partnership. These have already been excluded through criteria #10 below, but the use of public landfills that accept out-of-county waste is risky because some have been known to abruptly close their doors to out-of-county waste due to public opposition, and the same could happen when waste disposal markets tighten and the county needs the landfill space for its own use. Privately-owned landfills can more reliably be expected to welcome waste from anywhere, so they are given preference (1 point toward the score) in this analysis. Since in-county county-owned landfills are also preferable, they're given one point as well.

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[14] JusticeMap, EJmap.org. https://ejmap.org/justice/

A smaller preference in the scoring (0.5) was given to landfills owned by smaller waste corporations outside of the two large waste disposal monopolies, since supporting a diversity of players in the market enhances competition and should help keeps costs at all landfills lower than where monopolistic practices enable landfill owners to charge higher tip fees.

Ownership data is available in Table 2.

7. Landfill gas management methods

Landfill impacts can vary based on how landfill gas is managed. Landfill gas is about half methane, half CO_2, and is contaminated with hundreds of toxic chemicals. Because of the toxicity of the contaminants, modern landfills are required to capture the gas. Historically that involved flaring off the gas, but many landfills now burn the captured gas for energy. Some inject into gas pipelines or use it to fuel gas-powered vehicles.

Most landfills now collect their gas, and those which burn for energy, particularly with internal combustion engines, are far more polluting than those which flare their gas. There are also concerns with how landfills are managed when operators seek to produce energy by maximizing gas generation, and manipulating the landfill in order to increase the proportion of methane in the gas. These practices reduce gas collection efficiency and cause more gas to escape than if the landfill were simply flaring its gas and managing to minimize gas formation and maximize gas collection.[15]

Scores were assigned based on landfill gas management practices, with a negative score for those that do not have gas capture systems in place, and with varying scores for how captured gas is burned, as follows:

Scoring:

1 = Flaring
0.85 = Direct thermal combined with flaring
0.7 = Cogeneration
0.5 = Pipeline injection, vehicle fuel, or leachate evaporation
0 = Internal combustion engines, gas turbines, or steam turbines
-1 = No gas collection system installed

Landfills using leachate recirculation, which keeps the landfill wet by cycling the leachate back into the landfill, causing more gas generation, were penalized with a score of -0.5 to -1 based on the frequency of leachate recirculation.

Landfill gas management data is available in Table 5.

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[15] See links to resources on landfill gas emissions in the top and sidebar at https://www.energyjustice.net/lfg and recommendations for better landfill management in the Zero Waste Hierarchy at https://www.energyjustice.net/zerowaste/hierarchy

8. Rainfall

Rainfall impacts landfill gas generation. Less rainfall means less gas generation, reduced leachate, and fewer odor problems.

Multiple rainfall maps were used to evaluate differences in rainfall for each landfill community.[16]

Scoring:

0.2 = 40.1-50 inches/year (light)
0.1 = close to the boundary between 40.1-50 and 50.1-60 inches/year (medium-light)
0 = 50.1-60 inches/year (medium)
-0.2 = 60.1-70 inches/year (high)

Rainfall scores are available in Table 5.

9. Future availability as incinerators retire

A few landfills (which wouldn't have been among the top dozen scoring landfills, anyway) were scored lower because they exist in areas where the waste market is likely to be tight due to proximity to aging trash incinerators. Except for the new incinerator at West Palm Beach, the state's nine trash incinerators are past their prime. These eight older incinerators are between 30 and 42 years old, averaging 36 years. Of the 52 commercial trash incinerators in the U.S. that have closed since 2000, their average age at closure is just 25 years.[17] It's rare that incinerators make it past the age of 40. As these incinerators continue to close as they reach their end of life, area landfill capacity will be needed to absorb that waste. Because of this, public landfills in Broward, Polk, and Manatee Counties were scored lower because the county hosts an old incinerator (-2) or sits adjacent to a county that does (-1).

Scoring for proximity to aging incinerators is in Table 3.

10. Acceptance of out-of-county municipal solid waste

Of the 63 potential landfills evaluated (all of those in Florida, and landfills in Georgia as far north as Atlanta), 33 were of them were eliminated from consideration because they are not accepting out-of-county municipal solid waste. Nearly all of these are publicly-owned landfills.

This was determined using state environmental agency databases and reports on waste types and tonnages accepted at each facility. These reports include the county-of-origin, making it possible to screen out those that are not accepting out-of-county MSW.

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[16] National Oceanic and Atmospheric Association. https://www.climate.gov/media/13728; http://upload.wikimedia.org/wikipedia/commons/d/d3/Average_precipitation_in_the_lower_48_states_of_the_USA.png; Florida Average Yearly Rainfall. https://www.eldoradoweather.com/climate/us-states/florida-average-rainfall.html; Georgia Average Yearly Rainfall. https://www.eldoradoweather.com/climate/us-states/georgia-average-rainfall.html

[17] "Incinerator Closures 2000-2024," Energy Justice Network. https://www.energyjustice.net/incineration/closures.pdf

CONCLUSIONS

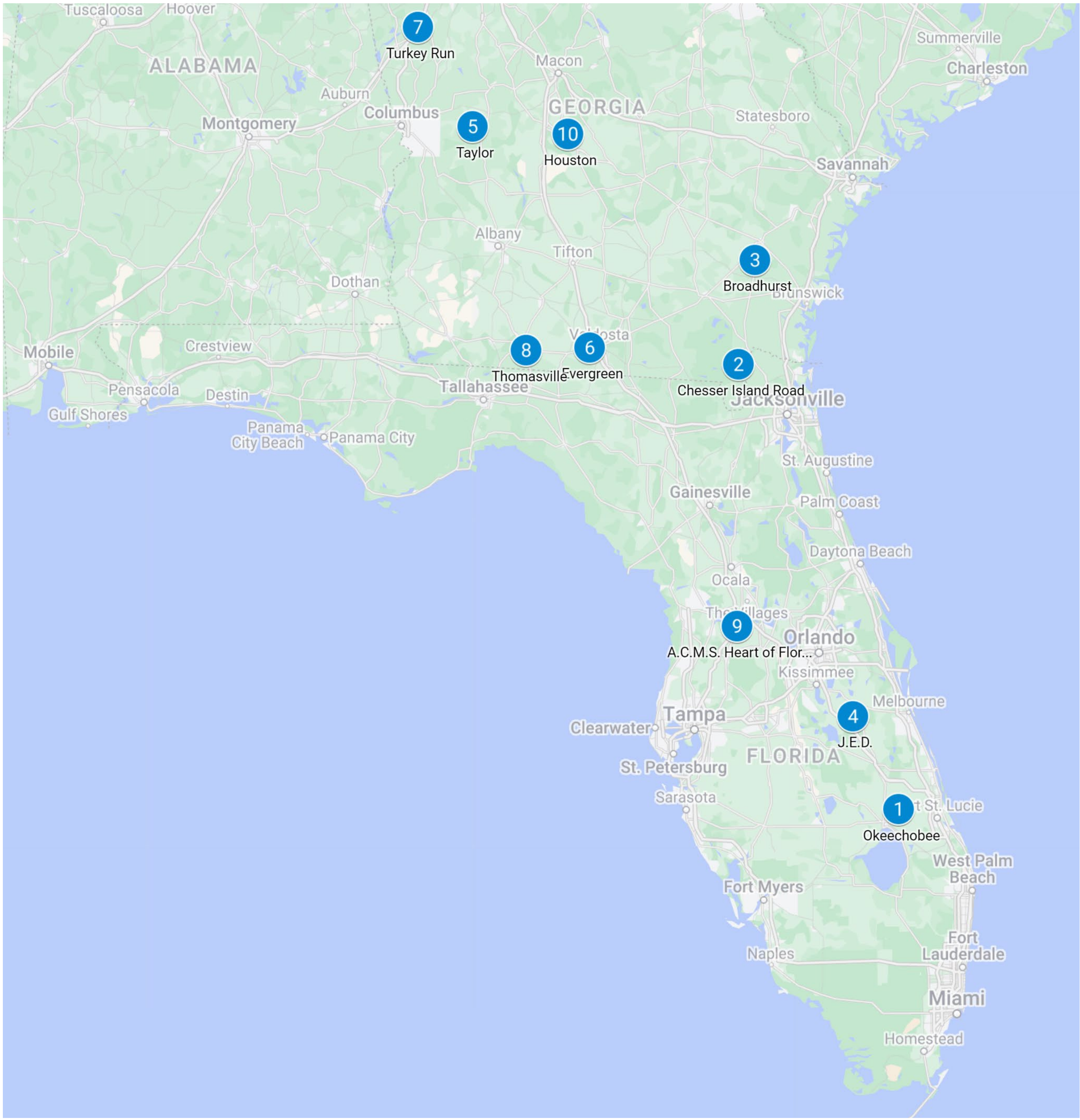
There are many viable landfills available as options for Miami-Dade County, some better than others. Okeechobee Landfill scored highest when combining all of the metrics outlined in this report, followed by Chesser Island and Broadhurst Landfills in Georgia, J.E.D. Landfill in Osceola County, FL, and Taylor County Landfill rounding out the top five. Okeechobee Landfill excelled largely due to its large available capacity, but also happens to be the closest of the top ten. J.E.D. Landfill is notable as the only one with no one living within three miles, yet it's also the one with the worst environmental compliance track record. A.C.M.S. Landfill came in 10th largely for its lack of capacity, but is in the process of expanding, which could elevate it to 3rd of 4th place among the options.

All three operating landfills in Miami-Dade County ranked poorly due to lack of capacity, proximity to large populations of nearby residents, being located in environmental justice communities (where a county decision to expand the landfill would be vulnerable to a legal complaint under Title VI of the Civil Rights Act), and/or long-term flood risk.

Below is a map of the top ten landfill options, followed Table 1, which summarizes the scores, and Tables 2-6 which provide the details on each landfill, much of which fed into the scores they received. All tables are sorted according to the scores in Table 1.

We recommend that Miami-Dade County issue separate requests for proposals (RFP) for hauling and disposal. Long-term contracts get lower prices from landfills. Criteria such as those used in this analysis should be used to score proposals for different landfills. No "put-or-pay" clause or minimum commitment of waste should be in disposal contracts, as it will stand in the way of Zero Waste efforts.

Top 10 Landfill Options for Miami-Dade County



View this map online here:
https://www.google.com/maps/d/u/0/viewer?mid=1CS9OuxPG41pOintuduLMLveFJiKKxlw

View maps of all landfills in Florida and Georgia here:
https://ejmap.org/Florida (choose layers: Landfill - Operating & Expand)
https://ejmap.org/Georgia (choose layers: Landfill - Operating & Expand)

Table 1: Top Scoring Landfills in Florida and Georgia

St	County	Landfill	Takes out-of-county MSW	RAINFALL SCORE	Close to incinerators	LANDFILL GAS MGMT SCORE	CAPACITY SCORE	OWNERSHIP SCORE	DISTANCE SCORE	RAIL ACCESS (CSX)	COMPLIANCE SCORE	POPULATION SCORE	EJ SCORE	OVERALL SCORE	RANK
FL	Okeechobee	Okeechobee / Berman Road Landfill	Y	0.2		0.5	4	1	1.4	0	0.0	0.00	0.1	7.2	1
GA	Charlton	Chesser Island Road Landfill	Y	0		1	3	1	0.0	0	0.0	0.00	1.1	6.1	2
GA	Wayne	Broadhurst Environmental Landfill	Y	0.1		1	3	1	-0.2	0	0.0	0.00	0.7	5.6	3
FL	Osceola	J.E.D. Solid Waste Management Facility	Y	0.1		-0.5	3	1.5	1.0	0	-2.1	0.00	1.0	4.0	4
GA	Taylor	WI Taylor County Disposal, LLC	Y	0.2		-1	1.5	1.5	-1.1	1	0.0	0.00	0.5	2.6	5
GA	Lowndes	Evergreen / Pecan Row Landfill	Y	0		0	2	1	-0.4	0	0.0	-0.06	-0.1	2.5	6
GA	Meriwether	Turkey Run Landfill	Y	0		0.3	2.5	1	-1.4	0	0.0	-0.02	0.1	2.4	7
GA	Thomas	City of Thomasville MSW Landfill	Y	0		1	1	0	-0.5	0	0.0	-0.08	0.4	1.8	8
FL	Sumter	A.C.M.S. / Heart of Florida Environmental	Y	0.1		-1	0	1.5	0.5	0	0.0	-0.04	0.6	1.7	9
GA	Houston	Houston County SR247 MSW Landfill	Y	0.2		-0.7	2	0	-1.0	0	0.0	0.00	0.8	1.3	10
GA	Twiggs	Wolf Creek Landfill	Y	0.2		0	2	1	-1.1	0	0.0	0.00	-1.0	1.1	11
FL	Jackson	Springhill Regional Landfill	Y	0		-0.2	2.5	1	-1.0	0	-0.1	-0.01	-1.3	1.0	12
FL	Miami-Dade	South Dade Solid Waste Disposal Facility	(N)	0		1	0	1	1.9	0	0.0	-1.43	-1.6	0.9	13
FL	Orange	Orange County Solid Waste Landfill	Y	0.1		0	1.5	0	0.8	0	-0.5	-0.36	-0.8	0.8	14
FL	Manatee	Manatee County / Lena Road Landfill	Y	0	-1	0.85	0	0	0.8	0	-1.0	-0.67	1.4	0.4	15
GA	Butts	Pine Ridge Landfill	Y	0		0	0	1	-1.3	0	0.0	-0.05	0.5	0.2	16
FL	Volusia	Tomoka Farms Road Landfill	Y	0.1		1	0	0	0.7	0	-1.8	-0.10	0.2	0.2	17
GA	Dougherty	Fleming/Gaissert Road Landfill	Y	0		0.7	2	0	-0.7	0	-1.4	-0.03	-0.5	0.1	18
GA	Crisp	Crisp County Landfill	Y	0.2		-2	2	0	-0.8	0	0.0	-0.01	0.5	-0.1	19
FL	Polk	North Central Landfill	Y	0.2	-1	-0.2	0	0	0.8	0	0.0	-0.22	0.3	-0.1	20
FL	Union	New River Regional Landfill	Y	0		-0.5	0	0	0.1	0	0.0	-0.07	0.0	-0.5	21
FL	Santa Rosa	Santa Rosa Central LF	Y	-0.2		1	1	0	-1.4	0	-1.7	-0.31	0.7	-0.9	22
GA	Atkinson	Atkinson County - SR 50 MSW Landfill	Y	0.1		-1.7	0	0	-0.3	0	0.0	-0.01	0.8	-1.2	23
GA	Camden	Camden County SR 110 MSW Landfill	Y	0		-1	0	0	0.0	0	-1.1	0.00	0.6	-1.5	24
GA	Toombs	Toombs County MSW Landfill	Y	0.2		-1	0	0	-1.0	0	-0.2	-0.03	0.2	-1.8	25
FL	Miami-Dade	North Dade Landfill	(N)	0		1	0	1	1.9	0	0.0	-3.97	-2.0	-2.1	26
FL	Madison	Aucilla Area Solid Waste Facility	Y	0		-2	1	0	-0.4	0	0.0	-0.03	-1.0	-2.4	27
FL	Broward	Monarch Hill Landfill	Y	0	-2	0	0	1	1.8	0	0.0	-2.50	-0.8	-2.5	28
GA	Lamar	Cedar Grove Landfill	Y	0		-1.5	0	0	-1.3	0	-0.1	-0.05	0.3	-2.6	29
FL	Miami-Dade	Medley Landfill	(Y)	0		0.5	0	1	2.0	0	-1.0	-3.43	-1.7	-2.6	30

Min	**-0.2**	**-2**	**-2**	**0**	**0**	**-1.4**	**0**	**-2.1**	**-4**	**-2.0**	**-2.6**
Max	**0.2**	**-1**	**1**	**4**	**1.5**	**2.0**	**1**	**0**	**0**	**1.4**	**7.2**
Range (weight)	**0.4**	**1**	**3**	**4**	**1.5**	**3.4**	**1**	**2.1**	**4**	**3.4**	**9.8**

Table 2: Landfill Location & Ownership

RANK	St	County	Landfill	Landfill Owner	Ownership Type	Ownership	Smaller company	OWNERSHIP SCORE
1	FL	Okeechobee	Okeechobee / Berman Road Landfill	WM	Private	1		1
2	GA	Charlton	Chesser Island Road Landfill	WM	Private	1		1
3	GA	Wayne	Broadhurst Environmental Landfill	Republic Services, Inc.	Private	1		1
4	FL	Osceola	J.E.D. Solid Waste Management Facility	Waste Connections, Inc.	Private	1	0.5	1.5
5	GA	Taylor	WI Taylor County Disposal, LLC	GFL Environmental USA Inc.	Private	1	0.5	1.5
6	GA	Lowndes	Evergreen / Pecan Row Landfill	WM	Private	1		1
7	GA	Meriwether	Turkey Run Landfill	WM	Private	1		1
8	GA	Thomas	City of Thomasville MSW Landfill	City of Thomasville, GA	Public			0
9	FL	Sumter	A.C.M.S. / Heart of Florida Environmental	Waste Connections, Inc.	Private	1	0.5	1.5
10	GA	Houston	Houston County SR247 MSW Landfill	Houston County Board of Commissioners, GA	Public			0
11	GA	Twiggs	Wolf Creek Landfill	WM	Private	1		1
12	FL	Jackson	Springhill Regional Landfill	WM	Private	1		1
13	FL	Miami-Dade	South Dade Solid Waste Disposal Facility	Miami-Dade County, FL	Public	1		1
14	FL	Orange	Orange County Solid Waste Landfill	Orange County Utilities, Solid Waste Division, FL	Public			0
15	FL	Manatee	Manatee County / Lena Road Landfill	Manatee County, FL	Public			0
16	GA	Butts	Pine Ridge Landfill	Republic Services, Inc.	Private	1		1
17	FL	Volusia	Tomoka Farms Road Landfill	Volusia County, FL	Public			0
18	GA	Dougherty	Fleming/Gaissert Road Landfill	Dougherty County Solid Waste Department, GA	Public			0
19	GA	Crisp	Crisp County Landfill	Crisp County, GA	Public			0
20	FL	Polk	North Central Landfill	Polk County Board of County Commissioners, FL	Public			0
21	FL	Union	New River Regional Landfill	New River Solid Waste Association, FL	Public			0
22	FL	Santa Rosa	Santa Rosa Central LF	Santa Rosa County, FL	Public			0
23	GA	Atkinson	Atkinson County - SR 50 MSW Landfill	Atkinson County, GA	Public			0
24	GA	Camden	Camden County SR 110 MSW Landfill	Camden County, GA	Public			0
25	GA	Toombs	Toombs County MSW Landfill	Toombs County, GA	Public			0
26	FL	Miami-Dade	North Dade Landfill	Miami-Dade County, FL	Public	1		1
27	FL	Madison	Aucilla Area Solid Waste Facility	Aucilla Area Solid Waste Administration, FL	Public			0
28	FL	Broward	Monarch Hill Landfill	WM	Private	1		1
29	GA	Lamar	Cedar Grove Landfill	Lamar County, GA	Public			0
30	FL	Miami-Dade	Medley Landfill	WM	Private	1		1

Table 3: Landfill Location, Tipping Fees, Distance, Rail Access, and Proximity to Aging Incinerators

RANK	St	County	Landfill	2022 Tipping fee ($/ton)	2021 Tipping fee ($/ton)	Distance (toll)	Distance (non-toll)	DISTANCE SCORE	RAIL ACCESS (CSX)	Close to incinerators
1	FL	Okeechobee	Okeechobee / Berman Road Landfill			128	130	1.4		
2	GA	Charlton	Chesser Island Road Landfill	$35.61		n/a	404	0.0		
3	GA	Wayne	Broadhurst Environmental Landfill		$25.00	445	444	-0.2		
4	FL	Osceola	J.E.D. Solid Waste Management Facility			190	199	1.0		
5	GA	Taylor	WI Taylor County Disposal, LLC		$55.59	587	619	-1.1	1	
6	GA	Lowndes	Evergreen / Pecan Row Landfill		*$52.57*	439	471	-0.4		
7	GA	Meriwether	Turkey Run Landfill		$34.98	n/a	682	-1.4		
8	GA	Thomas	City of Thomasville MSW Landfill		$21.00	472	505	-0.5		
9	FL	Sumter	A.C.M.S. / Heart of Florida Environmental	$64.47		273	297	0.5		
10	GA	Houston	Houston County SR247 MSW Landfill		$15.50	563	596	-1.0		
11	GA	Twiggs	Wolf Creek Landfill		*$61.83*	599	625	-1.1		
12	FL	Jackson	Springhill Regional Landfill			565	595	-1.0		
13	FL	Miami-Dade	South Dade Solid Waste Disposal Facility			29.4	23.6	1.9		
14	FL	Orange	Orange County Solid Waste Landfill	$37.10		n/a	239	0.8		
15	FL	Manatee	Manatee County / Lena Road Landfill			239	232	0.8		-1
16	GA	Butts	Pine Ridge Landfill		$57.75	n/a	652	-1.3		
17	FL	Volusia	Tomoka Farms Road Landfill			253	259	0.7		
18	GA	Dougherty	Fleming/Gaissert Road Landfill		$38.97	514	547	-0.7		
19	GA	Crisp	Crisp County Landfill		$35.75	520	552	-0.8		
20	FL	Polk	North Central Landfill	$36.50		228	237	0.8		-1
21	FL	Union	New River Regional Landfill			n/a	389	0.1		
22	FL	Santa Rosa	Santa Rosa Central LF			655	687	-1.4		
23	GA	Atkinson	Atkinson County - SR 50 MSW Landfill	$35.00		471	468	-0.3		
24	GA	Camden	Camden County SR 110 MSW Landfill	$24.74		399	398	0.0		
25	GA	Toombs	Toombs County MSW Landfill		$26.50	535	601	-1.0		
26	FL	Miami-Dade	North Dade Landfill			n/a	18	1.9		
27	FL	Madison	Aucilla Area Solid Waste Facility			443	472	-0.4		
28	FL	Broward	Monarch Hill Landfill			n/a	38.9	1.8		-2
29	GA	Lamar	Cedar Grove Landfill		$26.00	n/a	652	-1.3		
30	FL	Miami-Dade	Medley Landfill			14.3	9.9	2.0		

Italicized prices are an average over the year.

Table 4: Landfill Open and Projected Closure Years, Capacity and Waste Acceptance Data

RANK	Landfill	Year Landfill Opened	Landfill Closure Year	Landfill Design Capacity (tons)	Waste in Place (tons)	Remaining capacity (tons)	Annual Waste Acceptance Rate (2022 tons)	Remaining years calculated	Remaining capacity	Closure year > 2050	30+ remaining years	CAPACITY SCORE
1	Okeechobee / Berman Road Landfill	1981	2078	241,553,464	37,538,675	204,014,789	2,203,649	91	2	1	1	4
2	Chesser Island Road Landfill	1992	2061	77,840,654	21,482,339	56,358,315	1,455,159	37	1	1	1	3
3	Broadhurst Environmental Landfill	1993	2165	64,334,141	14,363,827	49,970,314	338,550	146	1	1	1	3
4	J.E.D. Solid Waste Management Facility	2004	2056	86,351,655	27,923,508	58,428,147	1,277,541	44	1	1	1	3
5	WI Taylor County Disposal, LLC	1989	2037	51,117,057	18,300,772	32,816,285	360,826	89	0.5		1	1.5
6	Evergreen / Pecan Row Landfill	1992	2081	30,231,984	13,705,888	16,526,096	343,023	46		1	1	2
7	Turkey Run Landfill	2010	2056	28,973,656	6,366,244	22,607,412	696,691	30	0.5	1	1	2.5
8	City of Thomasville MSW Landfill	1975	2085	6,812,419	3,883,896	2,928,523	144,145	18		1		1
9	A.C.M.S. / Heart of Florida Environmental	2013	2035	19,337,669	5,764,620	13,573,049	995,902	12				0
10	Houston County SR247 MSW Landfill	1987	2227	21,780,180	4,120,888	17,659,292	213,369	81		1	1	2
11	Wolf Creek Landfill	1992	2064	23,191,228	7,727,092	15,464,136	374,712	39		1	1	2
12	Springhill Regional Landfill	1983	2069	52,837,321	17,443,060	35,394,261	852,060	40	0.5	1	1	2.5
13	South Dade Solid Waste Disposal Facility	1979	2036	21,184,000	20,932,186	251,814	581,817					0
14	Orange County Solid Waste Landfill	1972	2088	64,559,937	36,079,131	28,480,806	1,159,080	23	0.5	1		1.5
15	Manatee County / Lena Road Landfill	1972	2041	20,947,500	11,678,873	9,268,627	346,218	25				0
16	Pine Ridge Landfill	1997	2041	37,266,389	25,610,923	11,655,466	954,365	10				0
17	Tomoka Farms Road Landfill	1977	2029	18,214,658	16,498,244	1,716,414	623,196	1				0
18	Fleming/Gaissert Road Landfill	1983	2050	8,916,325	4,831,285	4,085,040	97,777	40		1	1	2
19	Crisp County Landfill	1973	2104	8,556,515	2,064,383	6,492,132	140,090	44		1	1	2
20	North Central Landfill	1977	2027	25,308,136	20,352,370	4,955,766	721,083	5				0
21	New River Regional Landfill	1992	2025		6,296,430		294,851					0
22	Santa Rosa Central LF	1978	2055	5,512,500	5,055,404	457,096	336,687	-1		1		1
23	Atkinson County - SR 50 MSW Landfill	1997	2042	2,832,219	1,353,412	1,478,807	100,305	13				0
24	Camden County SR 110 MSW Landfill	1992	2033	3,184,719	2,460,285	724,434	100,533	5				0
25	Toombs County MSW Landfill											0
26	North Dade Landfill	1952	2025		14,775,246		197,780					0
27	Aucilla Area Solid Waste Facility	1992	2053	3,039,272	1,463,511	1,575,761	58,719	25		1		1
28	Monarch Hill Landfill	1965	2030	84,293,492	77,055,682	7,237,810	1,516,359	3				0
29	Cedar Grove Landfill	1984	2041	5,529,420	1,907,284	3,622,136	233,421	14				0
30	Medley Landfill	1980	2030	44,636,263	35,560,166	9,076,097	1,379,566	5				0

Table 5: Landfill Environmental Compliance, Rainfall, and Landfill Gas Management

RANK	Landfill	Compliance [ECHO]	Actions in last 5 yrs [ECHO]	# of Quarters in Non-Compliance (3 yrs)	COMPLIANCE SCORE	RAINFALL SCORE	Leachate Recirculation	Leachate Recirculation Frequency (Past 10 Yrs)	Landfill Gas Collection System?	Flares in Place?	Landfill Gas Management	LANDFILL GAS MANAGEMENT SCORE
1	Okeechobee / Berman Road Landfill	0	0.0	0	0	0.2	No	Not used	Yes	Yes	Pipeline Inject.	0.5
2	Chesser Island Road Landfill	0	0.0	0	0	0	No	Not used	Yes	Yes	Flaring	1.0
3	Broadhurst Environmental Landfill	0	0.0	0	0	0.1	No	Not used	Yes	Yes	Flaring	1.0
4	J.E.D. Solid Waste Management Facility	2	1.5	7	-2.1	0.1	Yes	Several/year	Yes	Yes	Leachate evap.; LFGTE-ICE	-0.5
5	WI Taylor County Disposal, LLC	0	0.0	0	0	0.2	Yes	Several/year	Yes	Yes	LFGTE-ICE	-1.0
6	Evergreen / Pecan Row Landfill	0	0.0	0	0	0	No	Not used	Yes	Yes	LFGTE-ICE	0.0
7	Turkey Run Landfill	0	0.0	0	0	0	Yes	< Once/year	Yes	Yes	Flaring	0.3
8	City of Thomasville MSW Landfill	0	0.0	0	0	0	No	Not used	Yes	Yes	Flaring	1.0
9	A.C.M.S. / Heart of Florida Environmental	0	0.0	0	0	0.1	No	Not used	No	No		-1.0
10	Houston County SR247 MSW Landfill	0	0.0	0	0	0.2	Yes	< Once/year	Yes	Yes	LFGTE-ICE	-0.7
11	Wolf Creek Landfill	0	0.0	0	0	0.2	No	Not used	Yes	Yes	LFGTE-ICE	0.0
12	Springhill Regional Landfill	0	0.0	2	-0.1	0	Yes	< Once/year	Yes	Yes	Leachate evap.; LFGTE-ICE	-0.2
13	South Dade Solid Waste Disposal Facility	0	0.0	0	0	0	No	Not used	Yes	Yes	Flaring	1.0
14	Orange County Solid Waste Landfill	0	0.0	10	-0.5	0.1	No	Not used	Yes	Yes	LFGTE-ST	0.0
15	Manatee County / Lena Road Landfill	0	2.0	0	-1	0	No	Not used	Yes	Yes	Direct Thermal / Flaring	0.9
16	Pine Ridge Landfill	0	0.0	0	0	0	No	Not used	Yes	Yes	LFGTE-ICE	0.0
17	Tomoka Farms Road Landfill	3	0.0	5	-1.75	0.1	No	Not used	Yes	Yes	Flaring	1.0
18	Fleming/Gaissert Road Landfill	1	0.5	12	-1.35	0	No	Not used	Yes	Yes	LFGTE-Cogen	0.7
19	Crisp County Landfill	0	0.0	0	0	0.2	Yes	Several/year	No	No		-2.0
20	North Central Landfill	0	0.0	0	0	0.2	Yes	< Once/year	Yes	Yes	Vehicle Fuel	-0.2
21	New River Regional Landfill	0	0.0	0	0	0	Yes	Several/year	Yes	Yes	Pipeline Inject.	-0.5
22	Santa Rosa Central LF	0	3.0	3	-1.65	-0.2	No	Not used	Yes	Yes	Flaring	1.0
23	Atkinson County - SR 50 MSW Landfill	0	0.0	0	0	0.1	Yes	< Once/year	No	No		-1.7
24	Camden County SR 110 MSW Landfill	1	0.0	12	-1.1	0	No	Not used	No	No		-1.0
25	Toombs County MSW Landfill	0	0.0	4	-0.2	0.2		Not used				-1.0
26	North Dade Landfill	0	0.0	0	0	0	No	Not used	Yes	Yes	Flaring	1.0
27	Aucilla Area Solid Waste Facility	0	0.0	0	0	0	Yes	Several/year	No	No		-2.0
28	Monarch Hill Landfill	0	0.0	0	0	0	No	Not used	Yes	Yes	LFGTE-GT	0.0
29	Cedar Grove Landfill	0	0.0	2	-0.1	0	Yes	> Once/year	No	No		-1.5
30	Medley Landfill	2	0.0	0	-1	0	No	Not used	Yes	Yes	Pipeline Inject.	0.5

LFGTE = Landfill gas-to-energy; GT = Gas turbine; ICE = Internal Combustion Engine; ST = Steam turbine

Table 6: Landfill Community Population, Race and Income Data

RANK	Landfill	Median Household Income (census block)	3 mile population	Indigenous %	Black %	Hispanic / Latine %	White %	POPULATION SCORE	ENVIRONMENTAL JUSTICE SCORE
1	Okeechobee / Berman Road Landfill	$57,826	20	0	0	25	65	0.00	0.14
2	Chesser Island Road Landfill	$49,053	174	0	0	2.3	96.6	0.00	1.11
3	Broadhurst Environmental Landfill	$40,375	125	0.8	0	5.6	86.4	0.00	0.68
4	J.E.D. Solid Waste Management Facility		0					0.00	1.00
5	WI Taylor County Disposal, LLC	$20,781	100	1	8	5	86	0.00	0.47
6	Evergreen / Pecan Row Landfill	$73,438	2,446	0.5	36.4	5.8	53.4	-0.06	-0.09
7	Turkey Run Landfill	$39,325	880	0.6	23.9	3.5	67.8	-0.02	0.05
8	City of Thomasville MSW Landfill	$66,667	3,394	0.6	21.1	3.9	70.9	-0.08	0.43
9	A.C.M.S. / Heart of Florida Environmental	$51,707	1,673	0.8	4.2	9.9	80.4	-0.04	0.60
10	Houston County SR247 MSW Landfill	$85,469	175	0	10.9	5.1	76	0.00	0.79
11	Wolf Creek Landfill	$47,837	182	0	58.2	6.6	34.6	0.00	-0.97
12	Springhill Regional Landfill	$31,649	469	0.6	59.9	2.3	30.9	-0.01	-1.26
13	South Dade Solid Waste Disposal Facility		57,196	0.4	17.6	68.5	10.6	-1.43	-1.57
14	Orange County Solid Waste Landfill	$69,026	14,595	0.5	12.6	41.9	33.8	-0.36	-0.78
15	Manatee County / Lena Road Landfill	$138,051	26,922	0.2	3.1	10.7	78.2	-0.67	1.39
16	Pine Ridge Landfill	$54,063	2,110	0.5	7.8	7.3	77.8	-0.05	0.53
17	Tomoka Farms Road Landfill	$55,770	4,164	0.4	8.9	10.3	68.7	-0.10	0.25
18	Fleming/Gaissert Road Landfill	$56,912	1,298	0	42.8	7.6	46	-0.03	-0.50
19	Crisp County Landfill	$48,565	486	0	17.7	2.9	78	-0.01	0.48
20	North Central Landfill	$54,712	8,829	0.9	2.7	22.8	70.3	-0.22	0.29
21	New River Regional Landfill	$57,045	2,908	0.2	31.7	5.7	61.1	-0.07	0.01
22	Santa Rosa Central LF	$66,571	12,451	0.8	5.6	5.5	79	-0.31	0.70
23	Atkinson County - SR 50 MSW Landfill	$43,292	215	1.4	1.4	10.2	88.4	-0.01	0.78
24	Camden County SR 110 MSW Landfill	$42,875	157	0	8.3	4.5	84.1	0.00	0.63
25	Toombs County MSW Landfill	$36,369	1,216	0.5	9.4	13.2	74.1	-0.03	0.23
26	North Dade Landfill	$39,096	158,662	0.2	34.5	54.6	6.4	-3.97	-2.00
27	Aucilla Area Solid Waste Facility	$39,375	1,075	0.3	57.1	3.2	36	-0.03	-1.01
28	Monarch Hill Landfill	$44,420	100,002	0.4	19.2	26.2	40.1	-2.50	-0.82
29	Cedar Grove Landfill	$37,355	1,964	0.1	17.3	2.4	76.2	-0.05	0.31
30	Medley Landfill	$76,765	137,043	0.2	0.9	92	5.4	-3.43	-1.65

Exhibit H

Original Image of 39 So.3d 1216 (PDF)

KeyCite Red Flag - Severe Negative Treatment
Receded From by Lieupo v. Simon's Trucking, Inc., Fla., December 19, 2019

39 So.3d 1216
Supreme Court of Florida.

Howard CURD, et al., Petitioners,
v.
MOSAIC FERTILIZER, LLC, Respondent.

No. SC08–1920
|
June 17, 2010.

Synopsis
Background: Commercial fishermen filed class action lawsuit against owner of fertilizer storage facility, alleging pollution from facility reduced the supply of fish and seeking economic damages. The Circuit Court, Hillsborough County, Richard A. Nielsen, J., dismissed the complaint. Fishermen appealed. The District Court of Appeal, 993 So.2d 1078, affirmed and certified questions.

Holdings: The Supreme Court, Quince, C.J., held that:

[1] injury or damage to property was not required for the fishermen to assert claim for strict liability under statute allowing private parties to sue for damages resulting from a discharge or other condition of pollution;

[2] fishermen were not precluded by economic loss rule from bringing negligence and strict liability claims;

[3] owner of fertilizer storage facility owed duty of care to fishermen, such as to give rise to cause of action sounding in negligence.

Questions answered.

Polston, J., concurred in part, and dissented in part, and filed opinion.

West Headnotes (7)

[1] **Appeal and Error** Statutory or legislative law

The Supreme Court's interpretation of a statute is a purely legal matter and therefore subject to the de novo standard of review.

6 Cases that cite this headnote

[2] **Statutes** 🔑 Language and intent, will, purpose, or policy

When construing a statute, the Supreme Court strives to effectuate the Legislature's intent; to determine that intent, the Court looks first to the statute's plain language.

4 Cases that cite this headnote

[3] **Environmental Law** 🔑 Civil liability; cleanup costs

Statute allowing private parties to sue for damages resulting from a discharge or other condition of pollution allowed commercial fishermen to allege a cause of action for strict liability to recover lost income from owner of fertilizer storage facility, after pollution from storage facility allegedly reduced available supply of fish, regardless of whether the fishermen did not own any real or personal property damaged by the pollution. West's F.S.A. § 376.313(3).

14 Cases that cite this headnote

[4] **Fish** 🔑 Injury to or destruction of fish

Commercial fishermen who claimed that pollution from fertilizer storage facility polluted fishing waters, reducing the available supply of fish, were not precluded by economic loss from bringing negligence and strict liability claims against owner of facility to recover damages for their loss of income; parties were not in contractual privity, and owner was not a manufacturer or distributor of a defective product that caused damage to itself.

28 Cases that cite this headnote

[5] **Negligence** 🔑 Elements in general

Four elements are necessary to sustain a negligence claim: (1) a duty, or obligation, recognized by the law, requiring the defendant to conform to a certain standard of conduct, for the protection of others against unreasonable risks; (2) a failure on the defendant's part to conform to the standard required: a breach of the duty; (3) a reasonably close causal connection between the conduct and the resulting injury, which is commonly known as "legal cause," or "proximate cause," and which includes the notion of cause in fact; and (4) actual loss or damage.

70 Cases that cite this headnote

[6] **Negligence** 🔑 Necessity and Existence of Duty

Negligence 🔑 Foreseeability

Negligence 🔑 Duty based upon statute or other regulation

The statute books and case law are not required to catalog and expressly proscribe every conceivable risk in order for it to give rise to a duty of care in order to support claim of negligence; rather, each defendant who creates a risk is required to exercise prudent foresight whenever others may be injured as a result.

11 Cases that cite this headnote

[7] **Fish** 🔑 Injury to or destruction of fish

> Owner of fertilizer storage facility near fishing waters owed duty of care to commercial fishermen to protect them from harm from leakage of pollutants into waters, such as to give rise to cause of action sounding in negligence; facility's activities created appreciable zone of risk within which owner was obligated to protect those who were exposed to harm, it was foreseeable that, were stored materials released into public waters, they would cause damage to marine and plant life as well as to human activity, and fishermen had a special interest within that zone of risk, an interest not shared by the general community, in being dependent on the waters to earn their livelihood.
>
> 14 Cases that cite this headnote

Attorneys and Law Firms

***1217** F. Wallace Pope, Jr. of Johnson, Pope, Bokor, Ruppel, and Burns, LLP, Clearwater, ***1218** FL, and Andra T. Dreyfus, Clearwater, FL, for Petitioners.

David Barnett Weinstein and Kimberly S. Mello of Greenberg Traurig, P.A., Tampa, FL, and Arthur J. England, Jr. of Greenberg Traurig, P.A., Miami, FL, for Respondent.

Charles W. Hall and Mark D. Tinker of Banker Lopez Gassler, P.A., St. Petersburg, FL, and Paul M. Smith and Michelle A. Groman of Jenner and Block, LLP Washington, DC, on behalf of General Dynamics Corporation and General Dynamics Land Systems, Inc., as Amicus Curiae.

Opinion

QUINCE, C.J.

This cause is before the Court for review of the decision of the Second District Court of Appeal in *Curd v. Mosaic Fertilizer, LLC,* 993 So.2d 1078 (Fla. 2d DCA 2008). In its decision the district court ruled upon the following questions, which the court certified to be of great public importance:

> DOES FLORIDA RECOGNIZE A COMMON LAW THEORY UNDER WHICH COMMERCIAL FISHERMEN CAN RECOVER FOR ECONOMIC LOSSES PROXIMATELY CAUSED BY THE NEGLIGENT RELEASE OF POLLUTANTS DESPITE THE FACT THAT THE FISHERMEN DO NOT OWN ANY PROPERTY DAMAGED BY THE POLLUTION?
>
> DOES THE PRIVATE CAUSE OF ACTION RECOGNIZED IN SECTION 376.313, FLORIDA STATUTES (2004), PERMIT COMMERCIAL FISHERMEN TO RECOVER DAMAGES FOR THEIR LOSS OF INCOME DESPITE THE FACT THAT THE FISHERMEN DO NOT OWN ANY PROPERTY DAMAGED BY THE POLLUTION?

Id. at 1079. We have jurisdiction. *See* art. V, § 3(b)(4), Fla. Const. For the reasons explained below, we answer the questions in the affirmative and quash the decision below.

FACTS AND PROCEDURAL HISTORY

In *Curd,* 993 So.2d 1078, the Second District Court of Appeal summarized the facts as follows:

> According to the allegations in [Howard Curd and several other commercial fishermen's (the fishermen)] complaint, [Mosaic Fertilizer, LLC (Mosaic)] owned or controlled a phosphogypsum storage area near Archie Creek in Hillsborough County. The storage area included a pond enclosed by dikes, containing wastewater from a phosphate plant. This wastewater allegedly contained pollutants and hazardous contaminants.
>
> The fishermen alleged that in the summer of 2004, the Hillsborough County Environmental Protection Commission and the Florida Department of Environmental Protection both warned Mosaic that the quantity of wastewater in the storage facility was dangerously close to exceeding the safe storage level. According to the complaint, on August 10, 2004, the Department of Environmental Protection warned Mosaic that a 100–foot section of the pond dike was three feet narrower than the minimum required width of 18 feet. It warned that only an inch or two of additional rain during the tropical season would raise the level of pollutants in the pond to the top of the dike. On September 5, 2004, the dike gave way and pollutants were spilled into Tampa Bay.
>
> The fishermen claim that the spilled pollutants resulted in a loss of underwater plant life, fish, bait fish, crabs, and other marine life. They do not claim an ownership in the damaged marine and ***1219** plant life, but claim that it resulted in damage to the reputation of the fishery products the fishermen are able to catch and attempt to sell. At least implicitly, they are alleging monetary damages in the nature of lost income or profits.
>
> The complaint included three counts. Count 1 attempted to allege a claim for statutory liability under section 376.313(3), Florida Statutes (2004). Count 2 alleged common law strict liability based upon damages resulting from Mosaic's use of its property for an ultrahazardous activity. *See, e.g., Cities Serv. Co. v. State,* 312 So.2d 799 (Fla. 2d DCA 1975). Count 3 alleged a claim of simple negligence. The trial court concluded that the language in chapter 376 did not permit a claim by these fishermen for monetary losses when they did not own any real or personal property damaged by the pollution. After initially permitting the fishermen to proceed on their claims of negligence and strict liability, the trial court ultimately ruled that these claims were not authorized under the economic loss rule. The fishermen then appealed the dismissal of their entire fourth amended complaint to [the Second District].

Curd, 993 So.2d at 1079–80.

On appeal, the Second District affirmed the trial court's order dismissing Curd's proposed class action lawsuit against Mosaic Fertilizer. *See* *Curd,* 993 So.2d at 1079. The court held that under traditional principles of negligence the fishermen failed to state a cause of action. *See* *id.* at 1083. The court reasoned that an action in common law either through strict liability or negligence was not permitted because the fishermen did not sustain bodily injury or property damage. The strict liability and negligence claims sought purely economic damages unrelated to any damage to the fishermen's property. Accordingly, the court further reasoned that Mosaic did not owe the fishermen an independent duty of care to protect their purely economic interests. *See* *id.* at 1082–83. Additionally, in evaluating the fishermen's statutory liability claim under section 376.313(3), the court concluded that there is no Florida precedent that permits a recovery for damages under section 376.313(3) when the party seeking the damages does not own or have a possessory interest in the property damaged by the pollution. *See* *id.* at 1084. Further, the court said that there is no express language from the Legislature stating that it intended the statute to create a wide array of claims by people indirectly affected by pollution. *See* *id.* The court also declined to read into the statute a legislative intent in section 376.313(3) to allow such economic claims based on the fishermen's unique relationship with the fish or based on the fact that the fishermen hold commercial fishing licenses. Moreover, the court declined to recognize such a right as a matter of tort law. *See* *id.* at 1085. The court was unconvinced that a special theory should be established under the common law for a narrow subset of the people who are indirectly or remotely injured by pollution. *See* *id.* at 1085–86.

Pursuant to article V, section 3, subsection (b)(4) of the Florida Constitution and Florida Rule of Appellate Procedure 9.030(a)(2)(A)(v), the Second District certified the questions above to be of great public importance. *See* *Curd,* 993 So.2d at 1079. We granted review to answer the certified questions.

ANALYSIS

Statutory Cause of Action

[1] We first address whether the private cause of action recognized in ***1220** section 376.313(3), Florida Statutes (2004),[1] allows commercial fishermen to recover damages for their loss of income despite the fact that the fishermen do not own any property damaged by the pollution. Our interpretation of a statute is a purely legal matter and therefore subject to the de novo standard of review. *See* *Kephart v. Hadi,* 932 So.2d 1086, 1089 (Fla.2006); *see also* *B.Y. v. Dep't of Children & Families,* 887 So.2d 1253, 1255 (Fla.2004) (noting that the standard of appellate review on issues involving the interpretation of statutes is de novo).

[2] When construing a statute, we strive to effectuate the Legislature's intent. *See, e.g., Borden v. East–European Ins. Co.,* 921 So.2d 587, 595 (Fla.2006) ("We endeavor to construe statutes to effectuate the intent of the Legislature."); *State v. J.M.,* 824 So.2d 105, 109 (Fla.2002) (noting that legislative intent is the polestar that guides a court's statutory construction analysis). To determine that intent, we look first to the statute's plain language. *See Borden,* 921 So.2d at 595. We have held that "when the statute is clear and unambiguous, courts will not look behind the statute's plain language for legislative intent or resort to rules of statutory construction to ascertain intent." *Id.* (quoting *Daniels v. Fla. Dep't of Health,* 898 So.2d 61, 64 (Fla.2005)). In reaching our conclusion that chapter 376, Florida Statutes (2004), allows a cause of action by these plaintiffs, we have construed several provisions of the chapter in pari materia and given effect to the various sections. *See* *E.A.R. v. State,* 4 So.3d 614, 629 (Fla.2009); *McDonald v. State,* 957 So.2d 605, 610 (Fla.2007); *Zold v. Zold,* 911 So.2d 1222, 1229–30 (Fla.2005).

Section 376.313(3), Florida Statutes (2004), provides as follows:

> Except as provided in s. 376.3078(3) and (11), *nothing contained in ss. 376.30–376.319 prohibits any person from bringing a cause of action in a court of competent jurisdiction for all damages resulting from a discharge or other condition of pollution covered by ss. 376.30–376.319.* Nothing in this chapter shall prohibit or diminish a party's right to contribution from other parties jointly or severally liable for a prohibited discharge of pollutants or hazardous substances or other pollution conditions. Except as otherwise provided in subsection (4) or subsection (5), *in any such suit, it is not necessary for such person to plead or prove negligence in any form or manner. Such person need only plead and prove the fact of the prohibited discharge or other pollutive condition and that it has occurred.* The only defenses to such cause of action shall be those specified in s. 376.308.

(Emphasis added.) The Second District Court of Appeal provided the following legislative history regarding section 376.313(3):

> Chapter 376 regulates the discharge of pollution. The first portion of this chapter was enacted in 1970 as the "Oil Spill Prevention and Pollution Control Act." *See* ch. 70–244, Laws of Fla. The legislature expanded the reach of chapter 376 when it enacted the Water Quality Assurance Act of 1983, ch. 83–310, Laws of Fla. Section 84 of chapter 83–310 effectively created a private cause of action for damages caused by pollution. Ch. 83–310, § 84, at 1885, Laws of Fla. ***1221** This provision is currently codified in section 376.313(3).

Curd, 993 So.2d at 1083.

The statute at issue is found within chapter 376, which is entitled "Pollutant Discharge Prevention and Removal." Section 376.315 of this chapter provides that "[s]ections 376.30–376.319, being necessary for the general welfare and the public health and safety of the state and its inhabitants, shall be liberally construed to effect the purposes set forth under ss. 376.30–376.319 and the Federal Water Pollution Control Act, as amended." Additionally, section 376.30, which gives legislative intent regarding pollution of surface and ground waters, provides in pertinent part that the preservation of surface and ground waters "can only be served effectively by maintaining the quality of state waters in as close to a pristine condition as possible, taking into account multiple-use accommodations necessary to provide the broadest possible promotion of public and *private interests.*" § 376.30(1)(c), Fla. Stat. (2004) (emphasis added). Section 376.30 further provides that the Legislature found and declared that escapes of pollutants "pose threats of great danger and damage ... to citizens of the state, and to *other interests deriving livelihood from the state.*" § 376.30(2)(b), Fla. Stat. (2004) (emphasis added).

[3] We find that section 376.313(3) and the language used in section 376.30 are clear and unambiguous, and we rely solely on their plain language to discover the legislative intent. Section 376.313(3) provides that "nothing ... prohibits any person from bringing a cause of action ... for all damages resulting from a discharge or other condition of pollution." § 376.313(3).[2] The language of the statute allows *any person* to recover for damages suffered as a result of pollution.[3] "Damage," as used in chapter 376, is defined as "the documented extent of any destruction to or loss of any real or personal property, or the documented extent, pursuant to s. 376.121, of any destruction of the environment and natural resources, including all living things except human beings, as the direct result of the discharge of a pollutant." *See* § 376.031(5). Moreover, the Legislature intended that the statute be liberally construed. *See* § 376.315 ("Sections 376.30–376.319 ... shall be liberally construed to effect the purposes set forth under ss. 376.30–376.319 and the Federal Water Pollution Control Act, as amended."). The title of section 376.313, "Nonexclusiveness of remedies and individual cause of action for damages under ss. 376.30–376.319," implies that a liberal construction should be applied under these circumstances. *See* *State v. Webb,* 398 So.2d 820, 824–25 (Fla.1981) (holding that when determining legislative intent, due weight and effect must be given to the title of the section because "the title is more than an index to what the section is about or has reference to; it is a direct statement by the legislature of its intent").

Importantly, in *Aramark,* 894 So.2d at 24, we held that section 376.313(3) creates a private cause of action by creating a damages remedy for the non negligent discharge of pollution without proof that the ***1222** defendant caused it. Therefore, a defendant can be held liable even without proof that it caused the pollutive discharge, that is, the plaintiff does not have to plead or prove negligence in any form. *See* *id.* at 23–24. We noted that the following factors demonstrate that section 376.313(3) creates a cause of action for strict liability regardless of causation:

> [T]he statute's provision of a damages remedy for the non negligent discharge of pollution; the defenses provided in the statute, including the inclusion of lack of causation as an affirmative defense; and other aspects of the

> statute such as its title, the cumulative remedies clause and the attorney's fees provision—when combined with the statutory directive that section 376.313(3) should be liberally construed....

Id. at 26 (citing Gary K. Hunter, *Statutory Strict Liability for Environmental Contamination: A Private Cause of Action to Remedy Pollution or Mere Legislative Jargon?,* Fla. Bar J., Jan. 1998, at 50, 51). We find that some of those factors are relevant in this case and would allow commercial fishermen to recover damages for their loss of income despite the fact that the fishermen do not own any real or personal property damaged by the pollution. Section 376.313(3) provides that "*in any such suit* ... [a] person need only plead and prove the fact of the prohibited discharge or other pollutive condition and that it has occurred." Mosaic contends that despite this language the fishermen are not entitled to economic damages because they do not own any property damaged by the pollution. First, it must be noted that under the definition of damages cited above, one can recover for damages to real or personal property but one can also recover for damages to "natural resources, including all living things." Furthermore, section 376.313(3) states that "[t]he only defenses to such cause of action shall be those specified in s. 376.308." Those defenses specified in section 376.308 include acts of war, acts by a governmental entity, acts of God, and acts or omissions by a third party. Because the statute does not specifically list the lack of property ownership as a defense, we find that defense, much as we found the omission of causation in *Aramark,* was deliberately omitted.

In sum, the Legislature has enacted a far-reaching statutory scheme aimed at remedying, preventing, and removing the discharge of pollutants from Florida's waters and lands. To effectuate these purposes, the Legislature has provided for private causes of action to any person who can demonstrate damages as defined under the statute. There is nothing in these statutory provisions that would prevent commercial fishermen from bringing an action pursuant to chapter 376.

The Economic Loss Rule

The Second issue before this Court is whether Florida recognizes a common law theory under which commercial fishermen can recover for economic losses proximately caused by the negligent release of pollutants despite the fact that the fishermen do not own any real or personal property damaged by the pollution. Because this case is before the Court on the trial court's dismissal of Curd's fourth amended complaint, we must take all the factual allegations in his complaint as true and construe all reasonable inferences from those facts in his favor. *See* *Florida Dept. of Health & Rehabilitative Servs. v. S.A.P.,* 835 So.2d 1091, 1094 (Fla.2002) (citing *Ralph v. City of Daytona Beach,* 471 So.2d 1, 2 (Fla.1983)). Our standard of review is de novo. *See id.* (citing *Execu–Tech Bus. Sys., Inc. v. New Oji Paper Co.,* 752 So.2d 582, 583 (Fla.2000)).

***1223** The Second District held that the fishermen's common law negligence and strict liability claims were barred by the economic loss rule and general tort law principles because the fishermen did not own any property damaged by the pollution. *See* *Curd,* 993 So.2d at 1080–81. The district court found that Mosaic did not owe the fishermen an independent duty of care to protect their purely economic interests. *See* *id.* at 1082–83. Relying on a negligence principle that the law generally protects interests in the safety of person and property, the district court concluded that the fishermen failed to state a cause of action for strict liability or negligence because they had sustained no bodily injury or property damage. *See* *id.* at 1082. Since the fishermen did not own a property interest in the fish or allege any bodily injury, the district court concluded that the fishermen's negligence and strict liability claims sought purely economic damages unrelated to any damage to the fishermen's property. Therefore, Mosaic did not owe an independent duty of care to protect the fishermen's expectation of profits. *See* *id.* at 1083.

[4] The Second District, in finding that the economic loss rule applied to the facts of this case, attempted to explain this Court's opinion in *Indemnity Ins. Co. v. American Aviation, Inc.,* 891 So.2d 532 (Fla.2004). In *American Aviation* we undertook a comprehensive look at the economic loss rule including its origin and scope. We clearly stated that the economic loss rule in Florida is applicable in only two situations: (1) where the parties are in contractual privity and one party seeks to recover damages in tort for matters arising out of the contract, or (2) where the defendant is a manufacturer or distributor of a defective product which damages itself but does not cause personal injury or damage to any other property. 891 So.2d at 536.[4]

Clearly neither the contractual nor products liability economic loss rule is applicable to this situation. The parties to this action are not in contractual privity. Moreover, the defendant in this case is not a manufacturer or distributor of a defective product that has caused damage to itself. Rather we have plaintiffs who have brought traditional negligence and strict liability claims against a defendant who has polluted Tampa Bay and allegedly caused them injury. Thus, the economic loss rule does not prevent the plaintiffs from bringing this cause. The plaintiffs' causes of action are controlled by traditional negligence law, which requires proof of duty, breach, and proximate cause, and by strict liability principles.

Common Law Causes of Action

In addition to finding that the fishermen's claims were barred by the economic loss rule, the Second District also found their claims barred because "Mosaic did not owe an independent duty of care to protect the fishermen's purely economic interests—that is, their expectations of profits from fishing for healthy fish." *Curd,* 993 So.2d at 1083. We hold that Mosaic did owe a duty of care to the fishermen, a duty that was not shared by the public as a whole.

As a general principle of common law negligence, some courts have not permitted recovery for purely economic losses when the plaintiff has sustained no bodily injury or property damage. *See* ***1224** *Union Oil Co. v. Oppen,* 501 F.2d 558, 563 (9th Cir.1974) (noting "the widely recognized principle that no cause of action lies against a defendant whose negligence prevents the plaintiff from obtaining a prospective pecuniary advantage"). The reasoning behind this general rule is that if courts allowed compensation for all losses of economic advantages caused by a defendant's negligence, a defendant would be subject to claims based upon remote and speculative injuries that he could not foresee. *See Oppen,* 501 F.2d at 563.[5] Courts have applied this general rule in a variety of ways. Some courts have concluded that the negligent defendant owes no duty to plaintiffs seeking compensation for such losses. *See, e.g., Byrd v. English,* 117 Ga. 191, 43 S.E. 419 (1903); *Chelsea Moving & Trucking Co. v. Ross Towboat Co.,* 280 Mass. 282, 182 N.E. 477 (1932); *Brink v. Wabash R.R.,* 160 Mo. 87, 60 S.W. 1058 (1901). In some cases, courts have invoked the doctrine of proximate cause to deny recovery. *See Byrd,* 117 Ga. 191, 43 S.E. 419; *Ross Towboat,* 280 Mass. 282, 182 N.E. 477. Other courts have relied on the remoteness of the economic loss. *See, e.g., Northern States Contracting Co. v. Oakes,* 191 Minn. 88, 253 N.W. 371 (1934). Consequently, the defendants were normally relieved of the burden to defend against such claims. *See Oppen,* 501 F.2d at 563.

Curd contends that commercial fisherman fall into a recognized exception to that general rule. Curd claims that the licensed commercial fishermen have a protectable economic expectation in the marine life that qualifies as a property right. Curd asserts that for years he and the other fishermen have been subjecting the fish, crabs, and other marine life within the polluted area to their "dominion." Curd asserts that because the State licensed the fishermen and created an economic expectancy, the Second District erred when it concluded that the fishermen did not "own" the marine life at the time of its destruction and thus suffered no property damage. *See* *Curd,* 993 So.2d at 1083.

In circumstances similar to this case, courts have allowed commercial fishermen to recover when the alleged injuries occurred on water as a result of activities that occurred on land. In *Leo v. General Electric Co.,* 145 A.D.2d 291, 538 N.Y.S.2d 844 (N.Y.App.Div.1989), commercial fishermen brought an action against the defendant, General Electric Company, for discharging approximately 500,000 pounds of polychlorinated biphenyls (PCBs) from two of its manufacturing plants into the Hudson River. *See* *id.* at 292–93, 538 N.Y.S.2d at 845–46. The marine life in the Hudson River, including the striped bass, absorbed the PCBs that collected on the river floor. *See id.* As a result, the New York State Department of Environmental Conservation imposed a ban on the sale of striped bass fished from the affected waters, and banned the fishing of striped bass anywhere in the State for either commercial or recreational purposes. *See* *id.* at 293, 538 N.Y.S.2d 844. The commercial fishermen, who earned their livelihood from fishing the affected waters, claimed that the sale of striped bass accounted for a substantial part of their income and that as commercial fishermen they had a special interest in use of public waters. That special interest, they claimed, was invaded by the defendant's pollution, and they alleged ***1225** that the defendant's public nuisance had and would continue to have a devastating effect upon their ability to earn a living. Accordingly, the fishermen sought damages and injunctive relief. *See id.*

The court agreed with the fishermen and held that the commercial fishermen did have standing to complain of the pollution of the waters from which they derived their living. *See* *id.* at 295, 538 N.Y.S.2d at 847 (citing *Louisiana ex rel. Guste v. The M/V Testbank,* 524 F.Supp. 1170 (E.D.La.1981), *aff'd,* 728 F.2d 748 (5th Cir.1984); *Pruitt v. Allied Chemical Corp.,* 523 F.Supp. 975 (E.D.Va.1981); *Burgess v. The M/V Tamano,* 370 F.Supp. 247 (D.Me.1973), *aff'd,* 559 F.2d 1200 (1st Cir.1977); *Carson v. Hercules Powder Co.,* 240 Ark. 887, 402 S.W.2d 640 (1966); *Hampton v. North Carolina Pulp Co.,* 223 N.C. 535, 27 S.E.2d 538 (1943); *Columbia River Fishermen's Protective Union v. City of St. Helens,* 160 Or. 654, 87 P.2d 195 (1939)). The court found that the fishermen suffered a peculiar or special harm, a diminution or loss of livelihood, which was not suffered by every person who fished the affected waters. Thus, the court determined that the fishermen's alleged harm was peculiar to them in their capacity as commercial fishermen, and it went beyond the harm done to members of the community. *See* *id.* at 295, 538 N.Y.S.2d at 847.

Moreover, in *Carson,* a licensed commercial fisherman brought an action against a powder company for injunctive relief and damages. The commercial fisherman had permission of the riparian owners of the land to fish a thirty-mile stretch of Bayou Meto, a non navigable stream. The fisherman commercially sold the fish. *See* 402 S.W.2d at 641. The fisherman claimed that the powder company, in the operation of its plant, had polluted the stream by discharging into it phenolic materials that killed fish, created a rotten egg odor, and made the fish inedible and unsalable. *See id.* The Supreme Court of Arkansas held that the powder company was liable to the fisherman. The supreme court concluded that even though the powder company corrected this condition, the powder company was liable for damages for loss of profits and damage to the fisherman's business. The supreme court reasoned that the fisherman had a substantial investment in a business and that it was his only means of livelihood. Therefore, by polluting the water, the powder company prevented the operation of this business, so it became directly liable for any damage to his business and loss of profits. *See* *id.* at 642.

Additionally, in *Columbia River Fishermen's Protective Union,* commercial fishermen brought an action against the operators of two plants, an insulating board company and a paper company, for discharging pollution into the river. The plaintiffs alleged that the pollution destroyed the fish, aquatic life, and its fishing nets. The plaintiffs contended that this caused irreparable injury. *See* 87 P.2d at 196–97. The Supreme Court of Oregon concluded that the commercial fishermen had a cause of action. *See id.* at 199–200. The court reasoned that the commercial fishermen had a special interest, distinct from that of the public, in fishing the rivers. In finding a cause of action, the supreme court found that deleting the fish from the rivers prevented the fishermen from pursuing their vocations and earning their livelihood. The court found a vital distinction between the rights of licensed fishermen who are accustomed to fishing in the river and the rights of other citizens of the state. *See id.* at 197.[6] *But see* ***1226** *Kuehn v. Milwaukee,* 83 Wis. 583, 53 N.W. 912, 912–14 (1892) (holding that a fisherman had no cause of action in equity because the fisherman was only one of a large number of fishermen affected by the alleged nuisance, he had no special privilege or right to fish in Lake Michigan, and he had no property damaged by the nuisance).

Some courts have also allowed commercial fishermen to recover against the polluter when both the activities and the alleged injuries occurred on water. In *Louisiana ex rel. Guste,* two ships collided which resulted in pollution of the waters by chemical cargo. *See* 524 F.Supp. at 1171. Because of the possibility that aquatic life was contaminated by the chemical, the United States Coast Guard temporarily closed a substantial number of square miles of Louisiana waterways and marshes to commercial fishermen, crabbers, oystermen, and shrimpers. *See id.* The commercial fishermen and other parties who used certain waters for business or recreation asserted various theories of liability, including maritime tort, and private causes of action pursuant to federal statute, the laws of the State of Louisiana, and the laws of the United States. *See id.* The defendants sought summary judgment on all claims for alleged economic loss, contending that the damages for which plaintiffs sought recovery were consequential results of the ships colliding in which no actual physical damage occurred. Therefore, defendants argued that the plaintiffs could not recover for mere business expectations or losses sustained solely from the negligent interference with contractual relations. *See id.*

The federal district court disagreed, holding that the collision of the ships and the resulting discharge of the toxic chemical "constituted a tortious invasion that interfered with the special interest of the commercial fishermen, crabbers, shrimpers and oystermen to use those public waters to earn their livelihood and the specific pecuniary losses which can be shown to have been incurred should be recoverable." *Id.* at 1174. The court reasoned that the fisherman were exercising their public right to make a commercial use of those waters. *See id.* (citing *Burgess,* 370 F.Supp. 247); *see also Pruitt,* 523 F.Supp. at 978 (noting that commercial fishermen were entitled to compensation for any loss of profits they could prove were caused by defendant's negligence because the entitlements presumably arose from a constructive property interest in Chesapeake Bay's harvestable species and the professional fishermen were entitled to recover despite the lack of any direct physical damage to their own property).

Other federal courts have held similarly. In *Oppen,* commercial fisherman brought an action for economic damages under a federal statute against oil companies for discharging raw crude oil over vast stretches of the coastal waters of Southern California. *See* 501 F.2d at 559–60. The court found that foreseeability was the crucial determinant as to whether the defendants owed a duty to the commercial fishermen to refrain from negligent conduct in their drilling operations. *See id.* at 568–69. Therefore, the issue that had to ***1227** be addressed was "whether the defendants could reasonably have foreseen that negligently conducted drilling operations might diminish aquatic life and thus injure the business of commercial fishermen." *Id.* at 569. The federal circuit court concluded that the defendants could have reasonably foreseen that the negligently conducted drilling operations might diminish aquatic life and injure the commercial fishermen's business. *See* *id.* The court reasoned that the dangers of pollution were known to all, and that the defendants understood the risks of their business. *See* *id.* Accordingly, the federal court held that the commercial fishermen had a cause of action to prove their case, and that the defendants were under a duty to commercial fisherman to conduct their drilling and production in a reasonably prudent manner so as to avoid the negligent diminution of aquatic life. *See* *id.* at 569–70. The court further noted that the plaintiff's injury was a pecuniary loss of a particular and special nature, limited to commercial fishermen. *See* *id.* at 570.

Likewise, in *Burgess,* a tanker discharged approximately 100,000 gallons of oil into the waters of Casco Bay in Maine. The plaintiffs, commercial fishermen, sought to recover economic damages incurred as a result of the discharge. *See* 370 F.Supp. at 248. The defendant contended that the plaintiff's economic interests were not legally cognizable because none of the fishermen had any property interest in the coastal waters, marine life, or shores claimed to have been injured by the spill. *See* *id.* at 249. The federal court disagreed. Although the court recognized that the fishermen had no individual property rights with respect to the aquatic life harmed by the oil pollution, the court concluded that the fishermen could state a claim for the tortious invasion of a public right because they had a special interest different from the general public to take fish from the coastal waters. *See* *id.* at 250. The court found that the fishermen's injury resulted from defendants' alleged interference with their direct exercise of the public right to fish. The court reasoned that it would be inappropriate for a person engaged in commercial fishing, who is dependent thereon for his livelihood, to be denied any right to recover for his pecuniary loss on the basis that his injury is no different from that sustained by the general public. *See* *id.*

[5] We conclude, as did many of the courts in the cases discussed above, that the defendant owed a duty of care to the commercial fishermen, and that the commercial fishermen have a cause of action sounding in negligence. Four elements are necessary to sustain a negligence claim:

> 1. A duty, or obligation, recognized by the law, requiring the [defendant] to conform to a certain standard of conduct, for the protection of others against unreasonable risks.
>
> 2. A failure on the [defendant's] part to conform to the standard required: a breach of the duty....
>
> 3. A reasonably close causal connection between the conduct and the resulting injury. This is what is commonly known as "legal cause," or "proximate cause," and which includes the notion of cause in fact.
>
> 4. Actual loss or damage....

Clay Elec. Coop., Inc. v. Johnson, 873 So.2d 1182, 1185 (Fla.2003) (alterations in original) (quoting *Prosser and Keaton on the Law of Torts* 164–65 (W. Page Keeton ed., 5th ed.1984)).

[6] Under Florida law, the question of whether a duty is owed is linked to the concept of foreseeability. We have held that duties may arise from four general sources: (1) legislative enactments or administrative ***1228** regulations; (2) judicial interpretations of such enactments or regulations; (3) other judicial precedent; and (4) a duty arising from the general facts of a case. *Clay Elec.,* 873 So.2d at 1185 (citing *McCain v. Fla. Power Corp.,* 593 So.2d 500, 503 n. 2 (Fla.1992)). The fourth category encompasses "that class of cases in which the duty arises because of a foreseeable zone of risk arising from the acts of the defendant." *McCain,* 593 So.2d at 503 n. 2. As we have explained:

> The statute books and case law ... are not required to catalog and expressly proscribe every conceivable risk in order for it to give rise to a duty of care. Rather, each defendant who creates a risk is required to exercise prudent foresight whenever others may be injured as a result. This requirement of reasonable, general foresight is the core of the duty element.

Id. at 503.

[7] In the present case, the duty owed by Mosaic arose out of the nature of Mosaic's business and the special interest of the commercial fisherman in the use of the public waters. First, Mosaic's activities created an appreciable zone of risk within which Mosaic was obligated to protect those who were exposed to harm. Mosaic's business involved the storage of pollutants and hazardous contaminants. It was foreseeable that, were these materials released into the public waters, they would cause damage to marine and plant life as well as to human activity. *See* *McCain,* 593 So.2d at 503 n. 2. Further, the commercial fishermen had a special interest within that zone of risk, an interest not shared by the general community. *See* *Oppen,* 501 F.2d at 568. The fishermen were licensed to conduct commercial activities in the waters of Tampa Bay, and were dependent on those waters to earn their livelihood. Mosaic's activities placed the fishermen's peculiar interests directly within the zone of risk created by the presence of its facility. As a result, Mosaic was obligated to exercise prudent foresight and take sufficient precautions to protect that interest. *See* *Kaisner v. Kolb,* 543 So.2d 732, 735 (Fla.1989) (“Where a defendant's conduct creates a foreseeable zone of risk, the law generally will recognize a duty placed upon [that] defendant either to lessen the risk or see that sufficient precautions are taken to protect others from the harm that the risk poses.”).

Here, the discharge of the pollutants constituted a tortious invasion that interfered with the special interest of the commercial fishermen to use those public waters to earn their livelihood. We find this breach of duty has given rise to a cause of action sounding in negligence. We note, however, that in order to be entitled to compensation for any loss of profits, the commercial fishermen must prove all of the elements of their causes of action, including damages.

CONCLUSION

For the reasons set forth above, we hold that the commercial fishermen have both a statutory and common law cause of action. Accordingly, we answer the certified questions in the affirmative and quash the decision of the Second District.

It is so ordered.

PARIENTE, LEWIS, LABARGA, and PERRY, JJ., concur.

POLSTON, J, concurs in part and dissents in part with an opinion.

CANADY, J., recused.

POLSTON, J., concurring in part dissenting in part.
I respectfully concur in part and dissent in part. Although I use different reasoning, ***1229** I agree with the majority's affirmative answer to the certified question of whether commercial fishermen can recover damages

for their loss of income pursuant to section 376.313, Florida Statutes (2004). However, I disagree with the majority's affirmative answer to the certified question of whether, under the facts of the case, commercial fishermen can recover economic losses proximately caused by the negligent release of pollution under Florida common law.

As an initial matter, I note that the majority decides the case for a more narrow class than those bringing the suit and more narrowly than the claims they allege. Although Curd's proposed class consists of "all fishermen *and those persons* engaged in the commercial catch *and sale* of fish,"[7] the majority's decision does not extend to distributors, seafood restaurants, fisheries, fish brokers, or the like whose incomes may have been affected by Mosaic's pollution. Additionally, the majority only addresses economic harm that resulted from the depletion of marine life and the resulting inability to harvest the commercial fishermen's usual yield-not from harm to reputation as alleged in the petitioner's complaint and mentioned by the Second District Court of Appeal. *Compare* majority op. at 1228 (discussing the diminution of aquatic life because of pollution) *with* *Curd v. Mosaic Fertilizer, LLC,* 993 So.2d 1078, 1079 (Fla. 2d DCA 2008) (noting that Curd asserts damage to reputation of fishery products); Petitioner's Fourth Amended Complaint (alleging loss of plant life, loss of fish and the resulting loss of revenue from inability to harvest fish, loss of crabs and other marine life, and damage to reputation of fishery products). Because the majority opinion does not extend to other class members beyond the commercial fishermen and does not extend to reputation damages, I do not address them.

I. STATUTORY LIABILITY

I agree with the majority that section 376.313(3) provides the commercial fishermen with a strict liability private cause of action. *See* *Aramark Unif. & Career Apparel, Inc. v. Easton,* 894 So.2d 20, 28 (Fla.2004) (holding that section 376.313(3) creates a strict liability cause of action).

As the Second District noted, chapter 376 contains two separately enacted antipollution laws. *Curd,* 993 So.2d at 1083. The first portion of chapter 376 was enacted in 1970 as the "Oil Spill Prevention and Pollution Control Act" and is currently codified in sections 376.011 through 376.21, Florida Statutes (2004).[8] *See* ch. 70–244, Laws of Fla. The 1970 enactment provides a cause of action for parties harmed by pollution of coastal waters and lands. *See* § 376.021, Fla. Stat. (2004) (entitled "Legislative intent with respect to pollution of coastal waters and lands"); § 376.041, Fla. Stat. (2004) ("The discharge of pollutants into or upon any coastal waters, estuaries, tidal flats, beaches, and lands adjoining the seacoast of the state in the manner defined by ss. 376.011–376.21 is prohibited.").

This 1970 enactment concerns pollution of the coastal waters, and the Legislature included a restrictive definition of damages, applicable only to the 1970 enactment. Specifically, section 376.031(5), Florida Statutes (2004), defines "Damage" as "the documented extent of any destruction to or loss of any real or personal property, or the documented extent, pursuant to s. 376.121, of any destruction of the environment and natural resources, including ***1230** all living things except human beings, as the direct result of the discharge of a pollutant." The Legislature specified in section 376.031 that the definition only applies to sections 376.011 through 376.21, namely the 1970 enactment. *See* § 376.031, Fla. Stat. (2004) ("When used in ss. 376.011–376.21, unless the context clearly requires otherwise, the term....").

In 1983, the Legislature expanded the reach of chapter 376 by enacting the "Water Quality Assurance Act," which is currently codified in sections 376.30 through 376.319, Florida Statutes (2004). *See* ch. 83–310, Laws of Fla. While the 1970 enactment created a remedy for those harmed by the pollution of coastal waters, the 1983 enactment provides a cause of action for those harmed by pollution of ground and surface waters. *See* § 376.30, Fla. Stat. (2004) (entitled "Legislative intent with respect to pollution of surface and ground waters"); § 376.302(1)(a), Fla. Stat. (2004) (prohibiting the discharge of "pollutants or hazardous substances into or upon the surface or ground waters of the state").

In contrast to the 1970 enactment, the 1983 enactment does not include a restrictive definition of damages. Instead, the 1983 enactment, which relates to ground and surface water pollution, provides for the recovery of "all damages." Specifically, section 376.313(3), Florida Statutes (2004) (emphasis added), states that "nothing ... prohibits any person from bringing a cause of action ... for *all damages* resulting from a discharge or other condition of pollution covered by ss. 376.30–376.319."

Curd filed his statutory cause of action relating to the pollution of surface and ground water under section 376.313 of the 1983 enactment.[9] Therefore, the "all damages" language of the 1983 enactment applies in this case, not the more restrictive definition of the 1970 enactment.

The plain meaning of "all damages" includes economic damages; and the Legislature has directed that section 376.313(3) be liberally construed. *See* § 376.315, Fla. Stat. (2004) ("Sections 376.30–376.319 ... shall be liberally construed to effect the purposes set forth under ss. 376.30–376.319...."). Consequently, the statute provides commercial fishermen (among others) with a private cause of action. If the statute is overly broad as suggested by the Second District,[10] that is an issue for the Legislature to address.

II. COMMON LAW LIABILITY

I disagree with the majority's holding that those responsible for pollution of ocean waters have a common law duty to protect the purely economic interests of those negatively affected by contamination of the sea. Unlike the majority, I do not believe that under Florida common law commercial fishermen have a unique or special interest that creates a duty to protect their purely economic interest in a healthy ocean.

As the majority explains, four elements are necessary to sustain a negligence claim: duty, breach of the duty, legal causation, and actual damages. *See* majority op. at 1227–28 (citing *Clay Elec. Coop., Inc. v. Johnson,* 873 So.2d 1182, 1185 (Fla.2003)). In 2004, this Court abrogated the ***1231** traditional tort requirement of personal or property damage. *See* *Indemnity Ins. Co. v. Am. Aviation, Inc.,* 891 So.2d 532, 543 (Fla.2004) (limiting the personal and property damage requirement to cases involving contractual privity or product defect and stating that "in general, actionable conduct that frustrates economic interests should not go uncompensated solely because the harm is unaccompanied by any injury to a person or other property"). Because the personal and property damage requirement no longer functions as a filter for unreasonable claims, the function of the duty element of negligence takes on a greater role to filter out the unwarranted claims. *See* *Am. Aviation,* 891 So.2d at 547 (Cantero, J., concurring) (stating that the duty element of traditional negligence should filter out undeserving claims that the personal and property damage requirement would have eliminated). Stated another way, "where the recovery of economic losses is sought on a theory of negligence, the concept of duty as a limiting principle takes on a greater importance than it does with regard to the recovery of damages for personal injury or property damage." *Onita Pac. Corp. v. Trs. of Bronson,* 315 Or. 149, 843 P.2d 890, 896 (1992).

Duty exists as a matter of law and generally can arise from four sources: legislative enactments, judicial interpretations of enactments, judicial precedent, or the general facts of the case. *Clay Elec.,* 873 So.2d at

1185. As this Court explained in *McCain v. Florida Power Corp.,* 593 So.2d 500, 502 (Fla.1992), regarding the fourth category, "[t]he duty element of negligence focuses on whether the defendant's conduct foreseeably created a broader 'zone of risk' that poses a general threat of harm to others." However, a proper application of *McCain* after *American Aviation* must include an analysis of how far-reaching the duty stretches because allowing recovery must have a sensible and just stopping point. *See Lemke–Wojnicki v. Kolodziaj,* 258 Wis.2d 950, 655 N.W.2d 212, 215 (Ct.App.2003); *see also Hamilton v. Beretta U.S.A. Corp.,* 96 N.Y.2d 222, 727 N.Y.S.2d 7, 750 N.E.2d 1055, 1060 (2001) ("[I]n determining whether a duty exists, courts must be mindful of the precedential, and consequential, future effects of their rulings, and limit the legal consequences of wrongs to a controllable degree." (internal quotation marks omitted)). The injury cannot be too remote from the negligence. *See Kolodziaj,* 655 N.W.2d at 215; *see also Lodge v. Arett Sales Corp.,* 246 Conn. 563, 717 A.2d 215, 223 (1998) ("In every case in which a defendant's negligent conduct may be remotely related to a plaintiff's harm, the courts must draw a line, beyond which the law will not impose legal liability.").

Courts have generally recognized that foreseeability in the duty context is not unlimited. *See, e.g., Scott v. Fla. Dep't of Transp.,* 752 So.2d 30, 33 (Fla. 1st DCA 2000) ("It is incumbent upon the courts to place limits on foreseeability, lest all remote possibilities be interpreted as foreseeable in the legal sense." (quoting *Fla. Power & Light Co. v. Macias,* 507 So.2d 1113, 1115 (Fla. 3d DCA 1987))); *Ransom v. Bethany Acad.,* No. A07–1769, 2008 WL 3289853 at *2 (Minn.Ct.App. Aug.12, 2008) ("[A]lthough foreseeability creates a duty of ordinary care, the Minnesota Supreme Court has recognized that there are limits to foreseeability."); *RK Constructors, Inc. v. Fusco Corp.,* 231 Conn. 381, 650 A.2d 153, 156 (1994) ("Many harms are quite literally 'foreseeable,' yet for pragmatic reasons, no recovery is allowed."); *People Express Airlines, Inc. v. Consol. Rail Corp.,* 100 N.J. 246, 495 A.2d 107, 116 (1985) (noting that "members of the general public, or invitees such as sales and service persons at a particular plaintiff's business premises, or persons travelling on ***1232** a highway near the scene of a negligently-caused accident ... who are delayed in the conduct of their affairs and suffer varied economic losses, are certainly a foreseeable class of plaintiffs" but stating that such a class would not be permitted to recover).

Additionally, it is insufficient to show that a defendant owed a duty to the world at large. *See* William L. Prosser, *Handbook of the Law of Torts,* § 36, at 166 (2d ed.1955); *Hamilton,* 727 N.Y.S.2d 7, 750 N.E.2d at 1060 ("The injured party must show that a defendant owed not merely a general duty to society but a specific duty to him or her...."). The purpose of the specific duty requirement is to avoid subjecting an actor to limitless liability to an indeterminate number of individuals conceivably injured by any negligence. *See Hamilton,* 727 N.Y.S.2d 7, 750 N.E.2d at 1060. This Court has stated that the concept of " '[d]uty' is not sacrosanct in itself, but only an expression of the sum total of those considerations of policy which lead the law to say that the particular plaintiff is entitled to protection...." *Gracey v. Eaker,* 837 So.2d 348, 354 (Fla.2002) (quoting William L. Prosser, *Handbook of the Law of Torts,* § 53, at 325–26 (4th ed.1971)).

Duty was appropriately limited in *TS & C Investments, LLC v. Beusa Energy, Inc.,* 637 F.Supp.2d 370 (W.D.La.2009), when local business owners (including truck stops, gas stations, and minimarts) brought a putative class action for economic damages sustained after an oil well blew out, causing closure of an interstate highway. The plaintiffs claimed damages for loss of business and economic opportunity. *Beusa,* 637 F.Supp.2d at 373. The court concluded that there was no independent duty to protect the claimants' economic interests. *Id.* at 381. Were the court to permit recovery, the argument could later be made that anytime the interstate closed due to negligent conduct, all impacted business people could seek economic damages occasioned by the interstate's closure on the grounds that drivers owed a duty to those businesses located within several miles of the interstate. *Id.* Even though economic injury to highway business was foreseeable, the court found recovery to be inappropriate under a duty-risk analysis. *Id.*

Here, the plaintiffs have suffered no personal injury. They have suffered no property damage. The only losses the commercial fishermen allege are economic in nature; and negligence claims for the recovery of economic losses must be predicated on some duty beyond the duty to exercise reasonable care to prevent foreseeable harm. *See* *Am. Aviation,* 891 So.2d at 546 (Cantero, J., concurring) (quoting *Onita Pac. Corp.,* 843 P.2d at 896). Defendants must have "an independent duty to protect [a] plaintiff's purely economic interests." *Id.* However, Mosaic had no such duty to the plaintiff; and if this Court allows commercial fishermen to recover under the foreseeability analysis in *McCain,* then liability will be limitless. Such expansive common law liability turns Mosaic and other similarly situated parties into insurers of the economic interests of all parties who can claim monetary loss because of pollution. The unrestricted imposition of liability on polluters for purely economic damages could create future liability "in an indeterminate amount for an indeterminate time to an indeterminate class." *Ultramares Corp. v. Touche,* 255 N.Y. 170, 179, 174 N.E. 441 (N.Y.1931); *see also* *City of Chicago v. Beretta U.S.A. Corp.,* 213 Ill.2d 351, 290 Ill.Dec. 525, 821 N.E.2d 1099, 1140 (2004) ("[T]he economic consequences of any single accident are virtually endless."). Therefore, denying common law recovery is appropriate.

***1233** Moreover, commercial fishermen in Florida do not have a "special" interest within the "zone of risk" the majority finds Mosaic to have created.[11] Rather, commercial fishermen are few among the tens of thousands of Floridians who earn their living from healthy ocean waters. For example, in 2006, beach tourism alone contributed $24.1 billion to the state's economy and provided 275,630 Floridians with jobs, earning them $7.7 billion. Center for Urban & Environmental Solutions, Florida Atlantic University, Florida Visitor Study 1 (2008) (prepared for Florida Department of Environmental Protection, Bureau of Beaches and Coastal Systems, DEP Contract No. BS014); *see also* Forrest J. Bass, *Calming the Storm: Public Access to Florida's Beaches in the Wake of Hurricane–Related Sand Loss,* 38 Stetson L.Rev. 541, 570–71 (2009) ("Sales tax revenues, parking fees, fines, and tourism dollars are all generated from recreational public beach access to Florida's beaches. For example, beach-related tourism directly generated $21.9 billion in 2000. This included $700 million in sales tax revenue and provided 442,000 jobs. Nearly one-third of non-resident tourists visited Florida's beaches in 2003. Florida ranks behind only California with regard to the size of its tourism revenues. Further, more tourists visit Miami Beach each year than Yellowstone, the Grand Canyon, and Yosemite combined.") (footnotes omitted); Erika Kranz, *Sand for the People: The Continuing Controversy Over Public Access to Florida's Beaches,* 83 Fla. B.J., June 2009, at 11 ("Florida is known worldwide for snowy-white beaches that provide peace, quiet, and natural beauty and also anchor the tourism that constitutes an essential part of the state's economy."); *State v. Osceola County Indus. Dev. Auth.,* 424 So.2d 739, 740 (Fla.1982) (stating that in 1980 tourism generated expenditures of over $17 billion, employed 580,000 Floridians with a $4 billion payroll, and generated state tax revenues of more than $785 million).

Although the majority rules that the commercial fishermen's state licenses set them apart from the general population,[12] if every state-licensed Floridian has a "special" or "unique" interest, then it seems there is endless "foreseeable" liability. Commercial fishermen are a small group, among thousands of licensed Floridians, who can claim economic damages from pollution of coastal waters. For example, hotels and restaurants near the beach, seafood truck drivers, beach community realtors, and yacht salesmen are all licensed by the State to conduct commercial activities that may be negatively affected by pollution of coastal waters. *See* § 509.241(1), Fla. Stat. (2004) (requiring public lodging and public food service ***1234** establishments to obtain licenses from the State); § 322.03(3)(b), Fla. Stat. (2004) (requiring Florida residents to obtain a commercial driver's license from the State in order to operate a commercial motor vehicle); §§ 475.15–475.161, Fla. Stat. (2004) (requiring real estate brokers and broker associates to obtain licenses from the State); § 326.004(1), Fla. Stat. (2004) (requiring yacht brokers and salesmen to obtain licenses from the State). Because the commercial fishermen have not demonstrated that Mosaic owed a specific, unique duty to protect their purely economic interests, I would disallow common law recovery in order to avoid subjecting defendants to limitless liability to an indeterminate number of individuals conceivably injured by any negligence.

III. CONCLUSION

Although I employ different reasoning, I agree with the majority's affirmative answer to the certified question on the commercial fishermen's statutory cause of action. However, unlike the majority, I would answer the certified question on the commercial fishermen's common law cause of action in the negative. I agree with the Second District that "Mosaic did not owe an independent duty of care to protect the fishermen's purely economic interests—that is, their expectation of profits from fishing for healthy fish." *Curd,* 993 So.2d at 1083.

Accordingly, I respectfully concur in part and dissent in part.

All Citations

39 So.3d 1216, 71 ERC 1005, 2010 A.M.C. 2211, 35 Fla. L. Weekly S341

Footnotes

[1] While the plaintiffs filed one of their causes of action under section 376.313, which provides for individual causes of action for pollution of surface and ground waters, it should be noted that section 376.205, Florida Statutes (2004), provides for individual causes of action for pollution of coastal waters and lands also.

[2] Although the statute is phrased in the negative, stating that it does not "prohibit" any person from bringing a cause of action, it does not necessarily follow that the statute does not actively *create* a cause of action. *See* *Aramark Uniform & Career Apparel, Inc. v. Easton,* 894 So.2d 20, 26 (Fla.2004).

[3] Some other state statutes provide that under similar circumstances a fisherman's claim would be permitted. *See* Del.Code Ann. tit. 7, §§ 6207–08 (2001); 35 Pa. Cons.Stat. Ann. § 6018.611 (2003); R.I. Gen. Laws 1956, § 46–12.3–4 (2007).

[4] We also noted that even in these two situations, the economic loss rules would not prevent the bringing of an action and recovery for intentional torts, such as, fraud, conversion, intentional interference, civil theft, abuse of process, and other torts requiring proof of intent. *American Aviation,* 891 So.2d at 543.

[5] After stating these general principles, however, the court, ultimately held that the defendants in that action, who were drilling for oil and caused vast quantities of crude oil to be released into the coastal waters of Southern California, owed a duty of care to the commercial fishermen to refrain from negligent conduct that would reasonably and foreseeably cause a diminution of the aquatic life in those waters.

[6] In addition, some courts have allowed business owners to recover when the alleged injuries occurred on water as a result of activities that occurred on land. *See* *Masonite Corp. v. Steede,* 198 Miss. 530, 23 So.2d 756 (1945) (concluding that a business owner could recover a judgment against a manufacturing plant for loss of profits she claimed she would have made in her business but for the plant's discharge that killed the fish in the Pascagoula River); *Hampton v. North Carolina Pulp Co.,* 223 N.C. 535, 27 S.E.2d 538 (1943) (holding that a fishery owner had stated a cause of action

for nuisance because the waste from a pulp mill had destroyed or diverted the run of the fish, which seriously injured or destroyed his business and diminished the value of his riparian property).

[7] Petitioner's Fourth Amended Complaint (emphasis added).

[8] These sections are currently known as the "Pollutant Discharge Prevention and Control Act." § 376.011, Fla. Stat. (2004).

[9] The majority correctly does not address whether Curd chose to file his cause of action under the appropriate section. The issue is not before us. I note that the majority's statutory ruling pertains to surface and ground water but the common law liability relates to the ocean.

[10] *See* *Curd,* 993 So.2d at 1084 ("[I]f this statute were given the expansive interpretation suggested by the fishermen, it would be very difficult to decide when damages were so remote that they were no longer damages.")

[11] *See* majority op. at 1228 ("[T]he commercial fishermen had a special interest within that zone of risk, an interest not shared by the general community." (citing *Union Oil Co. v. Oppen,* 501 F.2d 558, 568 (9th Cir.1974))). Courts, such as the *Oppen* court, have acknowledged the special interests of commercial fishermen while creating an exception to the economic loss rule, which generally prohibits recovery for economic damages without personal or property damage. *See* *Oppen,* 501 F.2d at 563–68. The existence of a general economic loss rule makes it possible for such courts to conclude that polluters have a duty solely to commercial fishermen without creating limitless, incidental liability for others. Because this Court abrogated the general requirement for personal or property damage in *American Aviation,* this option is no longer available to this Court, and the cases regarding the commercial fishermen exception cited by the majority are inapplicable.

[12] *See* majority op. at 1228 ("The fishermen were licensed to conduct commercial activities in the waters of Tampa Bay, and were dependent on those waters to earn their livelihood.").

Exhibit I

Original Image of 286 So.3d 143 (PDF)

KeyCite Yellow Flag - Negative Treatment
Distinguished by Schiffbauer v. Wal-Mart Stores, Inc., S.D.Fla., April 18, 2023

286 So.3d 143
Supreme Court of Florida.

Charles L. LIEUPO, Petitioner,
v.
SIMON'S TRUCKING, INC., Respondent.

No. SC18-657
|
December 19, 2019

Synopsis
Background: Individual who responded to accident scene to tow tractor-trailer brought an action against trucking company, alleging it was strictly liable for injuries he suffered when he came in contact with spilled battery acid. The Circuit Court, Hamilton County, William R. Slaughter, II, S.J., entered jury verdict in favor of plaintiff. Company appealed. The District Court of Appeal, Wolf, J., concluded that 1970 Pollutant Discharge and Control Act's definition of damages precluded cause of action for personal injuries. Question was certified.

[Holding:] The Supreme Court, Polston, J., held that private cause of action in 1983 Water Quality Assurance Act permitted recovery for personal injury; receding from *Curd v. Mosaic Fertilizer, LLC,* 39 So. 3d 1216.

Ordered accordingly.

West Headnotes (5)

[1] **Environmental Law** Private right of action; citizen suits

Private cause of action in 1983 Water Quality Assurance Act permitted recovery for personal injury; receding from *Curd v. Mosaic Fertilizer, LLC,* 39 So. 3d 1216. Fla. Stat. Ann. §§ 376.031(5), 376.313(3).

1 Case that cites this headnote

[2] **Appeal and Error** Statutory or legislative law

A certified question presenting an issue of statutory construction is reviewed de novo.

2 Cases that cite this headnote

[3] **Environmental Law** Private right of action; citizen suits

Plain meaning of "all damages" in 1983 Pollutant Discharge and Control Act under provision addressing discharge or other condition of pollution included personal injury damages. Fla. Stat. Ann. § 376.313(3).

[4] **Statutes** Language

A court's determination of the meaning of a statute begins with the language of the statute.

6 Cases that cite this headnote

[5] **Statutes** Plain language; plain, ordinary, common, or literal meaning

If the language of a statute is clear, the statute is given its plain meaning, and a court does not look behind the statute's plain language for legislative intent or resort to rules of statutory construction.

7 Cases that cite this headnote

Application for Review of the Decision of the District Court of Appeal – Certified Great Public Importance/ Certified Direct Conflict of Decisions, First District - Case No. 1D17-2065 (Hamilton County)

Attorneys and Law Firms

Michael J. Damaso, II, and Jackson W. Adams of Wooten Kimbrough, P.A., Orlando, Florida; and Peter D. Webster of Carlton Fields, Tallahassee, Florida, for Petitioner

Jason Gonzalez and Amber Stoner Nunnally of Shutts & Bowen, LLP, Tallahassee, Florida, for Respondent

Kansas R. Gooden of Boyd & Jenerette, P.A., Jacksonville, Florida; and Andrew S. Bolin of Bolin Law Group, Tampa, Florida, Amicus Curiae Florida Defense Lawyers Association

Frank A. Shepherd of Gray Robinson, Miami, Florida; and William W. Large of Florida Justice Reform Institute, Tallahassee, Florida, Amicus Curiae Florida Justice Reform Institute

Opinion

POLSTON, J.

***144** We review the decision of the First District Court of Appeal in *Simon's Trucking, Inc. v. Lieupo*, 244 So. 3d 370, 374 (Fla. 1st DCA 2018), a case in which the First District certified the following question of great public importance:

> DOES THE PRIVATE CAUSE OF ACTION CONTAINED IN SECTION 376.313(3), FLORIDA STATUTES, [of the 1983 Water Quality Assurance Act] PERMIT RECOVERY FOR PERSONAL INJURY?

For the reasons explained below and receding from precedent as requested by Lieupo, we answer the certified question in the affirmative.[1]

I. BACKGROUND

The First District set forth the pertinent facts as follows:

> Lieupo filed a complaint against Simon's Trucking, alleging it was strictly liable for injuries he suffered after one of its tractor-trailers was involved in an accident while transporting batteries, spilling battery acid onto the highway. Lieupo alleged he responded to the scene to tow away the truck and came into contact with the battery acid, which caused him serious personal injuries. He filed his complaint under section 376.313(3), Florida Statutes, [of the 1983 Water Quality Assurance Act] which imposes strict liability for the discharge of certain types of pollutants.
>
> Simon's Trucking argued that Lieupo could not seek recovery under section 376.313(3) because that statute did not permit recovery for personal injury. The trial court rejected this argument, and the case proceeded to trial. The jury found the battery acid caused Lieupo's injuries and awarded him a total of $5,211,500 in damages.

Id. at 371 (footnote omitted).

On appeal, the First District reversed the trial court's decision based on this Court's decision in *Curd v. Mosaic Fertilizer, LLC*, 39 So. 3d 1216 (Fla. 2010). *Lieupo*, 244 So. 3d at 371, 374. In *Curd*, 39 So. 3d at 1221, the majority applied the 1970 Pollutant Discharge Prevention and Control Act's (the "1970 act") definition of "damage" to a claim for economic loss brought by commercial fishermen under the 1983 Water Quality Assurance Act (the "1983 act"). The 1970 act defines "damage" as "destruction to or loss of any real or personal property ... or ... any destruction of the environment and natural resources, including all living things *except human beings*, as the direct result of the discharge of a pollutant." *Lieupo*, 244 So. 3d at 373 (emphasis added) (quoting *Curd*, 39 So. 3d at 1221 (quoting § 376.031(5), Fla. Stat.)).

The First District concluded that "the majority [in *Curd*] intended its *in pari materia* application of the definition of damages from the 1970 act to the fishermen's cause of action brought under the 1983 act to be its holding, [not] merely dicta." *Id.* Therefore, the First District concluded that it was "required to apply the 1970 act's definition of damages here, which precludes [Lieupo]'s cause of action for personal injuries." *Id.* at 374.

II. ANALYSIS

[1] **[2]** **[3]** We agree with the First District that it was required to apply this Court's decision in *Curd* and that this Court's application of the definition of "damage" from the 1970 act was part of this Court's holding in *Curd*. However, because it is not supported by the plain meaning of the ***145** 1983 act, we now recede as requested by Lieupo from *Curd*'s incorrect application of the 1970 act's definition of "damage" to a claim brought under the 1983 act.[2]

[4] **[5]** A court's determination of the meaning of a statute begins with the language of the statute. *Lopez v. Hall*, 233 So. 3d 451, 453 (Fla. 2018) (citing *Holly v. Auld*, 450 So. 2d 217, 219 (Fla. 1984)). If that language is clear, the statute is given its plain meaning, and the court does not "look behind the statute's plain language for legislative intent or resort to rules of statutory construction." *City of Parker v. State*, 992 So. 2d 171, 176 (Fla. 2008) (quoting *Daniels v. Fla. Dep't of Health*, 898 So. 2d 61, 64 (Fla. 2005)).

As the First District summarized,

> [c]hapter 376 regulates the discharge and removal of certain pollutants. The two portions of chapter 376 at issue in this case are the Pollutant Discharge [Prevention] and Control Act, passed in 1970 and codified at sections 376.011–376.21, Florida Statutes (the "1970 [a]ct"), and the Water Quality Assurance Act, passed in 1983 and codified at sections 376.30–376.317, Florida Statutes, (the "1983 act"). *The 1970 act is intended to protect coastal waters and adjoining lands, whereas the 1983 act is intended to combat pollution to surface and ground waters*. §§ 376.021, 376.041, 376.30(1)(b), & (2)(b), Fla. Stat.

Lieupo, 244 So. 3d at 371-72 (emphasis added); *see, e.g.*, § 376.021, Fla. Stat. (2011) (entitled "Legislative intent with respect to pollution of coastal waters and lands"); § 376.041, Fla. Stat. (2011) ("The discharge of pollutants into or upon any coastal waters, estuaries, tidal flats, beaches, and lands adjoining the seacoast of the state in the manner defined by ss. 376.011–376.21 is prohibited.").

Specifically, section 376.031(5), Florida Statutes (2011), of the 1970 act defines "damage" as "the documented extent of any destruction to or loss of any real or personal property, or the documented extent, pursuant to s. 376.121, of any destruction of the environment and natural resources, including all living things *except human beings*, as the direct result of the discharge of a pollutant." (Emphasis added.) However, section 376.031 plainly specifies that the definition only applies to sections 376.011 through 376.21, namely the 1970 act. *See* § 376.031 ("When used in ss. 376.011–376.21, unless the context clearly requires otherwise, the term ... 'Damage' means ....").

To be clear, before 1990, the 1970 act did not include a definition of "damage" in its definitions section. In 1990, the Legislature amended the definitions section of the 1970 act (section 376.031) to include the restrictive definition of "damage" applicable only to the 1970 act. *See* ch. 90-54, § 10, at 145, Laws of Fla. Then, in 1996, the Legislature amended the language in the cause of action section of the 1970 act. The language in section 376.205 was changed from "all damages" to "damages, as defined in s. 376.031" (the definitions section of the 1970 act), which limited the damages recoverable under the 1970 act to those defined in section 376.031. *See* ch. 96-263, § 13, at 1030, Laws of Fla.

While the 1970 act involves pollution of coastal waters and adjoining lands, the 1983 act provides a cause of action for those harmed by pollution of ground and surface waters. *See* § 376.30, Fla. Stat. (2011) (entitled "Legislative intent with respect to pollution of surface and ground waters"); § 376.302(1)(a), Fla. Stat. (2011) ***146** (prohibiting the discharge of "pollutants or hazardous substances into or upon the surface or ground waters of the state or lands"). Section 376.315, Florida Statutes (2011), provides that "[s]ections 376.30–376.317, being necessary for the general welfare and the public health and safety of the state and its inhabitants, shall be liberally construed to effect the purposes set forth under ss. 376.30–376.317 and the Federal Water Pollution Control Act, as amended." Additionally, section 376.30(2)(b) provides that the Legislature found and declared that spills, discharges, and escapes of pollutants "as a result of procedures taken by private and governmental entities involving the storage, *transportation*, and disposal of such products pose threats of great danger and damage to the environment of the state, to citizens of the state, and to other interests deriving livelihood from the state." (Emphasis added.)

In contrast to the 1970 act, the 1983 act does not and never has included any definition of damages in its definition section. *See* § 376.301, Fla. Stat. (2011) (setting forth the definitions for sections 376.30-376.317, 376.70, and 376.75 (the 1983 act)). Instead, the 1983 act provides for the recovery of "all damages." Specifically, section 376.313(3), Florida Statutes (2011), of the 1983 act states as follows:

> Except as provided in s. 376.3078(3) and (11), nothing contained in ss. 376.30–376.317 prohibits any person from bringing a cause of action in a court of competent jurisdiction for *all damages resulting from a discharge or*

> *other condition of pollution* covered by ss. 376.30–376.317. Nothing in this chapter shall prohibit or diminish a party's right to contribution from other parties jointly or severally liable for a prohibited discharge of pollutants or hazardous substances or other pollution conditions. Except as otherwise provided in subsection (4) or subsection (5), in any such suit, it is not necessary for such person to plead or prove negligence in any form or manner. Such person need only plead and prove the fact of the prohibited discharge or other pollutive condition and that it has occurred. The only defenses to such cause of action shall be those specified in s. 376.308.

(Emphasis added.)

In this case, because Lieupo filed his cause of action under section 376.313(3) of the 1983 act, the "all damages" language of the 1983 act applies, not the more restrictive definition of the 1970 act that expressly only applies to the 1970 act. The plain meaning of "all damages" includes personal injury damages. *See Merriam-Webster's Collegiate Dictionary* 29 (10th ed. 1994) (defining "all" as "the whole amount or quantity of" and "as much as possible"); *Black's Law Dictionary* 471 (10th ed. 2014) (defining "damages" as "[m]oney claimed by, or ordered to be paid to, a person as compensation for loss or injury"); *Merriam-Webster's Collegiate Dictionary* 291 (10th ed. 1994) (defining "damage" as "loss or harm resulting from injury to person, property, or reputation"); *see also* *State v. Brake*, 796 So. 2d 522, 528 (Fla. 2001) ("[W]here a statute does not specifically define words of common usage, such words are construed in their plain and ordinary sense."). Moreover, the Legislature has directed that section 376.313(3) be liberally construed. *See* § 376.315, Fla. Stat. (2011) ("Sections 376.30–376.317 ... shall be liberally construed to effect the purposes set forth under ss. 376.30–376.317 ...."). Accordingly, section 376.313(3) of the 1983 act provides for the recovery of personal injury damages. If the text of the statute is overly broad as suggested by Simon's Trucking, that is an issue for the Legislature to address.

***147** The majority in *Curd* applied the incorrect definition of "damage" to determine that the 1983 act allows commercial fishermen to recover damages for their loss of income. Specifically, the majority in *Curd* applied the 1970 act's definition of "damage" to a claim brought pursuant to the 1983 act. Section 376.031(5), the 1970 act's definition of "damage," excludes "destruction" to "human beings," which would preclude the recovery of personal injury damages. However, as explained above, the language of the 1970 act's definitions section clearly states that the definitions only apply to sections 376.011 through 376.21, namely the 1970 act. *See* *Curd*, 39 So. 3d at 1230 (Polston, J., concurring in part and dissenting in part).

III. CONCLUSION

For the above reasons, we answer the certified question in the affirmative and hold that the plain meaning of "all damages" in section 376.313(3) of the 1983 act includes personal injury damages. As requested by Lieupo, we hereby recede from *Curd*'s incorrect application of the 1970 act's definition of "damage" to a claim brought under the 1983 act. Accordingly, we quash the First District's decision and remand for proceedings consistent with our decision.

It is so ordered.

CANADY, C.J., and LABARGA, LAWSON, and MUÑIZ, JJ., concur.

All Citations

286 So.3d 143

Footnotes

1 We have jurisdiction. *See* art. V, § 3(b)(4), Fla. Const.

2 The certified question presents an issue of statutory construction, which we review de novo. *Borden v. East-European Ins. Co.*, 921 So. 2d 587, 591 (Fla. 2006).

End of Document

Exhibit J

IN THE UNITED STATES DISTRICT COURT
FOR THE DISTRICT OF MARYLAND

STATE OF MARYLAND,

Plaintiff,

v.

W.L. GORE & ASSOCIATES, INC.,

Defendant.

Civil Action No. 1:24-cv-03656

COMPLAINT

Plaintiff, the State of Maryland (the "State"), by and through Anthony G. Brown, Attorney General of Maryland, and counsel, on behalf of the Maryland Department of the Environment ("MDE"), the Maryland Department of Health ("MDH"), and the Maryland Department of Natural Resources ("DNR"), files this complaint against Defendant W. L. Gore & Associates, Inc. ("Gore" or "Defendant") to address Gore's releases of so-called "forever" chemicals into Maryland's environment for more than fifty years and in support thereof alleges as follows:

INTRODUCTION

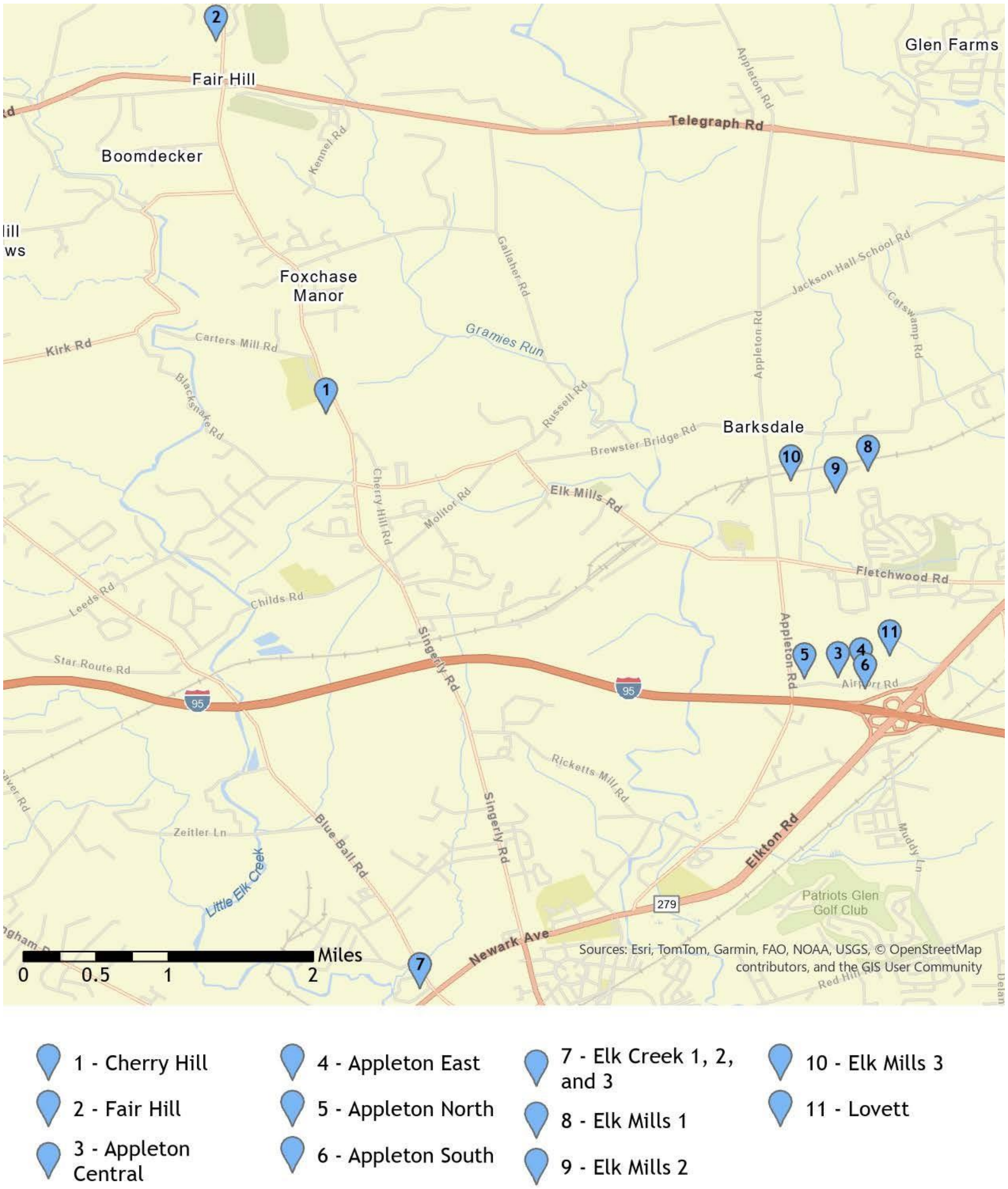
1. The State of Maryland owns and holds in trust the public lands, waters, and resources within its boundaries, and is responsible for the preservation and perpetuation of those natural resources. The State also works to protect the health, safety, and welfare of its residents. The State brings this action to redress Gore's contamination of Maryland's natural resources with toxic per- and polyfluoroalkyl substances ("PFAS"), including but not limited to the hazardous substance perfluorooctanoic acid ("PFOA").

2. PFAS are synthetic chemical compounds that are used in a variety of consumer products, including GORE-TEX®, Scotchgard®, and Teflon®. PFAS are toxic, mobile and persistent in the environment, and cause extensive and long-lasting environmental contamination.

3. Gore's manufacturing operations involved the use of polytetrafluoroethylene ("PTFE") and PFAS in its manufacturing processes. Gore has caused widespread PFAS contamination from its multiple facilities in and around Elkton, Maryland by way of decades-long releases of PFAS into the environment.

4. Gore owns and operates 14 facilities in and around Elkton, Maryland, including several facilities clustered at the same locations. All 14 facilities are within 10 miles of each other and within the same zip code.

5. At and in the vicinity of at least 13 of those facilities, Gore released, discharged, dumped, and/or emitted PFAS that have entered the State's environment through multiple pathways, contaminated its natural resources, and put its residents' health at risk. A map depicting these is below.



6. At the same time that Gore was profiting from the products it manufactured in Maryland, it knew for decades that PFOA was toxic and posed significant risks to human health and the environment and failed to warn the State or the communities living around its facilities of the dangers posed by its PFAS. Instead, Gore concealed those dangers to protect its corporate image and limit its liability.

7. Gore knew that PFAS, now commonly referred to as "forever" chemicals, were persistent and would remain in the environment for hundreds of years, leaving a toxic legacy for generations to come.

8. Gore's acts and omissions concerning the PFAS released from its facilities have caused significant PFAS contamination in the State's drinking water, groundwater, surface water, soil, sediment, wildlife, other natural resources, and property held in trust or otherwise owned by the State.

9. Maryland residents living near Gore's facilities have been and continue to be exposed to PFAS through contaminated drinking water and ingestion or inhalation of contaminated soil and dust, among other ways.

10. Exposure to PFAS may lead to significant negative health effects, including but not limited to:

- Reproductive effects including decreased fertility and pregnancy-induced hypertension;
- Developmental effects in children including low birth weight, accelerated puberty, bone variations, or behavioral changes;
- Increased risk of certain cancers, including prostate, kidney, and testicular cancers;
- Immune system effects, including reduced vaccine response;
- Interference with the body's natural hormones; and
- Increased cholesterol and/or increased risk of obesity.[1]

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[1] Environmental Protection Agency, Our Current Understanding of the Human Health and Environmental Risks of PFAS, https://www.epa.gov/pfas/our-current-understanding-human-health-and-environmental-risks-pfas (last visited Dec. 18, 2024).

11. Although Gore knew that its manufacturing in Cecil County would release PFAS into the environment, endanger people and natural resources, and require significant expense to remediate, it concealed that information from the State and the public.

12. For decades, the State was unaware of the risks posed by Gore's activities in Maryland as a result of Gore's concealment.

13. Although Gore is now conducting a limited investigation into the extent of PFAS contamination around its facilities, this investigation comes decades after Gore knew of the potential risks. Moreover, Gore has not fully delineated the scope of that contamination and has concluded that some sites do not warrant *any* PFAS sampling. While the full extent of PFAS contamination from Gore's facilities is not yet understood, the State already has incurred costs necessary to investigate, treat, and remediate the contamination that Gore has caused.

14. Maryland therefore brings this action to hold Gore responsible for the consequences of Gore's releases of PFOA and other PFAS into Maryland for more than 50 years. Despite its knowledge regarding the potential risks to human health and the environment, and its awareness of the need to abate and mitigate PFAS releases from its Maryland operations, Gore failed, for decades, to prevent PFAS releases into the air, lands, and waters around its facilities.

15. Gore created and profited from its PTFE and "expanded" PTFE ("ePTFE") products while using Maryland's natural resources as a dumping ground for PFAS. Thus, Gore—and not Maryland's residents—must pay to address the PFAS contamination from its facilities.

16. Gore contaminated soil, groundwater, surface waters, and drinking water supplies in Maryland with PFOA and other PFAS via aerial emissions and discharges to water. Gore's actions have contaminated the State's natural resources and have put Maryland residents' health at risk. Through this complaint, the State seeks to (a) recover all past and future costs to investigate, remediate, and restore lands and waters of the State contaminated by PFOA and other PFAS discharged and emitted from Gore's 13 facilities in and around Elkton; (b) abate the public nuisance created by Gore's PFAS emissions, discharges, and releases; and (c) obtain damages for injuries resulting from the contamination.[2]

JURISDICTION AND VENUE

17. Jurisdiction is proper in this Court under 28 U.S.C. § 1331 (civil action under the laws of the United States) and 28 U.S.C. § 2201 (declaratory relief). Jurisdiction is also proper in this Court under 42 U.S.C. § 9613(b) (the Comprehensive Environmental Response, Compensation, and Liability Act ("CERCLA")). Pursuant to 28 U.S.C. § 1367(a) the Court has supplemental jurisdiction of all other claims that form part of the same case or controversy under Article III of the United States Constitution.

18. This Court has personal jurisdiction over Gore because Gore will be served with process in Maryland; transacts business in Maryland; performs work in Maryland;

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[2] This action does not assert any liability on Gore's part regarding the use, manufacture, or sale of aqueous film-forming foam ("AFFF") or fluorosurfactants that were designed for and specifically incorporated into AFFF. For the purposes of this complaint, the term "PFAS" does not include AFFF or fluorosurfactants that were designed for and specifically incorporated into AFFF.

contracts to supply goods in Maryland; manufactures products or performs services in Maryland; caused tortious injury in Maryland; engages in a persistent course of conduct in Maryland; derives substantial revenue from manufactured goods, products, or services used or consumed in Maryland; and/or has interests in or uses real property in Maryland.

19. Venue is proper in this Court pursuant to 28 U.S.C. § 1391(b)(1) as Gore is considered a resident of Maryland, under 28 U.S.C. § 1391(c)(2) as an entity over which this Court has personal jurisdiction. Venue is also proper under 28 U.S.C. § 1391(b)(2) and 42 U.S.C. § 9613(b), because the events and/or omissions giving rise to the State's claims occurred in Maryland, and the property that is the subject of the action is situated in Maryland.

PARTIES

Plaintiff, the State of Maryland

20. The State brings this action (a) directly in its own right, (b) in its *parens patriae* capacity, (c) as trustee of Maryland's natural resources, and (d) under its police powers.

21. The State has an interest as a sovereign and natural resource trustee in protecting the natural resources of the State from contamination. The contamination of the State's natural resources by PFAS constitutes injury to the person and property of the State's residents and to the natural resources of the State, which are held in trust by the State on behalf of all its residents. The State may, for the common good, exercise all the authority necessary to protect its interests and those of its residents.

22. The State, as the public trustee, is empowered to bring suit to protect the corpus of the trust (i.e., the natural resources) for the beneficiaries of the trust (i.e., the public). Protection of the natural resources of the State is a matter of public concern in which the State has an interest apart from that of particular individuals who may be affected.

23. The State brings this action pursuant to its police powers, which include but are not limited to its powers to prevent and abate pollution of the natural resources of the State, to prevent and abate nuisances, and to prevent and abate hazards to the environment and to the public health, safety, and welfare.

24. The State holds significant direct property interests in State-owned lands that have been contaminated by PFAS from Gore's operations, including but not limited to the Fair Hill Natural Resources Management Area.

25. The State, through its Attorney General, also brings this action under Title 7 of the Environment Article, which empowers the Secretary of the Environment, through the Attorney General, to bring suit against any person who "stores[s], discharge[s], treat[s], or dispose[s] of a controlled hazardous substance in this State except: (1) in a controlled hazardous substance facility; and (2) in accordance with [Subtitle 2]." Md. Code Ann. Envir. §§ 7-224, 7-263(a).

26. The State, through its Attorney General, also brings this action under Title 9 of the Environment Article, which empowers the Secretary of the Environment, through the Attorney General, to bring suit against any person who "discharge[s] any pollutant into the waters of this State" without a permit. Envir. §§ 9-322, 9-339(a).

27. The responsibilities of the Attorney General include the investigation, commencement, and prosecution of civil suits on the part of the State. *See* Md. Const. art. V, § 3. "[T]he Attorney General has general charge of the legal business of the State." Md. Code Ann. State Gov't § 6-106.

28. The State is also authorized to seek response costs and declaratory relief from responsible parties, like Gore, pursuant to CERCLA, 42 U.S.C. § 9607, *et seq.*

29. As a result of Gore's acts and omissions as alleged herein, the State has suffered and will continue to suffer injuries to its natural resources and has incurred and will continue to incur costs; to monitor, treat, remediate, and remove PFAS; and to provide oversight of such activities.

Defendant, W.L. Gore& Associates, Inc.

30. Defendant W.L. Gore & Associates, Inc., is a Delaware corporation with its principal place of business at 555 Paper Mill Road, Newark, Delaware 19711. Gore is authorized to conduct business in Maryland. Gore's registered agent for service in Maryland is The Corporation Trust, Incorporated, 2405 York Road, Suite 201, Lutherville, Maryland 21093-2264.

31. Gore was founded in 1958 by Wilbert "Bill" Gore, a chemical engineer and chemist who worked for E.I. DuPont de Nemours and Co. ("DuPont") before he left to start Gore with his wife, Genevieve Gore. Today, Gore is a privately held, global materials science company that reports at least $4.8 billion in annual revenues. Gore specializes in the manufacture of fluoropolymer products and their application in a variety of products, including high-performance fabrics used in GORE-TEX® brand products and products

used in the healthcare, life sciences, mobile electronics, automotive, textiles and apparel, and aerospace industries.

32. Gore is the owner and operator of 13 industrial properties in or around Elkton, several of which are clustered in groups at multi-facility campuses. These facilities, which are the subject of this Complaint (and referred to collectively as the "Gore Facilities"), include:

a. The Cherry Hill facility, located at 2401 Singerly Road, Elkton, Maryland 21921 ("Cherry Hill");

b. The Fair Hill facility, located at 101 Lewisville Road, Elkton, Maryland 21921 ("Fair Hill");

c. The four Appleton facilities (collectively, "Appleton"):

  (1) Appleton Central, located at 301 Airport Road, Elkton, Maryland 21921;

  (2) Appleton East, located at 201 Airport Road, Elkton, Maryland 21921;

  (3) Appleton North, located at 401 Airport Road, Elkton, Maryland 21921;

  (4) Appleton South, located at 100 Airport Road, Building 1, Elkton, Maryland 21921;

d. The three Elk Creek facilities, Elk Creek 1, 2, and 3 (collectively, "Elk Creek"), all located at 295 Blue Ball Road, Elkton, Maryland 21921;

e. The three Elk Mills facilities (collectively, "Elk Mills"):

  1) Elk Mills 1, located at 501 Vieves Way, Elkton, Maryland 21921;

  2) Elk Mills 2; located at 402 Vieves Way, Elkton, Maryland 21921;

  3) Elk Mills 5; located at 105 Vieves Way, Elkton, Maryland 21921; and

f. The Lovett facility, located at 101 Lovett Drive, Elkton, Maryland 21921.[3]

FACTUAL ALLEGATIONS

A. PFAS Endangers Maryland's Environment and Residents.

33. PFAS are highly fluorinated synthetic chemical compounds that include carbon chains containing at least one carbon atom on which all hydrogen atoms are replaced by fluorine atoms. The carbon-fluorine bond is one of the strongest bonds in chemistry and imparts to PFAS their unique chemical properties.

34. The PFAS family, including PFOA, GenX, PFHxA, and PFHpA, has characteristics that cause extensive and long-lasting environmental contamination.

35. PFAS are mobile and persistent in the environment. Because they are water soluble, PFAS quickly spread once introduced into the environment. PFAS also persist in the environment indefinitely because of their multiple carbon-fluorine bonds, which are exceptionally strong and stable, and are resistant to metabolic and environmental degradation processes. Removing PFAS from drinking water sources, soil, groundwater, and other natural resources requires specialized, and expensive, treatment systems. In short, once released, PFAS migrate through the environment, resist natural degradation, contaminate soil, groundwater, and drinking water, and are difficult and costly to remove.

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[3] Gore operates an additional facility in Elkton, the Left Bank facility, located at 505 Blue Ball Rd., Bldg. 310, Triumph Industrial Park, Elkton, Maryland 21921. The State is not at this time asserting claims with regard to the Left Bank facility, based on Gore's representations that no manufacturing occurred there and that "extruded scrap PTFE material" was stored there but in a covered warehouse with no exposure to stormwater. The State reserves all rights in connection with any PFAS contamination resulting from Gore's use of the Left Bank facility.

36. PFAS bioaccumulate and biopersist in animals and are toxic to their health. Because several PFAS, including PFOS and PFOA, are excreted from individual organisms only slowly, ongoing low-level exposure results in a buildup of PFAS within the body. As a result, PFAS can also biomagnify, meaning that their concentration in organic tissue increases as they are consumed up the food chain.

37. PFAS are toxic and cause significant adverse effects to human health. PFOA exposure is associated with numerous adverse health effects in humans, including increases in serum lipids (i.e., high cholesterol); decreases in antibody response to vaccines; high blood pressure and preeclampsia during pregnancy; decreased birthweight, testicular and kidney cancers, ulcerative colitis, and thyroid disease.

38. In March 2021, the Environmental Protection Agency ("EPA") issued a final determination to regulate two PFAS, PFOS and PFOA, as contaminants under the Safe Drinking Water Act, 42 U.S.C. §§ 300f *et seq*. In March 2023, EPA proposed a regulation to establish drinking water standards for PFOS, PFOA, PFHxS, PFNA, PFBS, and HFPO-DA.

39. EPA finalized the proposed Safe Drinking Water Act regulation on April 10, 2024, and published it on April 26, 2024. *See* PFAS National Primary Drinking Water Regulation, 89 Fed. Reg. 32532 (Apr. 26, 2024) (to be codified at 40 C.F.R. Parts 141 and 142). The maximum contaminant level ("MCL") for PFOA is set at 4.0 parts per trillion (or "ppt"), which is at or near the level of detection under current methods. The MCL for HFPO-DA is 10 ppt. EPA "expects that over many years the final rule will prevent PFAS

exposure in drinking water for approximately 100 million people, prevent thousands of deaths, and reduce tens of thousands of serious PFAS-attributable illnesses."[4]

40. Also in April 2024, EPA finalized health-based Maximum Contaminant Level Goals ("MCLGs") for these PFAS, including an MCLG of zero for PFOA, and 10 ppt for HFPO-DA.

41. In addition, EPA finalized the designation of PFOS and PFOA as hazardous substances under CERCLA in April 2024, finalizing a rule that had been proposed in September 2022.

42. At the same time, EPA worked on a parallel track to establish health advisory limits for PFAS and regulate it through other environmental laws. On June 15, 2022, EPA lowered the Health Advisory Limits for PFOA and PFOS. The new interim Health Advisory Limits are 0.004 ppt for PFOA and 0.02 ppt for PFOS.

43. EPA has also sought to limit certain PFAS in manufacturing. In January 2023, EPA proposed a significant new use rule under the Toxic Substances Control Act (15 U.S.C. § 2601 *et seq.*) for inactive PFAS, i.e., PFAS that are currently on the statute's Chemical Substance Inventory but have not been used in manufacturing or processing since 2006. The proposed rule would require any person to notify EPA 90 days before commencing the manufacture, import, or processing of any of the designated PFAS for a significant new use, so that EPA can make a determination that the significant new use

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[4] https://www.epa.gov/sdwa/and-polyfluoroalkyl-substances-pfas (last visited Dec. 18, 2024).

does not pose an unreasonable risk of injury to health or the environment or, if it cannot make that determination, take regulatory action as necessary.

44. Industrial facilities that manufacture or use PFAS, such as the Gore Facilities, are a major source of PFAS contamination in the environment. PFAS are released from these facilities to the land, into the water, and, significantly, through air emissions, which can lead to PFAS contamination in soils, surface water, groundwater, and other natural resources. Widespread contamination has been found around several manufacturing facilities where PFOA was used or released, including but not limited to Chemours' Washington Works facility in Washington, West Virginia, where PTFE has been manufactured since the 1950s.[5]

45. The PFAS that has contaminated Maryland's environment as a result of Gore's actions and omissions will not degrade, and the contamination will persist until the PFAS is removed.

B. Gore's History Regarding PFOA, APFO, and Use in the Creation of Gore's PTFE and ePTFE.

46. Since Gore was founded in 1958, the company has used PTFE to manufacture a variety of products in a wide range of fields, including performance fabrics, electronics, medical devices, and polymer processing.[6]

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[5] The facility was previously owned and operated by E.I. DuPont de Nemours and Company (n/k/a EIDP, Inc.) ("DuPont").

[6] https://www.gore.com/about/the-gore-story#our-history (last visited Dec. 18, 2024).

47. In 1969, Gore invented "expanded" PTFE, or "ePTFE," by stretching PTFE. Expanded PTFE is used in a variety of applications and industries.

48. Gore processes fluoropolymers, primarily PTFE, at its Elkton manufacturing locations. Gore has acknowledged that, historically, it used PTFE products containing residual ammonium perfluorooctanoate ("APFO"), which is the ammonium salt form of PFOA. Gore used APFO largely in the form of fine powder, but it also purchased and used an aqueous PTFE product—referred to as an "aqueous dispersion"—at its Maryland facilities. Gore knew that both the fine powder and the dispersion forms of PTFE that it used contained APFO, and it has stated that APFO concentrations were higher in the dispersions than in the fine powders.

49. Upon information and belief, separate and apart from PFAS contained in PTFE products, Gore also directly used PFOA or other PFAS in its activities at one or more of the Gore Facilities., including in the form of PTFE scrap material.

50. APFO dissociates to PFOA in water, and Gore's use of these products in its manufacturing operations caused PFOA and other PFAS to be released from the Gore Facilities, including in process waste streams and air emissions.

51. Gore has used and continues to use PTFE in the manufacturing process at 11 of the Gore Facilities in Elkton. Specifically, Gore used PTFE aqueous dispersions at Cherry Hill, Fair Hill, Appleton South, all three Elk Creek facilities, and Elk Mills 1. In addition, the Appleton East, Appleton North, Appleton Central, and Elk Mills 5 facilities processed solvents, coatings, and other materials known to contain APFO.

52. Gore has not conducted any sampling to determine whether PFOA or other PFAS are detected around the remaining two Gore Facilities—Lovett and Elk Mills 2—because it claims that those facilities did not process APFO-containing materials. That Gore did not process APFO-containing materials at those facilities, even if true, would not necessarily mean that they do not contain PFAS. For example, to the extent manufacturing did not begin at the Lovett facility until after PTFE suppliers had moved away from the use of PFOA, the manufacturing processes could still be the source of other types of PFAS contamination, including Gen X. And Gore's conclusion that Elk Mills 2 need not be sampled is based solely on a review of historical records. Testing is required at these facilities to determine the accuracy of Gore's assumptions.

53. Upon information and belief, for decades, Gore purchased PTFE products from DuPont, which manufactured, marketed, and sold PTFE products, including those under the brand name Teflon®.

54. Until around 2013, DuPont used APFO (the ammonium salt form of PFOA) at its own Washington Works plant to manufacture several PTFE fluoropolymer products, including fine powders, fluorinated ethylene propylene ("FEP"), and aqueous dispersions. PFOA was used to aid polymerization in DuPont's processes and remained in the products that went to DuPont's customers. For decades, Gore used those same products at its facilities in Elkton.

55. Until 2002, DuPont purchased APFO from The 3M Company ("3M"). 3M's APFO contained additional PFAS as impurities, including perfluorohexanoic acid ("PFHxA") and perfluoroheptanoic acid ("PFHpA").

56. By around 2013, DuPont had transitioned to using GenX to manufacture its PTFE fine powders, FEP, and aqueous dispersions. GenX is a family of chemicals including HFPO dimer acid ("HFPO-DA"), which is a PFAS chemical. Upon information and belief, GenX would also have been present as an impurity in the PTFE that Gore obtained from DuPont and used in the Gore Facilities following the transition from PFOA to GenX.

57. In 2015, DuPont transferred its fluoropolymer business to the newly spun-off Chemours Company ("Chemours"). Chemours continues to make PTFE fine powders, FEP, and dispersions at the Washington Works plant.

58. Upon information and belief, Gore remained a customer through the transition from DuPont to Chemours, and from the use of PFOA to HFPO-DA. In 2018, Chemours awarded Gore the "Plunkett Grand Prize," named after the DuPont chemist who discovered Teflon. The Plunkett Award pre-dates DuPont's spinoff of Chemours. The award, which Gore has won many times, "recognize[s] advancements in products and applications across the Chemours fluoropolymer portfolio, including Teflon™ fluoropolymers" and "[e]ntrants are evaluated for the innovation and value of products, technologies, and applications *enabled by Chemours portfolio of fluoropolymers*."[7] Chemours' decision to give Gore this award three years after DuPont's spinoff of Chemours suggests that Gore continued to use Chemours' fluoropolymer products—e.g., PTFE products.

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[7] https://www.gore.com/news-events/press-release/plunkett-award-for-high-temperature-capacitors (emphasis added; last visited Dec. 18, 2024).

C. Gore Knew, or Should Have Known, of the Harm Caused by its PFAS Contamination.

59. Gore was founded by a prior DuPont employee, and, upon information and belief, Gore and DuPont maintained a close customer relationship for many decades. For example, at least one individual, Dr. Jack Hegenbarth, had detailed knowledge regarding the potential risks of PFOA and the potential for environmental contamination from fluoropolymer manufacturing. Dr. Hegenbarth took his years of knowledge and experience working with PFOA issues at DuPont to Gore in or about 1990.

60. DuPont began using PFOA in the 1950s, and shortly thereafter developed an understanding of the dangers associated with PFAS.

61. For example, DuPont scientists issued internal warnings about the toxicity associated with its PFOA products as early as 1961, including that PFOA caused adverse liver reactions in rats and dogs. DuPont's Toxicology Section Chief opined that such products should be "handled with extreme care" and that contact with the skin should be "strictly avoided."

62. In 1978, based on information it received from 3M about elevated and persistent organic fluorine levels in workers exposed to PFOA, DuPont initiated a plan to review and monitor the health conditions of potentially exposed workers to assess whether any negative health effects were attributable to PFOA exposure. This monitoring plan involved obtaining blood samples from the workers and analyzing the samples for the presence of fluorine.

63. By 1979, DuPont had data indicating that its workers exposed to PFOA had a significantly higher incidence of health issues than did unexposed workers. DuPont did not share these data or the results of its worker health analysis with its customers, the general public, or government entities.

64. The following year, DuPont internally confirmed, but did not make public, that PFOA "is toxic," that humans accumulate PFOA in their tissues, and that "continued exposure is not tolerable."

65. Not only did DuPont know that PFOA accumulated in humans, it was also aware that PFOA could cross the placenta from an exposed mother to her unborn child. In 1981, DuPont conducted a blood sampling study of pregnant or recently pregnant employees. Of the eight women in the study who worked with fluoropolymers, two—or 25%—had children with birth defects in their eyes or face, and at least one had PFOA in the umbilical cord.

66. DuPont reported to EPA in March 1982 that results from a rat study showed PFOA crossing the placenta if present in maternal blood, but it concealed the results of the study of its own plant workers, which revealed the same risk in humans.

67. DuPont was long aware that the PFAS it was releasing from its facilities could leach into groundwater used for public drinking water. DuPont's Medical Director had warned as early as 1982 about the potential for community exposure to PFOA through air emissions venting from the dryers in the PTFE fine powders process at Washington Works.

68. On or about May 22, 1984, DuPont held a meeting at its corporate headquarters in Wilmington, Delaware to discuss health and environmental issues related to PFOA (which DuPont called "C-8") and DuPont's potential liability (the "1984 Meeting").

69. By the time of the 1984 Meeting, DuPont was aware that PFOA had been detected in drinking water around the Washington Works plant, including across the Ohio River in Ohio (confirming the Medical Director's earlier warning that PFOA was traveling by air).

70. The employees in attendance at the 1984 Meeting spoke of the PFOA issue as "one of corporate image, and corporate liability." They were resigned to DuPont's "incremental liability from this point on if we do nothing" because DuPont was "already liable for the past 32 years of operation." They also stated that the "legal and medical [departments within DuPont] will likely take the position of total elimination" of PFOA use in DuPont's business and that these departments had "no incentive to take any other position."

71. Dr. Jack Hegenbarth was one of the attendees at the 1984 Meeting. Two years earlier, on September 28, 1982, Dr. Hegenbarth was copied on an internal DuPont memorandum reporting results of a study done on female employees after they had been transferred out of the PTFE manufacturing area, indicating that PFOA is retained in the blood.

72. In 1983 and 1984, both before and after the 1984 Meeting, internal DuPont memoranda discussed options for abating PFOA air emissions from Washington Works,

including in the PTFE fine powders area. Dr. Hegenbarth is listed as a recipient on these documents. For example, a memorandum dated April 5, 1984, evaluated various emission control options but concluded that none was practical, in part because they would increase the cost of manufacturing PTFE fine powders. That same memorandum noted that "thermal destruction" (i.e., incineration or thermal oxidation) had the highest chance of success, but a subsequent memorandum dated June 28, 1984, expressed a specific concern about "continued emission of C-8 to the atmosphere."

73. DuPont's knowledge of human health risks from PFOA exposure continued to grow during the 1980s and 1990s. By 1988, DuPont began treating PFOA internally as a possible human carcinogen. Yet throughout the following decade, DuPont increased its use and emissions of PFOA despite mounting evidence that it posed a serious risk to human health. For example, in 1999 DuPont received preliminary results from a health study showing that monkeys, even when given the lowest doses of PFOA, suffered liver enlargement, with one so ill it had to be euthanized. Upon information and belief, Gore had access to this and other data indicating PFOA's toxicity.

74. Dr. Hegenbarth later became a Gore employee, in or about 1990, at Gore's Cherry Hill facility. Dr. Hegenbarth took with him his knowledge about PFOA's biopersistence, its ability to contaminate off-site drinking water, and its toxicity. Upon information and belief, Dr. Hegenbarth's role included advising Gore on how to minimize PFOA emissions at its Maryland facilities. Yet PFOA contamination has been found at every Gore facility tested to date.

75. Dr. Hegenbarth was not the only DuPont employee with fluoropolymers experience to move to Gore. Mr. Richard Baillie was a chemical engineer who worked in several roles in DuPont's fluoropolymers operations between 1980 and 1996. In or about March 1996, he joined Gore where, according to Mr. Baillie, he played a "key role" in understanding and dealing with the "PFOA issue."

76. While still with DuPont, Mr. Baillie was listed as one of many recipients of a September 28, 1994, memorandum attaching a report by Roger Zipfel titled "C8 Ammonium Perfluorooctanoate Fluorosurfactant Strategies and Plans" ("The Zipfel Report"). The Zipfel Report noted that the "slow clearance of C-8 from human blood" "justifies the setting of a low permissible exposure" and evaluated strategies for reducing environmental emissions—for example, replacing PFOA "with other less toxic materials." The Report also discussed implementing certain engineering controls to reduce exposure, including scrubbing the emissions from the PTFE fine powders area of DuPont's Washington Works plant. The Zipfel Report estimated the final fate of PFOA used at multiple DuPont manufacturing sites and concluded that, for 1993, more than 14,000 pounds of PFOA left those plants in products destined for customers like Gore. Mr. Baillie, therefore, was on notice that the PTFE products that Gore purchased from DuPont and used in the Gore Facilities contained PFOA.

77. In addition, Appendix A to the Zipfel Report detailed DuPont's "1994 C-8 Plan," one aspect of which was to "Initiate C-8 recycle and recovery from U.S. Gore." The Zipfel Report thus suggests that DuPont and Gore explored recapturing PFOA the waste streams at the Gore Facilities for DuPont to "recycle" and re-use.

78. An internal DuPont document dated January 7, 1997, identifies Gore as a potential "first tier" external customer for evaluations of replacement surfactants, including in the manufacture of expanded PTFE. DuPont also noted that Gore had "shown interest" in products made with a polymer processing aid other than PFOA, and that Gore had a "self-imposed C-8 containment practice."

79. Gore's interest in alternatives to PFOA suggests that it had access to information from DuPont (and potentially other suppliers) about the adverse health effects of PFOA and had concerns about potential releases from its Maryland facilities.

80. In February 1998, a DuPont internal document called "C-8 Integrated Program Plan," prepared by Roger Zipfel and others, noted that: "Our customers, with the exception of Gore, have not expressed concern about C-8." It further states the goal of "assist[ing] customers in decreasing emissions and exposure."

81. In May 2000, DuPont in-house counsel Bernie Reilly sent a personal email reporting 3M's announcement that it would stop making PFOA, "an essential ingredient in the Teflon polymer we sell to big customers like Gore." The email noted, "We knew the material [3M] sold us, a surfactant, also is very persistent and also gets into blood, but so far no signs it has hurt anyone. If it does we are really in the soup because essentially everyone is exposed one way or another."

82. In September 2001, Mr. Reilly sent another personal email saying the following about PFOA: "The compound is an 8 carbon fully fluorinated chain with an ammonia group on the end, a perfect surfactant for our Teflon fine powders that are used

by Gore to coati [sic] fabric. It is very persistent in the environment, and on top of that, loves to travel in water and if ingested or breathed wants to stay in the blood."

83. In 2004, EPA filed an administrative enforcement action against DuPont based on its failure to disclose toxicity and exposure information for PFOA, in violation of the federal Toxic Substances Control Act ("TSCA") and Resource Conservation and Recovery Act ("RCRA"). DuPont eventually settled the lawsuit by agreeing to pay over $16 million in civil administrative penalties and undertake supplemental environmental projects. EPA called the settlement the "largest civil administrative penalty EPA has ever obtained under any federal environmental statute." Upon information and belief, Gore was aware of this public action.[8]

84. In May 2006, the EPA Science Advisory Board stated that PFOA cancer data are consistent with guidelines suggesting exposure to the chemical is "likely to be carcinogenic to humans." Upon information and belief, such findings were known to Dr. Hegenbarth, Mr. Baillie, and Gore.

85. Upon information and belief, Gore was regularly in communication with DuPont employees regarding PFOA issues, including potential methods for abating emissions into Maryland.

86. Indeed, DuPont stated in a public letter to EPA in 2006 that its "[a]queous dispersions contain higher amounts of residual PFOA (ca. 0.2% or 2000 ppm) some of

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[8] https://www.epa.gov/archive/epapages/newsroom_archive/newsreleases/fdcb2f665cac66bb852570d7005d6665.html (last visited Dec. 18, 2024).

which can be emitted from processor plant facilities." DuPont further promised to share its PFOA emissions abatement technology with competitors and customers.[9]

87. Despite this history, Gore continued to use and/or make PTFE at its Elkton facilities, without taking adequate measures to contain PFOA and prevent widespread contamination of the State's natural resources. Upon information and belief, Gore, through these discussions with DuPont and otherwise, knew of the potential for releases of PFOA to air and water from the Gore Facilities and yet did not inform the State.

D. Maryland's Affected Natural Resources

88. Maryland law establishes the State's right and obligation to protect its natural resources. As set forth by the statutory sections below, the State is the steward of the Maryland environment.

89. "The protection, preservation, and enhancement of the State's diverse environment is necessary for the maintenance of the public health and welfare and the continued viability of the economy of the State and is a matter of the highest public priority." Md. Code Ann. Nat. Res. § 1-302(b).

90. Pursuant to statute, "[e]ach person has a fundamental and inalienable right to a healthful environment[.]" Nat. Res. § 1-302(d).

91. "Because the quality of the waters of this State is vital to the public and private interests of its residents and because pollution constitutes a menace to public health and welfare, creates public nuisances, is harmful to wildlife, fish and aquatic life, and

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[9] https://www.epa.gov/sites/default/files/2015-08/documents/dupontresponse.pdf (last visited Dec. 18, 2024).

impairs domestic, agricultural, industrial, recreational, and other legitimate beneficial uses of water, and the problem of water pollution in this State is closely related to the problem of water pollution in adjoining states, it is State public policy to improve, conserve, and manage the quality of the waters of the State and to protect, maintain, and improve the quality of water for public supplies, propagation of wildlife, fish and aquatic life, and domestic, agricultural, industrial, recreational, and other legitimate beneficial uses." Envir. § 4-402.

92. The "quality of the waters of this State is vital to the interests of the citizens of this State[.]" Envir. § 9-302(b). "[B]ecause pollution is a menace to public health and welfare, creates public nuisances, harms . . . and impairs domestic, agricultural . . . and other legitimate beneficial uses of water . . . it is the policy of this State: (1) To improve, conserve, and manage the quality of the waters of this State; (2) To protect, maintain, and improve the quality of water for public supplies . . . and (3) To provide that no waste is discharged into any waters of this State . . . to protect the legitimate beneficial uses of the waters of this State." *Id.*

93. "The General Assembly determines and finds that lands and waters comprising the watersheds of the State are great natural assets and resources." Envir. § 4-101.

94. "It is the policy of the State of Maryland to: . . . (3) Protect the State's natural resources, including the fish and wildlife of the Potomac River, the Chesapeake Bay, and all other waters and waterways of the State." Envir. § 5-5B-03.

95. The "waters of the State" include both surface and underground waters within the boundaries of the State or subject to its jurisdiction. *See* Envir. § 5-101.

96. Under the Maryland Environmental Standing Act, the "General Assembly finds and declares that the natural resources . . . of the State of Maryland are in danger of irreparable harm occasioned by the use and exploitation of the physical environment." Nat. Res. § 1-502.

97. PFAS contamination from Gore's facilities has injured and continues to injure the waters and property of the State and the property, health, safety, and welfare of Maryland's residents.

98. The discharge of PFAS from the Gore Facilities into drinking water constitutes a public nuisance because such discharges create a "condition that is dangerous to health and safety," including a "contaminated water supply" and an "inadequately protected water supply." Md. Code Ann., Health-Gen. § 20-301(a).

99. The State owns lands throughout Maryland that it maintains for the benefit of the public, such as parks and wildlife management areas.

100. The State holds its waters in trust for the State's residents and has an obligation to protect public interests in these waters though, among other things, maintaining the environmental quality of its waters.

101. The State's natural resources include its waters, such as the springs, streams, wetlands, groundwater, ocean waters, and estuaries, within its boundaries or otherwise subject to its jurisdiction.

102. Natural resources and State-owned properties have been injured by past and ongoing contamination caused by PFAS attributable to Gore.

103. PFAS have been found in groundwater, surface water, and soils in Cecil County, Maryland, and the State anticipates that additional PFAS contamination of natural resources will be uncovered as its investigation continues.

104. Because PFAS does not break down in the environment, PFAS contamination will persist in the State's natural resources, damaging their intrinsic value and impairing the public benefits derived from their use and enjoyment.

105. The current and future residents of the State have a substantial interest in having natural resources uncontaminated by PFAS, as do the tourism, recreation, fishing, and other industries that rely upon maintaining a clean environment for their businesses, patrons, and tourists to visit and enjoy.

1. Groundwater

106. Groundwater is a critical and finite natural resource for the people of the State, as the State relies on groundwater for drinking water, irrigation, and agriculture.

107. Maryland relies on groundwater for drinking water supplies. It is the most common form of drinking water supply.

108. In addition to serving as a source of water for drinking, agriculture, and other uses, groundwater is an integral part of the overall ecosystem in the State. Groundwater provides base flow to streams and influences surface water quality, wetland ecological conditions, and the health of aquatic ecosystems. Groundwater keeps water in rivers during times of drought.

109. Groundwater promotes the movement of water and nutrients within and among the State's bodies of water and wetlands, prevents saltwater intrusion, provides subsurface stabilization, and helps to maintain critical water levels in freshwater wetlands.

110. Groundwater and the State's other natural resources are unique resources that help sustain the State's economy.

111. PFAS contamination mobilizes in and through groundwater sources to reach areas beyond the initial source of contamination. This contamination adversely affects the groundwater.

2. Surface Water

112. Surface water is a critical ecological resource of the State. In addition to serving as a source of drinking water, surface water in Maryland is also used for recreational, industrial, agricultural, and other commercial purposes.

113. Surface water also provides aesthetic and ecological values, including supporting aquatic ecosystems, nearby communities, and the residents of the State.

114. PFAS are mobile and persistent in water and can spread great distances from the point of discharge.

3. Sediments and Soils

115. Given the nature of PFAS contamination, PFAS from Gore's facilities has also contaminated soils and sediments.

116. PFAS-contaminated soil poses a risk to human health. PFAS in the soil column serve as a continuing source of contamination of groundwater and other resources of the State. PFAS in sediments, as well as in surface water, support the potential increase

of PFAS concentrations in fish and agricultural resources. PFAS-contaminated soil and dust can also be inadvertently ingested and/or inhaled, and plants grown for food can uptake PFAS from the soil.

4. Biota

117. Biota, including the State's flora and fauna, are critical ecological resources.

118. PFAS contamination threatens animal and plant species because PFAS can cause damage to the liver and immune system of animals and has been shown to damage cell structure and organelle functions in plants.

119. Natural resource injuries to biota in the State negatively impact not only the individual species directly involved, but also the capacity of the injured ecosystems to regenerate and sustain life into the future.

120. In addition, PFAS are subject to biomagnification in the food chain and contaminated biota can therefore be an additional exposure pathway for humans.

E. Gore's PFAS Have Contaminated the State's Natural Resources, Including Sources of Drinking Water, and Gore is Liable for Costs to Remediate and Restore Those Resources.

121. The State's natural resources have been contaminated with PFAS as a result of Gore's acts and omissions. Gore has caused the contamination of the State's groundwater, surface water, drinking water, and other resources, and exposed the State's residents to substantial health risks.

122. In addition, the proximity of the Gore Facilities to one another, and the fact that PFOA and other PFAS can contaminate widespread areas through aerial deposition

from industrial sites, mean that certain natural resources may be affected by more than one facility.

123. The State's investigation is continuing. The investigation is necessary to ascertain the full scope of the contamination attributable to the Gore Facilities and return the natural resources to the condition in which they existed prior to the impact of these contaminants.

124. Gore is liable for the cost of investigation, remediation, and restoration of all the property, soils, sediments, waters, and other natural resources contaminated with their PFAS, as well as for the State's loss of past, present, and future uses of such contaminated natural resources.

125. Most critically, PFAS contamination of groundwater and surface water is impacting the State's drinking water sources. Gore is liable for all of the costs necessary to investigate and treat in perpetuity any and all drinking water wells and sources of drinking water adversely affected by its PFAS.

126. On February 16, 2023, MDE sent a letter to Gore identifying it as a potentially responsible party in relation to the Cherry Hill facility and requested Gore to conduct or participate in the systematic investigation of PFAS contamination at the facility and surrounding area. Gore initially agreed to conduct limited drinking water sampling within a .25-mile radius of the facility. MDE was forced to bear the cost of expanding the sampling program to homes within a one-mile radius.

127. In addition, DNR bore the cost of sampling and treating the groundwater at the Fair Hill Natural Resources Management Area, which is next to the Gore Fair Hill

facility. Sampling conducted by DNR from Spring 2023 through Fall 2024 consistently showed PFOA contamination above the 4 ppt MCL at several locations, as high as 39 ppt. These sampling results also indicate contamination with other PFAS carboxylic acids, including PFHxA and PFHpA.

128. Data collected to date show exceedingly high levels of PFOA around the Cherry Hill and Fair Hill facilities. At addresses on Singerly Road, directly across from the Cherry Hill site, multiple residences showed concentrations of PFOA in their drinking water as high as 800 ppt. The surface water in a small stream nearby returned a result of 740 ppt PFOA. Near the Fair Hill site, PFOA was found in drinking water sources above 100 ppt PFOA. And at the Appleton South site, several samples of on-site groundwater had PFOA concentrations well above 1,000 ppt. These locations are also contaminated with PFHpA, PFHxA, and other PFAS.

129. To the State's knowledge, elevated levels of PFOA have been found around each of the Gore Facilities that has been tested.

COUNT I
PUBLIC NUISANCE

130. The State incorporates by reference the preceding paragraphs as though fully set forth herein.

131. Groundwater, surface water, sediments, soils, and biota are natural resources of the State held in trust by the State.

132. The use, enjoyment, and existence of uncontaminated natural resources is a right common to the general public.

133. The contamination of groundwater, surface water, sediment, soils, and biota with Gore's PFAS constitutes a physical invasion of the State's natural resources and, upon information and belief, real property owned by the State. That same contamination is also an unreasonable and substantial interference, both actual and potential, with (i) the exercise of the public's common right to these natural resources; (ii) the State's special status and authority regarding the natural resources of the State; (iii) the State's ability to protect, conserve, and manage the natural resources of the State, which are by law precious and invaluable public resources held by the State in trust for the benefit of the public; and (iv) the rights of the people of the State to enjoy their natural resources free from interference by pollution and contamination.

134. As long as these natural resources contain PFAS caused by Gore's conduct, the public nuisance continues.

135. Until these natural resources are restored to their pre-injury quality, Gore is liable for the creation and continued presence of a public nuisance in contravention of the public's common right to clean natural resources.

136. Gore discharged PFAS into the natural resources of the State knowing that this would create a public nuisance. Moreover, Gore continued discharging PFAS even after it understood the mobile, persistent, bioaccumulative, and toxic nature of PFAS in the environment.

137. Gore committed each of the above-described acts and omissions with actual malice or with a wanton and willful disregard of persons who foreseeably might be harmed by those acts or omissions.

COUNT II
TRESPASS

138. The State incorporates by reference the preceding paragraphs as though fully set forth herein.

139. Gore's intentional and/or negligent conduct caused PFAS to enter, invade, intrude upon, injure, trespass, and threaten to trespass upon properties the State owns or over which it holds a possessory interest.

140. PFAS released from the Gore Facilities continue to be located on or in the State's property, including but not limited to the Fair Hill Natural Resources Management Area.

141. Gore knew with substantial certainty or should have known that its acts would contaminate the State's property.

142. Gore is therefore liable for trespass and continued trespass.

143. Gore did not and does not have authority, privilege, or permission to trespass upon the aforesaid possessory property interests.

144. The State has never consented to the trespasses alleged herein.

145. Gore has refused and failed to terminate its trespasses, despite being put on notice to do so by the State through its policies, statutes, regulations, orders, and other means.

146. Gore's trespass is of a continuing nature and has produced a long-lasting negative effect upon the property of the State, as Gore knew or had reason to know at all times relevant hereto.

147. Based on its conduct, Gore has, at all times relevant to this action, created, caused, maintained, continued, substantially contributed to, substantially participated in, and/or assisted in the creation of such trespass. Based on its knowledge of the properties and manner of distribution, use, and storage of PFAS, as alleged herein, Gore was or should have been aware that contamination of the State's property was inevitable or substantially certain to result from its conduct.

148. As a direct and proximate cause of Gore's conduct, the State has suffered and continues to suffer damages from Gore's conduct and the presence of PFAS in the State's property, including without limitation incurring costs to assess, investigate, monitor, analyze and remediate contamination, costs to prevent PFAS from injuring additional property of the State, and costs to restore and replace the State's impacted natural resources whose use has been lost or degraded.

149. Gore committed each of the above-described acts and omissions with actual malice or with a wanton and willful disregard of persons who foreseeably might be harmed by those acts or omissions.

COUNT III
NEGLIGENCE

150. The State incorporates by reference the preceding paragraphs as though fully set forth herein.

151. Gore had a duty to the State to ensure that PFAS were not released as a result of the transport, storage, use, handling, release, spilling, and/or disposal of its PFAS and did not injure groundwater, surface water, sediment, soils, and biota in Maryland.

152. Gore had a duty to the State to exercise due care in its manufacturing and other operations at the Gore Facilities.

153. Gore breached these duties by, among other things, failing to conform to the requisite standard of care.

154. Groundwater, surface water, sediments, soils, biota, and other natural resources where Gore's PFAS have come to be located have become contaminated with PFAS as a direct and proximate result of Gore's negligence.

155. As a direct and proximate result of the contamination of the environment from Gore's PFAS, the State has incurred, is incurring, and will continue to incur investigation, clean-up and removal, treatment, monitoring, and restoration costs and expenses for which Gore is liable.

156. Gore committed each of the above-described acts and omissions with actual malice or with a wanton and willful disregard of persons who foreseeably might be harmed by those acts or omissions.

COUNT IV
ENVIRONMENT ARTICLE, TITLE 7, SUBTITLE 2 CLAIM
(Unauthorized Discharge of Controlled Hazardous Substances)

157. The State incorporates by reference the preceding paragraphs as though set forth at length herein.

158. MDE is charged with the responsibility of enforcing Title 7, Subtitle 2 of the Environment Article, which governs the control, handling, storage, disposal, and remediation of hazardous substances, including controlled hazardous substances. Envir. §§ 7-220 through 7-222 and 7-256 through 7-266. The Attorney General is also authorized to prosecute claims arising under Title 7, Subtitle 2 on behalf of the State. Envir. § 7-268.

159. Maryland prohibits the discharge or disposal of a controlled hazardous substance in the State of Maryland except in a controlled hazardous substance facility and in accordance with Title 7, Subtitle 2 of the Environment Article. Envir. §§ 7-222 through 7-224.

160. "Hazardous substance" means any substance defined as a hazardous substance under § 101(14) of CERCLA or identified as a controlled hazardous substance by MDE in the Code of Maryland Regulations ("COMAR"). Envir. § 7-201(l); COMAR 26.13.01.03.

161. "Controlled hazardous substance" is any substance identified by MDE as a hazardous substance, including those substances identified under § 101(14) of CERCLA. Envir. § 7-201(b); COMAR 26.13.01.03.

162. "Discharge" is defined as the addition, introduction, leaking, spilling, or emitting of a pollutant into waters of the State; or placing a pollutant in a location where the pollutant is likely to pollute waters of the State. Envir. § 7-201(h).

163. "Release" means the addition, introduction, leaking, spilling, emitting, discharging, escaping, or leaching of any hazardous substance into the environment. Envir. § 7-201(s).

164. PFOA is defined as a hazardous substance under § 101(14) of CERCLA.

165. PFOA therefore is also a controlled hazardous substance under Title 7, Subtitle 2 of the Environment Article.

166. Gore does not have a permit to release or discharge PFOA, or any other PFAS that qualify as controlled hazardous substances, into groundwater or surface water.

167. Gore has discharged controlled hazardous substances into the waters of the State and is liable for civil penalties up to $25,000 per violation. Envir. § 7-266(a). Each day a violation occurs is a separate violation under Title 7, Subtitle 2 of the Environment Article.

168. The State also is entitled to reimbursement for amounts spent under § 7-220 of the Environment Article in response to Gore's release or threatened release of hazardous substances at the Gore Facilities. Envir. § 7-221.

169. The State further is entitled to injunctive relief due to Gore's historic and ongoing discharges of controlled hazardous substances into the natural resources of the State. Envir. § 7-263

170. Gore committed each of the above-described acts and omissions with actual malice or with a wanton and willful disregard of persons who foreseeably might be harmed by those acts or omissions.

171. The State's investigation remains ongoing, and it reserves the right to seek full recovery for additional violations of Title 7, Subtitle 2 of the Environment Article that are discovered in its investigation.

COUNT V
ENVIRONMENT ARTICLE, TITLE 9, SUBTITLE 3 CLAIM
(Unauthorized Discharge of Pollutants & Wastes)

172. The State incorporates by reference the preceding paragraphs as though set forth at length herein.

173. MDE is charged with the responsibility of enforcing Title 9, Subtitle 3 of the Environment Article, which governs water pollution. Envir. §§ 9-334 through 9-344. The Attorney General is also authorized to prosecute claims arising under Title 9 on behalf of the State. Envir. § 9-344.

174. Under Title 9, Subtitle 3 of the Environment Article, a person may not discharge any pollutant into waters of the State without a discharge permit issued by the Department. Envir. §§ 9-322, 323. Subtitle 3 also prohibits the unpermitted "discharge of any wastes . . . regardless of volume[.]" COMAR 26.08.03.01A(1).

175. "Discharge" is defined as "(1) [t]he addition, introduction, leaking, spilling, or emitting of a pollutant into waters of the State; or (2) [t]he placing of a pollutant in a location where the pollutant is likely to pollute waters of the State." Envir. §§ 9-101(b); *see also* COMAR 26.08.01.01B(20).

176. "Waste" is defined to include industrial waste—which refers to any material resulting from any industrial, manufacturing, trade, or business process—and all other "liquid, gaseous, solid, or other substances which will pollute any waters of this State." COMAR 26.08.01.01B(98); *see also id.* at (40).

177. "Pollutant" is defined to mean: "(1) any waste or wastewater that is discharged from . . . an industrial source, or (2) any other liquid, gaseous, solid, or other substances which will pollute any waters of the State." Envir. § 9-101(g).

178. "Pollution" is defined as any contamination or other alteration of the physical, chemical, or biological properties of any waters of the State, including a change in temperature, taste, color, turbidity, or odor of the waters, or the discharge or deposit of any organic matter, harmful organism, or liquid, gaseous, solid, radioactive, or other substance into the waters of this State, that will render the waters harmful or detrimental to: (1) public health, safety, or welfare; (2) domestic, commercial, industrial, agricultural, recreational, or other legitimate beneficial uses; (3) livestock, wild animals, or birds; or (4) fish or other aquatic life. Envir. § 9-101(h); COMAR 26.08.01.01B(67).

179. The "Department may bring an action for an injunction against any person who violates any provision of [Subtitle 3] or any rule, regulation, order, or permit adopted or issued by the Department under [Subtitle 3]." Envir. § 9-339(a). The "court shall grant an injunction without requiring a showing of a lack of an adequate remedy at law." *Id.* § 9-339(c).

180. Gore is responsible for unauthorized discharges of PFAS into the waters of the State. As Gore violated and continues to violate Title 9, Subtitle 3 by discharging PFAS throughout the State, MDE is empowered to seek an injunction ordering Gore to investigate and fully delineate the scope of PFAS contamination for which Gore is responsible and to ensure the clean-up of such contamination so that the waters of the State are in the same state they existed prior to the discharges. Envir. §§ 9-339(c); 9-302(b)(1) ("To improve,

conserve, and manage the quality of the waters of this State . . . ."); Envir. § 9-302(b)(2) ("To protect, maintain, and improve the quality of the water . . . .").

181. Because Gore discharged PFAS into the waters of this State, it "shall reimburse the Department for the reasonable costs incurred by the Department in conducting environmental health monitoring or testing, including the costs of collecting and analyzing soil samples, surface water samples, or groundwater samples for the purpose of assessing the effect on public health and the environment of the [Gore's] discharge[s]." Envir. § 9-342.2; *see* COMAR 26.14.01.04.

182. Gore has discharged PFAS into the waters of the State and is liable for civil penalties up $10,000 per violation. Envir. § 9-342. Each day a violation occurs is a separate violation under Title 9, Subtitle 3.

COUNT VI
ENVIRONMENT ARTICLE, TITLE 9, SUBTITLE 4 CLAIM
(Injunctive Relief)

183. The State incorporates by reference the preceding paragraphs as though set forth at length herein.

184. PFAS are "dangerous contaminant[s]" because when they are "present in a public water system, they present an imminent and substantial danger to the health of individuals." Envir. § 9-405(a).

185. Upon receipt of information that PFAS "[are] present in or likely to enter a public water system," the Secretary of MDE "may take any action necessary to protect the health of the individuals whose health is or would be endangered" by the PFAS. Envir.

§ 9-405(b)(1). The actions the Secretary of MDE may take include suing “for injunctive or other appropriate relief.” *Id*. § 9-405(b)(2)(ii).

186. To stop PFAS from entering public water systems, the Secretary of MDE may seek an injunction that orders Gore to investigate and fully delineate the scope of PFAS contamination for which the Gore is responsible and to ensure the clean-up of such contamination so that the water is in the same state it was in prior to the discharges.

COUNT VII
COMPREHENSIVE ENVIRONMENTAL RESPONSE, COMPENSATION, AND LIABILITY ACT, 42 U.S.C. § 9607(A)

187. The State incorporates by reference the preceding paragraphs as though set forth at length herein.

188. Under CERCLA, 42 U.S.C. §§ 9601, *et seq*., owners and operators of facilities are liable for “all costs of removal or remedial action incurred by . . . a State,” occasioned by a “release, or a threatened release which causes the incurrence of response costs, of a hazardous substance,” “damages for injury to, destruction of, or loss of natural resources, including the reasonable costs of assessing such injury, destruction, or loss resulting from such a release,” and other forms of compensation. 42 U.S.C. § 9607(a).

189. Gore has, at all relevant times, been an “owner” and/or “operator” of each of the Gore Facilities.

190. There have been “releases,” 42 U.S.C. § 9601(22), of “hazardous substances,” 42 U.S.C. § 9601(14), from Gore’s facilities, including releases or threatened releases of PFOA and other PFAS substances exhibiting similar characteristics. Upon information and belief, these releases or threatened releases are ongoing.

191. The State has incurred and will continue to incur necessary costs of response pursuant to CERCLA Section 107(a), all of which are consistent with the national contingency plan, as a result of releases and/or threatened releases of hazardous substances at and from the Gore Facilities. 42 U.S.C. § 9607(a)(4)(A).

192. Upon information and belief, the State has incurred and/or will incur damages for injury to, destruction of, or loss of natural resources, including the reasonable costs of assessing such injury, destruction, or loss resulting from such releases and/or threatened releases of hazardous substances from the Gore Facilities. 42 U.S.C. § 9607(a)(4)(C).

COUNT VIII
COMPREHENSIVE ENVIRONMENTAL RESPONSE, COMPENSATION, AND LIABILITY ACT, 42 U.S.C. § 9613(g)

193. The State incorporates by reference the preceding paragraphs as though set forth at length herein.

194. CERCLA § 113(g)(2) provides in pertinent part: "In any action described in this subsection the court shall enter a declaratory judgment of liability for response costs or damages that will be binding on any subsequent action or actions to recover further response costs or damages." 42 U.S.C. § 113(g)(2).

195. The Declaratory Judgment Act further states: "In a case of actual controversy within its jurisdiction . . . any court of the United States, upon the filing of an appropriate pleading, may declare the rights and other legal relations of any interested party seeking such declaration." 28 U.S.C. § 2201.

196. An actual controversy now exists because Gore is liable under CERCLA § 107(a) for all costs and damages compensable to the State in connection with the release or threatened release of hazardous substances from the Gore Facilities. 42 U.S.C. § 9607(a).

197. The State seeks a judicial declaration of rights pursuant to CERCLA § 113(g)(2), binding on Gore in any subsequent action or actions to recover response costs or other damages incurred by the State, as appropriate and in the interest of justice. 42 U.S.C. § 9613(g)(2).

PRAYER FOR RELIEF

WHEREFORE, the State requests that this Court enter judgment against Gore as follows:

a. Finding Gore liable for all costs to assess, investigate, mitigate, clean up and remove, remediate, restore, treat, monitor, and otherwise respond to PFAS contamination from Gore’s facilities so the contaminated natural resources are restored to their original condition;

b. Finding Gore liable for all damages to compensate the residents of the State for the lost use and value of its natural resources during all times of injury caused by PFAS and for such orders as may be necessary to provide full relief to address risks to the State, including, but not limited to, the costs of:

i. Past and future testing of natural resources where Gore's PFAS were transported, stored, used, handled, released, spilled, and/or disposed and, thus, likely caused PFAS contamination;

ii. Past and future treatment of all natural resources where Gore's PFAS were transported, stored, used, handled, released, spilled, and/or disposed and which contain detectable levels of PFAS until restored to non-detectable levels; and

iii. Past and future monitoring of the State's natural resources where Gore's PFAS were transported, stored, used, handled, released, spilled, and/or disposed as long as there is a detectable presence of PFAS, and restoration of such natural resources to their pre-discharge condition;

c. Ordering Gore to pay for all costs related to the investigation, cleanup, restoration, treatment, and monitoring of PFAS contamination of the State's natural resources attributable to Gore's PFAS;

d. Ordering Gore to pay all damages to the State at least equal to the full cost of restoring the State's natural resources to their original condition prior to the PFAS contamination attributable to Gore's PFAS;

e. Ordering Gore to pay all compensatory damages for economic damages and for the lost value (including lost use) of the State's natural resources as a result of the PFAS contamination attributable to Gore's PFAS of such natural resources;

f. Ordering Gore to pay all other damages sustained by the State in its public trustee, *parens patriae*, and regulatory capacities as a direct and proximate result of Gore's acts and omissions alleged herein;

g. Awarding all compensable costs and damages available to the State under CERCLA Section 107(a);

h. Declaring Gore liable for all past and future response costs pursuant to CERCLA Section 113(g)(2);

i. Entering an order against Gore to abate or mitigate the PFAS contamination that it caused by its PFAS;

j. Entering an order requiring Gore to investigate and delineate the full extent of all contamination for which Gore is responsible;

k. Enjoin Gore from causing further PFAS releases;

l. Entering an order requiring Gore to establish an abatement fund to ensure the cleanup of its PFAS contamination, so that the waters of the State are in the same state they existed prior to PFAS discharges;

m. Awarding the State compensatory damages in an amount to be determined by the trier of fact;

n. Awarding the State punitive damages in an amount to be determined by the trier of fact;

o. Awarding the State costs and fees in this action, including reasonable attorneys' fees, incurred in prosecuting this action, together with prejudgment interest, to the full extent permitted by law; and

p. Awarding the State such other relief as this Court deems appropriate.

DEMAND FOR JURY TRIAL

The State demands a trial by jury on all claims for which a jury trial is available.

Dated: December 18, 2024

ANTHONY G. BROWN
Attorney General of Maryland

/s/Patricia V. Tipon
Patricia V. Tipon (28786)
Julie Kuspa (21432)
Matthew Zimmerman (01222)
Assistant Attorneys General
Office of the Attorney General
1800 Washington Boulevard, Suite 6048
Baltimore, Maryland 21230
patricia.tipon@maryland.gov
matthew.zimmerman@maryland.gov
julie.kuspa@maryland.gov
(410) 537-3061

ADAM D. SNYDER (25723)
Assistant Attorney General
Office of the Attorney General
120 E. Baltimore Street
Baltimore, Maryland 21202
adam.snyder1@maryland.gov
(410) 767-1409

and

SCOTT E. KAUFF (20260)
ALEXANDER LATANISION*
DEREK Y. SUGIMURA (28600)
LAW OFFICES OF JOHN K. DEMA, P.C.
One Central Plaza
11300 Rockville Pike, Suite 112
Rockville, Maryland 20852
skauff@demalaw.com
alatanision@demalaw.com
dsugimura@demalaw.com
(202) 309-0200

MATTHEW K. EDLING*
STEPHANIE D. BIEHL*
ASHLEY B. CAMPBELL*
PAUL M. STEPHAN*
SHER EDLING LLP
100 Montgomery St. Ste. 1410
San Francisco, California 94104
matt@sheredling.com
stephanie@sheredling.com
ashley@sheredling.com
paul@sheredling.com
(628) 231-2500

WILLIAM J. JACKSON*
JOHN D.S. GILMOUR*
KELLEY DRYE & WARREN LLP
515 Post Oak Blvd
Houston, Texas 77027
bjackson@kelleydrye.com
jgilmour@kelleydrye.com
(713) 355-5000

/s/Melissa E. Byroade
MELISSA E. BYROADE (31335)
ERIN HODGE*
KELLEY DRYE & WARREN LLP
Washington Harbour
3050 K Street NW, Suite 400
Washington, D.C. 20007
mbyroade@kelleydrye.com
ehodge@kelleydrye.com
(202) 342-8823

Special Counsel for the State of Maryland

**Pro hac vice applications forthcoming*