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Subject: New written testimony for LU-24-027
Date: Wednesday, June 11, 2025 10:34:57 AM
Attachments: Andrews et al 2023.pdf
PFAS burden Pam Castle.pdf
Costello and Lee 2024.pdf
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Good morning,

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Thank you,
Pam Castle

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Discussion

Discussion. Has the human population become a sentinel for the adverse effects of PFAS contamination on wildlife health and endangered species?

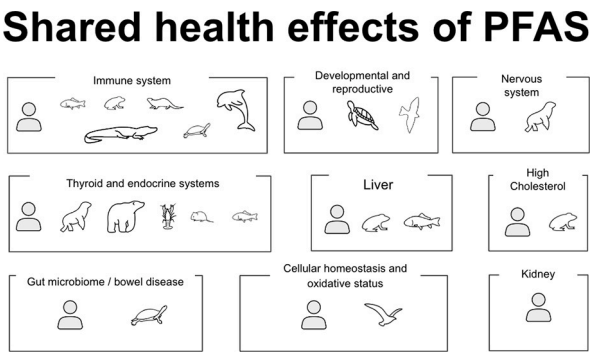
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HIGHLIGHTS

- *Per-* and polyfluoroalkyl substances (PFAS) are global environmental contaminants.
- PFAS are detected in wildlife globally.
- There is concordance between findings on PFAS health effects in wildlife and humans.
- Humans can be regarded as a sentinel species for adverse health impacts of PFAS.
- Wildlife conservation must consider health impacts of chemical contaminants.

GRAPHICAL ABSTRACT



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ABSTRACT

Global contamination with *per-* and polyfluoroalkyl substances (PFAS) poses a threat to both human health and the environment, with significant implications for ecological conservation policies. A growing list of peer-reviewed publications indicates that PFAS can harm wildlife health and that the adverse effects associated with PFAS exposure in wildlife are in concordance with human epidemiological studies. The correlation of cross-species data supports a unique perspective that humans can be regarded as a sentinel for PFAS effects in other species. The health harms due to PFAS are potentially most concerning for populations of endangered and threatened species that are simultaneously exposed to PFAS and other toxic pollutants, and also face threats to their survival due to habitat loss, degradation of ecosystems, and over-harvesting. Human epidemiological studies on the PFAS doses associated with health harm present a rich source of information about potential impacts on wildlife health due to PFAS. Our analysis suggests that national and international efforts to restrict

Abbreviations: DDT, Dichlorodiphenyltrichloroethane; NHANES, National Health and Nutrition Examination Survey; PCB, Polychlorinated biphenyl; PFAS, *Per-* and polyfluoroalkyl substances; PFBS, Perfluorobutane sulfonic acid; PFCA, Perfluoroalkyl carboxylic acid; PFDA, Perfluorodecanoic acid; PFDoDA, Perfluorododecanoic acid; PFHpA, Perfluoroheptanoic acid; PFHxA, Perfluorohexanoic acid; PFHxS, Perfluorohexane sulfonic acid; PFNA, Perfluorononanoic acid; PFOA, Perfluorooctanoic acid; PFOS, Perfluorooctane sulfonate; PFSA, Perfluoroalkyl sulfonic acid; PFTeDA, Perfluorotetradecanoic acid; PFTrDA, Perfluorotridecanoic acid; PFUnDA, Perfluoroundecanoic acid; POP, Persistent organic pollutant; ppt, parts per trillion; U.S. EPA, United States Environmental Protection Agency.

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the discharges of PFAS into the environment and to clean up PFAS-contaminated sites present an opportunity to protect wildlife from chemical pollution and to advance species conservation worldwide.

1. Using humans as sentinels to better understand chemical pollution impacts on the biodiversity crisis

In a classic ecotoxicological approach, monitoring of indicator species such as sensitive bird populations has provided information about the health harms due to bioaccumulative pollutants such as PCBs and DDT, including information highly relevant to human health (Bowerman et al., 2002; Solla et al., 2016). As the global wildlife and biodiversity conservation crisis deepens, there is an urgent need to better protect wildlife from the threats of chemical pollution. One such path towards this goal is to shift the perspective so as to understand how what we know from studies in humans applies to wildlife health. The use of animals as toxicity sentinels has been explored and developed for decades as an approach to protect human health. In contrast to this anthropocentric perspective, more recent frameworks emphasize a One Health approach that considers shared health risk for all species, which may provide better opportunities to protect human and wildlife health simultaneously within a global ecosystem (Destoumieux-Garzon et al., 2018). However, adoption of such frameworks has been slow, with examples so far limited to acute high dose chemical exposures or infectious disease risk. Just a handful of publications to-date explored cross-species extrapolation from humans to animals, which may be appropriate when humans are the most extensively studied exposed population (Beasley, 2009; Rabinowitz et al., 2009; Rabinowitz et al., 2008).

Per- and polyfluoroalkyl substances (PFAS) are recognized as pervasive contaminants associated with harmful effects on the global environment and all species that live on the planet (Evich et al., 2022). We use the term "PFAS" in this Discussion to describe this entire broad class of anthropogenic fluorinated chemicals that have been manufactured globally since the 1950s (Kwiatkowski et al., 2020; U.S. Environmental Protection Agency, 2023a; Wang et al., 2021). Research literature presents overwhelming evidence from studies in humans about the health harms of PFAS exposures. Here we explore the idea that humans can be used as sentinels for wildlife health to enable conservation measures for wildlife and endangered species. Loring and co-authors have previously noted "... the versatility of the word 'sentinel', which generally describes a sentry or guard, but also can be used to refer to the act of keeping watch, as well as the position from which watch is kept" (Loring et al., 2017). Here we adapt this wider epistemological and policy perspective within which we seek to explore the concept of humans as a sentinel species.

A quickening loss of biodiversity threatens species and ecosystems across the globe with significant negative implications both for food security and human health world-wide (World Health Organization and Secretariat of the Convention on Biological Diversity, 2015). Estimates published in 2023 suggest that 48 % of species globally are experiencing population declines, a feature of population dynamics classified as a symptom of potential extinction (Finn et al., 2023). Anthropogenic influences can impact wildlife health, species diversity, and species survival through alteration of physical, biological and chemical composition of terrestrial and aquatic ecosystems (Rillig et al., 2021). In particular, the chemical contamination of the environment with industrial chemicals, pesticides, and a variety of pollutants discharged by the global economy is both a considerable burden (Persson et al., 2022; Sigmund et al., 2022) and a growing peril (Naidu et al., 2021). Yet, the existing frameworks of industrial chemical regulation in individual countries and globally have not yet given sufficient consideration to the impact of toxic chemicals on wildlife and have not yet developed effective approaches to species conservation and wildlife protection in the face of pervasive anthropogenic pollution.

There is a growing recognition that conservation of biodiversity plays a fundamental role in sustaining all aspects of society. The need to address global threats to biodiversity, and action steps for doing so, were outlined in the Kunming-Montreal Global Biodiversity Framework adopted in December 2022 at the 15th meeting of the Conference of Parties to the UN Convention on Biological Diversity. One of the eight primary targets includes reducing pollution risks from pesticides and highly hazardous chemicals "to levels that are not harmful to biodiversity and ecosystem functions and services, considering cumulative effects" (Convention on Biological Diversity, 2022). The Framework emphasizes the need for an integrated whole-of-government and civil society approach to reduce threats to biodiversity and calls for urgent actions to protect biodiversity that should be initiated immediately.

2. PFAS health effects have been extensively studied in humans

The human health effects of PFAS exposure have been well studied in areas of severe local contamination such as the C8 science panel studies in the mid-Ohio River Valley in the United States (Frisbee et al., 2009) and in human studies evaluating health impacts with exposure differentials across the general population. State, federal, and international human health risk assessments for regulatory action on some PFAS have drawn on the vast literature of likely causal associations between PFAS exposure and adverse health effects in humans to establish health protective exposure limits. The U.S. Environmental Protection Agency 2023 toxicity assessment for the proposed drinking water standard for perfluorooctane sulfonic acid (PFOS) in drinking water included 423 epidemiological studies, 45 animal studies, 89 toxicokinetic studies, 22 physiologically-based pharmacokinetic and pharmacodynamic modeling (PBPK) studies and 301 mechanistic studies (U.S. Environmental Protection Agency, 2023a). A similar number of studies were included in the proposed drinking water standard for perfluorooctanoic acid (PFOA) (U.S. Environmental Protection Agency, 2023b). An effort to catalog the studies on health and toxicological effects for PFAS other than PFOA and PFOS identified 1067 studies (505 in people, 385 in animals) for 29 different PFAS compounds (Pelch et al., 2022).

The well-studied link between PFAS exposure in humans and associated health harms along with widespread contamination of wildlife provide an important opportunity to explore the potential for humans to serve as sentinels for PFAS impacts in other species. In this Discussion, our goal is to bring to the attention of the research community and government decision makers the threat PFAS pollution poses to wildlife health, emphasizing that health impacts of globally significant chemical contaminants must be considered in wildlife conservation strategies.

As a result of uncontrolled discharges into the environment, PFAS now pollute water, soil, air and living organisms, and are found in the blood of people around the world (Evich et al., 2022). In the past two decades, research worldwide revealed that exposure to PFAS can cause numerous health harms, as observed in experimental toxicology studies with laboratory animals and in epidemiological studies of highly exposed human populations in the vicinity of contaminated sites as well as in the general population (Fenton et al., 2021; Sunderland et al., 2019). While initial health studies regarding PFAS exposure focused on two most-studied members of this class, PFOA and PFOS, a growing body of scientific research finds that exposure to numerous types of PFAS can harm health (Pelch et al., 2022; Temkin et al., 2020), acknowledging the importance of considering the health impacts of PFAS mixtures (Goodrich et al., 2023).

3. PFAS are detected in wildlife globally

Unlike other classes of industrial chemicals with an identified impact

on wildlife, such as polychlorinated biphenyls (PCBs) that have been phased out from use globally, various types of PFAS continue to be produced, as new fluorochemicals are developed to replace PFOA and PFOS whose use was either phased out or decreased globally. Further, as documented by environmental monitoring studies, there is seemingly no place on the planet untouched by PFAS contamination. A study of the snow on Mount Everest found evidence of PFAS presence, perhaps partially related to PFAS leaching from climbing gear, as well as due to air deposition from the global atmospheric PFAS pollution (Miner et al., 2021).

The manufacturers of PFAS have known for decades that these substances can accumulate in the bodies of living organisms. In 1998, testing by 3M chemical manufacturing company could only find uncontaminated human blood by using archived samples from the Korean War fought in 1950–1953 (3M, 2000). Subsequent testing sponsored by 3M in 2001 documented the ubiquitous presence of PFOS in a global collection of tissue samples for turtles, fish, birds such as bald eagles and albatrosses, and marine mammals such as seals and polar bears (Giesy and Kannan, 2001).

Wildlife are exposed to PFAS through their habitat, such as contaminated waterways (Teunen et al., 2021), as well as their diet (Boisvert et al., 2019). Various PFAS can accumulate in living organisms following exposure to contamination. However, unlike other persistent pollutants, such as organochlorines, PFAS do not tend to accumulate in fat, but rather tend to bind to proteins (Butt et al., 2010). Measurement of PFAS in blood, eggs and other tissues is often limited to a select number of compounds due to analytical capabilities or other limitations in wildlife studies. The set of approximately two dozen PFAS commonly included in various monitoring studies may only account for a fraction of this wide class of chemicals and could underestimate exposure as non-targeted analysis has shown (Barrett et al., 2021; McDonough et al., 2019; Spaan et al., 2020; Wang et al., 2021). Current targeted methods may not be keeping pace with the shifting patterns in global manufacturing of PFAS materials (De Silva et al., 2021).

Hundreds of studies reported the accumulation of various PFAS in animals from a variety of taxa. PFAS are found in wildlife on every continent from manatees in the U.S. state of Florida to fish in Ethiopia and Uganda, flamingos in the Caribbean, whales and dolphins in Hawaii, polar bears across the Arctic, ducks in Australia, sea turtles in Mexico, seafood caught off the coast of Bangladesh and freshwater fish across the United States (Barbo et al., 2023; Vendl et al., 2022; Vendl et al., 2021). For additional resources and visualization of the studies that detected PFAS in the environment, we direct the readers to the U.S. Environmental Protection Agency website (https://echo.epa.gov/trends/pfas-tools) as well as the global map of PFAS contamination in wildlife developed by our organization (Environmental Working Group, 2023, https://www.ewg.org/interactive-maps/pfas_in_wildlife/map/) to view a visualization of a subset of studies documenting this far-reaching contamination.

4. Cross-species extrapolation to assess PFAS toxicity

Current thinking behind traditional toxicity testing is moving towards greatly reducing the use of animals in testing. As alternatives to animal testing, there is now more focus on examining cell-based, computational, and mechanistic approaches to toxicity assessment. Such advanced methods would increase the understanding of biological mechanisms and pathways of adverse effects and their conservation between species and thereby support the research efforts to extrapolate effects across species. For example, the Key Characteristics approach and the Adverse Outcomes Pathways approach (LaLone et al., 2021) offer conceptual frameworks for gathering and utilizing data across species to understand the shared biological linkages that can be used to predict adverse effects (Arzuaga et al., 2019; Germolec et al., 2022; La Merrill et al., 2020; Lind et al., 2021; Rusyn et al., 2021; Smith et al., 2020). Additionally, analytical technologies like transcriptomics and metabolomics which can assess biomarkers of adverse health effects, such as oxidative stress, DNA damage and changes in hormonal levels, are rapidly being developed in laboratory model species and in humans, and are also applied to investigations of wildlife health. Within this framework, researchers can leverage existing data, with a considerable amount of information from humans, as well as generate data from non-lethal and non-invasive sampling methods to predict potential adverse effects from PFAS exposure to a diversity of species.

PFOA, PFOS, and other compounds in the PFAS chemical family have been linked with a broad array of toxic effects in humans and laboratory animals, including harm to the immune system, harm to reproduction and fetal development, changes in the endocrine balance and increased risk of various cancers (National Academies of Sciences, 2022). Here we briefly review the literature on PFAS toxicity in environmental field studies of wildlife, which indicate similar target organ systems and health impacts in wildlife as humans and model organisms, from PFAS exposure (Acosta-Tlapalamatl et al., 2022; Beaulieu and Costantini, 2014; Bedia, 2022). Importantly, while not covered in this review, PFAS exposure has been also shown to affect the health of domesticated animals, including companion pets and livestock (Rock et al., 2023). Published studies, described in more detail below (Table 1), include reports of the impact of PFAS on hormonal levels in polar bears (Pedersen et al., 2016) and hooded seals (Gronnestad et al., 2018); the immune systems and disease resistance in fish (Guillette et al., 2020), alligators (Guillette et al., 2022) and marine mammals (Ishibashi et al., 2008; Kannan et al., 2006; Soloff et al., 2017); and hatching success in birds and emergence success in turtles (Custer, 2021; Custer et al., 2014) (Wood et al., 2021), as well as metabolic alterations associated with PFAS exposure in turtles (Beale et al., 2022), among other effects. Of note, associations with health parameter changes were reported for several different members of the PFAS family, including the sum of more than 20 PFAS, PFOS and other long chain PFAS, as well as new compounds from PFAS class with regional contamination patterns such as Nafion byproduct 2. Table 1 summarizes the health effects in wildlife associated with summed PFAS exposure and/or individual PFAS, while Table 2 indicates the number of studies where each compound was reported in association with these health effects.

It is important to recognize that reports on PFAS and wildlife health published in peer-reviewed scientific literature to date are likely a significant underestimate of all possible impacts of PFAS on wildlife, due to the practical difficulties of carrying out studies in the field as well as ethical and legal considerations protecting the health and survival of already endangered species. The PFAS-associated health effects described in the studies cited in Table 1 are indicative of chemical stressors that can impact both species fitness and survival and therefore can be considered a factor influencing population dynamics, with a potential connection to overall biodiversity loss (Groh et al., 2022). Impacts on the immune, nervous and endocrine systems, reproduction and development and metabolism are reported consistently by the published studies on different taxa.

4.1. Immune system

The immune system is one of the most sensitive targets of PFAS toxicity, as observed by human studies that examined reduced vaccine response and increased susceptibility to infection and disease (Budtz-Jorgensen and Grandjean, 2018; European Food Safety Authority, 2020). Similarly, studies in reptiles, mammals, fish and birds have shown immune system impacts of PFAS exposure. Researchers studying alligator populations in the industrially contaminated Cape Fear River Basin (North Carolina, United States) observed an association between elevated levels of PFAS and altered biomarkers of innate immune activity related to autoimmune response (Guillette et al., 2022). Additionally, an increased incidence of skin lesions and unhealed wounds was observed, which is uncharacteristic for American alligators, and may be similar to the elevated risk of slow-healing wounds associated

Table 1
Recent studies of impacts of PFAS on wildlife health from field studies.

Species	Continent/Region	System affected and findings
Invertebrate and fish		
Crayfish (*Faxonius immunis, F. rusticus, F. virilis*)	North America, (Northern Michigan, USA)	Behavioral impacts Increased exposure to PFAS (sum of 24 compounds) was associated with decreased foraging in crayfish (Coy et al., 2022).
Bluegill (*Lepomis macrochirus*)	North America, (Northern Michigan, USA)	Behavioral impacts Increased exposure to PFAS (sum of 24 compounds) was associated with changes in swim speed (Coy et al., 2022)
Striped bass *(Morone saxatilis)*	North America (North Carolina, USA)	Liver and immune system Elevated PFOS and Nafion byproduct 2 concentrations in serum were associated with altered liver biomarkers. Elevated PFOS was associated with alterations in the immune system in fish (Guillette et al., 2020).
Amphibians		
Black spotted frog (*Rana nigromaculata)*	Asia (Zhejiang province, China)	Liver Higher levels of PFAS, particularly PFOA, PFOS, and 6:2 Cl-PFESA, were correlated with liver biomarkers that indicated disrupted lipid metabolism (Lin et al., 2022).
Reptiles		
American alligator (*Alligator mississippiensis*)	North America (North Carolina, USA)	Immune system Elevated total PFAS (sum of 23) were associated with increased lysozyme activity in serum (an indicator of innate immunity), autoimmune-like traits and increased incidence of skin lesions (Guillette et al., 2022).
Freshwater turtle (*Emydura macquarii macquarii*)	Australia	Immune system Changes in the gut microbiome, gut inflammation and gut metabolic changes were observed in turtles exposed to elevated total PFAS concentrations (sum of 47) (Beale et al., 2022).
Hawksbill turtle (*Eretmochelys imbricata*)	North Pacific Ocean	Reproduction Higher concentrations of PFUnDA and PFTrDA in eggs were correlated with reduced emergence success of the hatchlings (Wood et al., 2021).
Birds		
Arctic black-legged kittiwakes (*Rissa tridactyla*)	Kongsfjorden, Svalbard	Cellular homeostasis and oxidative status Oxidative status markers were correlated with higher plasma concentrations of PFDoDA, PFTrDA, PFTeDA, PFUnDA, PFOS (Costantini et al., 2019).
Tree swallows (*Tachycineta bicolor*)	North America (Minnesota, Wisconsin, Michigan, USA)	Reproduction Decreased hatching success was associated with PFOS concentrations in eggs (Custer et al., 2014).
Arctic black-legged kittiwakes (*Rissa tridactyla*)	Kongsfjorden, Svalbard	Reproduction and endocrine system PFDoDA concentration was related to lower hatching success. PFNA was related to changes in body mass in male kittiwakes and PFTrDA and PFTeDA were related to decreased baseline corticosterone in both sexes (Costantini et al., 2019).
Mammals		
Bottlenose dolphin (*Tursiops truncatus*)	North America (South Carolina, USA)	Immune system PFOS exposure was associated with increased T cell proliferation and proinflammatory cytokine production, a marker of chronic immune activation (Soloff et al., 2017).
Hooded seals (*Cystophora cristata*)	East Greenland	Endocrine system Elevated total PFAS (sum of 8) were associated with changes in both concentrations and ratios of thyroid hormones in mothers and pups (Gronnestad et al., 2018).
Polar bear *(Ursus maritimus)*	East Greenland	Nervous system and endocrine system Higher concentrations of PFAS (both sum of 10 perfluoroalkyl carboxylic acids and sum of 4 perfluorosulfonic acids) in brain tissue were associated with increased levels of steroid hormones in brain tissue (Pedersen et al., 2016).
Bank voles (*Myodes glareolus)*	Europe (Norway)	Nervous system and endocrine system Higher PFAS concentrations (sum of 10 perfluoroalkyl carboxylic acids and 5 perfluorosulfonic acids) in the liver were associated with elevated dopamine levels in the brains and lower testosterone levels in muscle of wild bank voles (Gronnestad et al., 2021b).
Southern sea otter (*Enhydra lutris nereis)*	North America (California, USA)	Immune system Higher concentrations of PFOS and PFOA in the liver were associated with higher rates of infectious disease (Kannan et al., 2006).

with autoimmune disease in humans (Shanmugam et al., 2017). In marine mammals, increased levels of PFOA and PFOS in the liver were significantly associated with disease mortality in sea otters (Kannan et al., 2006), and environmental exposure to PFOS in bottlenose dolphins was associated with immune system dysregulation (Foguth et al., 2020; Soloff et al., 2017). Exposure to PFAS mixtures as well as exposure

Table 2
Studies where specific PFAS exposure was associated, either as individual compound or as a part of the summed PFAS, with the health outcomes in Table 1. Where total PFAS sum was reported, all PFAS within that summation are listed. PFAS names are provided in footnote.

PFAS*	Beale et al., 2022	Costantini et al., 2019	Custer et al., 2014	Coy et al., 2022	Gronnestad et al., 2018	Gronnestad et al., 2021b	Guillette et al., 2020	Guillette et al., 2022	Kannan et al., 2006	Lin et al., 2022	Pedersen et al., 2016	Soloff et al., 2017	Wood et al., 2021	Number of studies listed in Table 1
PFOS	*	*	*	*	*	*	*	*	*	*	*	*		12
PFOA	*			*	*			*	*	*	*			7
PFNA	*	*		*	*	*		*			*			7
PFTrDA	*	*		*	*	*					*		*	7
PFUnDA	*	*		*	*	*					*		*	7
PFDA	*			*	*	*		*			*			6
PFDoDA	*	*		*	*	*					*			6
PFHxS	*			*	*	*		*			*			6
PFBS	*			*		*		*			*			5
PFHpA	*			*		*		*			*			5
PFHxA	*			*		*		*			*			5
PFTeDA	*	*		*		*					*			5

* PFAS reported in 4 studies in Table 1: PFBA (perfluorobutanoic acid). PFAS reported in 3 studies in Table 1: PFPeA (perfluoropentanoic acid), PFDS (perfluorodecane sulfonic acid), 6:2 FTS (6:2 fluorotelomer sulfonate). PFAS reported in 2 studies in Table 1: PFPeS (perfluoropentane sulfonic acid), PFNS (perfluorononane sulfonic acid), PFHpS (perfluoroheptane sulfonic acid), Nafion byproduct 2, 7H-Perfluoro-4-methyl-3,6-dioxaoctanesulfonic acid), N-MeFOSE (N-ethyl perfluorooctane sulfonamido ethanol), N-MeFOSAA (N-Methylperfluorooctane sulfonamidoacetic acid), N-MeFOSA (*N*-methyl perfluorooctane sulfonamide), N-EtFOSE (N-ethyl perfluorooctane sulfonamido ethanol), N-EtFOSAA (N-ethylperfluorooctane sulfonamidoacetic acid), N-EtFOSA (N-ethyl perfluorooctane sulfonamide), HFPO-DA (GenX) (Hexafluoropropylene oxide-dimer acid), FOSA (perfluorooctanesulfonamide), 8:2 FTS (8:2 fluorotelomer sulfonate), 4:2 FTS (4:2 fluorotelomer sulfonate). PFAS reported in 1 study in Table 1: R-PSDA (Formerly Nafion Byproduct 4) (Perfluoro-4-(2-sulfoethoxy) pentanoic acid), PMPA (Perfluoro-2-methoxypropanoic acid), PFPrS (perfluoropropanesulfonic acid), PFPeDA (Perfluoropentadecanoic acid), PFOSA (Perfluorooctanesulfonamide), PFODA (Perfluorooctadecanoic acid), PFO5DoDA (TAFN4) (Perfluoro-3,5,7,9,11-pentaoxadodecanoic acid), PFO4DA (Perfluoro-3,5,7,9-butaoxadecanoic acid), PFO3OA (Perfluoro-3,5,7-trioxaoctanoic acid), PFO2HxA (Perfluoro(3,5-dioxahexanoic) acid), PFHxDA (Perfluorohexadecanoic Acid), PFECHS (Perfluoroethylcyclohexane Sulfonate). PFDoDS (perfluorododecane sulfonic acid), PEPA (perfluoro-2-ethoxypropanoic acid).

NVHOS (1,1,2,2-tetrafluoro-2-(1,2,2,2-tetrafluoro-ethoxy)ethane sulfonic acid), Nafion Byproduct 1 (Perfluoro-3,6-dioxa-4-methyl-7-octenesulfonic acid), Hydro-EVE Acid (2,2,3,3-Tetrafluoro-3-{[1,1,1,2,3,3-hexafluoro-3-(1,2,2,2-tetrafluoroethoxy)propan-2-yl]oxy}propanoic acid), FHxSA (perfluorohexane sulfonamide), FBSA (perfluorobutane sulfonamide).

FASA (perfluoroalkane sulfonamide), ADONA (trade name for 4,8-dioxa-3H-perfluorononanoate), 9Cl-F53B (9-Chloroeicosafluoro-3-oxaundecane-1-sulfonic acid), 8Cl-PFOS (Sodium 8-chloroperfluoro-1-octanesulfonate), 8:2 FTCA (8:2 fluorotelomer carboxylic acid), 8:2 DiPAP (8:2 fluorotelomer phosphate diester), 6:6 PFPiA (6:6 perfluorophosphinic acid), 6:2 FTCA (6:2 fluorotelomer carboxylic acid), 6:2 Cl-PFESA (6:2 chlorinated polyfluorinated ether sulfonate), 5:3 FTCA (5:3 fluorotelomer carboxylic acid), 11Cl-F53B (11-Chloroeicosafluoro-3-oxaundecane-1-sulfonic acid), 10:2 FTS (10:2 fluorotelomer sulfonate), 10:2 FTCA (10:2 fluorotelomer carboxylic acid).

to PFHxS increased parasite infections in different species of frogs (Brown et al., 2021; Lech et al., 2022). The inability to heal from injury or increased susceptibility to disease may impact both individual survival and overall population of a species.

4.2. Nervous system

Several PFAS have been detected in the brains of laboratory animals, humans, as well as environmentally exposed wild animal populations (Foguth et al., 2020; Shang et al., 2023). While the extent of known PFAS impacts on the human nervous system are somewhat limited, evidence from laboratory animal studies, especially for exposures during development and early life, suggest that PFAS, particularly PFOA and PFOS, can harm the nervous system, causing changes in motor activity and function, and impaired learning and memory (Starnes et al., 2022). In wild bank voles caught in a PFAS-contaminated ski area in Norway, higher PFAS concentrations were associated with higher dopamine levels in the brain compared to voles caught from a reference site (Gronnestad et al., 2021a). The study authors exposed mice in the lab to a mixture of PFAS based on the concentrations at the contaminated site and found disruption of dopamine homeostasis (Gronnestad et al., 2021b). In another study, environmental exposure to PFAS in crustaceans (crayfish) and fish (bluegills) was associated with altered behavioral outcomes, including decreased foraging for crayfish, and altered swim speeds in fish (Coy et al., 2022).

4.3. Endocrine system

Studies suggest that PFAS disrupt levels of thyroid hormones in the body in both humans (Kim et al., 2018) and other species with thyroid-dependent development, like amphibians (Flynn et al., 2022). Additionally, PFAS have been implicated in altering circulating levels of reproductive hormones at different life stages in humans and laboratory animals and endocrine-mediated diseases, effects that may be related to the ability of PFAS to bind directly with sex steroid nuclear hormone receptors (Mokra, 2021). In wild bank voles from a contaminated ski area, some associations were observed between PFAS concentrations and lower testosterone in muscle tissue (Gronnestad et al., 2021b). In a study of hooded seal mothers exposed to complex contaminant mixtures, PFAS contributed to hormone disruption and were the most important predictor of thyroid hormone levels (Gronnestad et al., 2018). Levels of perfluoroalkyl carboxylic acids (PFCAs) and perfluorosulfonic acids (PFSAs) in brain tissue were associated with increased levels of steroid hormones in the brains of polar bears studied in Greenland (Pedersen et al., 2016). Prolonged periods of exposure to elevated levels of steroid hormones could adversely affect reproduction rates and fetal development.

4.4. Reproduction and developmental

In human studies, PFAS impacts on reproduction and development include preeclampsia, low birth weight, altered breastfeeding duration, reduced fertility among others (Blake and Fenton, 2020; Rickard et al., 2022). For wildlife species, reproductive success is critical, especially for a species population that is threatened or endangered. PFAS exposure, happening at the same time with other anthropogenic pressures, may be contributing to decreased reproduction success in wildlife. In hawksbill sea turtles, a critically endangered species, emergent success was negatively correlated with levels of perfluoroundecanoic acid (PFUnDA) and perfluorotridecanoic acid (PFTrDA) in eggs (Wood et al., 2021). Maternal transfer of PFAS to eggs has been observed in both freshwater and sea turtles (Beale et al., 2022; Wood et al., 2021). A negative association with PFOS in tree swallow eggs and hatching success was observed along with total nest failure in nests with eggs that had PFOS concentrations at or above 150 ng/g (Custer et al., 2014). In contrast, a subsequent study by the same team observed no effects of PFAS exposure on reproduction in tree swallows at another field study site (Custer et al., 2019). It is important to acknowledge that not all published studies detect and report a consistent association of individual PFAS exposures and distinct, measurable adverse outcomes. This diversity of findings could be due to a variety of factors in the population at the site of study, such as site-specific mixtures of PFAS, or the presence of co-occurring contaminants causing or contributing to overall population health effects (Custer, 2021).

4.5. Impacts on metabolism, including liver function and oxidative stress

Blood serum biomarkers can be useful tools indicative of immune and reproductive functions as well as the overall health of populations in wild species where other types of health parameters may be difficult to study. Elevated levels of PFOS were associated with increased activity of blood serum biomarkers indicative of adverse impacts to the liver and immune system in striped bass from the Cape Fear River (North Carolina, United States) (Guillette et al., 2020). In Arctic black-legged kittiwakes, analysis of blood serum biomarkers indicated higher protein oxidative damage and lower plasma antioxidants in birds with higher levels of longer chain PFAS (Costantini et al., 2019). Additionally, freshwater turtle eggs exposed to PFAS contained significantly elevated levels of purine metabolites which may alter eggshell structure and egg physiology and were depleted of lipids needed for growth and development. Overall, higher PFAS exposure may affect egg development and the health of the turtle hatchlings. Analysis of the eggshells showed lower levels of essential calcium and higher levels of lipids and metabolites indicative of weakened physical structure (Beale et al., 2022). In field studies in black spotted frogs, associations were found with PFOS, PFOA and 6:2 Cl-PFESA and lipid metabolism-related indices suggesting that PFAS exposure may disrupt lipid metabolism in frogs collected from a contaminated industrial site (Lin et al., 2022). While changes in lipid metabolism, endocrine system, and immune defense may not be immediately lethal to an individual animal's health, these changes in biomarkers indicate significant alterations in hepatic, endocrine and immune systems, with a potential to contribute to overall population decline (Shi et al., 2023).

5. Shared health concept

There is strong concordance between adverse health effects observed in epidemiological research in humans and in wildlife studies in association with PFAS exposure (Fig. 1), supporting an argument that humans can be regarded as a unique sentinel species for the identification of the potential health harms from PFAS to other species. Distinctly, the amount of data available from epidemiological studies as well as humans' large population, geographic spread, and widespread health monitoring make *Homo sapiens* potentially a true global sentinel for the impacts of toxic chemicals on biological organisms. Future research will provide more information on the species differences in sensitivity to individual PFAS and to the overall load of combined exposure to PFAS as a class. Meanwhile, just as wild or domesticated animals have been both regarded and used as early warning sentinels for the impacts of toxic chemicals, humans can also serve in this role, especially for PFAS health harms to other animals, since similar levels of PFAS have been measured, for example, in seals, dolphins, dogs, horses, alligators and humans.

We posit that the mobility of humans does not preclude their potential as a sentinel species. Of note, epidemiological studies that can provide valuable information for cross-species extrapolation about PFAS effects between species were conducted for populations that have stayed in their local area for a long time, such as the C8 health study (Frisbee et al., 2009), the Faroes island study of PFAS effects on the vaccine response (Grandjean et al., 2012), and worker health studies (Olsen et al., 2003). Further, in general population exposures, published data suggest that a significant portion of overall PFAS exposure is due to local

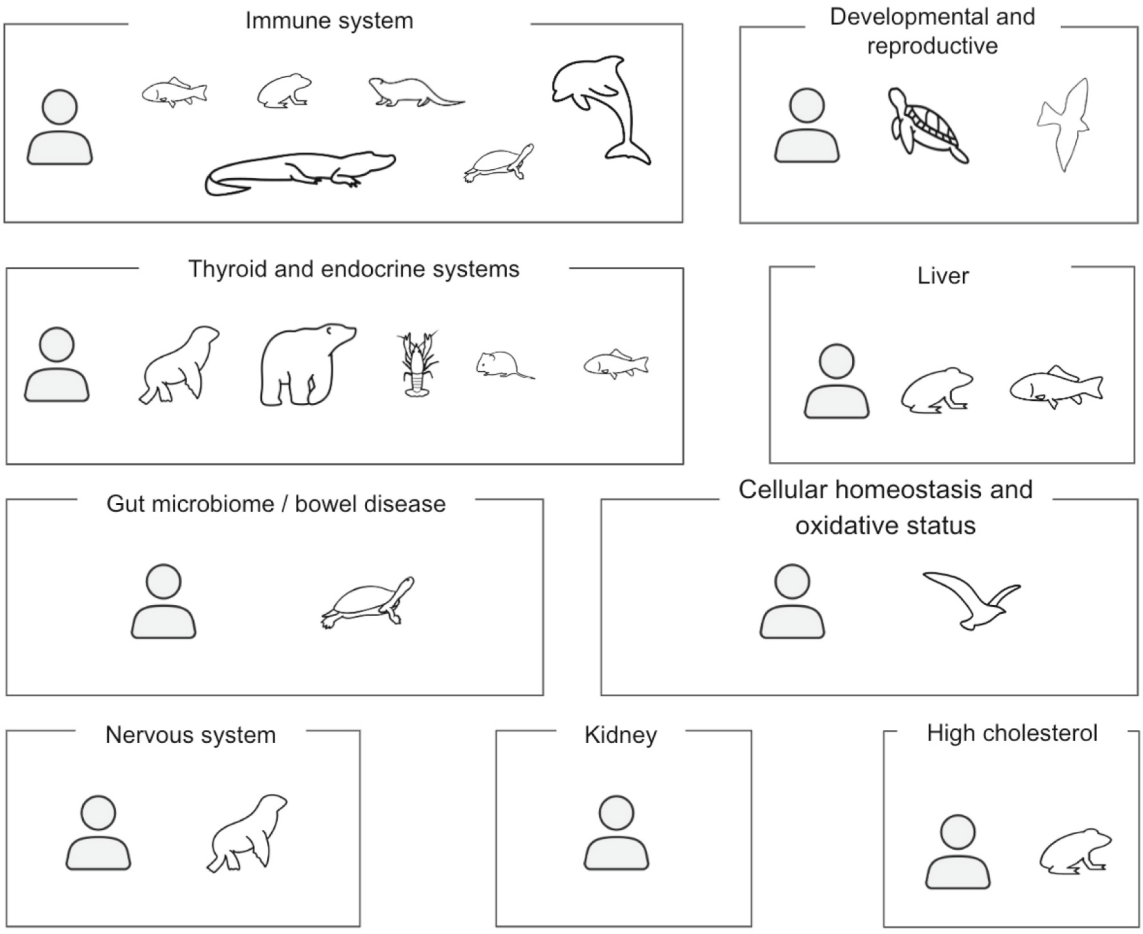


Fig. 1. Adverse health effects on organ systems and biomarkers observed in association with PFAS exposure in epidemiological research and wildlife studies. Individual wildlife study references are included in Table 1. Immune system: *Homo sapiens,* Striped bass *(Morone saxatilis),* American alligator (*Alligator mississippiensis*), Black spotted frog (*Rana nigromaculata),* Southern sea otter (*Enhydra lutris nereis),* Freshwater turtle (*Emydura macquarii macquarii*), Bottlenose dolphin (*Tursiops truncatus*). Developmental and reproductive system: *Homo sapiens,* Hawksbill turtle (*Eretmochelys imbricata*), Tree swallows (*Tachycineta bicolor*). Nervous and endocrine systems: *Homo sapiens*, Crayfish (*Faxonius immunis), Bluegill (Lepomis macrochirus),* Hooded seals (*Cystophora cristata*), Polar bear *(Ursus maritimus),* Bank voles (*Myodes glareolus)*. Liver: *Homo sapiens*, Black spotted frog (*Rana nigromaculata),* Striped bass *(Morone saxatilis)*. Gut microbiome/bowel disease: *Homo sapiens*, Freshwater turtle (*Emydura macquarii macquarii*). Cellular homeostasis and oxidative stress: *Homo sapiens*, Arctic black-legged kittiwakes (*Rissa tridactyla*). Thyroid: *Homo sapiens*, Hooded seals (*Cystophora cristata*). Kidney: *Homo sapiens*. High cholesterol: *Homo sapiens*, Black spotted frog (*Rana nigromaculata)*.

exposures, through dietary patterns and locally contaminated drinking water supplies (Sunderland et al., 2019). Migratory species of birds, large mammals and reptiles such as aquatic marine turtles can carry out annual journeys of distance, difficulty and diversity of geographic areas that may be well out of range for most humans on the planet. Finally, and irrespectively of whether a "sentinel" or "comparison" populations are localized or mobile, the atmospheric transport of PFAS and global contamination have put all species and populations at risk of PFAS exposures.

Conceptualizing humans as a sentinel species for other animals is supported by the One Environmental Health approach that views human health, wildlife health and ecosystem health as one (Perez et al., 2018). Given the extent of global PFAS contamination (Cousins et al., 2022) and the ubiquitous PFAS exposure of nearly all biota on the planet, this interconnected approach is needed to understand and confront the universal threat to human health and to global biodiversity from PFAS exposure. Further, a translation of the concept "humans as sentinels" to environmental risk assessment points to the value of using human data where available to aid the development of policy measures for protecting wildlife populations.

Examples of the cross-species extrapolation approach comparing human and wildlife health data already exist in the literature. Some of the highest levels of PFAS reported globally in marine mammals have been measured in bottlenose dolphins from the Charleston, South Carolina (U.S.) urban coast and have been compared to levels in humans that have been exposed occupationally (Fair et al., 2010; Fair et al., 2012). Levels of PFOS ranged from 49 ng/g in plasma from dolphins in Bermuda to 1171 ng/g in dolphins near Charleston, South Carolina from a 2005 study (Houde et al., 2005). Mean PFOS levels in serum in workers at a 3 M facility were reported to be 1.32 ppm, equivalent to 1320 ng/g (Olsen et al., 2003). Comparing human exposure with exposure for bottlenose dolphins may be useful because of dolphin's longevity, top-predator status and high site fidelity in an urban area (Fair et al., 2019). Published studies linked immune system dysregulation in dolphins with exposure to PFOS (De Guise et al., 2019; Fair et al., 2014; Soloff et al., 2017). Nevertheless, a substantial gap in data availability is evident when comparing the number of health studies conducted on wild animal populations to studies in humans. Given the availability of hundreds of studies in humans, epidemiological data could serve as the basis for conservation risk assessments and the development of conservation policies for other mammals or species where such data are lacking.

Available data suggest that the adverse health effects observed in humans due to PFAS exposure are relevant and applicable to all animal

species generally, and across the globe. The health impacts documented in wildlife, as detailed in Table 1, are similar to adverse health endpoints with likely causal associations to PFAS exposure in humans (Agency for Toxic Substances and Disease Registry (ATSDR), 2021) (EPA 2023B). Further, the graphical representations in Fig. 1 and Fig. 2 highlight both reported and anticipated health effects due to PFAS exposure in wildlife and correlate the measured PFAS levels in the blood of wildlife from the studies in Table 1 with the PFAS levels in humans that have documented health impacts.

While the concordance in types of health impacts observed in humans and other species point to the value of cross-species extrapolation for this analysis, the uncertainties associated with dose-response curves for adverse effects in PFAS in different species remain a scientific challenge. In Fig. 2, the circulating concentrations of PFAS do not represent the probability of the frequency of occurrence of specific health effects. Rather, Fig. 2 documents the fact of global PFAS contamination in wildlife at levels that in many circumstances are vastly greater than concentrations shown in epidemiological studies to impact human health parameters. In parallel, Table 2 documents the co-occurrence of different PFAS compounds measured in wildlife health studies, pointing to the relevance and necessity of both assessing and mitigating against the adverse effects of PFAS mixtures.

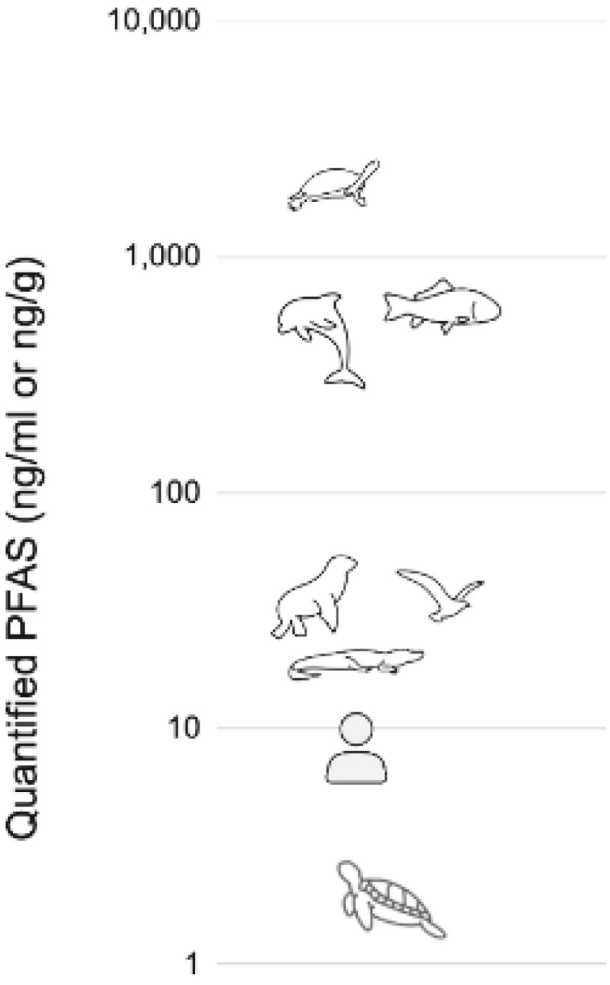


Fig. 2. Concentration of PFAS in animals in studies documenting health impacts from exposures, as detailed in Table 1, compared to the median levels in the U.S. population for 2017–2018. PFAS concentrations are cited according to how they were reported in original references. From lowest to highest: Hawksbill turtle (*Eretmochelys imbricata*, 1.74 ng/g median PFAS in plasma with a median of 31 ng/g in eggs), human (*Homo sapiens*, 16.7 ng/ml median PFOS in serum), American alligator (*Alligator mississippiensis,* 18.6 ng/ml median total PFAS in serum), Arctic black-legged kittiwakes (*Rissa tridactyla*, 39.9 ng/g mean PFAS in plasma), Hooded seals (*Cystophora cristata*, 40 ng/g median PFOS in plasma), Striped bass (*Morone saxatilis*, 551 ng/ml median total PFAS in serum), Bottlenose dolphin (*Tursiops truncatus*, 571 ng/g median PFOS in plasma), Freshwater turtle (*Emydura macquarii macquarii*, 1992 ng/ml average total PFAS in plasma). Frequency of associations of different PFAS compounds with reported health effects in different studies is shown in Table 2, with most associations reported for PFOS.

For practical reasons, the number of studies linking specific health effects in wildlife with PFAS exposure remains limited. Nevertheless, the published literature clearly demonstrates that exposure to PFAS in the environment and elevated levels of PFAS in animals' bodies are associated with changes in key health parameters, such as hormonal levels, immune function, reproduction, and growth of the young, which have also been observed in controlled toxicological studies in laboratory animals as well as human epidemiological cohorts. Changes in these health indicators have significant relevance to individual animal's reproduction and survival and the overall health and stability of wildlife populations and ecosystems.

Available data do not allow a definitive assessment of whether humans are the most susceptible species to the toxic effects of PFAS exposure, or whether other species, including species shown in Fig. 1 and Fig. 2 have a more sensitive population-level dose-response relationship for PFAS. For a read-across or cross-species extrapolation approach, uncertainties remain in the context of exposure of life-histories across species that may vary drastically. Further, exposures may differ, especially for aquatic species that may be exposed simultaneously to numerous contaminants during developmental phases and throughout their life. Yet, given the dramatic and tragic rate of the disappearance of endangered species and shrinking of wildlife populations globally, all measures should be taken to consider pollution reduction as part of the essential framework for protecting threatened and endangered species, and a cross-species extrapolation approach holds, in our view, valuable potential for both research and policy.

It is important to acknowledge the difficulty of conducting wildlife health studies in general, especially in endangered species, as well as the appropriate ethical barriers and legal restrictions on conducting lethal sampling of wildlife. These practical considerations preclude a comprehensive assessment of the full scope of PFAS impacts on wild animals' health. Additionally, given the reality of global PFAS contamination of the environment and wildlife, identifying a true, uncontaminated reference population generally proves impossible, since even samples of animals collected from the geographic sites without known sources of industrial discharges still contain PFAS (Beale et al., 2022). All these factors make it especially urgent and timely to use a read-across or cross-species extrapolation approach so as to draw on all available data to assess potential damage to the ecosystems and specific animals due to PFAS contamination, and to develop and implement measures to prevent such damages.

Scholars and environmental advocates across the world are raising the alarm on the critical state of anthropogenic pollution and its effects on human health globally (Cousins et al., 2022; Groh et al., 2022). As emphasized in previous reports, the planetary boundary for chemical pollution has likely been crossed (Persson et al., 2022). Yet, with the human population continuing to grow, we can anticipate the continued increase of chemical manufacturing and with it, the potential for growing pollution levels in the environment. Given this reality, both regulation and minimization of environmental discharges of industrial chemicals are essential measures for protecting human and wildlife health as well as for wildlife and endangered species conservation on our planet. Protection of a sentinel species from a particular toxic chemical impact would begin the process to protect all species on the planet. In this process, it is especially important to consider the cyclical nature of

PFAS pollution (Stoiber et al., 2020), so as to ensure that remediation policies and practices do not end up shuffling the PFAS pollution load from one site to another. Systematic approaches to tackling the PFAS contamination crisis should be the focus to protect both humans and wildlife.

Summarizing our argument, we posit that the use of human data as a sentinel approach can serve as a foundation for ecological conservation policies that are urgently needed now. While PFAS exposures of humans may be conceptually similar to PFAS in large mammalian carnivorous species such as dolphins and polar bears, the similar health effects in different taxa are documented (Table 1, Fig. 1). Building on this concordance of recorded health effects, future research of the mechanistic basis for toxicological outcomes due to PFAS exposure can provide a framework for extrapolation across species as well as potential explanations of variability (Celander et al., 2011; LaLone et al., 2021). Finally, while human epidemiological data might not provide a full set of information relevant to the environmental impacts of PFAS contamination, this corpus of data is the most extensive dataset on a single species that is currently available and it can shed light on a wide variety of toxic effects elicited by PFAS.

6. Research needs

Contamination of living organisms around the globe with anthropogenic environmental pollutants, such as persistent organic pollutants or POPs, has been known for decades. Top predators in terrestrial and aquatic food webs often carry the heaviest body burden of such pollutants, even decades after regulatory action restricting the manufacturing of these persistent substances (Kean et al., 2021; Travis et al., 2020). In co-occurrence with chlorinated and brominated contaminants, PFAS have emerged as ubiquitous pollutants with planetary-wide impact (Cousins et al., 2022). Most species, including species included in the International Union for Conservation of Nature Red List, are exposed to the PFAS family of chemicals that, as human epidemiological evidence indicates, may compromise basic life functions critical to survival of individuals and populations: the ability to reproduce and to resist disease and injury.

Animals in the wild are exposed to PFAS through the water, air, soils and sediments, and the food chain. Those species also face habitat loss, over-harvesting and other anthropogenic threats to their survival. PFAS exposures are potentially most concerning for endangered and threatened species populations that are simultaneously exposed to a mix of toxic pollutants (Routti et al., 2019). With widespread wildlife exposure to mixtures of PFAS and other persistent organic pollutants, additional research on the health impacts of these chemical mixtures is urgently needed. The chemical concentrations of individual pollutants or pollutant mixtures varies in different geographic sites for both people and different wildlife species. It remains to be investigated what might be potential limits on the concordance of adverse health effects of exposure to similar pollutants in humans and in wildlife species. Further studies on characterizing the exposure, dose-response relationship, and toxicological outcomes in various species remain necessary. Yet, such data limitations should not preclude prompt, coherent and comprehensive action on protecting the wildlife species populations that still remain on this planet. A consideration of humans as sentinel species is a promising approach that would be refined with the generation and dissemination of new research studies.

Additional research needs to apply the question of chemical pollutant impacts on population dynamics and trends in population health, which is an under-investigated topic to date. For example, the documented detrimental effects of PFAS exposure on the immune system can make species susceptible to other stressors. Additionally, large aquatic animals that suffer injuries from plastic pollution could potentially face reduced resistance to infections due to their body burden of PFAS. Together, the cumulative pressures of chemical exposures, habitat loss and climate change may act jointly, exacerbating the loss of biodiversity. As habitat loss increases, species are increasingly living closer and closer to human populations (Simkin et al., 2022). The density of the human population has been observed to correlate with the levels of PFAS measured in wildlife living in close proximity (Guillette et al., 2022; Wood et al., 2021), indicating both elevated PFAS exposure and degraded habitat quality. As documented by numerous studies, the current loss of species and the rate of species loss is unprecedented (IPBES, 2019; Isbell et al., 2022) and understanding the role of chemical exposures in this global extinction will be critical for attempting to conserve biodiversity (Mueller et al., 2023).

We hope that future research will also focus on developing models to characterize specific adverse health effects such as reproductive harm in wildlife, thus facilitating environmental health assessments while avoiding any lethal sampling of wildlife populations (Finn et al., 2023). We look forward to the greater acceptance of the importance of biomarker effects in ecotoxicology studies and the application of the toxicological read-across and cross-species extrapolation concepts for developing effective policies for species conservation. Positively, recent emphasis on reducing animal testing and efforts to focus on cross-species extrapolation and new approach methodologies to leverage existing toxicity data (LaLone et al., 2021) may help to protect all species. These efforts align with the One Health approach and point out the connectedness of all species that share the planetary environment. Methods that focus on examining molecular, cellular and organ-level changes can be more predictive compared to whole-organism endpoints, which reduces animal testing and broadens the possibilities for extrapolating toxicological data across a variety of species.

7. Recommendations

Analysis presented in this article supports the call for an integrated government and society approach to reduce threats to biodiversity due to PFAS pollution and for urgent actions to advance species conservation that should be initiated immediately:

1. Biodiversity conservation policymakers need to consider chemical pollution reduction as part of the essential framework for protecting threatened and endangered species.
2. In situations where a direct wildlife health study may be impractical or not appropriate due to endangered species status, a read-across and cross-species extrapolation approach incorporating all available data sets can aid policy measures to protect wildlife populations anthropogenic contaminants.
3. Where available, human epidemiological studies should be included as one of the sources of reference data for developing policies for species conservation. The epidemiological dataset for PFAS presents a compelling case study for such an approach.
4. International measures controlling PFAS pollution and preventing future discharges of PFAS into the environment are essential for protecting both human health and global biodiversity.
5. Protection of wildlife populations from PFAS pollution has the potential to contribute to ecosystem stability and sustainable development goals connected with ecosystem functions and services.
6. Industrial chemical registrations and approvals should be developed and implemented to prevent adverse impacts of PFAS on critical habitats, wildlife populations, and endangered species, by ending ongoing releases to the environment and capturing PFAS in discharges from manufacturing plants, wastewater treatment plants, landfills, and other types of commercial and residential PFAS discharges.
7. More research is needed to develop technologies for assessing and remediating PFAS contamination in the environment and for the cleanup of contaminated sites.

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Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

No data was used for the research described in the article.

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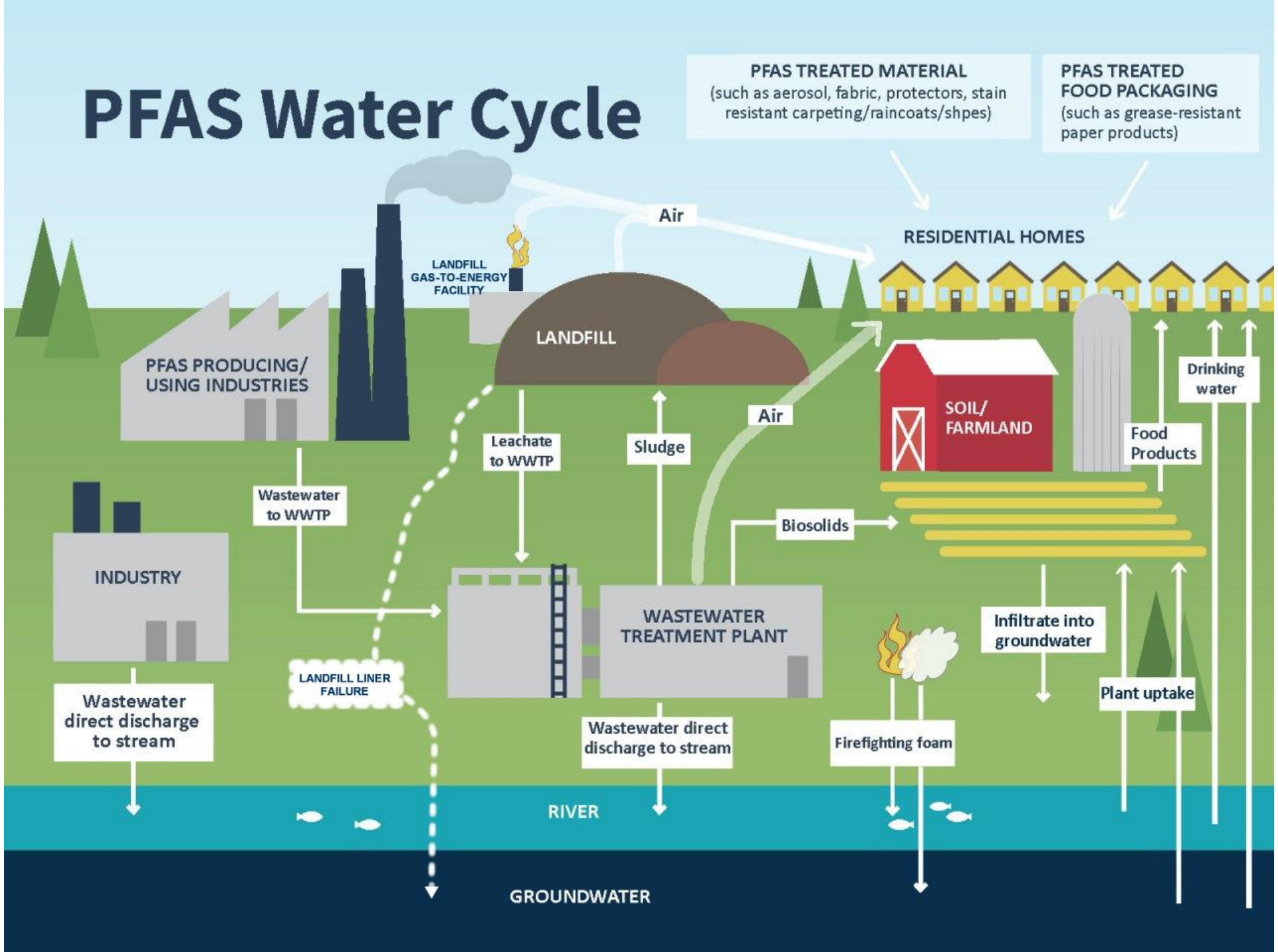
Planning commission members,

Thank you for volunteering your time and energy on this commission. Making a decision about the application to expand Coffin Butte landfill is a time-consuming process and it is clear that you are working to understand the issues and to make the right decision for the people of Benton County. I appreciate your care and commitment.

As I sat through the hearings, it became clear to me that the burden of dealing with the PFAS in Coffin Butte Landfill was not fully appreciated. I have a Master's degree in pharmacology/toxicology, taught college biology and genetics for 20 years and have access to the scientific literature through my college's library. It seemed important that someone provide some background about why the PFAS in both landfill gas and leachate are such a vexing problem. While it is common practice to dispose of landfill leachate by mixing it with municipal wastewater and processing them together, it is not an appropriate way to dispose of leachate. Municipal wastewater treatment does not mitigate PFAS contamination and, in fact, it can create more PFAS that are then released into the environment. One verbal testimony referred to the strength of the C-F bonds in PFAS molecules. These strong bonds are hard to break making PFAS hard to destroy. This results in molecules that stay in the environment for a long time – hence PFAS were given the moniker "forever chemicals".

The only solution to the issue is to make less leachate by reducing the quantity of material being landfilled, by reducing the amount of organic matter that is landfilled, by disposing of municipal solid waste in landfills in dry environments like Columbia Ridge Landfill in Gillam County Oregon, and to appropriately handle landfill leachate separately from municipal wastewater. The operators of Coffin Butte Landfill must treat leachate on site, ridding it of PFAS prior to release into the environment.

Expanding the current landfill worsens the PFAS and leachate problems. This comes at environmental and economic costs and while we cannot quantify them, it is extremely unlikely that any current financial gains will pay for these future financial costs.



SOURCE

This figure was modified from the original EPA figure to include the movement of PFAS to the air via landfill gas and to groundwater via landfill leachate leakage.

PFAS present human health risks

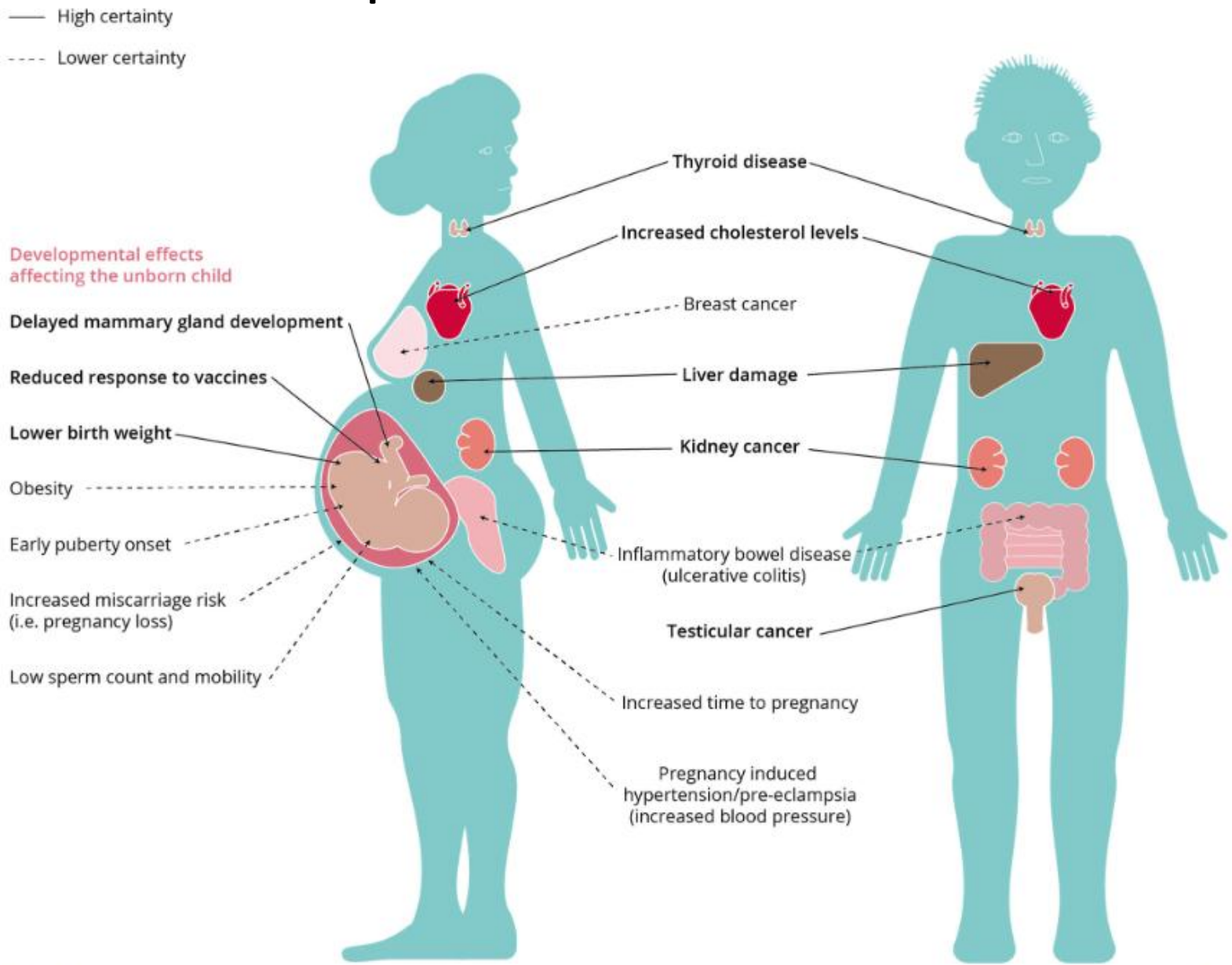
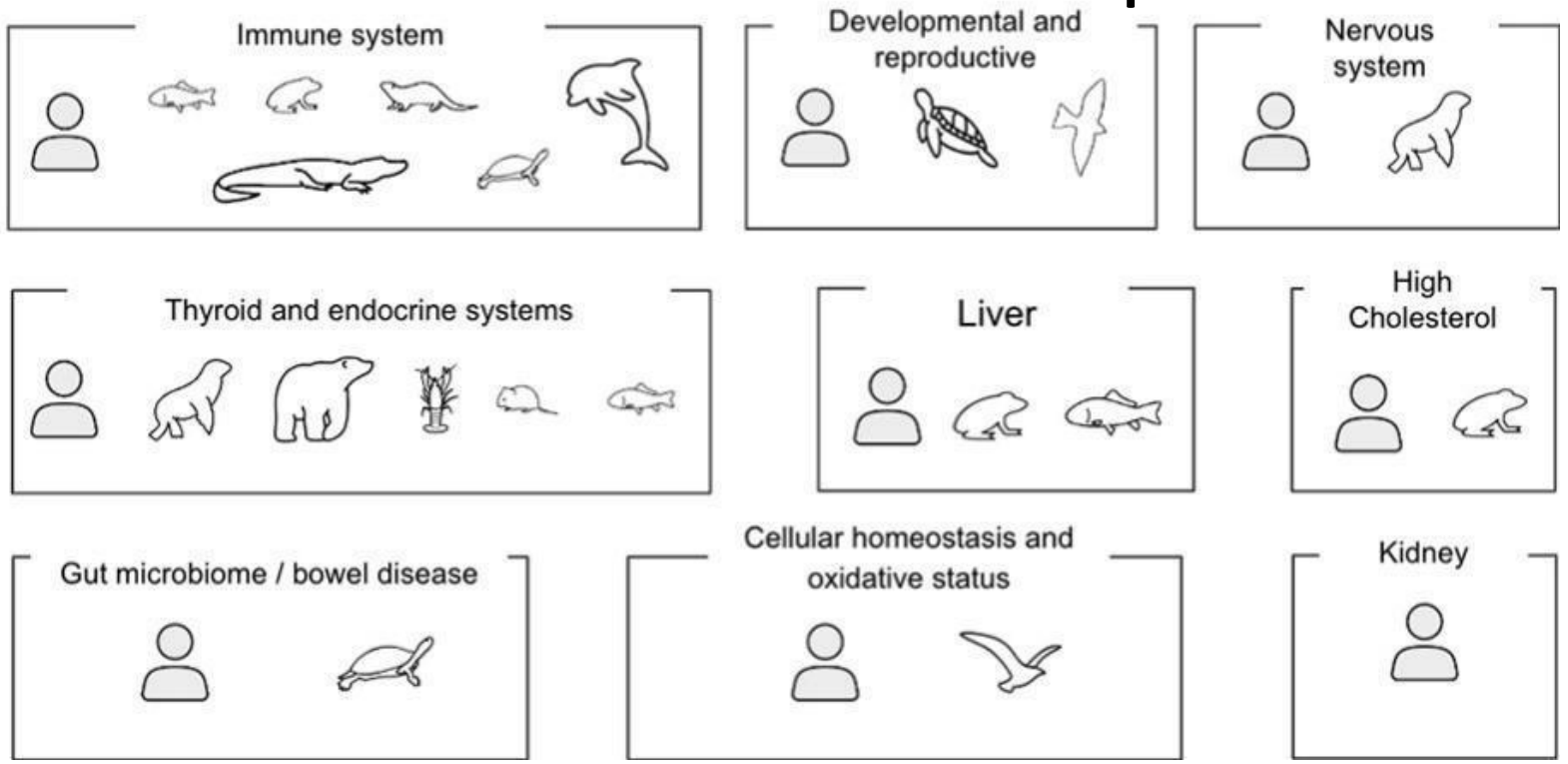


FIGURE 1: Effects of per- and polyfluoroalkyl substances on human health. Used with permission from European Environment Agency (2019). Original sources for this figure: National Toxicology Program (2016), C8 Science Panel (2012), IARC Working Group on the Evaluation of Carcinogenic Risks to Humans (2017), Barry et al. (2013), Fenton et al. (2009), and White et al. (2011b).

From Fenton et al. 2021

Wildlife suffer the same effects when exposed to PFAS



Andrews et al. 2023 highlights:

- PFAS are detected in wildlife globally.
- There is concordance between findings on PFAS health effects in wildlife and humans.
- Humans can be regarded as a sentinel species for adverse health impacts of PFAS.

Plants also bioaccumulate PFAS and exhibit metabolic and phytotoxic responses

Costello and Lee (2024) found:

- “Land application of PFAS-containing media results in plant uptake of short-chain PFAS into plant components that are consumed by humans, utilized as fodder, or land-applied. Long-chain compounds typically sorb to or are taken up into plant roots and largely remain in the roots while short-chain compounds have a higher propensity to translocate into plant shoots.”

- “Also in lettuce, PFOS and PFOA altered amino acid metabolites, dipeptides, fatty acids, lipids, phytol, purine nucleosides, phenolic antioxidants, and flavonoids, as a result of stress responses to DNA injury, photosynthesis inhibition, and impact to energy metabolism.”

- That plant enzymes were found to convert precursors into more bioavailable PFAS.

Summary:
Plants bioaccumulate PFAS from soil and those PFAS have been shown to disrupt cellular processes. In addition, plants can convert PFAS into forms that are more disruptive to biological processes.

Landfill leachate composition is complex and contains:

- Water from precipitation as well as deposited waste and the decomposition of that waste.
  - Coffin Butte Landfill receives a great deal of rainfall adding to this component of leachate.*
- Organic materials from food and landscape waste, animal waste, paper, etc.
  - Coffin Butte Landfill has been documented to receive animal carcasses like cows from Tillamook county, increasing its organic material burden.
- Inorganics like ammonia, sulfates, heavy metals, etc.**
- Xenobiotics like pesticides, industrial chemicals, pharmaceuticals (Wang and Qiao 2024) and PFAS (Neill and Megson 2024).**

*As water percolates through the landfill matrix, contaminants are leached out of the landfill contents. Limiting the amount of water passing through the landfill contents limits the contaminants released by the contents. Landfills should be in dry climates.

**Items like batteries, televisions and other electronics that have been disposed of at Coffin Butte Landfill contribute to the levels of these contaminants.

It is important that I point out that processes that occur in landfills convert precursors into PFAS, so Coffin Butte Landfill also likely generate PFAS despite the assertions of the applicant (Tolaymat et al. 2023).

Landfills in wet climates produce more PFAS-laden leachate

- Lang et al. (2017) found that even though they contained only 47% of the total landfill waste, landfills in wet regions of the US (>30” precipitation/year) generated 79% of the leachate formed.

- In the US, <1% of PFAS released into leachate at landfills does so in arid climates (Lang et al. 2017).

- Coffin Butte Landfill is situated terribly due to the large amount of rainfall it receives. No landfill should be located in this type of wet location.

For some leachate components, even very low concentrations are of high importance.

Table 1
Composition of municipal solid waste leachate [32,33].

Components	Range	Unit
pH	7.29–8.61	mg/L
COD	580–1821	mg/L
BOD	76–701	mg/L
Total suspended solids	191–740	mg/L
Volatile suspended solids	72–329	mg/L
Total nitrogen	95–401	mg/L
Organic nitrogen	29–46	mg/L
Ammonia nitrogen	66–364	mg/L
Total phosphorus	1.4–15.7	mg/L
Magnesium	126–419	mg/L
Calcium	192–430	mg/L
Chlorides	490–1190	mg/L
Volatile dissolved solids	1248–2175	mg/L
Total dissolved solids	2969–6384	mg/L
Cadmium	0.022–0.13	mg/L
Zinc	0.22–0.435	mg/L
Chromium	0.05–0.08	mg/L
Mercury	0.005–0.017	mg/L
Ethylbenzene	2.4–89	µg/L
Xylene	9.3–225	µg/L
Toluene	0.3–182	µg/L

"Landfill leachate contains a variety of micropollutants, including synthetic compounds like- phthalic acid esters (PAEs), polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), pharmaceuticals and personal care products (PPCPs), pesticides, microplastics (MPs), per- and **polyfluoroalkyl substances (PFAS),** cleaning detergents, microplastic, steroids, etc. [22,54]. *Despite being in very lower concentrations (µg L− 1 to ng L− 1 or even lower), micropollutants still significantly adversely affect ecosystems, the economy, and humans [55].*" (Kumar et al. 2023)

During the April 29 hearing, the applicant attempted to downplay the environmental and health risks associated with PFAS by stating that they were measured in parts per trillion (ng/L), describing this as a "trace" amount. The sheer volume of leachate and, as indicated by Kumar et al.(2023), the low concentrations required to cause harm, mean that their concentration in parts per trillion are of concern. The applicant also appeared to suggest that the fact that it takes work and expertise to measure PFAS was an excuse for not doing so. That is part of the cost of operating a business that concentrates PFAS – they've trucked them in from around Oregon, they have the responsibility to prevent their spread in the environment. Monitoring is required.

Leachate treatment must use processes to remove PFAS. If not, they can inadvertently <u>increase</u> PFAS levels.

- Because the processes used have generally not been appropriate, treatment of leachate at landfills to reduce harmful components and improve water quality has been shown to ***increase*** PFAS in the UK (see figures from Neill and Megson 2024 below), in Spain (Fuertes et al. 2017), and in the US (Chen et al. 2022).

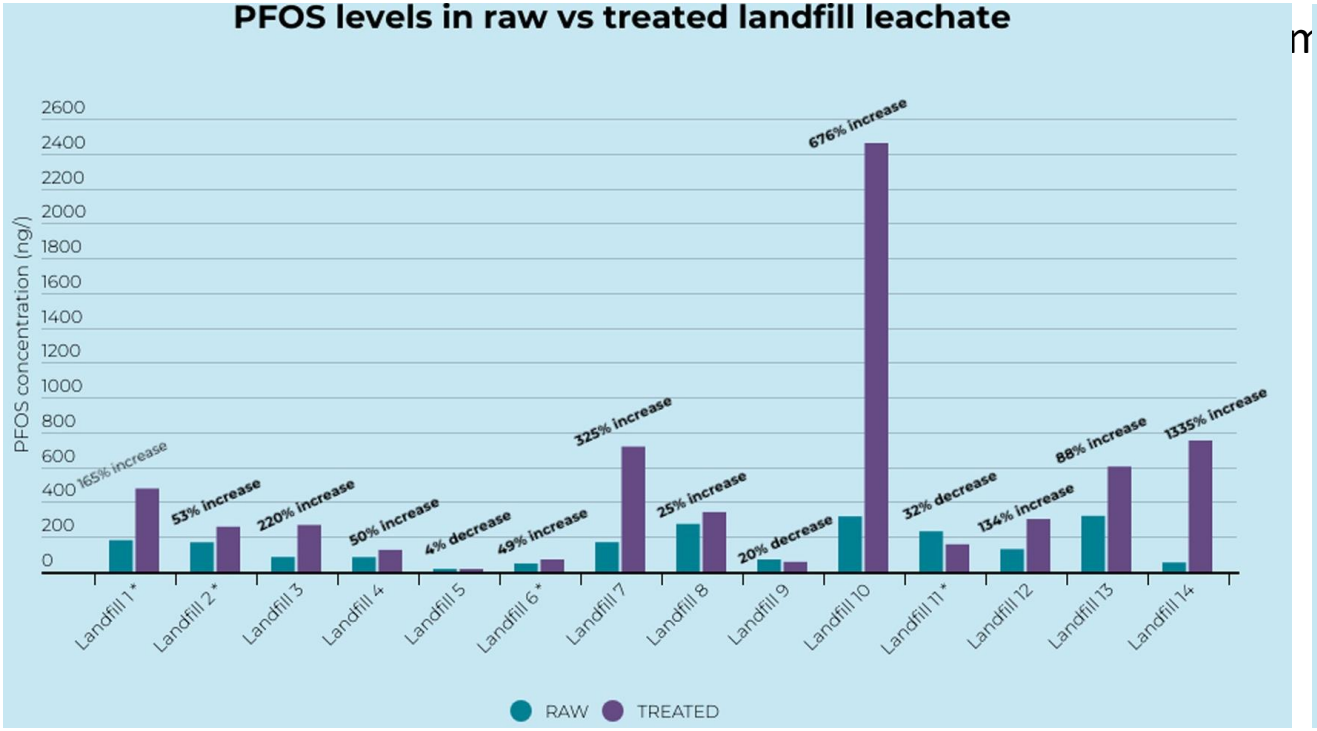


Figure 1. PFOS concentrations in raw and treated landfill leachate. *Represents average value for landfills with multiple treated or raw samples.

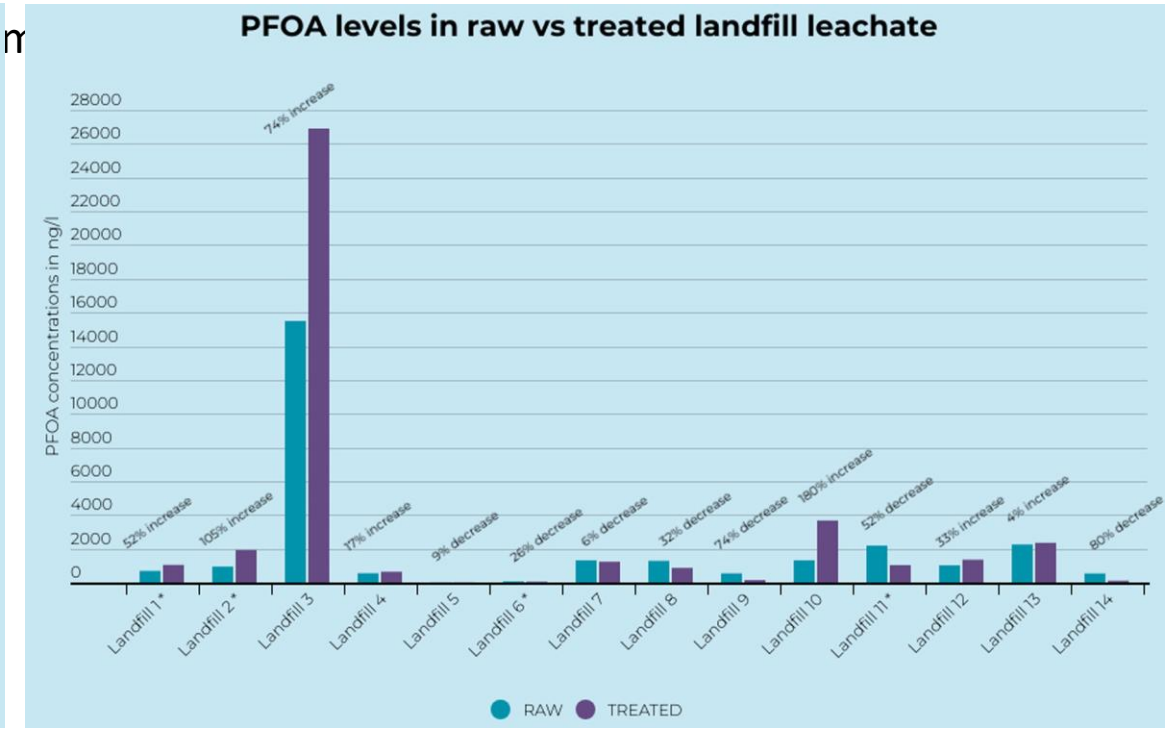


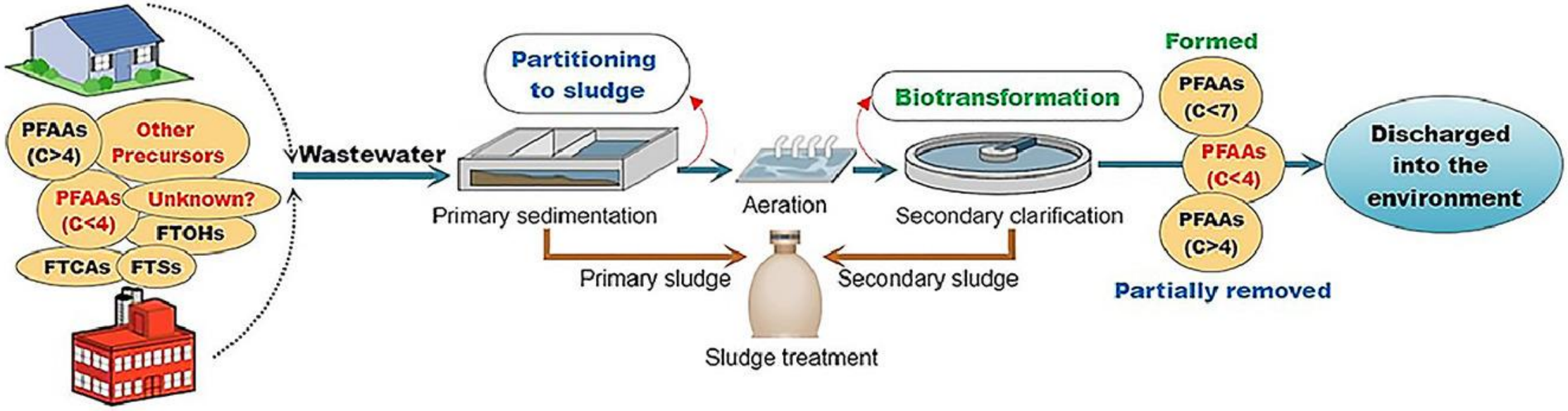
Figure 2. PFOA concentrations in raw and treated landfill leachate. *Represents average value for landfills with multiple treated or raw samples.

Wastewater treatment plant processes are not designed to remove PFAS

- A 2023 review summarizes the problem with treatment of leachate at wastewater treatment plants as “the treatment of ammonia and COD relies on chemical or biological oxidation, which do not effectively treat PFAS but often have the unintended effect of transforming precursor PFAS to terminal PFAS” (Tolaymat et al. 2023).

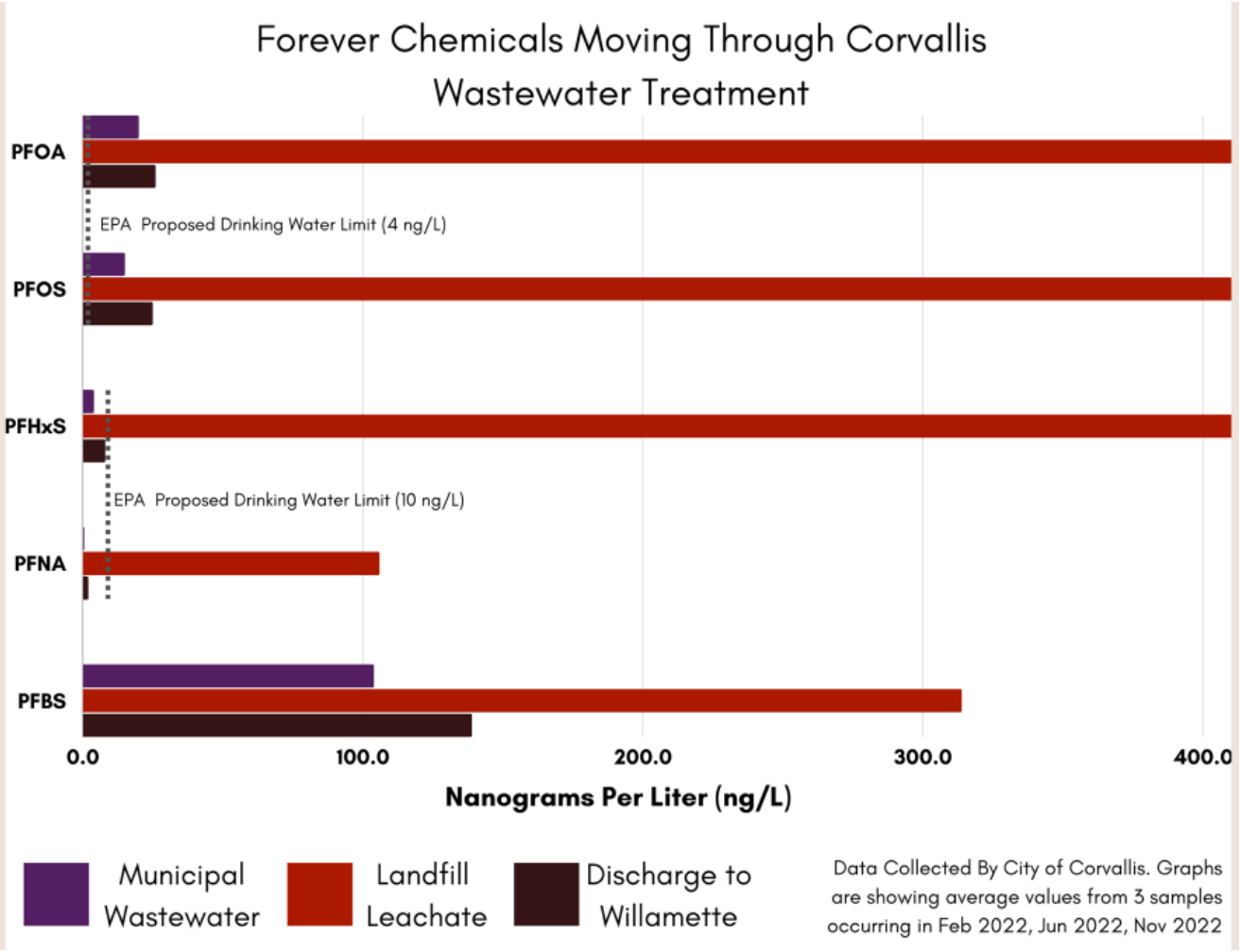
- The biological processes used to mitigate organic matter can convert precursors into PFAS resulting in higher PFAS levels in effluent compared to influent (Lenka et al. 2021).

- Short-chain PFAS (designated C<4 below) are generated (Lenka et al. 2021). Short-chain PFAS are highly mobile and extremely persistent (Brendel et al. 2021).



Most of the PFAS moving through the Corvallis WWTP comes from Coffin Butte Landfill leachate. The same could be said for probably any WWTP receiving landfill leachate.

Landfill leachate is especially high in PFAS



It is important to note that the PFAS not discharged to the Willamette River in effluent have not been destroyed. Instead, they are in the biosolids applied to local properties and in the air around the WWTP.

Wastewater treatment plant biosolids are applied to land in Benton County

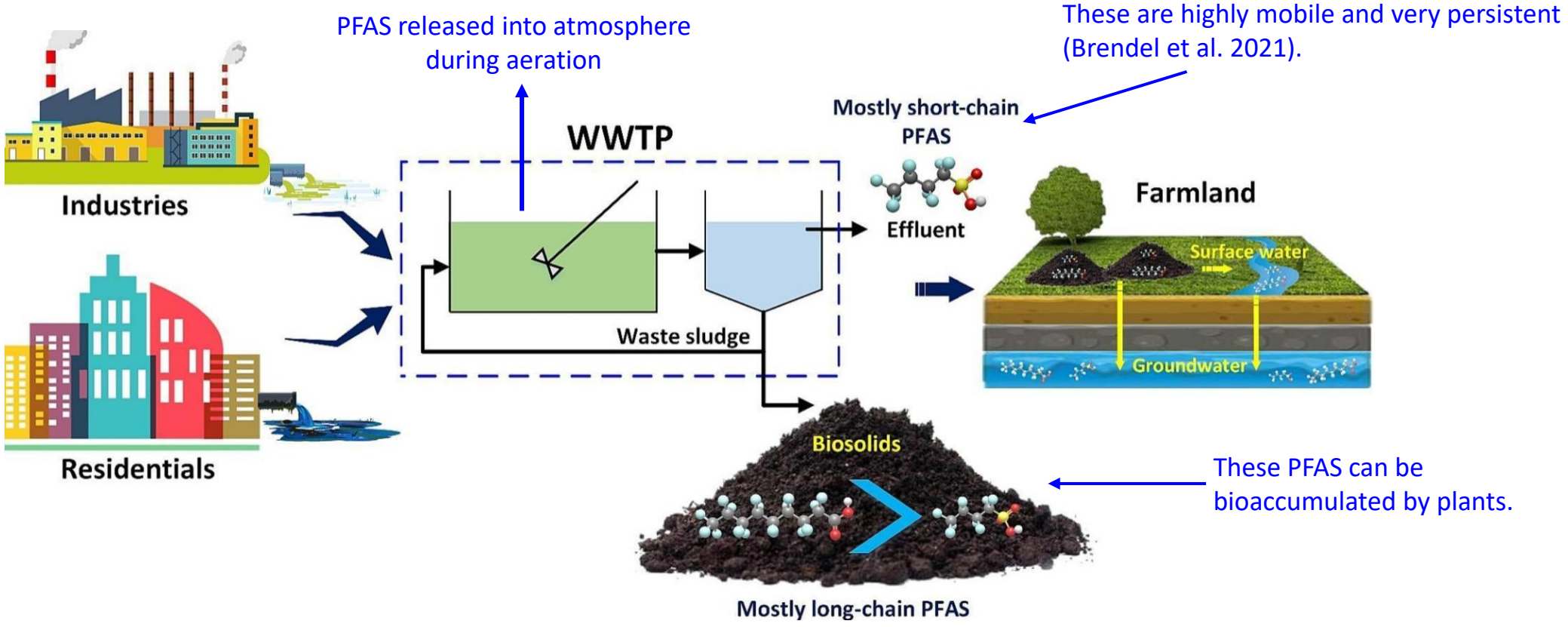
The Biosolids Management Plan available on the City of Corvallis' webpage is from 2007 and I find no reference to PFAS in it; heavy metals and pathogens are the focus (see Corvallis Biosolids pdf).

Given the persistent nature of PFAS, spreading biosolids on any soil is a bad idea. For example, biosolids used on future hay fields will result in livestock consuming PFAS that the grasses take up from the treated soil. Those grasses may have worsened the problem by converting PFAS into more bioavailable PFAS. Those PFAS may then accumulate in the livestock affecting them and suitability for human consumption.

Summary of why landfill leachate should not be disposed of with municipal wastewater

WWTPs do not treat PFAS (Tolaymat et al. 2024) and those PFAS leave the WWTP via effluent (to the Willamette River), via aerosols at aeration tanks and via biosolids applied to the land, often agricultural land. In short, PFAS from the landfill are distributed into the environment where they persist (Behnami et al. 2024).

The Corvallis and Salem WWTPs rely heavily on aeration (biological oxidation), a process that results in airborne PFAS and conversion of precursor to PFAS.



It IS possible to treat leachate and remove the PFAS but…

- Since the various harmful components of leachate require different treatments, a treatment facility that does this while removing PFAS requires the design, building and operation of a coordinated, nuanced treatment system (Wang and Qiao 2024). Otherwise, they risk generating *more* PFAS.

- "PFAS treatment technologies have advanced significantly in recent years, demonstrating promising applicability for commercial use. Despite this progress, challenges remain, including high energy consumption, high-maintenance requirements for high-strength matrices, reduced performance in the presence of background contaminants, and the production of harmful byproducts" (Sabba et al. 2025).

- I bring this to your attention because I observed some exchanges during the hearings that seemed to indicate that all parties were under the impression that PFAS mitigation is a relatively simple process when it is not. It is complex, expensive, and not the responsibility of Benton County.

- The landfill operators have shifted the PFAS burden onto Corvallis, Salem, Benton County, and Marion County by sending leachate to WWTPs and by not preventing the escape of PFAS-laden fugitive landfill gas. The operators of Coffin Butte Landfill must build a treatment facility to remove PFAS from leachate on site. This is their problem to solve. They have chosen to concentrate PFAS from around Oregon at Coffin Butte Landfill knowing that its wet location causes it to produce a great deal of PFAS-laden leachate, they must deal with the result of the choice to continue to concentrate PFAS in an inappropriately-situated landfill.

PFAS from landfills enter surface water and groundwater

- Direct leakage of leachate:
    - Although geomembrane liners have a 98% collection efficiency, imperfections allow approximately 1.2 billion L of municipal solid waste landfill leachate to enter groundwater directly every year (Tolaymat et al. 2023).
- WWTP effluent:
    - PFAS are not eliminated or mitigated at WWTPs. Millions of gallons of leachate have passed through the city of Corvallis' and Salem's WWTPs each year and the PFAS in the leachate that do not migrate with the biosolids or aerosolized during aeration into enter the Willamette River with WWTP effluent.

PFAS from landfills are spread to soils and air

WWTP Biosolids:

PFAS from landfill leachate enter the environment when the biosolids are spread on local fields. Lenka et al.'s 2021 review stated:

- "The occurrence of PFAS in biosolids and its consequences are often overlooked (Letcher et al. 2020), although biosolids and sludge from secondary treatment are notable routes for PFAS emission into the environment (e.g. groundwater, soil, vegetation etc.) (Ebrahimi et al. 2021, Stoiber et al. 2020, Zhang et al. 2013). PFCAs, PFSAs, precursors (e.g. FTOHs) (Guerra et al. 2014) and novel fluorinated polymers (Letcher et al. 2020) have been detected in biosolids worldwide, including North America, Europe, Africa, Asia, and Australia (Moodie et al. 2021)."

Air:

- It has recently been found that landfill gas (LFG) is a major route of PFAS entering the environment (Lin et al. 2024).

- WWTP processes cause aerosolization of PFAS which then enter the atmosphere (Vierke et al. 2011).

Landfills are a forever problem

“Landfills have historically been the primary waste disposal method in Europe, leading to numerous legacy sites requiring extensive aftercare to safeguard human health and the environment. Long-term aftercare of sanitary engineered landfills requires insight in the risk associated with possible emission of contaminants from the landfill waste body. In this research we developed an approach where data measured at the landfill sites are combined with a model in order to quantify the amount of mass that will emit from the waste body over a certain period of time. This is important information for landfill operators and responsible authorities in order to develop approaches for landfill aftercare. Especially because after-care is expected to last for a very long time (up to several centuries)” (Heimovaara and Wang 2025).

The responsible authorities in Benton County must consider this prospect when making decisions about the landfill.

Regulations will present problems on the road ahead

- The EPA drinking water rule for maximum PFAS levels has been established and was to be enforced in 2029, although that date was just moved to 2031.

- The EPA is already examining landfill effluent and biosolids with respect to allowable PFAS levels. Although no rules currently exist for WWTPs regarding mitigation or decreasing PFAS levels in effluent or sludge, I would expect that rules will come in the future.

- There are several EPA sheets regarding various aspects of PFAS in sewage/sludge, in drinking water and in landfill effluent that I attached to the email used to send this PowerPoint.

Thus far, Benton County has been extremely myopic in its consideration of the potential problems of Coffin Butte Landfill. The focus has been, and appears to continue to be, the problems associated with the working life of the landfill, not what comes after. The long-term problems are significant and the county will only worsen the burden associated with them by approving the application for expansion. Some of those problems will center around future mitigation/remediation/emissions requirements.

If Benton County approves the application for expansion of Coffin Butte Landfill, it will continue to fail to meet its own goals, guidelines and policies.

- As described previously, PFAS originating at Coffin Butte Landfill are spread to the air, soil and water in several ways.

- Continuing to allow this spread means the county is not meeting the **Benton County Comprehensive Plan policy 6.1.2** that dictates that the county must manage air, water, and land resources to insure their protection, conservation, restoration or enhancement. The same is true for policies **6.1.4** and **6.3.1** which call for the county to consider these types of impacts when making land use decisions and to take steps to minimize degradation of our air, water, and land.

- With regard to landfills and solid waste, comprehensive plan policies **6.5.4** and **6.5.8** direct the county to make sure that our solid waste handling, disposal and storage is safe and environmentally sound – allowing the unmitigated spread of PFAS does not meet the requirements in these two policies.

Allowing this problem to worsen by approving an expansion, is antithetical to everything Benton County and Corvallis purport to represent. Why have these policies at all if they play no role in decision-making?

The expansion does not meet the CUP criteria BCC 53.215(1)

- The possibility of widespread PFAS contamination seriously affects the use of landfill-adjacent properties (and WWTP-adjacent and biosolid application-adjacent properties).

    - The application does not meet **BCC 53.215(1)** since an expansion would mean a worsening of all the PFAS-related issues – more air, soil, and surface and groundwaters will be affected further interfering with the ability of residents to use their properties.

The expansion does not meet the CUP criteria BCC 53.215(2)

- Carrying out the PFAS mitigation required to meet EPA standards when they are defined and enforced will be a further economic burden on all of us.
    - We know that the landfill operators have a finite window of responsibility for maintaining the landfill once closed.
    - That leaves area residents faced with a potentially enormous financial bill associated with the contamination and continued production of leachate. Thus, an expansion does not meet **BCC 53.215(2)**.

It is time to do the right thing in Benton County

We have a chance to make a solid-waste handling course-correction and that is likely only if the expansion is denied. A denial of expansion will help prevent further environmental damage and likely save Benton County money in the long run. Benton County's future financial burden associated with dealing with local PFAS contamination will likely be high – far higher than any fees collected by the county for allowing Coffin Butte Landfill's operation. It is also my opinion that it has suffered a damaged reputation for allowing this problem to continue for so long.

Benton County must deny the application for expansion of Coffin Butte Landfill. The trash being trucked in should be transported to Columbia Ridge Landfill which produces much less leachate due to limited precipitation. The small volume of leachate that is produced stays on site being recirculated as part of their landfilling process. The extraordinarily thick clay layer beneath the landfill provides extra environmental protection. If a leak were to occur, the clay would buffer against the PFAS and other contaminants migrating to groundwaters. Columbia Ridge Landfill produces much less fugitive LFG, closing another route by which PFAS enter the environment. In short, Columbia Ridge Landfill truly concentrates and sequesters PFAS and other contaminants while Coffin Butte Landfill concentrates and then distributes PFAS and other contaminants into the environment via LFG and leachate sent to WWTPs where it is not treated for PFAS. For these reasons, diverting trash to Columbia Ridge Landfill is the appropriate course of action.

I strongly urge you to deny the application for expansion of Coffin Butte Landfill. It does not meet the CUP criteria and it is not right for Benton County or its residents.

Respectfully,

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Use of references

I wanted to include a note on my use of references in the event there are any science types on the board. Many of the references I used are review papers and I did that for two reasons. The first is that they are often the most readable for those who aren't in that field of study. Second, they summarize the current state of affairs with respect to a topic – I selected reviews that were as recent as possible to make sure the information was up-to-date.

Normally, one would cite the references from the primary literature, those papers reporting the findings on some very specific aspect of a topic. Using primary literature references would mean that I'd need to cite and attach far more papers.

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EMERGING CONTAMINANTS (D CHIANG AND C WALECKA-HUTCHISON, SECTION EDITORS)

Sources, Fate, and Plant Uptake in Agricultural Systems of Per- and Polyfluoroalkyl Substances

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Abstract

Purpose of Review Per- and polyfluoroalkyl substances (PFAS) are a family of > 4700 recalcitrant compounds, many of which are ubiquitous in the environment. This review aims to (1) identify PFAS sources and fate processes relevant to agricultural systems and (2) expand on plant uptake mechanisms and plant responses to PFAS.

Recent Findings The number of PFAS being quantified in studies involving soil, water, and plants is increasing. Transformation of precursors that tend to stay in the rhizosphere can lead to long-term PFAS reservoir to plants. Some PFAS are readily taken up, particularly the shorter-chain PFAS, and can evoke metabolic responses and phytotoxic effects at high concentrations. PFAS translocation from roots to shoots occurs through both active and passive transport mechanisms. Both PFAS uptake and effects vary between and within species.

Summary As new PFAS emerge, it will be necessary to continue expanding the list of PFAS quantified in land-applied media and assessing their accumulation potential in plants. While controlled laboratory or greenhouse studies have merit, comprehensive field studies are needed to provide clarity on PFAS fate and their relative risk in agricultural systems. Field studies should include identifying site-specific PFAS sources, quantifying a broader suite of PFAS and identifying potential precursors, evaluating plant uptake of replacement PFAS, reporting of soil properties and climatic conditions, and assessing risk of impacts to source and irrigation waters. This information can be utilized to inform future studies towards evaluating and mitigating risks to our food chain associated with PFAS in agricultural systems.

Keywords PFAS · PFOS · PFOA · Soil amendments · Phytotoxicity · Crops · Vegetables

Introduction

Per- and polyfluorinated alkyl substances (PFAS) are a widely utilized chemical family [1–3] now referred to as forever chemicals due to their high thermal and chemical stability as well as minimal biodegradability [4]. The perfluoroalkyl chain length (C_nF_{2n+1}–), the functional head group (e.g., carboxylic acid, sulfonate, alcohol, phosphate, amino, or other groups), which may be attached to a nonfluorinated hydrocarbon entity, and structural differences in the fluoroalkyl chain itself (e.g., branching, ether moiety) govern their behavior, subclassification, and nomenclature [5]. Some PFAS are highly bioaccumulative and have been reported in blood serum and breast milk of humans throughout the globe [6–8] and have been associated with adverse health effects [9]. Two classes within the PFAS family that have received the most attention are the perfluoroalkyl sulfonic acids (PFSAs) and the perfluoroalkyl carboxylic acids (PFCAs) [10•] of which perfluorooctane sulfonic acid (PFOS) and perfluorooctanoic acid (PFOA), respectively, are the most researched to date. Due to their highly bioaccumulative nature, PFOS and PFOA have been voluntarily phased out and are on the Stockholm Convention persistent organic pollutant lists Annex B (restricted) and Annex A (eliminated), and perfluorohexane sulfonic acid (PFHxS) is under review [11]. A range of less studied and often more mobile PFAS that are substitutes for PFOS and PFOA [12] have been found to be equally persistent and often with similar adverse effects, albeit often at higher concentrations [9], thus leading to additional management and regulatory challenges.

This article is part of the Topical Collection on *Emerging Contaminants*

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The primary PFAS exposure routes researched to date are drinking water, meat, fish, milk, food packaged in PFAS-containing materials, and indoor dust inhalation [13–16]. More recently, research has included evaluating PFAS exposure from commonly consumed agricultural products (grains, cereals, vegetables, fruits); therefore, it is crucial to understand PFAS sources to plants and the processes controlling how much of each PFAS is taken up by various plant types. Sources of PFAS to agricultural field crops consist of atmospheric deposition via precipitation or particle-bound PFAS, and absorption from the gaseous phase [17, 18], irrigation with contaminated well water or treated wastewater effluent [19–23], pesticides [24•, 25], aqueous film-forming foams (AFFFs) used to extinguish fires [26], and land-applied sewage sludge or biosolids [27•, 28–32, 33•], as well as industrial or municipal compost [34, 35]. Likewise, food grown in hydroponic and greenhouse systems can be impacted by contaminated water as well as other sources. Relative PFAS contributions from different sources will vary depending on location-specific activities and facilities. For example, PFAS contributions from atmospheric deposition have been especially concerning for crops grown near fluorochemical manufacturing facilities [36, 37]. AFFFs are typically the major PFAS source of highly contaminated water near fire training areas; however, impacts to agriculture are generally limited to irrigation with AFFF-contaminated water [26] which could significantly increase exposure risks [38]. Much less is known about the relative PFAS contribution to crops from land-applied biosolids and composts, which serve as both a valuable resource for improving crop yields and a waste management strategy. Also, factors and mechanisms controlling PFAS uptake by plants are not well understood. A critical understanding of these sources and mechanisms is necessary to determine the associated risks, trade-offs, and need for regulatory action [39–41].

Aim and Scope

This review will address PFAS in agricultural systems with an emphasis on land application scenarios, PFAS reservoirs in the rhizosphere, soil–plant transfer, and plant health. A recent review by Ghisi et al. [42•] summarized research on PFAS uptake and occurrence in grains, vegetables, and fruit prior to 2018 with an emphasis on comparing bioaccumulation trends between plants and impacts of environmental factors with a primary focus on C4 and C8 PFCAs and PFSAs. Studies after 2018 have addressed an expanded list of PFAS and began to address precursor contributions to plant uptake. Precursors are PFAS that can degrade to PFCAs or PFSAs, which are the terminal metabolites and collectively referred to as perfluoroalkyl acids (PFAAs). Precursors are often less mobile than PFAAs, thus they can serve as a PFAA reservoir in the rhizosphere. The previous review did not include several field studies that provide valuable information on soil–plant transfer of PFAS under field conditions. In addition, insight into uptake mechanisms and effects to plant health were not specifically addressed. Therefore, this review expands upon the work by Ghisi et al. [42•] accordingly. In addition, hydroponic, pot, and field studies will be compared and contextualized to inform future studies. Recent studies on uptake of PFAS in different agricultural plants are synthesized and discussed in light of previously hypothesized uptake mechanisms and trends. Furthermore, studies on plant responses to PFAS contamination are discussed. Immediate data gaps are highlighted to guide future plant uptake studies, clarify potential PFAS sources in agricultural systems, and better inform relative risk of PFAS exposure through our food chain.

The Expanding List of PFAS

At least 4700 PFAS have been used in industrial and commercial processes and thus potentially released to the environment [3, 43]. PFAS include anionic, cationic, zwitterionic, and neutral compound groups with a range of functional groups (e.g., carboxylic acid, sulfonic acid, ether sulfonate, sulfonamide, sulfonamido acetic acid, phosphate diester, acrylate, amido ammonium salt, amido betaine, and others) [44, 45]. In addition, for a given PFAS, there can be multiple isomers depending on the manufacturing process, e.g., PFOS produced from an electrochemical fluorination process yields multiple branched isomers in addition to the linear isomer [5]. Initial PFAS occurrence studies targeted only PFOS and/or PFOA, and without isomer delineation. Research involving additional PFAS has increased as awareness of the numbers of PFAS in the environment, analytical method development, and analytical standard availability have increased. Table 1 provides a classification and acronym summary for the PFAS that have been included in the plant uptake studies reviewed. PFAS in these studies were quantified using liquid chromatography (LC) paired with mass spectrometry (MS), tandem mass spectrometry (MS/MS), high-resolution mass spectrometry (HRMS), or quadrupole time of flight MS (QToF-MS) [46, 47]. Targeted quantification is limited to availability of standards, which currently includes approximately 60 PFAS. Suspect and nontarget screening with advanced MS instrumentation (e.g., QToF-MS) may be used to identify other PFAS including precursors and intermediates in the pathways to PFAAs. Precursor transformation to PFAAs typically involves multiple transformation steps with different degradation rates and some quite stable intermediates. Degradation rates, metabolites, and metabolite yields vary with each PFAS class as well as environmental factors commonly known to affect microbial degradation [48]. For example, aerobic half-lives of fluorotelomer alcohols are < 1 week with yields to PFOA as high as 40% whereas aerobic soil

Table 1 PFAS included in analytical methods used in plant uptake studies along with acronyms and chemical formulas

Class	PFAS	Acronym	Formula
Carboxylic Acid (PFCA)	Trifluoroacetic acid	TFA	CF_3COOH
	Perfluoropropanoic acid	PFPrA	CF_3CF_2COOH
	Perfluoro-butanoic acid	PFBA	$CF_3(CF_2)_2COOH$
	Perfluoropentanoic acid	PFPeA	$CF_3(CF_2)_3COOH$
	Perfluorohexanoic acid	PFHxA	$CF_3(CF_2)_4COOH$
	Perfluoroheptanoic acid	PFHpA	$CF_3(CF_2)_5COOH$
	Perfluorooctanoic acid	PFOA	$CF_3(CF_2)_6COOH$
	Perfluorononanoic acid	PFNA	$CF_3(CF_2)_7COOH$
	Perfluorodecanoic acid	PFDA	$CF_3(CF_2)_8COOH$
	Perfluoroundecanoic acid	PFUnDA	$CF_3(CF_2)_9COOH$
	Perfluorododecanoic acid	PFDoA	$CF_3(CF_2)_{10}COOH$
	Perfluorotridecanoic acid	PFTrDA	$CF_3(CF_2)_{11}COOH$
	Perfluorotetradecanoic acid	PFTeDA	$CF_3(CF_2)_{12}COOH$
	Perfluorohexadecanoic acid	PFHxDA	$CF_3(CF_2)_{14}COOH$
	Perfluorooctadecanoic acid	PFODA	$CF_3(CF_2)_{16}COOH$
Sulfonic Acid (PFSA)	Perfluoroethane sulfonate	PFEtS	$CF_3CF_2SO_3$
	Perfluoropropane sulfonate	PFPrS	$CF_3(CF_2)_2SO_3$
	Perfluorobutanesulfonate	PFBS	$CF_3(CF_2)_3SO_3$
	Perfluoropentanesulfonate	PFPeS	$CF_3(CF_2)_4SO_3$
	Perfluorohexanesulfonate	PFHxS	$CF_3(CF_2)_5SO_3$
	Perfluoroheptanesulfonate	PFHpS	$CF_3(CF_2)_6SO_3$
	Perfluorooctanesulfonate	PFOS	$CF_3(CF_2)_7SO_3$
	Perfluorodecanesulfonate	PFUdA	$CF_3(CF_2)_9SO_3$
Perfluoroalkane Sulfonamide (FASA)	Perfluorooctane sulfonamide	FOSA, PFOSA	$CF_3(CF_2)_7SO_2NH_2$
	N-methylperfluorooctane sulfonamide	N-MeFOSA	$CF_3(CF_2)_7SO_2NHCH_3$
	N-ethylperfluorooctane sulfonamide	N-EtFOSA	$CF_3(CF_2)_7SO_2NHCH_2CH_3$
Perfluoroalkane Sulfonamido Ethanol (FASE)	N-Methylperfluorooctanesulfonamidoethanol	N-MeFOSE	$CF_3(CF_2)_7SO_2N(CH_3)(CH_2CH_2OH)$
	N-ethylperfluorooctanesulfonamidoethanol	N-EtFOSE	$CF_3(CF_2)_7SO_2N(CH_2CH_3)(CH_2CH_2OH$
Chlorinated Polyfluoroalkyl Ether Sulfonate (Cl-PFAES)	6:2 chlorinated polyfluoroalkyl ether sulfonate	6:2 Cl-PFAES	$Cl(CF_2)_6O(CF_2)_2SO_3$
	8:2 chlorinated polyfluoroalkyl ether sulfonate	8:2 Cl-PFAES	$Cl(CF_2)_8O(CF_2)_2SO_3$
Perfluoroalkane Sulfonamido Acetic Acid (FASAA)	N-methyl perfluorooctane sulfonamido acetic acid	N-MeFOSAA	$CF_3(CF_2)_7SO_2\ N(CH_2)(CH_2CO_2)$
	N-ethyl perfluorooctane sulfonamido acetic acid	N-EtFOSAA	$CF_3(CF_2)_7SO_2\ N(CH_2CH_3)(CH_2CO_2)$
Fluorotelomer Phosphate Diester (DiPAP)	6:2 fluorotelomer phosphate diester	6:2 DiPAP	$[CF_3(CF_2)_5(CH_2)_2\ O]_2PO_2$
	8:2 fluorotelomer phosphate diester	8:2 DiPAP	$[CF_3(CF_2)_7(CH_2)_2\ O]_2PO_2$
	10:2 fluorotelomer phosphate diester	10:2 DiPAP	$[CF_3(CF_2)_9(CH_2)_2\ O]_2PO_2$

Table 1 (continued)

Class	PFAS	Acronym	Formula
Fluorotelomer Sulfonic Acid (FTSA)	6:2 fluorotelomer sulfonic acid	6:2 FTSA	$CF_3(CF_2)_5(CH_2)_2SO_3$
Fluorotelomer Carboxylate (FTCA)	3:3 fluorotelomer carboxylic acid	3:3 FTCA	$CF_3(CF_2)_2CH_2CH_2COOH$
	5:3 fluorotelomer carboxylic acid	5:3 FTCA	$CF_3(CF_2)_4CH_2CH_2COOH$
	7:3 fluorotelomer carboxylic acid	7:3 FTCA	$CF_3(CF_2)_6CH_2CH_2COOH$
	6:2 fluorotelomer carboxylic acid	6:2 FTCA	$CF_3(CF_2)_5CH_2COOH$
	8:2 fluorotelomer carboxylic acid	8:2 FTCA	$CF_3(CF_2)_7CH_2COOH$
	6:2 unsaturated fluorotelomer carboxylate	6:2 FTUCA	$CF_3(CF_2)_4CF{=}CHCOOH$
	8:2 unsaturated fluorotelomer carboxylate	8:2 FTUCA	$CF_3(CF_2)_6CF{=}CHCOOH$
Fluorotelomer Alcohol (FTOH)	6:2 fluorotelomer alcohol	6:2 FTOH	$CF_3(CF_2)_5(CH_2)_2OH$
	8:2 fluorotelomer alcohol	8:2 FTOH	$CF_3(CF_2)_7(CH_2)_2OH$
	10:2 fluorotelomer alcohol	10:2 FTOH	$CF_3(CF_2)_6(CH_2)_2OH$

degradation of fluorotelomer sulfonates, which is both a precursor and an intermediate of some larger S-containing PFAS, is much slower with desulfonation being the rate-limiting step [49].

Other approaches for estimating the contribution of unknown PFAS loads include conducting a fluorine mole balance using bulk organofluorine measurements or chemical assays such as the total oxidizable precursor (TOP) assay, particle-induced gamma-ray emission (PIGE) method, and adsorbable organic fluorine (AOF) or extractible organic fluorine (EOF) paired with combustion ion chromatography (CIC) methods [50]. Understanding the contribution of the unknown PFAS is helpful when making cross-study comparisons between ∑PFAS, ∑PFSAs, and ∑PFCAs in targeted studies, due to variations between media and precursor degradation in temporal studies. For example, the PFOS precursor sulfluramid (active ingredient EtFOSA), which is a pesticide used in the region to control leaf-cutting ants, was proposed as the most likely source of measured PFOS in Brazilian soils (0.020–3.77 ng/g) [24•]. In a carrot uptake study using soil fortified with EtFOSA, temporal generation of FOSAA and FOSA (intermediates to PFOS) and PFOS with yields up to 34% was attributed to EtFOSA degradation [25]. Likewise, in a wheat uptake study using FOSA-fortified soil, degradation to PFOS was observed in both the soil and the earthworms [51]. A small subset of these now known precursors is starting to be routinely quantified; however, currently, there are still many known precursors that are often not quantified. In addition, there are likely still many unknown and difficult to quantify precursors that may be a significant fraction of the total PFAS present and could serve as a continual source of PFAAs depending on degradation potential and degradation rates. For example, a recent study looking at several Canadian biosolids found two side-chain fluorinated polymer surfactants used in consumer products at higher concentrations than the total for targeted PFAAs [33•]. Therefore, although the list of PFAS typically quantified is likely to expand rapidly over the next few years, PFAS occurrence and potential PFAA reservoir are likely underestimated, especially in, but not limited to, earlier studies.

PFAS Sources and Presence in Soils and Growing Media

To provide a foundation to discuss PFAS availability to plants in different growing media, typical PFAS types including precursors and concentration ranges are briefly detailed in this section. PFAS have become so ubiquitous in the environment that finding pristine soils is rare. A recent review by Vedagiri et al. [52] of soils in North America with no known PFAS

point sources found background levels of PFOS and PFOA ranged from 0.018 to 2.55 ng/g ($n = 38$ studies) and 0.059 to 1.84 ng/g ($n = 40$), respectively. A global survey of 32 PFAS in soils taken from 62 locations with limited to no anthropogenic impact revealed PFAS presence in every sample; long-range transport was identified as the source [53]. Of the data reviewed, 33 samples from North America only reported PFOS and PFOA values. Globally, ∑PFCAs (C6–C14) ranged from 0.03 to 14.3 ng/g, and ∑PFSAs (C6–C10) ranged from < LOD to 3.27 ng/g. Other PFAS quantified are intermediates and precursors in the PFAA degradation pathway including a series of ten FTUCAs and ten FTCAs, which ranged from < LOD to 0.106 ng/g and < LOD to 0.048 ng/g in soils, respectfully. Comparison across the globe indicated that the northern hemisphere (especially Asia and North America) had higher PFAS concentrations than the southern hemisphere, which is consistent with results from previous atmospheric distribution studies and oceanic studies [53].

Biosolids and irrigation waters are the most prominently studied sources of PFAS in agricultural soils. PFAS are found in multiple water resources around the world [52], which when utilized as an irrigation source can lead to higher PFAS loads in soils then global background concentrations. Irrigation with well water, treated wastewater effluent, and surface water is essential to agriculture in many regions, and thus is a source of PFAS to plants that is difficult to avoid. Of 13 PFAS tested in agricultural soils from a Ugandan wetland irrigated with surface water receiving effluent discharged by wastewater treatment plants (WWTPs), PFOS was the most abundant (36–50% of ∑PFAS, 0.6–3.0 ng/g dw) > PFOA (6.2–15%, 0.48–0.91 ng/g dw) > PFHxA (4.7–18% dw, 0.2–0.5 ng/g dw) [23]. In a South Korean rice paddy irrigated with surface water impacted by effluent from WWTP receiving textile industry waste, the soil contained ∑PFAS ranging from 0.12 to 13.9 ng/g dw [54]. Zhang et al. [55] reported ∑PFAAs in irrigated agricultural fields in China ranging from 0.29 to 4.28 ng/g (mean 1.62 ng/g) with PFOA (39% of ∑PFAAs) > PFDA (16%) > PFBA (9%) > PFHxA (7%) = PFNA (7%). In home gardens in the USA irrigated with landfill leachate-contaminated groundwater, several PFAAs, including PFOA (0.11–3.0 ng/g), PFOS (0.57–12.0 ng/g), PFBA (0.04–13.0 ng/g), PFPeA (ND—0.57 ng/g), PFHxS (ND—0.24 ng/g), and PFHxA (ND—0.66 ng/g), were detected in over 70% of the soil samples from 20 home gardens [22•]. Comparatively lower concentrations were found in 3 irrigated gardens outside the contaminated zone: PFOS (0.93–2.1 ng/g), PFOA (0.29–0.54 ng/g), and PFBA (0.073–0.49 ng/g).

Waste-derived soil amendments and fertilizers, including biosolids and composts made from biodegradable solid wastes, are frequently land-applied as an organic matter source that improves soil structure and water-holding capacity as well as a slow-release source of nitrogen and other micro- and macronutrients [56–58]. Concerns that these waste-derived soil amendments are significant sources of PFAS to the environment have been increasing over the past decade [27•, 30, 33, 35, 59, 60]. Biosolids, which are prepared from the sludge produced in the wastewater treatment process, inherently contain higher concentrations of the more sorptive longer perfluoroalkyl chains and larger precursors than the corresponding wastewater effluent. Longer-chain PFAS refer to those that have six or more perfluoroalkyl carbons. Degradation of these precursors throughout the sludge treatment process and after land application can lead to increased concentrations of the shorter-chain PFAAs [61]. PFAS loads to soil from biosolid application depend on the PFAS sources to WWTPs, the PFAS concentrations in the biosolids, and biosolid application rates [33•, 62, 63]. Furthermore, given the persistence of PFAS, multiple biosolids applications may result in higher PFAS concentrations [63, 64]. Sepulvado et al. [64] investigated PFAS at a site in Illinois where biosolids were applied over a 3-year period. Various loading rates and soil types were utilized, along with 6 different biosolids that contained a mean ∑PFAS concentration of 433 ng/g. PFOS (80–219 ng/g), MeFOSAA (63–143 ng/g), EtFOSAA (42–72 ng/g), and PFOA(8–68 ng/g) were detected most frequently in biosolid samples [64]. PFAS leaching at the site increased with decreasing carbon chain length, thus greater leaching of short-chain PFAS was observed [64]. While Sepulvado et al. [64] did include known PFAA precursors MeFOSAA and EtFOSAA in their study, other precursors yet to be identified may have been present. For example, in soils from a biosolid-amended field in Ontario, Chu and Letcher [30] found, for the first time, two side-chain fluorinated polymer surfactants (S1 and its shorter-chain replacement S2) from Scotchgard™ fabric protectors totaling ~240 ng/g dw, which was one hundred times higher than the total sum of 22 other more typically targeted PFAS [30]. Later, they found these same PFAS in biosolids from numerous wastewater treatment plants in Canada [33•]. While studies have yet to be conducted to quantify the degradation of these larger PFAS, PFAS used in textiles and leather for stain and water resistance may be a long-term source in domestic wastewater with potential to generate PFAAs.

Most commercially available waste-derived products sold in local or major store chains, which are typically purchased for use in urban and sub-urban gardens, golf courses, and public and homeowner lawns, have been found to contain PFAS. Kim Lazcano et al. [59] found 9.0–181 ng/g of total PFAAs in commercially available municipal biosolid-based products. PFAS are also present in non-biosolid urban waste-derived composts made from compostable paper products, food wastes, and tree and grass clippings. Total PFAAs in organic waste composts ranged from 31 to 75 ng/g for those containing food packaging with PFAS concentrations being lower in composts of only food and yard wastes (≤ 18.89 ng/g) [35, 59]. All these biosolid and non-biosolid waste-derived

composts contained PFOA and PFOS even though phase out was several years earlier than when these materials were produced. Several PFAS precursors were also detected with the most common being the 6:2 diPAP and 6:2 FTS [35, 59]. Likewise, total PFAAs up to 80 ng/g dw were found in biowaste from biogas production [31]. Overall shorter-chain PFAAs were higher in municipal solid waste- and biowaste-derived amendments compared to biosolid-based products [31, 35, 59]. Other waste-derived materials used as soil amendments that have not been studied for PFAS occurrence include paper mill wastes [65] which are expected to contain PFAS associated with PFAS-based paper coatings. Other PFAS sources applied to agricultural soils include PFAS-based pesticides, e.g., sulfluramid as exemplified already, or pesticides that may contain PFAS in their formulations as an ingredient in the surfactant carrier of which little is known.

PFAS production or use in manufacturing can lead to PFAS additions to agricultural soils through atmospheric deposition and/or contaminating water used for irrigation. Soil samples in agricultural fields located in five cities near fluorochemical manufacturing sites were commonly found to contain PFAS with detection frequency of PFBA (18.2%) > PFOA (31.8%), > PFDA (11.4%) > PFNA(12.6%) [66]. Liu et al. [36] reported total PFAS concentrations in agricultural soils of 79.9 to 200 ng/g and 2.09 to 3.75 ng/g within 0.3 km and 10 km, respectively, of a fluorochemical plant. PFOA made up the greatest portion (83.4%, up to 181 ng/g) of the total quantified with the remainder being primarily the shorter-chain PFCAs [36].

Use of AFFFs has led to release of a wide range of both PFAS types and concentrations to the environment. For example, in Australia, 60 different PFAS were detected in AFFF-impacted soil, including novel PFAS identified with nontarget techniques [26]. Several PFSAs, PFCAs, chlorinated perfluoroalkyl sulfonate (Cl-PFSAs), ketone perfluoroalkyl sulfonates, FASA, perfluoroalkyl sulfinates, perfluoroalkyl sulfonamidoethanols, fluorotelomer sulfonates, and perfluoroalkyl sulfoamido amines as well as one fluorotelomer thioamido sulfonate were detected. This exemplifies the need to include nontarget screening in future studies, particularly at, but not limited to, AFFF-impacted sites. High total PFAS concentrations are often observed at fire training sites, e.g., Dauchy et al. [67] reported up to 710,126 ng/g in surface soils and 300 to 8300 ng/L in groundwater.

PFAS Sorption and Mobility Characteristics

PFAS interactions with soil moderate PFAS mobility and plant availability, thus the long-term dynamics of PFAS in agricultural systems. Within a class of PFAS (e.g., PFCAs, PFSAs), PFAS sorption to soils increases with increasing perfluoroalkyl chain length [68]; therefore, leaching and bioavailability increase with decreasing chain length [64, 69]. For a given chain length, sorption tends to be higher for PFSAs compared to PFCAs [68]. Cationic and some zwitterionic PFAS tend to have a high affinity to agricultural soils, since healthy soils typically contain a significant cation exchange capacity [45, 70]. Although soil organic carbon (OC) generally increases sorption of organic contaminants, in a review by Li et al. [60], no consistent trend was apparent between OC and PFAS sorption by soils across the globe. Although parameters that may affect sorption were not provided in all papers, Li et al. [60] were able to evaluate trends between sorption, pH, and cation concentrations. They found pH to be significantly correlated to sorption for only EtFOSAA ($n = 10$), MeFOSAA ($n = 10$), and PFDS ($n = 10$), while a general trend of increasing PFAA sorption with increasing cation concentration and overall salinity was evident [60]. Correlations to other soil properties such as exchange capacities were constrained because property information was most often not reported [60]. Using PFAA sorption data obtained for a single soil and a single concentration but with different solution pH and cation presence (Al, Ca, and Na), Campos Pereira et al. [68] generally observed inverse correlations between OC-normalized sorption coefficients and pH. Both PFAAs and acid functional groups in soil organic matter (SOM) become more anionic with increasing pH, thus lower sorption with higher pH may be expected for anionic PFAAs. Cation effects were more varied and complicated since their addition can invoke pH changes as well. Overall, the authors attributed the effects observed to changes in the net charge of organic matter, which can be impacted by pH as well as cation charge and concentration. They also hypothesized that longer-chain PFAAs preferentially sorb to the humin SOM fraction whereas sorption of the shorter chain was more to humic and fulvic acid domains [68]. Although data were limited, results indicated that different OM sources (e.g., OM arising from different crop and management strategies including land-applied waste-derived amendments) may alter PFAS retention, thus mobility and plant uptake potential. Factors not considered by Campos Pereira et al. [68] include nonlinear sorption behavior [71] and the effect of cation charge and concentration on PFAA clustering [72].

Inorganic anions can also impact PFAA sorption. Addition of phosphate decreased sorption of PFOS due to site-specific competition with the greatest impact being in soils with low OM content and high ferric oxide content [71]. Given the importance of phosphorus as a nutrient in agricultural systems, further studies on the effect of phosphorus speciation and concentration on sorption of additional PFAS are warranted. Lastly, other sorption mechanisms unique to PFAS as surfactants include the potential for enhanced retention at the air–water interface in the vadose zone [73, 74] (the unsaturated zone of soil above the water table including the root zone).

In unsaturated soils, air–water partitioning of PFOS [74] and PFOA [73] at higher concentrations typical of AFFF-impacted sites accounted for 32% and up to 75% of the retention by soil, respectively. This indicates that rain events that lead to saturation may temporarily mobilize sorbed PFAS. The role of interfacial tension at PFAS concentrations that may be released from typical waste-derived soil amendments versus AFFFs is yet to be determined.

Uptake into Agricultural Plants

The plant uptake studies reviewed include hydroponic studies ($n = 17$), pot studies ($n = 19$), and field studies ($n = 14$) (Table S1). Table 2 provides a summary of the plant species studied, reported uptake and comparisons between study types for the most commonly studied PFAAs, and the range in number of PFAS that have been included in at least one study for the given species organized by classification. A total of 63 field crops and produce types have been studied with the most studied reported in Table 2. Grains, leaf vegetables, and root vegetables were the most studied plants, with sweet fruits being the least studied. Generally, vegetables and vegetative plant components had higher PFAS concentrations than grains and fruits. In home gardens, floret vegetables had higher concentrations of PFAS than other plant categories [22]. Studies indicate that long-chain PFAAs will be less susceptible to leaching and less likely to transfer from soils to plants or be translocated compared to the shorter-chain replacements which are more water soluble [12, 35, 75•]. Likewise, precursors can serve as a reservoir of smaller more mobile and bioavailable PFAS (intermediate metabolites and subsequent terminal PFAAs) [51]. The source strength of the precursor reservoir will depend on degradation rates and PFAA yield potential [48], which has not been determined for many precursors especially those more recently identified.

Synthesizing data from plant uptake studies can be challenging due to reported differences between plant species, within plant species (different cultivars), and within the different plant components (root, shoot, leaves, fruit) [42•, 82, 92•], as well if data are reported on a wet or dry weight basis. To compare between treatments and different systems, several ratios have been utilized including the root concentration factor (RCF, ratio of root to soil concentration), translocation factor (TF, ratio of shoot to root concentration) or sometimes called the foliage to root concentration ratios (FRCFs), shoot concentration factor (SCF, ratio of shoot to the growing media), and bioconcentration factor (BCF, ratio of the plant to growing media). In addition, a transpiration stream concentration factor (TSCF) can be used to describe translocation of PFAS via the transpiration stream; however, TSCF is not highly reported and is better suited to hydroponic studies; plants grown in soil require a porewater analysis to calculate TSCF [102•].

In studies using different ratios of PFAS between compartments in various plant systems, the PFAS characteristic appears to be the single most consistent factor affecting TFs and RCFs [20, 36, 62, 75•, 74–76, 77•, 79–81, 102•, 103, 104], which allots some insight into uptake mechanism. RCFs generally increased with increasing chain length and were greater for PFSAs compared to PFCAs of similar chain length [76, 80, 102•, 104], whereas TFs generally increased with decreasing chain length [20, 36, 62, 75•, 76, 79, 80]. However, there are exceptions to these general trends attributed to plant-specific mechanisms to be discussed in the mechanism section [91, 105•]. For a given PFAS, there are also notable trends with PFAS concentrations in the growing media. Current trends described in the literature indicate that root uptake concentrations increase with increasing PFAS concentrations [82, 91, 96]. This leads to relatively constant RCF values for given PFAS unless sorption to tissues reaches a maximum (e.g., nonlinear sorption to tissues) in which case the RCFs may decrease [91]. Across many species, short-chain PFAAs readily uptake and translocate throughout the plants [36, 42•, 75•, 79, 104], thus they tend to have higher TFs than longer chain. Short-chain PFAAs translocated to edible portions of crops more often than long-chain PFAAs [66], partially because short-chain PFAS sorb less to soils [106] and thus are more bioavailable [22•, 104, 107]. Also, it is plausible that with smaller size PFAS, plant root barriers may have a reduced impact on restricting their translocation [75•] as will be discussed in regard to plant uptake mechanisms.

Study type and design will affect PFAS bioavailability, the contact period during which the plant can uptake PFAS and/or PFAS effects on the plant. In hydroponic systems, overall PFAS availability is high compared to soil since it is not moderated by sorption; however, the root surface area where uptake occurs can be reduced in a hydroponic system. For example, red chicory root development in hydroponic systems was reduced; thus, uptake surface area decreased compared to chicory grown in soil [102•]. Most studies, particularly in pot and hydroponic studies, invoke PFAS treatments to seedlings (germination occurs in a different environment); thus, potential effects on germination and early growth are missed. Field studies often include impacts on germination since seeds are planted rather than seedlings and have additional variables that can impact PFAS availability for plant uptake such as potential for compounds to leach out of the root zone and climate variability. Also in most pot studies, leaching is minimized or prevented; thus, short-chain PFAS remain in the root zone unlike what is likely to occur in the field. For example, decreases were reported in short-chain PFAS concentrations in the soil profile throughout a growing season due to leaching [64, 106], whereas in a pot study short-chain PFAS became enriched in the bottom layers of the pot compared to the longer-chain PFAS and remained accessible for

Table 2 Comparison between three study types (A = hydroponic, B = pot studies, C = field studies) conducted on agricultural plants for PFBA (C4), PFHxA (C6), PFOA (C8), PFBS (C4), PFHxS (C6), and PFOS (C8) with parenthetical indicating total carbon chain length

Plant	Study type (# of studies)			Results										References
	A	B	C	PFAS class (range studied within the class for a given study)		Concentration ranges (ng/g dw)			RCF (range)		TF (range)			Hydroponic[a], pot[b], field[c]
						A	B	C	B	C	A	B	C	
Wheat	5	4	2	PFCA (1–9)	PFBA	200–1400	3170–406,080	1–1100	18–19	2.7–43.91	0.7–2.5	1.2–2.3	0.6–1.4	[20, 75–78][a]
					PFHxA	500–2000	810–8060	0.3–135	0.2–6.8	2.8–4.3	–	1.1–1.9	0.3–1.0	[79–82][b]
				PFSA (0–3)	PFOA	100–10,800	30–341,000	1–809	0.05–2	1.3–4.9	0.02–0.3	0.1–1.1	0.2–0.5	[36, 62] [c]
					PFBS	–	310–15,280	< LOQ–60	0.4–1.6	NA- 1.8	–	0.1–1.3	NA–0.4	
				Cl-PFAES (0–2)	PFHxS	–	150–7240	< LOQ–68	0.1–2	NA- 2.8	–	0.2–1.2	NA–0.2	
					PFOS	500–28,600	100–76,900	0.2–12,500	0.02–1.4	0.8–6	0.018	0.1–1	0.2–1	
Ryegrass		3		PFCA (1)	PFBA	–	–	–	–	–	–	–	–	[82–84] [b]
				PFSA (1)	PFHxA	–	–	–	–	–	–	–	–	
				FASA (0–2)	PFOA	–	1–100,300	–	2.35	–	–	0.6	–	
					PFBS	–	–	–	–	–	–	–	–	
				FASAA (0–1)	PFHxS	–	–	–	–	–	–	–	–	
					PFOS	–	1–42,000	–	1.4–1.5	–	–	0.1–2.1	–	
Maize		5	5	PFCA (0–13)	PFBA	–	30–35,200	< LOQ–1449	–	11–25.5	–	–	0.7–1.3	[28, 82–85] [a]
					PFHxA	–	30–2800	< LOQ–116	–	1.9–11.3	–	–	0.6–0.7	[22, 23, 36, 63, 86][c]
				PFSA (1–4)	PFOA	–	< LOQ–640	< LOQ–2478	–	1.1–2.4	–	–	0.1–0.3	
					PFBS	–	2–1800	< LOQ–100	5	–	–	–	–	
				FASA (0–2)	PFHxS	–	< LOQ–7980	< LOQ–0.05	2.6	–	–	–	–	
					PFOS	–	< LOQ–23100	< LOQ–0.2	8.8	–	–	–	–	
				FASE (0–2)										
				FASAA (0–2)										
				FTSA (0–1)										
				FTCA (0–7)										
				FTOH (0–3)										
				DiPAP (0–3)										
				Cl-PFESA (0–2)										
Rice		1	1	PFCA (10–13)	PFBA	–	ND	0.01–2.4	–	–	–	2.2–60	–	[19] [b]
					PFHxA	–	ND	0.02–1	–	–	–	4.3–6.2	–	[54][c]
				PFSA (4–5)	PFOA	–	ND–1.73	< LOQ − 1.9	–	–	–	0.1–1.6	–	
					PFBS	–	ND	< LOQ–0.03	–	–	–		–	
				FASA (0–2)	PFHxS	–	ND	< LOQ–3.8	–	–	–	0.1–0.5	–	
					PFOS	–	ND	< LOQ–55.5	–	–	–	0.1–0.2	–	
				FASAA (0–1)										

Table 2 (continued)

Plant	Study type (# of studies)			Results									References
	A	B	C	PFAS class (range studied within the class for a given study)		Concentration ranges (ng/g dw)			RCF (range)		TF (range)			Hydroponic[a], pot[b], field[c]
						A	B	C	B	C	A	B	C	
Soybean	1	2	1	PFCA	PFBA		–	223–2378	–	69.5	–	–	0.7	[87][a]
				(0–9)	PFHxA		–	5.5–212	–	6.4	–	–	0.6	[83, 84] [b]
				PFSA	PFOA	ND	120–1340	0.3–3967	–	1.7	–	0.1	0.002	[36] [c]
				(0–3)	PFBS	0.02–0.15	–	< LOQ	–	–	0.1	–	–	
				FASA	PFHxS	0.02–0.2	–	< LOQ	–	–	0.1	–	–	
				(0–2)	PFOS	0.3–15	20–720	0.12.3	2.3–4.7	5.6	0.02	0.1–1.2	0.8	
				FASAA (0–2)										
Alfalfa		3	1	PFCA	PFBA	–	–	ND	–	–	–	–	–	[83, 84, 88] [b]
				(0–11)	PFHxA	–	–	ND	–	–	–	–	–	[89] [c]
				PFSA	PFOA	–	1311.5–4310.3	ND	10.3	–	–	0.304	–	
				(1–4)	PFBS	–	–	ND	–	–	–	–	–	
				FASA	PFHxS	–	–	ND	–	–	–	–	–	
				(0–3)	PFOS	–	10–481.4	ND	2.3–3.12	–	–	0.131–1.1	–	
				FASAA (0–2)										
Lettuce	3	8	3	PFCA	PFBA	< LOQ–84	< LOQ–25,100	< LOQ–2365	–	–	2.7	–	–	[21, 90, 91] [a]
				(1–11)	PFHxA	< LOQ–14.4	< LOQ–9460	< LOQ–72	–	–	0.7	–	–	[34, 63, 83, 84, 88, 92–94][b]
				PFSA	PFOA	0.02–135	< LOQ–2523	< LOQ–1038	–	6	0.3	0.2	–	[22, 36, 63] [c]
				(0–5)	PFBS	< LOQ–17	< LOQ–3670	< LOQ–1.6	–	–	0.3	–	–	
				FASA	PFHxS	< LOQ–30	< LOQ–1250	< LOQ–0.5	–	–	0.2	0.1–1.6	–	
				(0–2)	PFOS	< OQ–205	< LOQ–279	< LOQ–3.5	0.2–14.6	3.8–3.9	0.1	0.02–0.4	–	
				FASAA (0–2)										
				DiPAP (0–1)										
Spinach		1	4	PFCA	PFBA	–	ND	ND–6.7	–	–	–	–	–	[28] [b]
				(9–13)	PFHxA	–	ND	ND–1.2	–	–	–	–	–	[22, 55, 66, 95] [c]
				PFSA	PFOA	–	ND–2.4	ND–6.7	–	–	–	–	–	
				(2–8)	PFBS	–	ND	ND–1.7	–	–	–	–	–	
				FASA	PFHxS	–	–	ND–0.1	–	–	–	–	–	
				(0–3)	PFOS	–	1–1.6	ND–0.1	–	–	–	–	–	
				FASAA (0–2)										
				FTCA (0–1)										
Cabbage	1		2	PFCA	PFBA	< LOQ–10.9	–	2.5–17.8	–	–	1.5	–	–	[96] [a]
				(9–11)	PFHxA	0.1–14.4	–	0.2–1.4	–	–	0.6	–	–	[22, 55] [c]
				PFSA (3)	PFOA	0.1–10.9	–	1.3–4.0	–	–	0.4	–	–	
					PFBS	0.2–31.5	–	ND–0.1	–	–	0.7	–	–	
					PFHxS	0.1–31.2	–	ND–0.1	–	–	0.4	–	–	

Table 2 (continued)

Plant	Study type (# of studies)			Results									References
	A	B	C	PFAS class (range studied within the class for a given study)		Concentration ranges (ng/g dw)			RCF (range)		TF (range)			Hydroponic[a], pot[b], field[c]
						A	B	C	B	C	A	B	C	
					PFOS	0.1–90.1	–	ND–0.08	–	–	0.2	–	–	
Celery		1	2	PFCA (7–9)	PFBA	–	< LOQ–80.8	433.2–517.8	17.3	133.5	–	1.4	0.8	[97] [b]
					PFHxA	–	7.2–137.1	18.4–19.9	4.8	16.2	–	1.3	1.1	[22, 36] [c]
				PFSA (3–5)	PFOA	–	2–111.3	75.4–218.2	1.4	2.7	–	0.3	0.3	
					PFBS	–	4.5–122.6	0.05–0.07	2.5	–	–	0.5	0.7	
					PFHxS	–	0.4–3.9	0.05–0.09	5	–	–	0.5	0.6	
					PFOS	–	17.2–209.8	0.07–0.11	4.2	1.8	–	0.2	0.6	
Cucumber		3	5	PFCA (1–13)	PFBA	–	17–63	0.1–13.6	–	–	–	–	–	[88, 98, 99] [b]
					PFHxA	–	ND–0.3	ND–1.4	–	–	–	–	–	[22, 37, 55, 66, 95] [c]
				PFSA (1–8)	PFOA	–	0.5–797	0.005–1.4	–	–	–	–	–	
					PFBS	–	3.8–15	ND–0.06	–	–	–	–	–	
				FASA (0–3)	PFHxS	–	0.2–0.3	ND–0.004	–	–	–	–	–	
					PFOS	–	ND–119	ND–0.06	–	–	–	–	–	
				FASAA (0–2)										
				FTCA (0–1)										
Zucchini	1		2	PFCA (9–13)	PFBA	< LOQ–2.6	–	1.5–2	–	–	0.6	–	–	[96] [a]
					PFHxA	< LOQ–9.6	–	0.03–0.1	–	–	1	–	–	[55, 95] [c]
				PFSA (3–5)	PFOA	< LOQ–28.9	–	ND–0.64	–	–	0.4	–	–	
					PFBS	< LOQ–19.0	–	ND	–	–	0.1	–	–	
				FASA (0–3)	PFHxS	< LOQ–19.8	–	ND	–	–	0.2	–	–	
					PFOS	< LOQ–81	–	ND	–	–	0.1	–	–	
				FASAA (0–2)										
				FTCA (0–1)										
Pumpkin	2		2	PFCA (2–13)	PFBA	< LOQ–0.6	–	< LOQ–638.1	–	–	0.6	–	–	[87, 100] [a]
					PFHxA	< LOQ–0.2	–	< LOQ–11.7	–	–	0.5	–	–	[36, 66] [c]
				PFSA (3–8)	PFOA	< LOQ	–	0.05–15.1	–	–	0.2	–	–	
					PFBS	< LOQ–0.05	–	< LOQ	–	–	0.4–0.5	–	–	
				FASA (0-1)	PFHxS	< LOQ–0.2	–	< LOQ	–	–	0.2–0.4	–	–	
				FTSA (0-1)	PFOS	< LOQ–15	–	< LOQ–0.09	–	–	0.2–0.3	–	–	
Pepper			5	PFCA (5–13)	PFBA	–	–	0.1–946.5	–	–	–	–	–	[22, 36, 37, 55, 95] [c]
					PFHxA	–	–	0.2–74.4	–	–	–	–	–	
				PFSA (3–5)	PFOA	–	–	0.007–39.3	–	–	–	–	–	
					PFBS	–	–	ND–0.2	–	–	–	–	–	
				FASA (0–3)	PFHxS	–	–	ND–0.005	–	–	–	–	–	
					PFOS	–	–	ND–0.2	–	–	–	–	–	
				FASAA (0–3)										

Table 2 (continued)

Plant	Study type (# of studies)			Results										References
	A	B	C	PFAS class (range studied within the class for a given study)		Concentration ranges (ng/g dw)			RCF (range)		TF (range)			Hydroponic[a], pot[b], field[c]
						A	B	C	B	C	A	B	C	
Eggplant			5	FTCA (1) PFCA (9–13) PFSA (3–8) FASA (0–3) FASAA (0–2) FTCA (1)	PFBA PFHxA PFOA PFBS PFHxS PFOS	–	–	0.5–4.5 ND–0.4 0.2–0.8 ND ND–0.001 ND	–	–	–	–	–	[22, 37, 55, 66, 95] [c]
Tomato	1	6	6	PFCA (9–13) PFSA (3–8) FASA (0–3) FASAA (0–2) FTCA (0–1)	PFBA PFHxA PFOA PFBS PFHxS PFOS	< LOQ–48 < LOQ–39.3 < LOQ–82.1 < LOQ–78.7 < LOQ–84.4 < LOQ–490	ND–121.9 ND–102.8 ND–190.2 ND–177.1 ND–7.7 ND–225.1	< LOQ–3.3 ND–1.4 ND–0.5 ND–0.3 ND–0.004 ND–0.2	5 1–1.4 1–4.4 0.7 1.8 0.2–4.5	– – – – – –	2.1 1.8 1.3 0.5 0.4 0.1	5.2–15.35 1.77–6.2 0.5–2.5 5.1 1.8 0.5–4.5	– – – – – –	[96][a] [28, 63, 88, 94, 97, 99] [b] [22, 37, 55, 63, 66, 95] [c]
Strawberry		2	2	PFCA (2–13) PFSA (2–4) FASA (0–2) FASAA (0–3) FASE (0–2) FTSA (0–1)	PFBA PFHxA PFOA PFBS PFHxS PFOS	–	< LOQ–11,200 < LOQ–1380 < LOQ–302 < LOQ–373 < LOQ–150 < LOQ–132	< LOQ < LOQ < LOQ < LOQ < LOQ–0.14 < LOQ	–	–	–	–	–	[88, 93] [b] [22, 101] [c‡]
Radish		4	3	PFCA (7–9) PFSA (3–5)	PFBA PFHxA PFOA PFBS PFHxS PFOS	–	< LOQ–13.7 7.3–44.5 8.1–2227 23.9–164.2 2.8–10.3 17–1250.3	1–1167.5 0.1–103.3 0.08–1879.8 < LOQ–0.5 ND–0.1 ND–1.8	2.9 1.2 0.8–3.0 1.3 2 0.7–2.6	10.8–15.2 3.1–14.2 0.7–1.4 NA NA NA–0.9	–	9.5 4.3 1.8–15.9 3.8 2.9 1.2–6.6	2.1–13.9 2.2–4.9 9.4–19.7 NA NA 0.6–30.8	[83, 84, 88, 97][b] [22, 36, 55] [c]
Carrot		4	4	PFCA (1–13) PFSA	PFBA PFHxA PFOA	–	36–130 3.1–144 ND–3567	0.02–865.8 ND–32.4 0.005–138.6	4.74 0.86 0.5–0.6	7.6–38.9 1.6–4.2 0.3–0.6	–	13.7 9.6–15.8 2.8–9.6	2.2–3.1 3.2–4.3 2.7–4.0	[25, 34, 94, 98][b] [22, 36, 55, 95] [c]

Table 2 (continued)

Plant	Study type (# of studies)			Results									References
	A	B	C	PFAS class (range studied within the class for a given study)		Concentration ranges (ng/g dw)			RCF (range)		TF (range)			Hydroponic[a], pot[b], field[c]
						A	B	C	B	C	A	B	C	
				(1–5) PFCA (1–13) PFSA (0–3) FASAA (0–2) FTCA (0–2) DiPAP (0–1)	PFBS PFHxS PFOS	– – –	– – 0.5–782	ND–1.0 ND–0.07 ND–1.73	– – 0.1–1.3	NA–3 NA–0.07 NA	– – –	– – –	NA–10.9 NA–1.4 NA–2.2	
Potato		2	2	PFCA (1–13) PFSA (1–5) FASA (0–3) FASAA (0–2) FTCA (0–1)	PFBA PFHxA PFOA PFBS PFHxS PFOS	– – – – – –	– – ND–234 – – ND–590	0.8[d] 0.006 [d] 0.01 [d] ND [d] 0.0008 [d] ND [d]	– – – – – –	– – – – – –	– – – – – –	– – – – – –	– – – – – –	[82, 98] [b] [22, 95] [c]

Additional PFAS classes studied for each plant type are reported PFAS class column. Plant species with only one study, or one study and studies by Bao et al. [37] or Scher et al. [22] which reported collective values by plant types are not listed. Spiked and biosolid amended pot studies are combined in category B. The PFAS class column indicates additional PFAS studied in reported plants; however, sufficient data were not available for comparisons across study types and within a given species for these additional PFAS

Variables reported include concentration ranges, root concentration factors (RCF), and translocation factors (TF)

< *LOQ* below method limit of quantitation, *ND* not detected, *NA* not applicable, *d* value estimate

plant uptake [104]. Thus, study design must be considered when interpreting plant uptake magnitudes and mechanisms.

Field studies on plant uptake of PFAS and the number of different PFAS analyzed have increased in recent years; eleven field studies are summarized here that were not previously reviewed [42•]. Two field studies noted a correlation between the level of short-chain PFAS taken up by crops and the volume of PFAS-contaminated irrigation water used [22•, 55]. For example, 60% of the variability in PFBA concentrations in home garden–grown produce was accounted for by differences in the volume of PFAS-contaminated water used to irrigate the gardens [22•]. For a variety of lab and greenhouse designed studies, PFCAs have been shown to translocate from plant roots to shoots more readily than PFSAs. However, field-scale translocation and soil–root transfer data are sparse; sampling rarely includes harvest of non-edible crop components. In cases where soil–root transfer was evaluated, PFAS contributions from atmospheric deposition confounded accurate TF and BCF comparisons, especially in plants grown near fluorochemical manufacturing. For example, field-grown lettuce at a site 0.3 km from fluorochemical manufacturing had up to 2370 ng/g PFBA and 1040 ng/g PFOA in lettuce and only 5 ng/g PFBA and 90 ng/g PFOA in soil [36], whereas lower concentrations in lettuce (no more than 270 ng/g PFBA and 200 ng/g PFOA) were observed in pot studies grown at similar soil PFAS concentrations (6 ng/g PFBA and 80 ng/g PFOA in soil at the time of harvest) [63]. Though costly, future field study designs should target inclusion of data that will enable estimations of BCFs and TFs to different plant domains to better ascertain uptake mechanisms and predict risks associated with accumulation in both human and animal edible components.

Studies that include additional PFAS classes as well as precursor and precursor degradation intermediates in plant studies are limited. A single study that included Cl-PFAESs found that the TF in wheat of 6:2 Cl-PFAES (a PFOS replacement) was similar to that of PFOS, while 8:2 Cl-PFAES had a lower TF, indicating that chain length effects occur within other PFAS classes similar to those observed for PFCAs and PFSAs [77•]. A single study addressed uptake of ultra-short-chain PFCAs (C2 and C3) [75•]. This study was with hydroponically cultivated wheat in which they found high uptake of C2 and C3 PFCAs into the roots and shoots due to multiple uptake mechanisms [75•]. As previously noted, EtFOSA-fortified soils produced a series of degradation products (FOSAA, FOSA, and PFOS) which increased when growing carrots [25]. Both the intermediate FOSA and terminal metabolite PFOS were found on the carrot peel as well as transported into the leaves along with small amounts into the carrot core whereas the precursor EtFOSA remained in the soil with only small amounts found on the peel [25]. Zhao et al. [108] exposed hydroponically cultivated wheat, soybean, and pumpkin to EtFOSA (1 mmol/mL) and proposed in vivo transformation of EtFOSA to FOSAA, FOSA, and PFOS in solution by microbes and further metabolic degradation in plant tissue to PFOS, PFHxS, and PFBS. Impurities in the EtFOSA standard lead to initial PFOS concentrations in the incubation solution; however, PFOS concentrations significantly increased in the presence of plant roots and associated microbes [108]. Similar results were observed for wheat with FOSA-fortified soils [51] and hydroponically grown soybeans and pumpkin [87] except that additional shorter-chain PFSAs (PFHxS and PFBS) were observed, particularly in the roots with some in the shoots, albeit at much lower concentrations than PFOS. PFOS is a known microbial degradation product of FOSA while PFBS and PFHxS are not [34]. The authors attributed the formation of short-chain PFSAs to a unique metabolic pathway. However, there are several noteworthy points to consider. First, the FOSA used was a technical grade (90–92% purity) with 8 mol% PFOS. Analysis targeted only FOSA, PFOS, and shorter-chain PFSAs and PFCAs; therefore, other impurities would have been missed such as shorter perfluoroalkyl chain sulfonamides, which would lead to PFSA degradation products of the corresponding chain length. In addition, in the case of wheat grown in both systems, higher levels of these short chains were found when grown in soils where microbial degradation would be higher [51]. Also in the hydroponic systems for soybean and pumpkin, small amounts of both PFBS and PFHxS were found in the plant controls (no FOSA added) [87]. Lastly, in order to form PFBS and PFHxS from FOSA, cleavage to produce a shorter fluorinated alkyl chain with reattachment of either a fluorine atom (if C-C cleavage occurred) or a sulfonate group (if C-S cleavage occurred) would have had to happen, which seems thermodynamically unlikely. Therefore, although the authors propose that production of PFBS and PFHxS from FOSA indicates a unique plant-mediated metabolism, it is much more plausible that these results are due to impurities in the technical FOSA used, for which additional support using oxidative stress measures was provided [87]. Enhanced degradation in the presence of plants and formation of additional metabolites relative to what is known to occur in microbially active soils have also been shown for 6:2 FTSA [87] and 8:2 diPAP [34]. Carrots enhanced 8:2 diPAP degradation and yielded additional PFCAs compared to what occurred in soil-only controls as well as what was observed with lettuce [34]. Whereas acid metabolites translocated, 8:2 diPAP remained primarily in plant portions in contact with the soil (e.g., carrot, lettuce heart) [34]. Therefore, plant-mediated degradation of 8:2 diPAP was likely due to carrot exudates rather than degradation within the plant. In 6:2 FTSA exposure to hydroponically grown pumpkin, series of PFCAs (C2–C7) were produced with the C2 and C3 acids not present in root exudates controls; therefore, the authors concluded some 6:2 FTSA biotransformation was occurring within the plant. Such complexities can complicate interpretation of PFAS-specific uptake.

Plant Uptake Mechanisms

Plant mechanisms involved in chemical uptake are plant-specific, but must first involve initial transfer to the root and adsorption to apoplastic plant tissue followed by translocation through the xylem in the central vasculature to the shoot and remainder of the plant. It is widely accepted that the Casparian strip, which is located in the root endodermis and made up of lignified structures that regulate water and mineral uptake by the roots, also serves to constrain translocation of contaminants from roots to the central vasculature, thus the rest of the plant [109]. The Casparian strip prevents passive transport of compounds across the endodermis, by sealing the nonliving spaces between cells and cell membranes (apoplast) [110]. The suberin lamella is an additional selective barrier that is formed in some plants by covering the endodermal cells with the hydrophobic polymer suberin [109].

Trends discussed in PFAS plant uptake studies are consistent with this hypothesis that the Casparian strip constrains translocation of long-chain PFCAs, PFSAs, FOSAs, and Cl-PFAESs, thus resulting in TF < 1 for these PFAS. Felizeter et al. [91] hypothesized that transfer of long-chain PFAS was inhibited by the Casparian strip while the more soluble shorter chains were transported with water into the foliage by comparing RCF, FRCF, and TSCF values for C4-C14 PFCAs and C4-C8 PFSAs in hydroponic studies with lettuce and in light of sorption to root isotherms. They showed that both RCFs and TSCFs had U-shaped curves with the minimum occurring with C7 PFCA while FRCFs consistently decreased with increasing chain length. They also showed that sorption to root isotherms were only linear for C4-C6 PFCAs and PFBS while all longer-chain PFAS exhibited nonlinear isotherms. Findings from hydroponic studies may not transfer well to studies with soils due to the strong chain-length–dependent interactions of PFAS with soils, thus affecting RCFs and subsequent translocation. Müller et al. [105•] further investigated this U-shaped curve phenomenon by measuring uptake, depuration, and translocation kinetics for C4–C10 PFAAs in hydroponically cultivated *Arabidopsis thaliana* in which they evaluated uptake kinetics to root and shoot independently. They also evaluated sorption to dead roots which they used to calculate sorption normalized concentration factors (SNCFs) to better differentiate between physical adsorption and active uptake. They observed preferential uptake of PFBA followed by slow and incomplete depuration. Thus, they proposed a 2-compartment uptake process occurs in which PFBA uptake by the root symplast is reversible whereas uptake by one or more compartments is not reversible, which may involve the central vasculature and/or organelles and be influenced by active transport from the roots [105•]. Other hydroponic studies with wheat also saw enhanced translocation of shorter-chain PFBA compared to the longer-chain PFAAs [75•, 78]. Different transport inhibitors were used to elucidate active transport mechanisms in hydroponically grown wheat exposed to C2–C8 PFAAs [75•] and in an Asian wetland plant (*Alisma orientale*) exposed to PFOA and PFOS [111]. Both studies observed Michaelis-Menten type uptake behavior and when coupled to the effects of inhibitors concluded that PFAA uptake is an energy-dependent active uptake process mediated by carrier proteins. Depending on the specific PFAA and plant species, transport of these carriers could involve anion channels and/or aquaporin channels [75•, 111]. For chicory grown hydroponically, Gredelj et al. [102•] did not observe a pronounced U-shape trend in PFAA (C4–C10) uptake behavior, which was attributed to using higher PFAA exposure concentrations than comparable studies [91, 105•]. Gredelj et al. [102•, 104] also exemplified that chain-length–dependent uptake patterns observed hydroponically may not compare directly to soils grown in soils due to PFAS-soil interactions.

Several studies indicate that root proteins and lipids may influence uptake and bioaccumulation of PFAS in plant tissues [76, 83, 84, 108, 111]. Wen et al. [83, 84] found positive correlations between PFOS and PFOA RCFs and root protein content [83] as well as between TFs and shoot to root protein ratios [84] across seven plant species (alfalfa, lettuce, maize, mung bean, radish, ryegrass, and soybean). Likewise for the same plants, Wen et al. [83] found that although root protein content could account for 55% of the variations in N-EtFOSAA plant concentrations, when combined with lipid content, 85% of the variation across species was predicted. This was in contrast to PFOS and PFOA in which RCFs were negatively correlated to lipid content. In hydroponic studies, H. Zhao et al. [76] attributed increasing root accumulation with increasing PFCA chain length to sorption to root lipids or membrane proteins resulting in lower translocation of the longer-chain PFCAs (e.g., PFDoA) compared to the shorter-chain PFCAs. Zhao et al. [108] found that RCFs for N-EtFOSA increased with increasing root lipid content of hydroponically grown pumpkin, wheat, and soybean. Wang et al. [111] demonstrated that PFOS and PFOA partitioned differently into the subcellular plant components of the hydroponically grown wetland plant *Alisma orientale*, with PFOA residing primarily in the water-soluble fractions of stems, roots, and leaves and PFOS residing primarily in cell walls of roots and leaves and the water-soluble fraction of stems. PFOA and PFOS increased in the cell walls (anionic) and organelles (15–30% lipid) of roots, stems, and leaves with time, and reached an approximate equilibrium after 8 days of exposure [111]. When plants were transplanted into new PFAS-free culture solutions, approximately 80% of PFAS originally taken up was released [111]. These results exemplify that PFAS properties impact uptake to various plant components as well as differences in the subcellular distribution within a plant component.

Contaminants other than PFAS can invoke responses by the Casparian strip and suberin lamellae. For example, maize

roots responded to Cd exposure by developing suberin lamella, which along with the Casparian strip can inhibit the translocation of Cd and other contaminants to the xylem and the remainder of the plant [112]. Co-contamination of PFOS or PFOA with Cd decreased the bioaccumulation of these PFAAs in both wheat and rapeseed; however, the PFAA TFs decreased in wheat only and increased in rapeseed. Interestingly, both the bioaccumulation and TFs of Cd were increased in both plant types in the presence of these PFAAs [81]. Although the authors hypothesized that the cause of co-contamination effects was due to impacts on sorption to soil, thus soil solution concentrations, this is not well supported given the inconsistent data trends. It is more likely that Cd invoked changes in the plant. Wang et al. [113] observed an increased presence of suberin lamella or inactivation of aquaporin channels in response to Cd, which may also decrease PFAS translocation [81]. Additionally, potential PFAS-induced damage to the barriers in root structure at high concentrations (200 mg/L) [78] may allow for increased uptake. Therefore, future studies should consider plant properties that may impact PFAS uptake in addition to sorption dynamics.

Identifying and differentiating PFAS source contributions especially in field studies can be difficult as previously noted. PFAS in irrigation water are readily bioavailable for plant uptake, as demonstrated by Gredelj et al. [104] who showed chicory irrigated with PFAA-containing water having an increased uptake of long-chain PFAAs into the edible heads whereas PFAA soil concentration had a stronger impact on PFAA root concentrations. Seo et al. [114] attributed higher concentrations in plants than expected from soil transfer alone to atmospheric deposition [36]. Seo et al. [114] showed that PFCAs, neutral PFAS, and PFBS commonly partition to the gas phase, while PFOS tends to partition onto airborne particles. Gaseous uptake by tree bark from multiple species in China was identified as the main uptake mechanism for short-chain PFAS, which associated well with high lipid content [115]. In contrast, the presence of long-chain PFAS was attributed to airborne particles being trapped in bark fissures given the lower concentrations in the smooth bark species [115]. The influence of particles and surface texture on PFAS concentrations may also be responsible for higher PFAS concentrations in potato peel than flesh [42•]. For example, recent work showed the presence of PFAS-containing soil mineral derived micron-sized particles on the surface of potatoes even after washing [95]. Therefore, while field studies are critical to bounding PFAS occurrence levels in plants, the multiple variables and variations that exist in field-scale studies confound data interpretation targeted at understanding uptake mechanisms.

Metabolic Responses and Phytotoxicity

PFAS presence and uptake can elicit a metabolic response in plants [21, 75•, 100, 116] which may include chemical modification, conjugation, and compartmentation. Superoxide dismutase (SOD), peroxidase (POD), catalase (CAT), glutathione (GSH), glutathione-S-transferase (GST), cytochromes P450 (CYP450), and malondialdehyde (MDA) are enzymes and metabolites that collectively serve as a defense against damage from superoxide radicals and reactive oxygen species (ROS) by preventing their formation or destroying those formed. PFAS exposure induces plant responses including enzymatic activity [78, 81, 87, 100, 103, 116] and metabolite production [21, 90] that may play a role in reducing phytotoxic impacts of PFAS within plants and/or promote in vivo degradation [87, 100].

Metabolic and phytotoxic responses were reported in crops and produce. Co-exposure of PFOS and PFOA caused several inorganic plant elements (Na, Mg, Cu, Fe, Ca, Mo, and K) to decrease in lettuce leaves while Zn increased, which may be due to ROS-induced ion channel leakage [21]. Also in lettuce, PFOS and PFOA altered amino acid metabolites, dipeptides, fatty acids, lipids, phytol, purine nucleosides, phenolic antioxidants, and flavonoids, as a result of stress responses to DNA injury, photosynthesis inhibition, and impact to energy metabolism [90]. PFAS exposure significantly increased chlorophyll, MDA, and POD, but slightly decreased SOD in rapeseed [81]. Co-contamination with Cd led to PFAS stress responses including decreased urease activity, POD activity, and biomass in rapeseed. In wheat, chlorophyll and MDA inhibition occurred [81]. Also, wheat shoot and root biomass decreased at higher PFAA exposure concentrations (2000 ng/g) while root biomass increased at lower exposure concentrations (200 ng/g) [79]. Additionally in wheat, exposures of PFOS at higher concentrations (100–200,000 μg/L) invoked changes in chlorophyll, protein synthesis, growth, SOD, and POD [78]. Chlorophyll *a* and chlorophyll *b* increased with exposure to long-chain PFAAs while exposure to short-chain PFAAs led to decreases in chlorophyll *a* [79]. Pumpkin and soybean exposure to FOSA increased SOD (soybean 19.2–30.8%, pumpkin 39.2–92.8%) and POD (soybean 19.2–20.7%, pumpkin 21.1–37.6%), likely in response to increased production of ROS [87]. GST and CYP450 also increased with FOSA exposure. Both enzymes are known to play a role in detoxification of contaminants, and thus may have also played a role in plant-mediated FOSA degradation [87]. CYP450 activity increased in pumpkin exposed to 6:2 FTSA and when CYP450 activity was reduced by addition of a CYP450 oxidation inhibitor (1-aminobenzotriazole), PFCA metabolites from 6:2 FTSA degradation decreased, supporting plant-mediated biotransformation [100]. Thus, metabolic responses may lead to some oxidative degradation of PFAS, which will be plant and PFAS dependent. However, stress

responses at the field-scale when irrigating fields or land applying waste-derived fertilizers have not been reported or observed. This is likely due to environmental concentrations typically being lower than those used in lab-based studies, the positive attributes of the amendments to the plants, and their ability to compensate for chemical-induced stress.

Conclusions and Data Gaps

The PFAS list previously considered to be of interest in soil-plant transfer studies has expanded from select PFAAs (mainly PFOS, PFOA, PFBA, and PFBS) to include PFAS precursors, additional PFAAs, and additional PFAS classes (FASAs, Cl-PFAESs, FASAAs, diPAPs, FTSAs, and FTCAs). Of the PFAS discussed, side-chain fluorinated polymer surfactants, diPAPs, FASAAs, and FTSAs, were identified in waste-derived fertilizers or fields treated with waste-derived fertilizers. Cl-PFAESs should also be considered, especially in China [12, 77]. Future field studies on plant accumulation would benefit from including either nontarget PFAS screening and/or utilizing non-specific tests to better evaluate the level of precursors present that may serve as a PFAA reservoir. Degradation rates and pathways (intermediates on the way to PFAAs) in agriculturally relevant soils and conditions are lacking for many precursors and those for recently identified are completely unknown [30, 33•]. Such studies are needed to elucidate the true potential for these precursors to contribute to PFAS intermediates and PFAAs becoming available in the root zone for uptake during the life of the plant. Field studies should also expand upon the number of PFAS quantified as increasingly more standards are becoming available. Future studies on PFAS in agricultural soils in both greenhouse and field studies should include sufficient details on soil and soil amendment properties that can influence PFAS mobility, and thus plant availability [60].

Land application of PFAS-containing media results in plant uptake of short-chain PFAS into plant components that are consumed by humans, utilized as fodder, or land-applied. Long-chain compounds typically sorb to or are taken up into plant roots and largely remain in the roots while short-chain compounds have a higher propensity to translocate into plant shoots. Uptake varied between species and within species and was associated with different plant properties including root lipids and proteins. However, more research is needed to make further inferences on the effects of plant characteristics on uptake. Uptake was shown to be an active carrier protein-mediated process in wheat, which contrasts with frequently cited passive transport processes that are constrained by the Casparian strip. Plant uptake depends on how long PFAS reside in the root zone and on PFAS concentrations. Therefore, the use of high concentrations and growing conditions that minimize leaching could explain why plants grown in pot studies generally have higher concentrations of short-chain PFAS than have been observed in field studies. However, when plants are grown in fields close to fluorochemical manufacturing plants, higher concentrations than predicted from PFAS concentrations in soils may be observed due to atmospheric deposition or irrigation with PFAS-contaminated water. Likewise, comparisons made between hydroponic and soil-based studies must take into consideration the impacts of soil-based sorption, air–water interface sorption variability, porewater available compounds in soil, and morphological growth differences (root structure and surface area). Lastly, in some cases, plant enzymes were found to enhance degradation rates of precursors and increase the yields of the more bioavailable PFAAs.

Knowledge on mechanisms of soil–plant transfer, phytotoxicity, and PFAS metabolism in plants is growing and can inform future decision-making regarding cultivation on PFAS-laden soils. Similar to how lower PFAS levels are found in agricultural fields than pot studies, phytotoxic effects were observed at exposure concentrations that are generally considerably much higher than those observed in field studies. Risk of PFAS exposure from plant uptake is multifaceted [38] and encompasses not only plants that are directly consumed, but also through the food web (e.g., bioaccumulation in livestock feeding on PFAS contaminated water, forage, and feed) [117]. Field studies that demonstrate plant uptake, fate of PFAS, and risks associated with land-applied media containing PFAS are needed to inform farmers, landowners, and policymakers on how farming practices will impact the food supply chain.

Future studies should aim to:

- Investigate paper mill waste, industrial byproducts, recycled water, and pesticides as potential sources of PFAS in agricultural systems and their impact on source water in agricultural operations
- Conduct additional field studies, particularly those involving land application of typical waste-derived fertilizers or treated effluent
- Include a larger suite of PFAS for quantification consistent with current knowledge as well as analytical approaches that identify allow insight into additional PFAS including precursors that may serve as a long-term release reservoir of more mobile PFAS
- Investigate soil–plant transfer of replacements compounds, e.g., GenX, ADONA, and others as they become known
- Include more realistic PFAS concentrations in studies designed to improve our understanding of plant-specific uptake mechanisms and plant responses to PFAS exposure
- Include studies that improve our ability to quantify the role of plant enzymes on enhanced precursor degradation in the rhizosphere and within the plant

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Compliance with Ethical Standards

Conflict of Interest On behalf of all authors, the corresponding author states that there is no conflict of interest.

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Landfill leachate treatment process is transforming and releasing banned per- and polyfluoroalkyl substances to UK water

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Landfills are a known source of PFAS pollution. Many have environmental permits allowing the discharge of treated leachate to controlled waters. In this article we compared leachate data for 17 PFAS from 17 different landfill sites across the UK. The results show that the landfill leachate treatment process (designed to improve water quality) is generating the banned PFAS; PFOA and PFOS. Approximately 80% of locations tested showed an increase in PFOS, with an increase of 1,335% in one sample. The highest concentrations of PFOS and PFOA in treated leachate were 2,460 ng L^{-1} and 26,900 ng L^{-1}, respectively. When compared against the environmental quality standard of 0.65 ng L^{-1} for PFOS this leachate could pose a significant concern. Landfill leachate treatment has proven effective for a wide range of different pollutants, but this research shows in several instances that it is not appropriate for PFAS remediation.

KEYWORDS

per- and polyfluoroalkyl substances, PFAS, PFOS, PFOA, landfill leachate, waste water treatment, effluent

1 Introduction

Per- and polyfluoroalkyl substances (PFAS) are persistent environmental pollutants of global concern. Perfluorooctanoic acid (PFOA) and perfluorooctanesulfonic acid (PFOS) were the first two PFAS to be listed under the Stockholm convention (Secretariat of the Stockholm Convention (UNEP), 2008). There is currently widespread contamination of PFOS and PFOA which is impacting the marine environment, freshwater environments and drinking water (Kurwadkar et al., 2022). PFOS and PFOA are not readily removed by conventional drinking or waste-water treatment processes, and growing evidence has shown that PFOA and PFOS can be generated from precursor compounds in chemical and biological processes. Xiao et al. (2012) identified a significant increase (up to 1,200%) in the wastewater concentrations of perfluorohexanoic acid (PFHxA) and PFOA at 22 out of 37 wastewater treatment plants. Similar increases have also been observed in other biological wastewater treatment plants (Guo et al., 2010; Schultz et al., 2006; Takagi et al., 2011; Xiao et al., 2012), and during the drinking-water disinfection processes (Appleman et al., 2014; Boiteux et al., 2017; Dauchy et al., 2012). Increases of 18–77% of the mass of perfluoroalkyl carboxylic acids PFCAs, including PFOA, have been reported after disinfection of ozone or chlorine. The authors concluded that this was caused by the transformation of unidentified precursors in surface water and other legacy precursor compounds (Boiteux et al., 2017).

These increases in PFAAs have been identified in a wide number of different treatment processes all over the globe. A survey of 15 U.S. water treatment plants, identified that the concentration of PFOA and PFOS in water was consistently higher after chemical disinfection treatments (Appleman et al., 2014). Similarly, increases in PFOA and PFOS concentrations in drinking-water processes have been reported in Japan, which was attributed to the transformation of precursor compounds (Takagi et al., 2011). The extremely strong carbon-fluorine bond makes PFAS inherently persistent to the biological treatment, with studies showing that the biological landfill leachate treatment process also resulted in elevated total PFAS concentrations in leachate after aeration (Fuertes et al., 2017; Solo-Gabriele et al., 2020). This indicates biotransformation of precursor compounds [such as fluorotelomer alcohols (FTOHs), fluorotelomer sulfonates (FTSs) and polyfluoroalkyl phosphate esters (PAPs)] in leachate to produce perfluoroalkyl acids (PFAAs) (Liu et al., 2021).

While this process has been studied in some global regions, to the best of our knowledge it has never been investigated or reported in the UK. The UK has a long industrial history where a large proportion of chemical waste was disposed of in landfill sites. Many of these are historic landfills, where, according to the Environment Agency definition, there is no pollution prevention and control permit or waste management license currently in force (Gov.UK, 2024). Climate change is resulting in more severe weather events and as a result the UK is experiencing increased frequency and severity of rainfall and coastal erosion. This is already putting pressure on these landfill sites, with nearly one in ten historic coastal landfill sites in England already at risk of erosion (Brand et al., 2018). Many of these historic landfill sites are also located within drinking water protection zones (Brand and Spencer, 2023) and so it is imperative that we understand the potential risk of PFAS from our industrial legacy.

Once a landfill operator in the UK has treated the leachate, they are legally allowed to discharge their waste directly into controlled waters via a Discharge to Water permit, this includes hazardous landfills (Gov.UK, 2024). Given that the treatment process has the potential to generate banned PFAS it is imperative we understand these risks further.

2 Methodology

In this study we evaluated leachate data from 17 landfill locations across the UK to identify if they are likely to be a significant source of PFAS to the environment. All data obtained for this report was obtained by an ENDS Report Freedom of Information Request. The data was collected by third-party contractors and formed part of a UK government analysis into landfill leachate for PFAS as part of a wider DEFRA-funded research project on persistent organic pollutants (Environment Agency, 2021). The data collected came from a range of operational, closed and historical landfills in England, they are predominantly classified as non-hazardous, accepting municipal solid wastes. All samples were collected between 2021–2022. The locations of the landfill sites and sample collection dates were not provided, we were informed this was because the contractors provided the Agency with an anonymized dataset. Where on-site leachate treatment facilities were available, raw and treated leachate (final effluent) samples were collected. The treatment facilities used biological, chemical, and physical technologies, the most commonly used in the UK landfill industry is biological sequence batch reactors (SBR) with alkaline-dosing. As the data set was anonymized we were unable to identify which technology was used at each location.

In total we obtained results for 48 samples. This included 32 raw samples and 16 treated samples from 17 different landfill sites, plus three blank samples. Duplicate samples were obtained from 4 locations (Landfills 1, 2, 6, & 11). Samples were prepared using solid phase extraction and quantified using isotope labeled internal standards, with analysis by HPLC-MS/MS (high pressure liquid chromatography coupled to a triple quadrupole mass spectrometer). Results were reported for 17 PFAS (perfluorooctyl sulphonate (PFOS), perfluorooctanoic acid (PFOA), 5: 3 fluorotelomer carboxylic acid (5, 3 FTCA), 6: 2 fluorotelomer sulphonate (6, 2 FTSA), 8: 2 fluorotelomer alcohol (8, 2 FTOH), perfluoropentyl sulphonate (PFPeS), perfluoroheptyl sulphonate (PFHpS), perfluoropentanoic acid (PFPA), perfluoroheptanoic acid (PFHpA), hexafluoropropylene oxide-dimer acid (HFPO-DA) (Gen-X), F53B (two components), perfluorobutyl sulphonate (PFBS), perfluorobutanoic acid (PFBA), perfluorononanoic acid (PFNA), perfluorohexanoic acid (PFHxA), perfluorohexyl sulphonate (PFHxS), perfluoroethylcyclohexane sulphonate (PFECHS)) with limits of detection of 0.065 ng L^{-1} for PFOS and PFOA. Data is presented in the Supplementary information.

As the data was obtained from a variety of different landfill sites, we urge caution comparing data between different locations. However, in this study we are focused on identifying overall trends and identifying relative trends in raw and treated leachate. These paired raw and treated samples were analyzed at the same time which facilitates this type of data analysis.

3 Results and discussion

Concentrations of all 17 PFAS in the raw leachate ranged from 1 ng L^{-1} to 87,100 ng L^{-1} (Table 1). The highest concentration for any PFAS in a treated sample was for perfluorobutyl sulphonate (PFBS), which was detected at a concentration of 44,800 ng L^{-1}. PFOS concentrations in the raw leachate samples ranged from 16–320 ng L^{-1} and 15.3–2,460 ng L^{-1} in the treated samples. These concentrations greatly exceed the current environmental quality standards (EQSs) for PFOS. The annual average EQS for inland surface waters is 0.65 ng L^{-1} and the annual average EQS for other surface waters is 0.13 ng L^{-1} (Environment Agency, 2019). The EU Water Framework Directive applies an EQS of 0.65 ng L^{-1} for PFOS in (inland) surface waters (European Union, 2013), with a threshold of 4.4 ng L^{-1} proposed for a group of 24 PFAS in surface and groundwater (European Environmental Bureau, 2023). In England, the Drinking Water Inspectorate's current guidance states that in England the acceptable levels of the two most common types of PFAS - PFOS and PFOA - is 100 ng L^{-1}. In the U.S. the proposed legal limit is 0.004 ng L^{-1} for PFOA and 0.0 2 ng L^{-1} for PFOS. There are currently no guidelines for other PFAS chemicals. There is currently no guidance in England on PFBS, although animal studies following oral exposure to PFBS have shown health effects on the thyroid, reproductive organs and tissues, developing fetus, and kidney (U.S. Environmental Protection Agency, 2024).

Data was reported for 17 Landfill sites, but only 14 locations had data from both raw and treated leachate. This paired data was used to provide a broad assessment of the effectiveness of 14 sites to reduce PFAS concentrations. Only one location (landfill 11) displayed lower concentrations of every PFAS in the treated samples relative to the raw samples. The majority of locations displayed a relative increase in

TABLE 1 Average concentrations and relative difference of the 17 PFAS in raw and treated leachate samples across the 14 paired locations.

		Landfill 1	Landfill 2	Landfill 3	Landfill 4	Landfill 5	Landfill 6	Landfill 7	Landfill 8	Landfill 9	Landfill 10	Landfill 11	Landfill 12	Landfill 13	Landfill 14	Number of locations with an increase	Number of locations with a decrease
PFOS	Raw	193	168	84	83	16	47	169	274	70	317	232	129	320	52		
	Treated	478	258	268	125	15	70	718	342	56	2,460	157	302	603	752	11	3
	x Increase	2.5	1.5	3.2	1.5	1.0	1.5	4.2	1.2	0.8	7.8	0.7	2.3	1.9	14.4		
PFOA	Raw	737	940	15,500	545	10	59	1,310	1,280	538	1,310	2,183	1,014	2,250	533		
	Treated	1,040	1,295	26,900	635	9	44	1,230	862	140	3,670	1,030	1,350	2,350	102	7	7
	x Increase	1.4	1.4	1.7	1.2	0.9	0.7	0.9	0.7	0.3	2.8	0.5	1.3	1.0	0.2		
5:3 FTCA	Raw	193	3,942	5,550	2,770	*0.001*	3.0	1790	1940	3,710	460	3,733	3,428	6,390	1,250		
	Treated	478	13	86	*0.001*	*0.001*	*0.001*	14	92	4.0	750	15	96	41	30	2	11
	x Increase	2.5	0.0	0.0	0.0		0.0	0.0	0.0	0.0	1.6	0.0	0.0	0.0	0.0		
6:2 FTSA	Raw	116	489	532	227	*0.001*	5.0	128	344	129	300	198	1,202	244	291		
	Treated	111	74	1,030	12	*0.001*	1.0	4.0	92	3.0	1800	109	108	101	94	2	11
	x Increase	1.0	0.2	1.9	0.1		0.2	0.0	0.3	0.0	6.0	0.6	0.1	0.4	0.3		
8:2 FTOH	Raw	*0.001*	*0.001*	*0.001*	*0.001*	*0.001*	*0.001*	*0.001*	*0.001*	*0.001*	*0.001*	*0.001*	*0.001*	1,630	1,130		
	Treated	*0.001*	*0.001*	*0.001*	*0.001*	*0.001*	*0.001*	*0.001*	*0.001*	*0.001*	*0.001*	*0.001*	*0.001*	*0.001*	*0.001*	0	2
	x Increase													0.0	0.0		
PFPeS	Raw	41	123	48	25	*0.001*	8.0	59	47	71	99	88	35	73	60	11	2
	Treated	257	153	103	149	*0.001*	4.0	465	223	86	100	42	157	241	82		
	x Increase	6.3	1.2	2.1	6.0		0.5	7.9	4.7	1.2	1.0	0.5	4.5	3.3	1.4		
PFPHpS	Raw	3.1	8.3	*0.001*	*0.001*	*0.001*	2.0	9.0	10	3.0	17	5.0	*0.001*	14	*0.001*	9	2
	Treated	14	*0.001*	*0.001*	11	*0.001*	4.0	28	14	5.0	89	*0.001*	17	18	*0.001*		
	x Increase	4.5	0.0		11,000		2.0	3.1	1.4	1.7	5.2	0.0	17,000	1.3			
PFPA	Raw	648	662	868	388	*0.001*	9.0	551	676	707	713	800	816	1,350	725	10	3
	Treated	1,680	859	1,210	856	*0.001*	22.5	1,460	1960	235	529	334	1,510	1,510	866		
	x Increase	2.6	1.3	1.4	2.2		2.5	2.6	2.9	0.3	0.7	0.4	1.9	1.1	1.2		
PFPHpA	Raw	336	408	624	256	*0.001*	18	607	572	398	463	459	310	695	362	9	4
	Treated	622	490	715	335	*0.001*	16.5	640	693	109	478	178	559	654	474		
	x Increase	1.9	1.2	1.1	1.3		0.9	1.1	1.2	0.3	1.0	0.4	1.8	0.9	1.3		

(Continued)

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TABLE 1 (Continued)

		Landfill 1	Landfill 2	Landfill 3	Landfill 4	Landfill 5	Landfill 6	Landfill 7	Landfill 8	Landfill 9	Landfill 10	Landfill 11	Landfill 12	Landfill 13	Landfill 14	Number of locations with an increase	Number of locations with a decrease
HFPO-DA (Gen-X)	Raw	*0.001*	*0.001*	*0.001*	*0.001*	*0.001*	1.0	2.0	1.0	1.0	*0.001*	*0.001*	*0.001*	*0.001*	*0.001*	1	1
	Treated	*0.001*	*0.001*	*0.001*	*0.001*	*0.001*	1.0	*0.001*	2.0	1.0	*0.001*	*0.001*	*0.001*	*0.001*	*0.001*		
	x Increase							0.0	2.0								
F53B	Raw	0.11	*0.001*	*0.001*	*0.001*	*0.001*	1.0	1.0	2.0	1.0	*0.001*	*0.001*	*0.001*	*0.001*	*0.001*	2	3
	Treated	*0.001*	*0.001*	*0.001*	*0.001*	*0.001*	1.5	2.0	*0.001*	*0.001*	*0.001*	*0.001*	*0.001*	*0.001*	*0.001*		
	x Increase	0.0					1.5	2.0	0.0	0.0							
PFBS	Raw	2,140	5,690	24,400	8,100	*0.001*	17	7,740	16,200	2,370	8,850	3,867	29,083	20,600	10,500	9	4
	Treated	18,200	14,200	25,000	9,370	*0.001*	21	10,800	29,400	2,280	6,930	1710	44,800	11,500	10,900		
	x Increase	8.5	2.5	1.0	1.2		1.2	1.4	1.8	1.0	0.8	0.4	1.5	0.6	1.0		
PFBA	Raw	823	4,603	3,010	1940	29	62	1820	1,580	1,080	2,700	1806	1,346	2,850	1,190	7	7
	Treated	3,190	2,150	2,970	2,160	22	61	2,310	2030	395	2,840	871	2090	2,800	1,550		
	x Increase	3.9	0.5	1.0	1.1	0.8	1.0	1.3	1.3	0.4	1.1	0.5	1.6	1.0	1.3		
PFNA	Raw	34	26	47	20	*0.001*	*0.001*	38	87	9	15	36	33	70	17	9	3
	Treated	54	33	76	29	*0.001*	*0.001*	82	43	3	87	17	60	126	52		
	x Increase	1.6	1.3	1.6	1.5			2.2			5.8		1.8	1.8	3.1		
PFHxA	Raw	1,443	1,743	3,240	1,080	13	78	2,320	2010	1880	1810	1806	1,688	4,190	1,350	7	7
	Treated	2,330	1,880	3,400	1,140	12	32	2080	2,150	572	1860	512	2,430	2,800	1,130		
	x Increase	1.6	1.1	1.0	1.1	0.9	0.4	0.9	1.1	0.3	1.0	0.3	1.4	0.7	0.8		
PFHxS	Raw	377	427	188	319	*0.001*	25	452	589	308	969	338	247	576	316	9	3
	Treated	726	362	321	407	*0.001*	37	738	686	141	1,050	140	566	952	316		
	x Increase	1.9	0.8	1.7	1.3		1.5	1.6	1.2	0.5	1.1	0.4	2.3	1.7			
PFECHS	Raw	50	36	16	26	18	62	39	13	4	108	13	41	284	12	8	6
	Treated	69	33	17	33	21	99	20	7	*0.001*	209	13	62	142	*0.001*		
	x Increase	1.4	0.9	1.1	1.3	1.2	1.6	0.5	0.5	0.0	1.9	1.0	1.5	0.5	0.0		

Red shading indicates a relative increase, green shading indicates a relative decrease, grey shading indicates no change. Values in italics represent limit of detection.

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straight chain PFAAs, and a relative decrease in branched PFAS. There were more instances where higher concentrations of PFAS were detected in treated leachate than raw leachate for PFOS, PFPeS, PFPHpS, PFPA, PFPHpA, PFBS, PFNA, PFHxS, and PFECHS. The branched PFAS 5: 3FTCA, 6: 2FTCA and 8: 2FTOH were the only PFAS with more locations displaying a relative decrease in concentrations (Table 1). These results indicate that relative increases in straight chain PFAAs is likely due to precursor transformation during the treatment process (Fuertes et al., 2017; Solo-Gabriele et al., 2020; Liu et al., 2021). This transformation can occur slowly in the environment but appears to be drastically accelerated during treatment at some locations.

When the data is investigated it reveals many instances where the treatment process appears to be increasing the concentrations of the banned PFAS (PFOA and PFOS) (Table 1). Eleven (11) of the 14 paired samples displayed a higher concentration of PFOS in the treated leachate than the raw leachate (Figure 1). In 9 of these samples the PFOS concentrations increased by over 50% and in 6 of the samples the PFOS concentrations increased by over 100%. A paired T-test revealed concentrations of PFOS were significantly higher in the paired treated samples. The results reveal some of the highest ever reported relative potential increases in PFOS concentrations. The concentration in Landfill 14 increased by 1,335%, from 52.4 ng L^{-1} to 752 ng L^{-1}, and in Landfill 10 by 676%, from 317 ng L^{-1} to 2,460 ng L^{-1} (Figure 1).

Seven (7) of the 14 paired samples displayed a higher concentration of PFOA in the treated leachate than the raw leachate (Figure 2). In six (6) of these samples the PFOA concentrations increased by over 50% and in two (2) of the samples the PFOA concentrations increased by over 100%. The highest relative increases were observed in Landfill 10 which displayed a potential increase of 180% from 1,310 ng L^{-1} in the raw sample to 3,670 ng L^{-1} in the treated sample (Figure 2). Despite these cases concentrations of PFOA in treated samples were not statistically significantly higher in the paired samples.

Relative increases in the straight chain PFAAs could not be explained by relative decreased in the 3 branched PFAS alone. As only targeted analysis was performed on the data it was not possible to ascertain what other PFAS were present in the raw leachate, or what degradation products are being generated from the treatment process. These results indicate that there are likely to be many more PFAS in these samples and so further analysis using total PFAS methods and non-targeted analysis would be recommended. Wastewater treatment plants can be effective at improving water quality for a wide variety of pollutants. However, our snapshot of data from UK treatment plants indicates that they are not effective at removing PFAS, and in many instances are generating the banned PFAS (PFOA and PFOS) through precursor transformation. This data was gathered from a targeted monitoring program however, future monitoring efforts would benefit from complimentary non-targeted and total PFAS methods to gain a more complete picture of what is entering our waterways.

4 Conclusion

The results of our study paint a concerning picture for the effectiveness of current efforts to treat PFAS in landfill leachate. As the locations of the landfill sites were not disclosed it is not possible to comment on the direct impact to the environment. However, the highest concentrations of PFOS and PFOA in treated leachate were 2,460 ng L^{-1} and 26,900 ng L^{-1}, respectively. When comparing these against the environmental quality standard of 0.65 ng L^{-1} for PFOS in inland waters, it would indicate that if this treated leachate was to be discharged directly to the environment it would pose a significant

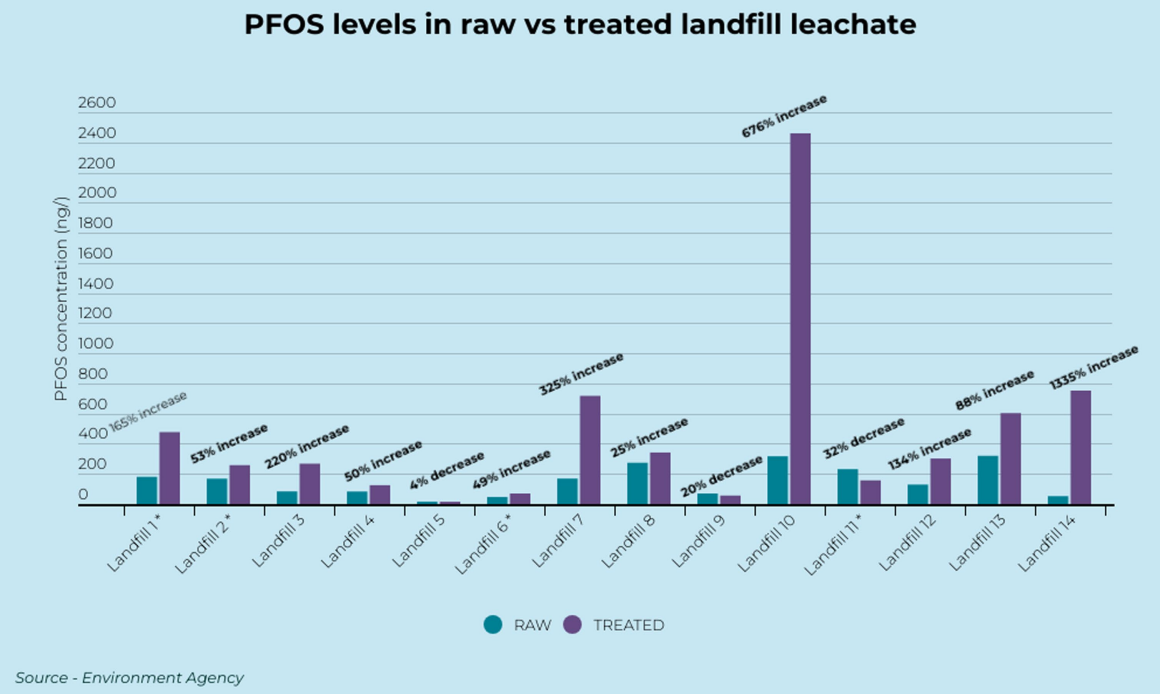


FIGURE 1
PFOS concentrations in raw and treated landfill leachate. *Represents average value for landfills with multiple treated or raw samples.

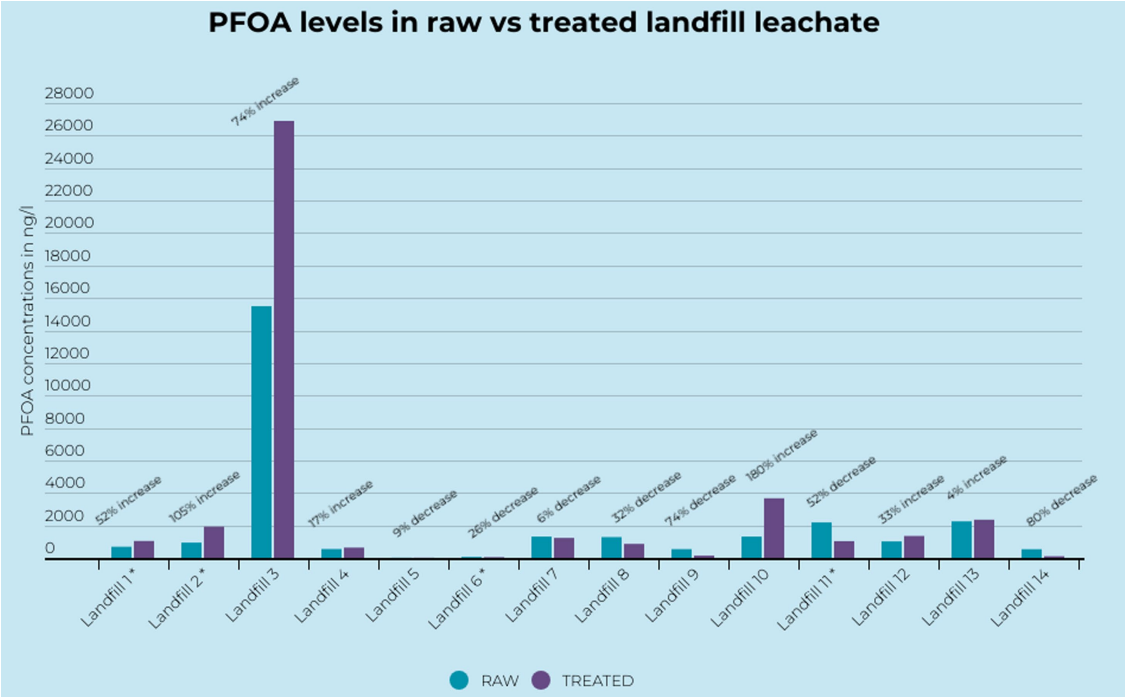


FIGURE 2
PFOA concentrations in raw and treated landfill leachate. *Represents average value for landfills with multiple treated or raw samples.

concern. It was disappointing that spatial and temporal data was not collected by subcontractors (or was removed and anonymized) as this limits what could be interpreted from the dataset. Further monitoring would be strongly recommended to compare the performance of different treatment types, and assess seasonal variation. We hope this manuscript will prompt further research of this nature. This will enable the development and use of the most appropriate technologies for leachate treatment. This information will help support the development of more appropriate monitoring programs and policies, both in the UK and globally.

PFOS and PFOA are listed under the Stockholm convention under Annex B (Restriction) Annex A (elimination) respectively. They pose a significant risk to the environment and PFOA is a class one carcinogen and PFOS is possibly carcinogenic to humans (Group 2B). This study shows that there is an urgent need for further investigation and to upgrade inappropriate leachate treatment works in the UK. In several cases the technology that is designed to protect humans and the environment appears to be doing the opposite and is actively creating considerable amounts of PFOS and PFOA.

Data availability statement

The original contributions presented in the study are included in the article/Supplementary material, further inquiries can be directed to the corresponding author.

Author contributions

PN: Conceptualization, Data curation, Formal analysis, Investigation, Visualization, Writing – original draft, Writing – review & editing. DM: Formal analysis, Writing – original draft, Writing – review & editing.

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Conflict of interest

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Supplementary material

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A comprehensive review of landfill leachate treatment technologies

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The management of landfill leachate presents a significant environmental challenge, necessitating a comprehensive and dynamic treatment approach. This comprehensive review delves into the critical issue of landfill leachate treatment, exploring its environmental impact, treatment technologies, regulatory frameworks, and the path towards sustainable management practices. This review explores the complexities of landfill leachate, emphasizing the need for sustainable waste management practices to safeguard environmental health. Our analysis highlights the evolution of conventional and advanced treatment technologies designed to mitigate these risks, focusing on membrane technologies, advanced oxidation processes, and the promising potential of emerging techniques such as adsorption and biological nutrient removal. These technologies are evaluated for their efficiency, cost implications, and sustainability impacts, underscoring the challenges and opportunities within the current landscape of leachate treatment. The review aims to provide insights into designing efficient and effective treatment systems through a detailed analysis of conventional and advanced treatment methods. By examining a case study in Changsha City, the effectiveness of a comprehensive treatment system integrating various technologies is demonstrated. The review underscores the interconnectedness of human activities, environmental health, and waste management, emphasizing the importance of a holistic approach. It stresses the continuous improvement of leachate treatment technologies and the adoption of sustainable practices to reduce the environmental footprint of landfills. Ultimately, it calls for integrating multiple treatment processes, economic considerations, and readiness to address future challenges in landfill leachate treatment, contributing to the advancement of sustainable waste management practices.

KEYWORDS

landfill leachate, treatment technologies, membrane technologies, advanced oxidation processes, waste treatment

1 Introduction

Landfill leachate, a liquid byproduct of the decomposition process in landfills, represents a significant environmental challenge that encapsulates the complexity of modern waste management and the imperative of safeguarding environmental health (Teng et al., 2021). As rainfall or other sources of moisture percolate through the waste material accumulated in landfills, they dissolve and suspend a myriad of substances present

in the waste, creating a potent mixture known as leachate (Mor and Ravindra, 2023). This concoction's composition can vary widely but typically includes dissolved organic matter, inorganic macro-components like salts and metals, and a range of xenobiotic organic compounds, reflecting the diverse nature of disposed materials (Kumar et al., 2023). Organic components, measured in terms of biochemical oxygen demand (BOD) and chemical oxygen demand (COD), indicate the leachate's potential impact on water bodies, while inorganic macro-components, such as ammonia, chlorides, sulfates, and heavy metals like arsenic, cadmium, chromium, lead, and mercury, underline the toxicological concerns associated with leachate (Al-Yaqout and Hamoda, 2020). Furthermore, the presence of xenobiotic compounds, which include pesticides, pharmaceuticals, and industrial chemicals, adds to the complexity of leachate, given their resistance to degradation and potential for bioaccumulation (de Oliveira et al., 2020). The environmental impacts of uncontrolled landfill leachate are profound and far-reaching. Leachate can seep into groundwater and surface water, contaminating aquatic ecosystems, wildlife, and human health (Essien et al., 2022). The infiltration of pollutants into water supplies can disrupt aquatic habitats, harm species diversity, and bioaccumulate the food chain, presenting significant risks to public health through consuming contaminated water and food (Sharma et al., 2023). Moreover, the heavy metals and persistent organic pollutants found in leachate are particularly concerning due to their toxicity, environmental longevity, and potential to cause long-term ecological damage (Hussein et al., 2021).

Recognizing the environmental hazards of landfill leachate, its treatment and management have become critical components of sustainable waste management practices (El-Saadony et al., 2023). Effective leachate treatment aims to neutralize its harmful components, making it safe for discharge or further use (Kumar et al., 2023). This necessitates a multifaceted approach that combines physical, chemical, and biological treatment processes tailored to the specific composition of the leachate and local regulatory requirements (Torretta et al., 2017). Physical treatments, such as sedimentation and filtration, remove suspended solids and particulate matter (Siddiqi et al., 2022). Chemical treatments, including precipitation, coagulation, and advanced oxidation, target dissolved pollutants, reducing their concentration to safer levels (Kurniawan et al., 2006). Biological aerobic and anaerobic treatments further degrade organic substances, transforming them into less harmful compounds (Kurniawan et al., 2010). The importance of landfill leachate treatment extends beyond the immediate goal of pollution control. It embodies the broader principles of environmental protection and sustainable waste management (Mukherjee et al., 2015). By mitigating the adverse impacts of leachate on ecosystems and human health, effective treatment practices contribute to the preservation of water quality, protect biodiversity, and ensure the wellbeing of communities (Senathirajah and Palanisami, 2023). Furthermore, advancements in leachate treatment technologies offer opportunities for resource recovery, such as the extraction of valuable materials or the reuse of treated water, aligning with the principles of the circular economy and resource efficiency (Puyol et al., 2017).

Treating landfill leachate is not just a technical challenge but a fundamental aspect of environmental stewardship and public health protection (Nath and Debnath, 2022). It reflects the evolving understanding of waste not merely as a byproduct to be disposed of but as a potential source of pollution that must be managed with foresight and responsibility. As such, the continuous improvement of leachate treatment technologies, alongside adopting more sustainable waste management practices, is essential to reducing the environmental footprint of landfills. This includes the development of more efficient and cost-effective treatment solutions and the implementation of waste reduction, reuse, and recycling initiatives that can minimize leachate generation in the first place (Kamaruddin et al., 2017).

Ultimately, the comprehensive management and treatment of landfill leachate highlight the interconnectedness of human activities, environmental health, and the need for a holistic approach to waste management (Torretta et al., 2017). This review aims to provides an integrated analysis of both conventional and advanced treatment technologies, emphasizing recent advancements and emerging techniques such as adsorption and biological nutrient removal. Unlike previous reviews that may focus solely on specific technologies or methodologies, this review evaluates a broad spectrum of treatment methods, offering a practical guidance for designing and optimizing treatment systems by evaluating the efficiency, cost implications, and sustainability impacts of various methods. Additionally, we incorporate a detailed case study from Changsha City, demonstrating the practical application and effectiveness of a comprehensive treatment system that integrates various technologies. This real-world example bridges the gap between theoretical discussions and practical implementation, providing actionable insights for practitioners and policymakers. By addressing the interconnectedness of human activities, environmental health, and waste management, our review advocates for a holistic approach that emphasizes continuous improvement of leachate treatment technologies and the adoption of sustainable practices. This comprehensive perspective makes our review an invaluable resource for engineers, environmental scientists, and waste management professionals seeking to implement effective and sustainable treatment solutions in diverse contexts.

2 Characteristics of landfill leachate

2.1 Physical and chemical properties of landfill leachate

Leachate embodies a dynamic concoction of substances, each contributing to its hazardous nature. Landfill leachate's physical, chemical, and biological characteristics delineate a complex and multifaceted environmental challenge that requires a nuanced understanding for effective management and treatment (Table 1) (Andreottola and Cannas, 2024; Castrillón et al., 2010; Naveen et al., 2017; Luo et al., 2020; Subiza-Pérez et al., 2023).

Physically, leachate is primarily a liquid matrix. However, its appearance, viscosity, and other physical properties can vary greatly depending on the landfill's age, the composition of waste, and the amount of precipitation the landfill receives (Anjum et al., 2023). Typically, young leachate is characterized by a high organic load, evident in its dark color and strong odor (Wijekoon et al., 2022). In

TABLE 1 Characteristics of landfill leachate and implications for treatment (Andreottola and Cannas, 2024; Castrillón et al., 2010; Naveen et al., 2017; Luo et al., 2020; Subiza-Pérez et al., 2023).

Aspect	Characteristics	Implications for treatment
Physical	Liquid matrix, varying appearance and viscosity, pH, temperature, conductivity	Influences treatment process selection, affects solubility and treatment effectiveness
Chemical	Dissolved organic/inorganic matter, heavy metals, xenobiotics (e.g., pharmaceuticals, pesticides), COD, BOD, ammonia	Requires targeted removal or neutralization of specific compounds, challenges due to persistence and toxicity of pollutants
Biological	Microbial communities, biodegradation of organic matter, presence of pathogenic organisms	Utilization of microbial activity for degradation, management of pathogenic or harmful microorganisms

TABLE 2 Physical and chemical properties of landfill leachate across different continents (Abunama et al., 2021a).

Parameter		Africa	Asia	Europe	North America	Oceania	South America
Organic	BOD (mg/L)	566.22	1305.85	1281.98	270.11	60.66	875.73
	COD (mg/L)	3026.81	5,983.26	5,890.65	1628.52	1594.5	3117.69
Inorganic	pH	7.72	7.72	8.03	7.95	7.81	8.13
	TKN (mg/L)	26.485	34.55	42.66	26.485	28.33	34.55
	AN (mg/L)	297.89	450.22	525.77	297.89	350.55	450.22
	TDS (mg/L)	10,403.1	11,456.3	15,382.9	3077.1	17,487.5	4364.6
Heavy metals	Cl (mg/L)	84.32	98.72	103.21	72.33	55.66	84.12
	Fe (mg/L)	7.11	25.99	41	22.04	4.88	26.28
	Cu (mg/L)	0.256	0.489	0.323	0.141	0.022	1.131
	Ni (mg/L)	0.45	0.56	0.52	0.32	0.24	0.45
	Zn (mg/L)	0.65	2.738	2.105	0.91	0.159	1.66
	Pb (mg/L)	0.331	0.55	0.54	1.441	0.006	2.524
	Cr (mg/L)	0.49	0.62	0.49	42.585	0.32	1.66
	Hg (mg/L)	0.221	0.33	0.32	0.723	0.0003	3.783
	As (mg/L)	0.0433	0.225	0.0525	33.8	0.1	0.221
	Cy (mg/L)	0.261	0.216	0.075	0.025	0	0.055

contrast, older leachate may appear clearer but contain higher concentrations of dissolved inorganic compounds (Abdel-Shafy et al., 2024). The physical properties of leachate, such as pH, temperature, and conductivity, are critical indicators of its overall composition and the potential challenges it presents for treatment (Abdel-Shafy et al., 2024). For instance, high conductivity in leachate signifies a high concentration of dissolved ions such as chlorides, sulfates, and ammonia, which can affect the efficiency of certain treatment processes. Similarly, pH levels can influence the solubility of metals and the effectiveness of chemical treatments, with acidic or alkaline conditions necessitating specific adjustments in the treatment strategy (Deng et al., 2020).

Chemically, leachate is a cocktail of dissolved organic and inorganic matter, heavy metals, and xenobiotic compounds (Sil and Kumar, 2017). The organic fraction is typically dominated by biodegradable substances like carbohydrates, fats, proteins, and persistent organic pollutants that resist natural degradation processes (Dignac et al., 2000). COD and BOD are key parameters used to quantify the organic strength of leachate, reflecting its potential impact on receiving environments and treatment systems (Naveen et al., 2017). Inorganic components include a range of salts, metals, and nutrients, with ammonia being particularly prevalent due to the nitrogenous decomposition of organic waste (Borah et al., 2020). Heavy metals such as arsenic, cadmium, chromium, lead, and mercury are of significant concern due to their toxicity and propensity to accumulate in ecosystems (Rahman and Singh, 2019). Moreover, the presence of xenobiotics—synthetic chemicals like pharmaceuticals, pesticides, and industrial chemicals—poses a unique challenge, as these compounds often exhibit resistance to degradation and can disrupt biological processes in treatment systems and natural environments (Priya et al., 2024). Table 2 presents a detailed analysis of the physical and chemical properties of landfill leachate from various regions around the world, showing significant variability across different regions (Abunama et al., 2021a). These variations highlight the importance of regional assessments in understanding leachate pollution potential and tailoring appropriate management strategies.

Biologically, leachate is a living, evolving entity, teeming with microorganisms, such as bacteria, fungi and protozoa, which are adapted to its unique chemical milieu. The microbial communities within leachate play a pivotal role in the biodegradation of organic matter, facilitating the natural attenuation of some pollutants (Qi et al., 2023). However, the diversity and activity of these microbial populations can be both a boon and a bane for leachate treatment (Dave et al., 2020). Bacteria such as *Pseudomonas* and *Clostridium* spp., and fungi like Aspergillus and Penicillium spp., are integral to breaking down organic pollutants and transforming hazardous substances. While certain bacteria and fungi are harnessed in biological treatment processes to degrade organic pollutants, the presence of pathogenic organisms or those capable of generating harmful byproducts necessitates careful management to prevent environmental and health risks (Mani et al., 2020). The methanogenic archaea, including Methanobacterium and Methanosarcina spp., also play a crucial role by converting waste into methane, highlighting the importance of microbial activity in both waste reduction and potential energy recovery.

The interplay between the physical, chemical, and biological characteristics of leachate significantly influences the selection and design of treatment processes. Physical attributes like suspended solids and temperature necessitate pre-treatment steps such as sedimentation and filtration to prevent system clogging and optimize subsequent treatment stages. Chemical properties, including pH, organic load, and the presence of toxic compounds, determine the need for neutralization, biological treatment, or advanced oxidation processes to ensure effective contaminant removal. Biological factors, such as microbial content and biodegradability, guide the choice of biological treatment methods, which might require nutrient supplementation for optimal microbial activity. Consequently, tailored and integrated treatment strategies are developed to address the specific challenges presented by leachate's complex composition, ensuring efficient and sustainable management to mitigate its environmental impact.

2.2 Factors on the composition of landfill leachate

The composition of landfill leachate is influenced by a myriad of factors, including the age of the landfill, the types of waste it contains, climatic conditions, and the specific landfill operating technologies employed (Mor and Ravindra, 2023). Each of these factors plays a critical role in determining the leachate's physical, chemical, and biological characteristics, thus influencing its potential environmental impact and the strategies required for its effective management and treatment (Table 2) (Cano et al., 2020; Golwala et al., 2022; Ma et al., 2022; Abdel-Shafy et al., 2024).

The age of a landfill significantly affects the composition of the leachate produced. In the initial stages of landfilling, leachate tends to have a high organic content, characterized by a high BOD and COD, due to the abundance of readily degradable organic matter (Lindamulla et al., 2022). As the landfill matures, the rate of organic matter decomposition decreases, leading to a reduction in BOD levels (Kulikowska and Klimiuk, 2008). However, the concentration of inorganic compounds, such as ammonia, chlorides, and heavy metals, may increase over time due to the continued degradation and leaching processes (Prieto-Espinoza et al., 2022). This temporal evolution in leachate composition necessitates adaptable treatment approaches that effectively address the shifting balance between organic and inorganic constituents (Saxena et al., 2022) (Table 3). In 2022, Liu et al. conducted a study on the molecular differences between young landfill leachate (YL) and mature landfill leachate (ML) (Liu J. et al., 2022). The results indicated that the COD of YL was 74,853 mg/L, while the COD of ML was 3,098 mg/L, suggesting a higher concentration of organic matter in young leachate. Regarding molecular composition, YL contained lower molecular weight organic compounds, primarily consisting of CHO compounds and aliphatic compounds. In contrast, ML predominantly contained CHON compounds and high oxygen highly unsaturated and phenolic compounds. The molecular structure of YL was relatively simple, with higher bioavailability and more straight-chain structures. ML, however, featured more oxygen-containing functional groups and benzene-ring structures, indicating a more complex molecular structure. In terms of chemical properties, YL exhibited lower nominal oxidation state of carbon and modified aromaticity index, indicating that its organic matter was in a more reduced state. Conversely, ML's organic matter had higher oxidation states and aromaticity, reflecting that over time, the organic matter in mature leachate had undergone extensive biochemical reactions, leading to more complex structures (Table 4).

Moreover, the nature of the waste material disposed of in a landfill is another critical determinant of leachate composition. Landfills containing a high proportion of organic waste, such as food scraps and yard waste, tend to produce leachate with high levels of organic pollutants and a greater potential for biological activity (Kjeldsen et al., 2002). Conversely, landfills with significant industrial or hazardous waste may generate leachate with higher concentrations of toxic metals and xenobiotic organic compounds (El-Saadony et al., 2023). The diversity of waste types, including e-waste, pharmaceuticals, and plastics, contributes to the complexity of leachate, requiring targeted treatment solutions capable of addressing a broad spectrum of contaminants (Srivastava et al., 2021).

Also, climatic conditions, including precipitation, temperature, and humidity, play a pivotal role in the generation and composition of leachate. High rainfall areas will likely produce larger volumes of leachate, potentially diluting its contaminant concentrations but increasing the overall load of pollutants that must be managed (Wijekoon et al., 2022). Temperature affects the rate of biochemical reactions within the landfill, with warmer conditions typically accelerating organic matter decomposition and, consequently, influencing the rate of leachate generation and its organic content (Mohammad et al., 2022). Humidity can also impact the moisture content within the landfill, affecting leachate production rates and composition (Abunama et al., 2021b).

Finally, a landfill's design and operational practices can significantly influence leachate composition (Ma et al., 2022). Modern landfills equipped with engineered liners and leachate collection systems are designed to minimize the direct contact of waste with groundwater, thereby reducing leachate generation and its potential contamination (Abdel-Shafy et al., 2024). Additionally, using daily covers can limit rainwater infiltration, further controlling

TABLE 3 Impact of various factors on the composition of landfill leachate (Kulikowska and Klimiuk, 2008; Lindamulla et al., 2022; Prieto-Espinoza et al., 2022; Saxena et al., 2022).

Factors	Options	Leachate composition
Landfill Age	Young Landfill (<10 years)	High in volatile organic compounds, BOD, and COD due to active decomposition of organic matter
	Middle-Aged Landfill (10–20 years)	Reduced levels of BOD and COD as the rate of organic decomposition decreases, increase in inorganic compounds such as ammonia
	Old Landfill (>20 years)	Lower BOD, higher concentrations of inorganic compounds, heavy metals, and stabilized organic compounds
Type of Waste Material	Organic Waste	High levels of BOD, COD, and volatile organic compounds due to decomposition of organic matter
	Industrial and Hazardous Waste	Increased concentrations of toxic metals, xenobiotics, and specific industrial chemicals
	Construction and Demolition Waste	Elevated levels of inorganic compounds, such as salts and heavy metals, lower organic content
	E-Waste	Presence of heavy metals (e.g., lead, mercury, cadmium) and potentially hazardous chemicals from electronic components
Climatic Condition	High Rainfall/High Humidity	Diluted concentrations of contaminants due to increased volume, potential for higher leachate production
	Arid/Dry	Concentrated leachate due to lower precipitation, reduced leachate volume but higher concentration of contaminants
	Cold Temperatures	Slowed decomposition rates leading to lower BOD and COD in the short term, potential for seasonal variation in leachate composition
Design or Operational Practice	Engineered Liners and Leachate Collection Systems	Minimizes the infiltration of external water, reducing leachate volume and potentially concentrating contaminants
	Daily Covers	Reduces rainwater infiltration, controlling leachate production and influencing its composition by limiting dilution
	Landfill Gas Management Systems	Affects the biochemical degradation processes within the landfill, potentially altering the organic content of the leachate

TABLE 4 Comparative Molecular Characteristics of Young and Mature Landfill Leachates in the example landfills in southwest China (Liu J. et al., 2022).

Property		Young landfill leachate	Mature landfill leachate
Age		New Landfill for temporary stockpile	27 years
COD		74,853 mg/L	3098 mg/L
DOM Species Count		1122	5,075
Main Compounds		Aliphatic Compounds, LOHUPC	HOHUPCs
Bioavailability		Higher	Lower
Nominal Oxidation State of Carbon		−0.8010	−0.0692
Aromaticity (AI_{mod})		0.1254	0.2464
Straight-chain Structures		More	Fewer
Benzene-ring Structures		Fewer	More
Proportion of different DOM classes	Low O HUPC	417.96 (23.75%)	2881.86 (27.86%)
	High O HUPC	0%	3863.53 (37.55%)
	Polyphenols and Polycyclic Aromatics	1%	8%
	Aliphatic Compounds	1320.62 (75.03%)	2767.64 (26.76%)

leachate volume and composition (Khasawneh et al., 2022). Landfill gas management systems, which capture and treat or utilize the methane produced during organic decomposition, can also impact the biochemical processes within the landfill, indirectly affecting leachate composition (Japperi et al., 2021).

In sum, the composition of landfill leachate results from complex interactions between the age of the landfill, the types of waste it contains, prevailing climatic conditions, and the technologies employed in landfill operation. Understanding these factors is crucial for the development of effective leachate

management strategies, as they directly influence the environmental impact of landfills and the technical and economic feasibility of leachate treatment solutions.

2.3 Typical values of pollutants in leachate from different regions

To provide a global perspective, typical values for different classes of contaminants in leachate reported from around the world are described below. These values highlight the variability in leachate composition and typical values for different classes of contaminants in leachate from different regions are critical for designing effective treatment strategies.

Organic pollutants like BOD and COD are critical for assessing the pollution levels. For instance, in tropical regions, average BOD values reported were 2435.16 mg/L in Africa, 3455.77 mg/L in America, and 2127.79 mg/L in Asia, while COD values were 7985.1 mg/L in Africa, 6185.17 mg/L in America, and 7504.32 mg/L in Asia (Lindamulla et al., 2022). In China, BOD_5 concentrations varied from 750 to 25,000 mg/L, with an average of 8,500 mg/L, and COD levels ranged from 2,000 to 62,000 mg/L, averaging 19,250 mg/L (Ma et al., 2022).

Inorganic pollutants like Total Dissolved Solids (TDS) and Ammonium Nitrogen (NH_4^+-N) also show significant variability. Polish landfills reported TDS values from 2,225 mg/L to 7,830 mg/L, while NH_4^+-N concentrations in tropical regions varied widely, with higher concentrations in younger leachates (Wdowczyk and Szymańska-Pulikowska, 2021). China reported NH4+-N levels ranging from 100 to 3,100 mg/L, with a mean of 1,300 mg/L (Ma et al., 2022). Brazil's ammonium nitrogen levels ranged between 0.4 and 1,800 mg/L, reflecting the diversity of landfill conditions and waste compositions across the country (Lindamulla et al., 2022).

Heavy metals, another major pollutant category, had varied concentrations with iron ranging from 0.019 mg/L in inactive Polish landfills to 38.73 mg/L in active ones, and zinc reaching up to 2,560 mg/L in Polish active landfills (Wdowczyk and Szymańska-Pulikowska, 2021). In Thailand, heavy metals like cadmium, chromium, copper, nickel, lead, and zinc were generally below 1 mg/L, though still posing potential ecological and health risks (Ma et al., 2022). This highlights the need for careful monitoring and tailored remediation strategies to address potential environmental and health impacts.

The Leachate Pollution Index (LPI) further highlights the pollution potential of leachate, being higher in open dumps compared to engineered landfills (Lindamulla et al., 2022). Young leachates typically show higher LPI values than older leachates, indicating a higher pollution potential in newer landfills. This global variability in leachate composition underscores the importance of site-specific assessments and tailored treatment approaches for effective leachate management.

3 Regulations and standards

Countries around the world are increasingly recognizing the importance of effective landfill leachate treatment as a critical component of waste management and environmental protection. The significant number of research publications on this topic, particularly from nations with large populations, underscores their commitment to addressing the challenges posed by waste and pollutants. Figure 1 shows the research on "landfill leachate treatment" collected from the Scopus database from 2014 to 2023. The top five countries are China (1,675 articles), India (500 articles), the United States (498 articles), Malaysia (418 articles), and Brazil (302 articles). These all reflect a positive stance in mitigating the impact of rapid industrial and urban development on the environment, and countries have also invested heavily in research to develop efficient landfill leachate treatment technologies, further highlighting the global recognition of the need for advanced and sustainable waste management practices. This joint effort is not only aimed at improving public health and environmental quality, but also at promoting global sustainable development.

3.1 Global regulations on landfill leachate management

International and national regulations governing the disposal and treatment of landfill leachate are crucial for protecting the environment and public health from the potential hazards posed by waste management practices (Daniel et al., 2021). These regulations are designed to set minimum standards for leachate management, ensure the safe handling of this potentially toxic byproduct, and encourage sustainable practices in the waste management industry (Table 5) (Oakes and Shank, 1979; Directive, 2003; Protection, 2003; Obradović et al., 2010; Pollutants, 2011; Gas, 2017; Bergesen et al., 2018; Bhawan and Nagar, 2020; Lin et al., 2022; Santana et al., 2022).

At the international level, the Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and their disposal is one of the most significant treaties (Ahmed, 2019). It impacts leachate management by restricting the international movement of hazardous wastes, which includes leachate, and encouraging proper disposal (Amos et al., 2024). Another important piece of international legislation is the Stockholm Convention on Persistent Organic Pollutants, which addresses the treatment and disposal of leachate due to its potential to carry organic pollutants that can persist in the environment (Omoto, 2014). The European Union (EU) has established stringent directives for waste management, including the Landfill Directive (1999/31/EC), which sets the standards for waste disposal in landfills, including the collection and treatment of leachate (Schiopu and Gavrilescu, 2010). It specifies that landfills must be designed and operated to collect and treat leachate to prevent groundwater and environmental contamination. Furthermore, the EU's Water Framework Directive (2000/60/EC) also affects leachate management by setting out a framework for the protection of inland surface waters, transitional waters, coastal waters, and groundwater, requiring that leachate be treated to prevent the pollution of water bodies (Kyriakopoulos, 2021).

The Resource Conservation and Recovery Act (RCRA) provides the framework for properly managing solid and hazardous waste in the United States (Hinds, 2022). Under Subtitle of RCRA, the Environmental Protection Agency (EPA) has established regulations that require municipal solid waste landfills to use

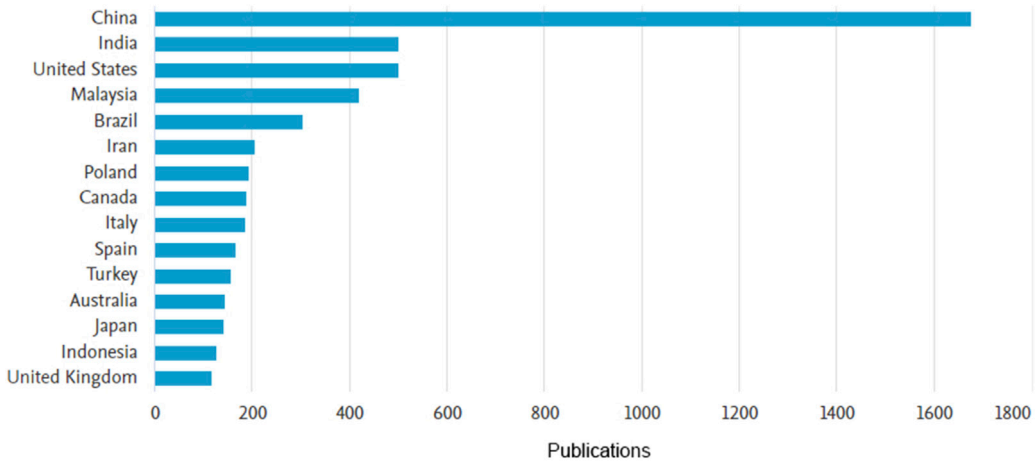


FIGURE 1
Number of publications from the top 15 countries on the topic "Landfill leachate treatment" collected in the Scopus database from 2014 to 2023.

TABLE 5 Global laws and regulations on landfill leachate management (Oakes and Shank, 1979; Directive, 2003; Protection, 2003; Obradović et al., 2010; Pollutants, 2011; Gas, 2017; Bergesen et al., 2018; Bhawan and Nagar, 2020; Lin et al., 2022; Santana et al., 2022).

Country	Regulation/Law	Key provisions
European Union	Landfill Directive (1999/31/EC)	Sets standards for waste disposal in landfills, including leachate treatment to prevent groundwater contamination
United States	Resource Conservation and Recovery Act (RCRA)	Requires municipal solid waste landfills to use liners and leachate collection systems to protect groundwater
China	Standard for Pollution Control on the Landfill Site of Municipal Solid Waste (GB 16889–2008)	Details requirements for leachate collection, treatment, and discharge, including limitations on contaminants
India	Hazardous and Other Wastes (Management and Transboundary Movement) Rules, 2016	Specifies criteria for handling, treatment, and disposal of hazardous waste, including leachate from landfills
Brazil	National Solid Waste Policy (Política Nacional de Resíduos Sólidos - PNRS, Law No. 12,305/2010)	Emphasizes the treatment and environmentally sound disposal of waste, including leachate management
Canada	Canadian Environmental Protection Act (CEPA)	Provides a framework for the safe management of hazardous waste and substances, including leachate
Australia	National Environment Protection (Assessment of Site Contamination) Measure 1999	Establishes guidelines for the assessment and management of site contamination, including leachate
Germany	Technical Instructions on Municipal Waste (TA Siedlungsabfall)	Specifies technical requirements for waste treatment facilities, including leachate management systems
Japan	Waste Management and Public Cleansing Law	Regulates waste disposal and cleaning operations, ensuring proper leachate control and treatment
South Africa	National Environmental Management: Waste Act, 2008	Lays down waste management measures to protect health and the environment, including leachate standards

composite liners and leachate collection systems to protect groundwater (Bonaparte et al., 2024). The National Pollutant Discharge Elimination System (NPDES), part of the Clean Water Act, regulates the discharge of leachate into surface waters, requiring permits and setting treatment standards (Hudiburgh, 2020). The EPA's treatment standards for hazardous wastes subject to land disposal restrictions include several key parameters that must be monitored under applicable regulations (Agency, 2024). These standards specify regulated hazardous constituents, which are chemicals or compounds that need to be treated to meet specific numerical limits before disposal. For instance, chloroform must be reduced to a concentration of 0.046 mg/L for wastewaters and 6.0 mg/kg for nonwastewaters. Characteristic wastes, such as those exhibiting ignitability, corrosivity, reactivity, or toxicity, require deactivation of these characteristics and treatment of any underlying hazardous constituents (UHCs) to meet universal treatment standards (UTS). Ignitable wastes must be treated by methods such as combustion or deactivation, depending on their total organic carbon content, to remove the ignitability characteristic. Corrosive wastes need deactivation to eliminate corrosivity, with high-level radioactive corrosive wastes requiring vitrification. Reactive wastes are generally treated by deactivation,

with reactive cyanide wastes having specific concentration-based standards, such as 590 mg/kg total cyanide and 30 mg/kg amenable cyanide for nonwastewaters, and 0.86 mg/L amenable cyanide for wastewaters. Toxicity characteristic wastes must be treated to meet numerical standards for metals, pesticides, and organics, as well as the UTS for UHCs. Additionally, alternative treatment standards are available for certain waste types like contaminated soil and debris, allowing for a reduction in hazardous constituent concentration by 90% or to levels not exceeding 10 times the UTS. These parameters ensure that hazardous wastes are adequately treated to minimize their environmental impact before land disposal.

In China, the Environmental Protection Ministry released the "Standard for Pollution Control on the Landfill Site of Municipal Solid Waste" (GB 16889–2008), which sets out leachate collection, treatment, and discharge requirements (Liu Q. et al., 2022). These include specific limitations on contaminants such as BOD, COD, ammonia nitrogen, total phosphorus, and heavy metals in the discharged leachate (Shi et al., 2021). India has enacted the Hazardous and Other Wastes (Management and Transboundary Movement) Rules, 2016, which set forth the criteria for identifying hazardous waste and its handling, treatment, and disposal standards, including leachate from landfills (Sadala et al., 2023). The Central Pollution Control Board (CPCB) of India provides technical guidelines for the secure landfill of hazardous waste, emphasizing the need for a comprehensive leachate management system (Kumar et al., 2007). In Brazil, the National Solid Waste Policy (Política et al. - PNRS) established under Law No. 12,305/2010 requires proper leachate management and emphasizes the treatment and environmentally sound disposal of waste (Galavote et al., 2022).

These examples demonstrate the common goal across different regulatory frameworks: minimizing environmental and health risks associated with leachate from landfills. National regulations often incorporate or build upon international guidelines, tailoring requirements to local conditions and available technologies. However, the effectiveness of these regulations often depends on the robustness of enforcement mechanisms and the commitment of local authorities to uphold standards. The significance of these regulations extends beyond environmental protection. They also have profound economic and social implications. Strict regulatory standards drive innovation in the waste management sector, leading to the development of new treatment technologies and practices that can reduce the volume and toxicity of leachate. Moreover, by requiring the safe disposal of leachate, regulations also prevent the long-term social costs associated with environmental cleanup and healthcare for communities affected by pollution (Fang et al., 2021).

3.2 Regulatory influence on landfill leachate treatment technology selection

Regulatory standards significantly influence the selection of landfill leachate treatment technologies. These regulations protect the environment and public health from the potential hazards posed by untreated or inadequately treated leachate (Tenodi et al., 2020). As such, they set specific benchmarks that leachate must meet before it can be released into the environment or treated further for potential reuse.

The impact of regulatory standards on the choice of treatment technology is multifaceted. Initially, these standards determine the permissible levels of various pollutants, such as organic matter, heavy metals, and nitrogen compounds, that can be present in the treated leachate (Salem et al., 2008). This directly affects the technology selection, as different processes vary in their ability to remove or neutralize specific contaminants (Ahmed et al., 2017). For instance, high BOD and COD levels might necessitate biological treatment processes, such as activated sludge systems (Haydar et al., 2007). At the same time, the presence of heavy metals may require chemical precipitation or ion exchange technologies (Saleh et al., 2022). Furthermore, regulations often specify the methods that must be used for testing and monitoring leachate quality (Brennan et al., 2016). These requirements can influence the choice of technology by necessitating systems that treat leachate effectively and allow for easy sampling and monitoring. Technologies that offer more control and predictability in their operation, such as membrane bioreactors, can become preferred options under stringent monitoring requirements (Galinha et al., 2018). Advanced treatment technologies like reverse osmosis, nanofiltration, and advanced oxidation processes become more prevalent in jurisdictions with strict environmental standards (Valdés et al., 2021). These technologies can achieve the low contaminant levels required for discharge into sensitive environments or for meeting the high-quality standards necessary for leachate reuse in irrigation or industrial processes (Torretta et al., 2017).

Moreover, regulations also impact the economic aspects of technology choice. Compliance with stringent standards often comes with higher operational and maintenance costs (Siddiqi et al., 2022). As a result, regulatory frameworks can indirectly incentivize the development and adoption of more cost-effective and energy-efficient technologies (Pan et al., 2015). For example, the push to meet low nitrogen limits could encourage the adoption of innovative nitrogen removal processes that consume less energy than traditional methods (Nourmohammadi et al., 2013). Regulatory standards also influence the long-term planning and scalability of treatment solutions. Facilities must consider current regulations and potential future tightening of standards (Bunce et al., 2018). This foresight leads to the selection of technologies that are compliant now and adaptable to future regulatory changes, thereby protecting the investment in treatment infrastructure over time. Environmental policies and regulatory standards can also foster the integration of multiple treatment technologies. To meet comprehensive standards that cover a wide range of contaminants, treatment plants may combine physical, chemical, and biological processes in a tiered treatment strategy. Such integration can achieve higher levels of purification than standalone systems.

In addition to these factors, regulations may promote sustainable practices by advocating for treatment technologies that minimize waste generation or allow for resource recovery (Capodaglio, 2017). Techniques that enable the recovery of byproducts, such as biogas production or the retrieval of precious metals, might be preferred in regions where sustainability is a regulatory goal (Chrispim et al., 2021).

In conclusion, the impact of regulatory standards on the choice of landfill leachate treatment technology is profound, driving innovation and influencing every aspect from the design phase to operational strategy. By setting the bar for environmental

TABLE 6 Conventional landfill leachate treatment technologies (Torretta et al., 2017; Dogaris et al., 2020; Ahmed et al., 2021; Younas et al., 2021; Siddiqi et al., 2022).

Treatment	Brief overview	Advantages	Limitations	Applicability	Representative technologies	Typical target pollutants	Removal efficiency
Physical	Involves the separation of substances without a chemical reaction, such as screening, sedimentation, and flotation	— Low energy requirement — Simple operations	— Often requires subsequent treatment — Limited effectiveness for dissolved contaminants	— Preliminary treatment — Suitable for large suspended solids	— Screening — Sedimentation — Flotation	— Suspended solids — Particulate matter — Colloidal particles	— Sedimentation: 50%–70% suspended solids — Filtration techniques: 85%–95% COD, 80%–90% BOD
Chemical	Uses chemical reactions to remove pollutants, including precipitation, oxidation, and ion exchange	— Effective for specific contaminants — Can achieve high levels of purification	— Can produce secondary waste — Chemicals used can be hazardous	— Versatile, can treat a range of leachate types — Used for heavy metals	— Precipitation — Oxidation — Ion Exchange	— Heavy metals — Dissolved organics — Inorganic compounds	— Coagulation-flocculation: 60%–90% suspended solids and 50%–70% COD — Precipitation:80%–95% heavy metals — Oxidation-reduction: 80%–90% color and 70%–85% COD — Fenton's reagent:75%–90% COD and 80%–95% organic pollutants
Biological	Employs microorganisms to degrade organic pollutants, categorized into aerobic and anaerobic processes	— Cost-effective — Environmentally friendly	— Sensitive to toxic compounds — May require long treatment times	— Ideal for organic-rich leachate — Commonly used in municipal wastewater	— Aerobic and Anaerobic Digestion — Membrane Bioreactors	— Organic matter (BOD, COD) — Nitrogen — Phosphorus	— Activated sludge: 70%–90% BOD and 60%–80% COD — Digesters: 80%–90% BOD and 70%–85% COD — Constructed wetlands: 60%–80% BOD, 50%–70% COD, and 40%–60% ammonia

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compliance, regulations ensure that the selected technologies not only mitigate the potential impact of leachate on the environment but also align with the broader goals of sustainability and public health protection.

4 Conventional landfill leachate treatment technologies

Landfill leachate treatment is a crucial component of modern waste management systems, aiming to mitigate the environmental impact of leachate discharge from landfills. As leachate comprises a complex mixture of dissolved and suspended contaminants, its treatment requires a multifaceted approach involving various conventional technologies. These include physical, chemical, and biological methods, each with principles, advantages, limitations, and practical considerations (Table 6) (Torretta et al., 2017; Dogaris et al., 2020; Ahmed et al., 2021; Younas et al., 2021; Siddiqi et al., 2022).

4.1 Physical methods

Physical methods for landfill leachate treatment involve processes that primarily rely on physical separation or filtration mechanisms to remove suspended solids and particulate matter (Renou et al., 2008). These methods typically include sedimentation, filtration, and membrane technologies (Bashir et al., 2016). Sedimentation involves allowing suspended solids to settle out of the leachate under the influence of gravity, facilitating their removal from the liquid phase (Wang et al., 2021). Filtration techniques, such as sand or activated carbon filtration, employ porous media to trap suspended solids and organic compounds as the leachate passes through the filter bed (Loh et al., 2021). Membrane technologies, including microfiltration, ultrafiltration, nanofiltration, and reverse osmosis, utilize semi-permeable membranes to remove contaminants based on their molecular size and charge selectively (Rai and Shrivastav, 2022). Air stripping is a physical treatment process that removes volatile contaminants, such as VOCs and ammonia, from landfill leachate by passing air through the liquid (De et al., 2019a; 2022). The process relies on Henry's Law to transfer contaminants from the liquid to the gas phase, using packed towers or aeration tanks to maximize contact surface area. Air stripping is effective, simple to operate, and can achieve high removal efficiencies (up to 90% for ammonia). However, it produces off-gas that requires further treatment to prevent air pollution and can be energy-intensive. Despite these challenges, air stripping is an essential part of leachate treatment, often used alongside other technologies to improve overall performance.

In terms of efficiency, sedimentation can remove 50%–70% of suspended solids. Filtration techniques, such as activated carbon filtration, can achieve removal efficiencies of 85%–95% for COD and 80%–90% for BOD (De et al., 2019b; Mojiri et al., 2019). Membrane technologies, including microfiltration, ultrafiltration, nanofiltration, and reverse osmosis, utilize semi-permeable membranes to selectively remove contaminants based on their molecular size and charge. Microfiltration and ultrafiltration can remove 90%–99% of suspended solids and bacteria, while nanofiltration is effective in removing 70%–90% of divalent ions and 50%–70% of monovalent ions (Peng, 2017). Reverse osmosis offers the highest removal efficiencies, capable of removing 95%–99% of dissolved solids, heavy metals, and organics (Peng, 2017).

The principal advantage of physical treatment methods lies in their ability to achieve high levels of contaminant removal, particularly for suspended solids and larger organic molecules (Saravanan et al., 2021). These methods are often effective in reducing turbidity and removing colloidal particles, thereby improving the treated leachate's visual clarity and overall quality (Zakaria et al., 2023). Additionally, physical treatment processes are generally straightforward to implement and operate, requiring minimal chemical additives and relatively low energy inputs compared to some chemical and biological methods (Nyabadza et al., 2023). However, physical treatment methods also have certain limitations and practical considerations. The potential for membrane fouling or clogging, particularly in membrane-based processes, can reduce treatment efficiency and increase operational costs (Asif and Zhang, 2021). Moreover, physical methods may not effectively remove dissolved contaminants such as heavy metals or organic compounds, which may require additional treatment steps (Saleh et al., 2022). Furthermore, membrane technologies' high capital and operational costs can limit their widespread adoption, particularly in regions with limited financial resources (Othman et al., 2022).

4.2 Chemical methods

Chemical methods for landfill leachate treatment involve the addition of chemical reagents or agents to facilitate the precipitation, coagulation, flocculation, oxidation, or neutralization of contaminants present in the leachate (Aziz et al., 2022). These methods aim to enhance the removal of dissolved metals, organic compounds, and other pollutants through chemical reactions that transform them into insoluble precipitates or aggregates that can be separated from the liquid phase (El-Saadony et al., 2023; Chen et al., 2024). Common chemical treatment processes include coagulation-flocculation, precipitation, oxidation-reduction, and advanced oxidation processes (AOPs) (Sengupta and Pal, 2021).

Coagulation-flocculation involves the addition of coagulants, such as ferric chloride or aluminum sulfate, followed by the addition of flocculants, such as polymers, to promote the aggregation of suspended solids and colloidal particles into larger flocs that can be easily separated from the leachate (Abujazar et al., 2022). Precipitation methods rely on adding chemicals that react with dissolved contaminants to form insoluble precipitates, which can then be removed by sedimentation or filtration (Pohl, 2020). Oxidation-reduction processes, such as ozonation or chlorination, involve adding oxidizing agents to degrade organic pollutants or reduce the concentration of toxic compounds (Wang and Chen, 2020). AOPs utilize powerful oxidants such as hydrogen peroxide or ozone, combined with UV radiation or catalysts, to generate highly reactive hydroxyl radicals that can mineralize organic contaminants into simpler, less harmful compounds (Pandis et al., 2022).

Regarding removal efficiency, coagulation-flocculation can remove 60%–90% of suspended solids and 50%–70% of COD (Teh et al., 2016). Precipitation is highly effective in removing

heavy metals, achieving removal efficiencies of 80%–95% for metals such as lead, cadmium, and chromium (Fu and Wang, 2011). Oxidation-reduction processes like ozonation can remove 80%–90% of color and 70%–85% of COD (Miklos et al., 2018). Advanced oxidation processes, such as the use of Fenton's reagent, can remove 75%–90% of COD and 80%–95% of organic pollutants (Babuponnusami and Muthukumar, 2014).

The primary advantage of chemical treatment methods lies in their versatility and effectiveness in targeting a wide range of contaminants present in leachate (Pisharody et al., 2022). Chemical processes can achieve high removal efficiencies for dissolved metals, organic compounds, and recalcitrant pollutants that may not be effectively treated by physical or biological methods alone (Nidheesh et al., 2022). Additionally, chemical treatment processes can often be tailored to specific leachate compositions and treatment objectives by adjusting the type and dosage of chemical additives used (Khoo et al., 2020). One of the common disadvantages of chemical treatment methods, particularly coagulation-flocculation and precipitation, is the generation of sludge. This sludge, a byproduct of the treatment process, consists of the aggregated contaminants and excess chemical reagents, posing a significant disposal challenge (Sharma et al., 2022). Using chemical reagents can also increase operational costs and pose health and safety risks to operators if improperly handled. The management of this sludge is crucial as it requires appropriate handling, treatment, and disposal to avoid secondary pollution. Chemical sludge typically needs to be dewatered and may require further stabilization or treatment before disposal in landfills or use in other applications. The disposal costs and environmental impacts associated with sludge management must be considered when evaluating the overall feasibility and sustainability of chemical treatment methods (Hou et al., 2023).

4.3 Biological methods

Biological methods for landfill leachate treatment harness the metabolic activity of microorganisms to degrade organic pollutants and remove nutrients from the leachate (Wu D. et al., 2015). These methods typically involve aerobic treatment, anaerobic treatment, and constructed wetlands, which utilize different microbial communities and environmental conditions to facilitate the leachate's biodegradation and transformation of contaminants (Kurniawan et al., 2010). Aerobic treatment involves exposing the leachate to oxygen-rich conditions, allowing aerobic bacteria to metabolize organic compounds and oxidize them into simpler, less harmful substances (Bhambore and Suresh Kumar, 2022). On the other hand, anaerobic treatment processes operate under oxygen-free conditions and rely on anaerobic bacteria to degrade organic matter through fermentation and methanogenesis, producing methane gas as a byproduct (Zamri et al., 2021). Constructed wetlands are engineered systems that mimic natural wetland ecosystems and utilize wetland vegetation, soil, and microbial communities to remove contaminants from the leachate through physical, chemical, and biological processes (Hassan et al., 2021). These systems provide an ideal habitat for the growth of wetland plants and microorganisms, which can absorb, adsorb, and metabolize pollutants in the leachate, improving its quality (Bakhshoodeh et al., 2020). Additionally, constructed wetlands can enhance nutrient removal and promote groundwater recharge by facilitating water infiltration and retention in the soil (Chand et al., 2022).

When it comes to biological methods for landfill leachate treatment, aerobic treatment processes, such as activated sludge, can remove 70%–90% of BOD and 60%–80% of COD (Peng, 2017). Anaerobic treatment using digesters can achieve removal efficiencies of 80%–90% of BOD and 70%–85% of COD, with the added benefit of producing biogas as a byproduct (Mukherjee et al., 2015). Constructed wetlands offer another effective solution, capable of removing 60%–80% of BOD, 50%–70% of COD, and 40%–60% of ammonia (Wu S. et al., 2015).

Here we need to mention a mature biological treatment technology that has been widely used in the field of wastewater treatment for many years, biological nutrient removal (BNR). BNR harnesses the metabolism of microorganisms to remove nutrients, notably nitrogen and phosphorus, from wastewater (Deng et al., 2023). BNR is crucial for preventing eutrophication in aquatic environments caused by nutrient overloads (Kabuba et al., 2022). BNR operates through various biological mechanisms, including nitrification, denitrification, and phosphorus uptake by specific bacteria, under different operational conditions to remove these nutrients (Rout et al., 2021a). Factors influencing the efficiency of BNR include biomass concentration, hydraulic retention time, temperature, and pH, which need to be optimized to enhance microorganism activity and nutrient removal (Mishra et al., 2022). The complexity of BNR systems, which can include multiple tanks and stages for different phases of nutrient removal, requires careful design and operation to achieve the desired outcomes (Al-Hazmi et al., 2022). BNR offers a cost-effective and environmentally friendly approach to nutrient removal, avoiding using chemicals and leveraging natural biological processes (Al-Hazmi et al., 2024). This technology requires for a large footprint due to the extensive infrastructure needed and the complexity of operation and maintenance, which demands skilled personnel and continuous monitoring (Abyar and Nowrouzi, 2023).

Biological treatment methods can achieve sustainable and cost-effective leachate treatment through natural processes (Chand et al., 2022). Biological processes are highly efficient in removing organic pollutants and nutrients from the leachate, often achieving high removal efficiencies under favorable environmental conditions (Li et al., 2021). Moreover, biological treatment methods are generally environmentally friendly and produce minimal secondary waste or harmful byproducts compared to chemical or physical methods (Sathya et al., 2023). Nevertheless, compared with chemical or physical methods, the required processing time is longer because the microbial degradation process can be relatively slow, especially under certain environmental conditions (Sharma, 2020). Additionally, biological treatment systems may be sensitive to fluctuations in temperature, pH, and other environmental parameters, which can affect treatment performance and reliability (Sundui et al., 2021). Moreover, toxic or inhibitory substances in the leachate, such as heavy metals or recalcitrant organic compounds, may limit the effectiveness of biological treatment processes and require additional pre-treatment or post-treatment steps to achieve desired treatment objectives (Ilmasari et al., 2022).

TABLE 7 Membrane technology in landfill leachate treatment (de Almeida et al., 2020; Magalhães et al., 2020; Yusuf et al., 2020; El Batouti et al., 2022).

Process	Introduction	Mechanism	Target pollutants	Removal efficiency	Factors affecting removal efficiency	Advantages	Limitations
RO	RO is a high-efficiency filtration process that removes contaminants by applying pressure to push water through a semi-permeable membrane	Solution/ diffusion; Size exclusion	Dissolved salts, metals, organics, and some microorganisms	Remove exceeding 99% for various contaminants	Membrane fouling level, feed water quality, operating pressure	High removal efficiencies for a wide range of contaminants; Produces high-quality effluent	High energy consumption; Generates concentrated waste stream
NF	NF is a selective filtration technology that removes specific ions and small molecules from water	Size exclusion; Electrostatic exclusion	Hardness ions (e.g., Ca, Mg), heavy metals, certain organic compounds	Up to 99.62% removal of PFAS.	Membrane fouling level, feed composition, operating pressure	Lower pressure and energy requirements than RO; Can selectively remove contaminants	Less effective for monovalent ions; Still produces a waste concentrate
UF	UF uses a semi-permeable membrane to remove suspended solids, bacteria, viruses, and large molecules	Size exclusion	Suspended solids, bacteria, viruses, macromolecules	Remove 90%–99% of suspended solids and bacteria	Membrane fouling level, pore size of the membrane, feed water turbidity	Low energy requirement; Effective pre-treatment for RO/ NF; High removal of particulates and microorganisms	Limited in removing dissolved ions and small molecules
MBR	MBR combines biological treatment with membrane filtration to degrade organic pollutants and remove solids	Biological degradation; Size exclusion	Organic pollutants, nutrients (N, P), suspended solids	High removal rates for BOD (85%–95%) and COD (70%–85%)	Biomass concentration, membrane fouling level, hydraulic retention time	High-quality effluent; Compact design; High organic and nutrient removal efficiency	Higher operational and maintenance costs due to fouling; Energy-intensive

In conclusion, conventional landfill leachate treatment technologies encompass diverse physical, chemical, and biological methods, each offering unique advantages, limitations, and practical considerations. Physical methods rely on separation or filtration mechanisms to remove suspended solids and colloidal particles from the leachate. In contrast, chemical methods involve the addition of chemical reagents to facilitate the precipitation, coagulation, or oxidation of contaminants. Biological methods harness the metabolic activity of microorganisms to degrade organic pollutants and remove nutrients from the leachate through natural processes. While each treatment method has its strengths and weaknesses, selecting an appropriate treatment technology depends on factors such as leachate composition, treatment objectives, regulatory requirements, and site-specific considerations. By understanding each treatment method's principles, advantages, limitations, and practicality, stakeholders can make informed decisions to design and implement effective leachate treatment systems that meet environmental standards and protect human health and the environment.

5 Advanced landfill leachate treatment technologies

5.1 Common technology

5.1.1 Membrane technology

Membrane technology has emerged as a pivotal solution in treating landfill leachate (Chen et al., 2021). This technology offers a versatile and efficient approach to removing contaminants, including dissolved salts, heavy metals, organic compounds, and particulates (El Batouti et al., 2021). The application of membrane technology in leachate treatment primarily involves reverse osmosis, nanofiltration, ultrafiltration, and membrane bioreactors, each serving unique roles in the purification process (Table 7) (de Almeida et al., 2020; Magalhães et al., 2020; Yusuf et al., 2020; El Batouti et al., 2022).

5.1.2 Reverse osmosis (RO)

Reverse osmosis is a cornerstone of membrane technology for leachate treatment, renowned for its ability to remove a wide array of contaminants, including dissolved salts and organic molecules (Feria-Díaz et al., 2021). The principle of RO involves applying pressure to the leachate across a semi-permeable membrane, which allows water molecules to pass while retaining the majority of dissolved solids (Mengesha and Sahu, 2022). This process is particularly effective in reducing leachate conductivity, making it a critical step in compliance with stringent discharge regulations (Keyikoglu et al., 2021). RO systems can achieve removal efficiencies exceeding 99% for various contaminants, rendering them indispensable in scenarios requiring high-quality effluent (Srivastava et al., 2022). However, the performance of RO systems is significantly influenced by factors such as membrane fouling, feed water quality, and operating pressure (Odabaşı et al., 2022). Membrane fouling, caused by the accumulation of particles and microorganisms on the membrane surface, can lead to decreased efficiency and increased operational costs (Sengar and Vijayanandan, 2022). To mitigate these effects, pre-treatment processes and regular maintenance are crucial. Additionally, generates a concentrated brine that requires further management

or treatment, presenting an environmental and operational challenge (Sengar and Vijayanandan, 2022). RO is particularly suitable for final polishing of treated leachate to meet discharge standards.

5.1.3 Nanofiltration (NF)

Nanofiltration occupies a niche between reverse osmosis and ultrafiltration, with a pore size that allows it to selectively remove divalent and multivalent ions, some monovalent ions, and organic molecules (Zhao et al., 2021). NF is particularly valued for its ability to soften water by removing hardness ions, such as calcium and magnesium, alongside a significant reduction in organic content and color (Patel et al., 2023). NF's selectivity allows it to target pollutants with lower pressure and energy requirements than RO, making it a cost-effective option for many treatment scenarios (Yadav et al., 2022). The process is well-suited for pre-treatment in RO systems or as a standalone treatment for leachates with moderate contamination levels (Ali, 2021). Like RO, NF also produces concentrate streams that require careful disposal or further treatment (Song et al., 2020). The efficiency of NF is contingent upon membrane fouling, feed composition, and operating pressure (Yang et al., 2024). Similar to RO, the presence of pre-treatment processes is vital to prevent fouling and ensure consistent performance. While NF offers the advantage of selective contaminant removal with reduced energy consumption, its limitations include less effectiveness for monovalent ions and the production of a waste concentrate that requires further treatment (Tian et al., 2021).

5.1.4 Ultrafiltration (UF)

Ultrafiltration is a membrane filtration process employing larger pore sizes than NF and RO, making it ideal for removing suspended solids, bacteria, viruses, and high molecular weight solutes (Urošević and Trivunac, 2020). Factors like membrane fouling, pore size, and feed water turbidity can impact UF's removal efficiency (Peters et al., 2021). Nonetheless, UF systems require lower energy to operate and serve as an excellent pre-treatment step for RO and NF processes by reducing the load of suspended solids and microbial content, thereby extending the lifespan of downstream membranes (El Batouti et al., 2022). UF is often used as a pre-treatment step to protect downstream RO or NF systems from fouling. It can typically remove 90%–99% of suspended solids and bacteria. The primary advantage of UF lies in its ability to operate at lower pressures, thereby reducing energy costs (Fan et al., 2020). It is also less susceptible to chemical damage, allowing for a broader range of chemical cleaning agents. Although UF is highly effective in removing particulates and microorganisms, its limitations lie in its inability to address dissolved ions and small molecules, underscoring the need for a comprehensive treatment approach that may include subsequent NF or RO stages (Castro-Muñoz, 2020).

5.1.5 Membrane bioreactors (MBR)

The membrane bioreactor technology integrates biological treatment processes with membrane filtration, offering a compact and efficient solution for degrading organic pollutants and removing solids (Neoh et al., 2016). In an MBR system, a bioreactor degrades organic pollutants through microbial activity, while a membrane filtration unit separates treated water from biomass and suspended solids (Deng et al., 2022). MBRs can achieve high levels of organic and nutrient removal, with the added benefit of producing a clarified, disinfected effluent (Hai et al., 2014). Integrating biological treatment and membrane separation enhances system stability and reduces the footprint compared to conventional treatment setups (Waqas et al., 2020). MBRs are suitable for leachate with high organic loads and in locations where space is limited. They can achieve high removal rates for BOD (85%–95%) and COD (70%–85%), along with effective solids separation. The performance of MBRs is influenced by factors such as biomass concentration, membrane fouling, and hydraulic retention time. Despite the challenges of operational and maintenance costs associated with fouling and energy consumption, MBRs provide significant advantages, including high-quality effluent, compact design, and efficient removal of organic matter and nutrients (Meng et al., 2007).

5.2 Advanced oxidation processes

The treatment of landfill leachate using AOPs has garnered significant attention due to the complexity and variability of leachate composition (Fang et al., 2021). AOPs are characterized by their ability to generate highly reactive species, particularly hydroxyl radicals, which can non-selectively degrade a wide range of pollutants (Anandan et al., 2020). Usually there are four primary AOPs: the Fenton process, ozonation, photocatalysis, and electrochemical oxidation (Table 8) (Rekhate and Srivastava, 2020; Wang and Zhuan, 2020; Pandis et al., 2022).

5.2.1 Fenton process

The Fenton process is a chemical treatment method that involves the reaction of hydrogen peroxide (H_2O_2) with ferrous iron (Fe^{2+}) to produce hydroxyl radicals (·OH·) (Benassi et al., 2021). These radicals are extremely reactive and can break down various organic pollutants, including dyes, pharmaceuticals, and other complex organic compounds in landfill leachate (Ateş and Argun, 2021). The mechanism of action in the Fenton process is homogeneous catalysis, where the reaction occurs in solution, allowing for the widespread generation of hydroxyl radicals (Liu et al., 2021). This process is particularly effective for treating organic-rich leachate, offering several advantages, such as high efficiency in degrading organic pollutants and operating under ambient conditions (Soltani et al., 2022). The Fenton process is effective for treating leachate with high concentrations of refractory organic compounds. It can remove 75%–90% of COD and 80%–95% of organic pollutants. However, the process is highly sensitive to operational conditions such as pH, temperature, and the ratios of Fe to H_2O_2 and reaction time (O'Dowd and Pillai, 2020). A major limitation of the Fenton process is sludge production from iron salts, necessitating further treatment steps (Ziembowicz and Kida, 2022). Additionally, precise control over the reaction conditions is crucial to prevent the consumption of hydrogen peroxide by excessive iron, which would otherwise reduce the efficiency of pollutant degradation (Liu et al., 2024).

TABLE 8 AOPs in landfill leachate treatment (Rekhate and Srivastava, 2020; Wang and Zhuan, 2020; Pandis et al., 2022).

Process	Introduction	Mechanism of action	Target pollutants	Removal efficiency	Factors affecting removal efficiency	Advantages	Limitations
Fenton Process	A chemical process using hydrogen peroxide (H_2O_2) and iron (Fe) catalysts to produce hydroxyl radicals for the oxidation of pollutants	Homogeneous catalysis	Organic pollutants, dyes, pharmaceuticals	Remove 35%–90% of COD and 80%–95% of organic pollutants	pH, temperature, Fe and H_2O_2 ratio, reaction time	Highly efficient in degrading organic pollutants; Can operate at ambient conditions	Production of sludge; Requires post-treatment to remove iron; Sensitive to operational conditions
Ozonation	The application of ozone (O_3), a strong oxidizing agent, to break down organic and inorganic materials	Direct oxidation	Color, odor, microorganisms, some organic compounds	Remove 80%–90% of color and 70%–85% COD.	Ozone dosage, contact time, water quality	Effective for color and odor removal; Can disinfect; No sludge production	High energy consumption for ozone generation; May form by-products
Photocatalysis	Uses light to activate a catalyst (typically TiO_2), generating reactive species that degrade contaminants	Heterogeneous catalysis	Organic pollutants, bacteria, viruses	achieved up to 86.14% removal of endosulfan, 96.2% of methylene blue, 97.3% of rhodamine B, and 84.67% of crystal violet dye	Light intensity, catalyst type and concentration, pH	Can be powered by solar energy; Low energy cost; No sludge	Catalyst recovery and reuse; May require UV light
Electrochemical Oxidation	An electrochemical reaction where pollutants are oxidized at the anode surface or in the bulk solution	Direct and indirect oxidation	Dyes, pharmaceuticals, complex organic molecules	Remove 75%–95% of COD and up to 90% of ammonia	Electrode material, current density, conductivity, pH	No chemical addition needed; Precise control over process	High energy consumption; Electrode degradation; Sludge generation; Management

5.2.2 Ozonation

Ozonation employs ozone (O_3), a potent oxidizing agent, to oxidize pollutants directly (Tripathi and Hussain, 2022). This process is highly effective for removing color, odor, and microorganisms from landfill leachate and degrading some organic compounds (Yang C. et al., 2021). The direct oxidation mechanism of ozonation allows for rapid reaction rates, making it a valuable option for enhancing the biodegradability of recalcitrant organic compounds in leachate (Yang Y. et al., 2021). Ozonation can remove 80%–90% of color and 70%–85% of COD. The efficiency of ozonation is influenced by ozone dosage, contact time, and the quality of water being treated. Its advantages include effective color and odor removal, disinfection capabilities, and the absence of sludge production (Hussain et al., 2022). However, ozonation requires significant energy to generate ozone gas, and the potential formation of by-products during the oxidation of certain pollutants can be a concern (Wang and Chen, 2020). Despite these limitations, ozonation remains a favored choice for specific applications within leachate treatment, particularly when targeting pollutants susceptible to oxidation by ozone.

5.2.3 Photocatalysis

Photocatalysis involves using light (typically UV) and a catalyst (commonly titanium dioxide, TiO_2) to produce reactive species capable of degrading pollutants (Gopinath et al., 2020). This process is governed by heterogeneous catalysis, where the catalyst remains in a solid phase, distinct from the liquid phase in which the pollutants are dissolved (Iervolino et al., 2020). Photocatalysis is effective against many contaminants, including organic pollutants, bacteria, and viruses, making it an attractive option for treating landfill leachate (Han et al., 2020). Factors affecting the efficiency of photocatalysis include light intensity, catalyst type and concentration, and the pH of the solution (Gusain et al., 2020). Photocatalysis offers the potential for low energy costs, especially when powered by solar energy, and the absence of sludge production (Ahmad et al., 2020). Catalyst recovery and reuse and dependence on UV light for optimal performance limit its widespread application (Kefeni and Mamba, 2020). Despite these challenges, photocatalysis presents a sustainable option for leachate treatment, especially in sunny climates where solar radiation can be effectively harnessed.

5.2.4 Electrochemical oxidation

Electrochemical oxidation utilizes an electrical current to facilitate the oxidation of pollutants at an electrode surface or within the bulk solution (Fitch et al., 2022). Through direct and indirect oxidation mechanisms, this method can target a broad spectrum of pollutants, including dyes, pharmaceuticals, and complex organic molecules (da Silva et al., 2021). This process is effective for treating leachate with a high load of organic pollutants and ammonia. Electrochemical oxidation can remove 75%–95% of COD and up to 90% of ammonia. One of the primary advantages of electrochemical oxidation is the absence of chemical additives coupled with precise control over the treatment process (Yang L. et al., 2021). This method offers an environmentally friendly approach to leachate treatment, albeit with considerations for

TABLE 9 Emerging technologies in landfill leachate treatment (Bhambri and Karn, 2020; Deng et al., 2020; Hasan et al., 2021; Rajesh Banu et al., 2021).

Technology	Introduction	Mechanism of action	Target pollutants	Removal efficiency	Factors affecting removal efficiency	Advantages	Limitations
Adsorption	A process where pollutants are captured on the surface of a material	Surface adsorption; Physical and chemical adsorption	Organic compounds, metals, dyes	Achieve up to 99% removal of emerging contaminants and 95.1% for strontium	Adsorbent material, contact time, concentration of pollutants, temperature	Low cost; Simplicity of design; No harmful by-products	Limited capacity; Requires regeneration or disposal of adsorbents
Ion Exchange	Involves the exchange of ions between a solid phase and a liquid phase to remove dissolved ions	Electrostatic exclusion	Heavy metals, hardness, radionuclides	Remove nutrients up to 98% for ammonium, 91.38% for arsenic, and over 99% colorants	Ion exchange capacity, flow rate, water chemistry	High specificity; Efficient for targeted contaminants	Regeneration of resins needed; Sensitive to competing ions
Simultaneous Partial Nitrification, Anammox, and Denitrification	An innovative biological process integrating partial nitrification, anammox, and denitrification to treat high-strength nitrogenous leachate efficiently	Partial Nitrification, Anammox, and Denitrification	Ammonia (NH^{4+}) Nitrite (NO^{2-}) Nitrate (NO^{3-})	SNAD achieves nitrogen removal efficiencies of 80%–94.56% for total inorganic nitrogen and over 90% for total nitrogen	Dissolved Oxygen Levels, Temperature, pH, Organic Carbon Availability, ammonia and Nitrite Concentrations	Cost-effective; High Efficiency; Low Sludge Production; Sustainable	Complex Control; Environmental Sensitivity; Startup Time; Monitoring Needs

high energy consumption and potential electrode degradation over time (Deng et al., 2020).

5.3 Other technologies

As environmental standards become increasingly stringent, adopting innovative and efficient treatment technologies is essential. Emerging technologies for treating landfill leachate are pivotal in addressing the environmental challenges posed by the complex and variable composition. These technologies, including adsorption, ion exchange, and biological nutrient removal (BNR), offer innovative solutions to remove many pollutants, from organic compounds and heavy metals to nitrogen and phosphorus (Table 9) (Bhambri and Karn, 2020; Deng et al., 2020; Hasan et al., 2021; Rajesh Banu et al., 2021).

5.3.1 Adsorption

Adsorption is a versatile and widely implemented method for contaminant removal, characterized by the adherence of pollutants onto the surface of an adsorbent material (Shen et al., 2023). This process can occur through physical adsorption, driven by weak van der Waals forces, or chemical adsorption, involving stronger covalent bonds (Rathi and Kumar, 2021). The mechanism is influenced by the surface area, porosity, and chemical nature of the adsorbent, as well as the properties of the pollutants (Ambaye et al., 2021). Adsorption is particularly effective for removing organic compounds, heavy metals, and dyes from leachate, making it a valuable tool for improving water quality (Saravanan et al., 2021). The adsorption efficiency is determined by factors such as the type and concentration of pollutants, contact time, adsorbent material, and environmental conditions like temperature (Rathi and Kumar, 2021). Activated carbon is among the most commonly used adsorbents due to its high surface area and porosity, although other materials like biochar, zeolites, and engineered nanomaterials are also employed (Sharma et al., 2022). The primary advantages of adsorption include its simplicity, cost-effectiveness, and the absence of harmful by-products. The finite adsorption capacity of materials, which necessitates their periodic regeneration or replacement (Sharma et al., 2024). Similar to coagulation-flocculation, the adsorption method also generates sludge, which consists of spent adsorbent materials saturated with contaminants (Kurniawan et al., 2021a). The management of this adsorption sludge is a critical issue, as it involves the disposal or regeneration of spent adsorbents. Regeneration can often be achieved through thermal, chemical, or biological methods to restore the adsorbent's capacity for reuse. However, these processes can be energy-intensive and costly, and not all adsorbents can be regenerated effectively. Disposal of non-regenerable adsorbents must be conducted in an environmentally safe manner, often requiring landfilling or incineration, which introduces additional costs and environmental concerns. The environmental impact and cost of managing adsorption sludge are important factors in assessing the overall sustainability of the adsorption process.

5.3.2 Ion exchange

Ion exchange is a specialized technique designed for removing dissolved ions from wastewater, functioning through the reversible exchange of ions between a liquid and a solid phase (Liu et al., 2023). Ion exchange relies on electrostatic interactions between charged particles, making it particularly suitable for targeting specific ionic contaminants such as heavy metals, hardness-causing ions (e.g., calcium and magnesium), and radionuclides (Liu et al., 2023). The performance of ion exchange systems is influenced by the resin or medium's ion exchange capacity, the leachate's flow rate, and the water's chemical composition, including the presence of competing

ions (Xu et al., 2024). Factors such as resin specificity, regeneration requirements, and operational parameters must be carefully managed to optimize performance (Clark et al., 2024). Ion exchange resins, synthetic or natural, are chosen based on their selectivity for certain ions, offering high specificity and efficiency in contaminant removal (Wang et al., 2024). Ion exchange offers the advantage of high specificity and efficiency for targeted contaminants, providing a tailored approach to leachate treatment (Aziz et al., 2023). The need for regular resin regeneration with chemicals like brine, which can generate secondary waste streams (Sahu, 2021). It is sensitive to competing ions, which can reduce the efficiency of targeted ion removal, requiring careful water chemistry management (Chen et al., 2023).

5.3.3 Simultaneous partial nitrification, anammox, and denitrification (SNAD)

SNAD is an emerging biological treatment technology that integrates partial nitritation, anaerobic ammonium oxidation (anammox), and denitrification processes within a single reactor (Singh et al., 2022). This integrated approach shows great potential for treating landfill leachate with high ammonia concentrations. The SNAD process involves three main steps: partial nitritation, which converts part of the ammonia into nitrite; anammox, which utilizes the remaining ammonia and the produced nitrite to generate nitrogen gas; and denitrification, which reduces any remaining nitrate to nitrogen gas. These processes occur simultaneously in the same reactor, achieving efficient nitrogen removal. SNAD achieves nitrogen removal efficiencies of 80%–94.56% for total inorganic nitrogen and over 90% for total nitrogen, while also reducing oxygen demand and sludge production compared to conventional methods (Zheng et al., 2016). SNAD technology offers several advantages for landfill leachate treatment, including high nitrogen removal efficiency, energy savings due to lower oxygen and organic carbon requirements compared to traditional nitrification-denitrification processes, reduced operational costs, smaller footprint due to the integration of three processes in one reactor, and less sludge production, which lowers sludge handling costs (Su et al., 2022). Despite its promising prospects, SNAD technology faces challenges such as the need for precise control of parameters like dissolved oxygen and pH, long startup times due to the slow growth of anammox bacteria, and sensitivity to temperature. Future research will focus on optimizing operational parameters, enhancing system stability, and exploring innovative reactor designs to further improve the application of SNAD technology in landfill leachate treatment.

In summary, when selecting an appropriate landfill leachate treatment technology, it is essential to consider the balance between efficiency, cost, and sustainability (Özdemir et al., 2020; Cherni et al., 2021; Dereli et al., 2021; Saadatlu et al., 2023). Membrane technologies are highly effective, capable of removing up to 99% of dissolved salts and other specific pollutants. However, their application involves significant capital and operational costs, particularly with RO, due to the expenses associated with energy use, membrane replacement, and maintenance. Despite these costs, advancements in technology are gradually improving the sustainability of membrane processes, although energy use and membrane disposal remain concerns. AOPs offer another robust solution, particularly for degrading resistant organic compounds and pathogens. Nevertheless, they come with high operational costs, mainly due to the chemical inputs and energy demands, such as for ozone generation or UV lamps. These factors also contribute to moderate sustainability concerns, especially regarding potential secondary pollution. Emerging technologies provide a more cost-effective and sustainable approach. While their efficiency varies depending on the materials and infrastructure used, they excel in targeting specific contaminants like heavy metals and nutrients. These technologies emphasize the use of recyclable materials and natural processes, thereby enhancing environmental friendliness. The choice of leachate treatment technology should be guided by specific treatment goals, regulatory requirements, and sustainability objectives, balancing efficiency with environmental and economic considerations to achieve effective and responsible leachate management.

6 Comprehensive treatment system

The cornerstone of addressing the leachate challenge lies in implementing a comprehensive treatment system that synergistically combines various treatment methodologies to remove contaminants effectively and meet or exceed regulatory discharge standards (Abdelfattah and El-Shamy, 2024). These systems are designed not only to mitigate the adverse environmental impacts of leachate but also to contribute to the sustainability of landfill operations by enabling the possibility of resource recovery and reuse of treated water (Mojiri et al., 2020).

6.1 Comprehensive treatment system

A comprehensive landfill leachate treatment system is a multi-faceted and dynamic approach that mitigates leachate contaminants' environmental impacts through a well-orchestrated sequence of processes (Figure 2) (Britz, 2020; Babaei et al., 2021; Fang et al., 2021; Righetto et al., 2021; Teng et al., 2021; Nath and Debnath, 2022). Initiating with pretreatment, the system begins by addressing the more tangible aspects of leachate—easily separable solids are removed, and the pH is carefully adjusted. This foundational step is about preparing the leachate for the complex journey ahead and safeguarding the intricate mechanisms of downstream processes from potential abrasion, blockages, and harmful chemical interactions. As the leachate advances, it encounters the primary treatment phase, a sophisticated interplay between biology and chemistry. Here, living microbes engage in a digestive ballet, utilizing both aerobic and anaerobic pathways to metabolize organic pollutants, thereby reducing the load of substances that demand oxygen in the water. Concurrently, a suite of physical-chemical treatments, each a targeted strike—coagulation, flocculation, and flotation—work in concert to clear out suspended solids, strip away color, and separate particulate organic matter. This systematic approach continues into the secondary treatment, where a more refined attack is launched on dissolved organic compounds that have eluded the grasp of biological processes. This stage employs the precision of AOPs, the fine sieve of membrane filtration technologies like reverse

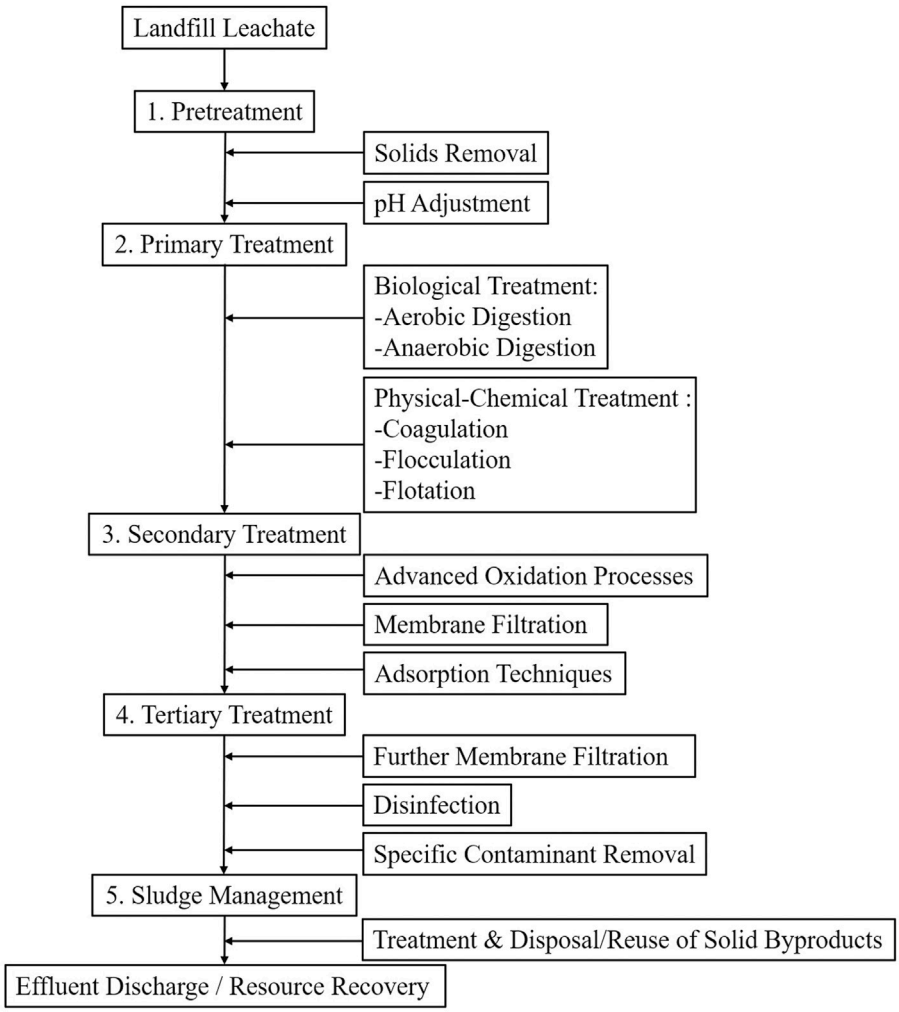


FIGURE 2
Components of a comprehensive treatment system.

osmosis, and the binding affinities of adsorption techniques, each method meticulously selected to ensure no contaminant is left unchallenged. Nearing the end, the leachate undergoes tertiary treatment—an exemplary display of purification, where the effluent is polished to meet stringent quality standards. Further, membrane filtration stands sentinel against the minutest of pollutants, disinfection processes wage war on pathogenic microorganisms, and additional specific treatments meticulously remove any lingering traces of heavy metals or nitrogen compounds.

The comprehensive landfill leachate treatment system's responsibility does not end with the effluent; sludge management remains a testament to the ethos of sustainability that underpins the entire operation (Torretta et al., 2017). This critical phase manages the solid byproducts of the leachate's treatment to minimize further environmental impact, reclaim valuable resources, and embody the cyclic nature of waste transformed into worth (Remmas et al., 2023). Implementing a comprehensive landfill leachate treatment system is a paragon of environmental responsibility and technological ingenuity, offering multifarious benefits that resonate far beyond the confines of waste management. Such systems are critical in preventing the percolation of hazardous contaminants into the groundwater and surface waters, safeguarding the aquatic ecosystems and public health against the potential onslaught of toxic substances (Ravindiran et al., 2023). By utilizing progressively targeted treatments that include biological degradation, chemical transformation, and physical separation, these systems adeptly reduce the concentrations of organic and inorganic pollutants, heavy metals, and pathogens to meet stringent regulatory standards (Rathod et al., 2024). Beyond the immediate environmental benefits, these systems embrace the principles of a circular economy, often allowing for the recovery of resources such as biogas, which can be converted to energy, and treated water, which can be repurposed for industrial or agricultural use (Zarei, 2020). This contributes to the conservation of natural resources and offers economic advantages by offsetting operational costs and generating revenue (Ekins and Zenghelis, 2021). Comprehensive treatment systems are committed to sustainable development as they adapt to the ever-changing waste composition and emerging contaminants, ensuring resilience against future environmental challenges. Thus, these systems' significance lies in their immediate efficacy in pollution control and their strategic contribution to long-term environmental sustainability and economic viability (Mojiri et al., 2020).

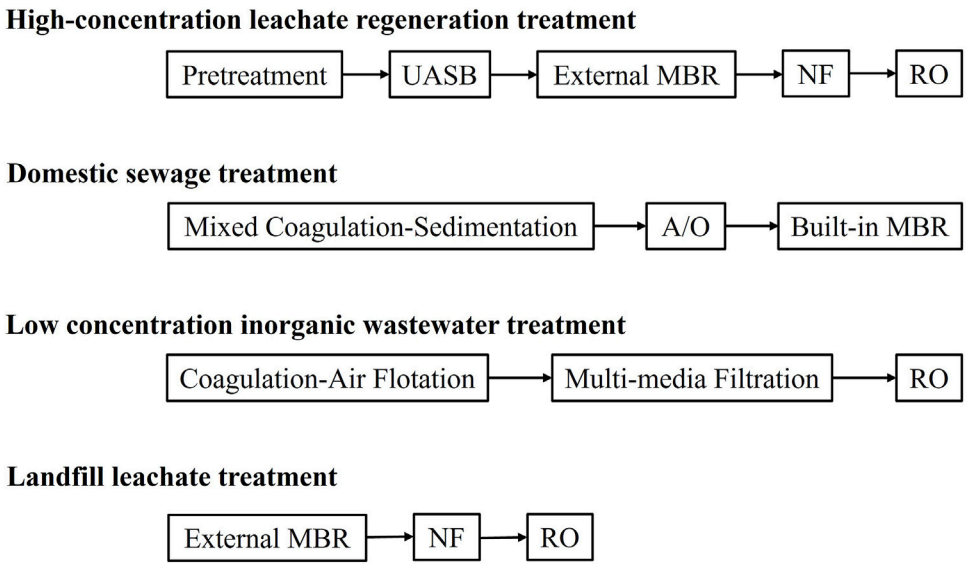


FIGURE 3
Comprehensive wastewater treatment system by Welle Group Co., Ltd, addressing high-concentration landfill leachate and domestic/low-concentration sewage through advanced multi-stage processes for effective contaminant removal and safe discharge or reuse.

6.2 Example of comprehensive treatment system for landfill leachate

In the bustling urban environment of Changsha City, the capital of Hunan Province, China (28.2282°N, 112.9388°E), located 35 km north in Qiaoyi Town, Wangcheng District, the Changsha Municipal Solid Waste Treatment Site stands as a pivotal solution to the city's waste management challenges. Operated by Welle Environmental Protection Technology Group Co., Ltd., this project exemplifies a cutting-edge approach to handling landfill leachate treatment (Welle Environmental Protection Technology Group Co.L, 2024). As the sole comprehensive co-processing site for municipal solid waste in Changsha, it is responsible for managing domestic waste from the city's expansive area, municipal sludge from 13 sewage treatment plants, and all leachate in the reservoir area. The facility is ingeniously divided into two functional units: landfill and incineration plant, with 80% of domestic waste processed through incineration for electricity generation and the remaining 20% allocated for landfilling. The core challenge in managing landfill leachate stems from its highly concentrated and variable composition, which can severely impact local water sources and the broader environment.

To tackle this, Welle Group Co., Ltd has designed a comprehensive leachate treatment system addressing both landfill leachate and domestic/low-concentration sewage (Figure 3). The process for high-concentration leachate regeneration is a testament to engineering excellence, employing a sophisticated combination of "pretreatment + up-flow anaerobic sludge blanket (UASB) + external membrane bioreactors (MBR) + nanofiltration (NF) + reverse osmosis (RO). This multi-stage approach ensures the thorough removal of contaminants, making the leachate safe for discharge or reuse. In addition to the high-concentration leachate treatment, the facility also addresses the treatment of domestic and low-concentration inorganic sewage. The domestic sewage treatment follows a "mixed coagulation and sedimentation + anaerobic-aerobic (A/O) + built-in MBR" process. In contrast, the low-concentration inorganic sewage is treated through "coagulation and air flotation + multi-media filtration + RO." These processes are designed to efficiently handle the varied waste streams entering the facility, ensuring that all forms of wastewater are effectively managed. The landfill leachate treatment, in particular, employs an "external MBR + NF + RO" process. This specific treatment path is crucial for dealing with the highly contaminated leachate from the landfill, effectively reducing its pollutant load before release or reuse. The concentrated liquid produced from these treatment processes undergoes advanced oxidation/reduction treatment, showcasing the facility's commitment to minimizing environmental impact. This liquid is then cleverly returned to the factory for consumption, embodying the principles of sustainability and resource recovery. The achievements of this project are both tangible and impactful. After undergoing the comprehensive treatment processes, the landfill leachate meets the stringent emission standards outlined in Table 2 of the "Pollution Control Standards for Domestic Waste Landfills" (GB16889-2008) (Table 10). Similarly, the effluent from the three sewage systems at the incineration plant complies with the "Water Quality for Industrial Water for Recycling of Urban Wastewater" (GB/T19923-2005), which sets the water quality standards for supplementary water in open circulating water systems. The project addresses the immediate need for waste and leachate treatment and contributes to resource conservation and pollution reduction goals. It stands as a testament to the power of technology and strategic planning in solving the complex environmental challenges of our time.

6.3 Optimizing integrated landfill leachate treatment systems

In environmental management, optimizing and modeling an integrated system for landfill leachate treatment serve as

TABLE 10 Emission standards in Table 2 of the 'pollution control standards for municipal solid waste landfills' (GB16889-2008).

Pollutant control item	Emission concentration limit
Color (Dilution times)	40
Chemical Oxygen Demand (COD_Cr) (mg/L)	100
Biochemical Oxygen Demand (BOD_5) (mg/L)	30
Suspended Solids (SS) (mg/L)	30
Total Nitrogen (TN) (mg/L)	30
Ammonia Nitrogen (NH_3-N) (mg/L)	25
Total Phosphorus (TP) (mg/L)	3
Fecal Coliform Group (CFU/L)	10,000
Total Mercury (Hg) (mg/L)	0.001
Total Cadmium (Cd) (mg/L)	0.01
Total Chromium (Cr) (mg/L)	0.1
Hexavalent Chromium (Cr^{6+}) (mg/L)	0.05
Total Arsenic (As) (mg/L)	0.1
Total Lead (Pb) (mg/L)	0.1

cornerstones for enhancing system efficiency, curbing expenditures, and adhering to environmental norms (Awewomom et al., 2024). The essence of this optimization lies in crafting a flexible treatment architecture that proficiently manages the intricate and diverse makeup of leachate while remaining nimble to fluctuations in waste inputs and legislative edicts. A synergy of suitable treatment modalities—such as UASB, MBR, NF, and RO—complements the specific leachate profile, ensuring that downstream processing is primed for optimal treatment (Huang et al., 2024). Operational parameters like retention time, aeration rate, and chemical dosing are meticulously fine-tuned to bolster the removal of pollutants, and real-time monitoring systems are deployed for on-the-fly adjustments. The model's sophistication extends to embodying resource recovery, advocating for repurposing byproducts like biogas into energy and advocating for the reuse of treated water.

The modeling facet delves into the assimilation of comprehensive data, which informs the simulation of the system's multifaceted treatment dynamics through advanced mathematical constructs (Ergene et al., 2022). These models are enriched with predictive analytics and scenario simulations, courtesy of machine learning algorithms, paving the way for a thorough understanding of potential operational outcomes. Validation through empirical data ensures the model's fidelity, while optimization algorithms distill the operational sweet spot that marries environmental efficacy with economic prudence (Podlasek, 2023). This ongoing, iterative process is honed by real-world performance feedback and shifting regulatory frameworks, propelling landfill leachate treatment toward unparalleled operational finesse.

7 Challenges and future

In addressing the complex challenges of landfill leachate treatment, it is crucial to consider the economic, operational, and emerging contaminant-related hurdles that impede the efficacy of current systems. This section delves into the multifaceted issues faced by waste management facilities, highlighting the significant financial burdens, operational difficulties due to leachate variability, and the rising threat of emerging contaminants like pharmaceuticals and microplastics. These challenges necessitate innovative solutions and strategic approaches to ensure effective, sustainable leachate management in the future.

7.1 Comprehensive challenges in landfill leachate treatment: Economic, operational, and emerging contaminant

The foremost economic challenge in landfill leachate treatment lies in the high capital and operational costs associated with advanced treatment systems (El-Saadony et al., 2023). Technologies such as RO, NF, and MBRs are effective in removing a wide range of contaminants but require substantial financial investment. These systems are not only expensive to install but also incur ongoing costs related to energy consumption, maintenance, and the need for skilled personnel to operate and monitor the processes. The financial burden is further amplified by the variability in leachate composition, which demands adaptable systems capable of handling fluctuations in pollutant concentrations. This variability necessitates continuous system optimization, leading to additional costs associated with adaptive management and process upgrades (Yadav et al., 2021). Table 11, which outlines the costs of various representative leachate treatment technologies, underscores the financial strain on waste management facilities (Gerald et al., 1998; Randall and Ubay Cokgor, 2000; Mahamuni and Adewuyi, 2010; Ju-Chang Huang, 2017; Ighalo et al., 2022; Pérez et al., 2022; Kurniawan et al., 2023). Moreover, the costs are not merely limited to initial capital outlays but extend to operational expenses that accumulate over the system's lifecycle (Abdelfattah and El-Shamy, 2024). As such, the economic challenges of landfill leachate treatment call for innovative approaches to reduce costs while maintaining treatment efficacy.

Operationally, the treatment of landfill leachate is fraught with difficulties due to the high variability of leachate quality. This variability is influenced by factors such as seasonal changes, waste composition, and the age of the landfill, all of which affect the concentration and types of pollutants present in the leachate. The dynamic nature of leachate composition requires continuous monitoring and adjustment of treatment processes to ensure consistent performance (Upadhyay et al., 2023). Operators must continuously monitor and adjust treatment processes, requiring skilled personnel and sophisticated control systems. The potential for system failure or suboptimal performance poses significant environmental risks, including groundwater contamination and non-compliance with discharge regulations, leading to potential fines and reputational damage. Addressing these operational challenges requires a holistic approach that integrates cost-effective treatment technologies with operational efficiency and

TABLE 11 Costs of representative leachate treatment technologies (Gerald et al., 1998; Randall and Ubay Cokgor, 2000; Mahamuni and Adewuyi, 2010; Huang, 2017; Ighalo et al., 2022; Pérez et al., 2022; Kurniawan et al., 2023).

Treatment method	Approximate cost (/m^3 of treated wastewater)
Adsorption	0.038–51,120 USD The main cost is depended on the adsorbent Adsorbent Cost Performance: - Very cheap: <1 $/mol chemical species - Most adsorbents: 1–200 $/mol chemical species - Very expensive: >200 $/mol chemical species
Air-stripping	0.0134 USD
Reverse osmosis	0.5 USD
Biological nutrient removal	0.615USD
Ultrafiltration	0.69 USD
Advanced oxidation processes	US/UV/O_3: 23.717 USDUS/UV/H_2O_2: 17.216 USDUS/UV: 6.821 USD

environmental stewardship (Kundariya et al., 2021). The development of more energy-efficient membrane technologies or the adoption of natural treatment solutions, such as constructed wetlands, offers promising avenues for reducing operational difficulties and associated costs (Kataki et al., 2021). For instance, the development of more energy-efficient membrane technologies or the adoption of natural treatment solutions, such as constructed wetlands, offers promising avenues for reducing operational difficulties and associated costs. Moreover, embracing circular economy principles, where treated leachate is reused for industrial or agricultural purposes, can help mitigate operational challenges while contributing to sustainability goals. (Bakan et al., 2022; Das et al., 2023).

The emergence of new contaminants, such as pharmaceuticals, personal care products, microplastics, and other micropollutants, presents a significant challenge to traditional wastewater and landfill leachate treatment systems (Kumar et al., 2023). These substances, often called emerging contaminants (ECs), have become a focal point for environmental scientists and engineers due to their potential to bypass conventional treatment processes and enter the aquatic environment, posing risks to wildlife and human health (Ahmed et al., 2021; Morin-Crini et al., 2021). Traditional wastewater and landfill leachate treatment systems were designed to remove organic matter, nutrients, and major pollutants (Lindamulla et al., 2022). However, these systems often lack the specificity to capture and degrade ECs effectively, many of which are designed to be biologically active and resistant to degradation (Teng and Chen, 2023). Pharmaceuticals, for example, are engineered to resist metabolic breakdown to ensure their efficacy, making them persistent in the environment (González-González et al., 2022). Microplastics, conversely, are resistant to biodegradation due to their synthetic polymer composition, accumulating in water bodies and potentially entering the food chain (Huang et al., 2021). The presence of ECs in water and leachate streams can significantly reduce the treatment efficiency of these systems (Rout et al., 2021b). Certain contaminants can inhibit microbial activity in biological treatment processes, reducing the degradation rate of organic pollutants and altering the microbial community structure (Aragaw, 2021). For example, antibiotics in wastewater can inhibit the growth of bacteria critical to the biological treatment processes, leading to a decrease in overall treatment efficiency (Zhu et al., 2021). Moreover, some ECs can form complex compounds with other elements in the wastewater, making them more difficult to remove and requiring additional treatment stages or more advanced treatment technologies (Shahid et al., 2021). Notably, based on the environmental risks and regulatory status of chemicals, and following technical feasibility and socioeconomic assessments, the Ministry of Ecology and Environment of China published the latest list of key controlled emerging contaminants in 2023 (Table 12) (China, 2022). The newly listed contaminants are now subject to environmental risk management measures, including those related to their manufacture, use, import, or export. Some contaminants have transition periods or exemptions. This initiative highlights the government's commitment to addressing emerging contaminants and urges relevant departments to take action to prevent environmental and public health risks. For landfill leachate treatment, the presence of these new pollutants complicates the process, often requiring upgrades or advanced technologies to effectively manage them. Proactively addressing these contaminants can improve treatment efficiency and reduce environmental contamination risks. It is worth mentioning that, Per- and polyfluoroalkyl substances (PFAS) are synthetic chemicals widely used for their water- and grease-resistant properties, found in various consumer products and industrial applications (Venkatesh Reddy et al., 2024). Their strong carbon-fluorine bonds make them highly persistent and resistant to conventional water treatment methods, posing significant challenges for removal (Jin et al., 2021). Traditional processes like coagulation, sedimentation, and standard filtration are largely ineffective against PFAS, necessitating the use of advanced technologies (Nguyen and Le Tran, 2024). Methods such as granular activated carbon adsorption, ion exchange, and high-pressure membrane filtration (nanofiltration and reverse osmosis) show varying degrees of success but often involve high operational costs and complex waste management (Lee et al., 2022). Emerging technologies like electrochemical treatment, plasma-based processes, and combined treatment approaches offer potential but require further development and investment (Blotevogel et al., 2023). The inclusion of PFAS-specific treatments increases operational complexity and costs for water treatment facilities, which must adapt to stringent regulatory standards. Research continues to focus on more efficient and cost-effective removal technologies to address the persistent nature of PFAS and protect public health and the environment.

The impact of emerging contaminants on the efficiency of wastewater and landfill leachate treatment systems presents a complex challenge that requires immediate and sustained attention. While these contaminants pose significant environmental and public health risks, they also provide an impetus for innovation in treatment technologies and strategies. Addressing this challenge will require a combination of technological advancement, regulatory action, public education, and international cooperation to ensure the safety and

TABLE 12 2023 List of Key Controlled New Pollutants posted by Ministry of Ecology and Environment of the People's Republic of China (China, 2022).

No.	New pollutant name	CAS no.	Main environmental risk control measures
1	Perfluorooctane sulfonic acid (PFOS) and its salts and perfluorooctane sulfonyl fluoride	1763–23–1 307–35–7 2795–39–3 29,457–72–5 29,081–56–9 70,225–14–8 56,773–42–3 251,099–16–8	1. Prohibit production. 2. Prohibit processing and use (except for certain uses such as firefighting foam until 31 Dec 2023). 3. Conduct mandatory clean production audits for companies using PFOS for firefighting foam. 4. Require environmental management clearance for import/export until 1 Jan 2024. 5. Manage as hazardous waste if no longer in use or declared as waste 6. Establish soil pollution prevention measures for enterprises involving PFOS.
2	Perfluorooctanoic acid (PFOA) and its salts and related compounds	—	1. Prohibit new production facilities. 2. Prohibit production and use (except for certain uses in semiconductor manufacturing, photography coatings, protective textiles, invasive medical devices, and other specified uses). 3. Conduct mandatory clean production audits for companies using PFOA. 4. Require environmental management clearance for import/export. 5. Manage as hazardous waste if no longer in use or declared as waste 6. Establish soil pollution prevention measures for enterprises involving PFOA.
3	Decabromodiphenyl ether	1163–19–5	1. Prohibit production and use (except for certain uses in textiles, plastic casings, polyurethane foam until 31 Dec 2023). 2. Conduct mandatory clean production audits for companies using decabromodiphenyl ether. 3. Require environmental management clearance for import/export until 1 Jan 2024. 4. Manage as hazardous waste if no longer in use or declared as waste 5. Establish soil pollution prevention measures for enterprises involving decabromodiphenyl ether
4	Short-chain chlorinated paraffins	85,535–84–8 68,920–70–7 71,011–12–6 85,536–22–7 85,681–73–8 108,171–26–2	1. Prohibit production and use (except for certain uses in rubber, leather, lubricants, decorative lights, paints, adhesives, metal processing, and flexible PVC until 31 Dec 2023). 2. Conduct mandatory clean production audits for companies using short-chain chlorinated paraffins. 3. Require environmental management clearance for import/export until 1 Jan 2024. 4. Manage as hazardous waste if no longer in use or declared as waste 5. Establish soil pollution prevention measures for enterprises involving short-chain chlorinated paraffins
5	Hexachlorobutadiene	87–68–3	1. Prohibit production, use, and import/export. 2. Ensure compliance with emission standards for related enterprises. 3. Manage as hazardous waste if no longer in use or declared as waste 4. Establish soil pollution prevention measures for enterprises involving hexachlorobutadiene
6	Pentachlorophenol and its salts and esters	87–86–5 131–52–2 27,735–64–4 3772–94–9 1825–21–4	1. Prohibit production, use, and import/export 2. Manage as hazardous waste if no longer in use or declared as waste 3. Establish soil pollution prevention measures for enterprises involving pentachlorophenol
7	Trifluralin	115–32–2 10,606–46–9	1. Prohibit production, use, and import/export. 2. Manage as hazardous waste if no longer in use or declared as waste
8	Perfluorohexane sulfonic acid (PFHxS) and its salts and related compounds	—	1. Prohibit production, use, and import/export. 2. Manage as hazardous waste if no longer in use or declared as waste
9	Dechlorane Plus and its isomers	13,560–89–9 135,821–03–3 135,821–74–8	1. Prohibit production, use, and import/export from 1 Jan 2024 2. Manage as hazardous waste if no longer in use or declared as waste
10	Dichloromethane	75–09–2	1. Prohibit production of paint removers containing dichloromethane 2. Prohibit use in cosmetics 3. Ensure compliance with volatile organic compounds content limits and emission standards. 4. Conduct regular environmental risk assessments. 5. Establish soil pollution prevention measures for enterprises involving dichloromethane. 6. Strictly control soil pollution risks

(Continued on following page)

TABLE 12 (*Continued*) 2023 List of Key Controlled New Pollutants posted by Ministry of Ecology and Environment of the People's Republic of China (China, 2022).

No.	New pollutant name	CAS no.	Main environmental risk control measures
11	Chloroform	67–66–3	1. Prohibit production of paint removers containing chloroform 2. Ensure compliance with volatile organic compounds content limits and emission standards. 3. Conduct regular environmental risk assessments. 4. Establish soil pollution prevention measures for enterprises involving chloroform. 5. Strictly control soil pollution risks
12	Nonylphenol	25,154–52–3 84,852–15–3	1. Prohibit use as pesticide adjuvants. 2. Prohibit use in the production of nonylphenol ethoxylates. 3. Prohibit use in cosmetics
13	Antibiotics	—	1. Strictly enforce prescription requirements for antibacterial drugs. 2. Manage antibiotic residues as hazardous waste. 3. Ensure compliance with emission standards for pharmaceutical industries
14	Phased-out group (hexabromocyclododecane, chlordane, mirex, hexachlorobenzene, DDT, α-hexachlorocyclohexane, β-hexachlorocyclohexane, lindane, endosulfan as a raw material and its isomers, and polychlorinated biphenyls)	—	1. Prohibit production, use, and import/export. 2. Manage as hazardous waste if no longer in use or declared as waste. 3. Strictly control soil pollution risks according to soil pollution risk management standards

sustainability of water resources in the face of emerging environmental threats.

7.2 Future research

Embracing the future of landfill leachate treatment demands a multi-faceted approach: pioneering new materials for adsorption and catalysis, harnessing microbial genetic engineering for targeted bioremediation, and committing to process intensification for enhanced efficiency. These innovative pathways herald a new era of sustainability and effectiveness in managing complex waste challenges.

7.2.1 New materials for adsorption and catalysis

Future research in landfill leachate treatment increasingly focuses on developing new materials for adsorption and catalysis, driven by the need for more efficient, cost-effective, and sustainable treatment solutions. These materials are at the forefront of addressing complex pollutants, including emerging contaminants that traditional treatment methods struggle to remove. Advanced adsorbents, such as modified biochars, nanocomposites, and metal-organic frameworks (MOFs), are being explored for their high surface area, porosity, and tunable surface chemistry, which can significantly enhance the adsorption of pollutants (Zango et al., 2020). For instance, biochars produced from agricultural waste can be engineered at the nano-scale to improve their adsorption capacity for specific contaminants like pharmaceuticals and heavy metals (Bhandari et al., 2023). Catalytic materials are equally crucial for breaking down pollutants through AOPs. New photocatalysts, including quantum dots and graphitic carbon nitride (g-C_3N_4), offer the potential for more efficient light absorption and utilization, enabling the degradation of contaminants under visible light and thus reducing energy consumption (Wang et al., 2020). The development of these materials aims to enhance the efficiency and specificity of contaminant removal and minimize secondary pollution and the use of harsh chemicals in the treatment process (Ahmed et al., 2021). Research also focuses on the regeneration and reuse of these materials to ensure the sustainability of the treatment processes (Baskar et al., 2022). Innovative approaches to regenerate spent adsorbents and catalysts, such as thermal, chemical, and ultrasonic methods, are being studied to extend their life and reduce operational costs (Ferella, 2020). Overall, exploring new materials for adsorption and catalysis represents a dynamic and promising area of landfill leachate treatment, offering potential breakthroughs in efficiency, sustainability, and tackling a broader spectrum of pollutants.

7.2.2 Microbial genetic engineering

Microbial genetic engineering stands at the forefront of biotechnological innovations in landfill leachate treatment (Saeed et al., 2022). The natural biodegradation processes employed by microorganisms offer a sustainable and efficient means of removing organic pollutants from leachate (Pisharody et al., 2022). However, leachate's complex and toxic nature often limits the effectiveness of native microbial communities (Nimonkar et al., 2022). Genetic engineering presents an opportunity to overcome these limitations by enhancing the metabolic capabilities of microorganisms, enabling them to degrade a wider array of contaminants, including recalcitrant and emerging pollutants (Haripriyan et al., 2022). The manipulation of microbial genomes involves introducing genes responsible for producing specific enzymes that can break down complex molecules or modifying existing metabolic pathways to increase their efficiency and specificity (Michalska et al., 2020). This can lead to the creation of microbial strains capable of rapid pollutant degradation, reduced biomass production, and enhanced tolerance to high pollutant concentrations and toxic substances (Maqsood et al., 2023). Ongoing research focuses on identifying key enzymatic pathways for the degradation of specific contaminants, developing genetic tools for efficient genome editing, and integrating genetically engineered microbes into existing treatment processes (Saravanan et al., 2022). Additionally, the ecological safety of releasing genetically modified organisms into environmental treatment

systems is a critical consideration, necessitating rigorous containment strategies and risk assessments.

7.2.3 Process enhancement

Process intensification in landfill leachate treatment aims to enhance the efficiency and effectiveness of treatment processes by integrating and optimizing physical, chemical, and biological methods. The goal is to achieve more with less—reducing the footprint, energy consumption, and operational costs of treatment systems while maximizing pollutant removal efficiency (Cardoso et al., 2021). This can involve the development of novel reactor designs that facilitate enhanced contact between the leachate and treatment agents, using AOPs to break down complex organic molecules, and integrating membrane technologies for selective pollutant removal and water recovery (Titchou et al., 2021). Process intensification can leverage the synergies between different treatment modalities, such as combining adsorption processes with biodegradation to handle a broader range of contaminants more effectively.

Future research directions include exploring hybrid treatment systems that integrate emerging materials for adsorption and catalysis with genetically engineered microbial processes, developing energy-efficient AOPs, and optimizing process parameters through advanced modeling and control strategies. The aim is to develop compact, modular treatment systems that can be easily adapted to varying leachate compositions and volumes, ensuring robust and sustainable leachate management strategies for the future.

7.2.4 Circular economy and sustainability in future landfill leachate treatment

The circular economy and sustainability principles are increasingly influencing the development and adoption of future landfill leachate treatment technologies, steering them towards more environmentally friendly and resource-efficient solutions (Bandala et al., 2021). This shift is a response to the growing awareness of the environmental impacts of waste management practices and the urgent need to transition towards more sustainable models of production and consumption. The role of circular economy concepts in shaping future leachate treatment technologies is profound, focusing on waste minimization, resource recovery, and the closed-loop reuse of materials and water (Yusuf et al., 2020).

At the heart of the circular economy is reducing waste and preventing pollution (Dincă et al., 2022). Future landfill leachate treatment technologies are being designed with a preventative approach, aiming to minimize the generation of leachate and its associated contaminants at the source (Bandala et al., 2021). This can be achieved through better landfill design, including impermeable liners and covers and waste pre-treatment to reduce its leachate-generating potential. By addressing the problem at its source, these technologies contribute to a more sustainable and less resource-intensive waste management system. Another cornerstone of the circular economy is the recovery and reuse of resources, which plays a significant role in shaping leachate treatment technologies (Puntillo et al., 2021). Emerging technologies are not only aimed at treating leachate to meet environmental discharge standards but also at recovering valuable resources from it (Li and Liu, 2021). Nutrient recovery, for example, can transform leachate into a source of nitrogen and phosphorus for agricultural use (Kurniawan et al., 2021b). Similarly, advanced treatment processes can recover water for irrigation or industrial applications, contributing to water conservation and reducing the demand for freshwater resources (Meese et al., 2022). The recovery of energy in the form of biogas from the anaerobic treatment of leachate further underscores the potential of these technologies to contribute to a more sustainable and energy-efficient waste management system (An et al., 2023). The circular economy encourages a systemic view of waste management, promoting the integration of leachate treatment technologies within broader waste and water management systems (Smol et al., 2020). Future technologies will likely be characterized by their ability to integrate with other treatment and recovery processes, creating synergies that enhance overall efficiency and sustainability (Kehrein et al., 2020). For example, integrating leachate treatment with solid waste management processes can facilitate the co-treatment of organic waste and leachate, optimizing resource recovery and minimizing environmental impacts (Mishra et al., 2023).

Collectively, the influence of circular economy and sustainability principles on the future of landfill leachate treatment technologies is transformative, driving innovation towards solutions that are effective in treating leachate and minimizing waste, recovering resources, and integrating seamlessly into a circular and sustainable waste management framework. As these principles continue to gain traction, they will undoubtedly shape the development of more sustainable, efficient, and integrated leachate treatment technologies, aligning waste management practices with the broader goals of environmental sustainability and resource conservation.

8 Conclusion

The environmental impact of untreated leachate is significant, with potential contamination of groundwater, surface water, and overall ecosystem health, necessitating robust and dynamic treatment solutions.

This comprehensive review of landfill leachate treatment technologies presents a global perspective, focusing on the physicochemical properties of leachate, regulatory frameworks, and specific country examples. The study highlights the complex composition of landfill leachate, which includes a mix of organic and inorganic pollutants, heavy metals, and xenobiotic compounds. The review also underscores the importance of global regulatory frameworks in shaping landfill leachate management practices. International agreements, such as the Basel Convention and the Stockholm Convention, along with national regulations from countries like the United States, China, India, and Brazil, are setting standards and encouraging the adoption of advanced treatment technologies.

Key findings emphasize the evolution of conventional and advanced treatment technologies. Traditional methods such as physical, chemical, and biological treatments remain fundamental. Physical methods, including sedimentation and membrane filtration, effectively remove suspended solids and larger organic molecules. Chemical treatments like coagulation-flocculation and advanced oxidation processes are crucial for targeting dissolved pollutants and heavy metals. Biological treatments utilize microbial processes to degrade organic pollutants and are particularly effective for nutrient removal.

Advanced treatment technologies, including MBRs, NF, RO, and AOPs), provide higher efficiency and are essential for meeting stringent regulatory standards. These technologies can remove a broader range of contaminants, including emerging pollutants such as pharmaceuticals and microplastics, which traditional methods may not adequately address. Economic considerations are also crucial in the treatment process. The high capital and operational costs of advanced leachate treatment systems pose significant challenges. Technologies such as RO, NF, and MBR are effective but require substantial investment in equipment, maintenance, and energy consumption. Addressing these economic challenges necessitates the development of cost-effective, energy-efficient solutions and the integrating of resource recovery practices, such as biogas production and water reuse, to offset operational costs. A case study from Changsha City illustrates the practical application and effectiveness of a comprehensive treatment system that integrates various technologies. This example provides valuable insights into designing integrated systems capable of meeting diverse regulatory requirements and addressing complex environmental challenges.

In conclusion, the continuous improvement of landfill leachate treatment technologies is vital. Adopting innovative and sustainable practices is essential to reduce the environmental footprint of landfills. Future research should focus on developing cost-effective, energy-efficient solutions and exploring emerging technologies for better pollutant removal. By emphasizing a global perspective and integrating multiple treatment processes, this review significantly contributes to advancing sustainable waste management practices and protecting environmental health.

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JW: Conceptualization, Data curation, Formal Analysis, Resources, Software, Writing–original draft, Writing–review and editing. ZQ: Conceptualization, Project administration, Supervision, Writing–original draft, Writing–review and editing.

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Critical Review

Per- and Polyfluoroalkyl Substance Toxicity and Human Health Review: Current State of Knowledge and Strategies for Informing Future Research

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Abstract: Reports of environmental and human health impacts of per- and polyfluoroalkyl substances (PFAS) have greatly increased in the peer-reviewed literature. The goals of the present review are to assess the state of the science regarding toxicological effects of PFAS and to develop strategies for advancing knowledge on the health effects of this large family of chemicals. Currently, much of the toxicity data available for PFAS are for a handful of chemicals, primarily legacy PFAS such as perfluorooctanoic acid and perfluorooctane sulfonate. Epidemiological studies have revealed associations between exposure to specific PFAS and a variety of health effects, including altered immune and thyroid function, liver disease, lipid and insulin dysregulation, kidney disease, adverse reproductive and developmental outcomes, and cancer. Concordance with experimental animal data exists for many of these effects. However, information on modes of action and adverse outcome pathways must be expanded, and profound differences in PFAS toxicokinetic properties must be considered in understanding differences in responses between the sexes and among species and life stages. With many health effects noted for a relatively few example compounds and hundreds of other PFAS in commerce lacking toxicity data, more contemporary and high-throughput approaches such as read-across, molecular dynamics, and protein modeling are proposed to accelerate the development of toxicity information on emerging and legacy PFAS, individually and as mixtures. In addition, an appropriate degree of precaution, given what is already known from the PFAS examples noted, may be needed to protect human health. *Environ Toxicol Chem* 2021;40:606–630. © 2020 SETAC

Keywords: Per- and polyfluoroalkyl substances; Perfluorooctane sulfonate; Perfluorooctanoic acid; Persistent compounds; Contaminants of emerging concern

INTRODUCTION

Per- and polyfluoroalkyl substances (PFAS) are ubiquitous in environmental media because of their prolific use in a variety of industrial and consumer products and processes (Jian et al. 2018; Sunderland et al. 2019). Widespread human exposure to PFAS in water, food, and air coupled with the lengthy environmental persistence and biological half-lives of some PFAS have led to measurable PFAS in the blood of nearly the entire population in developed countries, with health effects reported globally (Kato et al. 2011; Khalil et al. 2016; Stubleski et al. 2016; Jian et al. 2018). Information needed to evaluate the potential risk of harm from PFAS includes the types of adverse health effects that might occur at environmentally relevant exposures, especially in sensitive life stages. Information is also needed regarding the mode(s) of action for

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PFAS toxicity, PFAS toxicokinetics in both humans and laboratory animal models, and dose–response relationships. Risk estimates can be used to inform public health exposure limits that will determine the need for exposure mitigation and environmental cleanup.

There are several challenges in obtaining the information needed to assess human health risk from the large number of PFAS with a wide range of structures and chemical properties (Buck et al. 2011; Wang Z et al. 2017; Organisation for Economic Co-operation Development 2018). Data on the identity, composition, and quantity of PFAS used in products and processes are often treated as confidential business information, hampering efforts to estimate exposure sources and routes. The Organisation for Economic Co-operation and Development's (OECD's) chemical inventory reports over 4000 substances that contain at least one perfluoroalkyl (–CnF2n–) moiety (Organisation for Economic Co-operation Development 2018), and the US Environmental Protection Agency (USEPA) has a curated list of over 8000 PFAS included, based on structure (US Environmental Protection Agency 2018) from the CompTox Chemicals Dashboard (Williams et al. 2017). The USEPA estimates that more than 600 PFAS are currently in commercial use (US Environmental Protection Agency 2019). Experimental studies of PFAS have been limited by funding and the availability of analytical standards, confounded by the prevalence of background contamination in laboratory materials, and challenged by physicochemical properties such as high surface activity that can interfere with and complicate measurements. Consequently, sufficient information to conduct quantitative risk assessment is currently available for only a relative few PFAS (Post 2020). Further, although typical human exposures involve various combinations of PFAS (Centers for Disease Control and Prevention 2017), only a few efforts address interactions of PFAS mixtures; and a well-founded, scientific basis on which to evaluate their combined toxic potential does not yet exist (Carr et al. 2013; Wolf et al. 2014; Zhou et al. 2017; Hoover et al. 2019; US Environmental Protection Agency 2020).

The Society of Environmental Toxicology and Chemistry (SETAC) North America held the focused topic meeting and workshop "Environmental Risk Assessment of PFAS" on 12 to 15 August 2019, covering a wide range of topics related to the characterization of health risks posed by PFAS. The overarching purpose of the meeting was to begin a scientific discussion on how best to approach studying, grouping, and regulating the large number of PFAS to which people and other species are potentially exposed (for charge questions and other details, see Johnson et al. 2020). We refer to these PFAS as "legacy" (those perfluoroalkyl acids for which there are accumulating health data but that may be phased out or decreased in use) and "emerging" (those which are being used as replacements, often with minimal health effects data). The objectives of the Human Health Toxicity section were to provide an assessment of the state of the science in understanding toxicological effects of PFAS and to explore and discuss strategies for advancing knowledge on the toxicity of individual and groups of PFAS.

CURRENT KNOWLEDGE OF PFAS TOXICITY IN HUMANS

Like other chemicals, PFAS are potentially capable of producing a wide range of adverse health effects depending on the circumstances of exposure (magnitude, duration, and route of exposures, etc.) and factors associated with the individuals exposed (e.g., age, sex, ethnicity, health status, and genetic predisposition). Aspects to consider when establishing the health effects of greatest concern are 1) effects for which evidence is the strongest (strength of evidence can come from consistency of effect across studies, strength of effect associations in epidemiological studies, and species concordance, as examples), and 2) effects for which potential impact is greatest (factors contributing to impact can include severity of effect, functional impairment, persistence, and specific age groups that are susceptible, as examples). Brief summaries of candidate PFAS health effects from human and experimental reports are provided in this section (Figure 1).

Immune function

Epidemiological studies have explored relationships between PFAS exposure and laboratory biomarkers of immunomodulation, such as vaccine responses. A doubling of perfluorooctane sulfonate (PFOS) in maternal serum was associated with a 39% ($p < 0.001$) reduction in diphtheria antibody concentration in children (age 5 yr), with increased odds of falling below clinically protective values against diphtheria and tetanus at age 7 yr. The authors noted that a "2-fold greater concentration of major PFCs [perfluorinated compounds] in child serum was associated with a difference of −49% (95% CI, −67% to −23%) in the overall antibody concentration" (Grandjean et al. 2012). Decreased immunological response persisted at age 13 yr (Grandjean et al. 2017). Adverse associations were also noted for responses to rubella, mumps, and *Hemophilus* influenza vaccinations in children and to vaccinations in adults (Granum et al. 2013; Looker et al. 2014; Stein et al. 2016; Abraham et al. 2020). In a single study, modest down-regulation of C-reactive protein response, a marker of human systemic inflammation, was also reported to be associated with perfluorooctanoic acid (PFOA) blood levels (Genser et al. 2015).

Disease outcomes linked with immunosuppression such as clinician-recorded diagnoses of childhood infections have also been associated with prenatal exposures to PFOS and perfluorohexane sulfonate (PFHxS) (Goudarzi et al. 2017). A pregnancy cohort study prospectively detected increased risk of airway and throat infections and diarrhea in children through age 10 yr, correlated with cord-blood PFAS measurements (Impinen et al. 2018, 2019). A recent review concluded that exposure to PFAS in infancy and childhood resulted in an immunosuppressive effect characterized by an increased incidence of atopic dermatitis and lower respiratory tract infections (Kvalem et al. 2020). Some of the immunological effects were sex-specific, but the authors cautioned that there were inconsistencies across studies (Kvalem et al. 2020).

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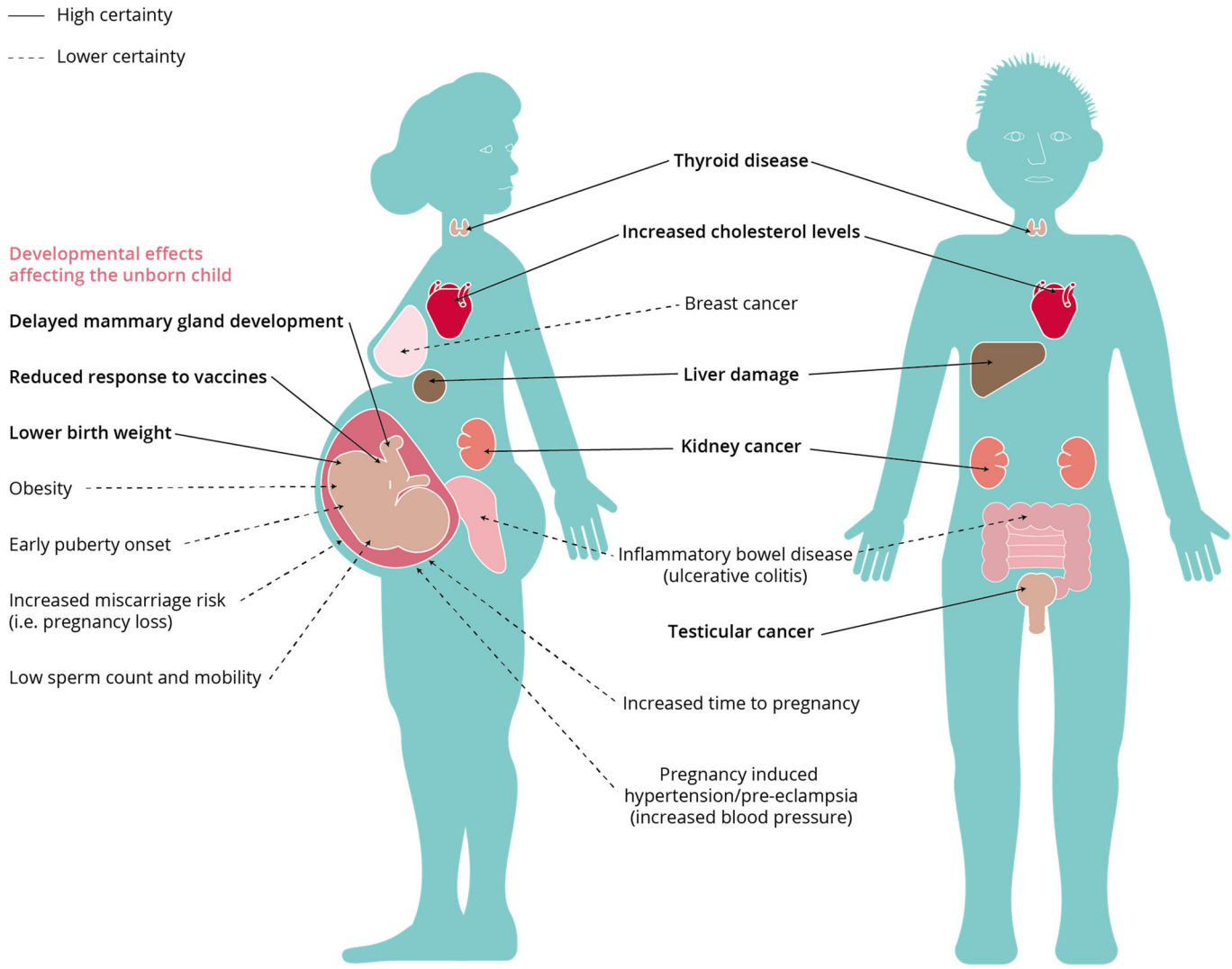


FIGURE 1: Effects of per- and polyfluoroalkyl substances on human health. Used with permission from European Environment Agency (2019). Original sources for this figure: National Toxicology Program (2016), C8 Science Panel (2012), IARC Working Group on the Evaluation of Carcinogenic Risks to Humans (2017), Barry et al. (2013), Fenton et al. (2009), and White et al. (2011b).

Overall, available data provide strong evidence that PFAS exposure can suppress the human immune response.

Population studies of immune hyperreactive diseases have resulted in mixed findings. Studies on childhood allergy and asthma outcomes have shown no association with PFAS (Impinen et al. 2018, 2019), whereas others have found substantial effects, including provocative evidence that subgroups of individuals not adequately immunized may be at an increased risk for disease a priori (Qin et al. 2017; Timmermann et al. 2017a). For example, a case–control study of Taiwanese children compared the first and fourth quartiles of serum measurements for 11 PFAS with asthma and other immune markers and reported confidence intervals well above 1.0 for PFOA and others (Qin et al. 2017). However, review articles concerning PFAS and childhood allergy and asthma offer nuanced, age- and sex-specific interpretations and advise against firm conclusions (Kvalem et al. 2020).

Chronic autoimmune outcomes, including thyroid disease (see section *Thyroid function*) and inflammatory bowel disease (IBD), have also been considered. A study in contaminated communities ($n = 32\,254$) detected an association between both prevalence and incidence of ulcerative colitis (UC) and PFOA exposure (linear trend $p = 0.0001$ [Steenland et al. 2013]). A worker study ($n = 3713$) found a higher prevalence ($p = 0.01$) and incidence ($p < 0.05$) of UC with increasing log PFOA serum concentrations (Steenland et al. 2015). A case–control study of children and young adults from a background exposure community in Atlanta, Georgia, USA, also found higher serum PFOA levels in patients with UC (Steenland et al. 2018b). In contrast to PFOA-related associations in US populations, a study of a contaminated community in Sweden ($n = 63\,074$) did not show a consistent association of IBD with any PFAS exposure (Xu et al. 2020b).

Recent, thorough reviews (National Toxicology Program 2016; DeWitt et al. 2019; Pachkowski et al. 2019) emphasize some key concepts: 1) there is concordance between animal studies and human epidemiological observations that PFAS modify the immune response, and 2) there are noted

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complexities in assuming dose–response continuums, including possible differences in life-stage vulnerability. Authors of these reviews note uncertainty about which outcome will be of most importance but agree that immunotoxicity should be included among sensitive human PFAS toxicity endpoints.

Thyroid function

The C8 Science Panelists concluded that there is a "probable link" of PFOA exposure to thyroid disease, with sex-specific outcomes in women (for hyperthyroid disease) versus men (hypothyroid disease) (C8 Science Panel 2012). Subsequent reviews drew attention to hypothyroid outcomes in women and children and to the possibility that populations with a priori circulating antithyroid peroxidase antibodies may be at additional risk (Coperchini et al. 2017). A broad childhood disease review noted "some evidence" that PFAS cause childhood hypothyroidism and characterized the number of studies as "limited" for childhood disease conclusions (Rappazzo et al. 2017). A meta-analysis of 12 child and adult studies that excluded populations with higher exposures noted that PFAS exposure is negatively associated with serum total thyroxine levels and that "PFAS could induce thyroid dysfunction and disease" (Lee and Choi 2017).

Human thyroid disease is mostly the result of an autoimmune response and is 5 to 10 times more prevalent in women than men (Tadic et al. 2018). Concerning PFAS and clinically diagnosed outcomes, women in the highest quartile of PFOA exposure (>5.7 ng/mL) reported clinical hypothyroid disease (odds ratio 2.2, 95% confidence interval [CI] 1.4–3.7) over 3 cycles of National Health and Nutrition Examination Survey (NHANES) data (1999–2006, $n = 3974$ adults), with similar findings in men (Melzer et al. 2010). The C8 Science Panel studies (median serum PFOA 26.1 ng/mL) found thyroid disease hazard ratios of 1.00, 1.24, 1.27, 1.36, and 1.37 across cumulative exposure quintiles in women (log-linear trend $p = 0.03$ [Winquist and Steenland 2014b]), with parallel hypothyroid findings in children aged 1 to 17 yr (Lopez-Espinosa et al. 2012). The Ronneby, Sweden, population experienced excess risk of thyroid disease in a discrete time period (1984–2005) among women (hazard ratio 1.29, 95% CI 1.05–1.57) that did not persist over time despite higher cumulative PFAS exposure (Andersson et al. 2019). The authors did not link exposure to hypothyroid outcome, noting a nonmonotonic dose–response relationship (Andersson et al. 2019).

Human population studies augment experimental data that PFAS interact with thyroid hormone binding proteins (Berg et al. 2015; Ren et al. 2016; Zhang J et al. 2016), one of several mechanisms by which PFAS can perturb feedback relationships between free thyroid hormone and the hypothalamic–pituitary–thyroid axis. Exposures to PFAS also interfere with thyroid peroxidase (TPO) enzyme activity in vitro (Song et al. 2012). Several PFAS studies have pursued this putative mechanism, finding that maternal and neonatal thyroid hormone outcomes were more readily detected in those with a priori abnormally high circulating anti-TPO antibodies (Webster et al. 2014, 2016). One case–control study investigated congenital hypothyroidism, a rare condition. Serum concentrations of PFOA (5.40 vs 2.12 ng/mL; $p < 0.01$), perfluorononanoic acid (PFNA; 1.93 vs 0.63 ng/mL; $p < 0.001$), perfluorodecanoic acid (PFDA; 0.52 vs 0.30 ng/mL; $p < 0.005$), and perfluoroundecanoic acid (0.98 vs 0.44 ng/mL; $p < 0.005$) were higher in the diagnosed newborns; and levels of several PFAS, including PFOA and PFHxS, were correlated with thyroid autoantibodies (Kim et al. 2016).

Thyroid disease is not the only concern. Clinicians are concerned about subclinically elevated thyroid-stimulating hormone (TSH) in early pregnancy because it may be associated with several possible adverse maternal and fetal outcomes (Forhead and Fowden 2014). This general concern has prompted numerous PFAS-exposure evaluations of corresponding TSH in maternal serum, cord blood, and newborns. A review of maternal and child biomarkers with PFAS exposure noted that higher TSH has been reported in 4 second-trimester studies (Ballesteros et al. 2017), but there are also conflicting findings. Studies measuring PFAS in the first trimester have also found associations between PFAS exposure and altered TSH levels in newborns, including nonmonotonic patterns of dose response that mirror the marked alterations of thyroid hormone levels during pregnancy (Inoue et al. 2019).

From the available studies, PFAS definitively alter human thyroid hormones and potentially contribute to thyroid autoimmunity but do not so far appear to be a cause of thyroid cancer (Barry et al. 2013; Vieira et al. 2013). Also, thyroid cancer is usually survived; thus, morbidity rather than mortality studies are useful.

Liver disease and cancer

The liver is a primary target organ for long-chain PFAS storage, and accompanying experimental evidence of toxicity includes hepatocyte fat infiltration, specific P450 (CYP) pathway induction, apoptosis, hepatocellular adenomas and carcinomas, and disrupted fatty acid trafficking that can be peroxisome proliferator–activated receptor alpha (PPARα)–dependent or –independent and present across species (Maestri et al. 2006; Cui et al. 2009; Wan et al. 2012; Huang et al. 2013; Perez et al. 2013; Filgo et al. 2015; Xu et al. 2016, 2020a; Yao et al. 2016; Zhang L et al. 2016b; Hui et al. 2017; Li et al. 2017a; Guillette et al. 2020; National Toxicology Program 2020a).

Population studies demonstrate significant associations of long-chain PFAS (>6 fluorinated carbons) exposure to higher liver enzymes, such as alanine aminotransferase in adults and adolescents (Sakr et al. 2007a; Gallo et al. 2012; Yamaguchi et al. 2013; Gleason et al. 2015; Attanasio 2019; Nian et al. 2019), including in longitudinal studies (Sakr et al. 2007b; Darrow et al. 2016). Following low-dose exposures, these associations may be more evident in obese participants (Lin et al. 2010; Gallo et al. 2012; Jain and Ducatman 2019e).

Based on experimental data (Martin et al. 2007; Wan et al. 2012; Wang et al. 2013; Das et al. 2017), nonalcoholic

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fatty liver disease (NAFLD) has been investigated as a clinical outcome of PFAS exposure mediating consistent population PFAS-altered liver enzyme findings. Studies with NAFLD cytokeratin C18 biomarkers have provided supportive evidence for PFAS inducing steatosis (Bassler et al. 2019). Metabolomic studies have been directed at potentially explanatory human glycerophosphocholine and fatty acid profiles (Kingsley et al. 2019; Salihovic et al. 2019; Wahlang et al. 2019). Processes which favor steatosis promote advanced liver disease including liver cancer in humans (Massoud and Charlton 2018; National Toxicology Program 2020a). Associations of PFAS with advanced human liver disease and liver cancer are technically hard to study for reasons including (and not limited to) lethality, selection of comparison populations, and alterations of excretion mechanics associated with disease states. In a clinic-based study, mostly obese (85%) children aged 7 to 19 yr with biopsy-proven NAFLD had more advanced disease associated with PFOS and PFHxS exposure as well as associations with lipid and amino acid pathways linked to NAFLD pathogenesis (Jin et al. 2020). However, an adult study reported that serum PFHxS was inversely associated with hepatic lobular inflammation in morbidly obese bariatric surgery patients (Rantakokko et al. 2015). A study of heavily exposed workers ($n = 462$, geometric mean serum PFOA of 4048 ng/mL) detected significantly increased incident mortality for cirrhosis (relative risk = 3.87, 95% CI 1.18–12.7) and liver cancer (relative risk = 6.69, 95% CI 1.71–26.2) compared to a regional population (Girardi and Merler 2019), whereas no PFAS association to cancer or advanced liver disease was reported in a 3M worker cohort or in the C8 Health study population (Lundin et al. 2009; Barry et al. 2013; Vieira et al. 2013).

Emerging animal toxicology and histology and human population data provide mechanistic clues that PFAS disrupt hepatic metabolism, leading to increased bile acid reuptake and lipid accumulation in liver (Salihovic et al. 2020; Schlezinger et al. 2020). A review of NAFLD and toxicant exposure concluded that PFAS are associated with early steatosis ("fatty liver"), the preclinical stage of NAFLD (Armstrong and Guo 2019).

Lipid and insulin dysregulation

Cross-sectional and longitudinal investigations indicate that PFAS increase serum total and low-density lipoprotein cholesterol in adults and children (Steenland et al. 2009; Frisbee et al. 2010; Nelson et al. 2010; Eriksen et al. 2013; Fisher et al. 2013; Fitz-Simon et al. 2013; Geiger et al. 2013; Fu et al. 2014; Starling et al. 2014; Winquist and Steenland 2014a; Skuladottir et al. 2015; Zeng et al. 2015; Koshy et al. 2017; Convertino et al. 2018; He et al. 2018; Seo et al. 2018; Dong et al. 2019; Lin et al. 2019; Li et al. 2020; Liu G et al. 2020), including clinically defined high cholesterol (Steenland et al. 2009; Winquist and Steenland 2014a; Lin et al. 2019). Studies of large populations, featuring wide exposure ranges, demonstrate that serum lipids rapidly increase beginning at background (1–10 ng/mL) serum concentration and then are followed by attenuating ("plateaued") cholesterol measurements as (log-transformed) exposures to long-chain PFAS increase (Steenland et al. 2009; Frisbee et al. 2010; Li et al. 2020). These findings suggest partially saturable mechanisms; thus, the cholesterol dose response at pharmacologic or acutely toxic doses should be viewed with caution; associations can be missed or may be misleading when an environmental range of exposure is absent. At background exposure levels, residual associations may be more detectable in obese participants (Timmermann et al. 2014; Jain and Ducatman 2019d), a finding congruent with experimental PFAS outcomes in rodents fed "Western" or high-fat diets (Tan et al. 2013; Quist et al. 2015; Rebholz et al. 2016). Human gene expression pathways provide support for an interaction of obesity and PFAS exposures and suggest possible sex differences (Fletcher et al. 2013). A pharmacokinetic model predicts that approximately half of the PFOS-exposed population would experience a >20% rise in serum cholesterol (Chou and Lin 2020). Risk-assessment implications for low-PFAS dose increases in cholesterol have been noted (New Jersey Drinking Water Quality Institute Health Effects Subcommittee 2017; Li et al. 2020), and a review of population and toxicity data concluded that dyslipidemia is the strongest metabolic outcome of PFAS exposure (Sunderland et al. 2019).

Human PFAS lipid findings may be related to experimental findings of induced adipogenesis, impaired bile acid metabolism/synthesis, strongly decreased CYP7A1 enzyme activity, altered fatty acid transport, and intracellular lipid accumulation with steatosis, including in PPAR-α-null or PPAR-α-humanized animals (Guruge et al. 2006; Lau et al. 2007; Bijland et al. 2011; Bjork et al. 2011; Wang et al. 2014; Filgo et al. 2015; Das et al. 2017; Salihovic et al. 2019; Zhang et al. 2019; Behr et al. 2020a; Liu S et al. 2020b; Schlezinger et al. 2020). Independent of PFAS exposure, similar alterations in metabolic pathways have been related to disrupted fatty acid beta-oxidation and increased free cholesterol in toxicology studies (Perla et al. 2017).

Cross-sectional studies of diabetes outcomes can be misleading for reasons discussed in the renal section (see section *Kidney disease, uric acid, and kidney cancer*). Emerging longitudinal and diabetes clinical trial data indicate that PFAS may increase human insulin resistance, associated with dysregulated lipogenesis activity (Alderete et al. 2019; Lin et al. 2019). Longitudinal studies of clinically diagnosed diabetes patients have sometimes associated PFAS exposures with diabetes (Sun et al. 2018) or with small changes in glycemic markers (Cardenas et al. 2017); however, diabetes associations to date are not consistent (Karnes et al. 2014; Cardenas et al. 2017; Donat-Vargas et al. 2019). Future studies should consider whether PFAS may instigate autoimmune diabetic outcomes in humans, as shown in experimental studies (Bodin et al. 2016). Experimental data reveal that PFAS activate G protein–coupled receptor 40, a free fatty acid–regulated membrane receptor on islet ß cells, stimulating insulin secretion (Qin et al. 2020; Zhang L et al. 2020).

Kidney disease, uric acid, and kidney cancer

Extended human half-lives of long-chain PFAS are attributed to active renal tubular reabsorption. Of concern, legacy PFAS

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such as PFOA and PFOS are concentrated in renal tissues, and histopathologic, molecular, oxidative stress, and epigenetic studies provide evidence of potential nephrotoxicity (Wen et al. 2016; Stanifer et al. 2018; Sakuma et al. 2019; Rashid et al. 2020). In addition, the strong influence of kidney reabsorption on the extended half-lives of long-chain PFAS is consistent with both human protein binding and experimental PFAS excretion data.

Human studies have associated legacy PFAS exposure to diminished glomerular filtration and/or defined chronic kidney disease in adults and children (Shankar et al. 2011; Watkins et al. 2013; Kataria et al. 2015; Blake et al. 2018). However, this outcome may be due to reverse causation (Watkins et al. 2013; Dhingra et al. 2017). Some reviews of the available epidemiologic and toxicologic evidence suggest causative links between PFAS and diminished kidney function and chronic kidney disease (Stanifer et al. 2018; Ferrari et al. 2019); these authors also note several knowledge gaps and uncertainty about which proposed mechanisms of action are most important. A propensity score approach to NHANES data (Jain and Ducatman 2019c; Zhao et al. 2020) and a study with repeated PFAS and health measures over an 18-yr period (Blake et al. 2018) recently concluded that PFAS exposure likely causes diminished renal glomerular filtration.

Uric acid, a biomarker of increased risk for renal disease (Obermayr et al. 2008), is also consistently associated with PFAS exposure in adults and children (Steenland et al. 2010; Geiger et al. 2013; Gleason et al. 2015; Kataria et al. 2015; Qin et al. 2016; Zeng et al. 2019), including a visible dose–response curve that begins at or near historic background levels in human populations (Steenland et al. 2010; Zeng et al. 2019). Serum PFAS concentrations exhibit an inverted U-shaped pattern related to glomerular filtration, initially exhibiting a modest accumulation as glomerular filtration begins to decrease and then decreasing in advancing renal disease, likely due to failure of normal strong reabsorption mechanisms in moderate to severe kidney disease (Jain and Ducatman 2019c). This finding is more dramatic across stages of glomerular filtration when there is also albuminuria (Jain and Ducatman 2019b). Studies suggest that the association of PFAS to uric acid is not due to reverse causation and is underestimated because the failing kidney excretes long-chain PFAS but retains uric acid. An implication is that population outcomes that occur in the presence of either albuminuria or moderate to severe renal disease such as hypertension (Jain 2020) increasing presence of and uric acid (a biomarker of renal disease; Jain and Ducatman 2019a; Zeng et al. 2019) can be underestimated in cross-sectional studies; in other words, the link between these health outcomes and PFAS exposure is obscured in these studies because of enhanced PFAS excretion patterns in the presence of either albuminuria or moderate to severe kidney disease. Furthermore, the strong influence of renal reabsorption on the long half-lives of long chain PFAS is consistent with both human protein binding of PFAS and experimental PFAS excretion rates in high-dose rodent studies (Cheng and Ng 2017).

Kidney cancer diagnoses have been increasing since 1975, a finding that is partially independent of improved detection, with 5-yr cancer-specific survival of approximately 80% (Gandaglia et al. 2014). The C8 Health studies noted longitudinal ($n = 32\,254$) increases of kidney cancer (hazard ratio = 1.10, 95% CI 0.98–1.24) and kidney cancer mortality (Steenland and Woskie 2012; Barry et al. 2013; Vieira et al. 2013). A review of 6 published studies found long-chain PFAS exposure associated with kidney cancer or kidney cancer mortality, with risks ranging from 1.07 to 12.8 (Stanifer et al. 2018). Subsequent preliminary data from the heavily exposed Veneto, Italy, population also suggest a significant increase in kidney cancer mortality with PFAS exposure (Mastrantonio et al. 2018). Evidence is accumulating for PFAS as a cause of chronic disease and kidney cancer. Study designs must consider the peculiar PFAS excretion mechanics involved in and associated with kidney disease.

Reproductive and developmental outcomes

Exposure to PFOA impairs human sperm motility and sperm penetration into viscous media (Sabovic et al. 2020; Yuan et al. 2020) and is longitudinally associated with lower sperm concentration and count and higher adjusted levels of luteinizing and follicle-stimulating hormones in young men (Joensen et al. 2009; Vested et al. 2013; Song et al. 2018). Serum concentrations of PFAS are also cross-sectionally associated with deleterious markers of semen quality (Louis et al. 2015; Pan et al. 2019).

Legacy and emerging PFAS have been found in follicular fluid (Kang et al. 2020). They appear to alter endometrial regulation such as progesterone activity in young women (Di Nisio et al. 2020b) and possibly menstrual cycle length (Lum et al. 2017). Associations with menarche and menopause may be substantially due to reverse causation because menstruation is a route by which women eliminate PFAS (Dhingra et al. 2017), partially explaining why men have higher PFAS levels than women in the same communities. Women on birth control and who do not menstruate or with poor cyclicity because of age, activity level, or disease may have elevated PFAS levels in comparison with menstruating women. Exposure to PFAS has been associated with endometriosis in the United States and in China (Louis et al. 2012; Campbell et al. 2016; Wang B et al. 2017a), but the specific PFAS associated with this effect vary among studies.

Time-to-pregnancy (fecundity) studies provide indirect evidence of changes in fertility. Methodologic considerations include maternal and paternal age, parity (which in turn affects serum PFAS), and health status. Among 1240 women in the Danish National Birth Cohort, PFOS exposure was associated with decreased fecundity (median serum PFOS 35.5 ng/mL; Fei et al. 2009). Reverse causation may explain this finding because it is duplicated in parous, but not among nonparous, women (Whitworth et al. 2012; Bach et al. 2015). Prospective odds of actual infertility in the Maternal–Infant Research on Environmental Chemicals cohort ($n = 1743$) at low-dose exposures were associated with PFOA (geometric mean 1.66 ng/mL; odds ratio = 1.31, 95% CI 1.11–1.53) and PFHxS

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(odds ratio = 1.27, 95% CI 1.09–1.48; Velez et al. 2015). The reported fertility rate improved following water filtration in a PFAS-contaminated community (incidence rate ratio 0.73, 95% CI 0.69–0.77 prior to filtration) along with measures of birth weight (Waterfield et al. 2020).

Per- and polyfluoroalkyl substances reliably move across the placenta and enter breast milk (Gyllenhammar et al. 2018; VanNoy et al. 2018); serum PFAS levels in young children generally exceed maternal serum concentrations (Fromme et al. 2010; Papadopoulou et al. 2016; Eryasa et al. 2019). Population studies provide evidence that breastfeeding duration and milk quantity are adversely affected by PFAS exposure (Romano et al. 2016; Timmermann et al. 2017b; Rosen et al. 2018).

A systematic review reported that PFOA exposure was associated with a small decrease in infant birth weight; the meta-analysis estimated that a 1-ng/mL increase in PFOA was associated with an approximately 19-g reduction (95% CI −29.8 to −7.9 g) in birth weight (Lam et al. 2014). The authors noted similarities in experimental studies (Johnson et al. 2014; Koustas et al. 2014) and concluded that there was "sufficient" human and corroborative toxicology evidence of a detrimental effect of PFOA on birth weight (Johnson et al. 2014; Koustas et al. 2014; Lam et al. 2014). However, another meta-subpopulation analysis, focused on early pregnancy or the time shortly before conception, detected only a small and nonsignificant association, which was less subject to bias (Steenland et al. 2018a). Different approaches to the possible confounding role of shifting glomerular filtration rates in pregnancy can affect interpretations; evidence suggests this consideration can, at most, only partially explain associations of PFAS exposure to decreased birth weight (Interstate Technology and Regulatory Council 2020; Wikstrom et al. 2020). A recent review of mostly prospective cohort studies ($n = 24$ studies) noted PFAS associated with altered fetal and postnatal growth measures, such as lower birth weight. Many ($n = 22$) of the relevant studies suggest developmental and childhood immunomodulatory effects, whereas 21 studies concerning neurodevelopment were inconclusive (Liew et al. 2018). The authors of the review noted methodologic challenges of developmental and newborn epidemiology, including consideration of critical exposure windows for developmental effects, the effects of breastfeeding and parity on maternal PFAS levels, and the variety of possible mechanistic explanations for growth outcomes, such as disruption of glucocorticoid and thyroid hormone metabolism in utero (Liew et al. 2018). Recent Faroe Island studies report that prenatal PFAS effects on thyroid hormone status do not support a causal relationship (Xiao et al. 2020).

Review articles suggest that prenatal exposure to PFOA may increase risk of subsequent childhood adiposity, noting that steroid hormones, retinoid X receptor, and other pathways may be contributing to this effect (Halldorsson et al. 2012; Hall and Greco 2019). Prospective evidence supports this relationship in adults with a high risk of diabetes (Cardenas et al. 2017). However, some well-performed community studies do not support this outcome in adults or children (Barry et al. 2014; Martinsson et al. 2020).

Based on several preliminary findings, supported by longitudinal follow-up studies (Stein et al. 2009; Savitz et al. 2012; Darrow et al. 2013; Avanasi et al. 2016a, 2016b), the C8 Science Panel concluded that PFOA is probably linked to pregnancy-induced hypertension or preeclampsia. Population-level evidence implicating additional PFAS having this effect has included studies with longitudinal designs (Huang et al. 2019; Wikstrom et al. 2019; Borghese et al. 2020). Experimental support includes PFAS effects on human trophoblast migration in vitro (Szilagyi et al. 2020) and recent evidence of PFOA and GenX (or hexafluoropropylene oxide dimer acid) effects on mouse placenta, as well as excessive gestational weight gain (Blake et al. 2020). However, a recent longitudinal study did not find an association of PFAS with pregnancy-associated hypertension (Huo et al. 2020).

The possibility that circulating PFAS may reduce bone mineral density has been investigated. Cross-sectional and practical trial associations have been found in adults (Lin et al. 2014; Hu et al. 2019; Di Nisio et al. 2020a), and there is emerging longitudinal evidence from a mother and child pair study indicating that children may also be affected (Cluett et al. 2019).

Testicular cancer diagnoses are increasing steadily, a trend unrelated to improved detection (Cheng et al. 2018; Park et al. 2018). Most patients diagnosed (>90%) will be cured and die of other causes; mortality studies therefore provide little help in understanding disease risk factors. The C8 Science Panel detected longitudinal evidence for increased testicular cancer risk (1.35, 95% CI 1.00–1.79) for cumulative PFOA exposure (Barry et al. 2013). There are ample supportive data of testicular damage following PFAS exposure, including strong evidence of endocrine disruption; but the cell-specific associations are different in humans (germ cell) than the outcomes in rodents (stromal).

Per- and polyfluoroalkyl substances have deleterious effects on conception, pregnancy, and infant development. The underlying birth weight data are mostly supportive, although the subsequent growth and adiposity literature is mixed. The most sensitive reproductive and developmental outcomes are a topic of ongoing discussion.

Outcomes replicated across populations, such as perfluorocarboxylate (PFCA) and perfluorosulfonate (PFSA) exposures associated with down-regulation of immune response; increases in cholesterol, liver enzymes, and uric acid; alterations in thyroid hormone binding proteins; growth deficits; and effects on breast milk and lactation, indicate priority areas for understanding mechanisms and health implications.

CURRENT KNOWLEDGE OF PFAS TOXICITY IN EXPERIMENTAL MODELS

Animal studies have focused most intensely on PFOA and PFOS, using laboratory rodents and, more recently, zebrafish as models. Perfluoroalkyl acids of varied carbon-chain lengths as well as a few replacement chemicals with ether linkages in the carbon backbone (such as GenX and 3H-perfluoro-3-[(3-methoxy-propoxy)propanoic acid], or ADONA) have also

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been examined, with outcome profiles thus far generally consistent with legacy chemicals. The varying extent of responses is likely related to toxicokinetic disposition (excretion or half-life) and relative potency and affinity of the individual chemical for binding to receptor proteins. Some PFAS (i.e., PFHxS, PFOA, and PFNA) have longer half-lives in mice than rats and typically much longer half-lives in humans (Table 1). These differences in elimination kinetics complicate the cross-species evaluation of toxicity. In addition, some PFAS (such as PFOA and PFNA) exhibit a profound sex difference in the rate of chemical elimination and bioaccumulation in the rat: females eliminate them much faster than males (Table 1). Sex differences in half-lives, although important, are much smaller in humans and have a different explanation. The mouse also typically has more limited sex-based PFAS elimination differences, making this species more amenable for extrapolation to humans, especially for mechanistic and toxicity evaluations.

In general, human health effects associated with PFOA and PFOS exposure (described in section *Current Knowledge of PFAS Toxicity in Humans*) have also been reported in animal models: hepatic/lipid metabolic toxicity, developmental toxicity, immune suppression, tumor induction, endocrine disruption, and obesity. These findings are often derived from well-controlled laboratory experiments in more than one species using wide dose ranges that are often orders of magnitude higher than typical human exposure, to account for differences in half-life across species. Some of the phenotypic findings are supported by in vitro mechanistic investigation and/or molecular queries on target tissues. Our understanding of the toxicologic properties of PFAS other than PFOA and PFOS is notably less advanced and, in the case of emerging replacements and by-products, completely unexplored.

Hepatic and metabolic toxicity

In rodent studies, dose-dependent increases in liver weight, in hepatocellular hypertrophy associated with vacuole formation, and with or without increased peroxisome proliferation have been observed with a significant body burden of PFAS, especially for the most persistent and potent long-chain homologs. Hepatocyte proliferation, necrosis, and apoptosis are outcomes occurring at relatively low doses. This is also true for a new replacement chemical, GenX, which altered liver histopathology and function and increased apoptosis in mice and fish (Blake et al. 2020; Guillette et al. 2020). Correspondingly, transcriptional activation of mouse and, to a lesser extent, human PPARα-related genes in liver was detected in adult-exposed models; activation of other nuclear receptors such as PPARγ, constitutive androstane receptor (CAR), and pregnane X-receptor (PXR) has also been reported. These nuclear receptors, metabolic sensors that regulate lipid and glucose metabolism and transport and inflammation, tend to be more responsive in tissues of rodents than in humans (Wolf et al. 2012; Rosen et al. 2017). Recent work using developmental models reports that

TABLE 1: Per- and polyfluoroalkyl substances serum half-life estimates in rat, mouse, monkey, and humans

	PFBS (C4)		PFHxS (C6)		PFOS (C8)		PFBA (C4)		PFHxA (C6)		PFHpA (C7)		PFOA (C8)		PFNA (C9)		PFDA (C10)		F-53B		GenX	
	F	M	F	M	F	M	F	M	F	M	F	M	F	M	F	M	F	M	F	M	F	M
Rat	0.6–4.0 h	2.1–4.5 h	1.8 d	6.8 d	62–71 d	38–41 d	1.0–1.8 h	6–9 h	0.4–0.6 h	1.0–1.7 h	1.2 h	2.4 h	2–4 h	4–6 d	1.4–6.4 d	31–55 d	59–75 d	40–80 d			8 h	3 h
Mouse	4.5 h	5.8 h	25–2 d	28–3 d	31–3 d	36–4 d	3 h	12 h	~1.2 h	~1.6 h			16 d	22 d	26–6 d	34–6 d					18 h	20 h
Cynomolgus Monkey	3.5 d	4.0 d	87 d	141 d	110 d	132 d	1.7 d		2.4 h	5.3 h			30 d	21 d								
Human	28 d		5.3–8.5 yr		3.4–5.0 yr		3 d		32 d		1.2–2.5 yr		2.1–3.8 yr		2.5–4.3 yr		15.3 yr					

GenX = hexafluoropropylene oxide dimer acid; PFBA = perfluorobutanoic acid; PFBS = perfluorobutanesulfonic acid; PFDA = perfluorodecanoic acid; PFHpA = perfluoroheptanoic acid; PFHxA = perfluorohexanoic acid; PFHxS = perfluorohexane sulfonate; PFNA = perfluorononanoic acid; PFOA = perfluorooctanoic acid; PFOS = perfluorooctane sulfonate.

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mitochondrial dysfunction is associated with hepatocellular hypertrophy in young adult mice (Quist et al., 2015) and that other fatty acid metabolism pathways are activated (Jones et al. 2003; Shabalina et al. 2016). Steatosis is also a common feature of PFAS chronic exposure in rodents. Exposure in rodent models typically decreases serum cholesterol, whereas elevations of circulating cholesterol levels have been reported in humans. The mode of action concerning serum cholesterol is debatable. For example, PFOA exposure increased liver weight, increased liver enzymes, and led to persistent histopathological changes (particularly damage to the bile duct) in livers of wild-type and PPARα-null rodent strains (reviewed in Division of Science and Research, New Jersey Department of Environmental Protection 2019). Many of these effects are reversible on cessation of PFAS exposure, and this observation has been interpreted by some as evidence of "adaptive" responses to exposure. However, this reversibility is irrelevant to ongoing environmental PFAS exposure (for instance, from drinking water) because exposure will persist until contamination is remediated. In summary, there is a strong confluence of animal toxicology and histology and human population data that PFAS disrupt hepatic metabolism and lead to lipid accumulation in liver, although the mechanism(s) is unclear. Effects on bile acid metabolism, mitochondrial perturbation, and cholestatic mechanisms deserve further investigation at human-relevant exposures.

Reproductive and developmental toxicity

Only a few reproductive toxicity studies of males and females are available, primarily focusing on long-chain PFAS. Profound developmental toxicity has been described following gestational and lactational exposure to PFOS, PFOA, and PFNA in mice (Thibodeaux et al. 2003; Lau et al. 2006; Das et al. 2015) and in mice and rats gestationally exposed to GenX (Conley et al. 2019; Blake et al. 2020). Neonatal morbidity and mortality were seen with exposure to high doses of legacy PFAS; growth deficits and developmental delays were noted in offspring exposed to lower doses. Evidence of lactation impairment was seen in mice at doses of 5 mg PFOA/kg body weight (White et al. 2007), leading to increased offspring mortality (Lau et al. 2006); recent studies have indicated a role of placental dysfunction in these adverse developmental outcomes (Blake et al. 2020). Deficits of mammary gland development were also observed in mice exposed to PFOA (doses of 1 mg/kg body wt and lower) during gestation, which persisted into adulthood, although these exposure levels did not alter body weight, lactational function, or neonatal growth of offspring (F1 or F2 mice; Macon et al. 2011; White et al. 2011b; Tucker et al. 2015). Systematic reviews support a relationship between in utero exposure to PFOA and PFOS and reduced fetal growth in animals and humans, and the relationship between PFOA and reduced fetal growth in mice was recently validated (Koustas et al. 2014; Blake et al. 2020). Also, PFAS are reported to have reproductive effects such as ovulation failure in mice (Zhang Y et al. 2020).

Immunotoxicity

A few long-chain PFAS (PFOS, PFOA, PFNA, and PFDA) have been shown to alter immune status in rodents and nonhuman primates. Effects are predominantly immunosuppressive and include reductions in thymus and spleen weights and associated immune cell populations, in numbers of circulating immune cells, in certain aspects of innate immunity (i.e., natural killer cell cytotoxicity), in infectious disease resistance, and in antibodies produced in response to an antigen (i.e., analogous to the vaccine response in humans). In their 2018 draft *Toxicological Profile for Perfluoroalkyls*, the US Agency for Toxic Substances and Disease Registry (ATSDR) noted changes to the aforementioned immune parameters observed in experimental rodents exposed to PFOA, PFOS, PFNA, PFHxS, PFDA, perfluorobutanesulfonic acid (PFBS), or perfluorobutanoic acid (PFBA; Agency for Toxic Substances and Disease Registry 2018). The US National Toxicology Program conducted a systematic review of the immunotoxicological literature for PFOA and PFOS and concluded that PFOA and PFOS were presumed to be immune hazards to humans based on a high level of evidence for suppression of antibody responses in experimental animals and a moderate level of evidence for suppression of antibody responses in humans (National Toxicology Program 2016). The ATSDR (Agency for Toxic Substances and Disease Registry 2018) also included a decreased antibody response to vaccines (PFOA, PFOS, PFHxS, and PFDA) and increased risk of asthma diagnosis (PFOA) among the list of adverse health effects in PFAS-exposed humans. Reduction in the antibody response to a vaccine, an adaptive immune function, is a well-accepted measure of immunotoxicity, is consistent with the mode of action for the effects of fatty acids on immune system function (Fritsche 2006), and is compelling evidence that the immune system is a sensitive target of PFAS.

Tumor induction

Per- and polyfluoroalkyl substances are not known to be directly mutagenic; PFOA, PFOS, and other tested PFAS show little or no evidence for induction of gene mutation, clastogenicity, or aneuploidy in vitro or in vivo by a direct mode of action (see EFSA Panel on Contaminants in the Food Chain [2020] for details). There is evidence that PFAS can induce DNA damage, such as strand breaks, and other genotoxic effects, secondary to oxidative stress (EFSA Panel on Contaminants in the Food Chain 2020). This occurs at concentrations or doses that are high relative to human environmental exposures to PFAS, and the mechanism is such that their dose–response will be sublinear. Hence, PFAS are unlikely to be of mutagenic concern in exposed populations.

In adult-exposed rodents and fish, PFOA and PFOS have been shown to induce tumors. Liver adenomas, pancreatic acinar cell tumors, and testicular Leydig cell adenomas have been detected in rats treated chronically with PFOA (IARC Working Group on the Evaluation of Carcinogenic Risks to Humans 2017) as well as its replacement, GenX (Caverly Rae et al. 2015). Following

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gestational and chronic exposure to PFOA, 58% of male rats demonstrated pancreatic tumors at the lowest dose administered (National Toxicology Program 2020b). This finding has spurred Minnesota and California policymakers to consider cancer as an endpoint in risk assessment, whereas the European Food Safety Authority (EFSA Panel on Contaminants in the Food Chain 2020) has the opinion that there is not adequate evidence for a link between exposure to PFAS and cancer risk in humans. This "tumor triad" profile has been associated with the PPARα-mediated molecular signaling pathway in rats exposed to high doses of PFAS. Consequently, liver tumors involving this mode of action are not considered relevant to humans at equivalent PFAS exposures (Post et al. 2017). The human relevance of PPARα-mediated pancreatic tumors in rodents remains to be determined. Liver lesions evident in PPARα-null mice exposed to PFOA during pregnancy and lactation (Filgo et al. 2015) suggest a non-PPARα-mediated liver response. Induction of liver tumors mediated by estrogen receptor (ER) activation has also been reported in fish (Tilton et al. 2008), and several non-PPARα-mediated hypotheses, including increased reactive oxygen species formation, oxidative stress, and mitochondrial dysfunction; decreased tumor cell surveillance by the immune system; and diminished gap junction cellular communication, are documented (IARC Working Group on the Evaluation of Carcinogenic Risks to Humans 2017; New Jersey Drinking Water Quality Institute Health Effects Subcommittee 2017).

Endocrine disruption

The primary evidence for the endocrine-disrupting potential of PFAS involves induction of hypothyroxinemia and reduction of serum testosterone in rats. An early review of PFAS endocrine-disrupting properties in humans concluded that the "thyroid may be one axis significantly affected by PFOA exposure while the animal toxicology literature is less certain due to technical issues" (White et al. 2011a).

The effects of PFAS on thyroid hormone status detected in animal studies differ from classical hypothyroidism, in that reduction of circulating total thyroxine is not accompanied by a compensatory increase of TSH. A possible mechanism for these effects may be related to the propensity of protein binding of legacy PFAS, which could lead to displaced total thyroxine binding to its carrier proteins (transthyretin and thyroxine-binding globulin). Human population studies augment animal data showing that PFAS interact with thyroid hormone binding proteins (Berg et al. 2015; Ren et al. 2016; Zhang J et al. 2016a), one of several mechanisms by which PFAS can perturb feedback relationships between free thyroid hormone available to cells (free total thyroxine) and the hypothalamic–pituitary axis. Some estrogenic effects of PFAS have also been illustrated by in vitro studies, although there is no evidence of direct transactivation of estrogen, androgen, or glucocorticoid receptors (Behr et al. 2018, 2020b).

The evidence for PFAS affecting ER signaling in humans and animals is mixed. Although studies have identified some PFAS as being without estrogenic activity (Behr et al. 2018; Borghoff et al. 2018; Gogola et al. 2019), others suggest an ability of PFAS to modulate or even activate ER-mediated effects (Benninghoff et al. 2010; Kjeldsen and Bonefeld-Jørgensen 2013; Wang et al. 2018; Bjerregaard-Olesen et al. 2019; Qiu et al. 2020), with some effects only observed in aquatic organisms (Wei et al. 2009; Chen et al. 2016, 2018). Microarray analyses of human primary hepatocytes confirmed that PFOA activated the ER pathway (Buhrke et al. 2015).

Neurotoxicity

Potential adverse effects of PFAS on the nervous system and functions have not been widely investigated. A few studies reported neurotoxicity of PFOS, PFHxS, and PFOA in cell culture systems (Slotkin et al. 2008), as well as altered behavioral responses (Goulding et al. 2017) and deficits in learning and memory ability in rodents (Viberg et al. 2013). In contrast, no significant developmental neurotoxic effects were seen from prenatal exposure to PFOS in USEPA guideline–based studies with rats (Butenhoff et al. 2009).

Obesity

Numerous cell-based assays in human and mouse preadipocytes and animal studies with and without high-fat diets have consistently shown that some PFAS have the potential to increase lipid production by adipocytes and fat pads (van Esterik et al. 2016). Exposure of pregnant mice to low doses of PFOA produced obesity in young adult female offspring (Hines et al. 2009; van Esterik et al. 2016), a finding that was recapitulated in Danish women exposed in utero to PFOA (Halldorsson et al. 2012). Both PFOA and GenX increased weight gain of pregnant mice (Blake et al. 2020), an effect also seen in women during pregnancy (Ashley-Martin et al. 2016), although discordant results have been reported in other studies (Barry et al. 2014; Ngo et al. 2014). These apparently disparate findings in experimental models may be associated with differences among mouse strains examined, exposure periods, statistical methodology, and/or the rodent diets used.

There are specific differences in human and rodent health outcomes that deserve further investigation: 1) cholesterol metabolism, 2) thyroid effects, 3) mode of action for liver effects (different or same), and 4) kidney transporter or other mode of action leading to large differences in half-life. However, species concordance in the 6 human health effects discussed in the present review supports a weight of evidence for these effect for the handful of extensively studied PFAS.

Human health advisory and guidance values for a few PFAS have been issued to date by the USEPA, the ATSDR, several individual state environmental agencies or health departments, as well as regulatory agencies in Canada and Europe that are largely (but not exclusively) based on toxicological findings in animal models. However, risk-assessment scientists have not reached consensus in selecting a singular apical endpoint as the basis for a point of departure for assessments. Three

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toxicological features of PFAS that have been commonly highlighted, based on their sensitivity (low dose effect), strength of evidence (robust corroborating studies with mechanistic support for human relevance), and corresponding findings noted in epidemiological investigation, are hepatotoxicity (and alterations in lipid metabolism), developmental toxicity, and immunotoxicity. It should be noted that apical endpoints that drive risk assessments often differ among individual PFAS, perhaps highlighting the complexity of these chemicals and the family of PFAS, in general.

IMPORTANCE OF TOXICOKINETICS IN UNDERSTANDING PFAS TOXICITY

Species and sex differences

Few of the substantial number of structurally diverse PFAS have been tested for toxicological effects. Some available toxicological information has come from studies in animals, where marked species and (in rat) sex differences in half-life for some PFAS (Table 1) have been observed and the relevance to humans is uncertain. These differences are due to toxicokinetic and toxicodynamic factors. There are also differences in mean PFAS serum levels between men and women in the same communities. Children may have elevated serum levels compared to parents, even with the same exposures (Emmett et al. 2006; Daly et al. 2018; Graber et al. 2019), for reasons relating to transplacental transfer, breastfeeding, and body mass (Emmett et al. 2006; Daly et al. 2018; Graber et al. 2019; Blake et al. 2020). Transplacental transfer of PFAS confers a substantial burden to the newborn infant. Because the infant has a smaller overall mass and blood volume, PFAS are concentrated, increasing PFAS per volume (Koponen et al. 2018). In addition, transfer of PFAS is common through lactation, and the longer a child breastfeeds, the higher the body burden (Gyllenhammar et al. 2018; VanNoy et al. 2018).

Effects of comorbidity on PFAS toxicokinetics

Factors affecting renal function can influence PFAS toxicokinetics. As discussed, opposing types of causation should be considered. Human toxicokinetics appear to vary bidirectionally with changing renal function, leading to nonmonotonic dose–response relationships and, depending on the study goal, possibly to errors in estimating disease associations. As progress is made in the field of PFAS toxicokinetics, new chemistries may have different clearance factors and nuances that vary by PFAS group or structures, and that will need to be investigated to accurately model half-lives in different exposure subgroups.

Sources of information on toxicokinetics in humans: strengths and limitations of studies

Some PFAS half-life data in humans were obtained from retired industry workers, particularly those who worked with PFOS, PFOA, and PFHxS (Olsen et al. 2007). Since then, these estimates have been modified slightly or confirmed with longitudinal data and modeling from contaminated communities once uncontaminated water options were provided (Bartell et al. 2010; Li et al. 2018). Other contemporary PFAS estimates are derived from biomonitoring studies of production workers, blood donors, study participants, and/or occupationally exposed cohorts (Olsen et al. 2009, 2017; Russell et al. 2013; Zhang et al. 2013). Some caution must be taken in using these data because variables affecting PFAS clearance may not be taken into consideration (age, sex, menstruation, disease, and medication status) and may contribute to confounding.

The challenge in determining a reliable human half-life in these types of studies is that exposure does not end with a clean water source, retirement, or a change of job and that continued exposures vary over potential depuration periods. Model components may also vary in subclasses. Children (small blood volumes and a large fraction of exposures comes from drinking), pregnant women (large increase in blood volume and water intake), parous women (transfer to fetus and breast milk), and athletes (water intake elevated) are examples of subpopulations with expected variation in half-life compared to adult men (Post et al. 2017). There will be more human estimates of PFAS forthcoming that involve variations in half-life (Post et al. 2017). Realistic computational modeling can help, so long as it clearly characterizes exposures and applicable populations. The continued goal should be to provide predictive values for those PFAS lacking actual measurements, based on chemical structures and trusted physiological parameters.

Physiologically based pharmacokinetic/toxicokinetic modeling in different-aged populations

In the blood and other tissues, PFAS toxicokinetics are influenced by their interactions with proteins (Andersen et al. 2006; Katakura et al. 2007; Nakagawa et al. 2008; Weaver et al. 2009; Figure 2). Certain toxicokinetic features are saturable, and thus dosing in toxicokinetic studies is of profound importance. Studies of renal reabsorption mechanisms in mammals show that reduced activity of transporters such as organic anion transporting polypeptide 1a1, through inactivation (e.g., genetic manipulation, castration, treatment with estrogen) or by saturation at increasing doses, leads to substantial reductions in half-lives of PFOA and PFOS (Andersen et al. 2006; Nakagawa et al. 2008; Weaver et al. 2009; Yang et al. 2009).

These protein-associated toxicokinetic processes were recently incorporated into a model for PFOA in the male Sprague-Dawley rat (Cheng and Ng 2017), which provides a useful platform to explore how changes in protein interactions might affect estimates of PFAS half-life (Figure 3). At high doses, it is typical to see clear biphasic behavior with rapid initial clearance, during which the serum half-life appears to be shorter especially at high enough doses that processes such as renal reabsorption are saturated, followed by a much longer tail

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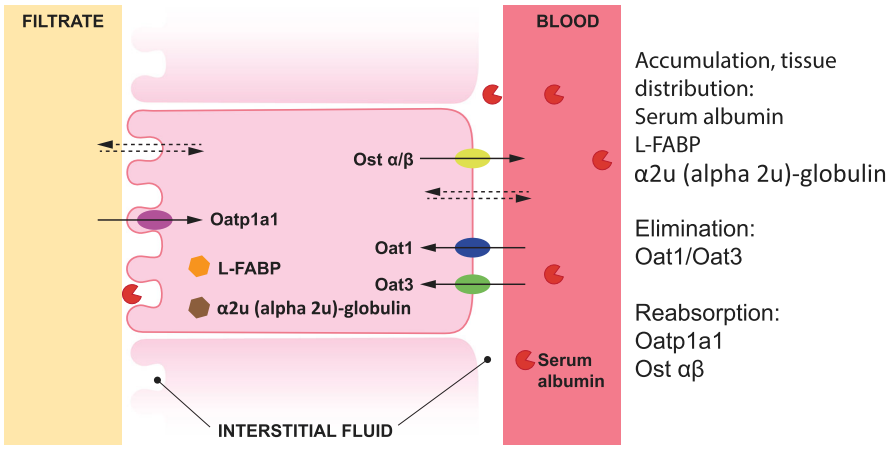


FIGURE 2: Example of proteins that are known to influence per- and polyfluoroalkyl substance toxicokinetics through binding (which affects tissue distribution and accumulation) and facilitation of membrane transport (which affects clearance and reabsorption). Illustrated for kidney and blood. L-FABP = liver fatty acid binding protein; Oat1 = organic anion transporting 1; Oatp1a1 = organic anion transporting polypeptide 1a1; Ost = organic solute transporter.

(Figure 3A). In a similar fashion, the magnitude of internal dose and rate of serum clearance can be profoundly influenced by proteins known to bind PFAS, such as serum albumin (Figure 3B). Increasing and decreasing the extent of reabsorption in the kidney increases and decreases the serum half-life, respectively (Figure 3C). Finally, the effect of saturating reabsorption is magnified when the half-life is longer because of increased serum binding (Figure 3D). In this case, taking an initial slope to calculate the serum half-life at high doses would lead to a profound underestimation.

Differences in protein expression, circulating levels, and even protein type across populations, sex, and species could lead to important species and sex differences in PFAS biological half-lives (Han et al. 2012); such differences should be

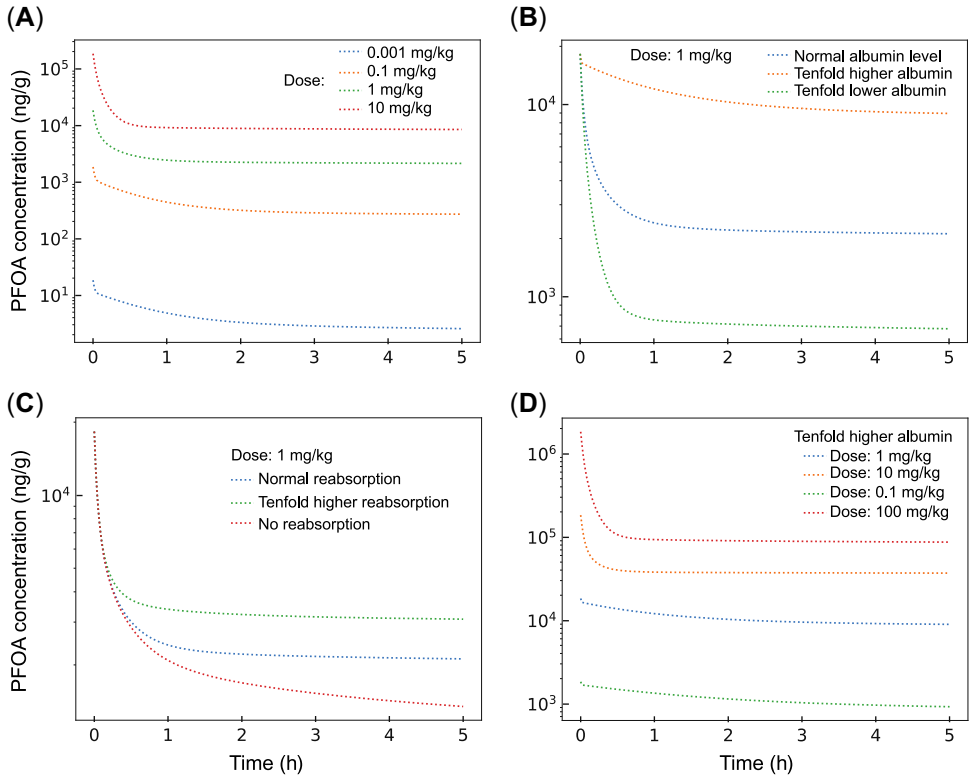


FIGURE 3: Simulations based on Cheng and Ng (2017), perfluorooctanoic acid (PFOA) toxicokinetic model for Sprague-Dawley rats. (**A**) Effect of dose on initial half-life. (**B**) Effect of higher and lower levels of serum albumin, which binds to PFOA, on serum clearance dynamics. (**C**) Effect of extent of reabsorption in kidney on serum half-life, based on organic anion transporting polypeptide 1a1 activity. (**D**) Effect of dose on elimination kinetics when half-life is longer because of higher albumin binding. Oat1 = organic anion transporting 1; Oat3 = organic anion transporting 3; Ost = organic solute transporter.

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investigated and taken into account in the extrapolation to human equivalent doses. Because expression of proteins may change at different life stages, clearance factors and toxicokinetics may also change.

Given the large number of species-, sex-, and age-specific differences that have been observed, coupled with the lack of data for many PFAS, the parameterization of complex physiologically based toxicokinetic models remains a persistent challenge. Therefore, lower-resolution models (e.g., one-compartment or few-compartment models) may be more appropriate for species and settings where insufficient data are available for reasonably accurate parameterization. Alternatively, in silico and in vitro methods are under development that could aid in parameterization in the absence of in vivo data, as discussed in the section *New approaches for developing PFAS toxicity information*.

SO MANY PFAS, SO LITTLE TIME: ACCELERATING THE PACE OF DISCOVERY

Importance of determining mode of action and adverse outcome pathways

Information on modes of action and/or adverse outcome pathways (AOPs) is invaluable in 1) establishing human relevance of experimental evidence, 2) assessing causality in epidemiological studies, 3) applying "read-across" to PFAS for which there is little toxicological information, 4) assessing risks from mixtures, 5) guiding development and interpretation of new approach methodologies, 6) informing the development of biomarkers in epidemiologic investigation, and 7) identifying potentially vulnerable subpopulations and life stage–specific effects (Meek et al. 2014; LaLone et al. 2017). Verified modes of action and AOPs can inform risk assessment based on intermediate effects and enable development of new methodology-based approaches to assess PFAS safety (Meek et al. 2014).

Postulated modes of action/AOPs for PFAS

Mechanistic studies have been performed on only a few PFAS. These have been shown to activate a range of putative molecular initiating targets, among which are the nuclear receptors PPARα, PPARγ, PPARβ/δ, CAR, PXR, liver X receptor α, and ERα (Bijland et al. 2011; Bjork et al. 2011; Rosen et al. 2017; Li et al. 2019). However, modes of action verified by agreed procedures (World Health Organization 2020) have been established for few reported effects of PFAS, and those that have been interrogated involve activation of PPARα and, at higher doses, CAR as molecular initiating events (Klaunig et al. 2012; Rosen et al. 2017). Several AOPs involving these molecular targets are in various stages of development (Organisation for Economic Co-operation Development 2020), but few have been endorsed by the OECD following its agreed procedures (Organisation for Economic Co-operation Development 2017). Demonstration of receptor activation alone is insufficient to establish involvement of a mode of action or AOP in an observed effect, for which an overall weight-of-evidence approach is necessary (World Health Organization 2020).

Andersen et al. (2007) provide a useful, albeit dated, review of possible PFAS modes of action. Established modes of action are restricted largely to the liver and include species-specific hepatic hyperplasia and liver tumors (Butenhoff et al. 2012; Elcombe et al. 2012; Corton et al. 2018). Available studies on PFBS, PFHxS, perfluorohexanoic acid, PFNA and PFDA suggest that they share molecular targets with similar consequences, albeit with differences in potency, in part due to differences in their excretion and protein-interaction kinetics (Zeilmaker et al. 2018). However, studies in vitro have established intrinsic differences in potency among PFAS analogues. Potency in activating PPARα showed some relationship with PFAS chain length (Wolf et al. 2008). A mode of action or AOP provides a causal chain of key events between chemical exposure and outcome. The established modes of action for PFOS and PFOA provide a causal explanation for development of liver tumors observed in rodents on exposure to these compounds, through activation of PPARα, and the possible relevance to humans. However, this does not mean that other effects of PFAS are due to activation of PPARα or that other pathways might not lead to liver tumors in humans, such as secondary to the primary effect of steatosis.

Until recently, there has been little study of modes of action/AOPs for effects of PFAS other than hepatic outcomes in rodents, particularly for critical effects, such as immunosuppression and developmental toxicity, and from PFAS other than PFOS and PFOA (EFSA Panel on Contaminants in the Food Chain 2020; Temkin et al. 2020). The ability of various PFAS to interact with and modify lipid metabolism is, however, an intriguing hypothesis (Xu et al. 1999; Jones et al. 2003; Andersen et al. 2007; Tan et al. 2013; Pouwer et al. 2019). Other putative molecular initiating/key events for PFAS, in addition to nuclear receptor activation, include gap junctional inhibition to disrupt cell–cell communication, mitochondrial dysfunction, interference of protein binding, partitioning into lipid bilayers, oxidative stress, altered calcium homeostasis, and inappropriate activation of molecular signals controlling cell functions. Many of these effects are consistent with a nonspecific action of PFAS on the cellular lipid membrane (Spector and Yorek 1985; Bourre et al. 1989; Dodes Traian et al. 2012; Casares et al. 2019). However, these alternative events lack robust evidence to support a specific pathophysiological role in the multifaceted effects of PFAS. A better characterization of the modes of action/AOPs for PFAS toxicities remains an important area of future investigation, necessary to improve our understanding of PFAS impacts on human health.

At present, there is insufficient evidence to determine which of, and to what extent, these molecular interactions play a pathophysiological role in observed adverse outcomes of PFAS (Michigan PFAS Science Advisory Panel 2018). Hence, there is a need to integrate such mechanistic information into a weight-of-evidence framework, first by establishing the mode of action or AOP linking a proposed chain of key events to an adverse outcome and then by demonstrating that at human exposure levels of PFAS the established AOP or mode of action is causal in the adverse outcome observed. The substantial advantage

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offered by such an approach is the ability to read across from representative members of appropriate PFAS groupings, based on quantitative information from new approach methodologies and exposure estimates. Hence, better characterization of the modes of action/AOPs for PFAS toxicities remains a critical area of future investigation and will allow us to understand which adversely PFAS-modified pathways must be interrogated prior to new chemicals joining this class. Predicting PFAS activity in the body should be the goal prior to approving novel PFAS for use.

New approaches for developing PFAS toxicity information

When it comes to determining which PFAS should be prioritized for further testing, there are too many chemicals, even in one subclass, for traditional approaches. Numerous creative and high-throughput methodologies are being developed and tested to provide valuable data on PFAS with no toxicity data.

Collaborative approaches. Problem formulation and approach must be guided by available equipment, funds, and technical staff, and important principles: 1) What biological activity and toxicology information can be generated in a *responsive time frame*? 2) Can this information be used to make public health decisions? 3) What are appropriate tools to bring to this problem (platforms, species/sex of cells used, metabolic competency of the model system, and data analysis)? 4) How do we organize, and what are the best mechanisms to report useful biological activity/toxicological information?

Developing "how" to evaluate potential health effects of new PFAS requires some thought to PFAS heterogeneity. Although subclass names have been suggested by several investigators (Buck et al. 2011; Wang Z et al. 2017; Sha et al. 2019), there is still disagreement on those groupings. In addition, half-lives and biological persistence are not predictable based on structure, and exposure routes may be complex. Given that traditional approaches to generate toxicity information are resource-intensive, new approach methodologies, which may include in vitro high-throughput toxicity screening and toxicokinetic testing, will be needed to inform further (in vivo) testing of PFAS.

One example of how agencies/institutes are collaborating to prioritize a list of PFAS needing further study is the REACT Program (Responsive Evaluation and Assessment of Chemical Toxicity). Scientists from the USEPA and the National Institute of Environmental Health Sciences (NIEHS) National Toxicology Program have joined forces to determine if read-across approaches would work. Essentially, they will use existing data for a data-rich substance (the source, e.g., PFOA or PFOS) as an anchor for a data-poor substance (the target, a novel PFAS), which is considered similar enough to the source substance to use the same data as a basis for the safety assessment. For example, the US National Toxicology Program 28-d PFAS or chronic PFOA data set (National Toxicology Program 2020c) could be used as an anchor. The goal is to group PFAS by biological activities and then use in vitro to in vivo extrapolation data and models to estimate oral equivalent exposures for PFAS. For example, multiple biological endpoints (Table 2) were chosen to generate data on 150 PFAS (Patlewicz et al. 2019), representing several structural subclasses for use in read-across.

Selecting assays shown in Table 2 based on PFOA and PFOS health effects covers a broad range of biology. However, because of the structural diversity of PFAS, biological activity of subclasses of PFAS may be missed; but this can be addressed in 2 ways. First, using transcriptomics as a screen, similar and unique pathways altered by different PFAS can be identified. Second, structure–activity relationships may predict potentially missing biological activities. As an example, Leadscope model predictions conducted at the NIEHS predicted biology that was covered in assays already chosen for evaluation, which increased confidence in the approaches chosen. Because model predictions are only as robust as data sets from which they are generated, these outputs should be used to identify assays for screening efforts and not as synonymous with toxicities induced by PFAS. Ultimately, the REACT program aims to prioritize PFAS for additional targeted testing and follow-up with in vivo studies as needed.

Molecular dynamics and protein interactions. Advances in computational tools, many developed for drug discovery, allow environmental and public health researchers to better anticipate some impacts of emerging contaminants even in the absence of substantial experimental data (Rabinowitz et al. 2008). For example, molecular docking and molecular dynamics to predict strengths of interactions between biomolecules and

TABLE 2: Fit-for purpose assays proposed in the REACT program

Endpoint of interest	Assay proposed
High-throughput transcriptomics	Metabolically competent human liver cells/MCF-7 (Tempo-Seq®)
Hepatotoxicity	2D HepaRG® cells
Developmental toxicity	Zebrafish embryo assay
Developmental neurotoxicity	Multielectrode array in neonatal cortical cells and neurite outgrowth
Immunotoxicity	Cytokine alterations in human vascular endothelial cells (BioSeek®)
Hepatic clearance	Metabolic clearance in 50 donor-pooled hepatocyte suspensions
Plasma protein binding	Serum protein binding assay using human serum
Enterohepatic recirculation	Qualyst B-CLEAR® hepatocyte transporter assay
In vitro disposition	In vitro disposition in cell lines under study

REACT = Responsive Evaluation and Assessment of Chemical Toxicity.

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contaminants can be an in vitro screening tool for assessing legacy and emerging PFAS for bioaccumulation potential, to identify potential sites of toxic action (Salvalaglio et al. 2010; Ng and Hungerbuehler 2015; Cheng and Ng 2018; Li et al. 2019) and to gain insights into toxic mechanisms (Sheng et al. 2018). Relatively strong binding with particular proteins (e.g., serum albumin, liver fatty acid binding protein) has already proven useful in correlating PFAS structure with potential for bioaccumulation (Ng and Hungerbühler 2014; Cheng and Ng 2017). Tools including molecular docking and molecular dynamics can correlate relative binding affinities of emerging PFAS with these target proteins and subsequently compare with affinities of legacy chemicals with known bioaccumulation potentials, thus providing a first-tier rapid screening mechanism (Luebker et al. 2002; Cheng and Ng 2018).

The use of fluorinated substances in pharmaceutical products has led to an unexpected data source for discovery of structural features in PFAS associated with various types of bioactivity. These data were recently used to train machine learning models to predict potential bioactivity for thousands of untested PFAS (Cheng and Ng 2019). Classification approaches such as these serve as preliminary screening tools for identifying PFAS as a first step in a tiered assessment when detailed mechanistic information is not available.

Addressing mixtures. Based on their potential for complex exposure patterns, PFAS are a mixtures issue. Communities with water-monitoring programs reporting PFAS concentrations demonstrated that they are exposed to mixtures of PFAS. This mixture may be from one or more point sources releasing multiple PFAS and/or PFAS by-products into the air and water, such as a Chemours plant in North Carolina, and suggest that exposures may be substantial (McCord and Strynar 2019). However, numerous other PFAS sources are known to impact community exposure to PFAS mixtures, such as landfill leachate, biosolids recycling, and aqueous film–forming foam contamination of drinking water sources, among others (Sunderland et al. 2019; Solo-Gabriele et al. 2020). Aqueous film–forming foam and other mixtures evident in drinking water, food packaging, health and beauty products, and food-based sources are often poorly characterized (Sunderland et al. 2019; Susmann et al. 2019).

Discussions on whether PFAS may be addressed using a relative potency framework or toxic equivalency factor approach are ongoing. Substances could be grouped by bioaccumulation and persistence (toxicokinetics), function (biology), molecular initiating events, with potency factors derived from several assays, or subclass (structural similarity).

SPECIAL CONSIDERATIONS IN FUTURE STUDY DESIGNS

Future epidemiological studies

Future human studies need to characterize immune outcomes including (and not limited to) immune effects from exposure in early pregnancy and possible roles of PFAS in initiating allergic and autoimmune processes, conditions for which a dose response is hard to predict. Interactions of immune pathways with liver and lipid toxicity deserve additional consideration.

Liver and lipid studies have reasonably characterized associations between PFAS and effects and should now address why and what to do about it. Characterization of possible a priori susceptibility, such as in the obese, is important. Human and animal lipid data suggest that future experimental studies should focus on mitochondrial toxicity, alterations in bile acid metabolism, cholestasis, and resultant steatosis. These outcomes are already known to be associated with altered serum lipids, liver enzymes, and uric acid in the human population regardless of PFAS (Cohen and Fisher 2013; Sattar et al. 2014; Arguello et al. 2015; Jensen et al. 2018).

Studies of human kidney markers related to PFAS exposures illustrate the importance of understanding physiology to inform study design choices and reasonable interpretations. These substances have complex excretion mechanics that vary with dose, state of the healthy or progressively diseased kidney, as well as a potentially additional causative effect on kidney disease outcome(s). Appropriate definition of biological and mechanistic targets and more precise investigation of PFAS subclasses will better inform study designs and research questions. For example, consistent reports of disrupted cholesterol metabolism should prompt mechanistic studies evaluating effects on steroid hormones that may influence cancer, fecundity, lactation, and developmental signals seen in human population data. More attention could be given to effects of PFAS on the hypothalamic–pituitary–gonadal axis and then reconsidered based on life stages.

The history of long-chain PFAS studies indicates that collaborative team approaches featuring clinical, epidemiologic, computational modeling, and laboratory toxicological expertise are needed. Future population designs and more sensitive analytical methodologies should address replacement chemicals, typically found as mixtures; study designs must account for shorter PFAS half-lives and unpredictable PFAS detection in exposed individuals/communities. Innovative use of biomarkers in specifically designated risk subpopulations (obesity, immune) will likely be important.

Sex differences, nonmonotonic dose responses, sensitive subpopulations

Although serum-level differences exist between men and women similarly exposed to individual PFAS, sex-dependent differences in half-life have not been reported in human populations for short-chain (PFBS, PFBA) or long-chain perfluoroalkyl acids thus far (Li et al. 2017b). Perhaps the half-life differences between the sexes is similar to interindividual variability and cannot be detected above background, or studies deriving data sets used to model half-lives were not designed to detect sex differences (convenience sampling or workers were mostly male, etc.). However, sex-specific elimination half-lives are defined (Table 1) for some PFAS in rodent models.

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In addition, developmental exposure studies in experimental models have consistently shown effects at lower doses than adult-only exposures and should be given priority in testing replacement chemicals. In vitro and alternative models that capture developmental susceptibility are encouraged. In summary, care should be taken in testing replacement PFAS in rodent or alternative (cell-based or zebrafish, for example) models to consider 1) the possibilities of sex-based differences in elimination half-lives, 2) dose range used (to include human relevant exposures), 3) life stage represented in the model system, and 4) variability of the response to enable the use of data generated for risk assessment.

Future experimental model studies

Experimental rodent studies have been essential in confirming PFAS health effects (liver and thyroid disease, lipid homeostasis), even when effects were not identical to those in humans; in some cases, novel targets (mammary and immune changes) were identified in animals. Future animal, cell-based, and high-throughput toxicity screening should enhance transparency in reporting to include blinded dose allocation, reporting of all data, adherence to Animal Research Reporting In Vivo Experiments (ARRIVE) guidelines (Kilkenny et al. 2010), and dose ranges that approach human relevance (adjusted to reflect the differences in elimination between species and potentially chronic exposures) so that they suitably inform systematic reviews that may be used in chemical regulation.

Model selection for health effects evaluation is critical. An appropriate model should be sensitive, be susceptible to the outcome(s) of interest (obesity, immune), and produce outcomes that will inform human health effects. Alternative research models, such as transgenic mice, zebrafish, developmental models for most affected target tissues, and diet-challenged designs in susceptible rodent strains, will strengthen our knowledge of PFAS-related health effects. Validation of fish neurobehavior models to inform mammalian, including human, developmental responses is needed.

Finally, advanced human cell-based platforms—that have been validated for relevant outcomes in humans—will facilitate concurrent screening of larger numbers of PFAS, but bioavailability of PFAS in the culture system needs to be understood because binding to media proteins or labware, the instability of some PFAS in some vehicles, and altered metabolism may exist in some cases (Gaballah et al. 2020; Liberatore et al. 2020).

Future alternative approaches

One way to determine the toxicity of the large number of PFAS compounds currently used in commerce is to develop quantitative structure–activity relationships (QSAR). Such QSAR attempt to define relationships between a PFAS compound structure with a specific biological activity or response that identifies or is a biomarker for toxicity. Few data are available for receptor binding of PFAS, mainly limited to a few PFCAs and PFSAs; and even between carboxylates and sulfonates of similar chain length substantial differences have been observed (Cheng and Ng 2017, 2018). If there are substantial differences between perfluoroalkyl carboxylic and sulfonic acids, which differ only in their acid head group, construction of successful QSAR for the large and diverse class of all PFAS will be particularly challenging. Several QSAR may be developed, each predictive of toxicity of a distinct class or subclass of PFAS, based on a unique functional moiety or other feature. Although this brings additional challenges in finding sufficient data for QSAR training and validation, big data approaches, such as the recently developed machine learning models to predict PFAS bioactivity (Cheng and Ng 2019), show promise for advancing these computational approaches at the screening level.

For example, it may be determined by affinity for receptor-specific binding and nonspecific interactions with cellular membranes that the specific toxic effect exhibits a multiphasic dose response reflecting 2 potential modes of action. In addition, the critical effect may change with levels of PFAS exposure. Add to this that people are typically exposed to PFAS mixtures, each of which may have a different affinity for a binding site and ability to impact cellular membrane fluidity, and the potential to predict PFAS toxicity becomes extremely complicated. In the foreseeable future, we may be limited to assessing PFAS toxicity using high-throughput assays designed to inform regulators as to the relative toxicity of PFAS mixtures or compounds. Such approaches are suited to the use of artificial intelligence (i.e., machine learning approaches) that integrate data from multiple sources to identify bioaccumulation potential, relevant pathways triggered, protein binding affinities, and modes of action involved in the development of individual and mixture toxicity of PFAS.

The utility of any future approach to determining PFAS toxicity must consider tissue-specific modes of action. Such an approach may rely on molecular interactions with specific binding sites on enzymes/storage/transport proteins or the nonspecific ability to alter cell membrane fluidity by which membrane-bound protein activities are altered within a particular organ/system. Regardless of the mode of action, model, and/or simulation, the predictive result should be biologically plausible and represent dose–effect responses across species.

CONCLUSION

Future research on the health effects of replacement PFAS and mechanistic studies on legacy PFAS must apply "lessons learned" such as those highlighted in the present review. There are only a handful of PFAS with enough health effects data for use in decision-making, as evidenced by state-led standard setting. There are numerous health effects reported for those PFAS tested, which sets this family of chemicals apart from many others and elevates the need for precautionary action. With hundreds of PFAS lacking health effects data, translational research teams using innovative methodologies and carefully designed studies will be critical to our state of knowledge on

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PFAS-related health effects and our enhanced strategies for informing risk assessment of this large family of chemicals.

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Micropollutants characteristics, fate, and sustainable removal technologies for landfill leachate: A technical perspective

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ABSTRACT

Waste disposal in landfills has gained momentum in the last few decades as it is one among the easy and cost-effective method for waste management for the accelerating global population. Heterogenous nature of the waste ending up in landfills makes it difficult to predict the fate of these pollutants making it difficult for its bio-monitoring and specific treatment. Among the major issues associated with landfill treatment, liquid percolation from this heterogenous waste substrate collectively referred as leachate poses the greater risk to environment through soil and ground water pollution. The diversity of micropollutants presents in landfill leachates ranging from microbial pathogens, pesticides, microplastics, pharmaceuticals and other harmful chemicals calls for crucial attention towards improving methods used for their treatment and removal. This review summarizes the key components of landfill leachates with specific emphasize on micropollutant content of leachate. Further the features of most recent and advanced technologies that were successfully explored for micropollutant removal from landfill leachates are presented in this review.

1. Introduction

Rapid industrialization, population growth and urbanization has triggered the generation of huge amounts of diverse waste materials often ending up in landfills as their final destination in its life cycle. This heterogeneous waste undergoes various physical, chemical and biological changes in landfills resulting in the liberation of more toxic micropollutants especially in the leachates originating from these landfill disposals [1]. Landfill leachates are produced due to a variety of processes in the landfills that include the release of inherent water content in the waste, percolation of rainwater through waste and enzymatic along with physicochemical degradation of the waste [2]. Irrespective of the mode of its origin, landfill leachates contain substantial amounts of micropollutants like organic compounds, pesticides, toxic residues, pathogens, heavy metals, dyes, microplastics and other harmful compounds that can pollute the soil and groundwater. These pollutants cause severe environmental pollution due to accelerated chemical oxygen demand (COD), presence of dyes and high nitrogen content due to ammonia and nitrogenous compounds [3]. High degree of variability is observed in landfill leachates based on the seasons, storage period and other geographical features [4]. Although the landfill leachate treatment is not a relatively new concept as it encompasses many conventional and currently existing technologies. But these methods cannot individually be recommended as fully effective in removing the micropollutants from

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landfill leachates [5]. This calls for development of integrated strategies and improvement of existing technology to cater to the heterogeneous nature of landfill leachates. Understanding the precise composition of the micropollutants in the landfill leachate is crucial to devise techniques for their removal and further treatment [6]. Despite several research endeavours related to landfill leachate treatment a wide knowledge gap still exists in this concept when these techniques are applied for onsite treatment over a long period of time. The performance and reproducibility of the results obtained in a laboratory scale when applied to the onsite treatment is one among the major challenges to the researchers working in this domain [7].

Taking into consideration the importance of this concept, the presented work critically reviews the emerging techniques and integrated approaches that can be used for the removal of micropollutants from the landfill leachates. This review work focuses equally on the compositional aspect of micropollutants to suggest and recommend the strategies for their mitigation. Further the limitations of each of the existing technologies and their possible improvement for revolutionizing the field of landfill leachate treatment is presented under various sections. Furthermore, the challenges and future directions in technological perspective for landfill leachate treatment are addressed.

2. Composition and characteristics of landfill leachate

During last decade, several countries in the world have seen continuous commercial and industrial growth to satisfy the need of affluent lifestyle which resulted into increased municipal and solid waste generation. Most commonly the municipal waste is dumped into landfill as it is the considered an economic option available. Leachate is generated when the inherent water present in the waste deposited in the landfills oozes out or when rain water mixes with the waste in landfills and percolates the soil to contaminate the nearby water sources. Landfill leachate is one of the major pollution caused by open dumping of municipal waste as the landfills are not engineered to collect liquid leachate [8,9]. The organic compound rich wastes in landfills goes through series of chemical and biological modifications. In new landfills which contains high amount of biodegradable waste, initially enters an anaerobic fermentation stage and produces volatile fatty acids (VFAs) [10]. Under high moisture content, acid fermentation takes place and releases free volatile fatty acids. This phase is known as acidogenic phase [11]. During the maturation phase of the landfill, methanogenic microbes grow and convert the volatile fatty acids into biogases like methane and carbon dioxide [12]. At this phase, the concentration of degradable organic matters is lower than the non-biodegradable matters [13]. As different types of wastes are dumped over the years, the different layer will be on different phases of conversion [14]. The characteristics of landfill leachate greatly depend on types of the wastes, state of decomposition, age of landfill, and the climate condition [15].

2.1. Characteristics of landfill leachate

Designing a treatment strategy for landfill leachate primarily requires an in-depth understanding about the composition and physicochemical characteristics of landfill leachate. This analysis serves as important checkpoints to evaluate the efficiency of the designed process, durability and reproducibility. There are different parameters to characterize the landfill leachate such as biological oxygen demand (BOD), chemical oxygen demand (COD), total organic carbon (TOC), pH, heavy metals and suspended solids [16]. The state of oxidation of carbon and biodegradability of organic compounds can be indicated by the ratio of BOD, COD and COD to TOC [17]. The composition of waste, seasons, temperature, precipitation and moisture content can alter the characteristics of leachate [11,18]. While oxidation reduction potential, concentration of metals, total and volatile suspended solids remains unaffected by seasonal changes, decreased COD and dissolved oxygen (DO) concentration results from high temperature in summer [19]. The landfill leachate composition and characteristics can be greatly affected by the age of landfill. Landfill older more than ten years is referred to as old while landfill between 5 and 10 years and older than 5 years are known as medium and young landfill respectively [20]. In new landfill leachate, low molecular weight organic compounds are present in high concentration. It has low pH and biodegradability index is also high. In addition, it contains several plasticizers and endocrine disruptors [21–23]. In stored landfill leachate (depending on their composition), high molecular weight components like humic acids and fulvic acid are present in high concentration, which increases the pH and decreases the biodegradability index. Stored landfill leachate contains low concentration of heavy metals as increased pH in stored landfill leachate reduces heavy metals solubility [14,24]. Some of the dissolved organic matter present in landfill leachate can absorb UV rays and light. This produces a dark color solution and it is known as chromophore dissolved matter [25,26]. Humic acids, fulvic acid and hydrophilic faction can be recovered when landfill leachate is treated with resin (combination of XAD-8 and cation exchanger) as the solubility of landfill leachate differs greatly [27]. The fourth fraction can be hydrophobic aromatic compounds which can interact with chlorine to produce by-products. Biodegradable compounds such as simple carbon chain structures are originated from initial decomposition of organic matter while the non-biodegradable components of landfill leachates are formed by condensation and polymerization of by-products from biodegradation [28,29]. Recently, pollutants like endocrine disrupting chemicals, personal and pharmaceutical care products such as antibiotics, stimulants, anti-inflammatory and beta blockers have been identified in landfill leachate. In addition to that disinfectants, their by-products and antibiotic resistance genes are also identified in landfill leachates which are considered to be a great threat to human health [30].

2.2. Composition of landfill leachate

Landfill leachate is composed of percolated rainwater, inherent water in waste, and water from the waste biodegradation. It contains dissolved organic matter, heavy metals, salts, chlorinated aliphatic and pesticides [31]. The detailed composition of municipal solid waste leachate is presented in Table 1.

Dissolved organic matter in landfill leachate is composed of various organic compounds such as low molecular weight volatile fatty acids, humic acid and fulvic acid [34].

Dissolved organic matter is a major portion of landfill leachate and it is a complex mixture. Therefore, it may affect the leachate composition.

Table 1
Composition of municipal solid waste leachate [32,33].

Components	Range	Unit
pH	7.29–8.61	mg/L
COD	580–1821	mg/L
BOD	76–701	mg/L
Total suspended solids	191–740	mg/L
Volatile suspended solids	72–329	mg/L
Total nitrogen	95–401	mg/L
Organic nitrogen	29–46	mg/L
Ammonia nitrogen	66–364	mg/L
Total phosphorus	1.4–15.7	mg/L
Magnesium	126–419	mg/L
Calcium	192–430	mg/L
Chlorides	490–1190	mg/L
Volatile dissolved solids	1248–2175	mg/L
Total dissolved solids	2969–6384	mg/L
Cadmium	0.022–0.13	mg/L
Zinc	0.22–0.435	mg/L
Chromium	0.05–0.08	mg/L
Mercury	0.005–0.017	mg/L
Ethylbenzene	2.4–89	μg/L
Xylene	9.3–225	μg/L
Toluene	0.3–182	μg/L

The composition of dissolved organic matter present in the leachate varies with the different phases of leachate biodegradation in the landfill. The concentration of fulvic acid varies from 6 to 30 % in aerobic and anaerobic leachates. Isolation and purification of leachate can help in better characterization of dissolved organic matter present [35]. Dissolved organic matter from leachate polluted groundwater showed the presence of fulvic acid (49 %), humic acids (8 %) and hydrophilic components (25 %) [11]. Leachate contains 95 % of low molecular weight volatile fatty acids and 1.3 % of high molecular weight acids [36]. In addition, it contains several volatile amines and alcoholic compounds. It has been observed that during the during methanogenesis compounds such as volatile fatty acids, amines and alcohols were absent.

Leachate also consists of humic substances which estimate about 60 % [37]. Composition of landfill leachate mixed with soil and water was observed to have high concentration of carbon, hydrogen, sulphur with low concentration of phenolic groups.

Maturation and stabilization of landfill can affect the concentration and composition of inorganic macro-components. During methanogenic phase with a pH range of 6.5–7.5, the concentration of magnesium (40–350 mg/L), calcium (20–600 mg/L), manganese (0.03–45 mg/L), iron (3–280 mg/L) and dissolved organic matter was low [38]. This is due to a high pH of the leachate. The sulphate concentration reported was low because methanogenic bacteria can reduce sulphate into sulphide. The concentration of chloride, potassium and sodium remain unchanged in methanogenic and acidogenic phases as the effects of sorption, precipitation and complexation on these compounds are negligible [39].

Ammonia and nitrogen are two important inorganic compounds present in landfill leachate. In leachate, ammonia is mainly generated from decomposition of proteins [40]. Ammonia is recalciterant to degradation. Therefore, it is present in high concentration in leachate. A study showed that the ammonia concentration was not reduced even after 30 years of aging of the landfill [41].

Heavy metals present in landfill leachate are low in concentration. The low concentration of heavy metal is due to sorption and precipitation of these heavy metals. Studies have shown that waste rich in organic matter harbours methanogenic bacteria which can absorb these heavy metals [42]. The anionic components such as sulphides (originated from reduction of sulphate) and carbonates are typically present in landfill. These sulphides and carbonates can interact and precipitate the cadmium (Cd), nickel (Ni), zinc (Zn), copper (Cu) and lead (Pb) [43,44]. Carbonates are present in high amount in landfill which have higher solubility to metals than sulphides although sulphide precipitation dominates the heavy metals immobilization. But it is not capable of chromium precipitation. Chromium is only precipitated by hydroxides. At neutral or high pH hydroxides and phosphates can also form precipitates with heavy metals. Sorption to colloids and complexation with inorganic and organic ligands are also known mechanisms for heavy metals immobilization. According to, a study reported that colloidal fraction is highly associated heavy metals. It was observed that significant concentration of chromium, lead and zinc are found in colloidal fraction [45].

Municipal solid wastes are the main source of xenobiotic compounds. Over last two decades, the concentration of xenobiotic compounds entering landfill has significantly decreased. Though countries with fewer restriction have seen presence of broad ranges of xenobiotic compounds [46]. The composition of xenobiotic compounds also varies with age, waste composition and landfill technologies. Mono-aromatic hydrocarbons such as benzene, xylenes, toluene, ethylbenzene and halogenated hydrocarbons like trichloroethylene and tetrachloroethylene are most abundantly found in landfill leachate In contrast, phenols and cresols are found in minor concentrations (in μgL^{-1}) [47]. Pesticides are class of xenobiotic compounds found in landfill leachate. Out of twenty one different pesticides identified in landfill, Mecoprop, Bentazon, 2-methyl-4-chlorophenoxyacetic acid (MCPA) are most commonly found along with chlorpropham, ametryn, dichlobenil, atrazine, hexazinon. Different surfactant used in shower gel and laundry detergents like benzene and naphthalate sulfonate are present in leachate in a concentration ranging from 1 to 10 mg/L [48]. Another type of surfactants, alkylphenol polyethoxylated and their metabolites are also found in landfill. Different phthalates like di-ethyl-phthalate, di-(2-hexylethyl)-phthalate, di-*n*-butyl-phthalate and their common metabolite phthalic acids are reported to be present in mg/L concentration in leachate [46]. The concentration of gasoline additive, methyl-tert-butyl-ether (MTBE) present in leachate was 30-35 $\mu g\ L^{-1}$. Synthetic cyclic ethers such as dioxanes and dioxolans with high odor and low odor intensity are generated from disposed coating and painting products and alkyle resin like materials. Adsorbable Organic Halogen (AOX) was present in the range of 200-5000 $\mu g\ L^{-1}$ [49,50].

3. Micropollutants & contaminants present in landfill leachate

Leachate is aroused from physicochemical changes in solid wastes that are hazardous and even have the potential to cause cancer [51]. Therefore, leachate is known as refractory wastewater and includes a wide range of pollutants in high concentrations. It is understood that inadequate management accompanies chemical generation. Leachate can be toxic, mutagenic, genotoxic, and have estrogenic effects which results in substantial harm to the ecosystem, including human beings [52]. Studies reveal that leachate has a complex chemical composition which may additionally contain several micropollutants in addition to common contaminants already present [53].

Landfill leachate contains a variety of micropollutants, including synthetic compounds like- phthalic acid esters (PAEs), polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), pharmaceuticals and personal care products (PPCPs), pesticides, microplastics (MPs), *per*- and polyfluoroalkyl substances (PFAS), cleaning detergents, microplastic, steroids, etc. [22,54]. Despite being exist in very lower concentrations ($\mu g\ L^{-1}$ to $ng\ L^{-1}$ or even lower), micropollutants still significantly adversely affect ecosystems, the economy, and humans [55]. Fig. 1 represents various micropollutants and their toxic effects on humans.

Although modern landfills are well constructed to minimize environmental harm. However, the production of landfill leachate being full of resistant micropollutants is an unavoidable side effect and contaminates the neighbouring aquatic systems [56]. Some of the most reported micropollutants are discussed in detail below.

3.1. Phthalate esters

Phthalic acid diesters (PAEs), known as phthalates, are lipophilic compounds widely engaged in many cosmetics and personal care product These include nail paints, shampoo, deodorants, hair gels, body wash etc. They are also added to plastics to increase the malleability and flexibility of polymer chains (like making vinyl plastics synthesis). Besides that, they are used in paper coating, paints, and adhesives-making industries [21]. Actually, the production of PAEs began in 1920 and increased after 1950, as a result of the increased use of polyvinyl chloride (PCV) in transparent bags, plastic toys, medical appliances, construction, and insulation materials [57]. Among diverse applications, plastics industry currently uses roughly 65 % of all phthalates and it uses most frequently the phthalates having high molecular weight (HMWPs) [58]. Since phthalates favour physical binding instead of chemical. They leak and migrate into the environment and pose substantial risk to the environment [59]. These substances exhibit environmental stability, lyophilic characteristics, and a propensity for bioaccumulation [60]. In addition, the polymer matrix's strong environmental resilience guarantees years of possible phthalate release from completed goods.

PAEs pose a major risk because the landfills lack suitable environmental protection systems (such as leachate collection systems). Further, their widespread existence and resistance to degradation lead

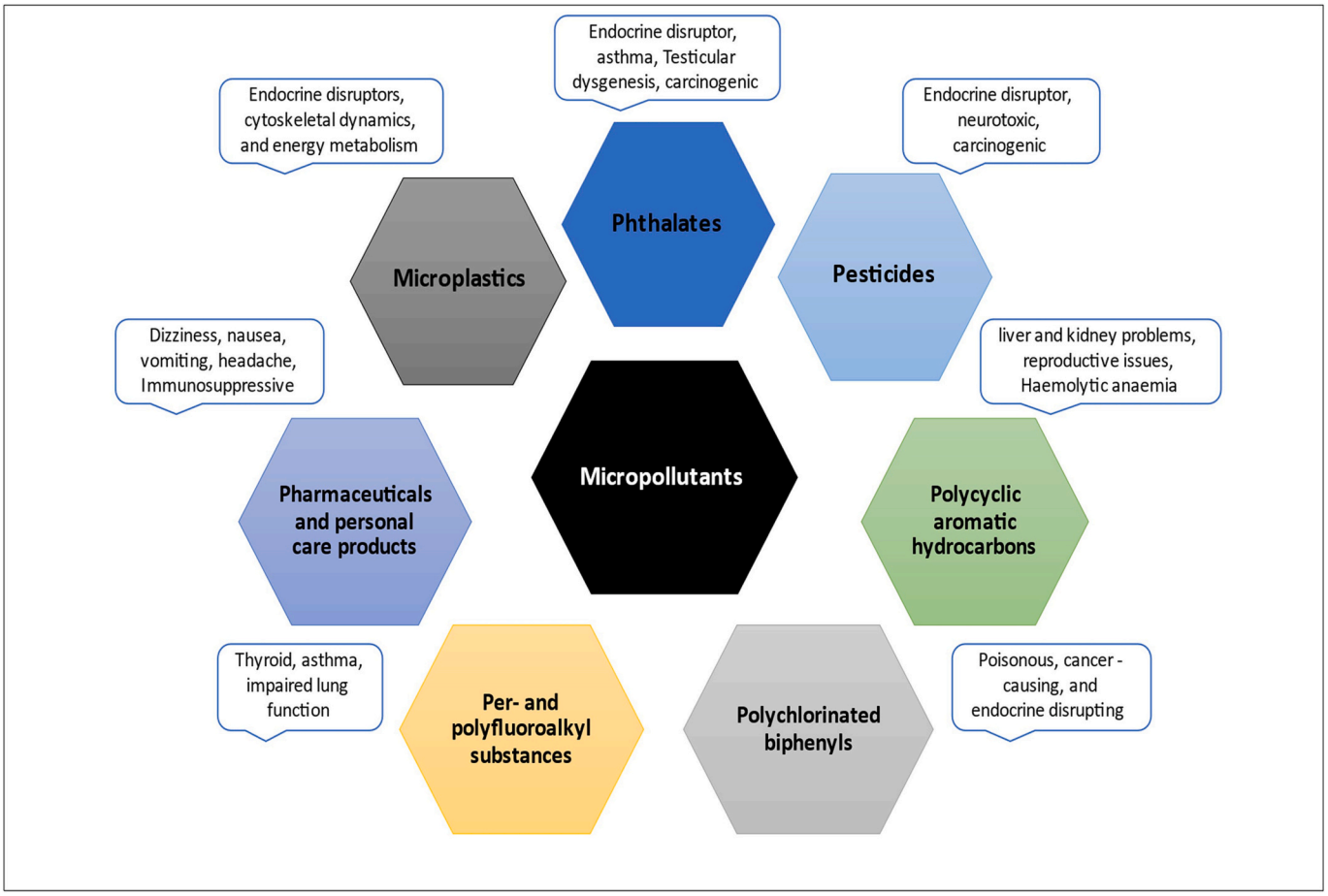


Fig. 1. Types of micropollutants present in landfill leachate and their harmful impacts on human beings.

to accumulation in aquatic habitats. Consequently, PAEs exposure not only risks the aquatic animals, it impacts their entire aquatic ecosystem including microorganisms, seaweed, fish, and invertebrates in seawater or limnetic zones [61]. Although it is believed that PAEs have low acute toxicity, but their breakdown products such as phthalic acid monoesters (PMEs) are harmful to animals [57]. They are detected in the gastrointestinal tract, and are regarded as endocrine disruptor (ED) chemicals by several environmental agencies like the US Environmental Protection Agency (USEPA) as well as European Environmental Agency (EEA) [62]. Besides the digestive system, they can be absorbed by the respiratory system, and some being more lipophilic including the most prevalent di (2-ethylhexyl) phthalate (DEHP) which can be absorbed through the skin [63]. After entering the body, they reach blood, and can cross the placental barrier [64].

Several PAEs, such as DEHP, dimethyl phthalate (DMP), diethyl phthalate (DEP), and dibutyl phthalate (DnBP) are recognised as hazardous micropollutants in different countries. Mohammadi et al., studied the leachate samples in Bushehr port (Iran) and reported a prohibitive range of PAEs of about 3.27 mg/ L, where DBP and DEPH are recorded as the most abundant ones [65]. Similarly, Kotowska et al., reported the presence of DEHP with the maximum concentrations up to 143 mg/ L in almost all the wastewater samples studied in Poland [66].

3.2. *Microplastics*

Microplastics (MPs) (<5 mm in size) are ubiquitous in landfill leachates and may enter the surrounding environments [67]. Depending upon their origin, MPs can be divided into- primary and secondary MPs. Primary MPs are intentionally produced of small-sized plastic, which can be traced in medications, textiles, cosmetics, and personal care goods. Whereas secondary MPs are produced as a result of the breakdown of massive plastic debris [68]. MPs are heterogeneous in nature and their characteristics are critical for their diffusion in the environment and bioavailability [69]. There are many studies coverings the prevalence, nature, sources, and effects of these MPs. MPs exert both direct and indirect negative impacts on living beings. These include disruption to the endocrine, cytoskeletal, and metabolism [60]. A research group investigated the impacts of polystyrene (PS) MPs in marine medaka (*Oryzias melastigma*) and observed tissue damage, sex-specific reproductive disruption, and transgenerational effects in the fish [70]. In another study, PS MPs exposure in mice resulted in excessive production of ROS, which lead to abnormal patterns of skeletal muscle regeneration, with suppressed myogenic differentiation of myosatellite cells and their redirection towards adipogenic differentiation [71]. Similar to this, repeated oral administration of polyethylene (PE) MPs to mice, results in higher neutrophils cells levels in the blood, tissue damage, disturbed immunological response, and altered numbers of live births, sex composition, and body weight of offsprings [72]. In another study, mice exposed to PE MPs during the gestation period caused metabolic disorders in the offsprings [73].

It is observed that substantial release of MPs into the environment generally happens due to improper waste management and anthropogenic activities. As a result, lot of plastic end up being buried in landfills and emerge as a significant source of microplastic pollution [74]. The plastic waste accumulated in these landfills is exposed to harsh environmental conditions. With complex biochemical reactions and physical changes such with varying pH (ranging from 4.5 to 9) MPs are transformed to harmful products [75]. All these actions cause the plastics to break into MPs resulting in smaller plastic debris.

However, the waste treatment management sector has not given

much consideration to the existence of MPs pollution. Research has not only examined the concentrations of MPs in leachates, but also assessed their risk towards ecosystem [51]. Mohammadi et al., reported an abundance of MPs with a size of >1000 μm in the leachate samples in Iran [65]. Su et al., observed a greater concentration (4.0–13.0 items L^{-1}) of MPs in leachate in Shanghai, China [76]. In the same region, polyethylene and polypropylene were the predominant polymers in landfills. The plastic dumped in these landfills lingers for decades, and the sizes vary between 100 and 1000 mm [77]. Despite their high levels, no study has looked at their long-term impacts on the environment.

3.3. Pesticides

Globally, around 1.2–2.5 million tons of pesticides are applied annually in agricultural system against the attack of fungi and other pest infestation [78]. They are used commercially as well as in domestic pest control. Their traces are found in landfills, incinerators, and composting plants. Pesticides include a variety of synthetic chemicals. The organochlorine pesticides possess slow degradation, high toxicity on living organisms, and long-range transmission capabilities. Therefore, these pesticides are banned in developed countries. But they are still used in several developing countries [79]. Nowadays, the use of pyrethroids has been growing over time, due to their superior effectiveness and less hazardous nature as compared to the previous insecticides like- organophosphate (OP), organochlorine (OC), and carbamates [80]. As a result, they became one of the three main groups of pesticides in 2015, controlling roughly 38 % of the market worldwide. The pyrethroids market stood at $1633.03 million globally in 2016 [80].

Numerous pesticides act as endocrine disruptors (EDs), neurotoxicants, immunotoxins, and carcinogens for both animals and humans. Pesticides like OCs, OPs etc., contribute to various life-threatening diseases like Alzheimer's and amyotrophic lateral sclerosis, but primarily to Parkinson's disease [81]. The pesticide exposure also leads to a higher incidence of coronary heart diseases, childhood brain tumours, and acute lymphocytic leukemia [82].

It is noticed that pesticide along with persistent chemicals, is frequently dumped as general garbage in municipal landfill sites without even prior separation or sufficient pre-treatment. During degradation into wastes, these toxic compounds get transported via leachate [83]. Currently, pesticides are leading emerging organic compounds, threatening aquatic life and the environment [84]. Varnosfaderany et al., studied the concentration of OCPs in the different samples of Zayandehrud River in Iran, and recorded high levels of total OCP concentrations ranging from 0.1 to 51.5 ng gL^{-1} dry weight in surface as well as in core sediments [85]. A research group investigated the prevalence of pesticides in the landfill leachate in Al Amirat, in Muscat. Numerous OCPs, including gamma benzene hexachloride (γ-BHC), delta hexachloride (δ-BHC), aldrin, heptachlor epoxide, endosulfan alpha, dichlorodiphenyldichloroethylene (Pp DDE), endrin, endosulfan beta, Dichlorodiphenyldichloroethane (Pp DDD) and endrine ketone were recorded from the samples. The two pesticides with the highest concentrations were endosulfan sulphate (range: 0.011–0.016 mg/L) and endrin aldehyde (0.003–0.063 mg/L). Although exceedingly low, their concentrations could hardly be ignored [86]. In another study of Al-Ghabawi landfill in the Ghabawi area, Jordan revealed the presence of DDTs (ranged from 2.16 μgL^{-1} to 3.38 μgL^{-1}), along with other OCPs in trace amounts in the leachate sample [87].

Fenton oxidation process is employed to remove a combination pf pesticides like chlorpyrifos, atrazine and chlorfenvinphos and observed to have seventeen fold increase in the removal of these micropollutants as compared to other conventional methods [88]. In another study with alachlor 33 %(*w*/w), degradation of the pesticide micropollutant was achieved using ozonation process [89].

3.4. Pharmaceutical and personal care products

Pharmaceuticals and personal care products (PPCPs) are regarded as emerging organic contaminants (EOCs), and have raised concerns among environmentalists. PPCPs with expiration dates are disposed of in landfills and accumulate in leachates. They generally exist in trace quantities between ng L^{-1} and g/L in drinking water, groundwater, surface water, wastewater, and other aquatic environments [90]. However, it is recorded that the majority of PPCPs exert adverse physiological effects, even at this low concentration [91]. The ongoing discharge of PPCPs into the environment could have unintended negative impacts on ecological species by contaminating water supplies for a longer period of time [92]. Once ingested, the PPCPs in the environment have the potential to bio-accumulate in living beings and have negative impacts. The harmful effects include antibiotic resistance, endocrine disruptions, and primary productivity inhibition [61]. As a result, it has become pertinent to consider their occurrence, distribution, migration, transformation, and potential dangers in our environment.

Yu et al., studied the effect of seasonal variations, like rain, on the levels of PPCPs in leachate from China and recorded a significant difference in the PPCPs concentration with 17.0 μg L^{-1} in the months from April to September, while only 3.8 μg L^{-1} in the remaining months [93]. Similarly, total PPCPs range from 0.39 to 349 μg L^{-1} was reported in the landfill leachates in Shanghai [90]. Interestingly, caffeine is also a commonly reported compound, with high concentrations in different water matrices. Its concentration generally ranges between 1 and 174 μg L^{-1} in surface water bodies, and sometimes reaches to 499 μg L^{-1} in Indian wastewater samples [94]. Similarly, in Japanese streams, caffeine concentration is reported to be around 5.3 μg L^{-1} [95]. Irrigation water source in Saudi Arabia is also tested with high concentrations of caffeine (21 μg L^{-1}), diclofenac (1.4 μg L^{-1}), paracetamol (3.1 μg L^{-1}), and diazinon (1 μg L^{-1}) [96]. The reported studies demonstrate that the micropollutants are not managed properly and reaches to the landfill leachate.

3.5. Polycyclic aromatic hydrocarbons

Polycyclic aromatic hydrocarbons (PAHs) are another category of common pollutants released into the environment via landfill leachate and generally originate from both natural and anthropogenic activities. They are originated from pyrolysis, car exhaust, and coal refinement processes [97]. These typically have two or more fused aromatic benzene rings, imparting lipophilic nature with comparatively high desorption activation energy. Based on their structures, PAHs can be divided into low and high molecular weight [98]. The water solubility of PAHs decreases almost linearly with increasing molecular mass. PAHs are also employed in manufacturing plastic goods, cosmetics, personal care items, and pesticides. However, their high mutagenic, carcinogenic, and teratogenic potential has drawn much attention in water, air, and soil pollution research. The data suggest that carcinogenicity potential increases with escalating molecular weight.

Additionally, they are suspected of being EDs and weakening the immune system [99]. They persist in nature for quite a long time due to their high chemical stability and resistance. These are bio-recalcitrant and have lower aqueous solubility, which did not let traditional physicochemical and biological processes to remove them. Furthermore, presence of low molecular weight contaminants hinders their removal [100]. Ates and Argun recorded the hazardous level of 10–58 μg L^{-1} PAHs in leachate present in Konya province, Turkey [101]. Similarly Qi et al., found the concentration of 5.81–1263 μg L^{-1} in the landfill leachate in China [102].

3.6. Per- and polyfluoroalkyl substances

Per- and polyfluoroalkyl substances (PFAS), are a large family of synthetic fluorinated compounds employed in an extensive range of

commercial applications since the 1950s. These possess exceptionally high thermal and chemical stability and are found extensively in municipal landfills. From landfilled wastes, the PFAS infiltrate the leachate through a chain of diverse physical, chemical, and biological processes. This process is common in landfill settings and can contaminate nearby soil and groundwater which is a threat to both human health and ecosystems [103]. Thus, the management of PFAS becomes challenging, mainly due to their widespread usage and recalcitrant degradation.

It is understood that the strongest bond in PFAS is the carbon and fluorine bond which is due to the extremely electronegative characteristic of fluorine. Thus, C—F chain segment the molecule shows high resistivity towards degradation [104]. Among the different PFAS, the most researched PFAS are- perfluorooctanoic acid (PFOA) and perfluorooctanesulfonic acid (PFOS). The MSW leachates can have PFOA concentrations in the range of 10 μg L^{-1} and PFOS about 1 μg L^{-1} in the United States [105], and in Europe [106].

PFAS began to draw the attention of scientific communities, regulatory organizations, and the general public in the late 1990s, because of their widespread occurrence [107]. The exposure of PFAS can leads to thyroid, asthma, impaired lung function, negative impacts on immune system responses, testicular tumours, and cancers of the liver and renal, kidney etc. [108]. As a result, the USEPA issued a drinking water health advice of 70 ng L^{-1} for the total of PFOA and PFOS, effective from May 2016 [109]. It is predicted that between 563 and 638 kg of PFAS is getting transferred from American landfills to the influent of municipal wastewater treatment plants (WWTPs) in 2013 [105]. It is noticed that the conventional leachate treatment processes are not good enough for the entire degradation of PFAS [110].

3.7. Polychlorinated biphenyls

Polychlorinated biphenyls (PCBs) are ubiquitous environmental pollutants with bioaccumulation potential in living beings. PCBs usually occur as a mixture of chlorinated biphenyl molecules containing 1–10 chlorines per molecule. There are 209 possible congeners of PCBs. They are persistent, lipophilic, hydrophobic, and prone to long-range transport of synthetic chemicals used in electrical equipment until the 1970s. After that, PCBs are also utilized in transformer cooling oils, lubricants in oils and greases, dielectric fluids of capacitors, insecticides, heat transfer fluids, preservatives, and impregnating agents [111]. Their wide use is the reason behind high PCB concentration detection in chlorinated goods or as by-products during the manufacture of solvents and polymer resins [112]. Environmental PCB exposure is a serious health problem due to its poisonous, cancer-causing, and endocrine-disrupting properties [113]. In the 1960–70s, there was a growing understanding of their environmental presence, and the health threats to life resulted in a federal ban on their sales and manufacture in 1979. Presently, roughly 100 t of PCBs is anticipated to be present only in Norway [114]. Tracking the release of PCB is a complicated task because the chief PCB sources might not reflect prior but rather an accidental generation. The Stockholm Convention mentions PCBs as compounds that may be generated and accidentally released from anthropogenic sources. Thus, it can be concluded that PCB issue might not get resolved without a more profound comprehension of its occurrence and fate during recycling and disposal. Table 2 discusses about the various

Table 2
A list of emerging micropollutants found in the landfill leachate.

Sr. no.	Micropollutants	Toxic effects	Country	Concentration (μg L^{-1})	Reference
1.	Phthalates				
	Diethyl phthalate (DEP)	Asthma, antioxidants enzymes alteration	Poland	<LOD-14.5	[66]
	Di(2-ethylhexyl) phthalate (DEHP)	Endocrine disruptor	Poland	73.9	[57]
			Poland	Up to 536	[115]
	Di-*n*-butyl phthalate (DBP)	Affect kidney and liver health.	China	Up to 83.3	[116]
	Dimethyl phthalate (DMP)	Testicular dysgenesis, obesity, asthma, antioxidants enzymes alteration	Poland	2.46–24.6	[66]
2.	Microplastics	Disruptions to the endocrine system, cytoskeletal dynamics, and energy metabolism	China	235.4[a]	[117]
			China	0.42 to 24.58[a]	[77]
			Serbia, Bosnia, and Herzegovina	640–2160	[118]
3.	Pesticides				
	Organochlorine pesticides	Endocrine disruptors, neurotoxicants, and carcinogens	China	0.03–0.20	[102]
	Endrin aldehyde	Immunotoxins and carcinogens	Oman	3–63	[86]
4.	Pharmaceuticals & personal care products				
	Diclofenac	Gastrointestinal problems	India	898	[119]
	Sulfamethazine	Dizziness, nausea, vomiting, headache	China	16	[102]
	Sulfadiazine	Breathing difficulty	China	0.23	[90]
	Camphor	Liver damage	USA	62.4	[120]
	Ibuprofen	Nausea, vomiting, heartburn, Immunosuppressive	India	136	[119]
	Caffeine	Increase in anxiety and disturbs sleeping	Saudi Arabia	21	[96]
			India	499	[94]
	Bisphenol A	Endocrine disruptor	Serbia, Bosnia and Herzegovina	700-2720	[118]
5.	Polyaromatic hydrocarbons (PAHs)				
	Acenaphthylene		Turkey	58.47	[101]
	Acenaphthene	Eye and skin irritation.	Turkey	25.40	[101]
			Kenya	1.0	[121]
	Naphthalene	Haemolytic anaemia	Kenya	0.06	[121]
	Benzo(*a*)pyrene	Reproductive issues	France	0.051	[122]
6.	Per- and polyfluoroalkyl substances				
	Perfluorooctanoic acid (PFOA)	Thyroid, asthma, impaired lung function	USA	10	[105]
	Perfluorooctanesulfonic acid (PFOS)	Testicular tumours and cancers of liver	USA	1	[105]
7.	Polychlorinated biphenyls (PCBs)	Poisonous, cancer-causing, and endocrine-disrupting	Oman	500–1400	[86]

LOD - limit of detection.
[a] Item/L.

micropollutants, their toxic effects, origin and determined concentrations.

4. Critical analysis on micropollutant removal technologies and future direction

The heterogeneous nature of the micropollutants present in the leachate makes it inevitable to use efficient and integrated strategies for the removal of micropollutants.

The major technologies used for the removal of micropollutants from landfill leachate are summarized in Fig. 2. Each of the listed technologies have the characteristic features and associated limitations that require critical focus in future to make them sustainable methods for micropollutant removal from landfill leachate.

4.1. Electrochemical treatment

Electrochemical oxidation method was reported to be a highly efficient method for the mineralization of micropollutants as it does not produce or leaves any toxic waste residues after the treatment period [123]. The efficiency of method depends upon the choice of electrodes. Although conventionally used anodes were also efficient but they contain critical raw materials (CRMs) that were highly prone to corrosion, low efficacy, easily deactivation. Therefore, newly introduced boron doped diamonds (BDD) contributed in high mineralization efficiency. Various studies have reported the removal of harmful pollutants such as polychlorinated dibenzo-p-dioxins and dibenzofurans, polychlorobiphenyls, polycyclic aromatic compounds, organochlorine pesticides by electrochemical method. A recent study carried out electrochemical treatment of leachate collected from municipal waste with the help of boron doped diamond (BDD) anodes [124]. The study reported the removal of >95 % poly- and perfluoroalkyl substances (PFAS) [125]. The collected samples treated with membrane bioreactor (MBR) such as raw leachate and leachate with no ammonia content were operated at the current densities between 200 and 800 A/m^2. It was reported that the absence of ammonia does not have any significant impact on mineralization process. Moreover, study reported the removal of 17 PFAS with estimated value ($\sum$17PFAS = 3456 ng L^{-1}). The maximum removal of PFAS was observed at 800 A/m^2. MBR treatment significantly enhanced the energy efficiency of ELOX leachate treatment. Another study reported the elimination of methiocarb and bisphenol-A through electrochemical method [126].

4.2. Fenton-like oxidation process

Heterogenous Fenton process is recently introduced process for the treatment of landfill leachate. A recent study used Iron Molybdophospahate (FeMoPO) as a catalyst at dose of 1.75 g/L with pH 2, Hydrogen peroxide (H_2O_2) concentration 9 mM which was capable of removing 84.9 % COD and 71 % TOC using Box-Behnken response surface design [127]. Another study by Singa et al. [128] used Photo-Fenton oxidation process using Fe^{2+} and H_2O_2 as oxidizing agent. It was observed that maximum COD removal of 68 % was observed at the dosages of Fenton reagents i.e., H_2O_2 hydrogen peroxide and Fe^{2+} were optimum at 0.09 mol/L and 0.03 mol/L respectively. In a study, zirconia supported copper catalyst (Cu/ZrO_2) in heterogenous Fenton oxidation was used [129]. It was observed that employed process showen high catalytic activity with catalyst dose of 200 mg/L at temperature of 70 °C with pH 5 and 30 mL/L dose of H_2O_2 dose for the time period of 150 min. A study carried out the removal of numerous micropollutants including polycyclic aromatic hydrocarbons (PAHs), phthalates, alkylphenols and pesticides using Fenton and ozone oxidation process [130]. The Fenton and ozone oxidation process was carried out at different doses and reaction time. It was observed that employed process was able to eliminate di-(2-Ethylhexyl) phthalate, 4-nonylphenol and 4-tert-octylphenol by 5–100 % but its efficiency reduced for PAHs including four and more ring members. Another study carried out the degradation of methylparaben using Fenton, photo-Fenton and electro-Fenton process [131]. It was observed that electro-Fenton process exhibited excellent results. Thetarget pollutant was removed in 10 min by using $Ti/Ru_{0,3}$ $Ti_{0,7}O_2$ as cathode where as other methods show degradation in 20 and 16 min by Fenton and photo-Fenton process respectively. The optimum conditions in electro-Fenton process were 4 mg/L Fe^{2+}, j = 25 mA/cm^2 and 0.05 mol/l sodium sulfate. A study, utilized landfill effluent with high COD and investigated processes such as electro-Fenton (EF) and 24hotoelectron-Fenton processes in a continuous-flow reactor using boron-doped diamond anode (Nb/BDD) [131]. The results showed the COD removal of about 66 %, 68 % and 89 % with an energy consumption of

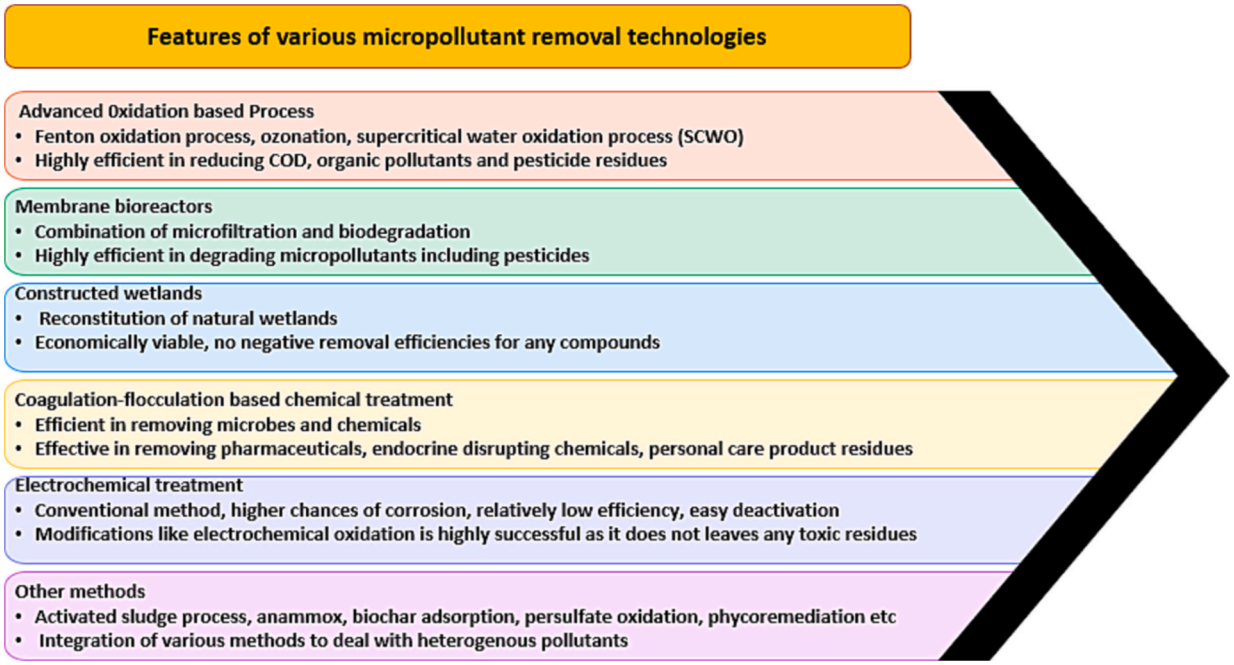


Fig. 2. Technologies for removal of micropollutants from landfill leachate.

19.41, 17.61 and 17.59 kWh kg COD^{-1} for electro-Fenton, photo electro-Fenton-UVA and photo electro-Fenton -UVC respectively during 4 h procedure. The byproducts formed during the process were the acetic and formic acids as well as NO_2^-, NO_3^-, and NH_4^+. A study carried out the biological and chemical degradation of fluconazole, clotrimazole, climbazole, and epoxiconazole by activated sludge. In the study, two novel Gram-negative bacterial strains- *Stenotrophomonas maltophilia* AsPCl2.3 and *Pseudomonas monteilii* LB2 were used. The method used UV treatment, Fenton reaction, and their combination [132]. It was observed that Fenton reaction easily removed all of target pollutants whereas biological method efficiently removed only climbazole. Similarly, another study carried out batch pilot plant experiments to study efficacy of Powdered Activated Carbon (PAC)-Activated Sludge (AS) tank for the removal of micropollutants [133]. It was reported that PAC with 2.5 mg dose contributed to 80 % pollutant removal. The characterization of removed organic matter was done with size exclusion chromatography coupled with organic carbon detection (SEC-OCD).

4.3. Ozonation

Ozonation is emerging as another productive candidate in oxidative treatment of landfill leachate. A study carried out the ozonation of atrazine, bisphenol A, carbamazepine and alachlor by semi-continuous method [134]. It was found that 0.95 g of ozone was consumed for the removal of >99 % of target pollutants bisphenol A, carbamazepine. A high ozone reactivity was observed during the process i.e., kO3 > 10^4 M^{-1} s^{-1}. The study also suggested that it is not authenticated to predict micropollutants level with hydroxyl formation yield. Yang and his co-workers proposed UVA254-based model for sensing of micropollutants using different spectroscopic measurements. Ozone based oxidative process application in high dissolved organic matter (DOM) was performed in a study [135]. The study was carried out using advanced oxidation process aided with heterogeneous catalyst along with O_3/H_2O_2 process for enhancement of indirect oxidation. It was reported that O_3/H_2O_2 process was more energy-efficient as compared to others. The catalytic process exhibited its efficiency when the chemical oxygen demand (COD) removal was <35 %. In another study introduced advanced oxidation process called as hybrid sono-electrocoagulation process for landfill leachate treatment [136]. Response surface methodology (RSM) and central composite design (CCD) approaches were used to study pH, reaction time, direct electrical current in developed process. It was observed that under the optimum conditions of pH: 5.7 with 80 min reaction time and 1.9 A direct current approximately 86 % COD was removed. A study employed packed bubble column for landfill leachates treatment and developed new method for biotransformation through ozonation [137]. The ozone columns were packed with lava rocks or metal pall rings (LBC or MBC). It was observed that with 8.35 mg dose of ozone, there was 32 % and 46 % COD removal in MBC and LBC. The highest O_3 utilization efficiency (89 %) and hydroxyl radical (OH) exposure rate (3.0×10^{-10} M s) were observed in LBC with lowest energy consumption (E_{EO}) for COD removal of 18 kWh/m^3. A recent study carried out the heterogeneous catalytic ozonation of landfill waste using Fe incorporated Ni − induced C−Al_2O_3−framework ($_{Ni}$CAF) composite [138]. Fe−$_{Ni}$CAF catalyst was synthesized for efficient catalytic ozonation. It was observed that at 4 g·L^{-1} dose of Fe−$_{Ni}$CAF, there was 100 % COD removal even at high concentration of pollutants. The efficacy of Fe−$_{Ni}$CAF/O_3 process was enhanced by the addition of hydrogen peroxide (H_2O_2) and peroxymonosulfate (PMS) due to the generation of hydroxyl radical (•OH) and singlet oxygen ($^1O_2$). A study carried out the ozonation of ^{14}C labelled micropollutants and converted them into ^{14}C labelled transformation products. $^{14}CO_2$ traps introduction shown that partial mineralization in all compounds except propyl-labelled bisphenol A and *O*-methyl labelled naproxen [139]. It was observed that after ozonization micropollutants shown high adsorption at powdered activated carbon. But adsorption rate decreased if high level of ozone dose was used during ozonation. A research group studied the impact of ozonation on adsorption properties of pollutants [140]. The studies revealed that ozonation decreases affinity of organic pollutants towards the activated carbon through site competition. They combined nitrification-Anammox, O_3/H_2O_2, and a granular activated carbon filtration (GAC) processes for the removal of total nitrogen, COD and micropollutants such as atrazine, alachlor, carbamazepine, and bisphenol A. The results revealed that there was >90 %, 80 % and 95 % removal of total nitrogen, COD and micropollutants respectively.

4.4. Supercritical water oxidation process (SCWO)

Degradation of naphthalene using supercritical water oxidation process (SCWO) was carried out in a recent study [141]. It was observed that employed method caused >98 % mineralization of naphthalene with the production of derivatives of aldehyde, propanoic acid, benzene acetic acid and benzo-furan as by products.

4.5. Constructed wetland for micropollutant removal

Constructed wetlands (CWs) are reconstitution systems of natural wetlands that are well-known for the betterment of water quality. A consortium of water, microorganisms, substrates, and plants can act simultaneously to either reduce or eliminate distinct compounds (pollutants) during the treatment. Micropollutants (MPs) have been inviting the attention of the scientists as they mostly present in trace amounts, and even after undergoing conventional treatments they ultimately end up in aquatic resources [142]. Aquaculture farms are potent sources of MPs because of the medications and artificial feeds used for aquatic animals. Influents, effluents and aquatic food products contain MPs whose removal is crucial to run an environmentally sustainable aquaculture [143]. Although both are capable of degrading organic MPs but advanced oxidation technologies are expensive than CWs. With this regard, CWs are opted due to their low cost of implementation, operation, and maintenance. Although CWs consume space and time, they sound interesting and safe due to the absence of toxic intermediate products. As study performed the removal of the European Union's eight priority micropollutants (polycyclic aromatic hydrocarbons) by three different treatment methods [144]. About 48 % of removal efficiency was achieved with CWs. No negative removal efficiencies for any compounds were reported in CWs by the authors. CWs performed relatively better than the stabilization ponds in eliminating the priority MPs. An interesting piece of research was accomplished [145] in which the distribution of MPs and lipids in various compartments of a two-stage Vertical Flow Constructed Wetland in France was investigated [145]. This would aid the operators to better understand the infiltration processes in the sludge treatment plants and develop strategies to specifically manage sludge. Most of the MPs were found in the top layers of the plant and their diffusion and settlement at the bottom lines was unavoidable.

Removal efficiency of acetaminophen (ACT) in hospital wastewater by vertical flow CWs and peroxidase enzymes generated by certain plants was studied in a study [146]. ACT is one of the major MPs in hospital wastewater as it is one of the highly prescribed drugs. *Scirpus validus*, a plant that generates peroxidase enzymes, was thoroughly checked for its capability to alleviate MPs' stress along with a pilot scale CW. Results showed that the vertical flow CW treatment plant removed initial high concentrations of ACT and nutrients to a recommended level for drinking purposes. The wastewater discharged finally met the standards of Thailand and Vietnam. A study monitored urban stormwater treatment and removal of heavy metals by CWs in all the seasons [147]. Removal efficiencies of MPs were more than97 percentage during most of the seasons. CWs can resolve the diffusion of MPs into the environment as the technology is decentralized with coexisting physical, chemical and biological processes. A vertical flow constructed wetlands (CW) was used with combination of substrates, i.e., gravel, sand, pine bark, zeolite and expanded clay to treat landfill leachate [148]. The

wetland systems were planted with *Phragmites australis*. It was observed that developed system removed heavy metals such as Zn, Ni, Cu, Cr by >50 % with total nitrogen reduction of about 70 %.

4.6. Advanced oxidation process

One of the robust and efficient means of removing the micropollutants (MPs) in the leachates, which are hitherto difficult to be eradicated by conventional means, is by exploiting the power of oxidation. Some of the recommended processes making use of this strategy include ozonation, powder activated carbon (PAC), granular activated carbon (GAC) and advanced oxidation processes (AOPs) [55]. Each of these specific techniques has its own merits and demerits. Though GAC helps adsorption and physical removal of most of the OMPs, its efficiency is dependent on the frequency of regeneration of GAC. The problem with ozonation is the generation of harmful by-products whose removal may require additional treatment later [149].

A recent study has revealed the successful removal of sixteen micropollutants, including alkylphenols and phthalates (both of which arising from the plastic waste, detergent packages and cosmetics), polycyclic aromatic hydrocarbons (PAHs) and few insecticides. The degradation potential was found t excellent, except for PAHs with four or more rings in their structure. Even though removal efficacy was found better for Fenton oxidation when compared to ozonation, the former showed more degradation products of higher molecular weight [88].

A broad range of organic micropollutants can be efficiently degraded by advanced oxidation processes (AOP) to increase the potability of water by ensuring the breakdown of complex organic material into smaller carbon chains containing more assimilable organic carbon (AOC). But this method has a few limitations - i) more AOC can potentially attract more bacterial growth ii) undesired by-products like bromate and nitrate, products of incomplete mineralization of OMPS etc. will be formed. A large-scale study conducted in the Netherlands combining AOP and Managed aquifer recharge and recovery (MARR) (has established that both these techniques complement each other in OMP removal, without compromising on the inorganic chemistry and biological stability of water. Hence the above method could be exploited as a promising technology for reducing the MPs in drinking water [150].

AOP is a recent approach in Europe, where the existing common methods for OMP removal include GAC and ozonation [151]. One of the first countries to implement legal regulations on the level of micropollutants in wastewater treatment plant effluents, was Switzerland. In their strategy, it is envisaged that such an advanced treatment method must have a very strong scientific basis, technical and economic feasibility, and societal acceptability. Therefore, only such holistic and sustainable approaches would turn out to be successful. The removal of OMPs by way of a tertiary water treatment was studied for four months in Germany In the process two methods were applied including: AOP (making use of Hydrogen peroxide and UV rays) and GAC with OMP removal efficiencies of about 76 % and 90 % respectively. Though the latter method had higher removal efficacy in the first few weeks, it showed a diminishing efficacy rate with time, whereas the AOP method when optimized at 40 ppm hydrogen peroxide and 10 kJ/m^2 UV-C light showen excellent OMP removal efficacy to the tune of almost 97 %. The low resource requirement and possible reuse of Hydrogen peroxide in the later stages, makes this approach more in line with circular economy [152].

Apart from OMPs, several metals contribute to inorganic MPs in wastewater, the removal of which are also being investigated. Chitosan was successfully used for removal of MPs owing to its properties like miscibility with other polymers, good hydrophobicity, biodegradability and low toxicity [153]. Another successful strategy was recently reported which made use of β–CD–CS–EDTA (a cross-linking polymer based on β-cyclodextrin (CD), chitosan (CS) and EDTA) for the removal of both OMPs and heavy metal ions simultaneously from wastewater [154].

4.7. Membrane bioreactors

Membrane bioreactors which combine biodegradation and microfiltration have been useful for treating old landfill leachates particularly when conventional processes turn inefficient. It is deemed more efficient due to separation by filtration instead of gravity and longer sludge retention time. Improved decomposition of MPs was observed in MBR systems due to their ability to maintain higher sludge concentrations thereby enhancing removal efficiencies [155]. With a view to understand the MP behavioural dynamics during MBR removal, modeling studies were carried out on sorption and biodegradation by calibrating various parameters [156]. First generation MBRs were costlier owing to their configuration wherein the membrane modules were located outside the aerobic tank used to supply oxygen for the microbes doing the biodegradation. The positive traits of this system include easier module replacement, cleaning and maintenance [156]. Later, different types of integrated and hybrid MBR configurations were successfully tested. One such hybrid MBR with nanofiltration capability was used for the treatment of textile industry effluent [157] and an electrically enhanced MBR was used for the removal of metals and organics [158].

Anaerobic-aerobic bioreactors result in lesser footprint when compared to the aerobic systems. Some of the specific advantages include handling large variation in organic loading, easier switching to non-working mode in case of lower load, conversion of 90 % biodegradable organics into biogas, generation of lower biomass and ability to tolerate high quantities of organics and fats [159]. An aerobic -anaerobic MBR was shown to successfully remove the pesticide atrazine from waste water [160]. Successful removal of MPs by utilizing the membrane biofouling layer was demonstrated using MBRs which were hitherto not possible by ultrafiltration or microfiltration membranes. MP removal efficiency of such a system was found to be dependent on the microbial flora forming the biofilm on the membrane surface [161]. The efficacy of MBRs depends on the type and composition of wastewater to be treated. Hence more experiments need to be performed to evaluate the performance of high retention and hybrid MBRs with a view to suit each country's requirements based on factors like scaling up, economic feasibility and long term sustainability.

4.8. Coagulation-flocculation based chemical treatment

Among the various MPs' removal processes, coagulation-flocculation has been playing a vital role in controlling the pollutants and microbes. It involves destabilization of particles in water/wastewater by neutralizing the particle charges, followed by agglomeration to end up with filterable and settleable macro-flocculants [162]. Coagulants and flocculants are organic/inorganic chemicals being used for the above mentioned process. The dissolved organic matter of leachate can influence the microbial community, changes the transport behavior of heavy metals, interact with different inorganic matter, leads to membrane fouling and decreases the effluent quality of the coagulation process [18]. A research group optimized and predicted the appropriate coagulant dose to eliminate organic MPs using four mathematical models based on turbidity and related environmental parameters [163]. Another research group performed preliminarily sorption potential of three different categories of MPs (pharmaceutical, endocrine-disrupting and personal care) on to polystyrene and polyethylene, which are widely used microplastics in recent times [164]. Microplastics serve as transport vehicles for MPs in the natural ecosystem and food chain. Researchers treated this sorption process with a coagulation method to remove the microplastics. Sorption of antihypertensive drugs (valsartan and losartan) with polystyrene was evident. No sorption of bisphenol A, parabens and sulfamethoxazole was observed. Addition of ferrous sulphate or magnesium sulphate was effective for polystyrene removal. Appreciable removal of polyethylene was obtained when magnesium sulphate was added. An elaborative study revealed that the removal efficiency was more notable with polystyrene (92.4 %) than

polyethylene (72.1 %). Similarly, another research group tried the addition of several coagulants and flocculants to raw wastewater which led to 66.6 to 97.3 % of microplastic transfer to the floating or settled sludge, ensuring effective microplastic removal in the primary clarifier [165]. Positive impact of aqueous solution of polyaluminium chloride resulted in 90 % removal. Authors concluded that this process of coagulation and flocculation would reinforce the safe usage of the sludge in the agricultural sector.

Combined effect of activated carbon with coagulants and polymers in treating municipal wastewater both in lab scale and in pilot plant scale was evidenced [166]. The MPs (suspended solids based) removal percentage was 84.40–91.30 % with activated carbon alone that increased to 88.90–93.60 % by addition of coagulant. The process became more viable by addition of polymer (flocculant). It was observed that as the size of flocs increased it made its separation easier. Combined sewer overflows (CSO) consist of overspill of untreated sewer wastewater during wet weather conditions. They act as reservoirs of different MPs. A study conducted by [167] consisted of coagulation, flocculation and sedimentation processes, followed by ozonation to enhance the primary treatment of CSO. The treated effluents exhibited insignificant levels of acute toxicity, enabling the feasibility of the combinations to treat such overflows with maximum removal of MPs. Geosmin and 2-methylisoborneol (2-MIB) are problematic algal MPs which are categorized as taste and odor compounds in drinking water treatment plants. Cyanobacteria and actinomycetes are the causative agents of such algal MPs. A researcher reported the effective application of powdered activated carbon (using ball mill) and permanganate oxidation combined with coagulation to control the taste and odor of MPs in conventional drinking water treatment plants [168]. The process results in simultaneous adsorption of MPs. Pilot scale trials in drinking water treatment plants revealed the suitability of the experiments with maximal adsorption. Coagulation-flocculation process also plays an inevitable role in the leather industries. One of the key wet-end processes in tanneries is dyeing which threatens the ecological equilibrium and increases chemical oxygen demand of tannery effluents. Another researcher group achieved significant dye removal in tannery wastewater, using coagulation-flocculation along with adsorption processes [169]. A novel activated carbon compound was prepared by pyrolysis (microwave-assisted) of an organic naive precursor extracted from the sludge of the tannery wastewater treatment plant and reached 85.2 % dye removal.

A study reported the use of a coagulation-flocculation process for the pre-treatment and removal of various micropollutants including surfactants and pharmaceuticals such as cationic surfactants, phenols and pharmaceuticals such as 5-methylbenzotriazole, ibuprofen, carbamazepine, and metoprolol [170]. This method involves the use of plant-based materials such as tannin-based product and use of tannin coagulant for nitrogen recovery using membrane contactor technology. Response Surface Methodology (RSM) was used for standardization of reaction conditions such as pH 7.3. This technology reported the removal of Iron (Fe) (92 %), Titanium (Ti) (91 %), Chromium (Cr) (68 %), Aluminium (Al) (52 %), Barium (Ba) (48 %) and Vanadium (V) (44 %). The employed combined technology also assisted the removal of TP, TOC and TN of 47 %, 15 %, and 4 % respectively and NH_3–N removal increased by 20 %. Another processes known as sequential coagulation and electro-oxidation (EO) processes were employed to treat landfill leachate and complex wastewater using Box-Behnken design of RSM [171]. This method involved the use of poly aluminium chloride (PAC) as a coagulant to reduce the high pollutant concentration before electro-oxidation process. In the electro-oxidation process multi-metal oxide doped Ti (MMO/Ti), Pt doped Ti (Pt/Ti), and graphite were used as anode and stainless steel was used as cathode. It was observed that method in which Pt/Ti anode was used under 1.25A current and pH 5, COD, UV_{254}, and CN removal efficiencies were observed to be 52.8, 68.1, and 85.6 % respectively.

4.9. Other methods

The complex and heterogeneous nature of micropollutants in landfill leachates calls for advanced and modified treatment processes that can effectively remove these pollutants with greater degree of reproducibility and sensitivity. Such methods are often combinations of conventional treatment methods or optimized processes derived from these traditional treatment processes. Nevertheless, each of these treatment strategies has their own advantages and limitations in removal of micropollutants from landfill leachates as summarized in Table 3.

Table 3
Comparison of treatment techniques used for micropollutant removal from landfill leachate.

Method	Advantages	Limitations	References
Activated sludge process	Compatible with any volume of leachate, Easy to implement and maintain	High energy and capital investment, Requires regular monitoring	[172]
Rotating biological contactors (RBC)	Low energy consumption, Relatively simple	High capital investment, Low performance efficiency	[173]
Anaerobic ammonium oxidation (Anammox)	Low cost technique	Long start up time	[174]
Biochar adsorption	Waste valorisation technique High efficiency in removing micropollutants	High power and energy investment for biochar formation	[175]
Persulfate oxidation	Broad range of pollutants can be targeted, Accelerated reaction speed	High cost of electrodes used, Short service life of electrode	[176]
Aerobic methane oxidation coupled to denitrification	Alternative solution for nitrogen removal of NO_3 and NO_2	High initiation and maintenance cost, Difficulty in process optimization	[177]
Membrane biofilm reactor	Most efficient in carrying out oxidation and reduction reaction for wide array of pollutants	Difficulty in controlling the growth of complex microbial community in biofilm	[178]
Sequential batch reactor	Flexibility in operation, High biomass retention, Resistance to shock loads	Automatic operation facilities, Excellent process control	[179]
Moving bed biofilm reactor	Can tolerate high amount of ammoniacal nitrogen content	High cost	[180]
Photo fenton process	Complete mineralization of organic contaminants in leachate	Precise optimization of process is difficult	[181]
Nanofiltration	Stability in operation, Efficient in removing dissolved organic matter	Limited life span of membranes, High maintenance cost	[182]
Phycoremediation	Eco Friendly method Relatively low cost	Sensitive to pH and temperature fluctuations	[183]
Electrocoagulation	Minimal operating cost, Improved efficiency of coagulation	High energy expenditure, Building up of toxic substances	[184]

5. Conclusion and future perspectives

The heterogenous nature of micropollutants present in the landfill leachates makes their removal and treatment a herculean task. Variability in the amount and nature of these micropollutants makes it difficult to define a single method for the treatment of these micropollutants. Conventional methods for removal of micropollutants although cost effective may not be sound enough to treat the emerging micropollutants due to the continuous change in waste ending up in landfills. Another major concern is to achieve a stable performance of the advanced technologies currently used for micropollutant removal with respect to varying season and geographical location. The influence of other physicochemical parameters in the vicinity of the landfills also needs careful consideration while designing advanced technique for micropollutant removal from landfill leachates. The issue related to sludge disposal and management should also be a focus area of research as sludge accumulation is often unavoidable during the leachate treatment. The success of any advanced treatment technology depends on its reproducibility, affordability, and performance durability. Development of integrated strategies connecting multiple existing technologies can open new avenues in dealing with the concern associated with heterogeneity of landfill leachates. Finally, the life cycle assessment of these treatment methods and their timely modification and improvement is also essential steps for the success of any micropollutant removal technologies.

Declaration of competing interest

It is declared that all the authors are mutually agreeing to submit this manuscript to the journal. It is also declared that co-authors do not hold any conflict of interest that may affect the publication of this manuscript.

Data availability

No data was used for the research described in the article.

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National Estimate of Per- and Polyfluoroalkyl Substance (PFAS) Release to U.S. Municipal Landfill Leachate

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Supporting Information

ABSTRACT: Landfills are the final stage in the life cycle of many products containing per- and polyfluoroalkyl substances (PFASs) and their presence has been reported in landfill leachate. The concentrations of 70 PFASs in 95 samples of leachate were measured in a survey of U.S. landfills of varying climates and waste ages. National release of PFASs was estimated by coupling measured concentrations for the 19 PFASs where more than 50% of samples had quantifiable concentrations, with climate-specific estimates of annual leachate volumes. For 2013, the total volume of leachate generated in the U.S. was estimated to be 61.1 million m^3, with 79% of this volume coming from landfills in wet climates (>75 cm/yr precipitation) that contain 47% of U.S. solid waste. The mass of measured PFASs from U.S. landfill leachate to wastewater treatment plants was estimated to be between 563 and 638 kg for 2013. In the majority of landfill leachate samples, 5:3 fluorotelomer carboxylic acid (FTCA) was dominant and variations in concentrations with waste age affected total estimated mass. There were six PFASs that demonstrated significantly higher concentrations in leachate from younger waste compared to older waste and six PFAS demonstrated significant variation with climate.



INTRODUCTION

Per- and polyfluoroalkyl substances (PFASs) are used in many consumer products including baking papers, microwave popcorn bags, carpet, upholstery, medical garments, food contact paper, nonstick cookware, dental floss, and outdoor clothing.[1−4] Many of these products are disposed in landfills at the end of their useful life and the presence of PFASs in landfill leachate is well documented, though the range of concentrations varies widely.[5−10] For example, the reported range of perfluorooctanoic acid (PFOA) concentrations in U.S. landfill leachate (n = 13) was 0.15−9.2 μg/L.[6,9] For Chinese landfill leachate, concentrations of PFOA have been reported to be as high as 214 μg/L.[5] The range of reported PFAS concentrations in landfill leachate is not surprising given the heterogeneity of municipal solid waste (MSW)[11] and the range of PFAS content on various products.[1−4,12]

The mass release of PFASs from landfills to wastewater treatment plants (WWTPs) is of interest as the U.S. EPA recently established advisory levels for perfluorooctanesulfonic acid (PFOS) and PFOA of 0.07 μg/L in drinking water.[13,14] WWTPs are not known to attenuate PFOA and PFOS, and previous studies have shown higher effluent PFOA and PFOS concentrations compared to the influent concentrations, with the transformation of precursor compounds during the biological treatment process as the likely source of the increase.[15−19] The mass of PFASs in collected leachate sent to WWTPs is a function of both leachate concentrations and leachate volume. Leachate volume will depend on the climate (i.e., rainfall rates, evapotranspiration) as precipitation is the major source of infiltration to landfills. Busch et al. (2010) documented ∼90 kg/yr release for 44 PFASs in treated leachate from all German landfills (∼1700 landfills).

Given the heterogeneity of waste disposed in landfills, there are many potential sources of variability in leachate PFAS concentrations. Concentrations could be influenced by infiltration volume (i.e., climate) as well as waste age and seasonal variability in infiltration. In addition, some landfills accept WWTP biosolids that have been reported to contain PFASs.[17] Previous studies on PFASs in leachate did not assess variations in concentrations based on climate.[5−10] The potential for leachate PFAS concentrations to change with time as concentrations of phased-out PFASs decrease (i.e., PFOA- and PFOS-based products) has not been evaluated. Benskin et al.

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(2012) demonstrated temporal PFAS concentration variations for a single landfill with one average waste age, with variations largely attributed to increases of C5, C6, C8, and C10 PFCAs and 8:2 fluorotelomer carboxylic acid (FTCA) during the spring months. However, no studies identified have evaluated the effect of climate or waste age on PFAS concentration.

The objective of this study was to characterize leachate PFAS concentrations in U.S. landfills operated in different climates and containing MSW of different ages, and to use concentration data with independent estimates of leachate volumes to estimate the mass of PFASs released from U.S. landfills. In addition, temporal variability was examined using samples collected from two landfills that were sampled five times over two-years.

MATERIALS AND METHODS

Site Selection and Descriptions. Prior to selecting a landfill for inclusion, landfill operators completed a questionnaire with information on waste sources and waste age, operating characteristics, and potential sample locations. Landfills were selected to include sites in different climates and containing refuse of different ages. Climate categories were adopted from the U.S. EPA which categorizes landfills based on annual precipitation: arid (<38 cm), temperate (38−75), and wet (>75).[18] The average waste age associated with a leachate sample (Table 1) was calculated from the mean of the date of initial waste placement and the sampling date (2013). This calculation of average waste age is imperfect because it assumes equal waste placement annually, but it allows for some analysis of the effect of waste age.

All participating landfills were publicly owned, the implication of which is that they were receiving primarily MSW and in some cases biosolids, but were less likely to accept a range of industrial wastes relative to privately owned facilities. Ultimately, 95 samples were collected from 18 landfills (Table 1), either directly from a valve after flushing, or using a polyethylene baler for leachate obtained from manholes and ponds. Most landfills were sampled twice, with approximately 6−16 months between samples, but two landfills (T and U) were sampled five times over a two-year period to examine temporal variability. Landfills H and N were only sampled once due to collection and shipping limitations.

The emphasis of the sampling strategy was to collect leachate as it leaves the landfill for offsite treatment at a WWTP. At some landfills, leachate was collected at additional locations to obtain samples from individual landfill cells to increase the size of the data set and/or to obtain waste-age specific leachate samples. To ensure that all landfills were weighted equally, all data from cells of the same waste age was averaged so that only one value is reported for each landfill at each time point.

Sample Collection and Storage. Landfill operators were provided with self-contained sampling kits with return shipping instructions. Sample collection took place from February 2013 to December 2014, with the majority of sample collection in 2013. Leachate was collected in a 1-L HDPE container and then poured into a 50 mL sample tube, sealed with parafilm, and frozen overnight onsite when possible. After freezing, samples were shipped on ice overnight to Oregon State University (OSU) for analysis. For Landfill O, the samples were shipped the same day as a freezer was not available.

For quality control, each sample kit included duplicate field and trip blanks (i.e., sampling containers with DI water only). Trip blanks remained sealed during sampling while field blanks were opened during sample collection to assess potential background contamination during sampling.

Table 1. Site Characteristics for U.S. landfills sampled

landfill[a]	number of sampling events	total samples collected	climate[b]	years of waste disposal	average waste age[c] (years)
K	2	10[d]	arid	1996−Present	9
L	2	2	arid	1970−Present	22
M	2	2	arid	1966−Present	24
E	2	2	temperate	1988−Present	13
F	2	4[e]	temperate	1993−2001	10.5
G	2	8[f]	temperate	1999−Present	7.5
T	5	10[g]	temperate	1975−Present	19.5
B	2	2	wet	1981−Present	16.5
H	2	2	wet	1998−Present	8
I	2	2	wet	1989−Present	12.5
J	2	2	wet	1996−Present	9
N	1	1[h]	wet	1998−Present	8
O	2	4[i]	wet	1970s−Present	>15
P	2	4[j]	wet	1970s−Present	>15
Q	2	2	wet	2001−Present	6.5
R	2	8[k]	wet	1977−Present	18.5
S	2	4[l]	wet	1997−Present	8.5
U	5	26[m]	wet	1980−Present	17
total		95			

[a]Landfills A, C, and D are not shown because these landfills were closed before 1980 and the data were not included in the national inventory. [b]"Arid" < ~38 cm of precipitation a year, "Wet" > ~75 cm a year, and "Temperate" precipitation between 38 and 75 cm a year as in reference23. [c]Average waste age was calculated as 2013 minus the initial waste placement date divided by two. [d]For Landfill K, the average concentration of the five cells at each time point were used in the national inventory model. [e]For Landfill F, the average concentration of the two cells at each time point were used in the national inventory model. [f]For Landfill G, the average concentration of the four cells at each time point were used in the national inventory model. [g]For Landfill T, only the four samples collected in March and October 2013 were used in the national inventory model. [h]For Landfill N, concentrations for the single sample were entered twice in the national inventory model. [i]For Landfill O, the average concentration of the two cells at each time point were used in the national inventory model. [j]For Landfill P, the average concentration of the two cells at each time point were used in the national inventory model. [k]For Landfill R, only the samples collected from the onsite lagoon were used in the national inventory model. [l]For Landfill S, the average concentration of the two cells at each time point were used in the national inventory model. [m]For Landfill U, only the samples collected from the onsite tank collected in February and October 2013 were used in the national inventory model.

Micro-Liquid−Liquid Extraction (Micro-LLE) and Liquid Chromatography Tandem Mass spectrometry

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(LC-MS/MS). Samples were analyzed for the aqueous concentrations of 70 PFASs comprising 14 compound classes (Supporting Information (SI) Table S1) using previously described methods for leachate analysis by LC/MS/MS.[6] Briefly, leachate samples were centrifuged, titrated to pH 7−8, and extracted with trifluoroethanol and ethyl acetate. Then, 900 μL of the extract was injected, using orthogonal column chemistries to separate classes of compounds and tandem mass spectrometry for individual compound detection and identification. Method detection limits were at low to subng/L levels. Analytes were divided into four tiers based on the availability of standards: quantitative (Qn), semiquantitative (Sq), screening (Sc), and qualitative (Ql). Quantitative (Qn) analytes (n = 29) had analytical standards, the measured accuracy fell within 90−110%, and precision was ≤20% RSD. Semiquantitative (Sq) analytes (n = 7) had analytical standards, but the measured accuracy did not fall within 90−110% and/or the precision was ≥20% RSD. Only a commercial reference material was available for qualitative (Ql) analytes (n = 16). No reference material was available for screening (Sc) analytes (n = 18) but whose composition fell within homologous series of compounds that differed only in chain length. For PFASs in the Sc and Ql categories, concentrations were estimated assuming equal molar response factors to structurally similar, quantifiable PFASs. Leachate samples were analyzed concurrently with samples presented in Allred et al. (2015) and Lang et al. (2016).

Quality Control. Concentrations in all 69 trip and field blanks were less than the limit of quantification (<LOQ) for all 70 PFASs measured, except for low level contamination in one field blank sample, from the first sampling event at Landfill Q (<3 ng/L for all compounds). The absence of quantifiable levels of PFASs in the blanks indicates that concentrations measured in the samples were attributable to the leachate and not background contamination.

Data Analysis of PFAS Concentrations. For the national inventory, the 70 PFASs were categorized into three groups based on the number of samples with concentrations above the LOQ. This grouping was implemented because a large number of PFAS concentrations were < LOQ. Individual PFASs were assigned to group 1 if over 50% of all samples were > LOQ. Mean concentrations were calculated for group 1 PFASs, as a majority of concentrations were at quantifiable levels. PFASs were assigned to groups 2 or 3 if the fraction of samples > LOQ was 20−50% or <20%, respectively. To evaluate the contribution of group 2 PFASs, maximum likelihood concentrations were calculated as outlined in Gibbons and Coleman (2001), with the equation in the SI. Group 3 compounds are presented graphically in the SI, but a mean was not calculated.

Leachate PFAS concentrations were analyzed to evaluate whether there were significant differences attributable to climate or average waste age (greater than or less than 10 years). For waste age evaluation, an average age of 10 years was selected because this resulted in two sample populations with similar numbers of samples in each group. To estimate national release, a separate mean concentration was used for each climate and waste age category for each PFAS.

Estimate of Mass Flow of PFASs in U.S. Landfill Leachate. eq 1 was used to estimate the mass release of individual PFASs in leachate from U.S. landfills in wet climates. This equation was repeated for landfills in temperate and arid climates, with the sum of the three climates equal to the total mass release for an individual PFAS in U.S. landfill leachate.

$$M_{p,\mathrm{wet}} = (\bar{x}_{\mathrm{p},<10,\mathrm{wet}} \times \mathrm{WIP}_{<10,\mathrm{wet}} + \bar{x}_{\mathrm{p},>10,\mathrm{wet}} \times \mathrm{WIP}_{>10,\mathrm{wet}}) \times \mathrm{LG}_{\mathrm{wet}}/\mathrm{SA} \quad (1)$$

where, $M_{\mathrm{p,wet}}$ is the mass release of PFAS p from landfills in wet climates (kg/yr), $\bar{x}_{\mathrm{p,a,wet}}$ is the average concentration of PFAS p for the set landfills with average waste age a, (i.e., >10 years or <10 years) from landfills in wet climates, $\mathrm{WIP}_{\mathrm{a,wet}}$ is the total mass of waste contained in U.S. landfills in Tg (1 Tg = 10^6 kg) from landfills in wet climates with waste age a, LG_c is the leachate generation rate for landfills in wet climates (m^3/ha-day), and SA is average mass of waste per surface area for U.S. landfills (Tg/ha). PFAS mass release was estimated for 2013, which was the year when most leachate samples were collected. The leachate generation rate was derived for three climate categories, as leachate generation varies with precipitation. SA was calculated for all U.S. landfills as it was not expected to vary with climate.

The probable range for the national release of total PFASs calculated here was estimated using a Monte Carlo analysis, which reflects uncertainties in PFAS concentrations and in predictors of leachate volume (LG_c and SA). For the Monte Carlo analysis, the total mass release of group 1 PFASs in U.S. landfill leachate was estimated as the sum of eq 2, repeated for landfills in wet, temperate, and arid climates. Group 1 PFASs defined as screening (Sc) or qualitative (Ql) in the LC-MS/MS method were not included in eqs 2 and 3. Group 2 PFASs were not included in the total release, as their average concentrations are only maximum likelihood estimates.

$$M_{\mathrm{T,wet}} = (\bar{x}_{\mathrm{T},<10,\mathrm{wet}} \times \mathrm{WIP}_{<10,\mathrm{wet}} + \bar{x}_{\mathrm{T},>10,\mathrm{wet}} \times \mathrm{WIP}_{>10,\mathrm{wet}}) \times \mathrm{LG}_{\mathrm{wet}}/\mathrm{SA} \quad (2)$$

where, $M_{\mathrm{T,wet}}$ is the total PFAS mass released to WWTP from landfills in wet climates, and $\bar{x}_{T,a,wet}$ is the average of the total measured group 1 PFAS concentrations for the set of n landfills in wet climates and each waste age (eq 3).

There are two methods that can be used to calculate the total, average PFAS concentration for a given climate and waste age category ($\bar{x}_T$). One alternative is the average of the total PFAS concentration from each landfill, which is not the same as the sum of the average concentration for each PFAS. To calculate M_T in eq 2, the sum of the group 1 PFAS concentrations from each landfill was used as in eq 3. This is because the total measured PFASs at each landfill was judged to be a more accurate estimate of total release than summing the average concentrations of individual PFASs.

$$\bar{x}_{\mathrm{T,a,c}} = \frac{\sum_l \sum_{\mathrm{p}} x_{l,\mathrm{p,a,c}}}{n_{\mathrm{a,c}}} \quad (3)$$

where, $\bar{x}_{\mathrm{T,a,c}}$ is the average of the sum of the measured group 1 PFAS concentrations for the set of n landfills in climate c and waste age a, $\sum_l \sum_p x_{\mathrm{l,p,a,c}}$ is the sum of the total group 1 PFAS concentrations from landfill n in climate c with waste age a, and $n_{\mathrm{a,c}}$ is the number of landfill samples analyzed in climate c with waste age a.

The Monte Carlo analysis was conducted with 10 variables including the total PFAS concentration for waste that is less than and greater than 10 years in age in each of three climates (6), the leachate generation rate in each climate (3) and the waste mass per surface area (1). To ensure model convergence,

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Table 2. Per- and Polyfluoroalkyl Substances Grouped Based on the Percent of Total Samples Analyzed above The LOQ[a]

Percent of Samples > LOQ	
>50%	PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFDA, 6:2 FTCA, 8:2 FTCA, 3:3 FTCA[c], 5:3 FTCA, 7:3 FTCA, PFBS, PFPS[c], PFHxS, PFOS, 6:2 FTSA, 8:2 FTSA[d], MeFBSAA[d], MeFPeSAA[e], MeFHxSAA[e], MeFHpSAA[e], MeFOSAA, EtFBSAA[e], EtFPeSAA[e], EtFHxSAA[e], EtFOSAA
20−50%	10:2 FTCA, 6:2 FTUCA, 8:2 FTUCA, 9:3 FTCA[e], PFHpS[c], 4:2 FTSA[d], FBSAA[e], FPeSAA[e], FHxSAA[e], FHpSAA[e], EtFHpSAA[e]
<20%	PFUnDA, PFDoDA, PFTriDA, PFTeDA[d], PFPeDA[c], PFHxDA[d], PFHpDA[c], PFOcDA[d], 4:2 FTCA[e], 4:2 FTUCA[e], 10:2 FTUCA[e], PFNS[c], PFDS, FOSAA[d], 4:4 PFPIA[e], 4:6 PFPIA[c], 6:6 PFPIA, 6:8 PFPIA, 8:8 PFPIA, 4:4 diPAP[c], 4:6 diPAP[e], 6:6 diPAP, 6:8 diPAP[c], 8:8 diPAP, 8:10 diPAP[c], 10:10 diPAP[c], 6:2 FTMAP[c], 6:2/8:2 FTMAP[c], 8:2 FTMAP[c], 8:2/10:2 FTMAP[c], 10:2 FTMAP[c], 8:8 SAmPAP[c]

[a]Full compound names and their abbreviations are listed in Table S1. [b]PFASs defined in Allred et al. (2014) as quantitative based on standard availability unless otherwise specified (n = 29). [c]PFASs defined in Allred et al. (2014) as qualitative based on standard availability and therefore not included in the national release model (n = 16). [d]PFASs defined in Allred et al. (2014) as semiquantitative based on standard availability (n = 7). [e]PFASs defined in Allred et al. (2014) as screening based on standard availability and therefore not included in the national release model (n = 18).

Table 3. Mean Concentrations (μg/L) for Group 1 PFASs Used in the Mass Release Model[a,b]

	wet		temperate		arid	
μg/L	<10 yrs. (n = 14)	>10 yrs. (n = 12)	<10 yrs. (n = 2)	>10 yrs. (n = 6)	<10 yrs. (n = 2)	>10 yrs. (n = 4)
PFBA	0.9 ± 1	1 ± 0.9	0.3 ± 0.01	1 ± 0.9	1 ± 0.3	0.3 ± 0.2
PFPeA	1 ± 1	0.8 ± 0.8	0.2 ± 0.01	0.9 ± 0.9	0.9 ± 0.04	0.3 ± 0.1
PFHxA	2 ± 2	1.8 ± 1.9	0.5 ± 0.1	2 ± 2	3 ± 0.1	0.6 ± 0.3
PFHpA	0.8 ± 0.8	0.6 ± 0.6	0.2 ± 0.04	0.6 ± 0.4	1 ± 0.1	0.1 ± 0.1
PFOA	1 ± 2	1 ± 1	0.2 ± 0.001	0.7 ± 0.6	1 ± 0.05	0.1 ± 0.04
PFNA	0.1 ± 0.1	0.1 ± 0.	0.01 ± 0.002	0.02 ± 0.01	0.1 ± 0.002	0.005 ± 0.003
PFDA	0.1 ± 0.1	0.05 ± 0.1	0.005 ± 0.003	0.01 ± 0.01	0.02 ± 0.001	0.003 ± 0.004
6:2 FTCA	0.9 ± 0.5	0.6 ± 0.7	1 ± 0.5	0.5 ± 0.4	3 ± 1	0.1 ± 0.1
8:2 FTCA	0.2 ± 0.2	0.1 ± 0.2	0.1 ± 0.03	0.1 ± 0.1	0.4 ± 0.01	0.01 ± 0.01
5:3 FTCA	4 ± 6	3 ± 3	4 ± 0.9	4 ± 3	15 ± 1	0.4 ± 0.4
7:3 FTCA	0.9 ± 1	0.4 ± 0.6	0.1 ± 0.02	0.2 ± 0.2	0.9 ± 0.4	0.03 ± 0.03
PFBS	0.5 ± 0.8	0.3 ± 0.5	0.02 ± 0.005	0.03 ± 0.02	0.3 ± 0.04	0.01 ± 0.01
PFHxS	0.3 ± 0.3	0.4 ± 0.2	0.1 ± 0.01	0.3 ± 0.2	0.5 ± 0.005	0.02 ± 0.01
PFOS	0.1 ± 0.2	0.1 ± 0.1	0.01 ± 0.001	0.1 ± 0.1	0.2 ± 0.05	0.003 ± 0.003
6:2 FTSA	0.2 ± 0.1	0.3 ± 0.6	0.03 ± 0.003	0.1 ± 0.1	0.3 ± 0.01	0.005 ± 0.009
8:2 FTSA	0.1 ± 0.	0.03 ± 0.05	0.001 ± 0.0002	0.03 ± 0.1	0.1 ± 0.05	0.0003 ± 0.0005
MeFBSAA	0.6 ± 0.6	0.4 ± 0.6	0.4 ± 0.1	0.2 ± 0.2	0.8 ± 0.01	0.04 ± 0.06
MeFOSAA	0.2 ± 0.3	0.2 ± 0.2	0.005 ± 0.003	0.1 ± 0.2	0.04 ± 0.01	0.001 ± 0.002
EtFOSAA	0.1 ± 0.2	0.1 ± 0.2	0.0003 ± 0.0004	0.1 ± 0.3	0.04 ± 0.01	0.001 ± 0.001
total	15 ± 16	11 ± 12	7 ± 1	11 ± 9	29 ± 1	2 ± 0.8

[a]One standard deviation is presented for each mean concentration. [b]Climates: "arid" less than ~38 cm of precipitation a year, "wet" above ~75 cm a year, and "temperate" annual precipitation between 38 and 75 cm a year.

10 000 realizations were conducted on the total PFAS mass flow (eq 2).

RESULTS AND DISCUSSION

The results for group 1 PFASs are presented first, as these were the only data used quantitatively for the national inventory. Results for group 1 PFASs are followed by the results for group 2 and 3 PFASs. As presented below, group 2 PFASs were less <1% of the total estimated PFAS release to landfill leachate.

PFASs Present above the Detection Limit in >50% of Samples (Group 1). The PFASs assigned to group 1 (i.e., >50% of samples exhibited quantifiable concentrations) belong to the PFCAs, FTCAs, PFSAs, fluorotelomer sulfonic acid (FTSAs), n-methyl fluoroalkyl sulfonamido acetic acids (MeFASAAs) and n-ethyl fluoroalkyl sulfonamido acetic acid (EtFASAAs) compound classes (Table 2). The widespread presence of C4−C10 PFCAs and C4, C6, and C8 PFSAs in the sampled leachate is consistent with previous observations of these PFASs in leachate[5−10] and as residuals on PFAS-treated products.[1−4,12] The presence of FTSAs, FTCAs, EtFASAA, and MeFASAAs in leachate is less well-documented. With the exception of FTSAs, these polyfluorinated substances are described as "intermediate environmental transformation products," as opposed to surfactants used on products.[12]

The mean concentrations for group 1 PFASs are presented in Table 3. The standard deviations are often greater than the mean and reflect considerable variability in each sample population (Table 3). Given the heterogeneity of landfills and the numerous factors that influence PFAS release to leachate (e.g., sorption, preferential flow paths of leachate through waste, uneven distribution of PFAS-containing products in landfills), this is not surprising.

Of the 19 group 1 compounds quantified, there were four for which all concentrations were > LOQ (i.e., C6−C8 PFCAs and PFHxS). For the remaining 15 compounds, the mean concentration across all landfill samples was essentially the same whether a value of the LOQ, 50% of the LOQ or 0 was used to calculate the mean (SI Table S4). The low effect of the substituted LOQ value on the mean concentrations is due to the small percentage of group 1 samples with a LOQ result and the low LOQs compared to measured concentrations. To

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Environ. Sci. Technol. 2017, 51, 2197−2205

calculate a mean concentration for cases in which samples were < LOQ, the concentration was assumed to be 50% of the LOQ.

The mean concentrations of group 1 PFCAs and FTSAs, PFBS, and PFHxS reported here are comparable to previously published concentrations for untreated landfill leachate.[15−19] The major difference in the PFAS concentrations reported here is the addition of 5:3 FTCA. This compound was not measured in the majority of previous landfill leachate studies, but the data in Table 3 shows that the mean 5:3 FTCA concentration is at least three times higher than PFOA and PFOS combined for leachate from all climates. While mean MeFOSAA and EtFOSAA concentrations reported here are less than previously reported,[15−19] the mean concentration of MeFBSAA reported here was higher. These changes in FASAA concentrations correspond to the switch to shorter chain compounds.

PFAS Concentration Variations with Waste Age. Analysis of variations in PFAS concentrations as a function of average waste age was examined for the 19 group 1 PFASs defined as quantitative or semiquantitative in the LC/MS/MS method (Table 2). For this analysis, data from all climates was combined and an unpaired two-sample Student's t test was performed on the log-transformed PFAS concentrations as a function of average waste age. There were only six PFASs that demonstrated significantly different means ($P < 0.05$) for older waste compared to younger waste (PFNA, 8:2 FTCA, 5:3 FTCA, PFBS, MeFBSAA, MeFOSAA, SI Table S5), with higher concentrations in leachate from younger waste in all cases. This could be due to decreases in concentrations with time or to changes in the types of PFASs used on products. For example, manufacture of PFBS and PFNA based products increased as alternatives to PFOS and PFOA products, and both PFBS and PFNA were higher in younger leachate.[12] Similar concentrations for old and new waste indicates that release from a given waste mass will continue for many years following placement.

PFAS Concentration Variations with Climate. When log-transformed group 1 PFAS concentrations were compared using analysis of variance (ANOVA), the means of six compounds were found to be statistically different based on climate (PFNA, PFDA, PFBS, PFOS, 6:2 FTSA, and MeFOSAA - SI Table S6). For all PFASs that differed by landfill climate, the largest concentrations were measured in leachate from wet climates, suggesting that leaching governed release for these compounds. Mean 5:3 FTCA concentrations for landfill leachate from wet, temperate, and arid climates were 2.1 ± 3.0, 4.1 ± 5.0, and 5.2 ± 7.6 μg/L, respectively (SI Table S6). While the mean 5:3 FTCA concentration was highest for the arid climate leachate, the large standard deviations masked any potential effect of climate on PFAS concentration.

Mass Disposed and Leachate Volume Estimates for U.S. Landfills in 2013. For estimates of the total mass of waste in U.S. landfills, 1980 was selected as the base year given the assumption that waste disposed after 1980 was managed in engineered landfills, which include leachate collection systems, though some landfills were operated without liners through the early 1990s.[20] Previous estimates for annual waste disposal in U.S. landfills demonstrate considerable variability, with annual disposal estimates varying from 150 to 284 Tg in 2013 (SI Figure S1).[11,24−26] The Powell et al. (2015) estimate for annual disposal was adopted here as it is the most recent and is the only one that utilizes certified reports from landfill operators for the majority of the data. For the years prior to 1990, the Powell et al. (2015) estimate was decreased at the rates described by the U.S. EPA.[11] For years after 1990, the Powell et al. (2015) estimate was assumed to be linear between the years where the mass was provided. The fraction of waste generated in each climate region was adopted from published estimates of disposal rates for each region (SI Figure S2).

For landfills in wet climates, leachate generation rates (SI Figure S3) were estimated using data from annual landfill operating reports to state agencies (n = 27), published data (n = 12), data provided by some of the landfills that participated in this study (n = 7) and data obtained from a consulting engineer (n = 6).[27,28] The wide distribution of leachate generation rates is likely due to variability in landfill geometry and operating conditions, which are known to affect infiltration into a landfill. Leachate generation rates for temperate and arid climates are summarized in SI Table S2.

The mass of waste contained per landfill surface area was estimated using landfill operating reports to state agencies (n = 18)[28] and data obtained from a consulting engineer (n = 95 - SI Table S3). The mass of waste contained per surface area was assumed to be constant across climates based on standard landfill operating practice. As such, the data were aggregated (SI Figure S4).

For 2013, the estimated total volume of leachate generated in the U.S. was 61.1 million m^3/yr (Table 4). Landfills in wet climates generated the majority of leachate in the U.S. (79%), even though only 47% of the waste is generated in this climactic region (Table 4). It is recognized that this estimate of leachate volume does not consider leachate that is recirculated to the waste and thus not sent off-site for treatment. Estimates of total leachate recirculation were not available but judged to be on the order of 10−20% of leachate generation in wet climates.

Table 4. Mean Values for Leachate Generation and the Mass of Waste Per Surface Area, Total Waste in Place for United States Landfills, And the Estimated Volume of Leachate Generated in 2013

	leachate generation (m^3/ha-day)[a]	mass per surface area (Tg/ha)[a]	total waste in place[b] (Tg)	volume of leachate for 2013 (million m^3/yr)
arid	0.04 ± 0.03	0.14 ± 0.098	1492	0.10
temperate	2 ± 3	0.14 ± 0.098	2602	13
wet	5 ± 5	0.14 ± 0.098	3648	48
total				61.1

[a]One standard deviation is presented for leachate generation and mass of waste per surface area. [b]Waste disposed from 1980−2013.

Annual PFAS Mass Release in U.S. Landfill Leachate. The national inventory of PFAS mass release based on eq 1 is given in Figure 1. The majority of estimated PFAS mass release was attributable to PFCAs (291 kg/yr) and FTCAs (285 kg/yr), with lower releases of PFSAs and their precursors (84 kg/yr). Both 5:3 FTCA and 6:2 FTCA were previously identified products of anaerobic 6:2 fluorotelomer alcohol (FTOH) degradation.[29] Whereas Buck et al. (2011) lists FTCAs as biotransformation intermediates, a recent publication demonstrates that FTCAs were present on Japanese consumer products.[30] PFAS release from arid climates was <1% of the total mass release. Although mean PFAS concentrations were statistically similar in different climates, the low leachate volume in arid climates resulted in low mass release (Figure 1). Total release of the seven group 1 PFASs defined as qualitative and screening was estimated to be ∼56 kg PFAS/yr, but these

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Environ. Sci. Technol. 2017, 51, 2197−2205

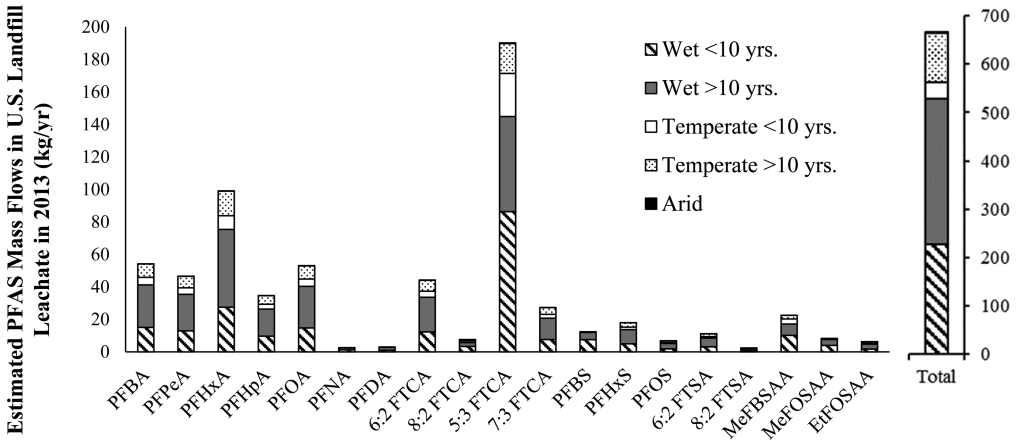


Figure 1. Group 1 PFAS release in U.S landfill leachate for 2013 demonstrating a dominance of compounds with five fluorinated carbons (PFHxA and 5:3 FTCA). Releases were calculated from mean concentrations in each climate and age category (Table 3). The individual columns are based on eq 1 while the total is based on eq 2.

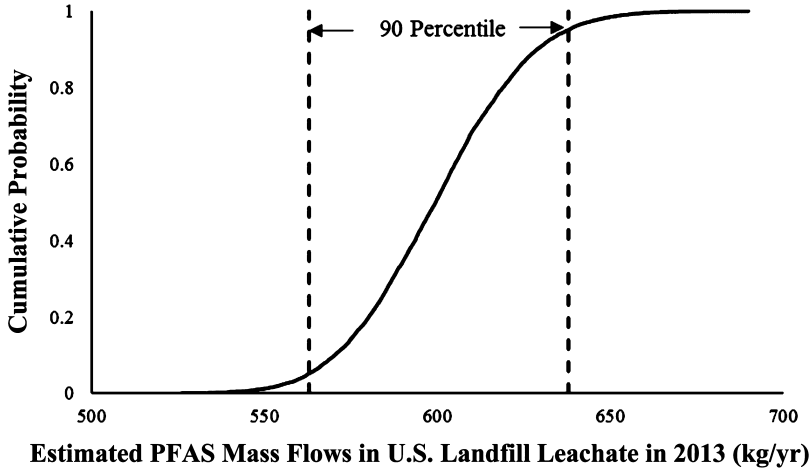


Figure 2. Distribution for total PFAS release in 2013 created using a Monte Carlo analysis with eq 2 and the distributions in SI Table S7.

compounds were not included in Figure 1 or the national release model as concentrations were not quantitative.

A Monto Carlo analysis was conducted to evaluate the uncertainty associated with the point estimate developed using eq 1. Model parameter distributions are described in SI Table S7 and Figure S7. The 90-percentile range for total group 1 PFASs of 563 to 638 kg/yr (~600 kg/yr with one significant figure - Figure 2). Since total oxidable precursor assays or total fluorine analyses were not completed in the current study, the actual inventory of PFASs in U.S. landfill leachate is likely higher than presented here, due to the presence of unmeasured PFASs.

PFASs Present above the Detection Limit in 20−50% of Samples (Group 2). Eleven PFASs were placed in group 2 (Table 2). The two classes of PFASs with the largest representation in group 2 were FASAAs and long chain FTCAs, but one PFSA, one unsaturated fluorotelomer carboxylic acid (FTUCA) and one FTSA were also present. The presence of a compound in group 2 indicates that the PFAS could be present in landfill leachate, but its presence is not widespread. Maximum likelihood estimates for group 2 PFASs that were quantifiable (10:2 FTCA, 8:2 FTUCA, and 4:2 FTSA, Table 5) are low relative to concentrations for group 1 PFASs (Table 3). The inclusion of group 2 PFASs added only ~1 kg/yr to PFAS mass flows (Figure 1).

Table 5. Concentration Ranges and Maximum Likelihood Estimates for Group 2 PFASs (20−50% of the Results Were > LOQ)

μg/L	concentration ranges	maximum likelihood estimate[a]
10:2 FTCA	ND−0.3	0.004
8:2 FTUCA	ND−0.2	0.005
4:2 FTSA	ND−0.02	0.001

[a]Maximum likelihood estimate calculated using methods outlined in Gibbons & Coleman (2001) using all samples (n = 95).

Long chain FTCAs are less likely to be present in leachate compared to their more soluble, shorter chain homologues, which were present in >50% of the leachate samples (group 1 − Table 2). FASAAs were present in a lower number of samples (group 2) compared to MeFASAAs and EtFASAA,s which were present in the majority of samples (group 1, Table 3). This is consistent with the lower production and use of FASAAs relative to the production of N-MeFASAAs and N-EtFASAAs derivatives.[31]

PFASs below the Detection Limit in the Majority of Samples (Group 3). There were 32 PFASs defined as group 3 PFASs because they were present in less than 20% of leachate samples (Table 2). Data for group 3 PFASs are only presented graphically (SI Figure S5) because the fraction of results that

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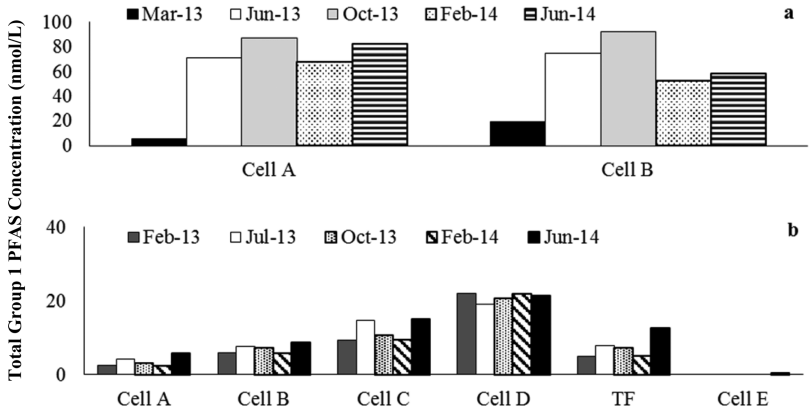


Figure 3. Total group 1 PFAS concentrations measured at (a) Landfill T and (b) Landfill U. The maximum whole method variability for the compounds in group 1 PFASs was 20%. For Landfill T, leachate was collected from two distinct areas with waste placement dates of 1975−1990 (Cell A) and 1990-present (Cell B). For Landfill U, Cell A contained the oldest waste (1980−1988) with each subsequent cell containing younger waste (Cell B: 1988−1993, Cell C: 1993−1998, Cell D: 1999−2014, and Cell E: 2014-present). Samples were also collected from a tank farm (TF), which collects and stores all leachate from Cells A−E prior to discharge for wastewater treatment. The leachate flows for Landfill U during the sampling period are presented in SI Figure S6.

were quantifiable was too low for a mean to be calculated. Long chain PFCAs (>C11), diPAPs, and PFPis were present at quantifiable levels in <20% of landfill leachate samples. One explanation for this is the sorption of these PFASs to several materials known to be present in MSW.[32] Since diPAPs were previously identified on food contact paper,[2] a product that is typically landfilled in the U.S., their absence in the majority of landfill leachate does not imply that they are not entering landfills, but rather that they are not being released to the aqueous phase. Allred et al. (2015) reported that diPAPs were present in the leachate in anaerobic laboratory scale landfill reactors filled with a single MSW sample, but concentrations were depleted by day 200 in both live and abiotic reactors. The PFPis are less likely to be present in MSW initially, as their primary use is pesticides.[12]

Temporal PFAS Concentration Variations. Five samples were collected over 17 months at Landfills T and U to evaluate temporal variability. The variability illustrated in Figure 3 is useful when considering the degree to which PFAS concentrations are representative of a landfill. At both Landfills T and U, variations appeared to be consistent with time across the landfill cells. For example, the March 2013 samples had the lowest group 1 PFAS concentrations in both Cells A and B, although the cells were physically separate, and Cell A had not accepted waste since 1990, whereas Cell B was accepting waste when sampled (Figure 3). Similar variations in PFAS concentrations in separate cells suggest that climate (i.e., rainfall) as opposed to internal processes in the buried waste was responsible for variations in concentrations (Figure 3). This contrasts the conclusion presented above that most PFASs concentrations were not statistically different based on the climate of the landfill sampled (SI Table S6). Benskin et al. (2012) also noted temporal changes for one landfill with increases over time in most of the PFASs analyzed as part of that study. The coefficients of variation (CV = stnd. dev./mean) for total PFAS concentrations in leachate from Cells A and B at Landfill T and Cells A to E at Landfill U were 53, 46, 37, 13, 33, 30 and 27%, respectively. In general, these values are lower than the variability in leachate samples between landfills (Table 3).

IMPLICATIONS AND LIMITATIONS

The estimate of total PFAS mass release in landfill leachate developed here (~600 kg/yr) represents the mass in collected leachate that was sent to wastewater treatment in 2013. Since 5:3 FTCA was the largest contributor to overall PFAS release in leachate, future work is needed to identify FTCA precursors in landfills and to estimate the attenuation of FTCAs during wastewater treatment. This study demonstrates that PFAS release from waste is slow relative to the quantities manufactured and applied to consumer products annually.[12] There are many reasons for the slow release including sorption, low PFAS solubility, bonding to polymers that are at most slowly degradable, and relatively low infiltration rates and subsequent waste flushing. However, the waste buried today will continue to release PFASs at similar rates for many years following placement (>10 yrs.). The slow release agrees with previous reports of PFASs release from MSW and MSW components (i.e., carpet and clothing).[21,22]

There are several limitations associated with the ~600 kg/yr estimate, and it is important to recognize other sources of PFASs that are attributable to landfills. First, there were 1540 landfills in the U.S. in 2013 and the estimated release is based on PFAS concentrations from 18 sites in different regions of the U.S.[33] Mass release is a function of both concentration and leachate volume. While leachate volume was estimated for the entire U.S. population of landfills, the extent to which the sampled landfills resulted in representative average PFAS concentrations is unknown, though variability in concentration data was incorporated in the Monte Carlo analysis.

The estimated release is based on ionic, nonvolatile PFASs quantified by LC-MS/MS and does not include volatile gas-phase PFASs typically measured by gas chromatography/mass spectrometry (GC/MS) (i.e FTOHs). Washington et al. (2014) reported that the degradation of fluorotelomer based polymers produced mostly volatile PFASs measured with GC-MS (>10 fold) compared to the nonvolatile PFASs measured by LC-MS/MS. An estimate of total PFAS release from landfills that includes volatile PFASs, would likely be higher than the estimates for leachate release of PFASs measured with LC-MS/MS alone, as presented here. Inclusion of additional new PFASs

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that can be measured with LC-MS/MS would further increase the estimate of release presented here.[34,35]

In the U.S., some MSW landfills likely receive waste from PFAS manufacturing and utilization facilities. This would include landfills near facilities that manufacture various PFASs and send their nonhazardous waste to MSW landfills, as well as facilities that utilize PFASs on their products and dispose of scrap material to an MSW landfill (e.g., carpet, textiles). For example, the total concentration of 12 PFASs in a Minnesota landfill that received wastewater treatment sludge from a 3 M manufacturing facility was 136 μ/L, as compared to the totals presented in Table 3 of 2−29 μ/L.[36] No landfill sampled in this study disclosed acceptance of waste from PFAS, carpet, or textile manufacturing.

An additional source of PFAS release to the environment is associated with unlined landfills. In 1988, there were an estimated 6000 landfills in the U.S., many of which did not have leachate collection systems such that leachate was released to the subsurface.[37] While some of these landfills later received low permeability covers that would reduce leachate flows, unlined landfills likely represent a significant source of PFASs to groundwater. Finally, there are other persistent organic pollutants in landfill leachate that may not be fully attenuated during leachate treatment and thus represent an additional source or surface water contamination.[38]

■ ASSOCIATED CONTENT

Supporting Information

The Supporting Information is available free of charge on the ACS Publications website at DOI: 10.1021/acs.est.6b05005.

Additional details regarding the PFASs analyzed and their acronyms, data for evaluation of changes in group 1 PFAS means with varying values substituted for a < LOQ result, group 1 PFAS concentration variations with climate and waste age, PFAS concentration correlations, and data used to estimate the annual leachate volume generated in the U.S. (PDF)

All PFAS concentration data for each landfill, the LOQ for each compound and the release for each compound (XLSX)

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Notes

The authors declare no competing financial interest.

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Review

A review of the occurrence, transformation, and removal of poly- and perfluoroalkyl substances (PFAS) in wastewater treatment plants

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Ultra-short chain PFAS
Precursors
Biodegradation
Adsorption

ABSTRACT

Poly- and perfluoroalkyl substances (PFAS) comprise more than 4,000 anthropogenically manufactured compounds with widescale consumer and industrial applications. This critical review compiles the latest information on the worldwide distribution of PFAS and evaluates their fate in wastewater treatment plants (WWTPs). A large proportion (>30%) of monitoring studies in WWTPs were conducted in China, followed by Europe (30%) and North America (16%), whereas information is generally lacking for other parts of the world, including most of the developing countries. Short and long-chain perfluoroalkyl acids (PFAAs) were widely detected in both the influents (up to 1,000 ng/L) and effluents (15 to >1,500 ng/L) of WWTPs. To date, limited data is available regarding levels of PFAS precursors and ultra-short chain PFAS in WWTPs. Most WWTPs exhibited low removal efficiencies for PFAS, and many studies reported an increase in the levels of PFAAs after wastewater treatment. The analysis of the fate of various classes of PFAS at different wastewater treatment stages (aerobic and/aerobic biodegradation, photodegradation, and chemical degradation) revealed biodegradation as the primary mechanism responsible for the transformation of PFAS precursors to PFAAs in WWTPs. Remediation studies at full scale and laboratory scale suggest advanced processes such as adsorption using ion exchange resins, electrochemical degradation, and nanofiltration are more effective in removing PFAS (~95–100%) than conventional processes. However, the applicability of such treatments for real-world WWTPs faces significant challenges due to the scaling-up requirements, mass-transfer limitations, and management of treatment by-products and wastes. Combining more than one technique for effective removal of PFAS, while addressing limitations of the individual treatments, could be beneficial. Considering environmental concentrations of PFAS, cost-effectiveness, and ease of operation, nanofiltration followed by adsorption using wood-derived biochar and/or activated carbons could be a viable option if introduced to conventional treatment systems. However, the large-scale applicability of the same needs to be further verified.

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Abbreviations

6:2 Cl-PFAES/ F-53B	6:2 chlorinated polyfluoroalkyl ether sulfonate	$Cl\text{-}C_6F_{12}\ OCF_2CF_2SO_3^-$
ADONA	Dodecafluoro-3H-4,8-dioxanonanoat	$C_7H_{12}O_4HNH_4$
di-SAmPAP	Perfluorooctane sulfonamidoethanol-based phosphate diester	–
EtFOSAA	Ethyl perfluorooctane sulfonamidoacetic acid	$C_8F_{17}SO_2N(C_2H_5)CH_2CO_2H$
FOSA	Perfluorooctane sulfonamide	$C_8F_{17}SO_2NH_2$
FOSAA	Perfluorooctane sulfonamidoacetic acid	$C_nF_{2n+1}\text{-}\ SO_2NCH_2CO_2H$
FOSE	Perfluorooctane sulfonamidoethanol	$C_nF_{2n+1}.SO_2NHCH_2CH_2OH$
FTCA	Fluorotelomer carboxylic acid	$C_nF_{2n+1}\text{-}CH_2COOH$
FTS	Fluorotelomer sulfonate	$C_nF_{2n+1}\text{-}CH_2CH_2SO_3^-$

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FTSA	Fluorotelomer sulfonic acid	C_nF_{2n+1}-$CH_2CH_2SO_3H$
FTUCA	Fluorotelomer unsaturated carboxylic acid	C_nF_{2n+1}-CF=CHCOOH
GenX/ HFPO-DA	Heptafluoropropylene-oxide-dimer-acid	$C_4HF_7O_3$
MeFOSAA	Methyl perfluorooctane sulfonamidoacetic acid	$C_8F_{17}SO_2N(CH_3)CH_2CO_2H$
n:2 FTOH	n:2 Fluorotelomer alcohol	C_nF_{2n+1}-CH_2CH_2OH
n:3 Acids	n:3 Saturated acids	C_nF_{2n+1}-CH2CH2COOH
N-EtFOSA	N-ethyl perfluorooctane sulfonamide	$C_8F_{17}SO_2N(C_2H_5)H$
NMeFOSA	N-methyl perfluorooctane sulfonamide	$C_8F_{17}SO_2N(CH_3)H$
PBSF	Perfluorobutane sulfonyl fluoride	$C_4F_9SO_2F$
PFAA	Perfluoroalkyl acids	–
PFBA	Perfluorobutyric acid	$C_4HF_7O_2$
PFBS	Perfluorobutanesulfonate/ sulfonic acid	$C_4F_9O_3$-SO_3^-/SO_3H
PFCA	Perfluorocarboxylic acid	C_nF_{2n+1}-COOH
PFDA	Perfluorodecanoic acid	$C_{10}HF_{19}O_2$
PFDPA	Perfluorodecylphosphonic acid	$C_{10}H_2F_{21}O_3P$
PFDS	Perfluorodecane sulfonate/sulfonic acid	$C_{10}F_{21}O_3$- SO_3^-/SO_3H
PFHpA	Perfluoroheptanoic acid	$C_7HF_{13}O_2$
PFHpS	Perfluoroheptanesulfonate/ sulfonic acid	$C_7F_{15}O_3$- SO_3^-/SO_3H
PFHxA	Perfluorohexanoic acid	$C_6HF_{11}O_2$
PFHxPA	Perfluorohexanephosphonic acid	$C_6H_2F_{13}O_3P$
PFHxS	Perfluorohexanesulfonate/ sulfonic acid	$C_6F_{13}O_3$- SO_3^-/SO_3H
PFNA	Perfluorononanoic acid	$C_9HF_{17}O_2$
PFOA	Perfluorooctanoic acid	$C_8HF_{15}O_2$
PFOPA	Perfluorooctanephosphonic acid	$C_8H_2F_{17}O_3P$
PFOS	Perfluorooctane sulfonate/sulfonic acid	$C_8F_{17}O_3$- SO_3^-/SO_3H
PFPeA	Perfluoropentanoic acid	$C_5HF_9O_2$
PFPeS	Perfluorpentanesulfonate/ sulfonic acid	$C_5F_{11}O_3$- SO_3^-/SO_3H
PFPrA	Perfluoropropionic acid	$C_3HF_5O_2$
PFSA	Perfluorosulfonate/sulfonic acid	C_nF_{2n+1}- SO_3^-/SO_3H
POSF	Perfluorooctane sulfonyl fluoride	$C_8F_{17}SO_2F$
PTFE	Poly tetrafluoroethylene	–
PVDF-CTFE	Poly vinylidene fluoride-co-chlorotrifluoroethylene	–
PVDF-HFP	Poly vinylidene fluoride-co-hexafluropropylene	–
TFA	Trifluoroacetic acid	$C_2HF_3O_2$
TFMS	Trifluoromethane sulfonic acid	CF_3SO_3H

1. Introduction

Per- and polyfluoroalkyl substances (PFAS), a class of artificially manufactured compounds, have been extensively used in various industrial and commercial products from the mid-twentieth century (Wang et al., 2017). Comprised of a hydrophobic fluorinated alkyl chain at one end and a hydrophilic functional group at the other (Arvaniti et al., 2014), PFAS (C_nF_{2n+1}-R; R: Functional group) (Meng et al., 2020, Buck et al., 2011) have high thermal and chemical stability owing to the strong C-F bonds (Mulabagal et al., 2018). This unique combination of physical and chemical properties, along with the hydrophobic and lipophobic moieties, make PFAS extremely suitable as water and oil repellants and friction resistants (Kwiatkowski et al., 2020). Due to the desired performance and production suitability, PFAS are manufactured massively for wide-scale industrial (manufacturing of paper, textiles, pesticides, leather, medical aids, oil, and minerals, metal plating etc.) and consumer applications (food packaging, cosmetics and personal 'care products, paints, inks, non-stick cooking utensils, surfactants, firefighting foams, and several other waterproof products) (Gagliano et al., 2020; Ji et al., 2020; Wang et al., 2017; Lu et al., 2017; Kotthoff et al., 2015). The general purpose of treating consumer products with PFAS is to impart an extended service life and durability. This practice, in turn, results in prolonged exposure of humans, wildlife, and the environment to PFAS (Herzke et al., 2012).

Although PFAS have been used widely since the 1950s, it was only after the first report of PFAS in wildlife and human serum in the early 2000s that scientific studies on the properties, occurrence, environmental fate, and health effects of PFAS accelerated (Cousins, 2015). Over the past few decades, PFAS have been widely detected in the environment (Nakayama et al., 2019), and they are ubiquitous in almost all aquatic matrices such as drinking water, surface water, groundwater, and coastal water (Wei et al., 2018). Humans can be exposed to PFAS through drinking water and dietary intake (Domingo & Nadal, 2019, Sunderland et al., 2019, Wei et al., 2018). Studies have reported that several acute as well as chronic human diseases, such as thyroid, asthma, anxiety, obesity, pediatric allergies, hyperuricemia, peroxisome proliferation, immune toxicity, kidney disorders, liver damage, and cardiovascular diseases, may be associated with PFAS exposure (Ruan et al., 2019, Jian et al., 2017, Suja et al., 2009). Other studies have demonstrated toxic effects, such as immunotoxicity, carcinogenicity, and hormonal disorders, of some perfluoroalkyl carboxylic acids (PFCAs) on animals (Poothong et al., 2020, Rand & Mabury, 2017).

Today, more than 4,000 PFAS are manufactured and used worldwide, including both conventional (e.g., PFOA and PFOS) and alternative PFAS (short-chain PFAS and newly identified fluorinated replacements of conventional PFAS) (Pan et al., 2019). Anticipating long-term exposure effects due to their persistent and bioaccumulative nature (Ateia et al., 2019a, (Wang et al., 2016), conventional PFAS were voluntarily phased out (Wang et al., 2014b, Krafft & Riess, 2015) due to regulations (e.g., US EPA drinking water guideline for PFOA and PFOS: 70 ng/L) (Li et al., 2020, Ateia et al., 2019b, Pan et al., 2019). Still, many studies have reported high levels of conventional PFAS in WWTP effluents (>100 ng/L) (Table 2 and 3). Alternative PFAS have also been detected abundantly in WWTPs by several studies with concentrations up to some hundred ng/L (Table 2 and 3). Short-chain PFAS, being highly mobile, are detected in all ecosystems (Brendel et al., 2018) and can be equally toxic as their longer-chain counterparts (Li et al., 2020).

The occurrence of PFAS in WWTPs has been reviewed earlier by Arvaniti & Stasinakis, 2015. However, more than 100 new monitoring studies have been published since 2015. Some involve aspects not covered earlier, such as the effect of different treatment stages of a WWTP on removal efficiencies and the transformation of PFAS precursors to perfluoroalkyl acids (PFAAs) during wastewater treatment, that allows a better understanding of the fate of PFAS in WWTPs. Overall, this article aims at providing a systematic overview of the up to date knowledge on the occurrence, fate, transformation, and removal of PFAS in WWTPs.

First, the sampling and analysis methods adopted in monitoring studies are discussed, followed by the global occurrence data of different types of PFAS in the influents and effluents of WWTPs. The removal efficiencies and mechanisms are then discussed in detail, emphasising the fate of PFAS in a full-scale treatment plant and the transformation of PFAS precursors during treatment. This comprehensive review on the levels and fate of conventional and alternative PFAS in WWTPs, along with their removal at different scales, allows the identification of research gaps. Those are compiled in the last section, along with suggestions for future research concerning alternative PFAS (especially ultra-short chain PFAS and PFAS precursors), transformation mechanism of PFAS precursors, and large scale applications of laboratory tested PFAS treatment techniques.

Table 1
Common sampling and analysis practices reported in the recent literature.

Process	Practices	References (selected)
Sampling		
Type	Grab and/or **Composite***	(Table 3)
Containers	**Polypropylene/High density polyethylene bottles**	(Nakayama et al., 2019, Nguyen et al., 2019, Stroski et al., 2019, Chen et al., 2018, Wang et al., 2018b, Elmoznino et al., 2018, (Chen et al., 2017b) Shi et al., 2015)
	Glass bottles	(Kibambe et al., 2020, Nakayama et al., 2019 Ciofi et al., 2018)
Preservation		
Refrigeration	**-20 to 4°C**	(Wang et al., 2020c, Nakayama et al., 2019, Wang et al., 2019a, Zhou et al., 2019, Wang et al., 2018b, Elmoznino et al., 2018, Houtz et al., 2018, Ciofi et al., 2018, Dauchy et al., 2017)
Acidification	Hydrochloric acid	(Nguyen et al., 2019)
	Nitric acid	(McCord et al., 2018)
	Formic acid	(Tian and Sun, 2019)
Other chemical addition	Formaldehyde	(Kibambe et al., 2020)
Time	**1 to 30 days between extraction and/or analysis**	(Zhou et al., 2019, Wang et al., 2018b, Elmoznino et al., 2018, Ciofi et al., 2018, Houtz et al., 2018)
Sample clean-up		
Filter material	**Polypropylene filter**	(Elmoznino et al., 2018)
	Glass fibre	(Wang et al., 2020c, Kibambe et al., 2020, Zhou et al., 2019, Jiawei et al., 2019, Coggan et al., 2019a, Wang et al., 2019a, (Chen et al., 2017b))
Solid-phase extraction	**Weak anion exchange cartridges**	(Wang et al., 2020c, Coggan et al., 2019b, Pan et al., 2019, Munoz et al., 2019, Björnsdotter et al., 2019, Zhou et al., 2019, Mullin et al., 2019, Pan et al., 2019, Jiawei et al., 2019, Houtz et al., 2018, Chen et al., 2018, Eriksson et al., 2017, Boiteux et al., 2016)
	Hydrophilic-lipophilic balance (HLB)	(Wang et al., 2020c, Pan et al., 2019, Mulabagal et al., 2018, Hopkins et al., 2018)
Sonification and Centrifugation	–	(Letcher et al., 2020, Singh et al., 2019, Nguyen et al., 2019)
Analysis		
Equipment	**Ultra/High performance LC MS/MS**	(Al Amin et al., 2020, Letcher et al., 2020, Kibambe et al., 2020, Nakayama et al., 2019, Zhou et al., 2019, Coggan et al., 2019b, Munoz et al., 2019, Nguyen et al., 2019, Elmoznino et al., 2018)

* The practices in **bold** are recommended for precise analysis of PFAS at trace concentrations by EPA method 537.1, Al Amin et al., 2020, Tian and Sun, 2019 and other references cited in this section.

2. Sampling and Analysis

Measuring PFAS at environmentally relevant concentrations (ng/L) comes with a series of associated challenges from sampling to analysis. The sampling, laboratory, and analytical components (e.g., sample collection equipment and containers, laboratory consumables, analytical grade water and solvents, sample preparation, and analytical instruments can potentially cause background interference owing to the ubiquitous presence of PFAS (Al Amin et al., 2020, Misa and Kennedy, 2018). Table 1 summarizes the commonly practised sampling and analytical procedures applied when monitoring PFAS in wastewaters.

Ideally, glass and fluoropolymer containers should be strictly avoided while collecting and storing samples (Lath et al., 2019). This is because of the tendency of PFAS to adsorb onto glass surface (Tian and Sun, 2019) and contamination from fluoropolymer (e.g., PTFE (Polytetrafluoroethylene)) containers (Chen et al., 2018) may result in inaccurate analysis. While recent studies have mostly used either polypropylene or high-density polyethylene (HDPE) bottles for the collection of samples (Table 1), the consequences of using glass containers on quantification remain ambiguous (Kibambe et al., 2020, Nakayama et al., 2019, Ciofi et al., 2018). Cleaning all the sampling equipment and containers with tap water followed by triple rinsing with ultra-pure water and methanol can reduce background contamination (Elmoznino et al., 2018).

It is essential to carefully consider the variableness for a WWTP (e.g., diurnal, seasonal) and representativeness of a sample without which the evaluation of PFAS is of the least value (Liu et al., 2020, Gallen et al., 2018, Berger et al., 2011). Many recent studies have relied on composite sampling for observing the presence of PFAS in WWTPs (Table 3). Grab samples (Wang et al., 2020c, Elmoznino et al., 2018) could be considered less representative as they fail to take into account the diurnal variations in wastewater flow and may not be apt to understand the diversity in PFAS' quality and quantity during the treatment. Though acidification (Kibambe et al., 2020, Nguyen et al., 2019) and refrigeration (Zhou et al., 2019, Dauchy et al., 2017) of biologically active samples, such as wastewater and sludge, can reduce the biodegradation potential of PFAS (precursors) (Tian and Sun, 2019), the excess protons in acidic water could lead to the adsorption of PFAS onto the container surface upon acidification (Ateia et al., 2019b). Most studies refrigerated samples at $\leq$ 4°C (Table 1), as recommended by Standard Methods (EPA 537.1, 2018, D-7979-19, 2019), to ensure the preservation of target PFAS until analysis.

The typically low concentrations of PFAS in wastewater (Table 3), along with the high levels of organic matter and inorganic salts, may result in inaccurate quantification of the target compounds (Liu et al., 2020, Al Amin et al., 2020, Huset et al., 2011). Hence, clean-up and concentration of wastewater are vital steps prior to PFAS analysis. Despite the capability of glass fibre, cellulose acetate, nylon, etc., to adsorb or release PFAS (Tian and Sun, 2019), filtration using glass fibres prior to extraction is often applied (Table 1). As polypropylene filters can also retain long-chain PFAS (Elmoznino et al., 2018), future studies could rely on centrifugation for pre-cleaning wastewaters (Table 1). At present, solid-phase extraction (SPE) using weak anion exchange (WAX) ad-

sorbents is the primary sample clean-up approach for wastewater matrices (Table 1 and Table 4), as WAX cartridges have shown better recoveries for short-chain PFAS than other polymeric cartridges such as Hydrophilic-Lipophilic Balance (HLB) (Pan et al., 2019, Mullin et al., 2019, Hopkins et al., 2018, Taniyasu et al., 2005).

Analytical sensitivities using direct injection has greatly improved over the last few years (< 1 ng/L), which can overcome the contamination risks associated with sample manipulation during SPE (Nakayama et al., 2019, Ciofi et al., 2018). Most studies have used Ultra/High-performance liquid chromatography to analyse PFAS with the mass spectroscopy usually operated in negative electron ionisation mode as most target PFAS are anionic (Nakayama et al., 2019). While the predominantly used reversed-phase chromatography columns work well for most anionic PFAS, separation of highly polar short-chain PFAS (especially ultra-short chain, C<4) is challenging (Tian and Sun, 2019) due to their higher solubility and poor retainability (Yeung et al., 2017). Since the prevalent detection techniques are not capable of detecting a wide range of precursors due to inadequate sensitivities and the unavailability of reference standards (Pan et al., 2019), further studies must adopt the total oxidisable precursors assay (TOPA) technique to analyse PFAS (Al Amin et al., 2020, Houtz et al., 2018). Blank samples, e.g., field, transport, procedural blanks, are necessary to ensure maximum accuracy (Liu et al., 2020, Houtz et al., 2018, Wang et al., 2018b, Chen et al., 2018, Shi et al., 2015). The emerging approach to evaluating PFAS contamination in wastewater is to implement complementary target and nontarget analyses. While the target analyses can quantify known PFAS, nontarget analyses are recommended to fully elucidate the types of PFAS that are present in a sample, including PFAS that are unexpected or unknown, by means of high-resolution mass spectrometry (HRMS) (Jacob et al., 2021).

While sample preservation and preparation are performed with maximum vigilance in the majority of studies, sampling is often not reported in detail. A special emphasis should be placed on the collection of contamination-free samples as it can be critical in trace analysis of PFAS (Feng et al., 2019, Concha-Graña et al., 2018, Berger et al., 2011). Some of the essential elements of sampling were unclear from the literature, such as standard deviations and limitations of sampling methods undertaken (Lorenzo et al., 2019). Hence, detailing the critical aspects of sampling, such as the type of sampling, duration and frequency of sample collection samples, procedure and duration of storage, and precautions necessary to minimize background contamination, would help guide future monitoring studies.

3. Occurrence of PFAS in WWTPs

3.1. Global distribution of monitoring studies

PFAS have been monitored in wastewaters worldwide (Table 2), but the geographical distribution of studies conducted over the last decade is largely unequal (Fig. 1). The majority of studies were conducted in China (31%), followed by Europe (30%) and North America (16%), whereas very limited data is currently available for other parts of the world, including Africa, Oceania, or South America. Emissions and occurrence of PFAS are expected to be higher in more urbanized and industrialized regions, which is a possible rationale behind the high number of studies in China (Wang et al., 2015c). Conversely, lesser studies are expected for countries with lower manufacturing output, such as Australia, where PFAS issues are mostly associated with the use of imported products usage (Coggan et al., 2019b). However, detection of significant amounts of PFAS (maximum reported concentrations of 440 ng/L in influents and 560 ng/L in effluents) in WWTPs around Australian regions (Gallen et al., 2018), together with recent developments in analytical capabilities, might have triggered many recent investigations in Australia (Table 2). At present, no/little information is available regarding the occurrence, fate and transformation of PFAS in WWTPs of least developed and developing countries (except China). Similar to other persistent organic pollutants, PFAS tend to move around the Earth through atmospheric and/or oceanic diffusion (Lam et al.,2017) due to their long-range mobility (Wang et al., 2019c). The data gaps regarding the occurrence and fate of PFAS in WWTPs in several parts of the world limits our understanding of their global production, usage, and transport. Although the lesser number of studies have been conducted in some countries, such as South Korea or Germany, in comparison to China or U.S., a larger number of WWTPs (>70) (Kwon et al., 2017, Ulrich et al., 2016) have been surveyed in a single study for some countries. Hence, comparison of countries based solely on the number of monitoring studies does not present a complete picture of PFAS occurrence considering the variableness in conducting monitoring studies worldwide and lack of such details in many studies.

3.2. PFAS monitored in WWTPs

To date, very few compounds among the vast PFAS family have been monitored. Perfluoroalkyl carboxylic acids (PFCAs) and perfluoroalkyl sulfonates/sulfonic acids (PFSAs) largely dominate (Letcher et al., 2020, Ruan and Jiang, 2017), whereas there is no data for a significant proportion of PFAS. Out of more than 4,000 PFAS currently present in the global market (Wang et al., 2017), most studies so far have focussed largely on two long-chain PFAS, which have been used for the longest time: PFOS and PFOA (Cohen Hubal, 2019). The production of long-chain PFAS has greatly decreased in the last two decades as a consequence of voluntary initiatives (e.g. PFOA, PFOS and related compounds phased out by 3M in 2000 (Krafft and Riess, 2015), national (e.g. PFOA production ban in Norway in 2013 (Li et al., 2020), and international restrictions (e.g. inclusion in the Stockholm convention in 2009 (Buck et al., 2011) (Askeland et al., 2020), Szabo et al., 2018). However, the proportional distribution of different PFAS detected in WWTPs worldwide indicates PFOA and PFOS were monitored and found in about 70% of studies (Fig. 2). The inability of natural environmental conditions to degrade these recalcitrant PFAS (Blum, 2015) with long half-lives (PFOS: >41 years and PFOA: >92 years) (Szabo et al., 2018, Olsen et al., 2008) could be a possible reason for their prevalence long after phasing out from the year 2000 onwards (Houtz et al., 2016, Krafft and Riess, 2015, Wang et al., 2009). Another long-chain PFAS, PFNA (C9), has also been reported in more than 40% of studies, after being previously detected in a wide range of consumer products and human samples (Kotthoff and Bücking, 2018).

An unfortunate consequence of these efforts to reduce contamination by conventional PFAS is the increased production and use of shorter chain (C4-C6) PFAS, which are now ubiquitous at relatively elevated concentrations in WWTPs (Phong Vo et al., 2020).

A significant proportion (40%) of studies reported short-chain PFAS (C≤6, e.g. PFBA, PFBS, PFHxS, and PFPeA) with the exception of PFHxA (C6), which was found in 69% of the total studies (Fig. 2). PFHxA has been extensively used as an alternative to PFOA over the last decade, which explains its increased occurrence in WWTPs (Shan et al., 2014). The detection of the above-mentioned short-chain PFAS in WWTPs is consistent with the studies reporting PFHpA, PFHxA, PFPeA, PFBA, and PFBS as the most widely found short-chain PFAS in the environment (Li et al., 2020) and PFHxS as the most widely used compound in stain and water repellent products (Wang et al., 2015a, The Danish Environmental Protection Agency, 2015). Despite their pervasive occurrence in WWTPs and other environmental media worldwide, no regulatory measures are in place regarding short-chain PFAS, which could be attributed to

Table 2
Global Distribution of PFAS in WWTPs from studies published during 2011-2020.

Regions	Number of WWTPs	PFAS detected	References
China			
Yanghe River, North China	6	PFBA, PFOA, PFOS	(Zhou et al., 2019)
Guangzhou province	3	PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFBS, PFHxS, PFOS	(Jiawei et al., 2019)
Wenzhou	-[#]	6:2 Cl-PFAES	(Wang et al., 2019c)
Daling river basin, Northern China	3	PFBA, PFOA, PFBS	(Wang et al., 2019a)
-	-	TFA, PFBA, 6:2 Cl-PFAES	(Chen et al., 2019)
Jiangsu Province	-	PFOA, PFSAs, H-PFCAs, Cl-PFCAs	(Wang et al., 2018b)
Qingdao City and Jiaozhou Bay	1	PFHxA, PFOA, PFNA, PFHxS, PFOS, FTCAs, FOSA, di-SAmPAP	(Wang et al., 2018a)
Guanting Reservoir	6	PFBA, PFPeA, PFHxA, PFOA, PFBS, PFOS	(Chen et al., 2018)
Liaoning Province, Shandong Province, Chinese Bohai Sea	-	PFHpA, PFOA, PFNA, PFDA, PFHxS, PFOS, FOSAA	(Chen et al., 2017a)
Wuxi City, Jiangsu Province	12	4:2, 6:2, 8:2 and 10:2 FTOHs	(Chen et al., 2017b)
Guangzhou City, South China	3	PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFBS, PFOS	(Pan et al., 2016)
-	-	PFOS, 6:2 Cl-PFAES, 6:2 and 8:2 FTSAs	(Ruan et al., 2015)
Xihe River	-	PFBA, PFOA, PFBS	(Wang et al., 2015b)
Yingcheng, Hubei	-	PFHxA, PFHpA, PFOA, PFOS	(Du et al., 2015)
Shanghai	2	PFPrA, PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFBS, PFHxS, PFOS	(Zhang et al., 2015)
Jinan, Shandong	-	C4-C7, C9-C11, PFOA	(Shi et al., 2015)
Taihu Lake	-	PFBA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFHxS, PFOS	(Guo et al., 2015)
Jiangsu Hi-tech	1	PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFBS, PFHxS, PFOS	(Shan et al., 2014)
Beijing, Shanghai, Guangzhou	28	PFPrA, PFBA, PFHxA, PFOA, PFNA, PFOS	(Zhang et al., 2013c)
Tianjin	6	PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFHxS, PFOS	(Sun et al., 2012)
Shanghai	25	PFPrA, PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFBS, PFHxS, PFOS	(Yan et al., 2012)
Shenyang	4	PFHxA, PFHpA, PFOA, PFNA, PFDA, PFHxS, PFOS, 8:2 FTUCA	(Sun et al., 2011b)
Canada			
-	20	Sidechain fluorinated polymer (C4-C8)	(Letcher et al., 2020)
Cambridge Bay, Victoria Island	1	PFOA, PFNA, PFDA, PFBS, PFOS	(Stroski et al., 2019)
-	20	PFHxA, PFHpA, PFOA, PFNA, PFDA, PFHxS. PFOS	(Guerra et al., 2014)
Ontario	1	PFBA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFSs and PFSAs (C 4, 6, 8, 10)	(Vierke et al., 2013)
USA			
-	-	PFHxA, PFBS, PFOS	(Ateia et al., 2019a)
North Carolina	-	GenX	(Tian and Sun, 2019)
Long Island Sound	12	PFPeA, PFHxA, PFOA, PFOS	(Elmoznino et al., 2018)
Oakdale, Minnesota	-	PFBA, PFPeA, PFHxA, PFOA, PFOS	(Scher et al., 2018)
San Franciso Bay, San Jose Santa Clara, East Bay Dischargers Association, East Bay Municipal Utilities District, Central Contra Costa, Palo Alto, San Mateo, Fairfield Suisun, San Francisco Airport Industrial (SFO)	8	PFBA, PFPeA, PFHxA, PFOA, PFNA, PFBS, PFHxS, PFOS, PFHxPA, PFOPA, 6:2 and 8:2 FTS, MeFOSAA, EtFOSAA	(Houtz et al., 2016)
Washington	4	PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFBS, PFOS	(Furl et al., 2011)
Pennsylvania, Maryland, Delaware	3	PFPeA, PFHxA, 6:2 and 8:2 FTOH, 6:2 FTS	(Wang et al., 2011)
Australia			
-	19	PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFBS, PFPeS, PFHxS, PFHpS, PFOS, PFDS, 6:2 and 8:2 FTS, 6:2 Cl-PFAES	(Coggan et al., 2019b)
-	2	PFBA, PFPeA, PFHxA, PFOA, PFBS, PFOS, 6:2 FTS	(Nguyen et al., 2019)
-	14	PFHxA, PFHpA, PFOA, PFNA, PFDA, PFHxS, PFOS	(Gallen et al., 2018)
Melbourne	1	PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFPeS, PFHxS, PFOS, PFDS, 6:2 FTS	(Szabo et al., 2018)
Queensland	2	PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFBS, PFHxS, PFOS	(Thompson et al., 2011)
Europe			
-	1	PFBA, PFPeA, PFHxA, PFOA, PFDA, PFBS, 6:2, 8:2 and 10:2 FTOH, 6:2 FTCA, 6:2 and 8:2 FTUCA, 5:3 acid	(Gremmel et al., 2017)
Netherlands			
Dordrecht	-	GenX	(Brandsma et al., 2019)
Papendrecht	-	PFBA, PFPeA, PFHxA	(Gebbink et al., 2017)
Switzerland			
-	45	PFCA (C4-C8), PFBS, PFOS	(Alder and van der Voet, 2015)
Zurich	20	PFOS	(Sun et al., 2011a)
Sweden			
-	-	TFA, TFMS	(Björnsdotter et al., 2019)
-	13	PFHxA, PFOA, 6:2 FTSA	(Gobelius et al., 2018)
Henriksdal, Gässlösa, Umeå	3	PFBA, PFPeA, PFHxS, PHHpA, PFOA, PFNA, PFBS, PFHxS, PFOS, 6:2 and 8:2 FTSAs, FOSA/FOSEs, FTCA/FTUCAs	(Eriksson et al., 2017)
Bromma, Bollebygd, Umeå	3	PFHxA, PFHpA, PFOA, PFNA, PFDA, PFBS, PFHxS, PFOS, PFDS	(Filipovic and Berger, 2015)
Italy			
Trissino (Vicenza Province)	-	PFOA, PFOS, other PFAS	(Domingo and Nadal, 2019)
Milan	6	C4-C7, PFOA, PFOS	(Castiglioni et al., 2015)

(*continued on next page*)

Table 2 (*continued*)

Regions	Number of WWTPs	PFAS detected	References
South Korea			
Hyung-san River, Gyeongju and Pohang	3	PFHxA, PFBS, PFOS	(Seo et al., 2019)
Seoul, Incheon, Daejeon, Daegu, Busan, Gwangiu	81	PFHxA, PFHpA, PFOA, PFNA, PFDA, PFHxS, PFOS, N-EtFOSA	(Kwon et al., 2017)
-	25	PFHxA, PFBA, PFBS, PFHxS	(Kim et al., 2016)
Japan			
Osaka	-	PFHxA	(Brendel et al., 2018)
Germany			
-	1	PFOA, PFHxS, 6:2 and 8:2 FTS	(Gremmel et al., 2017)
Halle	-	PFHxA, PFOA, PFNA, PFBS, PFOS	(Shafique et al., 2017)
Bavaria	71	PFHxA, PFHpA, PFOA, PFNA, PFDA, PFBS, PFHxS, PFOS, PFDS, PFOSA	(Ulrich et al., 2016)
River Alz	-	ADONA	(Wang et al., 2013)
Hesse	5	PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFBS, PFHxS, PFOS	(Llorca et al., 2012)
South Africa			
Gauteng Province	3	PFBA, PFPeA, PFHxA, PFOA, PFDA, PFHxS, PFOS	(Kibambe et al., 2020)
Spain			
Albufera Natural Park, Valencia	-	PFPeA, PFHxA, PFOA, PFHxS, PFOS	(Pico et al., 2012)
Alzira, Loja, and Cuenca	16	PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFBS, PFHxS, PFHpS, PFOS	(Campo et al., 2014)
France			
North France	1	PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFDA, 6:2 and 8:2 FTSA, 6:2, 8:2, 10:2 and 12:2 FTOH	(Dauchy et al., 2017)
Nancy	1	PFOA, PFOS	(Pasquini et al., 2014)
India			
Chennai	-	PFOA, PFOS	(Sunantha and Vasudevan, 2016)
Kenya			
Bungoma, Busia, Kakamega, Kisumu, Kisii, Mumias	6	PFHxA, PFOA, PFNA, PFDA, PFHxS, PFOS, PFDS, PFOSA	(Chirikona et al., 2015)
Vietnam			
Hanoi	-	PFOA, PFOS	(Duong et al., 2015)
Red River	-	PFOA, PFNA, PFDA, PFOS	(Duong et al., 2015)
Thailand			
Bangkok	2	PFBA, PFHxA, PFHpA, PFOA, PFNA, PFBS, PFOS	(Tabtong et al., 2015)
Central and Eastern Thailand	2	PFHxA, PFHpA, PFOA, PFNA, PFDA, PFHxS, PFOS	(Kunacheva et al., 2011)
Nigeria			
Lagos, Oyo, Ogun	10	PFHxA, PFHpA, PFOA, PFNA, PFDA, PFBS, PFHxS, PFOS	(Sindiku et al., 2013)
Finland			
Turku, Espoo, Helsinki, Porvoo	4	PFHxA, PFOA, PFDA, PFOS	(Perkola and Sainio, 2013)
Greece			
Athens, Mytilene	2	PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFBS, PFHxS, PFHpS, PFOS	(Arvaniti et al., 2012)

[#] No information available

the limited knowledge about their toxic effects and human exposure pathways. Recent literature suggests, shorter-chain PFAS might behave similarly in terms of persistence and bio-accumulation as their longer-chain counterparts (Ateia et al., 2019b) and can be equally hazardous as observed during experiments on animals (Sunderland et al., 2019). It is noteworthy that PFHxS, having similar toxicity potential as PFOA (Szabo et al., 2018), has been proposed for inclusion as a persistent organic pollutant (POPRC-15/1) under the Stockholm Convention (Kotthoff and Bücking, 2018). The relative lack of information regarding levels of short and ultra-short chain PFAS in wastewaters is particularly concerning as the demand for short-chain PFAS, especially in developing countries, is likely to increase their release into the environment in the coming years (Li et al., 2020).

The occurrence of precursors of PFAS and their fate in WWTPs is least studied, which calls for future research in this field as the production of precursors of PFAS outweighs that of PFCAs and PFSAs (Letcher et al., 2020). PFAS precursors (e.g., FTOHs) are widely used for manufacturing of various products such as fabrics, papers, surfactants, water and stain-resistant products, and firefighting foams (Lewis et al., 2016, Liu et al., 2015, Shi et al., 2015 Wang et al., 2012), and, hence, eventually make their way to WWTPs (Zhang et al., 2013b), where they can transform into PFAAs (Brendel et al., 2018, Houtz et al., 2016). Recently, two new side-chain polymers (perfluorooctane sulfonamide-urethane polymer and perfluorobutane sulfonamide-urethane polymer), which could potentially transform into polyfluorinated sulfonamides (e.g. FBSA and FOSA) and then short-chain PFSAs (e.g. PFBS), were reported for the first time in Canadian WWTP sludge with maximum concentrations of 105 and 2,051 ng/g d.w respectively (Letcher et al., 2020). Perfluoroalkyl ether acids (PFESAs and PFECAs) and fluorotelomer acids (FTSA and FTCA) are two main classes of PFAS which are widely used as alternatives to conventional PFAS in electroplating, processing aids, metal surface treatment, plastics manufacturing, etc. (Wang et al., 2019c, Wang et al., 2015d), and, hence, they are likely to be present in WWTPs. However, very few studies have evaluated the occurrence and fate of such compounds in WWTPs. Hence, the role of WWTPs as point sources of alternative PFAS to the aquatic environment requires deeper investigations.

Table 3
Influent and effluent concentrations for different PFAS categories, together with the associated sampling details.

Compounds	Influent (ng/L)	Effluent (ng/L)	[5]Removal efficiency (%)	Grab	Composite (Duration)	Country	Remarks	References
ΣPFAS	10 to 15	14 to 24	-40 to -60	✓	–	Jordan	[1]Ef>In	(Shigei et al., 2020)
	–	110	–	✓	(24 h)	Australia	–	(Coggan et al., 2019b)
	–	538	–	–	–	China	–	(Jiawei et al., 2019)
	*d	*d	–	✓	–	China	–	(Wang et al., 2018b)
	–	83	–	–	–	China	–	(Wang et al., 2018a)
	–	103 to 443	–	–	–	China	–	(Chen et al., 2016)
	20 to 232	16 to 234	-1 to +20 -	–	(24 h)	China	[2]Poor removal	(Pan et al., 2016)
	2	3	-50	–	–	Thailand	Ef>In	(Tabtong et al., 2015)
	–	130±20	–	–	(24 h)	Canada	–	(Vierke et al., 2013)
	0 to 91	0 to 107	-8	✓	–	China	Ef>In	(Zhang et al., 2013c)
	–	219	–	✓	–	USA	–	(Furl et al., 2011)
	761	903	-19	✓	–	Thailand	Ef>In	(Kunacheva et al., 2011)
	55	94	-71	✓	–	Australia	Ef>In	(Gallen et al., 2018)
ΣPFAA	81±25	–	–	–	(24 h)	Australia	–	(Nguyen et al., 2019)
ΣPFCA	930	780	+16	✓	–	China	Poor removal	(Wang et al., 2020c)
	20	40	-100	–	(7 days)	Sweden	Ef>In	(Eriksson et al., 2017)
	–	137	–	–	(24h)	Italy	–	(Castiglioni et al., 2015)
	–	1,580	–	–	–	Switzerland	–	(Alder and van der Voet, 2015)
ΣPFSA	490	260	+47	✓	–	China	Poor removal	(Wang et al., 2020c)
	5	6	-20	–	(7 days)	Sweden	Ef>In	(Eriksson et al., 2017)
	–	5	–	–	(24 h)	Italy	–	(Castiglioni et al., 2015)
Ultra-short chain compounds								
PFPrA	17±2	14±2	+18	✓	–	China	Poor removal	(Zhang et al., 2015)
Short-chain compounds								
PFBA	5	4	+20	✓	(24 h)	Spain	Poor removal	(Lorenzo et al., 2019)
	130	490	-277	✓	(3 days)	USA	Ef>In	(Houtz et al., 2018)
	24	20	+17	–	–	China	Poor removal	(Zhou et al., 2019)
	–	19	–	✓	–	USA	–	(Houtz et al., 2016)
	–	>800	–	–	–	China	–	(Wang et al., 2015b)
PFPeA	2	1	+50	–	(24 h)	Spain	[3]Ef<In	(Lorenzo et al., 2019)
	3	2	+33	–	–	China	Poor removal	(Zhou et al., 2019)
	200	1,690	-745	✓	(3 days)	USA	Ef>In	(Houtz et al., 2018)
	–	32	–	–	(24 h)	USA	–	(Elmoznino et al., 2018)
	–	10	–	✓	✓	USA	–	(Houtz et al., 2016)
	27	76	-181	–	(24 h)	Greece	Ef>In	(Arvaniti et al., 2012)
PFHxA	2	3	-50	–	(24 h)	Spain	Ef>In	(Lorenzo et al., 2019)
	4	4	0	–	–	China	Poor removal	(Zhou et al., 2019)
	–	18	–	–	–	China	–	(Jiawei et al., 2019)
	350	1,370	-291	✓	(3 days)	USA	Ef>In	(Houtz et al., 2018)
	–	27	–	–	(24 h)	USA	–	(Elmoznino et al., 2018)
	10	18	-80	✓	–	Australia	Ef>In	(Gallen et al., 2018)
	–	21,200	–	✓	(24 h)	France	Poor removal	(Dauchy et al., 2017)**
	–	24	–	✓	–	USA	–	(Houtz et al., 2016)
	7±2	13±200	-86	–	(24 h)	Sweden	Ef>In	(Filipovic and Berger, 2015)
	–	911 to 10,300	–	–	–	China	–	(Shan et al., 2014)
	–	4	–	–	(24 h)	Finland	–	(Perkola and Sainio, 2013)
	20±2	42±4	-110	✓	–	China	Ef>In	(Zhang et al., 2015)
	6	10	-67	–	–	USA	Ef>In	(Xiao et al., 2012)
PFBS	#n.d	17	–	–	(24 h)	Spain	[4]n.d in In but present in Ef	(Lorenzo et al., 2019)
	3	2	+33	–	–	China	Poor removal	(Zhou et al., 2019)
	1 to 195	1 to 186	-9 to +5	–	(24 h)	China	Ef>In to Poor removal	(Pan et al., 2016)
	–	3	–	✓	–	USA	–	(Houtz et al., 2016)
	–	>1,500	–	–	–	China	–	(Wang et al., 2015b)
	39 ± 1	386 ± 43	-890	✓	–	China	Ef>In	(Zhang et al., 2015)
Long-chain compounds								
PFHpA	2	20	-900	–	(24 h)	Spain	Ef>In	(Lorenzo et al., 2019)
	60	940	-1,467	✓	(3 days)	USA	Ef>In	(Houtz et al., 2018)
PFOA	3	65	-2,067	–	(24 h)	Spain	Ef>In	(Lorenzo et al., 2019)
	6	6	0	–	–	China	Poor removal	(Zhou et al., 2019)
	–	431	–	–	–	China	–	(Jiawei et al., 2019)
	100	110	-10	✓	(3 days)	USA	Ef>In	(Houtz et al., 2018)
	–	36	–	–	(24 h)	USA	–	(Elmoznino et al., 2018)
	5	22	-340	✓	–	Australia	Ef>In	(Gallen et al., 2018)
	3 to 12	4 to 13	-33 to -8		(24 h)	China	Ef>In	(Pan et al., 2016)
	–	23	–	✓	–	USA	–	(Houtz et al., 2016)
	3	8±1	-167	–	(24 h)	Sweden	Ef>In	(Filipovic and Berger, 2015)
	12	–	–	–	–	Kenya	–	(Chirikona et al., 2015)
	10	10	0	–	(24 h)	France	Poor removal	(Pasquini et al., 2014)
	–	29	–	–	(24 h)	Finland	–	(Perkola and Sainio, 2013)
	204 ± 10	341 ± 16	-67	✓	–	China	Ef>In	(Zhang et al., 2015)
	2 to 91	13 to 107	-550 to -18	–	–	China	Ef>In	(Zhang et al., 2013c)
	7	11	-57	–	–	USA	Ef>In	(Xiao et al., 2012)
	92	100 to 145	-33	–	–	China	Ef>In	(Sun et al., 2012)
	26,000	28,000	-8	✓	–	China	Ef>In	(Sun et al., 2012)

(*continued on next page*)

Table 3 (*continued*)

Compounds	Influent (ng/L)	Effluent (ng/L)	[5]Removal efficiency (%)	Grab	Composite (Duration)	Country	Remarks	References
PFNA	–	9	–	✓	–	USA	–	(Houtz et al., 2016)
PFDA	n.d*	3	–	✓	(24 h)	Spain	n.d in In but present in Ef	(Lorenzo et al., 2019)
PFHxS	7	18	-157	–	(24 h)	Spain	Ef>In	(Lorenzo et al., 2019)
	–	28	–	–	–	China	–	(Jiawei et al., 2019)
	–	5	–	✓	–	USA	–	(Houtz et al., 2016)
	430	500	-16	✓	(3 days)	USA	Ef>In	(Houtz et al., 2018)
PFHpS	2	3	-50	–	(24 h)	Spain	Ef>In	(Lorenzo et al., 2019)
	2,950	1,450	+51	✓	(3 days)	USA	Ef<In	(Houtz et al., 2018)
	7	NA	–	–	–	USA	–	(Xiao et al., 2012)
PFOS	11	35	-218	–	(24 h)	Spain	Ef>In	(Lorenzo et al., 2019)
	–	35	–	–	–	China	–	(Jiawei et al., 2019)
	–	18	–	–	(24 h)	USA	–	(Elmoznino et al., 2018)
	6 to 11	3 to 23	-109 to +50	–	(24 h)	China	Ef>In	(Pan et al., 2016)
	–	15	–	✓	–	USA	–	(Houtz et al., 2016)
	2±1	1	+50	–	(24 h)	Sweden	Ef<In	(Filipovic and Berger, 2015)
	–	28	–	–	(24 h)	Finland	–	(Perkola and Sainio, 2013)
	1 to 32	1 to 67	-109	–	–	China	Ef>In	(Zhang et al., 2013c)
	2 to 176	1 to 75	+50 to +57	✓	–	China	Ef<In	(Sun et al., 2012)
Precursor compounds								
6:2 FTS	2,400	4,790	-100	✓	(3 days)	USA	Ef>In	(Houtz et al., 2018)
	–	3	–	✓	–	USA	–	(Houtz et al., 2016)
8:2 FTS	700	330	+53	✓	(3 days)	USA	Ef<In	(Houtz et al., 2018)
FTSA	4	3	+25	–	(7 days)	Sweden	Poor removal	(Eriksson et al., 2017)
6:2 FTAB	8,712,250	4,228,250	+51	✓	(24 h)	France	Ef<In	(Dauchy et al., 2017)**
M4	1,459,500	85,000	+94	✓	(24 h)	France	Ef<In	(Dauchy et al., 2017)**

The removal efficiency of the wastewater treatment (% Removal) is based on the reported effluent and influent concentrations (negative values indicate an increase in PFAS concentration during the treatment).

[1] Ef>In: Concentration of compound in effluent > Concentration of compound in influent; Removal efficiency (%) < 0

[2] Poor removal: 0 ≤ Removal efficiency (%) < + 50

[3] Ef<In: Concentration of compound in effluent < Concentration of compound in influent; Removal efficiency (%) ≥ + 50

[4] n.d in In but present in Ef: Not detected in influent but present in effluent

[5] Removal efficiencies (%) are calculated from the influent and effluent concentration values, which were rounded to the nearest whole numbers.

** The influent and effluent concentrations (ng/L) are the average of concentrations of PFAS in the homogenization pool (Table S4, (Dauchy et al., 2017)) and effluent (Table S7, (Dauchy et al., 2017)) respectively, during May, July, October, December 2013.

* d: Detected

n.d: Not detected

3.3. Concentrations of PFAS in influents

Table 3 summarises the concentrations (ng/L) of different PFAS classes detected in the influent of WWTPs across the world. PFAAs tend to be more abundant and present at higher concentrations in WWTPs relative to other classes of PFAS. According to Filipovic & Berger (2015), industrial wastes are a primary source of PFAAs found in WWTP. For instance, Gallen et al. (2018) suspected a local industry to be responsible for high levels (444 ng/L) of PFAAs found in the influent of a WWTP. Similarly, influents receiving wastes from an electroplating industry carried elevated PFAS content with short-chain PFAAs (C4-C8), accounting for more than 60 % of the total PFAS (Jiawei et al., 2019). In some studies conducted prior to 2015, PFOA concentrations were as high as 92–66,000 ng/L in WWTP influents (Sun et al., 2012). The lower levels of PFOA typically reported after 2015 (3–100 ng/L) (Kim et al., 2021, Lorenzo et al., 2019, Zhou et al., 2019, Gallen et al., 2018, Houtz et al., 2018, Pan et al., 2016), can be attributed to the united efforts undertaken to reduce the production of conventional PFAS (see section "PFAS monitored in WWTPs" above).

Following PFOA, PFHxA has been the second most abundant in terms of occurrence in WWTP influents. Dauchy et al. (2017) found PFHxA at an unusually elevated concentration (~4,000 ng/L) in the influent of a WWTP that received raw effluents from a fluorochemical manufacturing plant. The technical performance of short-chain PFAS being lower than long-chain PFAS (Ateia et al., 2019b) larger quantities of short-chain PFHxA are typically required to achieve similar performance as long-chain PFOA. This, in turn, can lead to the massive production and abundance of PFHxA (Shan et al., 2014). More studies elucidating the reason behind such a high prevalence of PFHxA and tracking its sources to WWTPs are needed.

In addition to short (C6-C4) and long (C>6) chain PFAAs, PFSAs (C9-C4) have also been detected at significant levels (up to 1,500 ng/L) in WWTP influents (Table 3). The abundance of PFOS in domestic wastes (Chen et al., 2012) and the use of PFBS as an alternative to conventional PFAS (Liang and Chang, 2019) have been stated earlier. Studies regarding ultrashort (C2-C3) chain PFAS in WWTPs are scarce, which could be attributed to the difficulty in analysing three or fewer carbon PFAS (see the section on sampling and analysis). Björnsdotter et al. (2019) confirmed the presence of TFA (C2) and PFPrA (C3) at concentrations as high as 14,000 and 53,000 ng/L in landfill leachates and groundwater samples, respectively. Another study in China detected PFPrA (17 ng/L) in the influent of a WWTP (Zhang et al., 2015). These findings highlight the need for a further investigation of WWTPs as potential sources of such ultrashort chain PFAS to groundwater and/or landfills.

Precursors of prevalent PFAS have been found at elevated concentrations in influents of some WWTPs (Table 3). Unlike PFAAs, the precursors' concentrations systematically decreased from influent to effluent (Dauchy et al., 2017, Houtz et al., 2018), probably due to their transformation into PFAS or other compounds during treatment (Chen et al., 2017b). In some cases, the precursors were reported in the influent but not in the effluent. For instance, Houtz et al. (2018), for the first time, reported two new classes of fluorotelomer thioether surfactants (FTSHCs) and fluorotelomer thioether alkylamido hydroxyl carboxylate (FTSAHCs) in the influent but did not find those in the effluent. Other new PFAS, e.g., 6:2 fluorotelomer sulfonamide alkylbetaine (6:2 FTAB) and 6:2 fluorotelomer sulfonamide propyl N,N dimethylamine (M4), have

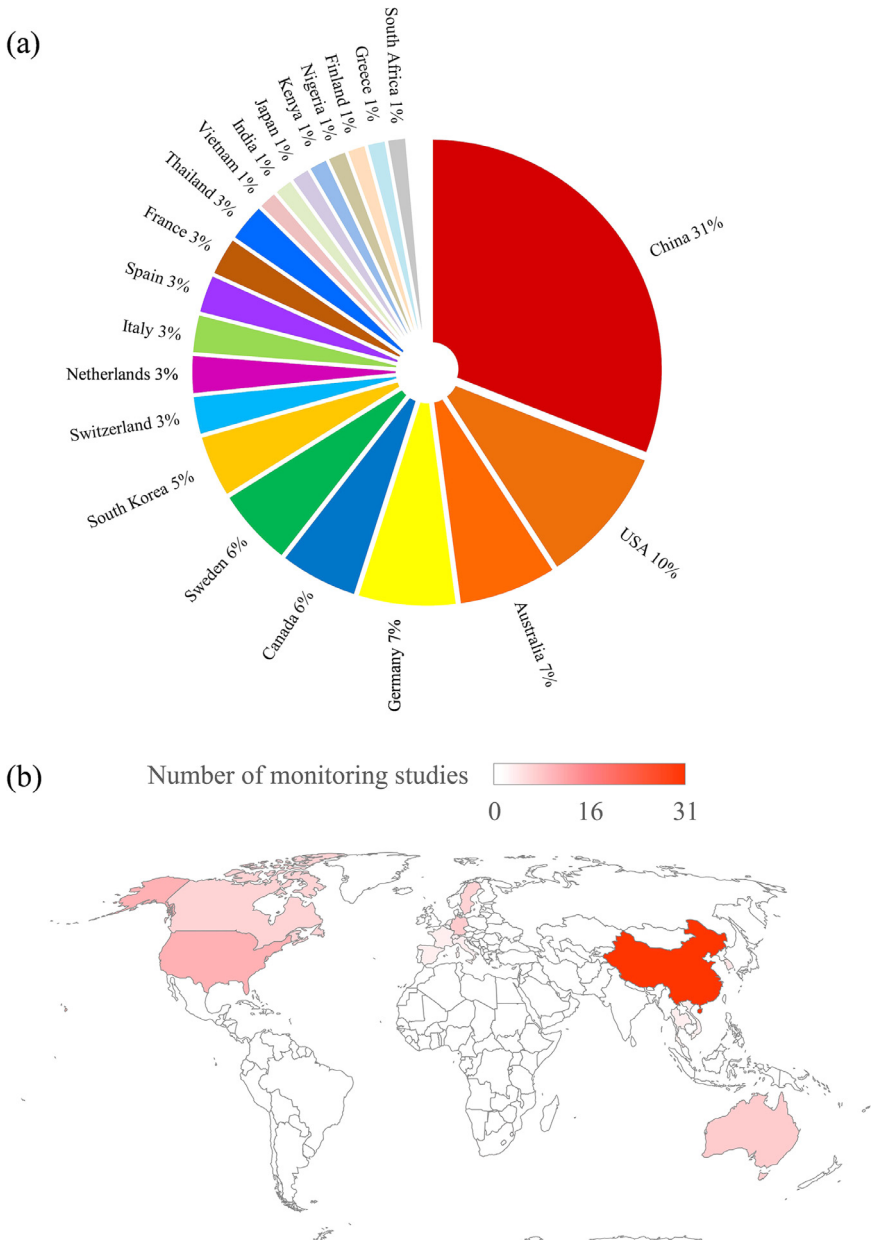


Fig. 1. Geographical distribution of studies monitoring PFAS in WWTPs during 2011–2020 in proportions (a) and number of studies across the globe (b).

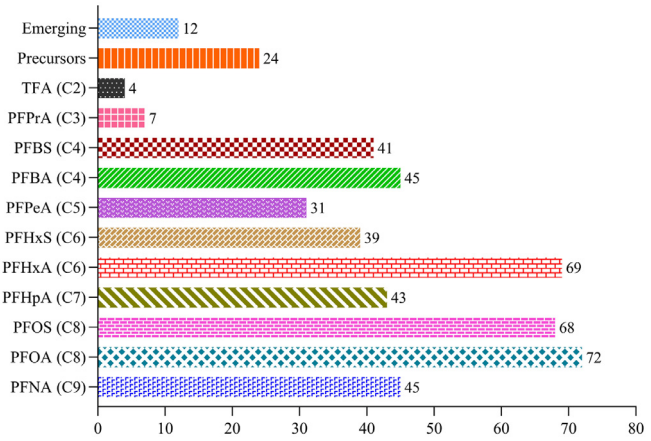


Fig. 2. Proportional distribution of selected PFAS found in monitoring studies. *Abundance of PFAS (%) was calculated based on 74 studies monitoring PFAS in WWTPs, E.g. PFHxA was detected in 69% of studies.

been reported at high concentrations (Table 3) in influent of a WWTP receiving effluent from a fluorochemical manufacturing facility (Dauchy et al., 2017). Thus, more monitoring studies involving new classes of PFAS and their precursors are recommended to understand their occurrence and fate in WWTPs.

Most monitoring studies conducted to date have focused on the occurrence of PFAS in the effluents than in influents of WWTPs. However, more work on PFAS in the influents could add to the existing knowledge about the source of newly discovered precursors to influents.

3.4. Concentrations of PFAS in effluents

Knowing the levels of PFAS in the effluents of a WWTP allows characterisation of the discharge into the environment. Effluents with comparatively lesser background matrix effect are easier to analyse than influents. This could be a possible reason behind the availability of more data on PFAS in effluents than

in influents (Table 3). High levels (1,100 ng/L) of PFAS has been detected in the effluents of WWTPs receiving wastes from industries (Castiglioni et al., 2015). WWTP effluents are significant point sources of PFAS to various environmental matrices, which receive wastewaters either through direct discharge (surface and coastal water) or application of recycled wastewaters (groundwaters, soils, and vegetation). Elmoznino et al. (2018) found WWTP effluent in the Long Island South, USA, to be a source of PFAAs to major tributary rivers. Similarly, a study conducted in the state of Washington, USA, suspected the effluent (about 200 ng/L of PFAAs) from a WWTP as the source of contamination to the surface water downstream (Furl et al., 2011). Inadequate removal or rather elevated levels of PFAS in the effluents has been observed in most of the studies (Table 3). For instance, in a recent study, Shigei et al. (2020) noticed that the combined concentration of three PFAS, i.e. PFPeA, PFOA, and PFNA, increased six times from influent to effluent. Similarly, the concentrations of short-chain PFAS, such as PFBA, PFPeA, and PFHxA, are often larger in the effluent than in the influent (Lorenzo et al., 2019, Houtz et al., 2018, Gallen et al., 2018, Filipovic et al., 2015, Arvaniti et al., 2012), which is much likely due to the degradation of precursors into recalcitrant shorter chain PFAS (Chen et al., 2017b). On the other hand, the concentration of longer chain PFAS (C>9) and precursors in the effluent decreased for some cases, which could be due to their sorption onto activated sludge (Zhou et al., 2019, Sun et al., 2012) and/or degradation to short-chain PFAAs (Zhang et al., 2015).

Houtz et al. (2018) suggested adsorption onto sludge and/or degradation to be responsible for the absence of novel PFAS precursors (FTSHCs and FTSAHCs), in the effluent in spite of their detection in the influent. Another study reported high removal (>80%) for two new fluorotelomer compounds: 6:2 FTAB and M4 in the final effluent after treatment (Dauchy et al., 2017). The actual reason behind the substantial elimination of such newly found PFAS in effluents is unclear and needs further research. However, an exception was observed in the case of a PFAS precursor, 6:2 FTS, whose concentration doubled after treatment (Houtz et al., 2018). Emerging PFAS, such as the ones mentioned above, have comparable levels of residue as their longer-chain counterparts and can potentially cause similar impacts as long-chain PFAS (Ruan and Jiang, 2017). Hence, in-depth investigation of newly discovered precursors and fluorinated compounds in terms of their occurrence and fate in WWTP would be helpful in better understanding such contradictory observations.

3.5. PFAS in biosolids

The occurrence of PFAS in biosolids and its consequences are often overlooked (Letcher et al., 2020), although biosolids and sludge from secondary treatment are notable routes for PFAS emission into the environment (e.g. groundwater, soil, vegetation etc.) (Ebrahimi et al., 2021, Stoiber et al., 2020, Zhang et al., 2013). PFCAs, PFSAs, precursors (e.g. FTOHs) (Guerra et al., 2014), and novel fluorinated polymers (Letcher et al., 2020) have been detected in biosolids worldwide, including North America, Europe, Africa, Asia, and Australia (Moodie et al., 2021). A recent study in Australia reported significant levels of PFAS (up to 910 ng/g) in biosolids (Moodie et al., 2021), despite the usage of imported products, rather than PFAS using/manufacturing industries, being predominantly responsible for the generation of PFAS wastes (Coggan et al., 2019b). Regulating various operational parameters (e.g., solid retention time, temperature) and solution chemistry (e.g., pH) have shown to reduce the PFAS loads in the effluent due to enhanced partitioning of PFAS into sludge (Ebrahimi et al., 2021, Zhang et al., 2013) (Section 4.1). However, these techniques result in the accumulation of PFAS in biosolids which could eventually leach into the environment (Rodríguez-Varela et al., 2021, Moodie et al., 2021, Guerra et al., 2014). Further research indicating the effects of various process parameters on the leaching and/or release of PFAS from biosolids into the environment is required.

PFAS enriched biosolids are disposed to landfills (Stoiber et al., 2020, Fuertes et al., 2017) which could be responsible for the abundance of PFAS (10 to 416,000 ng/L), including short and ultra-short chain compounds (Björnsdotter et al., 2019), in landfill leachates throughout the world (Hamid et al., 2018). Concerns regarding the long-term release of PFAS from solid waste disposal facilities and associated risks are emerging due to observed emissions of PFAS from landfills into various environmental matrices (Wang et al., 2020a). While incineration of PFAS loaded wastes and evaporation of volatile PFAS (primarily precursors, e.g. FTOHs) can discharge PFAS (including harmful fluorinated greenhouse gases) into the atmosphere (Stoiber et al., 2020, Wang et al., 2020a, Hamid et al., 2018), leaching of PFAS can contaminate the groundwater and soil (Moodie et al., 2021, Stoiber et al., 2020). The potential of commercial landfills to release PFAS into the air, water, and soil require immediate attention. Moreover, treating landfill leachates through WWTPs is widely practised (Stoiber et al., 2020). Hence, PFAS could be just transferring between wastewater and solid waste treatment facilities rather than getting removed from the environment, and hence more research is needed in this field.

4. Removal of PFAS through wastewater remediation techniques

The type of treatment and the physicochemical characteristics of PFAS significantly affect their removal efficiencies and composition (Kibambe et al., 2020, Ateia et al., 2019b, Chen et al., 2018, Rahman et al., 2014). Table 4 summarizes the various treatment techniques and their effectiveness for the removal of PFAS from wastewater, along with the effect of the background matrix. A drastic increase in the concentrations of PFAS from influent to effluent has been observed in many studies (Table 3) and is discussed in more detail in the following section.

4.1. Full-scale treatment

4.1.1. Conventional WWTPs: Overview

Very few studies have investigated the levels of PFAS at various stages of a WWTP (Kibambe et al., 2020, Wang et al., 2019a, Zhou et al., 2019). The design, capacity, and operating conditions such as hydraulic and sludge retention time, feed concentrations, temperature, flow rate, and mixed liquor suspended solids decide the fate of PFAS in a WWTP (Kibambe et al., 2020). For instance, longer retention time and higher temperature led to the formation of PFAAs, which increased their levels in the effluent (Guerra et al., 2014). On the other hand, unique operating parameters of higher mixed liquor suspended solids (5,000 mg/L) and longer sludge retention time (45 days) enhanced the removal of PFAS from the aqueous phase significantly in a WWTP using recirculation of activated sludge (Kibambe et al., 2020). Other factors such as the source of wastewaters (domestic, industrial, urban runoff and/or agricultural), climatic conditions, and atmospheric depositions influenced the concentrations of different PFAS and their removal from the aqueous phase in WWTPs (Ratola et al., 2012).

Along with process and external factors, the type of PFAS is also crucial. For example, PFAAs, the most abundant class of PFAS, which are usually present in anionic forms, are non-volatile and recalcitrant to biodegradation (Ateia et al., 2019b). Whereas neutral PFAS, which are mainly precursors such as FTOHs and perfluoroalkyl sulfonamidoethanols, are volatile and/ semi-volatile (Wang et al., 2021 (Ateia et al., 2019b), Hamid et al., 2018) and hence, can be released into the atmosphere from WWTPs

Table 4
Overview of treatment techniques tested for the removal of PFAS from wastewater at the laboratory scale (Lab), pilot or full scale.

Treatment	Target PFAS	Scale	Initial conc.	Details	Mechanism	Time	Efficiency (%)	Matrices effect	References
Electrochemical degradation	ΣPFAS	Full	1,642 μg/L	*Electrodes:* boron doped diamond (BDD) anode and stainless-steel cathode, *Current density:* 50 mA/cm^2 *Pre-treatment of WW:* SPE*	Decarboxylation and defluorination of PFCA carboxyl radical by OH radical oxidation	10 h	~ 100	TOC: removal > 90%	(Gomez-Ruiz et al., 2017)
Adsorption	PFOS	Full	146 ng/L	*Sludge:* 45 days aged *Pre-treatment of WW:* SPE	adsorption onto activated sludge	–	94	–	(Kibambe et al., 2020)
Adsorption	ΣPFAS	Full	200 ng/L	*Pre-treatment of WW:* SPE	adsorption onto activated sludge	–	82	–	(Kibambe et al., 2020)
Biological filtration	ΣPFAS	Full	18.5 ng/L	*Pre-treatment of WW:* SPE	–	–	41	–	(Kibambe et al., 2020)
Biological deodorization of raw water	ΣPFAAs	Full	208 ng/L	Biologically induced deodorization to raw influent *Pre-treatment of WW:* SPE	–	–	-70	–	(Wang et al., 2019a)
Adsorption	C3 to C11 PFCAs, C4, C6, C8 PFSAs, and C8 FOSA	Pilot	3 to 5 μg/L	*Adsorbent:* biochar filtration (BF), BF with active biofilm and BF with inactive biofilm *Pre-treatment of WW:* SPE	Hydrophobic interaction and hydrophilic functional groups interaction	22 weeks	58, 42, and 22	*Biofilm:* decreased availability of adsorption sites *Protein:* led to lower removal of PFOS and PFOA than carbohydrate content	(Dalahmeh et al., 2019)
Adsorption	PFAA (PFPeA, PFHxA, PFOA and PFOS)	Pilot	4 to 18 ng/L	*Adsorbent:* granular activated carbon *Pre-treatment of WW:* SPE	Hydrophobic interactions	–	61	*Biomass:* pore blockage	(Inyang and Dickenson, 2017)
Adsorption	GenX	Lab	70 ng/L to 5 mg/L	*Adsorbent:* purolite® A860, strongly basic anion exchange (IX) resin *Pre-treatment of WW:* microfiltration and ultrafiltration	Ion exchange	10 min	100	*Organic ions:* pore diffusion *Inorganic ions:* limited the availability of adsorption sites	(Dixit et al., 2020a)
Adsorption	PFOA, PFOS, PFBA, PFBS, and GenX	Lab	10 μg/L	*Adsorbent*: anionic ion exchange (IX) resin *Pre-treatment of WW:* SPE	Ion exchange	–	99	Organic and inorganic ions: competed with PFAS for adsorption sites	(Dixit et al., 2020b)
Adsorption	PFOA and PFOS	Lab	–	*Adsorbent:* anionic ion exchange resin (IX) *Pre-treatment of WW:* SPE	Ion exchange	40 min	70	*Organic matter:* more NOM required more dosage of resin	(Dixit et al., 2019)
Adsorption	ΣPFAS: PFOA, PFOS, PFBA, PFBS, GenX, ADONA, F-53B & other	Lab	1,000 ng/L	*Adsorbent:* DMAPAA-Q hydrogel polymers *Pre-treatment of WW:* SPE	Hydrophobic interaction: long-chain PFAS and PFSA Electrostatic interaction: other PFAS	< 2 h	100	*Inorganic anions:* adsorption of short-chain PFAAs decreased by an increase in anion concentrations which compete for adsorption sites and/or salt-out PFAS	(Ateia et al., 2019a)
Adsorption	PFOA	Lab	150 mg/L	*Adsorbent:* 1/19-MF-VT, Fe_3O_4 nanopowder and fluorinated vermiculite mixed at a ratio (mass) of 1:19 *Pre-treatment of WW:* centrifugation	Fast and selective adsorption for PFOS due to the fluorophilicity of C–F chains	4 h	98	–	(Du et al., 2017)

(*continued on next page*)

Table 4 (*continued*)

Treatment	Target PFAS	Scale	Initial conc.	Details	Mechanism	Time	Efficiency (%)	Matrices effect	References
Adsorption	PFAA (PFOA or PFBA)	Lab	1 µg/L	*Adsorbent:* biochar *Pre-treatment of WW:* SPE	Intra-particle diffusion	7 days	0 to 94	*Organic ions:* blockage of adsorption sites to PFOA.	(Inyang and Dickenson, 2017)
Adsorption	PFHxA, PFHpA, and PFOA	Lab	314, 364, and 828 mg/L	*Adsorbent:* bamboo-derived activated carbon *Pre-treatment of WW:* filtration	Hydrophobic interaction	33.5 h	< 80	*Divalent cations:* (Ca^{2+} and Mg^{2+}) neutralized negative surface charges of adsorbents forming bridges between PFAS	(Du et al., 2015)
Adsorption	PFHxA, PFHpA, and PFOA	Lab	314, 364, and 828 mg/L	*Adsorbent:* resin IRA67 *Pre-treatment of WW:* filtration	Anion exchange	33.5 h	< 80	*Anions:* decreased PFCAs solubility by salt-out effect, promoted the formation of PFCAs micelles, decreased removal of PFCAs by SO_4^{2-}	(Du et al., 2015)
Adsorption	PFHpA, PFHxS	Lab	5 µg/L	*Adsorbent:* WWTP sludge	Bridging effect, physisorption and coagulation	3 days	–	*Divalent cations:* bridging effect *Trivalent ions:* coagulation and adsorption	(Wang et al., 2015a)
Electrical degradation	PFOA	Lab	62 ng/L	*Electrodes:* boron-doped diamond (BDD) anode and Fe10MnC cathode	Cathodic electro-Fenton and anodic oxidation	4 h	96	–	(Wang et al., 2019b)
Algae and aerobic degradation	PFHxPA, PFOPA and PFDPA	Lab	100 µg/L	Biodegradation: Desmodesmus subspicatus and aerobic microorganisms in wastewater effluent *Pre-treatment of WW:* ultrasonication and centrifugation	–	> 10 days	0	–	(Llorca et al., 2018)
Photocatalysis	PFOA	Lab	30 mg/L	*Catalysts:* titanium dioxide (TiO_2), Indium Oxide (In_2O_3) and Gallium Oxide (Ga_2O_3); UV light source: UVB *Pre-treatment of WW:* filtration	In_2O_3 and Ga_2O_3: direct hole oxidation TiO_2: indirect oxidation of hole from OH radicals	180 min	0 to 23	–	(Lopes da Silva et al., 2017)
Nanofiltration (NF) and electrochemical degradation of NF concentrate	PFHxA	Lab	60 to 200 mg/L and 870 mg/L	*Filter:* DowFilm NF270 membrane (pressure: 20 bar) *Electrodes:* bipolar boron doped diamond electrode *Current density:* 50 A/m^2	Diffusion and electrochemical oxidation	1 h total recirculation and 1.5 h	99 and 98	–	(Soriano et al., 2017)
Aerosol based separation	6:2 FTSA	Lab	8.5 mg/L	*Design:* fritted glass set-up; pore diameter - 16-40 µm, solution: 0.2 M H_2SO_4 *Pre-treatment of WW:* LLE**	Transport of PFAS to the surface with the help of air bubbles generated in aqueous solution followed by the formation of PFAS enriched aerosols and their subsequent removal	< 1 h	~ 100	Higher ionic strength led to higher removal of 6:2 FTSA	(Ebersbach et al., 2016)

* SPE: Solid Phase Extraction
** LLE: Liquid-Liquid Extraction

(Fuertes et al., 2017). To date, the knowledge regarding the emission of volatile and/or semi-volatile PFAS from WWTPs into the atmosphere and the impact of various treatment processes is limited. Due to the complex wastewater matrix, biological and physicochemical reactions, and the unique properties of PFAS, WWTPs directly or indirectly serve as either source or sink for various PFAS (Chen et al., 2018). Hence, the efficiency of different treatment processes in removing PFAS and their formation in WWTPs needs further study.

4.1.2. Adsorption

In general, adsorption and biodegradation are the two processes responsible for the removal of contaminants from the aqueous phase in WWTPs. However, PFAS (long, short, ultra-short chain and other new alternate PFAS) are least susceptible to biodegradation due to the strong C-F bond (513 ± 10 kJ mol^{-1}) (Rodríguez-Varela et al., 2021), and hence adsorption onto sludge is the primary mechanism for their removal from the aqueous phase in WWTPs (Kibambe et al., 2020). The distribution coefficient (K_d) is an important parameter to evaluate the distribution of PFAS between sludge (adsorption onto sludge) and wastewater (dissolved phase) during wastewater treatment (Li et al., 2021). Factors like pH and composition of the wastewater and the structure of PFAS influences the partitioning of PFAS (Wang et al., 2020b). For example, with a decrease in pH (9 to 2), the PFAS partitioning to sludge intensified, resulting in increased sorption onto sludge in acidic conditions (Ebrahimi et al., 2021). The chemical structure of PFAS, especially the length of the carbon chain, dictates the partition behaviour of that PFAS between liquid and solid phases (Mussabek et al., 2019). For instance, the hexagonal ring structure of perfluoroethylcyclohexane sulfonate was associated with enhanced adsorption onto sediments (Liu et al., 2019b) while the ether bonds present in ADONA was detrimental for its adsorption onto suspended particles (Sun et al., 2016).

Studies suggest short-chain PFAS tend to partition to aqueous phase while long-chain PFAS partitions to the solid phase in the aquatic environment (Coggan et al., 2019b). A similar trend was observed in WWTPs where partitioning of PFAAs to sludge increased with an increase in chain length (Li et al., 2021, Mussabek et al., 2019, Coggan et al., 2019b, Pan et al., 2016) owing to the strong affinity of hydrophobic compounds to organic matter (Li et al., 2021). Moreover, the type of functional group in a PFAAs also influences its adsorption onto sludge. Wang et al. (2015a) found the K_d of PFHxS to be higher than that of PFHpA, suggesting a stronger affinity of more acidic sulfonate group towards organic content in the sludge than that of carboxylate groups (Arvaniti et al., 2014, Chen et al., 2012). Further studies utilizing K_d for understanding the effect of PFAS structure on its adsorption onto sludge could generate new insights. Very few studies have investigated the influence of PFAS headgroup charge (anionic, cationic, and zwitterionic) on its partitioning at liquid-liquid and solid-liquid interfaces (Ebrahimi et al., 2021, Xiao et al., 2019, Brusseau & Van Glubt, 2019). While most PFAS being anionic are partitioned to the sludge through hydrophobic and/or London−van der Waals interactions, cationic and zwitterionic compounds may be electrostatically attracted to negatively charged sites in the sludge along with other forces mentioned above (Xiao et al., 2019). The organic and inorganic matter present in wastewater is also crucial for predicting the partitioning behaviour of PFAS (Ebrahimi et al., 2021) in wastewater-sludge system which is discussed in later sections (4.2.1.1 and 4.2.1.2). As controlling the PFAS loads to influent may be gradual and is not a viable option at present, optimization of the process parameters, such as maintenance of lower pH and monitoring and regulating organic and inorganic fractions in wastewater to enhance PFAS partitioning to sludge, could be beneficial.

The partition behaviour of PFAS between water and solids (e.g. sediments, suspended particles, and soils) have been studied more in coastal, surface and groundwater systems (Wang et al., 2020b, Lee et al., 2020, Liu et al., 2019ab, Mussabek et al., 2019, Li et al., 2019, Dong et al., 2018) in comparison to wastewater and sludge in WWTPs (Ebrahimi et al., 2021, Li et al., 2021, Wang et al., 2015a, Coggan et al., 2019b). In addition, most of the previous studies have focused on the two legacy PFAS, i.e., PFOA and PFOS (Liu et al., 2019b). Insights on the partition behaviour of short/ultra-short chain and precursor PFAS between sludge-wastewater and effects of treatment type on their partition are insufficient. Further studies on K_d of sludge-wastewater systems can provide a better understanding of the fate, bioaccumulation, adsorption mechanism and transport of PFAS during and after treatment.

4.1.3. Transformation

Biological remediation techniques (e.g. aerobic/anaerobic digestion, biological filtration) applied during wastewater treatment influence the transformation of PFAS precursors in WWTPs, which ultimately affects the concentrations of PFAS in effluents (Chen et al., 2012). While typical WWTP may remove longer chain PFAAs from wastewater as a result of their adsorption onto sludge, shorter chain PFAS are formed during treatment (Houtz et al., 2016, Guerra et al., 2014). Short-chain PFAS are more persistent than their long-chain counterparts, resulting in longer-term occurrence (Li et al., 2020). Therefore, future studies should focus on the effective remediation of shorter chain PFAS from wastewaters using cost-effective, robust techniques with practical applicability.

Numerous studies have reported the formation of PFAAs after secondary stage (especially, anaerobic/anoxic/oxic oxidation) treatment (Table 3) (Wang et al., 2019a, Zhou et al., 2019, Houtz et al., 2018, Chen et al., 2012). FTOHs and fluorotelomer sulfonates (FTS) constitute a significant class of PFAS precursor compounds that can degrade into perfluoroalkyl acids (PFAAs) ($C \geq 4$), when undergoing aerobic and anaerobic treatment (Hamid et al., 2020a, Hamid et al., 2020b, Zhou et al., 2019, Coggan et al., 2019b, Chen et al., 2018, Hamid et al., 2018, Liu et al., 2015). Biodegradation of FTOHs to PFCAs through microbes has been well studied in laboratories (Chen et al., 2017b). Fluorotelomer carboxylic acids (FTCAs) and unsaturated fluorotelomer carboxylic acids (FTCUAs) are other intermediary compounds derived from fluorotelomer precursors of PFAS, which can eventually lead to the formation of PFAAs as end products (Hamid et al., 2018). The transformation of PFAS precursors to product PFAS in WWTPs is shown in Table 5.

While many studies have revealed the transformation of PFAS precursors into perfluoroalkyl carboxylic acids (PFCAs), $C \geq 4$, the ability of precursors to transform into ultrashort-chain PFCAs (C2-C3) remains unexplored. A few studies suggested that ultrashort-chain PFCAs (e.g., trifluoroacetic acid (TFA) and perfluoropropionic acid (PFPrA)) are usually generated at fluoropolymer manufacturing industries (Chen et al., 2019), but overall knowledge regarding the degradation of ultra-short chain PFAS from their respective precursors in WWTPs is limited.

A significant rise in the levels of PFAS for WWTPs applying biological deodorization to raw influent (Wang et al., 2019b) enunciates the need for evaluating the effect of specific microorganisms used for deodorization methods which caused ~70 % increase in the concentrations of PFAS. Aerobic treatment also has the potential to degrade PFAS precursors (Zhang et al., 2013b), which was confirmed by Guerra et al. (2014), who observed high levels of PFAAs in the secondary sludge of a WWTP adopting aerobic oxidation. Zhang et al., (2015) also reported a significant increase of PFAS in the final effluent of a WWTP using sequencing batch reactor system. Overall, PFAS precursors' transformation to PFAAs was prominent in WWTPs applying biological treatment methods

Table 5
Transformation of PFAS precursors to PFAS.

Product	Transformation	Matrix	Precursors	References
TFA, $C_2HF_3O_2$	Biochemical treatment	Landfill Leachate	Unknown C2–C4 PFAA precursors	(Wang et al., 2020a)
	Aerobic biotransformation	Landfill soil	6:2 FTOH	(Sun et al., 2020)
	Thermolysis (~400°C to ~650°C)	Landfill Leachate	PVDF-HFP, PVDF-CTFE and PTFE	(Cui et al., 2019)
	Reaction with $^{\bullet}$OH radicals	Industrial by-product	Semifluorinated alkanes [SFAs, F(CF2) n(CH2) mH]	(Wang et al., 2014b)
PFPrA, $C_3HF_5O_2$	Biochemical treatment	Landfill Leachate	Unknown C2–C4 PFAA-precursors	(Wang et al., 2020a)
PFBA, $C_4HF_7O_2$	Aerobic and anaerobic biotransformation	WWTP sludge, landfill leachate	8:2 FTOH	(Hamid et al., 2020a, Li et al., 2018, Yu et al., 2018, Yu et al., 2016)
	Aerobic biotransformation	WWTP sludge, landfill leachate	6:2 FTOH	(Sun et al., 2020, Yu et al., 2018)
	Aerobic biotransformation	Landfill Leachate	6:2 FTS	(Hamid et al., 2020a, Hamid et al., 2020b)
	Hydrothermal liquefaction (HTL) enhanced biotransformation	WWTP sludge	8:2 FTUCA, 7:3 FTCA and 8:2 FTS	(Yu et al., 2020)
	Aerobic biotransformation	Landfill Leachate	4:2 FTOH, 2-(trifluoromethyl) acrylic acid (TFMAA)	(Sun et al., 2020)
	Biochemical treatment	Landfill Leachate	Unknown C2–C4 PFAA-precursors	(Wang et al., 2020a)
	Photodegradation	WWTP effluent	2H,2H,3H,3H Perfluoroundecanoic acid (2H,2H,3H,3H-PFUnDA)	(Liu et al., 2019a)
	Thermolysis of persulfate ($S_2O_8^{2-}$)	WWTP effluents	8:2 FTCA	(Wang et al., 2018a)
	Reaction with $^{\bullet}$OH radicals, photodegradation and/or biodegradation	WWTP sludge	6:2 fluorotelomer based and PBSF based alternatives	(Wang et al., 2013)
	Biotransformation	WWTP sludge	5:3 acid (C5F11CH2CH2CO2H)	(Wang et al., 2012)
PFPeA, $C_5HF_9O_2$	Aerobic biotransformation	Landfill leachate	6:2 FTS	(Hamid et al., 2020b)
	Aerobic biotransformation	Landfill Leachate	4:2 FTOH	(Sun et al., 2020)
	Photodegradation	WWTP effluent	2H,2H,3H,3H-PFUnDA	(Liu et al., 2019a)
	Aerobic and anaerobic biotransformation	WWTP sludge	8:2 FTOH	(Li et al., 2018, Yu et al., 2018, Yu et al., 2016)
	Thermolysis of persulfate ($S_2O_8^{2-}$)	WWTP effluents	8:2 FTCA	(Wang et al., 2018a)
	Photodegradation	WWTP effluent	6:2 FTUCA	(Anumol et al., 2016)
	Biotransformation	WWTP effluent	POSF	(Wang et al., 2014a, Wang et al., 2014b)
	Biotransformation	WWTP sludge	5:3 acid (C5F11CH2CH2CO2H)	(Wang et al., 2012)
PFHxA, $C_6HF_{11}O_2$	Aerobic and anaerobic biotransformation	WWTP sludge, landfill leachate	8:2 FTOH	(Hamid et al., 2020a, Yu et al., 2018, Yu et al., 2016, Zhang et al., 2013b)
	Aerobic biotransformation	WWTP sludge, landfill leachate	6:2 FTOH	(Sun et al., 2020, Yu et al., 2018, Lewis et al., 2016, Zhang et al., 2013b)
	Aerobic biotransformation	Landfill Leachate	6:2 FTS	(Hamid et al., 2020a, Hamid et al., 2020b)
	Hydrothermal liquefaction (HTL) enhanced biotransformation and ozonation	WWTP sludge and WWTP effluent	8:2 FTUCA	(Yu et al., 2020, Glover et al., 2018)
	Hydrothermal liquefaction (HTL) enhanced biotransformation	WWTP sludge	7:3 FTCA and 8:2 FTS	(Yu et al., 2020)
	Photodegradation	WWTP effluent	2H,2H,3H,3H-PFUnDA	(Liu et al., 2019a)
	Degradation	Air	Fluorotelomer acrylate polymers	(Seo et al., 2019)
	Thermolysis of persulfate ($S_2O_8^{2-}$)	WWTP effluent	C8 PFAA precursors, 8:2 FTCAs	(Wang et al., 2018a, Ye et al., 2014)
	Ozonation	WWTP effluent	6:2 FTUCA	(Glover et al., 2018)
	Biotransformation	–	POSF	(Wang et al., 2014a)
	Reaction with $^{\bullet}$OH radicals, photodegradation and/or biodegradation	Biota	6:2 fluorotelomer based and PBSF based alternatives	(Wang et al., 2013)

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Table 5 (*continued*)

Product	Transformation	Matrix	Precursors	References
PFHpA, $C_7HF_{13}O_2$	Aerobic biotransformation	WWTP sludge, landfill leachate	8:2 FTOH	(Hamid et al., 2020a, Yu et al., 2018, Yu et al., 2016)
	Aerobic biotransformation	Landfill leachate	6:2 FTS	(Hamid et al., 2020b)
	Photodegradation	WWTP effluent	2H,2H,3H,3H-PFUnDA	(Liu et al., 2019a)
	Ozonation	WWTP effluent	6:2 FTUCA and8:2 FTUCA	(Glover et al., 2018)
	Biotransformation	–	POSF	(Wang et al., 2014a, Wang et al., 2014b)
	Biotransformation	WWTP sludge	PFCAs, 7:3 acids (C7F15CH2CH2CO2H)	(Wang et al., 2012)
PFOA, $C_8HF_{15}O_2$	Aerobic and anaerobic biotransformation	WWTP sludge, landfill leachate	8:2 FTOH	(Hamid et al., 2020a, Li et al., 2018, Yu et al., 2018, Yu et al., 2016, Zhang et al., 2013b, Gauthier & Mabury, 2005)
	Aerobic biotransformation	Landfill leachate	6:2 FTS	(Hamid et al., 2020b)
	Oxidation	WWTP effluent	8:2 FTCA	(Wang et al., 2018a, Ye et al., 2014)
	Anaerobic biotransformation		6:2 FTOH	(Zhang et al., 2013b)
PFNA, $C_9HF_{17}O_2$	Aerobic biotransformation	WWTP sludge	8:2 FTOH	(Yu et al., 2018, Yu et al., 2016)
	α-oxidation	WWTP effluent	8:2 FTCA	(Wang et al., 2018a)
6:2 FTCA, 6:2 FTUCA	Aerobic and anaerobic biotransformation	WWTP sludge, landfill leachate	6:2 FTOH	(Lewis et al., 2016, Zhang et al., 2013b)
8:2 FTCA, 8:2 FTUCA	Aerobic and anaerobic biotransformation	WWTP sludge	8:2 FTOH	(Zhang et al., 2013b)
5:3 acid (C5F11CH2CH2CO2H)	Aerobic and anaerobic biotransformation	WWTP sludge, landfill leachate	6:2 FTOH	(Lewis et al., 2016, Zhang et al., 2013b, Wang et al., 2012)
7:3 acid (C7F15CH2CH2CO2H)	Aerobic and anaerobic biotransformation	WWTP sludge	8:2 FTOH	(Zhang et al., 2013b, Wang et al., 2012)
PFBS, $C_4F_9SO_3H$	–	WWTP effluent	FOSA	(Wang et al., 2018a)
	Oxidation	WWTP sludge	NMeFOSA, FOSA, and diperfluorooctane sulfonamidoethanol-based phosphate	(Ye et al., 2014)
	Chemical degradation	Industrial by-product	PBSF	(Wang et al., 2013)
PFOS, $C_8HF_{17}SO_3H$	Biotransformation	WWTP sludge	FOSA	(Land et al., 2018)
	Oxidation	WWTP effluent	NMeFOSA, FOSA and diperfluorooctane sulfonamidoethanol-based phosphate	(Ye et al., 2014)
	Aerobic and anaerobic biotransformation	WWTP sludge	N-ethyl perfluorooctane sulfonamido ethanol (N-EtFOSE)	(Rhoads et al., 2008)
5:2 FTOH	Biotransformation	WWTP sludge	6:2 FTOH	(Lewis et al., 2016)

(Houtz et al., 2018), which are likely to break only the C-C bond resulting in the formation of shorter chain PFAS (Gagliano et al., 2020). The weaker adsorption of PFAS onto primary sludge than secondary sludge could be due to the hindrance from background ions (see section 4.2.1.2). The enhanced transformation of PFAS precursors during biological treatment could also be responsible for the higher detection of PFAS in the final effluent after secondary treatment (Zhang et al., 2015). Microorganisms could gain energy for their growth from thermodynamically favoured reductive defluorination (Arvaniti and Stasinakis, 2015) provided strong electron donors are available to facilitate such reactions. Thus, future studies should focus on the performance of anaerobic microorganisms capable of degrading PFAS.

The efficiency of biological filters in removing PFAS from wastewater is dependent on various parameters such as sludge age and thickness. Biological filters are inefficient in removing short-chain PFAS ($C \leq 6$) from wastewater. Observed removal efficiencies of about 60% for PFOA and $< 20\%$ for PFAAs ($C \leq 6$) by biological filtration confirmed the inability of microbial degradation to remove shorter chain PFAS (Kibambe et al., 2020). When thickened, the biofilm can act as both aerobic (outer layer) and anaerobic (inner layer), which could be possibly influencing PFAS rejection (Kibambe et al., 2020). Biological activated carbon filters using granular activated carbon (GAC) perform more efficiently than biological filters in PFAAs removal from wastewater due to the strong ability of GAC to adsorb PFAAs ($C > 7$) (Pramanik et al., 2017). Though biological filters/biological activated carbon filters showed considerable removal for PFAAs (~50 %) (Kibambe et al., 2020, Wang et al., 2019a), activated sludge treatment outperformed both with $\geq$ 90 % removal efficiency (Kibambe et al., 2020, Castiglioni et al., 2015).

Further investigation to understand the effects of different biological treatments on the fate of PFAS precursors, long-chain, short-chain and ultra-short chain PFAS are necessary for drawing tangible inferences. In addition to WWTPs, landfills are also important sites where PFAS precursors can transform into PFAS. The biotic conditions prevailing at landfills and subsequent biochemical treatment of landfill leachates are favourable for this transformation to occur (Hamid et al., 2020a, Sun et al., 2020, Wang et al., 2020a). Discharging leachates into WWTPs and dumping WWTP sludge in landfills are common practices, which may contribute towards PFAS levels in both media (Seo et al., 2019, Hamid et al., 2018). To our knowledge, there are no insights available on this interrelation to date, which requires further investigation.

4.2. Lab-scale investigations

4.2.1. Adsorption

Among all the processes explored for the remediation of PFAS from wastewater, adsorption is the most studied at the laboratory scale (Table 4). Adsorption using ion exchange resins has shown promising results with $\geq$ 99 % removal efficiency for PFAS along with dissolved organics (>90%) (Dixit et al., 2021, Dixit et al., 2020a). However, results obtained at high initial concentrations (5–1,000 mg/L) of PFAS in synthetic and spiked wastewaters are not necessarily applicable to lower and more realistic concentrations (<1,000 ng/L). The formation of nitrosamine precursors during regenerating amine-based anion exchange (IX) resins documented earlier is also concerning (Flowers and Singer, 2013). Biochar, on the other hand, successfully removed PFAS at a lower concentration (1,000 ng/L) (Inyang and Dickenson, 2017). Biochar filtration post-secondary treatment (after removal of most of the organic matter) could facilitate the removal of PFAS through adsorption (Dalahmeh et al., 2019). Modification of biochar surfaces to improve PFAS adsorption, especially for short-chain PFAS, should also be further studied. Other novel adsorbents, such as hydrogel polymers and magnetic fluorinated adsorbents, showed excellent results in minimum time (2–4 h) (Ateia et al., 2019a, Du et al., 2017). However, the applicability of these adsorbents at large scale treatment facilities with lower concentrations of PFAS and complex background matrix needs to be confirmed.

4.2.1. 1 Role of PFAS carbon chain length

To date, the effect of alkyl chain length on PFAS adsorption have been investigated mostly with activated carbon, biochars, and resins in aqueous and/or synthetic wastewater solutions (Gagliano et al., 2020). Hydrophobic interactions have been found to be the most prominent mechanism responsible for the adsorption of PFAS onto various adsorbents (Ateia et al., 2019a, Dalahmeh et al., 2019, Du et al., 2014). Hydrophobicity of perfluoroalkyl compounds, in turn, depends on the length of the carbon chain; with an increase in the length of the carbon chain from C5-C15, the hydrophobicity and, hence, their adsorption onto sludge increases (Zhang et al., 2013a). Shorter chain PFAS, being less hydrophobic and more water soluble, are less likely to get adsorbed onto the sludge in WWTPs (Brendel et al., 2018, Zhang et al., 2013a). Hence, the length of the carbon chain of a PFAS plays a crucial role in the adsorption behaviour of that compound. As wastewater has a different composition (organic and inorganic matter) from other aquatic matrices (e.g. surface, coastal and/or groundwater) (Phong Vo et al., 2020), PFAS interacts differently with adsorbents in wastewater. Hence, more studies are required to understand the role of chain length during adsorption of PFAS onto sludge in wastewater matrix. The interaction by short and long-chain PFAS with sludge has been studied using activated carbons and polymers, which suggest short-chain PFAS interact with organic matter through electrostatic repulsions (Dalahmeh et al., 2019) and pore blockage, while long-chain PFAS tend to have hydrophobic interactions (Gagliano et al., 2020). Since organic matter constitutes a major fraction of sludge, the adsorption mechanism between short/ultra-short chain PFAS and sludge could be studied further, as discussed in 4.1.2.

4.2.1. 2 Role of background matrices

Earlier reviews have discussed in detail the interaction between inorganic and organic ions and organic contaminants (Kah et al., 2017). The organic matter (e.g., proteins, lipids, carboxylic and amino acids, natural and/or dissolved organic matter (Gagliano et al., 2020) present in wastewater (Ebrahimi et al., 2021, Liu et al., 2020) impact the adsorption of PFAS onto sludge in WWTPs (Zhang et al., 2019). Only a few studies have investigated the role of organic ions on PFAS removal, among which most are concentrated on the effect of humic and fulvic acids (Banks et al., 2020), and least is known about the impact of dissolved organic matter on PFAS removal from wastewater. (Dixit et al., 2020b) found organic matter (natural organic matter, humic acid and fulvic acid from surface water) had a negative impact (~ 55% decrease in adsorption capacity) on the adsorption of PFAS using ion exchange (IX) resins as organic ions caused blockage of pores through diffusion and competed with PFAS for adsorption sites (Dixit et al., 2021, Dixit et al., 2019). On the contrary, Wang et al. (2015a) found that the adsorption of PFAS to sludge reduced significantly after the removal of organic matter from the sludge through thermal treatment. Hence, the authors suggested that PFAS has a higher affinity for sludge from secondary treatment with higher biomass. Zhang et al. (2013a) observed that protein in the sludge had a positive effect on the adsorption of PFAS and deduced that certain amide groups present in proteins could possibly be aiding the adsorption of PFAS onto sludge. While another study found protein

in the wastewater influent led to lower removal of PFOS and PFOA (Pramanik et al., 2017).

Similar to organic ions, inorganic ions can have both positive and negative effects on the adsorption of PFAS (Kah et al., 2020, Banks et al., 2020, Zhang et al., 2019). Divalent cations, such as Ca^{2+} and Mg^{2+}, can form cation-bridge between the negatively charged PFAS and sludge, thereby improving adsorption (Zhang et al., 2013a). In contrast, monovalent cations present in wastewater, such as Na^{+}, have been found to hinder the adsorption of PFAS (Arvaniti et al., 2014). Anions, such as SO_4^{2-}, PO_4^{3-}, NO_3^{-}, Cl^{-}, and CrO_4^{-}, strongly compete with PFAS for adsorption sites (Dixit et al., 2021, Ateia et al., 2019a), thereby interfering with the adsorption of PFAS on sludge (Deng et al., 2010). Ateia et al. (2019a) observed a rapid decrease (10–20 %) in the adsorption of short-chain PFCAs with an increase in anion concentrations. Whereas, high background anions in the real wastewater matrix caused the salting-out effect, which compensated the hindrance by SO_4^{2-} by reducing the solubility of PFAS (Du et al., 2015).

Such conflicting observations in terms of the effect of organic and inorganic ions on the treatment of PFAS in wastewater (Table 4) generates scope for more research on the impacts of different types of organic compounds and inorganic ions (cations and anions) on the removal of short-chain PFAS and precursors at environmentally relevant concentrations (<1,000 ng/L) (Ateia et al., 2019a). Further, the effect of co-existing PFAS in the presence of background ions is yet to be explored. Indeed, the rate of adsorption decreased significantly when three PFCAs (C6, C7 and C8) were treated simultaneously in wastewater with resins, which was attributed to the competitive adsorption between short and long-chain PFAS (Du et al., 2015). To date, most studies have investigated the effects of organic and inorganic ions separately on individual PFAS in aqueous solutions at concentrations not relevant to the environment. Outcomes of the simultaneous effects of various organic and inorganic ions on the removal of co-existing short-chain (<C4) PFAS and/or precursors would be relevant for real-world conditions. Further, the chemical interactions between various ions and PFAS, leading to the differences in adsorption behaviour, could provide useful mechanistic insights.

4.2.2. Other treatment techniques

Advanced treatment processes like adsorption, electrochemical degradation, nanofiltration, and reverse osmosis (Dixit et al., 2020a, Ateia et al., 2019a, Wang et al., 2019b, Soriano et al., 2017) are comparatively more effective than conventional processes (Kibambe et al., 2020, Wang et al., 2019a) in terms of PFAS removal. For example, a WWTP based on conventional treatment showed negative removal (-27 %) for PFAS, whereas a WWTP using advanced treatment (reverse osmosis) achieved 86 % removal (Tabtong et al., 2015). Similarly, a membrane bioreactor system was found to be more effective in removing long-chain PFCAs than conventional activated sludge treatment (Pan et al., 2016). However, it is noteworthy that some treatments like ozonation, UV, chlorination and sand filtration have been found ineffective in removing PFAS (Gagliano et al., 2020, Pan et al., 2016).

Nanofiltration (NF) has been efficient (>98%) in removing PFAS along with dissolved organic carbon and mineral hardness from drinking water (Franke et al., 2021). The efficiency of NF in removing PFAS while simultaneously achieving other water quality targets in wastewater needs validation by further studies. A study that conducted NF followed by electrochemical degradation could achieve ~ 99 % removal of C6 PFAA in less than 3 h for PFAS concentrations <1,000 mg/L (Soriano et al., 2017). Similarly, electrochemical treatment processes using electrodes like boron-doped diamond anode and stainless-steel cathode could achieve >95% degradation of PFAS with high and low initial concentrations of 1,642 µg/L (Gomez-Ruiz et al., 2017) and 62 ng/L (Wang et al., 2019b) within 10 h. Hence, the application of electrochemistry for degradation of short-chain PFAS and PFAS precursors at lower concentrations could be explored more rather than combining it with other techniques.

Treatment trains should involve processes that are not efficient individually, while their combined efficiency is higher in terms of PFAS removal along with the ease of operation and cost-effectiveness. For instance, ozonation (Wang et al., 2019b) and photocatalysis (Lopes da Silva et al., 2017) were ineffective in removing PFAAs from wastewater matrix individually, while the combination of ozonation and photocatalysis led to the degradation of PFAS as the reactions between $^{\bullet}OH$ and PFAAs precursors can initiate degradation reactions forming recalcitrant shorter chain PFAS (Wang et al., 2019a, Wang et al., 2018a, Huang et al., 2016). Similarly, efficient treatment (>98%) of PFAS from drinking water using NF followed by subsequent adsorption of PFAS (85–99%) from NF concentrate (Franke et al., 2021) calls for more study using such treatment train practices for effective remediation of PFAS in wastewaters.

Another unique aerosol-based separation technique for remediation of precursor 6:2 FTSA achieved ~100 % removal within 1 h (Ebersbach et al., 2016). However, the initial concentration of the spiked PFAS in wastewater was unrealistically high (~ 10 mg/L), which is one common limitation in experimental studies published to date (Table 4). More experimental data carried out at realistic concentrations (<1,000 ng/L) (Li et al., 2020) and involving short-chain PFAS (C < 4), and matrix effects are needed. The knowledge regarding the treatment of PFAS precursors from wastewater at environmentally relevant concentrations is also limited.

5. Conclusions and research recommendations

5.1. Conclusions

The lack of monitoring data from many parts of the world limits our understanding of the global distribution of PFAS. The inaccessibility of advanced analytical facilities and the need to meet more pressing wastewater quality issues in many developing countries could be responsible for the dearth of PFAS monitoring studies in these regions. Ultra-short chain PFAS, precursors of PFAS, and new generation replacements of conventional PFAS are understudied, which could be attributed to their analytical method limitations and the complexity of TOPA and non-target analysis. The relative ease of PFAS analysis in effluents, having lower background interference than in influents, could be responsible for fewer studies of PFAS in influents than effluents. The partitioning of longer chain PFAS to sludge and the transformation of precursors to shorter chain PFAAs during biological treatment influence the concentrations of PFAS in the sludge and the effluents. The release of volatile compounds is a possible route for the transfer of precursors from wastewater to air and remains unexplored currently. While most studies monitored the removal of PFAS from the aqueous phase (effluents), their fate after partitioning to sludge and/or biosolids is relatively unexplored, which compels us to think if the mere shifting of PFAS loads from one phase to the other has been misunderstood as removal to date.

5.2. Research recommendations

- The limited information about PFAS levels in developing and least developed countries necessitates future studies in these regions.

- To date, no study has critically reviewed the sampling procedures despite it being critical in monitoring PFAS, considering the high risk of background contamination.
- Monitoring ultra-short chain PFAS, precursors of PFAS, and new generation PFAS in wastewater and studying their transformation pathways is required as some of these compounds can exhibit similar persistence and bioaccumulating capabilities as their longer-chain counterparts.
- More monitoring data for PFAS in the influents could add to the existing knowledge about the sources of short-chain PFAS and newly discovered PFAS precursors in WWTPs.
- While adequate studies reveal PFAS precursors' transformation into perfluoroalkyl carboxylic acids (PFCAs), C $\geq$ 4, the likelihood of precursors to transform into ultrashort-chain PFCAs (C2-C3) remains unexplored.
- Investigating the degradability (e.g., biodegradation, photodegradation, ozonation, etc.) of synthetic precursor compounds at a laboratory scale could help better understand the mechanisms responsible for their transformations and effective treatments.
- The effects of different biological treatments, especially the effects of aerobic and anaerobic microbes, on PFAS precursors' transformation to PFAAs need to be explored.
- The reported rise in the levels of PFAS after the application of biological deodorization in WWTPs necessitates studying the effects of specific microorganisms used for the deodorization process.
- More mass balance studies of PFAS and their precursors in WWTPs, accounting for masses in all three phases, are advised as their removal reported in WWTPs so far could be simply the result of a transfer of mass load from one phase to another within a WWTP.
- The common practice of discharging landfill leachates into WWTPs and dumping WWTP sludge and/or biosolids in landfills necessitates studying the fate of PFAS throughout such interconnected systems.
- The combined effects of various organic (e.g. humic and fulvic substances, lipids and proteins, and natural and/or dissolved organic matter) and inorganic ions (cations, anions, and zwitterions) on the removal of co-existing short-chain PFAS and precursors need further studies due to the real-world implications.

Declaration of Competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Air concentrations and particle–gas partitioning of polyfluoroalkyl compounds at a wastewater treatment plant

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Environmental context. Polyfluoroalkyl compounds, widely used chemicals in consumer and industrial products, are global pollutants in the environment. Transport mechanisms and environmental pathways of these compounds, however, are not yet fully understood. We show that a wastewater treatment plant can be an important source for polyfluoroalkyl compounds to the atmosphere where they have the potential to be transported long distances.

Abstract. An air sampling campaign was conducted at a wastewater treatment plant (WWTP) to investigate air concentrations and particle–gas partitioning of polyfluoroalkyl compounds (PFCs). Samples were collected at an aeration tank and a secondary clarifier using both active high volume samplers and passive samplers comprising sorbent-impregnated polyurethane foam (SIP) disks. Water to air transport of PFCs was believed to be enhanced at the aeration tank owing to aerosol-mediated transport caused by surface turbulence induced by aeration. Mean air concentrations of target PFCs at the aeration tank were enriched relative to the secondary clarifier by factors of ~19, ~4 and ~3 for ∑fluorotelomer alcohols (FTOHs) (11 000 *v.* 590 pg m^{-3}), ∑perfluorooctane sulfonamides & perfluorooctane sulfonamidoethanols (FOSAs & FOSEs) (120 *v.* 30 pg m^{-3}) and ∑perfluoroalkyl carboxylates & perfluoroalkyl sulfonates (PFCAs & PFSAs) (4000 *v.* 1300 pg m^{-3}) respectively. The particle associated fraction in the atmosphere increased with increasing chain length for PFCAs (from 60 to 100%) and PFSAs were predominantly bound to particles (~98%). Lower fractions on particles were found for FTOHs (~3%), FOSAs (~30%) and FOSEs (~40%). The comparison of the active and passive air sampling showed good agreement.

Additional keywords: atmosphere, passive air sampler, PFC, PFOA, PFOS, WWTP.

Introduction

Polyfluoroalkyl compounds (PFCs), such as perfluorooalkyl carboxylates (PFCAs) and perfluorooalkyl sulfonates (PFSAs), have been detected in a variety of environmental media and also in remote regions[1]; including rivers,[2] oceans,[3] the atmosphere,[4] wildlife,[1] and in humans.[5] PFCAs and PFSAs are persistent, bioaccumulative and toxic.[6–8] Fluorotelomer alcohols (FTOHs), perfluorooctane sulfonamides (FOSAs) and perfluorooctane sulfonamidoethanols (FOSEs) are precursors for PFCAs and PFSAs.[9–11] These precursors are volatile and can be transported in the atmosphere.[12–14]

There is still uncertainty about the origin of PFCs in the environment and the transport pathways of PFCAs and PFSAs are still under discussion. Wastewater treatment plants (WWTPs) are known to be point sources for PFCAs and PFSAs in rivers.[15,16] PFCs can be further transported via ocean currents to remote regions.[3] In terms of atmospheric pathways, the long-range transport of precursors can contribute to the occurrence of PFCs in remote regions.[17,18] However, the amounts from transport of PFCAs and PFSAs in ocean currents or from their precursor compounds in the atmosphere are not sufficient to fully explain the levels detected in remote regions.[13,17] Additionally, atmospheric PFCA and PFSA concentrations in urban areas cannot be explained by only considering the degradation of precursors.[19]

Recently, a laboratory experiment and a model have shown the transport of perfluorooctanoic acid (PFOA) in the gas-phase in its neutral form.[20,21] It was suggested that PFOA originated

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from aerosol-mediated transport from a water body into the atmosphere.[20] PFCAs and PFSAs have been detected in the particle-phase in the atmosphere; however, only a few studies have reported their presence in the gas-phase.[19,22]

The aim of this study was to generate more information on aerosol-mediated sources of PFCs by investigating the air concentrations and particle–gas partitioning of PFCs emitted at two locations in a WWTP – the aeration tank and the secondary clarifier. A secondary aim was to compare measurements conducted using the active high volume air sampler with time-integrated measurements using a sorbent-impregnated polyurethane foam (SIP) disk passive air sampler.

Experimental methods

Chemicals

The target analytes included 10 PFCAs (C_4–C_{12}, C_{14}), 4 PFSAs (C_4, C_6, C_8, C_{10}), 3 FTOHs (6 : 2, 8 : 2, 10 : 2 FTOH), 3 FOSAs, 2 FOSEs, 1 PFOSEA and 19 mass-labelled internal standards (see Tables A1 and A2 in the Accessory publication, available at http://www.publish.csiro.au/?act=view_file&file_id=EN10133_AC.pdf).

Sampling

Sampling took place at a WWTP in Ontario, Canada, during spring 2010. At this sampling site two sampling locations were chosen: one at the aeration tank, where activated sludge is added to the wastewater to remove organic materials; and the other at the secondary clarifier, where the sludge is allowed to settle, to be separated from the water. The main difference between the two sampling sites is that air is blown into the wastewater at the aeration tank, to create an aerobic environment for the microbes. This generates a turbulent and bubbling surface at the aeration tank whereas the surface of the secondary clarifier is relatively calm.

High volume air samples (~140 m^3 per sample) were collected for 24 h, twice per week over a period of 6 weeks at an aeration tank and a secondary clarifier at a WWTP. Glass fibre filters (GFFs) (Pall Corporation, Quebec, QC, Canada, Type A/E Glass 102-mm diameter baked at 250°C before sampling) were used to collect the particle-phase, whereas PUF/XAD/PUF cartridges (precleaned large PUF plug, Supelco, Oakville, ON, Canada, 7.6-cm length, 6-cm diameter, 15 g of XAD-2 (SupelcoTM-2), Supelco) were used for trapping gas-phase compounds. In addition, SIP disks were deployed in duplicate for 37 days during the same time period at both sampling sites to provide a time-integrated sample. To prepare SIP disks, precleaned PUF disks (diameter 14 cm, thickness 1.35 cm, surface area 365 cm^2, mass 4.4 g, volume 207 cm^3, density 0.0231 g cm^{-3}, Tisch Environmental, Cleves, OH, USA) were coated by dipping in a hexane and ground Amberlite XAD-4 slurry (styrene-divenylbenzene, Supelco) and then drying, according to the method described in Shoeib et al.[23]

The effective air volume (V_{air}) for passive samples was calculated using[24]:

$$V_{air} = K_{SIP-air} \times V_{SIP}\left[1 - \exp\left(-\frac{d \times k_a}{K_{SIP-air} \times D_{film}}\right)\right] \quad (1)$$

V_{SIP} is the volume of the SIP disks (2.10×10^{-4} m^3), d the deployment time (37 days), D_{film} the thickness of the SIP disks (5.67×10^{-3} m) and k_a the air-side mass-transfer coefficient (108 m day^{-1}) calculated as the ratio of the sampling rate and the area of the SIP disks (3.7×10^{-2} m^2). The sampling rate was determined during another study ($r = 4$ m^3 day^{-1}).[24] SIP–air partitioning coefficients ($K_{SIP-air}$) and the slope of octanol–air partitioning coefficients (K_{OA}) from the literature[23,25,26] were used for the determination of temperature dependent $K_{SIP-air}$ values (average temperature during sampling period 9°C). The sample volume for FTOHs, FOSAs and FOSEs ranged from 120 to 140 m^3.

For PFCAs and PFSAs, an effective air volume of 150 m^3 was calculated based on the duration of 37 days and a SIP disk sampling rate of 4 m^3 day^{-1} according to Genualdi et al.[24]

Field blank samples for all sample media were collected by exposing them for 1 min at the sampling site and then treated them like real samples. Total suspended particles (TSPs) were determined gravimetrically by weighing the GFFs before and after sampling and dividing the mass by the air sample volumes. Furthermore air temperature was measured at the sampling sites.

Extraction and analysis

The extraction method and analysis was similar to methods used elsewhere.[23,24] All samples were spiked with mass-labelled internal standards before extraction. The PUF/XAD/PUF cartridges and the SIP-disks were extracted using Soxhlet apparatus with petroleum ether (6 h, 240 mL, ~20–40 cycles) for FTOHs, FOSAs and FOSEs and thereafter with methanol (10–14 h, 240 mL, ~50–70 cycles) for PFCAs and PFSAs. The GFFs were extracted by sonication using dichloromethane for FTOHs, FOSAs and FOSEs (three times using 12 mL for 20 min and then combining extracts) and methanol for PFCAs and PFSAs (five times using 12 mL for 20 min and then combining extracts). The two fractions for each sample media were treated separately. All fractions were concentrated by rotary evaporation (Büchi, Flawil, Switzerland) and nitrogen blow down. The methanol fractions were cleaned with Envi-Carb.[27] The petroleum ether and dichloromethane fractions were applied to sodium sulfate columns for removing moisture. After clean up, 80% of the dichloromethane extract was combined with the corresponding methanol extract, because PFCAs and PFSAs were found to be partially extracted into dichloromethane. The methanol fractions were treated by adding 50% water and injections standards $^{13}C_8$-PFOS and $^{13}C_8$-PFOA, whereas Me_2FOSA was added as the injection standard to the other fractions.

Gas chromatography mass spectrometry (GC-MS) in positive chemical ionisation mode (PCI) (Agilent Technologies, Mississauga, ON, Canada, 7890 A GC system) was used for analysis of FTOHs, FOSAs and FOSEs (except for PFOSA). The separation of target compounds was performed on a DB-WAX column (30 m, 0.25-mm inner diameter, 0.25-μm film, J&W Scientific, Folsom, CA, USA). The injection volume was 2 μL and using splitless injection (200°C). The oven temperature program is given in the Accessory publication (Table A3). Helium was used as carrier gas at a flow rate of 1.3 mL min^{-1}. Methane was used as the reaction gas.

Instrumental analysis of PFCAs and PFSAs (including PFOSA) was performed using high pressure liquid chromatography (Agilent 1100 Series) tandem mass spectrometry (HPLC-MS/MS) (Applied Biosystems, Toronto, ON, Canada, 4000 QTRAP) in the electrospray negative ionisation mode at atmospheric pressure. For separation, a pre-column (C_8, 4-mm length, 2-mm diameter, Phenomenex, Torrance, CA, USA) and a Luna column (C_8 (2), 50-mm length, 2-mm

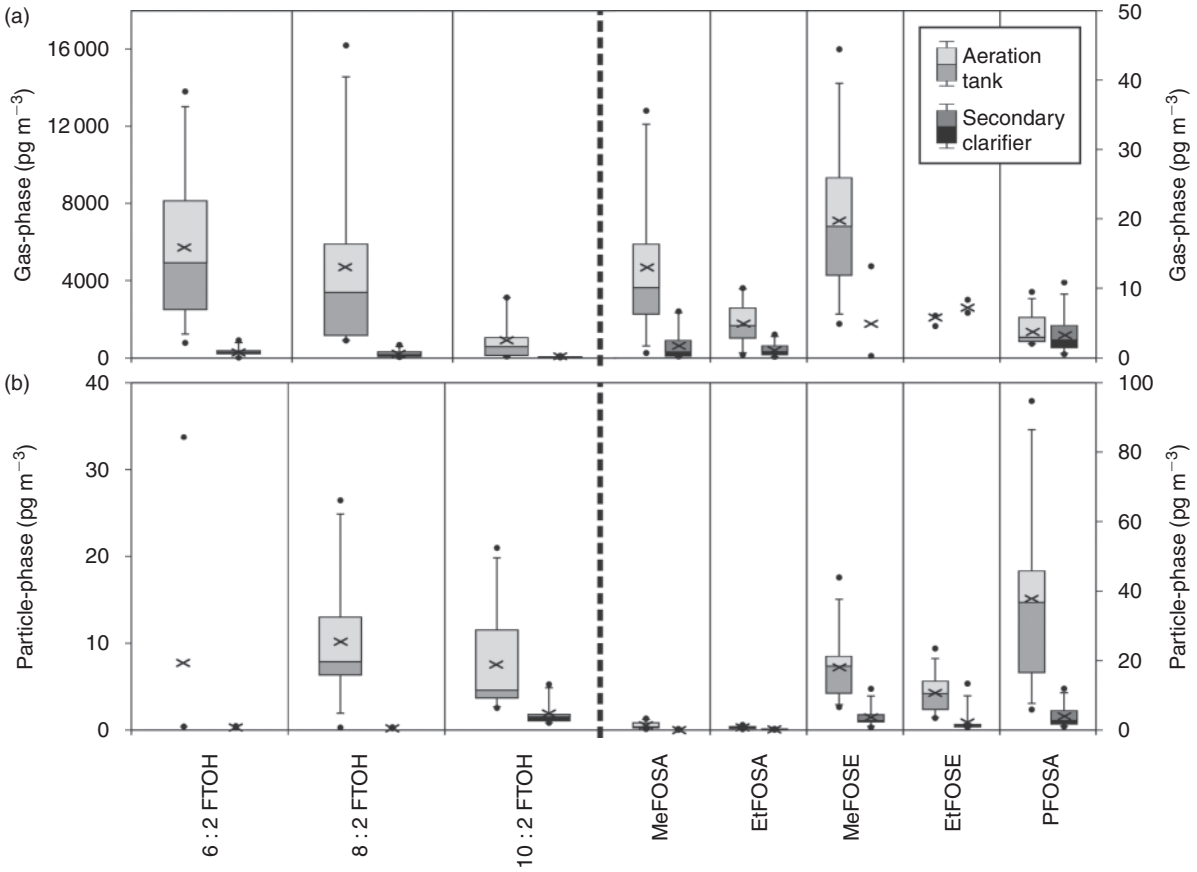


Fig. 1. Box-whisker plots for FTOH, FOSA and FOSE concentrations in the gas (a) and particle-phase (b) at the aeration tank and the secondary clarifier. The boxes show median concentrations and the 25th and 75th percentiles; 10th and 90th percentiles are indicated by the whiskers and the dots represent the minimum and maximum concentrations. The mean concentrations are indicated with an ×. If more then 25% of the data for one compound were below the IDL only minimum, maximum and mean values are shown.

diameter, 3-µm particle size, Phenomenex) was used. Methanol and water, each with 10-mM ammonium acetate, were used as the mobile phase. The flow was set to 0.25 µL min^{-1} and the gradient is given in the Accessory publication (Table A4). The injection volume was 25 µL.

Quantification was performed based on response factors of the target compounds and their corresponding internal standards. The ratio of both response factors was used for recovery correction. The calibration curves included eleven points for FTOHs (0.3–3800 ng mL^{-1}), seven points for FOSAs and FOSEs (0.12–115 ng mL^{-1}) and eight points for PFCAs and PFSAs (0.005–5.0 ng mL^{-1}). Instrument detection limits (IDLs) were calculated by extrapolating instrument response in blank samples to a concentration that would give a S/N value of three. A further limit of detection (LOD) calculated as three times the standard deviation (s.d.) of the blanks was used. Concentrations below the blank levels and below the IDL were set to half of the LOD for statistical analysis. In cases where the substitution for a particular chemical was required for more than 25% of the data set, only the mean value was presented in Figs 1 and 2 as the other statistical parameters are subject to bias by substituting a constant value.[28] Compounds not detected above the IDL in any of the samples were excluded from further investigations.

Prior to extraction, small punches (0.7 cm^2) of seven GFFs from each sampling site were analysed for organic carbon using a Thermal Optical Transmission box (Sunset Laboratory, Tigard, OR, USA). PFC concentrations for the relevant GFFs were corrected based on the punched area.

Results and discussion

Quality control

Concentrations in blank samples ranged from <IDL to 7 pg m^{-3} in PUF/XAD/PUF cartridges and GFFs and from <IDL to 10 pg m^{-3} in SIP disks. All results were corrected for blanks. Details of IDLs and the blank levels for individual compounds are given in Tables A5 and A6 in the Accessory publication.

LODs ranged from 0.4 to 13 pg m^{-3} for FTOHs, FOSAs and FOSEs and from 0.01 to 39 pg m^{-3} for PFCAs and PFSAs (see Table A6 in the Accessory publication).

Recoveries for internal standards of target PFCs in PUF/XAD/PUF cartridges ranged from 5.9 ± 1.8% for ^{13}C-6 : 2 FTOH to 100 ± 18% for ^{13}C-10 : 2 FTOH; from 72 ± 13% for d_3-MeFOSA to 230 ± 37% for d_9-EtFOSE and from 21 ± 11% for $^{13}C_2$-PFDoDA to 180 ± 96% for $^{13}C_2$-PFBA. The low recoveries for ^{13}C-6 : 2 FTOH are associated with the high volatility of this compound and the resulting evaporative losses during Soxhlet extraction and concentration. Signal enhancement caused by solvent or interfering compounds might be responsible for high recoveries, i.e. for d_9-EtFOSE. For GFFs, the recoveries ranged from 9.4 ± 2.8% for ^{13}C-6 : 2 FTOH to 150 ± 25% for d_9-EtFOSE and from 36 ± 8.5% for $^{13}C_2$-PFBA to 67 ± 16% for $^{13}C_2$-PFHxA. For the SIP disks recoveries ranged from 6.4 ± 1.0% for ^{13}C-6 : 2 FTOH to 310 ± 40% for $^{18}O_2$-PFHxS. All samples (including blanks) were recovery corrected using appropriate factors based on recoveries presented above and summarised in Table A7. The poor recoveries

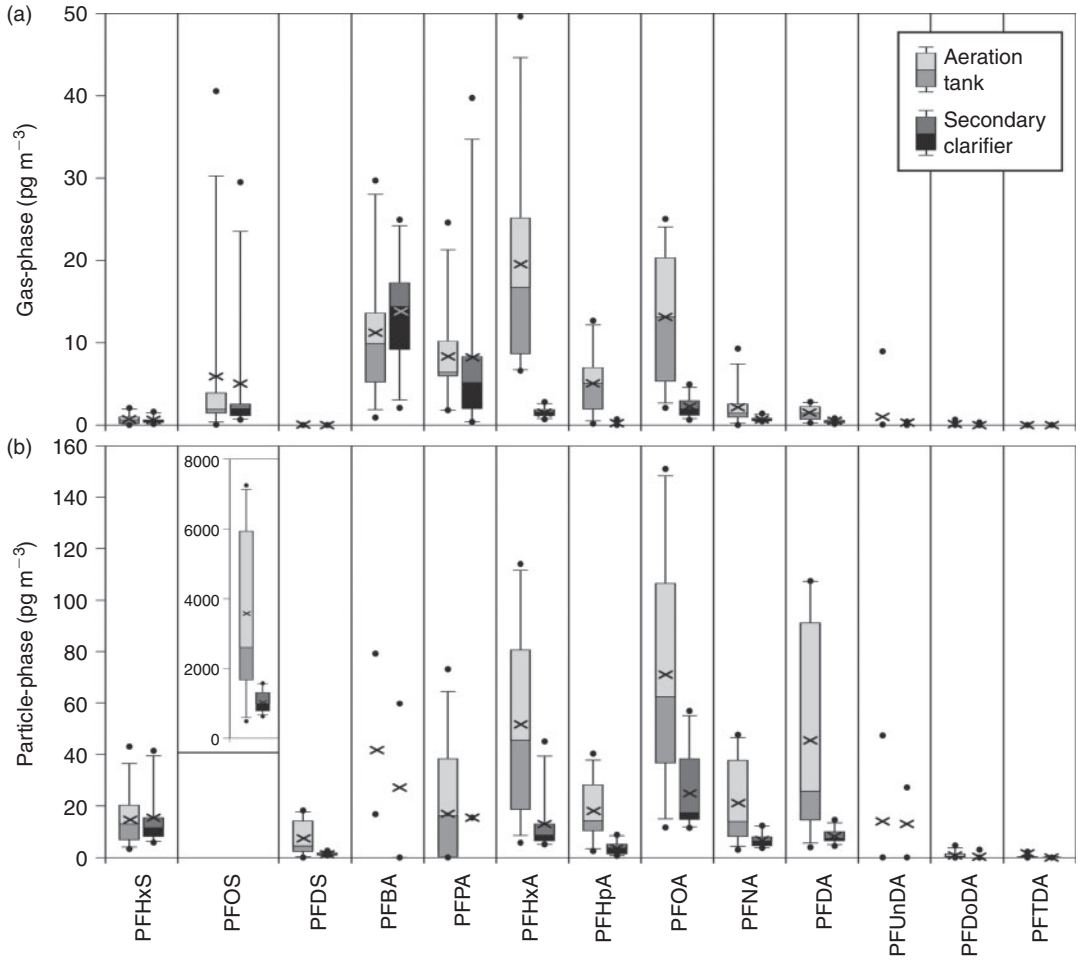


Fig. 2. Box-whisker plots for PFCA and PFSA concentrations in the gas (a) and particle-phase (b) at the aeration tank and the secondary clarifier. The boxes show median concentrations and the 25th and 75th percentiles; 10th and 90th percentiles are indicated by the whiskers and the dots represent the minimum and maximum concentrations. The mean concentrations are indicated with an ×. If more then 25% of the data for one compound were below the IDL only minimum, maximum and mean values are shown.

for 6 : 2 FTOH cause greater uncertainty in the derived air concentration for this compound.

FTOH, FOSA and FOSE air concentrations

Gas- and particle-phase air concentrations for FTOH, FOSA and FOSE are shown in Fig. 1 and are summarised with results from passive samples in Table 1. Detailed results are in Tables A8–A11 in the Accessory publication. PFOSEA was not detected above the IDL in any sample and was therefore excluded from further investigations.

Gas-phase samples were dominated by FTOHs with a mean concentration of 11 000 pg m^{-3} at the aeration tank and 590 pg m^{-3} at the secondary clarifier. Mean ∑FOSA & FOSE concentrations were 43 pg m^{-3} at the aeration tank and 16 pg m^{-3} at the secondary clarifier. Conversely, ∑FTOH exhibited the lowest particle-phase concentrations (25 pg m^{-3} at the aeration tank and 1.9 pg m^{-3} at the secondary clarifier), whereas for ∑FOSA & FOSE particle-phase concentrations were higher at 69 and 11 pg m^{-3} respectively. The mean ∑FTOH concentrations at the aeration tank were 18 times higher for the gas-phase and 13 times higher for the particle-phase compared with the secondary clarifier (*t*-test, $P < 0.001$ and $P < 0.005$ respectively). Mean ∑FOSA & FOSE concentrations were approximately a factor of three to six higher at the aeration tank compared with the secondary clarifier (*t*-test, $P < 0.001$ for gas- and particle-phase respectively). These results point to the important role of the aeration process in emitting high concentrations of PFCs to the atmosphere.

Composition

The composition of FTOHs, FOSAs and FOSEs in each sample is shown in Figs A1 and A2 in the Accessory publication. The profile of FTOHs in the gas-phase was dominated by 6 : 2 FTOH (54%) > 8 : 2 FTOH (38%) > 10 : 2 FTOH (8%) and was similar at both sampling sites. In the particle-phase at the aeration tank, all of the FTOHs were detected (8 : 2 FTOH (47%) > 10 : 2 FTOH (35%) > 6 : 2 FTOH (18%)) whereas at the secondary clarifier only 10 : 2 FTOH was detected above the IDL. For FOSAs and FOSEs the gas-phase profile was different at the two sites. The aeration tank samples were dominated by MeFOSE (47%) and MeFOSA (27%), whereas the secondary clarifier samples were dominated by PFOSA (39%) and MeFOSA (24%). FOSAs and FOSEs in the particle-phase showed a similar pattern at the aeration tank and the secondary

Table 1. Individual PFC concentrations in the gas-phase and particle-phase and from SIP disk passive air samplers at the aeration tank and secondary clarifier in picograms per cubic metre (minimum–maximum and average in parentheses)

	Aeration tank			Secondary clarifier		
	Gas-phase	Particle-phase	SIP disk passive air samples	Gas-phase	Particle-phase	SIP disk passive air samples
6 : 2 FTOH	780–14 000 (5700)	0.36–34 (7.7)	11 000–12 000 (12 000)	0.31–910 (330)	0.36–0.36 (0.36)	670–910 (790)
8 : 2 FTOH	900–16 000 (4700)	0.250–27 (10)	5700–5800 (5800)	27–670 (220)	0.25–0.25 (0.25)	310–350 (330)
10 : 2 FTOH	72–3100 (920)	2.5–21 (7.5)	780–860 (820)	6.2–110 (41)	0.82–5.2 (1.9)	41–48 (45)
∑FTOH	3300–33 000 (11 000)	4.5–79 (25)	18 000–19 000 (18 000)	34–1700 (590)	1.4–5.8 (2.5)	1100–1300 (1200)
MeFOSA	0.71–36 (13)	0.27–3.3 (1.4)	13–14 (14)	0.23–6.7 (1.9)	0.1–0.66 (0.4)	0.79–1.0 (0.90)
EtFOSA	0.46–10 (4.9)	0.25–1.5 (0.8)	5.7–5.9 (5.8)	0.19–3.4 (1.2)	0.1–0.32 (0.23)	1.2–1.6 (1.4)
MeFOSE	4.9–44 (20)	6.6–44 (18)	16–18 (17)	0.54–13 (5.6)	0.77–12 (3.7)	4.5–4.9 (4.7)
EtFOSE	4.1–7.0 (6.5)	3.4–24 (11)	8.5–9.4 (8.9)	<6.96–8.3 (7.1)	0.81–13 (2.2)	1.8–2.3 (2.0)
PFOSA	2.3–9.5 (4.3)	5.9–95 (38)	4.9–10 (7.7)	0.54–11 (3.3)	0.97–12 (4.0)	0.0–0.0 (0.0)
∑FOSA & FOSE	11–100 (43)	23–120 (69)	52–55 (54)	11–31 (16)	4.9–31 (11)	6.9–9.3 (8.1)
PFHxS	0.06–2.1 (0.78)	3.3–43 (15)	0.62–0.87 (0.74)	0.12–1.6 (0.59)	5.8–41 (15)	0.65–0.91 (0.75)
PFOS	1.1–41 (5.8)	480–7200 (3900)	220–260 (240)	0.65–30 (4.7)	620–1600 (1100)	21–38 (30)
PFDS	0.01–0.01 (0.01)	0.04–18 (7.4)	0.01–0.01 (0.01)	0.01–0.01 (0.01)	0.84–2.6 (1.4)	0.01–0.01 (0.01)
PFBA	0.9–30 (11)	9.3–79 (42)	13–15 (14)	2.1–25 (14)	1.4–62 (31)	6.7–13 (9.9)
PFPA	1.7–25 (8.7)	0.99–73 (22)	6.1–6.7 (6.4)	0.42–40 (8.8)	19–19 (19)	0.34–2.4 (1.4)
PFHxA	6.6–50 (20)	5.7–110 (52)	18–20 (19)	0.7–2.8 (1.6)	5.0–45 (13)	1.2–1.7 (1.4)
PFHpA	0.16–13 (5.1)	2.5–40 (18)	5.2–6.0 (5.6)	0.04 –0.81 (0.54)	0.66–8.8 (3.8)	0.85–1.0 (0.94)
PFOA	2.1–25 (13)	12–150 (71)	8.1–11 (9.7)	0.63–4.9 (2.3)	12–57 (25)	2.0–3.1 (2.5)
PFNA	0.69–9.3 (2.1)	3.0–48 (21)	1.1–1.2 (1.1)	0.45–1.4 (0.72)	3.7–12 (6.8)	0.41–0.66 (0.54)
PFDA	0.25–2.8 (1.4)	4.0–110 (46)	2.0–2.2 (2.1)	0.14–0.81 (0.42)	4.5–15 (8.3)	0.28–0.38 (0.33)
PFUnDA	0.03–9.0 (1.2)	0.59–47 (14)	1.2–1.2 (1.2)	0.06–0.41 (0.33)	2.0–27 (15)	012–0.35 (0.24)
PFDoDA	0.09–0.64 (0.24)	0.09–4.7 (1.1)	0.15–0.21 (0.18)	0.06–0.31 (0.11)	0.08–3.1 (1.3)	0.02–0.05 (0.04)
PFTDA	0.004–0.004 (0.004)	0.02–2.2 (0.31)	0.01–0.01 (0.01)	0.0–0.0 (0.0)	0.03–0.36 (0.17)	0.01–0.01 (0.01)
∑PFCA & PFSA	25–120 (69)	849–7600 (3900)	280–320 (300)	17–86 (34)	760–1700 (1100)	35–61 (48)
∑PFC	3300–33 000 (11 000)	29–130 (94)	18 000–19 000 (19 000)	47–1790 (600)	6.0–36 (13)	1100–1300 (1200)

clarifier with dominant compounds being PFOSA (53 and 40% respectively), MeFOSE (27 and 38% respectively) and EtFOSE (~17% at both sites).

Comparisons with other measurements

It is interesting to compare the magnitude of air concentrations for the various PFCs measured at the WWTP to other studies. This comparison will give some sense of the importance of the WWTP as a point source to air. Air concentrations of FTOH in the urban area of Toronto ($\sum$FTOH = 81 pg m^{-3} sum of particle and gas-phase[29]) were two orders of magnitude lower than at the aeration tank and one order of magnitude lower than at the secondary clarifier. Thus WWTPs seem to be an important point source for FTOHs. Differences were less drastic for other PFCs. For instance, FOSA and FOSE concentrations were approximately six times higher at the aeration tank compared with urban areas ($\sum$FOSA & FOSE = 19 pg m^{-3} sum of particle and gas-phase[19,29]) and the FOSA & FOSE concentrations at the secondary clarifier were generally in the same range as in urban areas.[19,29] Ongoing studies at this WWTP are attempting to quantify the emission fluxes to air so that WWTPs as a whole can be assessed in terms of their contribution to the atmospheric burdens of PFCs.

PFCA and PFSA air concentrations

Gas- and particle-phase results for PFCA and PFSA in air are shown in Fig. 2 and further summarised in Table 1 and in Tables A8–A11 in the Accessory publication.

Gas-phase concentrations of PFCAs and PFSAs ($\sum$PFCA & PFSA 70 pg m^{-3} at the aeration tank and 34 pg m^{-3} at the secondary clarifier) were one to three orders of magnitude lower compared with FTOHs; however, they were two times higher than FOSAs and FOSEs. In contrast, $\sum$PFCA & PFSA concentrations in the particle-phase were 500 and 100 times higher than the $\sum$FTOH and $\sum$FOSA & FOSE concentrations respectively. PFOS was the dominant compound in the particle-phase. PFOS concentrations (average 3600 pg m^{-3} at the aeration tank and 1000 pg m^{-3} at the secondary clarifier) were one to three orders of magnitude higher than concentrations of the other PFCs. It is interesting to note that mean $\sum$PFCA & PFSA concentrations were significantly higher at the aeration tank compared with the secondary clarifier (factor of 1–4, *t*-test $P < 0.012$ for the gas-phase, $P < 0.003$ for the particle-phase). In former studies investigating other WWTPs, higher PFC concentrations were reported in treated effluent in comparison to influent wastewater, though mass flow charts from Schultz et al. indicate similar concentrations in the aeration tank and secondary clarifier.[15,30] This indicates that the observed differences in air concentrations are most likely associated with enhanced mass transfer (water to air transfer) of PFCAs and PFSAs owing to the aeration process rather than to differences in wastewater concentrations of PFCAs and PFSAs between the aeration tank and secondary clarifier.

Composition

The composition of PFCAs and PFSAs in each sample is shown in Fig. A3 in the Accessory publication. The profile of PFCAs and PFSAs in the gas-phase was different at the two sampling sites. At the aeration tank, PFHxA was dominant (29%), followed by PFOA and PFBA (both ~19%). The contributions of the remaining compounds were 10% (PFPA)

and lower. Samples from the secondary clarifier were dominated by PFBA (48%), followed by PFPA (15%) and PFOS (11%). The contributions of the remaining compounds were below 9%. These results indicate that the different treatment processes of the wastewater at the WWTP caused different air emission signatures for the PFCAs and PFSAs. However, in general, PFOS and the short chain PFCAs (C_4–C_8) were the dominant compounds in the gas-phase at both the aeration tank and the secondary clarifier, whereas the contribution of the longer chain PFCAs (C_9–C_{14}) was very low (<3%). PFOS was also dominant among the PFCAs and PFSAs in the particle-phase (~91% of ∑PFCA & PFSA) at both sites. The next dominant PFCA and PFSA after PFOS was PFOA (3% at the aeration tank and 5% at the secondary clarifier), followed by PFHxS, PFHxA and PFDA (each >0.5%). The dominance of PFOS and PFOA on atmospheric particles has been reported in the literature[19]; however, the dominance of PFOS as observed in this study was not previously observed or reported. The dominance of PFOS (and PFOA) in particles might reflect the pattern of PFCs used in industrial and consumer products.[31,32]

Comparisons with other measurements

Chemical ratios are sometimes used to compare or differentiate sources. In the current study the ratio of PFOS and PFOA, i.e. PFOS/PFOA, observed at the WWTP could be compared with other locations to gain some sense of whether the WWTP as a point source could be contributing substantially to the broader contamination of the atmosphere. The ratio PFOS/PFOA in the gas-phase from the WWTP (0.4–2.1) was in the same range as the ratio from an urban area (0.5),[19] which suggests a possible contribution of the WWTP to urban air or at least that the source for urban air is similar. However, for the particle-phase, PFOS/PFOA was more than 100 times higher at the WWTP (42–50) compared with an urban area (0.3).[19] This may indicate that the particle-phase signature at the WWTP is a localised or short-lived source.

Most literature reports of PFCAs and PFSAs focus on the particle-phase exclusively. For instance, Dreyer et al. reported ∑PFCA and ∑PFSA concentrations in the particle-phase in Germany of 1.0 and 1.3 pg m^{-3} respectively.[4,33] These concentrations are one to three orders of magnitude lower compared with concentrations at the WWTP in this study. In a study from New York State, PFCAs and PFSAs were measured in the gas and particle-phase.[19] These concentrations (i.e. ∑C_6,C_{10} PFSA & $C_{7–12}$ PFCA 8.0 and PFOS 2.3 pg m^{-3})[19] were also one to three orders of magnitude lower compared with the present study (210 and 3600 pg m^{-3} at the aeration tank and 110 and 1000 pg m^{-3} at the secondary clarifier).

In summary, the measured concentrations of PFSAs and PFCAs at the WWTP were greatly elevated compared with other studies, even for urban areas. This highlights the importance of WWTPs as point-source emitters of these compounds to the atmosphere.

Correlations of atmospheric concentrations with sampling parameters

The influence of various meteorological and particle parameters (e.g. ambient air temperature, TSP and particle OC content) on PFC air concentrations were investigated.

Air temperature, which ranged from −0.2 to 12.5°C during high volume sampling, showed a positive correlation for FOSEs and FOSAs in the gas phase, i.e. MeFOSA and MeFOSE ($P<0.05$). This is likely owing to greater evaporation of these compounds at higher temperatures. However, other PFCs did not exhibit this correlation. It is likely that evaporation from wastewater is governed more by the temperature of the wastewater (versus the air temperature) which is much less subject to variability. A positive correlation of the PFC concentration with the air temperature was found previously[34]; however, owing to a weak correlation, it was assumed that other factors may have also had an influence.[33]

The OC contents of the particles from the two sites were not significantly different (4.0 ± 1.1% OC at the aeration tank, $n=7$, and 3.3 ± 0.9% OC at the secondary clarifier, $n=7$; t-test $P=0.2$) and no correlation with atmospheric PFC concentrations were found. The TSP concentration was significantly higher at the aeration tank (120 ± 29 μg m^{-3}, $n=12$) in comparison to the secondary clarifier (75 ± 20 μg m^{-3}, $n=12$, t-test $P<0.001$) but again no correlation with atmospheric PFC concentrations were found. The higher TSP concentration above the aeration tank is likely the result of aerosol generation and the release of wastewater particulates to air.

Particle–gas partitioning

The percent on particles for the various target PFCs is summarised in Fig. 3. This was calculated as the concentrations in the particle-phase (in picograms per cubic metre) divided by the sum of concentrations in the gas and particle-phase (in picograms per cubic metre) and multiplied by 100. There was no substantial difference in particle–gas partitioning of target compounds collected at the aeration tank versus the secondary clarifier and so average values are represented in Fig. 3.

Of the target PFCs, the FTOHs showed the smallest particle-phase percentages that were typically less than 10%. Particle bound fractions for FTOHs found during ship-based measurements for 8 : 2 FTOH and 10 : 2 FTOH were higher compared with results from the present study (up to 23%,[29] 26% for 8 : 2 FTOH and 15% for 10 : 2 FTOH[35]).

Particle-phase percentages increased for MeFOSA and EtFOSA (~19 and ~15% respectively) and were even higher for MeFOSE, EtFOSE and PFOSA (38–70%). Other studies showed lower particle-phase percentages from land-based measurements (i.e. MeFOSA & EtFOSA <10%, EtFOSE 14%[4]), whereas ship-based measurements were similar to the results for the WWTP (i.e. MeFOSA & EtFOSA ~15%,[4] MeFOSE 30%,[29] EtFOSE 57%[4]).

The PFSAs had the highest particle associated fractions with almost 100% bound on particles. PFCAs were also mainly particle-associated and this increased generally according to chain length as: PFBA (~64%) ≈ PFPA (~68%) < PFHxA (~78%) < PFHpA (~80%) < PFOA (~86%) ≈ PFNA (~88%) < PFDA (~95%) > PFUnDA (~89%) > PFDoDA (~80%) < PFTDA (~100%). This pattern is likely owing to the decrease in vapour pressure (which favours the condensed state) with increasing chain length for the neutral forms of PFCAs.[36,37] Only one study is available for comparison with these results.[19] Particle-phase percentages reported for New York State were 60% for PFHpA, 40% for PFOA and PFNA and 30% for PFDA, PFDoDA and PFOS.[19] Moreover, the shorter chain PFCA, PFHpA, had the highest particle-bound fractions and the longest chain PFCA, PFDA, had the lowest particle-phase fractions.[19] At this time, we have no explanation for these contradictory results.

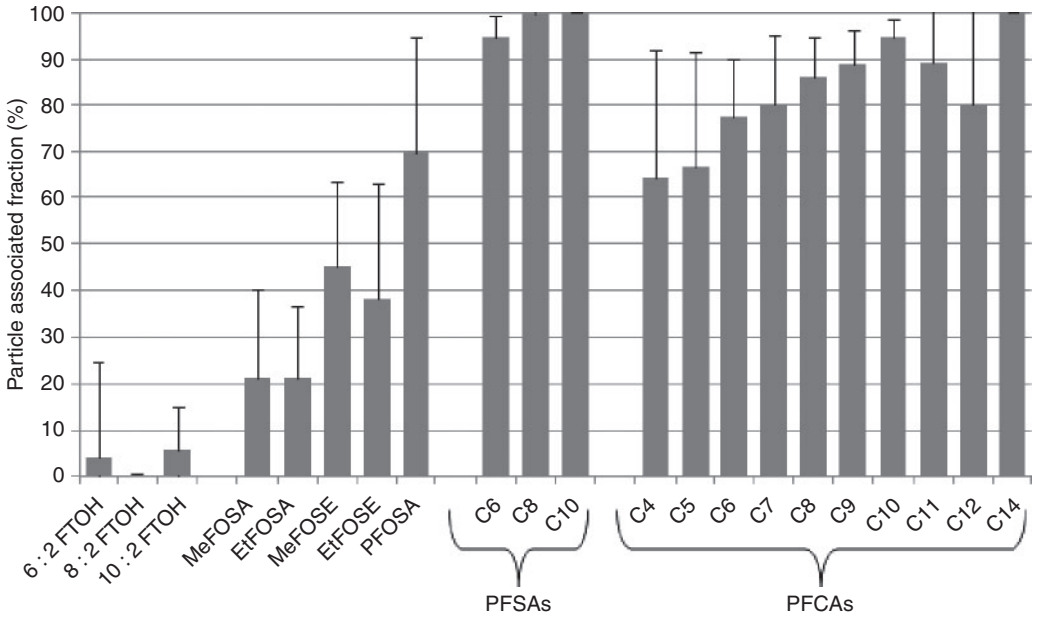


Fig. 3. Particle associated fractions (%) for individual PFCs in air at the aeration tank and secondary clarifier.

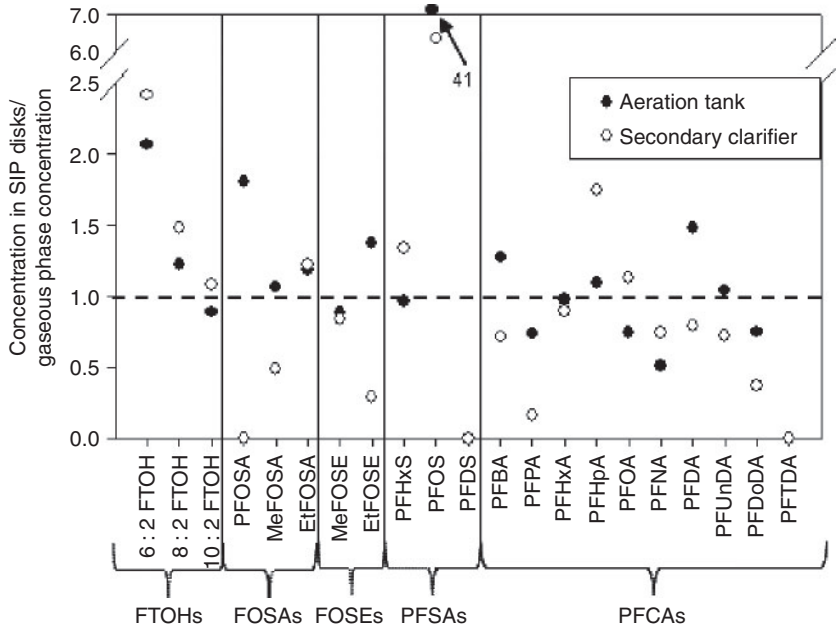


Fig. 4. Ratio between individual PFC concentrations in the SIP disks passive air samples (pg m^{-3}) ($n = 2$) and gas-phase from active air sampling (pg m^{-3}) ($n = 12$). The dashed line represents perfect agreement.

Comparison of active v. *passive sampling techniques*

It is interesting to compare the gas-phase air concentrations from the intermittent high volume samples against the time-integrated concentrations derived using the passive samples. Differences can be expected for several reasons: (i) differences in the sampling time, i.e. the high volume samples were collected on two consecutive days each week and represented ~29% of the time that was sampled by the continuous and time-integrating passive samplers. Thus, high and low air concentration episodes that could offset the true time-integrated air concentrations might not have been captured by this 29% of the time window; (ii) although passive samplers better cover the entire duration of the study, there is greater uncertainty with these derived concentrations owing to uncertainties in the sampling rates; (iii) collection of particles on the SIP disks is known to occur[38] and this may result in an overestimate of the gas-phase concentration for compounds that are particle-associated (this issue is discussed further, below); and (iv) general analytical errors that contribute to uncertainties.

Despite these potential uncertainties and confounding factors, the agreement between gas-phase concentrations derived from high volume samples *v*. SIP disk passive air samples was fairly good as shown in Fig. 4 with detailed results presented in

Table 1. The concentrations differed by a mean factor of 1.5 for FTOHs, 0.96 for FOSAs, 0.85 for FOSEs, 1.1 for PFHxS, 24 for PFOS and 0.8 for PFCAs.

Sampling artefacts

Previous laboratory investigations have shown that PFCAs may adsorb to filters (GFFs and quartz fibre filters (QFFs)) and therefore particle-phase concentrations derived from sampling techniques using GFFs and QFFs could be overestimated.[39] The results of the present study and the comparison of active versus passive samples provides some insight into this issue. PFCAs and PFSAs were found predominantly in the particle-phase. However, the presence of PFCAs in the gas-phase and the good agreement between gas-phase concentrations using high volume sampler and passive air sampler concentrations indicate that this sampling artefact has a relatively minor influence on the gas-phase concentrations. However, we acknowledge that the high concentrations of target compounds at the WWTP may not provide the best conditions for detecting this artefact. It may be more important at lower air concentrations.

Particle-phase sampling by the SIP disks is another 'artefact' that complicates the comparison of results from high volume samples and from SIP disks. It has been shown that the SIP disk sampling chamber allows ~10% of the ambient particles to be sampled and so the SIP disk is not just a gas-phase passive sampler.[38] The net effect is demonstrated well by the results for PFOS (see Fig. 4). In the high volume samples, the PFOS particle-phase air concentrations are more than two orders of magnitude larger than the gas-phase concentration. The air concentration derived from the SIP disk passive air samples, which represents mainly the gas-phase and ~10% of the particle-phase,[38] fall somewhere in between. In this case the 10% of ambient particles that are sampled by the SIP disk outweigh the gas-phase contribution. More studies are required to further elaborate and quantify particle-phase sampling by passive samplers and sorption artefact for filters.

Conclusion

This study demonstrates the importance of WWTPs as point sources of PFCs to the atmosphere. The aeration process in particular is shown to be a key emission process for both gas-phase and particle-associated PFCs. Aerosol-mediated transport is believed to account for the higher amounts of particle-associated PFCs in air near the aeration tank. This pathway is likely also to be important in open water bodies as aerosols are generated and released to air via sea spray and wave action. Passive and active samplers are shown to be complementary and comparable air sampling approaches for PFCs.

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Letter

Landfill Gas: A Major Pathway for Neutral Per- and Polyfluoroalkyl Substance (PFAS) Release

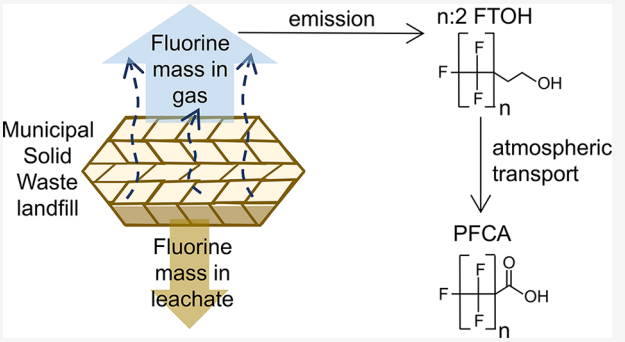
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ABSTRACT: The undisclosed and ubiquitous use of perfluoroalkyl and polyfluoroalkyl substances (PFAS) in consumer products has led to a growing issue of environmental pollution, particularly within the solid waste community, where the fate of volatile (neutral) PFAS in landfilled refuse is not well understood. Here, three municipal solid waste landfills in Florida were assessed for neutral PFAS in landfill gas and ionic PFAS in landfill leachate to compare the relative mobility between the two pathways. Landfill gas was directly sampled using a high volume, XAD-2 resin based sampling approach developed for adsorption and analysis of 27 neutral PFAS. Across sites, 13 neutral PFAS were identified from fluorotelomer alcohol (FTOH), fluorotelomer olefin (FTO), secondary FTOH, fluorotelomer acetate (FTOAc), and fluorotelomer methyl acrylate (FTMAc) classes; however, FTOHs dominated concentrations (87−97% total neutral PFAS), with most detections surpassing utilized calibration levels. Even under conservative assumptions, the mass of fluorine leaving in landfill gas (32−76%) was comparable to or greater than the mass leaving in landfill leachate (24−68%). These findings suggest that landfill gas, a less scrutinized byproduct, serves as a major pathway for the mobility of PFAS from landfills.



KEYWORDS: *volatile, emissions, GC, fluorotelomer alcohol*

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1. INTRODUCTION

Widespread per- and polyfluoroalkyl substance (PFAS) contamination has been a mounting environmental concern due to their chemical persistence and toxicity to human and biotic health.[1−4] While numerous industries are being confronted with PFAS-related management challenges, the burden of remediation and PFAS removal has often fallen on downstream industries—namely, the solid waste sector.[5−9] Discarded, PFAS-laden consumer products including textiles, wood products, and packaging and commonly landfilled industrial byproducts like MSW incineration ash and wastewater biosolids are known contributors to PFAS loading in landfills.[10−16] Existing research suggests most discarded PFAS mass is retained within landfills[9,17] with liquid-phase byproducts of waste decomposition, leachate and gas condensate, currently considered prevalent pathways for PFAS mobilization.[2,7,9] However, the extent of PFAS release to another major byproduct, landfill gas (LFG), has remained largely unscrutinized.

The bulk of PFAS characterization studies focus on nonvolatile/semivolatile (ionic) perfluoroalkyl acids (PFAAs) measured in liquid and solid matrices, in part because of a high presence and awareness of these species within the PFAS community but largely because analytical capabilities for ionic PFAS measurement are better established.[18−21] Volatile (neutral) PFAS are also utilized in consumer products[13,22−27,27] and have been determined in a few studies on ambient air surrounding landfills and near wastewater treatment plants,[28−32] but a lack of volatile analytical standards and latency in methodological development has hindered the progression of gas phase research in environmental matrices. Whereas PFAS characterization in leachate is established, concentrations ranging from thousands to tens of thousands of nanograms per liter are commonly encountered;[33−38] only two studies characterize volatile PFAS directly in LFG.[39,40] Titaley et al. identified fluorotelomer alcohol (FTOH), fluorotelomer acrylate (FTAc) and fluorotelomer olefin (FTO) homologues in LFG with combined concentrations ranging from 4,600 to 14,000 ng m^{-3} across three landfills. Goukeh et al., only assessing FTOHs, identified higher combined concentrations than Titaley et al., finding ∼18,000 ng m^{-3} (sum of 6:2 and 8:2

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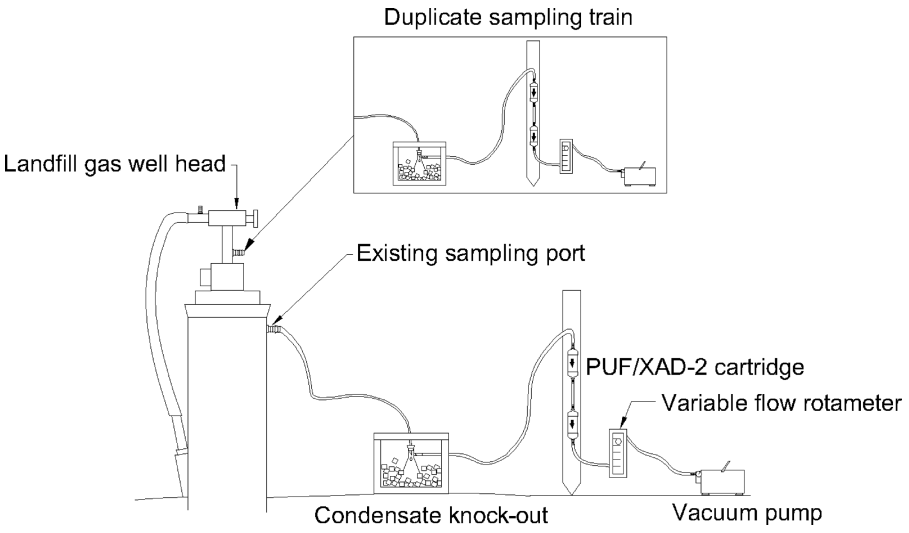


Figure 1. Developed system for sampling neutral PFAS directly from landfill gas well heads.

FTOH) in the one LFG sample examined. These studies suggest PFAS variability in LFG, which motivates further investigation, deploying higher sampling volumes[39] and larger analyte lists[40] to understand the potential presence of other neutral PFAS and distribution among landfills of different regions, compositions, and sizes.

With the ongoing development of PFAS regulation,[19] understanding the partitioning behavior of PFAS in major repositories like MSW landfills grows increasingly critical to minimize environmental and human risk. Unlike leachate, LFG is not always captured by collection systems, and management varies broadly across landfills, ranging from no treatment (i.e., passive venting) to some treatment (i.e., flaring, LFG to energy projects), but current treatment, if any, is not intended for PFAS.[41,42] Emerging research suggests the toxicity of volatile species (specifically 6:2 FTOH) to be significantly higher than their ionic counterparts via the inhalation pathway (a main route of exposure for volatile compounds).[43−47] Further, degradation of neutral species to ionic PFAAs once emitted to the atmosphere is well established.[48−58] The potential for long-range atmospheric transport of PFAS from landfills underscores the importance of considering neutral species and their fate during management to prevent further environmental contamination of highly scrutinized PFAAs such as perfluorooctanoic acid (PFOA) and perfluorooctanesulfonic acid (PFOS). As the only studies on LFG primarily identified FTOHs in LFG, the magnitude and significance of other neutral species remains unclear.

Here, LFG was sampled directly from gas well heads at three MSW landfill locations in Florida using a higher volume sampling protocol. XAD-2 resin sandwiched between polyurethane foam (PUF) was utilized for PFAS capture, then samples were analyzed for 27 volatile/semivolatile (neutral) PFAS via targeted gas chromatography high resolution mass spectrometry (GC-HRMS). To contextualize release in the gas phase, leachate was also collected at each landfill and analyzed for ionic PFAS (n = 93) using ultrahigh pressure liquid chromatography tandem mass spectrometry (UHPLC-MS/MS). The observed LFG and leachate concentrations were normalized on a mass of fluorine basis to compare the potential mobility in gas versus leachate matrices. This study provides foundational data critical for understanding the role of landfills in anthropogenic PFAS release and for informing LFG management.

2. METHODS AND MATERIALS

For brevity, materials and methods associated with ionic PFAS analysis in landfill leachate are provided in section 1 of the Supporting Information (Tables S-1 through S-4).

2.1. Standards and Reagents. Targeted neutral PFAS (≥97% purity, n = 27) were purchased from Wellington Laboratories Inc. (Guelph, ON, Canada), SynQuest Laboratories (Alachua, FL), and Chiron (Stiklestadveien, Trondheim, Norway). Nine classes of neutral PFAS (perfluoroalkane sulfonamides (FASAs), perfluoroalkane sulfonamidoethanols (FASEs), fluorotelomer acetates (FTOAcs), fluorotelomer methyl acrylates (FTMAcs), fluorotelomer iodides (FTIs), fluorotelomer secondary alcohols (sFTOHs), FTOHs, FTAcs, and FTOs) were measured using eight isotopically labeled internal standards (IS) from FASA, FASE, FTOH, and FTMAc classes for quantitation (Table S-5).

2.2. Sample Preparation and Collection. Polyvinyl chloride (PVC) cartridges filled with 4−5 g of Amberlite XAD-2 resin retained between two polyurethane foam (PUF) discs were utilized for PFAS capture.[29,59,60] Before use, XAD-2 sorbent was made PFAS-free through sequential Soxhlet extractions.[61] All cartridge components, sampling vessels, and tubing were sonicated in a mixture of Liquinox and PFAS-free water, rinsed, and then sonicated in methanol and methanol rinsed before use. Once dried and assembled, cartridges were stored in individually sealed polyethylene bags at 4 °C until sampling.

As neutral compounds were the focus of this investigation, aerosolized/particulate-bound PFAS were not specifically targeted for capture; however, a condensate collection system was included to prevent moisture interference. The developed sampling system (Figure 1) consisted of a condensate knockout (borosilicate, barbed Erlenmeyer flask contained in a cold box), two PUF/XAD-2 cartridges (installed in-series), a rotameter for flow control, a portable vacuum pump, and PFAS-free Tygon tubing. Before each sampling event, gas well head connection to the larger landfill gas collection system was disabled to create a neutral to positive pressure, workable for flow through the sampling system, then gas composition/

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Table 1. Average Concentrations (n = 2) of 13 Neutral PFAS (ng m^{-3}) from Three Municipal Solid Waste Landfills in Florida (Site Characteristics Are Provided in Table S-9)[a]

	concentration (ng m^{-3})					
analyte	landfill 1	E	landfill 2	E	landfill 3	E
4:2 FTOH	220		ND		57	
6:2 FTOH	*>9,900*	**170,000**	*>6,000*	**22,000**	*>6,500*	**62,000**
8:2 FTOH	*>6,800*	**200,000**	*>6,000*	**140,000**	*>6,500*	**740,000**
10:2 FTOH	*>5,100*	**14,000**	*>3,000*	**23,000**	*>5,000*	**120,000**
12:2 FTOH	860		1,400		5,000	
5:2 sFTOH	*>2,900*	**8,800**	*>1,700*	**9,000**	*>1,900*	**5,900**
7:2 sFTOH	320		*>1,300*	**13,000**	*>1,400*	**11,000**
8:2 FTO	2,500		1,300		550	
10:2 FTO	650		840		540	
12:2 FTO	97		580		160	
8:2 FTOAc	610		90		490	
10:2 FTOAc	99		19		140	
6:2 FTMAc	3,800		56		150	

[a]Concentrations of 6:2, 8:2, and 12:2 FTOH and 5:2 and 7:2 sFTOH consistently exceeded the upper limit of developed calibration ranges; therefore, both a minimum concentration (assuming the highest calibration concentration) and a maximum extrapolated concentration are provided. Italicized values denote a minimum concentration. Column "E" presents average maximum concentrations. "ND" denotes non-detect measurements. FTAcs, FASAs, FASEs, FTIs, and 8:2 FTMAc were not detected in any samples. Analyte acronyms and details are provided in Table S-5.

temperature was recorded using an Optimax Biogas analyzer (MRU Instruments, Humble, TX). Duplicate sampling trains were connected to existing gas well sampling ports. Approximately 1,200 L was sampled through each train at a flow rate of 5 L min^{-1}. After sampling, PUF/XAD-2 cartridges were sealed and individually stored at ≤4 °C for transport/storage. Quality control (QC) procedures are provided in the SI, section 2.

2.3. Extraction and Analysis. Spent XAD-2 from each cartridge was weighed and transferred to a 50 mL polypropylene centrifuge tube and vortexed, and approximately 2 g aliquoted for extraction. Samples were spiked with a mixture of mass labeled IS (Table S-5), rotated end-over-end for 18 h in 4 mL of 75/25% (v/v) ethyl acetate and methanol, and centrifuged for 10 min at 4,000 rpm. Supernatants were transferred to 15 mL centrifuge tubes, and the extraction process was repeated, combining supernatants from the two-fold extraction. Extracts were concentrated to 3 mL via gentle nitrogen evaporation, aliquoted, and stored no more than 30 days at −20 °C until analysis. QC details are provided in the SI, section 2 (Table S-6 and Figure S-1).

Targeted analysis of 27 neutral PFAS by positive chemical ionization (PCI) with selected ion monitoring (SIM) was conducted using a Thermo Scientific TRACE 1310 gas chromatograph coupled to a Thermo Scientific Orbitrap Exploris GC 240 mass spectrometer (GC-HRMS; see SI, section 2 for details regarding GC separations and instrumentation). A 12-point external calibration curve (from 1 to 2,000 pg μL^{-1}) was developed for quantitation, prepared through serial gravimetrically derived dilutions of primary stock solutions. A mixture of mass labeled IS at concentrations of 150 pg μL^{-1} was added to each calibration level. When a labeled standard was not available for a compound, a labeled standard with a similar retention time or structure was utilized for quantitation (Table S-5).

3. RESULTS AND DISCUSSION

Unexpectedly, several neutral PFAS concentrations in LFG exceeded the implemented calibration levels. Because of considerable exceedance for some compounds, dilution would reduce IS below instrument detection; therefore, in instances where sample concentrations exceeded calibration limits, two concentrations are presented (Equation S-1): a minimum value which assumes the highest calibration concentration and a maximum extrapolated concentration. Fluorine mass release calculations utilize minimum values, preventing overextrapolation while providing a conservative estimate for leachate comparison. Even under these assumptions, substantial concentrations of neutral PFAS, higher than those previously observed, were identified. Future assessments should deploy shorter sampling durations to refine findings.

3.1. Neutral PFAS in Landfill Gas. Except for 4:2 FTOH in one landfill, 13 PFAS were detected in duplicate samples across the three sites (site characteristics are provided in Table S-9). Observed concentrations are displayed in Table 1. At minimum, combined concentrations of neutral PFAS in LFG ranged from 22,000 to 33,000 ng m^{-3}. Considering extrapolated values, total concentrations ranged from 210,000 to 940,000 ng m^{-3}, an order of magnitude higher than those previously reported in LFG.[39]

3.1.1. FTOHs and sFTOHs in Landfill Gas. Like previous studies on LFG and air surrounding landfills, FTOHs dominated neutral PFAS concentrations;[28,31,32,39,40] however, extrapolated concentrations in this study surpassed previous reports in LFG, in some cases by 2 orders of magnitude, and were more comparable (although much lower) to concentrations recently identified in soil vapor near a PFAS manufacturing facility.[55] While there are uncertainties given the degree of extrapolation, the magnitude of FTOHs found in this study compared to existing research suggests fundamental differences potentially related to sampling methodology (e.g., much larger sampling volumes) and/or sampled landfill characteristics (e.g., waste type, age, air intrusion), although these data were not available for comparison. Across the three sites, 6:2, 8:2, and 10:2 FTOH, combined, made up 87 to 97% of total concentrations, but 8:2 FTOH alone constituted 50 to 79%. The shortest and longest analyzed homologues, 4:2 and

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12:2 FTOH, were significantly lower in concentration (Table S-10). This is supported by previous FTOH distributions determined from source fluoro-telomer polymers[62] and observations in LFG, urban air, and air surrounding wastewater treatment/landfill sites.[28−31,39,40,63] Concentrations of 12:2 FTOH were of similar magnitude to those in Titaley et al., but 4:2 FTOH has not been detected in LFG, suggesting MSW landfills to be a previously unidentified potential source of atmospheric 4:2 FTOH.[39]

Secondary FTOHs have not been targeted in gas-phase landfill research but have been identified in condensate associated with LFG collection systems.[37] As intermediary byproducts of 6:2 and 8:2 FTOH biodegradation to PFAAs, 5:2 and 7:2 sFTOH, were unsurprisingly elevated, they were at least an order of magnitude lower than respective parent FTOH homologues.[64−66] All detections of 5:2 sFTOH and two out of three detections of 7:2 sFTOH were above calibration, combined sums attributed to 2 to 10% of total concentrations.

3.1.2. Other Neutral PFAS in Landfill Gas. Other neutral PFAS fell within acceptable calibration ranges and together accounted for 0.22 to 1.9% of total concentrations. FTO homologues have been encountered in other LFG and ambient air studies, but in past assessments 8:2 and 10:2 FTO were below limits of quantitation and 12:2 FTO concentrations were consistently an order of magnitude higher than those reported here.[39,59] To the authors' knowledge, 8:2 and 10:2 FTOAc and 6:2 FTMAc have not been determined in LFG. FTOAcs are not commonly assessed analytes but are associated with fluoropolymer textile treatments and have been identified in one indoor air study from Japan.[67,68] Similarly, 6:2 FTMAc has only been analyzed in a few studies on cosmetics and wastewaters but at lower concentrations.[69−71]

3.2. Comparative Fluorine Mass Release between Landfill Byproducts. Normalizing PFAS concentrations on a fluorine basis allows comparisons to be drawn between different matrices and PFAS types (e.g., gas−liquid, neutral−ionic, precursor−terminal). This methodology is widely used to assess the "mass balance" of PFAS within systems, given that the long-term environmental fate of measurable PFAS is transient, whereas the mass of fluorine is conserved.[17,72,73]

Here, the same approach is utilized to compare the PFAS mobility in leachate versus LFG pathways. Neutral (Table 1, minimum values) and ionic (Table S-4) PFAS concentrations in LFG and leachate from this study were individually normalized to a mass of fluorine (Equation S-2) using compound specific fluorine mass fractions (Table S-8). Summed fluorine masses in leachate and LFG were then scaled according to site-specific annual generation volumes reported for each landfill (Table S-9).[41] A caveat of this comparison is the absence of measurements for neutral species in leachate and ionic species in LFG; however, the literature suggests FTOHs (the dominant neutral class identified) predominantly exist in the gaseous phase, while PFAAs exist in liquid or particulate phases.[28,74] Subsequent research should assess neutral and ionic compounds in both matrices to validate findings and further elucidate the PFAS behavior in landfills.

Even utilizing minimum concentrations observed in LFG, equal magnitudes of fluorine release are observed between LFG and leachate at each site (Figure 2)—contrasting from existing estimates of PFAS mass flow from landfills.[9] Existing

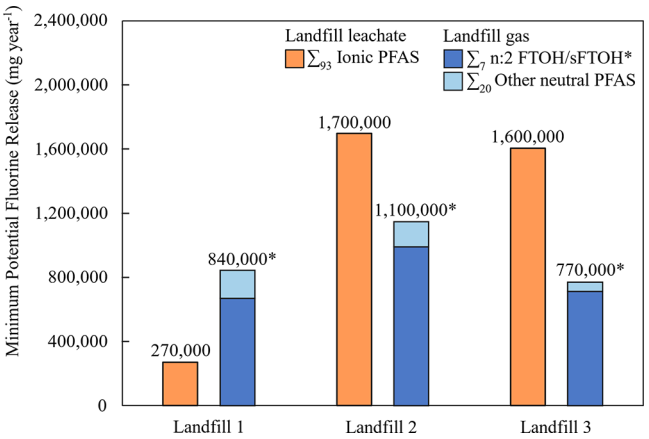


Figure 2. Annual fluorine mass release in landfill leachate versus landfill gas (LFG) from three municipal solid waste landfills in Florida. Fluorine masses in leachate are derived from ionic PFAS ($\sum_{93}$ PFAS) concentrations measured in leachate from each site (Table S-4) multiplied by the annual leachate generation volume and scaled using each detected compound's fluorine mass fraction (Tables S-8, S-9). The same methodology was applied for neutral PFAS ($\sum_{27}$ PFAS) in LFG by using the average of minimum concentrations (Table 1). Asterisked (*) values denote input FTOH/sFTOH concentrations which were above calibration levels developed for this study and therefore assumed to be at the highest calibration concentration. Consequently, these findings should be viewed as minimum values which conservatively estimate the magnitude of PFAS mobility in leachate versus LFG.

estimates, based on limited data, suggest that most PFAS mass mobilized from landfills releases through leachate (~62%).[9] However, our data from Landfill 1, showing over 76% fluorine release in LFG, along with substantial masses released by LFG in Landfills 2 and 3 (at minimum 40% and 32%, respectively), indicate that LFG may serve as an equal, likely greater, conduit of PFAS mobility from landfills than leachate, concurring with previous reactor studies on FTOH volatilization and neutral/ionic PFAS assessments of select waste materials.[17,75,76]

At least 79 to 92% of the fluorine mass in LFGs were derived from FTOH/sFTOH classes, with minimal contribution from FTOs, FTOAcs, and FTMAcs. In this conservative assessment, fluorine from LFG surpassed leachate in only Landfill 1. Although actual fluorine emission from LFG is higher than reported here, the elevated ratio of gas-to-leachate generation at Landfill 1 likely caused this difference (Table S-9). Landfill 2, the largest site, demonstrated the highest combined fluorine release from leachate and LFG, followed by Landfill 3, and then Landfill 1, corresponding to descending waste mass in place at each location.

4. IMPLICATIONS

This study provides fundamental data about neutral PFAS in LFG from MSW landfills. Unexpectedly, FTOH/sFTOH detections in LFG from this study exceeded implemented calibration levels; subsequent research should deploy shorter sampling durations. Regardless, even under more conservative assumptions these findings suggest that LFG, largely unscrutinized for PFAS, contains similar or greater magnitudes of PFAS compared to leachate, mostly attributed to midlength FTOH homologues. As landfills can be viewed as unabating PFAS repositories, the significance of LFG management in mitigating the long-term, long-range atmospheric transport of

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neutral PFAS, and subsequently derived PFAAs, cannot be understated. Unlike landfill leachate, LFG collection systems (when in place) are not fully efficient, collecting an estimated ~50−70% of generated biogases.[77] Though this is a considerable collection efficiency of biogas and presumably neutral PFAS, management of captured LFG fractions varies globally, from no treatment to degrees of carbon filtration and thermal treatment (i.e., flaring, advanced renewable natural gas technologies). Because the feasibility of PFAS destruction through thermal treatment remains unclear, research is needed to determine the treatment/removal efficiency of existing LFG management technologies. Considering the range of LFG capture efficiency, the retention and emission of neutral PFAS via fugitive emissions (i.e., migration through the waste layer) should also be examined, along with the role of landfill waste type, age, and temperature in neutral PFAS variability.

■ ASSOCIATED CONTENT

Supporting Information

The Supporting Information is available free of charge at https://pubs.acs.org/doi/10.1021/acs.estlett.4c00364.

Acronyms, structures, and instrumental parameters for neutral PFAS via GC-HRMS and ionic PFAS via LC-MS/MS; detailed QC information and results; and fluorine mass balance/extrapolation details and assumptions (PDF)

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Notes

The authors declare no competing financial interest.

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Short-chain perfluoroalkyl acids: environmental concerns and a regulatory strategy under REACH

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Abstract

Background: Short-chain PFASs (per- and polyfluoroalkyl substances) are widely used as alternatives to long-chain PFASs. Long-chain PFASs become gradually regulated under REACH (EC No. 1907/2006) and other international regulations, due to having persistent, bioaccumulative and toxic properties and/or being toxic for reproduction. The increasingly used short-chain PFASs are assumed to have a lower bioaccumulation potential. Nonetheless, they have other properties of concern and are already widely distributed in the environment, also in remote regions. The REACH Regulation does not directly address these emerging properties of concern, complicating the implementation of regulatory measures. Therefore, this study illustrates these environmental concerns and provides a strategy for a regulation of short-chain PFASs within REACH.

Results: Short-chain PFASs have a high mobility in soil and water, and final degradation products are extremely persistent. This results in a fast distribution to water resources, and consequently, also to a contamination of drinking water resources. Once emitted, short-chain PFASs remain in the environment. A lack of appropriate water treatment technologies results in everlasting background concentrations in the environment, and thus, organisms are permanently and poorly reversibly exposed. Considering such permanent exposure, it is very difficult to estimate long-term adverse effects in organisms. Short-chain PFASs enrich in edible parts of plants and the accumulation in food chains is unknown. Regarding these concerns and uncertainties, especially with respect to the precautionary principle, short-chain PFASs are of equivalent concern to PBT substances. Therefore, they should be identified as substances of very high concern (SVHC) under REACH. The SVHC identification should be followed by a restriction under REACH, which is the most efficient way to minimize the environmental and human exposure of short-chain PFASs in the European Union.

Conclusion: Due to an increasing use of short-chain PFASs, an effective regulation is urgently needed. The concerns of short-chain PFASs do not match the "classical" concerns as defined under REACH, but are not of minor concern. Therefore, it is of advantage to clearly define the concerns of short-chain PFASs. This might facilitate the following restriction process under REACH.

Keywords: REACH, SVHC, PFASs, Candidate list, Restriction, Regulation, Per- and polyfluoroalkyl substances, PFAAs

Background

Some representatives of per- and polyfluoroalkyl substances (PFASs) are highly stable organic compounds, due to the C–F bond being one of the strongest in organic chemistry [1]. Compared to hydrocarbons, PFASs have enhanced technical properties, such as a higher surface activity and better dielectric properties, and possess at the same time a higher thermal stability, an increased chemical resistance and a physiological inertness [2]. Due to these properties, PFASs are well suited for manifold applications and have been used extensively in various industrial and consumer applications since the 1950s, e.g., impregnation of textiles, paper, hard metal plating,

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in paints, the production of fluoropolymers, etc. [3, 4]. During the production and life cycle of the products, a certain fraction of PFASs is inevitably emitted into the environment (e.g., [5]). Many commonly used PFASs can degrade under environmentally relevant conditions to perfluoroalkyl acids (PFAAs), regarded as final degradation products (e.g., [6]). Besides the direct emission of PFAAs, e.g., from production and life cycle of certain fluoropolymers, indirect emissions via degradation of the precursor substances are of high relevance regarding the global emissions [7]. Also, side-chain fluorinated polymers, in particular the group of side-chain fluorotelomer-based polymers, were shown to be degraded to PFAAs under environmental conditions in the range of decades to centuries [8] and are therefore considered to be precursors. As highly persistent final degradation products of many other PFASs, PFAAs are a relevant group regarding environmental and human exposure, even though being a small substance group within the whole PFASs group, consisting of at least 3000 substances [9].

The concern of possible risks of PFAAs to the environment and human health was broadly unrecognized until the turn of the millennium, when two groups of PFAAs, long-chain perfluoroalkyl carboxylic acids and long-chain perfluoroalkane sulfonic acids, have been shown to be ubiquitously present in biota and humans (e.g., [10, 11]). Since becoming aware of the ubiquitous distribution and the hazardous properties (see [12]) of long-chain PFASs (i.e., long-chain PFAAs and their precursors), voluntary measures of manufacturers and regulatory action have been taken place to reduce the emissions. Perfluorooctane sulfonic acid (PFOS), its salts and perfluorooctane sulfonyl fluoride (PFOSF) are listed as persistent organic pollutant (POP) under the Stockholm Convention and perfluorooctanoic acid (PFOA) and related precursors are proposed as POP. Under the European Chemicals Regulation (REACH EC No. 1907/2006), several long-chain perfluoroalkyl carboxylic acids (C_8–C_{14}) and the long-chain perfluorohexane sulfonic acid are identified as substances of very high concern (SVHC) and included in the REACH candidate list. In the EU restriction, proposals for the perfluoroalkyl carboxylic acids on the candidate list are in progress, but only implemented for PFOA its salts and related substances up to the present. In summary, global measures to ban long-chain PFAAs of concern are ongoing and need to be continued.

With the regulation of long-chain PFASs and due to voluntary measures, non-regulated PFASs have been more extensively used as alternatives. Information of these replacements are not easily accessible, however, it is known that short-chain PFASs (i.e., short-chain PFAAs and respective precursors, see Fig. 1 for terminology) are important and broadly used alternatives [13]. Due to historical emissions mainly as unintended by-products during the manufacturing of long chain PFASs in the past, short-chain PFASs are already present in environmental compartments [7, 14]. The replacement of long-chain PFASs with short-chain PFASs and in general other PFASs is highly criticized [15–17], as short-chain PFASs have also relevant properties of concern.

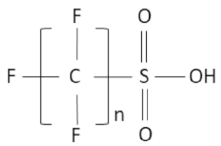
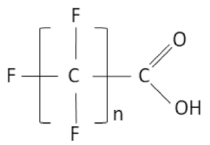
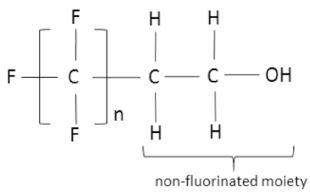
Perfluoroalkane sulfonic acids Long-chain: n ≥ 6 Short-chain: n < 6 Example short-chain representative: PFBS (n = 4)	
Perfluoroalkyl carboxylic acids Long-chain: n ≥ 7 Short-chain: n < 7 Example short-chain representatives: PFBA (n = 3), PFHxA (n = 5)	
Example of short-chain precursors 4:2 fluorotelomer alcohol (n = 4) 6:2 fluorotelomer alcohol (n = 6)	

Fig. 1 Terminology of per- and polyfluoroalkyl substances

Long- and short-chain PFAAs were originally distinguished, since short-chain PFAAs are assumed to have a lower bioaccumulation potential and improved environmental properties compared to long-chain PFAAs [3, 18]. However, short-chain PFAAs are as persistent as long-chain PFAAs have different, but not less alarming properties of concern, and are already widely distributed in the environment [e.g., 19–27]. The awareness of these concerns needs to be raised to obtain a fast regulation of short-chain PFASs. Therefore, this study (i) summarizes and highlights the concerns of short-chain PFAAs, (ii) investigates the need for a regulation and (iii) provides a strategy for how a regulation could be implemented in REACH.

Results and discussion

Uses and sources of environmental exposure

According to the publicly available information on REACH registrations, short-chain PFAAs, e.g., PFBA (perfluorobutanoic acid), PFBS (perfluorobutanesulfonic acid) and PFHxA (perfluorohexanoic acid) have not been registered so far. The last registration deadline under REACH for substances which are manufactured or imported between one and 100 tons per year is 31 May 2018. However, registration obligations do not include substances in imported articles. In contrast to the short-chain PFAAs, various short-chain precursor substances are registered in high tonnages (> 100 tons per annum) and are mainly used for manufacturing polymers. Those polymers may most likely have wide dispersive uses. In general, short-chain PFASs can substitute long-chain PFASs in manifold applications [28]. Therefore, they are used comparably to long-chain PFASs in many consumer products and industrial applications requiring inert or repellent characteristics. Due to a partially limited technical performance in comparison with long-chain PFASs, higher amounts of short-chain PFASs are usually used in compensation (e.g., in wax and polish) [28]. During the production and life cycle of the products, inevitably releases of short-chain PFASs into the aquatic and terrestrial environment occur [29]. There are vast amounts of applications, but only scarce information on uses of short-chain PFASs. Additionally, the degradation of side-chain fluorinated polymers might contribute significantly to emissions of short-chain PFASs in the long-term, as demonstrated by an environmental fate modeling analysis by Li et al. [30]. Besides degradation of the polymers in the long-term, non-bound residues can easily reach the environment and may thus contribute to future releases [29, 30]. Due to the hundreds of various precursors, assessing the uses of all precursors is nearly impossible. The Swedish Chemicals Agency [4], the Norwegian Environment Agency [31] and the Danish Ministry of the Environment [32] provided detailed information on the use of short-chain PFASs to the extent possible. In the following, information for some use areas is summarized.

In recent years, surfactants in foam-based fire-fighting agents, used to extinguish flammable liquids or gases, shifted from long-chain PFASs (mainly C8-based) to short-chain PFASs (mainly C6-based). The stability and surface-active properties of PFASs enable a film formation on top of burning liquids, extinguishing the fire and preventing burn back. Nowadays, a number of foam-based fire-fighting agents contain short-chain PFASs. The use of fire-fighting agents results in direct contaminations of the environment when there is no possibility of collecting the fire water and subsequent disposal [33, 34].

Textiles, leather, carpet, apparel, and upholstery are desired to be oil, water and dirt repellent. For some papers and food contact materials, greaseproof properties are required. Therefore, these items can be impregnated with side-chain fluorinated polymers, consisting of non-fluorinated carbon backbones connected to short-chain PFAS side-chains such as fluorotelomer alcohols.

Short-chain PFASs are furthermore used for example as mist suppressing agents in hard metal-plating [35], in the electronic industry due to insulating and water repellent properties [36], in cosmetic products to make the products oil and water repellent [37], in paints and inks to improve wetting, smoothness and flow [36], in medical devices [4] and the oil production [32].

Overview of concerns of short-chain PFAAs

This section specifies concerns attributed in general to short-chain PFAAs. The properties of concerns only attributed to certain PFAAs or precursor substances are not addressed. However, precursors share the common concern of degradation to short-chain PFAAs as final degradation products. It is therefore possible to include precursors in a regulation of short-chain PFAAs (see "Strategy for regulation of short-chain PFASs under REACH").

Short-chain PFAAs are extremely persistent

Based on the high energy of the carbon–fluorine bond [2], it can be assumed that short-chain PFAAs are extremely persistent similar to the persistence of long-chain PFAAs [38, 39]. They do not undergo abiotic or biotic degradation at all under environmental conditions and are considered highly stable transformation products in which several precursors ultimately degrade into [13, 40]. This extreme persistence is regarded as an incalculable hazard itself, as short-chain PFAAs will stay in the environment for decades to centuries [41]. The possible risks of such extremely persistent organic fluorochemicals for humans and the environment have been

emphasized by leading scientists [16, 17]. Each released PFAS molecule remains in the environment, meaning that it is impossible to reverse exposure. Thus, levels of these substances will most probably increase over time in certain compartments. As a consequence, long-term effects on humans and wildlife may be possible in the future, when certain effect thresholds are reached.

Short-chain PFAAs have a low adsorption potential and are very mobile

The determination of the physicochemical properties of short-chain PFAAs requires, due to the unusual substance properties, specific methods. Estimates of the physicochemical properties are available (PFBS, PFBA and PFHxA [42, 43]: logK_{ow} (neutral form) = 2.82–4.6, water solubility > 20 g/L, logK_{oa} = 6–6.7, pK_a < 1, log-K_{oc}: 2.7–3.6), showing that short-chain PFAAs are very mobile, which is also confirmed by their environmental distribution [44, 45]. Due to this mobility short-chain PFAAs effectively reach water bodies which is of special concern regarding human exposure: Drinking water resources are highly sensitive to contamination with short-chain PFAAs [19, 46]. Due to the low adsorption potential, short-chain PFAAs will not bind to particles and stay mainly dissolved in the water phase.

In a larger scale, once emitted short-chain PFAAs can only hardly be removed from water

Long-chain PFAAs can be removed from water with activated carbon filters. However, this is not as effective for short-chain PFAAs due to the low adsorption potential [47]. Hence, at a large scale short-chain PFAAs can only hardly, if at all, be removed from the environment with the main methods available today [46, 48, 49]. Some promising methods are very costly and just applied at a laboratory scale [50]. The absence of effective measures on a larger scale is problematic in particular with regard to contaminated drinking water reservoirs. Since short-chain PFAAs are not expected to degrade chemically and biologically, the concentrations in contaminated areas will only decline with a further spatial distribution, provided that no further release occurs.

Short-chain PFAAs have the potential for long-range transport. Already today, short-chain PFAAs are monitored in remote regions and show a wide spread distribution in biotic and abiotic compartments

Due to the high mobility caused by the high aqueous solubility and relatively low adsorption potential of short-chain PFAAs, they have a higher potential for long-range transport compared to the long-chain homologues [43]. Additionally, volatile precursors are likely to be transported to remote regions via the atmosphere and then being degraded to short-chain PFAAs. Monitoring data show that already today short-chain PFAAs are present in remote areas [20, 51, 52] and have a wide spread distribution in biotic and abiotic compartments. Table 1 shows examples of monitoring data from Europe. In a few cases, increasing concentrations of short-chain PFAAs in the environment and biota are already observed; for example, for PFBS in dolphins from the South China Sea [53], for PFHxA in water samples near a fluoropolymer production plant in Japan [54] and for PFBA in snow from remote regions in the European Alps [25, 52]. As demonstrated in a recent review by Land et al. [55], a generalized prediction of temporal trends in environmental concentrations of short-chain PFAAs is currently not feasible due to limited monitoring data and insignificant time trends shown in most of the studies. Historical releases of short chain PFASs are hard to quantify [7]. Because the substances are not degradable, historical releases, i.e., as unintended by-products during the manufacturing of long chain PFASs contributed to the environmental stocks to a reasonable amount and impede the observation of significant trends. However, due to the phase-out of long-chain PFASs [18, 56], manufacturing and use of short-chain PFAAs and related substances are very likely to further increase in the near future. Thus, emissions of short-chain PFAAs will increase as well. Along with the expected increasing emissions, short-chain PFAAs will further enrich in the environment leading to increased background concentration levels, in the long-term especially in the aquatic systems [57].

The permanent exposure to short-chain PFAAs results in continuous and poorly reversible concentrations in organisms. There is a risk of adverse effects on humans and the environment, which will rise with increasing exposure

Considering that the exposure to short-chain PFAAs is unlikely to be stopped shortly, there will be increasing continuous and poorly reversible environmental background concentrations of short-chain PFAAs. Consequently, organisms and humans will be permanently exposed to short-chain PFAAs, resulting in continuous and poorly reversible internal concentrations. The poorly reversible internal concentrations in organisms are caused by the persistence of short-chain PFAAs and their continuous presence in the environment. Therefore, the organismal elimination efficiencies are of secondary relevance. This becomes more evident when considering that the half-lives of short-chain PFAAs in the environment exceed their half-lives in the organisms by far. Cousins et al. [41] addresses this approach in more detail, showing exemplary that the contamination of drinking

Table 1 Example monitoring data from Europe regarding PFBS, PFBA and PFHxA

Sampling	Short-chain PFAA and concentration[a]	Reference/sampling year
Tab water [ng/L]		
France, 8 locations	PFBS: 3.2 (32%);	[19]
	PFHxA: nd	Sampling year: 2014
Germany, 26 locations	PFBS: 2.7 (42%)	[21]
	PFHxA: 2 (23%)	
	PFBA 2 (19%)	
Spain, 84 locations	PFBA: 10 (52%)	[22]
	PFHxA: 4.7 (18%)	Sampling year: 2010–2011
	PFBS: 8.3 (35%)	
Surface water [ng/L]		
Spain, Ebro	PFBA: 35.2 (58%)	[23]
	PFHxA: 1.7 (8%)	Sampling year: 2010
	PFBS: nd	
Spain, Guadalquivir	PFBA: 214.3 (92%)	
	PFHxA: nd	
	PFBS: 10.1 (8%)	
Germany, Elbe	PFBA: 2.6 (100%)	[24]
	PFBS: 7.5 (100%)	Sampling year: 2015
	PFHxA: 1.5 (100%)	
Germany, Saale	PFBA: 0.5 (100%)	
	PFBS: 4.3 (100%)	
	PFHxA: 4 (100%)	
Snow [ng/L]		
European Alps	PFBA: 0.69	[25]
	PFHxA: 0.06	Sampling year: 2008
Sediments [ng/g dw]		
Baltic sea	PFBS: 0.00017 (75%)	[26]
	PFHxA: 0.115 (100%)	Sampling year: 2013–2014
Organisms [ng/g ww]		
Baltic sea		[26]
Zooplankton	PFBS/PFHxA: nd	Sampling year: 2013–2014
Herring	PFBS/PFHxA: nd	
Sprat	PFBS/PFHxA: nd	
Guillemot egg	PFBS: 0.0035 (100%)	
	PFHxA: 0.0026 (50%)	
Ingolstadt, Germany		[27]
Wild boar	PFHxA: 0.49 (66%)	Sampling year: 2011–2012
	PFBA: 0.73 (100%)	
	PFBS: 0.25 (53%)	
Svalbard, Norway		[20]
Reindeer	PFBA: 0.42 (56%)	
Ebro, Spain		[23]
Fish	PFBS: 4.9 (69%)	Sampling year: 2010
	PFBA: 0.6 (31%)	
	PFHxA: 268.4 (56%)	

[a] Mean values, including frequency detected

water resources with short-chain PFAAs leads to a poorly reversible exposure in humans, comparably to a contamination with long-chain PFAAs. Due to the high mobility of short-chain PFAAs and the missing suitable water treatment technologies for drinking water, short-chain PFAAs will remain in the water resources over time (this scenario proves true, see "Case example Rastatt"). This results in a permanent exposure via drinking water, independently of the elimination half-lives of short-chain PFAAs.

Nevertheless, elimination half-lives of short-chain PFAAs are non-negligible: Depending on the species, they range from a couple of hours to several days in some mammals [58–60], with longer half-lives in humans (e.g., 30 days for PFHxA [61]). The elimination half-lives are shorter compared to some long-chain homologues, but nevertheless of concern, as short-chain PFAAs will still remain in organisms with continuous exposure. Some short-chain PFAAs have considerable protein binding potentials [62, 63]. A high protein binding potential is of toxicokinetical relevance: the blood can easily distribute short-chain PFAAs within the body resulting in a potential to enrich particularly in blood rich tissues. Data from the exposure of mammals show that short-chain PFAAs can be widely distributed in the organism [64, 65].

Possible adverse effects, especially long-term effects, caused by a high protein binding potential are uncertain. In general, there is a high level of uncertainty, if the permanent exposure with low concentrations of short-chain PFAAs may cause adverse effects in organisms. There is insufficient data on the toxic properties of short-chain PFAAs [66], and therefore, sub-lethal long-term effects cannot be excluded. There might be unconsidered sensible organisms, certain development stages or effects on multiple generations, being not considered when assessing the toxicity of short-chain PFAAs. Mixture toxicity of different PFASs or the toxicity of PFASs in combination with natural and anthropogenic stressors is largely unknown [57].

The distribution of short-chain PFAAs in terrestrial systems and along food chains is unknown. Short-chain PFAAs are known to enrich in edible parts of plants

Short-chain PFAAs are known to enrich in plants and due to their high water solubility and low adsorption potential, especially in leaves and fruits [67–69]. This enrichment in the edible parts of plants is higher compared to long-chain PFAAs and might pose a risk regarding a distribution along the food chain, which has not yet been investigated. In contaminated soil, especially in former arable land, the accumulation in the edible parts of plants is of concern due to human exposure (exemplary shown in "Case example Rastatt"). Field studies do not allow drawing conclusions on distributions of short-chain PFAAs along the food chains. Especially, the accumulation in terrestrial organisms, as shown for long-chain PFAAs [70], cannot be excluded [71].

Case example Rastatt

In the surroundings of Rastatt (Baden-Wuerttemberg, Germany), 480 hectares of former arable land are contaminated with short-chain PFAAs and precursors. The pollution was detected in 2013 and has probably been caused by the longstanding application of compost mixed with sludge from paper production, contaminated with various precursors. Over time, the precursors contained in the soil degraded to short-chain PFAAs and were enriched in plants. Local authorities derived thresholds for short-chain PFAAs in food (in the µg/kg range) [72], based on guidance values for drinking water [73]. Pre-harvest monitoring showed that the concentration of short-chain PFAAs in some crops exceeded these thresholds, preventing the use as food. Crops enriching high amounts of short-chain PFASs are recommended not to be cultivated for consumption on these contaminated fields (e.g.,asparaguses, strawberries and other vegetables). Despite the recommendations for cultivation, threshold values are in parts exceeded to date. This might be due to a strong influence of insufficiently predictable abiotic factors, such as organic carbon content, on the enrichment of short-chain PFASs in plants [74]. Over time, the very mobile short-chain PFASs and precursors in the soil also wash out into the groundwater. Irrigating crops with contaminated water are causing further emissions to soil and uptake into plants. Two groundwater wells for drinking water production had to be closed [72].

For several years, the dimension of the contamination and the problems arising with the contamination of short-chain PFASs has not been recognized. Estimations addressing all adverse environmental effects and socio-economic costs resulting from the contamination are not available, but according to information given by the water work solely the cost for water treatment with charcoal filters amounts to several million Euros. Until now, no practicable solution for removing the short-chain PFASs from the soil or groundwater has been found. Effective solutions, such as a removal and replacement of the top soil or "pump and treat" methods, would be far too expensive. Furthermore, there are still high concentrations of short-chain PFASs and unknown precursors contained in the soil and there is no applicable solution for stopping them from reaching the groundwater. Research to find an appropriate solution is ongoing.

This Rastatt case clearly shows that once emitted to the environment, short-chain PFAAs cause irreversible contaminations, and thus causing high socio-economic costs and threats to man and environment. Besides Rastatt,

there is a large scale contamination of drinking water with short-chain PFASs in Uppsala known. The contamination in Sweden is mainly caused by the use of aqueous film forming fire-fighting foams at a military airport [33].

Strategy for regulation of short-chain PFASs under REACH

PFAAs themselves are not produced in high tonnages. Primarily manifold precursor substances degrading in the environment to PFAAs are responsible for their emission to the environment and subsequently to humans. Those precursor substances are defined by a similar length of the perfluorinated carbon chain compared to the short-chain PFASs, connected to a non-fluorinated moiety (see Fig. 1).

A regulation of every single precursor is impracticable due to the unknown but likely vast amount of precursors on the market. For an effective regulatory measure, it is mandatory to address both PFAAs and their precursor substances, considering also emissions from imported articles. These considerations have already been outlined by Vierke et al. [39] regarding the long-chain perfluoroalkyl carboxylic acid PFOA. Precursors registered under REACH are mainly used for manufacturing polymers according to industry. Those polymers (i.e., side-chain fluorinated polymers) contain unbound residues of PFASs precursors or the polymers may breakdown over time and release PFAS precursors [8]. Thus, polymers are included in such a regulation as well.

In short, as a first step identification as SVHC according to Article 57 of the REACH regulation needs to be considered. Such substances become included in the REACH candidate list. Article 57 of the REACH Regulation allows to include substances on the candidate list that (i) are classified as carcinogenic, mutagenic or toxic for reproduction (CMR) category 1A or 1B, (ii) have persistent, bioaccumulative and toxic (PBT) or (iii) very persistent and very bioaccumulative (vPvB) properties according to REACH (Annex XIII), or (iv) have an equivalent level of concern as CMR or PBT/vPvB substances. The inclusion of a substance in the candidate list results in obligations for the manufacturer to provide information to downstream users and to consumers (if > 0.1% proportion in articles). This regulatory measure alone is not sufficient to control the emissions of short-chain PFASs. But the inclusion into the candidate list is a strong signal itself since it contains all substances of very high concern identified within the EU. For substances on the candidate list REACH foresees authorisation as the regulatory measure. In such case, using the substance is only allowed if authorization was granted. However, an authorisation does apply only for the substance itself and does not include any precursor substances. Furthermore, the authorisation procedure does not include imported articles, probably contributing to the emissions of short-chain PFASs in Europe. Consequently, starting the authorisation procedure would not constitute an effective risk management option. A restriction (Article 67) can cover the placing of a substance on the market, the content of a substance in (imported) articles, the use of such a substance on its own or in a mixture and respective precursors. Thus, a general restriction can address all relevant emission routes and is considered the appropriate regulatory measure for short-chain PFASs under REACH. To decide if certain exemptions might be appropriate, further detailed information on uses and alternatives are necessary. Once in force, the restriction is valid for the European Economic Area (EEA), having a population of more than 500 million people.

The equivalent level of concern

With the current knowledge, short-chain PFASs seem not to fulfill the bioaccumulation or toxicity criteria of REACH Annex XIII. Due to the special properties of short-chain PFASs, they might not fulfill the "classical" properties of concern of SVHCs, being clearly defined within Article 57(d and e) of the REACH Regulation. However, given the properties of concern (see "Overview of concerns of short-chain PFAAs") and the already existing pollutions (see "Case example Rastatt"), there is an unequivocal need for regulation of short-chain PFASs. Within the REACH Regulation, it is possible to identify substances as SVHC that have an equivalent level of concern compared to CMR or PBT/vPvB substances, as defined in article 57(f) (see "Strategy for regulation of short-chain PFASs under REACH"). So far, the equivalent level of concern approach is applied for endocrine disrupting chemicals, sensitisers and specific target organ toxicity after repeated exposure only. However, in our opinion short-chain PFASs are also of equivalent level of concern compared to PBT/vPvB substances and can be identified as SVHC for the following reasons:

Article 1 of the REACH Regulation states that the provisions of the REACH Regulation are underpinned by the precautionary principle. The realization of the precautionary principle is apparent when looking at the vPvB criteria for the SVHC identification (Article 57(e)). Even though a substance does not cause any toxic effect, it can be identified as SVHC because it is due to its intrinsic properties expected to remain in the environment and to accumulate in organisms. The reasoning is that (i) the level of uncertainty in identifying long-term effects cannot be estimated with sufficient accuracy (vB) and (ii) consequences of an underestimation of adverse effects are not easily reversible by regulatory action (vP), i.e., the effect is occurring or is likely to occur at a certain point in time, and even if there is immediate regulatory

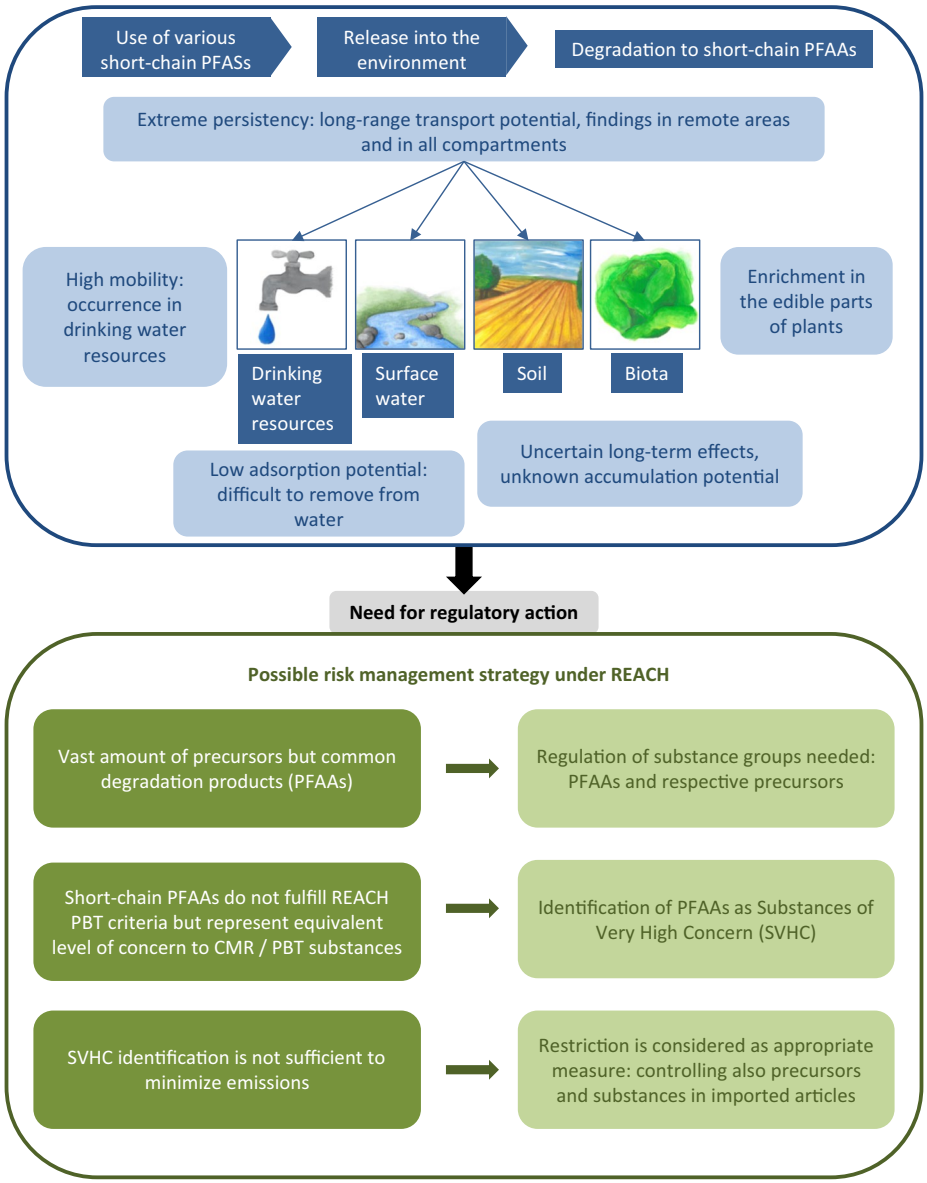


Fig. 2 Concerning properties of short-chain PFASs and a possible risk management strategy under REACH

action to prevent further emission, the adverse effects will continue. The extreme persistence and other intrinsic properties (e.g., low adsorption potential) of short-chain PFASs unquestionable fulfill the second concern. In practice, this is shown in the "case example Rastatt". The first issue, that long-term effects cannot be estimated with sufficient accuracy, is in case of vPvB substances met by the very high bioaccumulation potential. Short-chain PFASs do not fulfill the criteria according to Annex XIII of the REACH Regulation for being bioaccumulative. However, as they will not degrade in the environment, there is a permanent and irreversible exposure of organisms. With ongoing emission, the environmental concentrations of short-chain PFASs will most probably increase, due to the lack of degradation and the fact that emerging emissions come on top of already released amounts resulting in an accumulation in certain environmental compartments. There is a high uncertainty about the toxicity of short-chain PFASs, especially regarding long-term effects (see "Overview of concerns of short-chain PFAAs"). If emissions continue and environmental concentrations of short-chain PFASs exceed a certain limit, it seems very likely that adverse effects may occur. Even if the releases of short-chain PFASs will stop

immediately, the already emitted short-chain PFASs do irreversibly remain in the environment (see "Case example Rastatt"). These considerations and the additional concerns, such as enrichment in the edible parts of plants and the accumulation in water resources (see "Overview of concerns of short-chain PFAAs"), show that short-chain PFASs are of equivalent level of concern compared to PBT/vPvB substances.

Conclusion

Short-chain PFASs are increasingly used and are considered to have properties of very high concern according to Article 57(f) of the REACH Regulation: They are of equivalent level of concern to PBT or vPvB substances. The outlined concerns related to short-chain PFASs, the already existing pollution, i.e., locally restricted very high pollutions (e.g., Rastatt and Uppsala) and global pollutions due to historical emissions, and very likely further increasing concentrations in the environment and biota illustrate that a fast regulation is mandatory. In accordance with the regulation of long-chain PFASs, the most effective way constitutes an SVHC identification of the short-chain PFAAs followed by a restriction, allowing also the regulation of precursors. The SVHC identification is not a prerequisite but will be advantageous, as the concerns attributed to short-chain PFASs do not comply with the "classical" concerns considered in the REACH Regulation and need to be clearly defined. In addition to a regulation by authorities, voluntary measures by industry could contribute substantially in reducing the emissions of short-chain PFASs.

Methods

A literature review was conducted to obtain available information on short-chain PFASs. The concerns of short-chain PFASs and its equivalent level of concern with regard to the REACH Regulation were discussed and the concern was specified at a workshop in Berlin, organized by the German Environment Agency (UBA) in October 2016. Regulatory experts and scientists from different EEA Member States, the European Commission, the European Chemicals Agency, USA and Australia participated.

Abbreviations

CMR: carcinogenic, mutagenic or toxic for reproduction; EEA: European Economic Area; K_{OA}: octanol–air partition coefficient; K_{OC}: organic carbon–water partition coefficient; K_{OW}: octanol–water partition coefficient; PBT: persistent, bioaccumulative and toxic; PFAA: perfluoroalkyl acid; PFASs: per- and polyfluoroalkyl substances; PFBA: perfluorobutanoic acid; PFBS: perfluorobutane sulfonic acid; PFHxA: perfluorohexanoic acid; PFOA: perfluorooctanoic; PFOS: perfluorooctane sulfonic acid; pK_a: acid dissociation constant; POP: persistent organic pollutant; REACH: European Chemicals Regulation, EC No. 1907/2006, Registration, Evaluation and Authorization of Chemicals; SVHC: substance of very high concern; vPvB: very persistent and very bioaccumulative.

Authors' contributions

SB, ÉF, CS, LV and ABE contributed in equal parts to this publication. This paper does not necessarily reflect the opinion or the policies if the German Federal Environment Agency. All authors read and approved the final manuscript.

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Competing interests

The authors declare that they have no competing interests.

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All relevant data are available in the references cited.

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Review of the fate and transformation of per- and polyfluoroalkyl substances (PFASs) in landfills☆

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ABSTRACT

A critical review of existing publications is presented i) to summarize the occurrence of various classes of per- and polyfluoroalkyl substances (PFASs) and their sources in landfills, ii) to identify temporal and geographical trends of PFASs in landfills; iii) to delineate the factors affecting PFASs in landfills; and iv) to identify research gaps and future research directions. Studies have shown that perfluoroalkyl acids (PFAAs) are routinely detected in landfill leachate, with short chain (C4-C7) PFAAs being most abundant, possibly indicating their greater mobility, and reflecting the industrial shift towards shorter-chain compounds. Despite its restricted use, perfluorooctanoic acid (PFOA) remains one of the most abundant PFAAs in landfill leachates. Recent studies have also documented the presence of PFAA-precursors (e.g., saturated and unsaturated fluorotelomer carboxylic acids) in landfill leachates at concentrations comparable to, or higher than, the most frequently detected PFAAs. Landfill ambient air also contains elevated concentrations of PFASs, primarily semi-volatile precursors (e.g., fluorotelomer alcohols) compared to upwind control sites, suggesting that landfills are potential sources of atmospheric PFASs. The fate of PFASs inside landfills is controlled by a combination of biological and abiotic processes, with biodegradation releasing most of the PFASs from landfilled waste to leachate. Biodegradation in simulated anaerobic reactors has been found to be closely related to the methanogenic phase. The methane-yielding stage also results in higher pH (>7) of leachates, correlated with higher mobility of PFAAs. Little information exists regarding PFAA-precursors in landfills. To avoid significant underestimation of the total PFAS released from landfills, PFAA-precursors and their degradation products should be determined in future studies. Owing to the semi-volatile nature of some precursor compounds and their degradation products, future studies also need to include landfill gas to clarify degradation pathways and the overall fate of PFASs.

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1. Introduction

Landfilling is one of the most common disposal methods for end-of-life consumer products (Renou et al., 2008). Engineered landfills are designed to contain solid waste and collect landfill leachate, while preventing migration of the contaminants to groundwater. Among the emerging contaminants, per- and polyfluoroalkyl substances (PFASs), detected in landfill leachate, are receiving attention due to their persistence, bioaccumulation potential and adverse effects on biota and humans (Houde et al., 2011). PFASs are a diverse group of aliphatic compounds containing one or more perfluoroalkyl moiety (C_nF_{2n+1}-). PFASs containing at least one perfluoroalkyl moiety are called polyfluoroalkyl substances (e.g., $CF_3CF_2CH_2COOH$). Perfluorinated substances are defined as aliphatic substances for which all of the H atoms attached to C atoms in the nonfluorinated substance from which they are notionally derived have been replaced by F atoms, except the H atoms present in any functional groups (Fig. S1 in supplemental information (SI)) (Buck et al., 2011).

Due to their unique surface-active properties and high chemical and thermal stability (Buck et al., 2011), PFASs are widely used in numerous consumer products (e.g. textiles, paper, non-stick cookware, carpets, cleaning agents) and industrial applications (e.g., metal plating, fire-fighting foams, electronics production,

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Abbreviations

DiPAP	Disubstituted fluorotelomer phosphate esters
EtFOSAA	Ethyl-perfluorooctane sulfonamide acetic acid
EtFOSE	Ethyl-perfluorooctane sulfonamidoethanol
FASA	Perfluoroalkane and *N*-alkyl perfluoroalkane sulfonamide acetic acid
FOSA	Perfluoroalkane and *N*-alkyl perfluoroalkane sulfonamide
FOSE	Perfluoroalkane and *N*-alkyl perfluoroalkane sulfonamidoethanols
FTCA	Fluorotelomer saturated carboxylic acid
FTI	Fluorotelomer iodide
FTOH	Fluorotelomer alcohol
FTP	Fluorotelomer polymer
FTSA	Fluorotelomer sulfonate
FTUCA	Fluorotelomer unsaturated carboxylic acid
MeFBSAA	Methyl-perfluorobutane sulfonamide acetic acid
MSW	Municipal solid waste
NF	Nanofiltration
PAP	Polyfluorinated phosphate ester
PEPE	Perfluoropolyether
PFAA	Perfluoroalkyl acid
PFAS	Per- and polyfluoroalkyl substance
PFBA	Perfluorobutanoic acid
PFBS	Perfluorobutane sulfonic acid
PFCA	Perfluoroalkyl carboxylic acid
PFDA	Perfluorodecanoic acid
PFHpA	Perfluoroheptanoic acid
PFHxA	Perfluorohexanoic acid
PFHxS	Perfluorohexane sulfonic acid
PFNA	Perfluorononaic acid
PFOA	Perfluorooctanoic acid
PFOS	Perfluorooctane sulfonic acid
PFPA	Perfluoroalkyl phosphonic acids
PFPeA	Perfluoropentaonoic acid
PFPiA	Perfluoroalkyl phosphinic acids
PFPrA	Pentafluoropropionic acid
PFSA	Perfluoroalkyl sulfonic acid
POP	Persistent organic pollutant
POSF	Perfluorooctane sulfonyl fluoride
RO	Reverse osmosis
SI	Supplemental Information
TOC	Total organic carbon
UF	Ultrafiltration
WWTP	Wastewater treatment plant

photography) (Arvaniti et al., 2014; Kissa, 2001). Among the most commonly detected perfluoroalkyl acids (PFAAs) in the environment, pefluorooctane sulfonate (PFOS) has been listed under Annex B of the Stockholm Convention on persistent organic pollutants (POPs) since 2009, restricting its production and use, except for a few exemptions; perfluorooctanoic acid (PFOA) is currently under review by the POPs Review Committee of the Stockholm Convention (Stockholm Convention, 2017). While PFAAs may be directly released into the environment during production, usage and disposal, polyfluoroalkyl substances – the "PFAA-precursors" – can also be transformed abiotically or biologically into PFAAs (see Fig. S1 in SI).

A variety of consumer products (e.g., paper, textiles, carpets) and packaging containing PFAAs and their precursors are sent to municipal landfills at the end of their useful lives. In many municipalities, biosolids containing PFASs are also landfilled (Guerra et al., 2014; Arvaniti et al., 2012). Following disposal, PFASs are released from the waste through biological and abiotic leaching (e.g., desorption) (Allred et al., 2015), as shown in Fig. 1. Depending on their physio-chemical properties, some anionic, water soluble PFASs (e.g., PFAAs) can be released with the landfill leachate (Yan et al., 2015; Benskin et al., 2012); on the other hand, neutral PFASs with low water solubilities and relatively high vapor pressures (e.g., fluorotelomer alcohols (FTOHs)) partition with landfill gas and are subsequently released to the atmosphere, if not captured efficiently by a gas collection system (Fig. 1). Most often, leachate from lined landfills are collected and sent to wastewater treatment plants (WWTPs) for treatment before their final disposal in surface water bodies. However, WWTPs, already burdened with PFAS from wastewater, are not equipped to remove these classes of contaminants, instead are acting as secondary sources of PFASs in the aquatic environment (Allred et al., 2015; Eggen et al., 2010). Given that solid wastes have been, and will continue to be, landfilled, it is critical to investigate landfills as long-term point sources of PFASs in the environment.

As more and more studies are published regarding environmental occurrence, fate and degradation of PFASs, it is important to systematically review the published literature to critically evaluate the state of knowledge and identify research gaps. Recent reviews of PFASs have addressed environmental biodegradation (Liu and Avendano, 2013), fate and removal of PFASs in drinking water treatment plants (Rahman et al., 2014), and WWTPs (Merino et al., 2016; Arvaniti and Stasinakis, 2015). A comprehensive review on the fate and transformation of PFASs in landfills is needed. This study critically reviews existing publications i) to summarize the occurrence of various classes of per- and polyfluoroalkyl substances (PFASs) and their sources in landfills, ii) to identify temporal and geographical trends of PFASs in landfills; iii) to delineate factors affecting PFASs in landfill; and iv) to identify research gaps and key future research directions.

2. Methodology

Based on an online database search (Web of Science, ScienceDirect and Google Scholar) of peer-reviewed articles, 14 journal articles were identified that reported PFAS concentrations in landfill leachate. Two studies reporting PFAS concentrations in ambient landfill air, three investigating degradation and leaching of PFASs inside landfills and one reporting leaching of PFASs through sodium bentonite (landfill barrier material) were also uncovered. While the subsequent sections in this paper are heavily based on these 18 articles (published between 2004 and 2017), additional citations from peer-reviewed journals are also cited to contextualize and explain the observations of the selected articles. The reported concentrations of PFASs are compiled in Table S1 (PFAAs) and S2 (perfluoroalkane sulfonamide derivatives and polyfluoroalkyl compounds) of SI. Concentration ranges and, where possible, median and other statistical values were calculated for studies reporting concentrations from multiple samples (from one or more landfills). During data analysis, below-quantification-limit values were assumed to be zero.

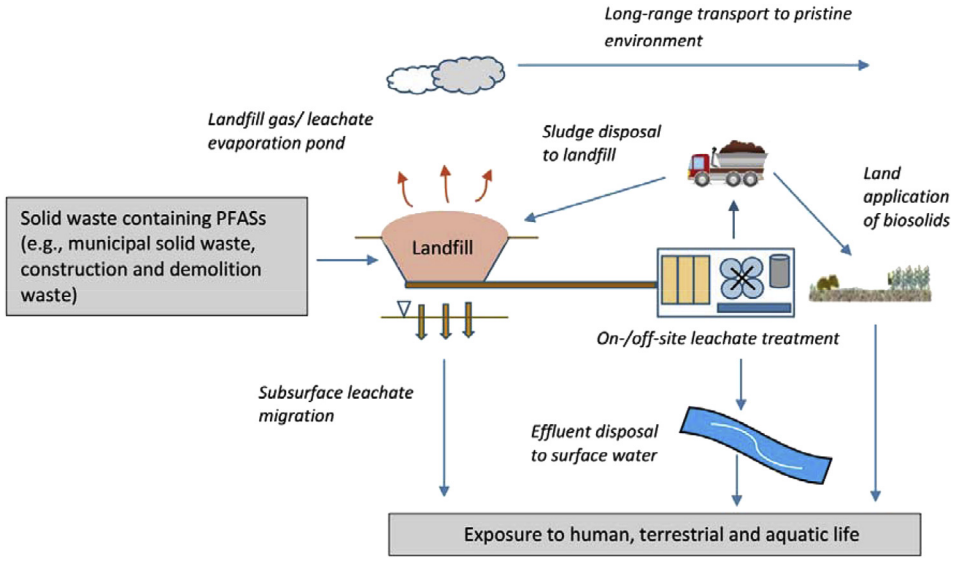


Fig. 1. Environmental pathways of per- and polyfluoroalkyl substances (PFASs) originating from solid wastes.

3. Per- and polyfluoroalkyl substances (PFASs) in landfills

3.1. Occurrence and sources of PFASs in leachate

3.1.1. Perfluoroalkyl acids (PFAAs)

Perfluoroalkyl carboxylic acids (PFCAs) and Perfluoroalkyl sulfonic acids (PFSAs), together known as PFAAs, are the most commonly studied PFASs in landfills. PFCAs with 4–14 carbon chain length and PFSA of mostly even chain length from C4-C10 have been reported in landfill leachate in the ng/L to μg/L range. Concentration ranges of PFAAs (C4-C10) in various countries are plotted in Fig. 2(a). Possible sources of PFAAs include consumer products (e.g., paper, textile, packaging, food contact paper, carpet) (Bečanová et al., 2016; Kotthoff et al., 2015; Ye et al., 2015), building materials (Bečanová et al., 2016), electronics (Bečanová et al., 2016) resulting from intentional addition of PFAAs during production and/or product use, and contamination with by-products impurities during production (Bečanová et al., 2016). Furthermore, PFAA precursors (e.g., FTOH, n:2 fluorotelomer carboxylic acids (n:2 FTCA) and n:2 unsaturated fluorotelomer carboxylic acids (n:2 FTUCAs)) present in the consumer products (Kotthoff et al., 2015; Ye et al., 2015) can degrade to PFAAs during product use and/or after disposal in the landfill (Allred et al., 2015; Lang et al., 2016).

3.1.2. Fluorotelomer-based substances

Fluorotelomer based compounds such as, n:2 FTCAs, n:2 FTUCAs, n:3 fluorotelomer carboxylic acids (n:3 FTCAs), fluorotelomer sulfonates (n:2 FTSAs) have been detected in landfill leachate (Lang et al., 2017; Allred et al., 2014; Benskin et al., 2012; Huset et al., 2011) and lab-scale landfill reactors (Lang et al., 2016; Allred et al., 2015) ranging from a few ng/L to μg/L (Fig. 2(b)). The FTCAs and FTUCAs are known degradation products of FTOHs (Buck et al., 2011), a major raw material of fluorotelomer polymers (FTPs), commonly used in textiles, upholstery, paper and carpets as surface protection agent (Rao and Baker, 1994). In addition to FTOH monomer released through biological (Washington et al., 2015; Rankin et al., 2014) and abiotic hydrolysis (Washington and Jenkins, 2015), residual FTOH present in FTPs (Dinglasan-Panlilio and Mabury, 2006) can biodegrade to FTCAs and FTUCAs, and subsequently to PFCAs in soil and activated sludge from WWTPs under both aerobic and anaerobic conditions (Liu and Avendano, 2013). The n:2 FTSAs can be released from consumer products applied with FTSA-containing surface protectors (Lang et al., 2016; Allred et al., 2015; Yang et al., 2014a), and by the degradation of complex fluorotelomer-based substances used in food packaging applications (Buck et al., 2011).

3.1.3. Perfluoroalkyl sulfonamide derivatives

Several unsubstituted, methyl- and ethyl-perfluoroalkane sulfonamide acetic acids (FASAAs) with C4-C8 carbon chain length have been reported in landfill leachates (Lang et al., 2017; Allred et al., 2014; Benskin et al., 2012; Huset et al., 2011) as shown in Fig. 2(b). Biodegradation of ethyl-perfluorooctane sulfonamidoethanol (EtFOSE), a major raw material of paper and packaging products (Buck et al., 2011), is said to form C8-based ethyl-perfluorooctane sulfonamide acetic acid (EtFOSAA) (Rhoads et al., 2008). Similar biodegradation pathways could be responsible for the shorter FASAA, MeFASAA and EtFASAA homologues resulting from methyl- and ethyl-perfluoroalkyl sulfonamidoethanols (FOSE) (Allred et al., 2014).

3.1.4. Polyfluoroalkyl phosphate esters (PAPs)

Detection of a few classes of polyfluoroalkyl phosphate esters (PAPs) (e.g., Di-substituted fluorotelomer phosphate esters (6:2–10:2 DiPAPs) and EtFOSE-based polyfluoroalkyl phosphate diester (DiSAmPAP)) have been reported in leachate (Allred et al., 2014; 2015, Lang et al., 2017) (see Table S2, in SI). PAPs are used in papers and synthetic fibers to impart oil and water repellency, in semiconductor materials and in personal care products (Liu and Liu, 2016). Microbial degradation of PAPs resulting in a mixture of FTCAs and PFCAs has been reported in activated sludge (Lee et al., 2010), and in aerobic soil (Liu and Liu, 2016; Lee et al., 2013), accounting for the infrequent detection of PAPs in leachate, despite their widespread use and high production volume (De Silva et al., 2012).

3.2. General observations and geographical trends of PFASs in leachate

Despite the high variabilities in PFAAs profiles and concentrations in landfill leachate reported in North America, Europe, China and Australia (Yan et al., 2015; Clarke et al., 2015; Allred et al., 2014; Bossi et al., 2008; Kallenborn et al., 2004), a few general trends emerge. For example, PFCAs are generally found to be the dominant

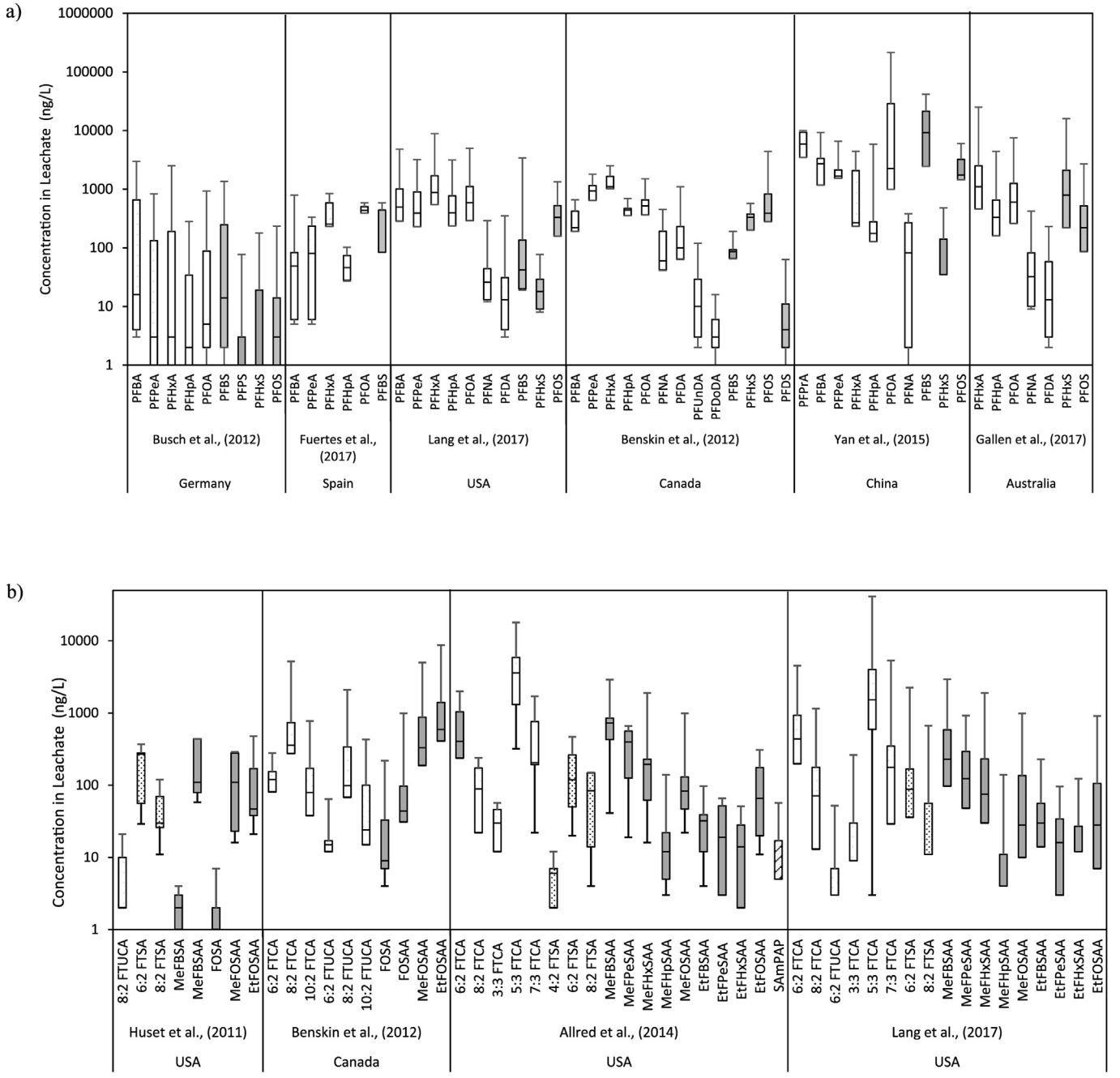


Fig. 2. Concentration ranges of (a) perfluoroalkyl carboxylic acids (PFCAs; plotted with white box) and perfluoroalkyl sulfonic acids (PFSAs; plotted with grey box); For countries with multiple studies, the study with highest number of landfill leachate samples are plotted in 2(a). Concentration ranges of (b) fluorotelomer carboxylic acids (plotted with white box), fluorotelomer sulfonic acids (black dotted box) and perfluoroalkane sulfonamide derivative substances (plotted with grey box) and polyfluorinated phosphate ester (plotted with slanted line) in landfill leachate in different countries. Analytes with ≥50% detection frequency has been plotted in 2(b). The box plots represent the 1st quartile, median and 3rd quartile values. The negative and positive error bars represent the difference between 1st quartile and minimum value, and maximum and 3rd quartile value, respectively. Note the logarithmic scale of Y-axis.
(Acronyms for 2(b): FTCA: fluorotelomer carboxylic acid, FTUCA: fluorotelomer unsaturated carboxylic acid, FTSA: fluorotelomer sulfonates, MeFASAA: methyl-perfluoroalkane sulfonamide acetic acid (C4-C8), EtFASAA: Ethyl-perfluoroalkane sulfonamide acetic acid (C4, C5, C6, C8), SAmPAP: 2 (*N*-ethylperfluorooctane-sulfonamido) ethyl phosphate).

PFASs (Fuertes et al., 2017; Allred et al., 2014; Li et al., 2012; Huset et al., 2011). Also, C4-C7 chain length PFAAs are more abundant than their longer-chain (≥C8) homologues (Fuertes et al., 2017; Li et al., 2012; Busch et al., 2010; Bossi et al., 2008; Kallenborn et al., 2004). Short-chain PFAAs are prone to preferential release and leaching from municipal solid waste (MSW), consistent with their higher aqueous solubilities and lower organic carbon-water partition coefficients relative to longer-chain PFAAs (Yan et al., 2015). In addition, the dominance of C4-C7 PFAAs could be related to the shift towards production of shorter-chain perfluorinated compounds since the early 2000's (Fig. 3). For example, 3M has commercialized surface treatment products containing C4-based side-chain fluorinated polymers since 2003 (Wang et al., 2013); 6:2 fluorotelomer-based side-chain fluorinated polymers have been registered in the Inventory of Effective Food Contact Substances Notifications of the United States Food and Drug

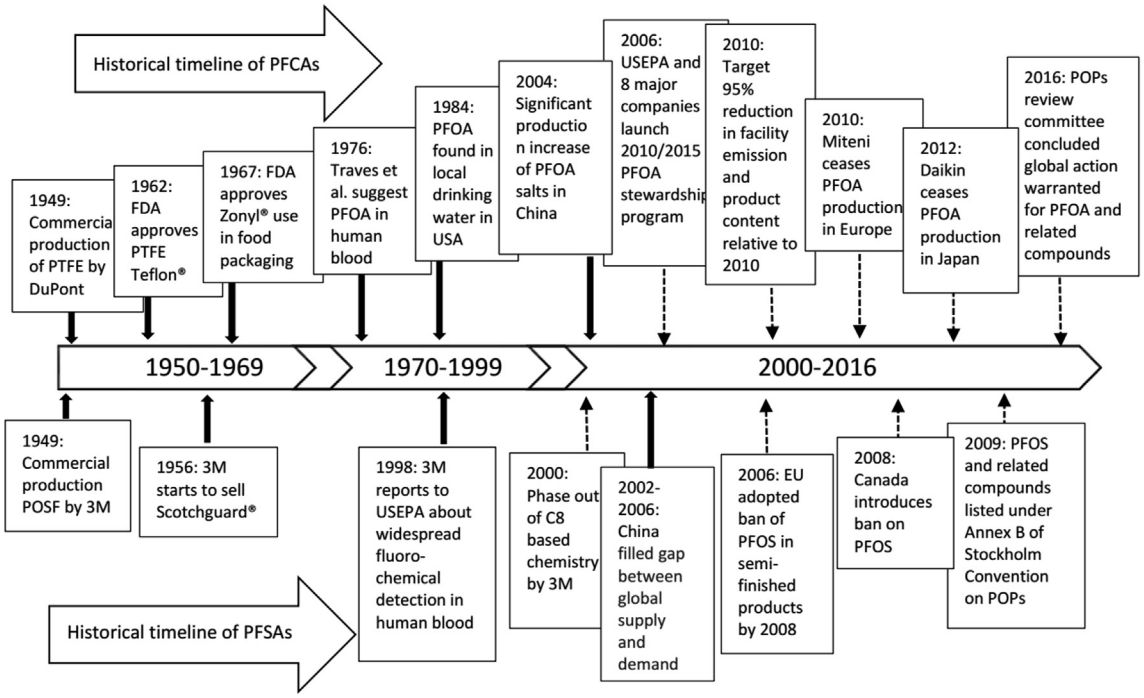


Fig. 3. Timeline of the production, commercialization and legislation of perfluoroalkyl carboxylic acids (PFCAs; at the top) and perfluoroalkyl sulfonic acids (PFSAs; at the bottom). Events and actions that may have resulted in increased concentrations in the environment and important findings are indicated by solid line arrow. Dotted arrows represent phase-outs and regulatory initiatives that may result in decreased concentrations in the environment. POSF (perfluorooctane sulfonyl fluoride) is the raw material of PFOS (perfluoroocatane sulfonic acid). Please note that the uneven time scale. Adapted from (Land et al., 2015; Lindstrom et al., 2011).

Administration since 2008 (Wang et al., 2013). Most of the landfill studies discussed here, involving sampling after 2010, likely reflect a product shift resulting from disposal of PFAS-containing consumer products with short residence life (e.g., food contact paper, packaging, other paper). For consumer products with longer residence times (e.g., carpet, upholstery, textiles), a time lag is expected before fluorinated alternatives used in these products reach measurable levels in leachate. The high frequency of PFOA detection in consumer products (Vestergren et al., 2015; Liu et al., 2014), together with its historical use in surface treatment agents of leather, textiles, paper and electronics (Wang et al., 2014a) explain the observed high detection frequencies and concentrations (often comparable to those of C4-C6 PFCAs) of PFOA in leachate. Studies have also reported that ∑PFAAs from landfill leachate (for a facility closed 2-4 decades ago), ranges from hundreds to a few thousands of ng/L (Gallen et al., 2016; Allred et al., 2014; Huset et al., 2011).

An increasing number of studies showing degradation of polyfluorinated compounds to PFAAs in the environment (Liu and Liu, 2016), along with increasing availability of chemical standards and improved analytical techniques, have led to recent studies (summarized in the next section) to investigate PFAA-precursors and their degradation products, as well as other classes of perfluorinated compounds (e.g., perfluoroalkyl sulfonamide derivatives) in landfill leachate. Some of the fluorotelomer-based (e.g., n:2 FTCA, FTUCAs) and *N*-alkyl FASAAs are frequently detected (Table S2) with concentration ranges (shown in Fig. 2(b)) that are comparable to and/or higher than those of PFCAs (>μg/L).

3.2.1. Concentration and trends in the USA and Canada

PFASs in landfill leachate in the USA have been studied by Huset et al. (2011), Allred et al. (2014) and Lang et al. (2017) (Table S1, in SI). PFCA contributed 20–90% of ∑PFASs (molar concentration basis), with concentrations from 10 to 8900 ng/L (Allred et al., 2014; Huset et al., 2011). While the median concentration reported by Allred et al. (2014) exceeded 1000 ng/L for all C4-C8 PFCAs, Huset et al. (2011) observed smaller concentrations (100–600 ng/L) for the same compounds. The difference in concentrations could result from variation in waste composition, climatic condition, age of the landfill and/or leachate management system (i.e., leachate recirculation vs flow-through), as indicated in Table 1 (see also section 3.5). PFSA concentrations in leachate have varied from 50 to 3200 ng/L in the USA, with median concentrations of a few hundred ng/L for perfluorobutane sulfonic acid (PFBS), perfluorohexane sulfonic acid (PFHxS) and PFOS (Allred et al., 2014; Huset et al., 2011), as shown in Fig. 2(a). While PFOS was detected in all leachate samples, its concentration was generally lower than that of PFBS and PFHxS (Allred et al., 2014; Huset et al., 2011). This dominance of shorter-chain PFSAs over historically used PFOS could be indicative of the transition towards C4-based chemistry after 2002, as shown in Fig. 3 (Vestergren et al., 2015; Huset et al., 2011; Lindstrom et al., 2011). Leachates from waste cells closed in 1993 or earlier also shows dominance of PFBS and PFHxS, indicating the role of C4-based chemistry even prior to 2002 (Wang et al., 2014b; Huset et al., 2011), in addition to the higher mobility of shorter chain PFSAs, leading to their release in aqueous phase (Higgins and Luthy, 2006).

PFCA concentrations in leachates collected from Canadian landfills have been in the range of tens to few thousands of ng/L (Li et al., 2012; Benskin et al., 2012). PFSAs, namely PFBS, PFHxS and PFOS, also varied within the same range, with median PFHxS concentration of 200 ng/L higher than for PFOS and PFBS in a cross-Canada study (Li et al., 2012). Landfill gas condensate was reported to contain C4-C8 PFAAs, with PFBS being the dominant compound at a concentration of 1000 ng/L (Li, 2011).

3.2.2. Concentration and trends in European Countries

PFAAs have been reported in several European countries, including Spain (Fuertes et al., 2017), Finland (Perkola and Sainio,

Table 1
Landfill locations and characteristics for leachate sampling sites.[a]

Reference	Woldegiorgis et al. (2006)	Busch et al. (2010)	Huset et al. (2011)	Benskin et al. (2012)	Perkola and Sainio (2013)	Allred et al. (2014)	Yan et al. (2015)	Gallen et al. (2016)	Gallen et al. (2017)	Lang et al. (2017)	Fuertes et al. (2017)
Location (Country)	Strandmossen, Djupdalen (Sweden)	(Germany)	Gulf Coast, Pacific Northwest, west coast, Mid-Atlantic states, Southeast (USA)	Pacific Northwest (Canada)	Espoo (Finland)	(USA)	Changzhou, Guangzhou, Nanjing, Shanghai, Suzhou (China)	(Australia)	(Australia)	Arid, temperate and wet climatic locations in USA	(Spain)
Landfill Sites	3	8 closed between 1979 and 2005, 14 active	4 active (since 1996), 2 closed (operated during 1982–1993)	2 active since 1960's	1 closed (1987–2007)	5 active (since 1990's), 1 closed (1975–1990)	4 active, 3 closed	6 active, 8 closed (operated between 1970's and 2010)	24 active, 3 closed (between 1993 and 2003)	18 active	2 active, 2 closed between 2015 and 2015
PFAS Analytes	13	43	24	24	4	70	14	14	9	70	16
Leachate System	not available	flow-through	recirculation, except one flow through system	flow-through and recirculated	not available	not available	flow through	flow through except one recirculated	recirculation and flow through system	recirculation	flow through
Sampling Year	November, 2005	not available	2006	February–June, 2010	October, 2009 and June, 2010	not available	Spring, 2013	February–April, 2014	August–December 2014	February 2013 to December 2014	March, 2015
Waste Type	not available	municipal and commercial	primarily municipal	primarily municipal	primarily municipal	municipal and commercial	municipal	primarily municipal and commercial	municipal, commercial, construction and demolition	primarily municipal	primarily municipal
Sampling Method	grab samples	not available	grab samples	grab sample	24-hr composite	grab samples	grab sample	grab sample	grab and composite samples	grab samples	grab sample
Leachate Treatment System	aerobic pond	biological and physical	not available	off-site at WWTP	not available	not available	off-site with two-stage process (MBR/RO or NF)	off-site at WWTP,	off-site WWTP, evaporation pond	Off-site WWTP	off-site with two-stage process (MBR/UF)
Estimated PFAS loading in leachate	not available	not available	not available	8-25 kg/y/landfill	not available	not available	3100 - 4000 kg/y (nationwide)	not available	not available	563 - 638 kg in 2013 (nationwide)	1 kg/y/landfill

[a] WWTP: wastewater treatment plant; MBR: membrane bioreactor; RO: reverse osmosis; UF: ultrafiltration; NF: nanofiltration; PFAS: per- and polyfluoroalkyl substance.

2013), Norway (Kallenborn et al., 2004), Sweden (Woldegiorgis et al., 2006), Denmark (Bossi et al., 2008), and Germany (Busch et al., 2010). Most of these studies (except for Fuertes et al., 2017; Busch et al., 2010) included less than 10 PFAAs, with concentrations reported to be in the range of <1–1800 ng/L. The median concentrations of all PFAAs were <550 ng/L, which is smaller than for the USA and Canada. Higher abundances of PFOS and PFOA compared to shorter chain PFAAs were also observed. However, more recent studies conducted in Spain (Fuertes et al., 2017) and Germany (Busch et al., 2010), reported higher abundances of shorter chain PFAAs (≤C7).

3.2.3. Concentration and trends in Australia

Gallen et al. (2016; 2017) reported the occurrence of PFAAs in leachates from a number of active and closed landfill sites in Australia, as shown in Table 1. The PFCA and PFSA concentrations of up to 5700 ng/L and 1900 ng/L, respectively, have been reported. While these ranges are slightly smaller than for the USA, the median concentrations were <550 ng/L for all PFAAs, except PFHxA (970 ng/L), similar to those reported in European Countries (Gallen et al., 2016).

3.2.4. Concentrations and trends in China

PFAAs have been reported in leachates from four active and three closed MSW landfills in China (Yan et al., 2015) with concentrations ranging from 70 to 214,000 ng/L for PFCAs (C4-C8) and 30 to 416,000 ng/L for PFSAs (Fig. 2(a)). While these ranges are orders of magnitude higher than for other countries, the median concentrations for most PFAAs (e.g., perfluorobutanoic acid (PFBA), PFOS, PFOA, PFBS, perfluoropentanoic acid (PFPeA)) in the range of several thousands ng/L were also higher than those reported in other countries. Observed high concentrations of PFAAs could be related to PFAS contents of the consumer products, and/or higher fractions of PFAS-containing wastes (e.g., construction and demolition waste, electronics, carpet, clothing) being disposed in these landfills. However, this could not be substantiated, as no survey of PFASs in consumer products in China is available to the best of our knowledge. Also, waste compositions of the Chinese landfills were not specified by Yan et al. (2015).

In contrast to other studies, PFOA (mean contribution 29%) was found to be the most abundant PFAA, followed by PFBS (26%) and pentafluoropropionic acid (PFPrA) (16%). Despite the high abundance of PFBS in leachate, suggesting a shift towards C4-based chemistry, high PFOS concentrations (1000–6000 ng/L) were reported in Chinese landfill leachates (Yan et al., 2015). It is noteworthy that, following the phase-out of PFOS by its largest manufacturer (3M) in the USA, production in China grew rapidly (see Fig. 3) from 50 tonnes/year in 2004 to current levels of 100–200 tonnes/y (Yang et al., 2014b; Xie et al., 2013). Since the addition of PFOS to Annex B of the Stockholm Convention in 2009, China has restricted its production and use, except for specific exemptions (Wang et al., 2016). Nonetheless, past manufacturing history could be a key factor underlying the observed high relative concentrations of PFOS. Other factors could be due to (1) long use lifetimes of major PFOS-containing products (e.g., carpets, textiles); (2) long residence of PFOS-containing MSW in landfill, and/or (3) on-going uses of PFOS-containing products in China (Yan et al., 2015).

3.3. Estimating PFASs mass discharged with landfill leachate

The mass of PFASs discharged with landfill leachate is a function of PFAS concentrations in leachate and leachate volume. As discussed in section 3.2, the total PFAS concentration in leachate is determined by the number/classes of PFAS analyzed and can be highly variable, depending on landfill- and waste-related factors (discussed in section 3.5 below). Similarly, leachate volume is highly dependent on climate (in particular, rainfall and subsequent infiltration into landfill), and may vary substantially from month to month, and from year to year (Gallen et al., 2017). A recent survey (Lang et al., 2017) of PFASs in landfill leachates in the USA (70 PFASs in 95 samples) estimated national release of 19 PFASs, with >50% of samples having quantifiable concentrations. A mass estimate was obtained by coupling waste age and climate-specific PFAS concentration estimates with climate-specific estimates of annual leachate volumes using Monte Carlo analysis. The estimates for the 19 PFASs ranged from 563 to 638 kg for the sampling year, 2013. PFCAs accounted for the majority of mass estimated (291 kg/yr), closely followed by FTCAs (285 kg/yr), with lower releases of PFSAs and their precursors (84 kg/yr) (Lang et al., 2017). This indicates that the estimated mass budget of PFASs, often based on measured PFAAs concentration only (Fuertes et al., 2017; Yan et al., 2015; Benskin et al., 2012; Busch et al., 2010; Oliaei et al., 2006), (<1–25 kg/yr per landfill as shown in Table 1) could grossly underestimate the total release of PFASs from landfills.

3.4. Occurrence of PFASs in landfill ambient air

While many of the PFAA-precursors and their degradation products (e.g., FTOHs, fluorotelomer iodides (FTIs), FOSEs) are semi-volatile in nature, the role of landfills as sources of PFAS gaseous emissions to the atmosphere has received little attention. Two studies (Ahrens et al., 2011; Weinberg et al., 2011) reported higher (2–30 times) PFAS concentrations in landfill ambient air compared to control sites that were presumably not contaminated with landfill emissions. Ambient landfill air predominantly contained FTOHs, with concentrations being >90% of total precursor compounds measured (see Table 2). While FOSAs and FOSEs were also detected, their concentrations were orders of magnitude lower than for the FTOHs (Ahrens et al., 2011; Weinberg et al., 2011). 8:2 FTOH was found to be the highest sole contributor (50–65% of the ∑FTOHs, FOSAs, FOSEs), followed by 6:2 FTOH (15–40%) FTOHs (Ahrens et al., 2011; Weinberg et al., 2011). Higher abundance of 8:2 FTOH compared with 6:2 FTOH has been reported (Jahnke et al., 2007; Shoeib et al., 2006) to be typical of urban air. This is also supported by a recent survey (Vestergren et al., 2015) of PFASs in consumer products in Norway imported from China, which showed that 6:2 and 8:2 FTOHs were the most abundant extractable PFASs measured.

PFAAs were also detected in the particulate phase (Weinberg et al., 2011) and gas phase (Ahrens et al., 2011) of ambient landfill air. PFBA, PFHxA, PFOA were detected most frequently and at higher concentrations compared to other PFAAs in the gas phase (Ahrens et al., 2011). This might indicate abundance of PFOA and shorter-chain PFCAs in landfill waste, or reduced availability of longer-chain PFCAs in air due to their higher affinity for solid particles (Arvaniti et al., 2012). This dominance of short and even chain length PFCAs is also consistent with the PFCA distribution in landfill leachates from 22 sites in Germany (Busch et al., 2010), indicating that this pattern is typical for landfill emissions. Although PFOS is frequently detected in landfill leachate, it exhibited very low air concentrations at the landfill sites (<5 pg/m^3), likely due to strong sorption of PFOS to landfill solids, efficient trapping of PFOS in landfill gas collection, and partitioning of PFOS to landfill leachate (Ahrens et al., 2011).

3.5. Factors affecting per- and polyfluoroalkyl substances (PFASs) in landfill

Following landfilling, PFASs undergo long-term leaching, as well

Table 2
Concentration ranges of various classes of per- and polyfluoroalkyl substances (PFASs) in ambient landfill air (pg/m^3).

∑PFAAs[a]	∑FTOHs[b]	∑FTAs[c]	∑FOSAs, FOSEs[d]	∑PFAS $_{WWTP}$/∑PFAS$_{reference\ site}$	Landfill status	Reference
130–320	2500–26,000	Not measured	60–120	5 to 30	active	(Ahrens et al., 2011)
5–10[e]	70–100	1–10	6–20	1.5 to 2.5	Closed for previous 6 years	(Weinberg et al., 2011)
<DL[f]-40[e]	120–660	<DL-20	7–20	1.5 to 3	active	(Weinberg et al., 2011)

[a] Perfluoroalkyl carboxylic and perfluoroalkyl sulfonic acids.
[b] Fluorotelomer alcohols.
[c] Fluorotelomer acids.
[d] Perfluoroalkane and *N*-alkyl perfluoroalkane sulfonamide and sulfonamidoethanols.
[e] Measured in particulate phase.
[f] Detection limit.

as degradation of precursor compounds, processes that are affected by the physio-chemical properties of the PFASs, as well as the landfill leachate (Yan et al., 2015). As the landfilled waste passes through successive stages of aerobic, acetogenic, and methanogenic stabilization stages, significant changes occur in the physio-chemical properties, such as pH and organic and inorganic constituents (Table S3 in SI) of the leachate (Renou et al., 2008), likely affecting the mobility and degradation of PFASs. In most cases leachate from various waste cells undergoing varying states of decomposition are collected together and subsequently sampled for PFAS analysis. This, along with the uncertainty surrounding waste input in various cells, makes it challenging to conduct any mass balance to better understand the PFAS release and transformation inside landfills. In addition, climatic factors (e.g., precipitation) affecting the moisture content inside landfills, and operating conditions (e.g., compaction of the waste, waste filling procedure, leachate recirculation) could also play an important role in determining the fate and transformation of PFASs in landfills.

3.5.1. Effect of leachate physiochemical properties

Several studies have reported increased mobility of PFAAs with increasing pH (Gallen et al., 2017; Yan et al., 2015; Benskin et al., 2012), possibly due to the altered electrostatic behavior of the sorbents (Higgins and Luthy, 2006). This observation is consistent with sorption studies of PFOS and PFOA to diverse adsorbents, which indicated decreased sorption with increasing solution pH due to protonation of the adsorbent surface, leading to fewer positive sites on the sorbent (Wang and Shih, 2011; Yu et al., 2009). Total organic carbon (TOC) has been weakly correlated with the PFAA concentration in leachate (Gallen et al., 2017; Benskin et al., 2012), possibly due to hydrophobic partitioning of the perfluorinated chain with organic matter. Electrical conductivity (a measure of ion concentration in solution) has been both positively (Benskin et al., 2012) and negatively (Yan et al., 2015) correlated with PFAA concentration in leachate. While earlier studies (You et al., 2010; Higgins and Luthy, 2006) indicated decreased mobility of PFAAs with increasing ionic strength, more recent work suggests that the effect of ionic strength of PFAA adsorption is quite complex and often ion-type and concentration specific. For example, multivalent cations can increase sorption by acting as bridges between anionic PFASs and negatively charged surfaces (Kim et al., 2015), whereas anions (e.g., Cl^-, SO_4^{2-} or $Cr_2O_7^{2-}$) have been reported to compete with anionic PFASs for adsorption sites, (in boehmite, chitosan and resins) leading to increased solubility of anionic PFASs (Du et al., 2014). This suggests that the observed seasonal variation of macro-constituents (e.g., Cl^-, Ca^{2+}, Mg^{2+}, SO_4^{2-}) in leachate (Kulikowska and Klimiuk, 2008) likely contributes to the observed variability and patterns of PFASs concentration in leachates. Sorption behavior is also affected by carbon chain length and the functional head group of the PFASs (Higgins and Luthy, 2006).

3.5.2. Biological processes inside landfills

Biodegradation is arguably one of the most important factors determining the fate of PFASs in landfills. Allred et al. (2015) and Lang et al. (2016) studied the evolution of PFASs into leachate using anaerobic landfill reactors (fed with MSW, carpets and clothing). While PFASs were released through a combination of biological (e.g., biodegradation) and abiotic (e.g., desorption) processes, the leachates from live bioreactors (producing methane) had on average 5 to 10 times higher ∑PFAS than the average for biologically inactive reactors (Lang et al., 2016; Allred et al., 2015). Following the onset of methanogenic conditions, concentrations of known biodegradation intermediates of PFAA precursors (i.e., methyl-perfluorobutane sulfonamide acetic acid (MeFBSAA), n:2 and n:3 FTCAs) increased steadily, with 5:3 FTCA becoming the single most concentrated PFAS (Allred et al., 2015). While the aforementioned studies have provided some valuable insights into the release of PFASs in landfills, further research including measurement of semi-volatile PFASs is needed to fully comprehend the transformation process.

Landfill operating conditions, such as leachate recirculation, could be important factors affecting biodegradation. For example, Benskin et al. (2012) reported order of magnitude lower ΣPFAS concentrations, (consisting entirely of PFAAs) in recirculated leachate, compared with flow-through leachate containing both PFAAs and their precursors (31–71% PFAAs and 29–69% PFAA-precursors). While it is possible that the PFAS profile observed in the single recirculated leachate sample was not representative of leachate from this site over the long term, another explanation could be that recirculating leachate back to the landfill facilitated more biodegradation of PFAA-precursors (Benskin et al., 2012). While Huset et al. (2011) observed slightly higher ∑PFAA-precursors in flow-through leachate, the few other studies which measured PFAA-precursors (Lang et al., 2017; Allred et al., 2014) did not identify the recirculated leachate sample. Therefore the observations of Benskin et al. (2012) need to be substantiated by other studies.

3.5.3. Effect of climate

Precipitation can dilute leachate, resulting in decreased PFAS concentration within a time frame of a single day (Gallen et al., 2017; Benskin et al., 2012). On the other hand, changes in the moisture content of wastes resulting from rainfall can affect hydrolysis reactions and bioactivity (through bacterial growth, mixing of leachate resulting in PFAS concentration change, etc.) inside landfill over longer periods (e.g., 2 weeks) (Benskin et al., 2012). However, Benskin et al. (2012) did not observe any significant correlations between any PFAS congener and cumulative 2-week precipitation for temporal leachate samples collected from a landfill. A more recent study (Lang et al., 2017) of 18 landfills located in different climatic conditions (wet, temperate and arid) observed that several PFAS compounds (perfluorononaic acid (PFNA), perfluorodecanoic acid (PFDA), PFBS, PFOS, 6:2 FTSA, and

methyl-perfluorooctane sulfonamide acetic acid (MeFOSAA)) had significantly higher concentrations in leachates from wet climates, suggesting that leaching governed release of these compounds.

3.5.4. Effect of waste age and waste composition

Statistically significant decreases in the concentrations of several PFASs with increasing landfill age have been observed in studies of multiple landfills (both active and closed) (Gallen et al., 2017; Lang et al., 2017; Busch et al., 2010). This trend could result from industrial significant shift of PFASs used in consumer products (Gallen et al., 2017; Lang et al., 2017). For example, Lang et al. (2017) observed higher PFBS and PFNA concentration in leachate from young waste, possibly due to increasing manufacture of PFBS- and PFNA-based products as alternatives to PFOS- and PFOA-based products. Decreases in the concentrations of PFASs in wastes (e.g., due to biodegradation) could account for the observed decreasing trend of PFAS degradation intermediates (e.g., 8:2 FTCA, 5:3 FTCA, MeFBSAA and MeFOSA) with landfill age (Lang et al., 2017). PFASs with similar concentrations in old and young wastes, will likely continue to be released for many years (Lang et al., 2017). Collected leachate samples often contained mixtures of leachate which have undergone various stages of degradation from various waste cells. As a result, it is impossible to comment on the effect of any specific stabilization stage on the PFAS concentration. The above-mentioned studies considered the total period of operation (final year minus opening year) to calculate the average age of the waste.

Lab-scale anaerobic reactors treating various types of refuse, such as MSW (Allred et al., 2015), carpets and clothing (Lang et al., 2016), have demonstrated the abundance of various classes of PFAS, indicating the importance of waste type in determining the PFAS concentration. For example, Lang et al. (2017) observed that short carbon-chain ($\leq$C6) PFAS were dominant in reactors treating waste carpets, whereas reactors treating clothing accumulated short-chain PFCAs, PFOA, and 8:2 FTSA. High variability in total PFAS release was observed, even for reactors receiving similar types of waste (i.e., clothing) (Lang et al., 2016). Gallen et al. (2017) also observed that leachate from operating landfills accepting >50% construction and demolition waste generally had higher PFAA concentrations compared to landfills accepting >50% MSW. Understanding PFAS trends due to variation in waste type could be challenging considering the variation in PFAS contents in similar categories of waste (Lang et al., 2016), the heterogenous nature of the solid waste and uncertainties surrounding the input of solid wastes to landfills.

4. Fate of PFASs in leachate treatment systems

One of the most common waste management practices is to send leachate to an off-site domestic WWTP. The fate and occurrence of PFASs in WWTPs have recently been thoroughly reviewed by Arvaniti and Stasinakis (2015), and are outside the scope of this paper. Other leachate management options include on-site pre-treatment, followed by off-site discharge at a WWTP, and complete treatment and discharge on-site (Townsend et al., 2015). Leachate treatment options can be broadly categorized as either physio-chemical treatment (e.g., coagulation-flocculation, chemical precipitation, membrane filtration, activated carbon adsorption, chemical oxidation) or biological treatment (e.g., activated sludge system, aerated lagoon) (Foo and Hameed, 2009; Renou et al., 2008). Similar technologies are also used for groundwater and drinking water treatment, and their effectiveness in removing PFASs has been reviewed elsewhere (Merino et al., 2016; Rahman et al., 2014). Therefore, the focus of this section is to discuss removal/fate of PFASs specifically resulting from landfill leachate treatment.

Several studies (Fuertes et al., 2017; Yan et al., 2015; Busch et al., 2010) reported an overall increase in PFAA concentrations following on-site biological leachate treatment, which is consistent with their persistent nature and possible formation from PFAA-precursors. The extent of formation observed was analyte- and site-specific, ranging between 10 and 250% for individual PFAAs (Yan et al., 2015). A wet air oxidation process contacting with ozone to create OH-radicals to degrade contaminants, also showed slightly higher (~5%) $\sum$PFAAs concentrations in the effluent leachate, but the increase was less than for biological treatment (Busch et al., 2010). An adsorption technique using activated carbon was reported to be somewhat effective (removal efficiency ranges between 70 and 99%) in removing PFAAs from leachate (Busch et al., 2010). High-pressure membrane filtration techniques such as reverse osmosis (RO) and nanofiltration (NF) removed >95% PFAAs directly from leachate (Busch et al., 2010) and from biologically-treated leachate (e.g., membrane bioreactor, followed by RO or NF) (Yan et al., 2015). On the other hand, Fuertes et al. (2017) found that ultrafiltration (UF) integrated with membrane bioreactors showed little or no removal of PFAAs. Despite the success of high-pressure filtration systems, disposal of PFAS-rich concentrate remains a challenging issue in need of careful consideration (Rahman et al., 2014).

5. Future research directions

While the presence of various classes of PFASs in landfill leachate is well documented in North America, several Northern European countries, Australia and China, no data exist for South and Southeast Asia, Southern Europe and Africa. Considering the lack of regulations limiting the manufacture and use of C8-based PFASs and the lack of pollution abatement measures such as leachate collection systems and lining materials (Ismail and Manaf, 2013), landfills in these countries could be a significant source of PFASs in the environment. This could undermine regulatory initiatives in some parts of the world, due to the long-range transport and persistence of some PFASs (such as PFAAs).

Widespread application of FTPs to consumer products (e.g., paper, textiles, leather) (Rao and Baker, 1994) imply that most FTP products will ultimately be landfilled (Washington and Jenkins, 2015). Despite the decade-long debate on their stability, recent studies show that FTPs can undergo abiotic (Washington and Jenkins, 2015) and biological hydrolysis releasing FTOHs, which then biodegrade to PFCAs with an estimated half-life range of 8–100 years (Washington et al., 2015; Rankin et al., 2014). Therefore, degradation of FTPs under landfill conditions (e.g., anaerobic, pH > 7) needs to be studied, including the measurement of semi-volatile compounds (e.g., FTOHs). As indicated by Washington et al. (2014), concentrations of volatile degradation products of FTP measured by GC-MS could be much more than an order of magnitude higher compared to non-volatile PFASs measured by LC-MS/MS.

To avoid significant underestimation of the total PFAS released with landfill leachate, PFAA-precursors and their degradations products (e.g., n:3 FTCA, FSAA) should be included in monitoring studies. Phillips et al. (2007) reported a 100-times smaller toxicity threshold of FTCAs compared to PFCAs for freshwater microorganisms. PFAA-precursor concentrations in leachate would provide valuable information from a water quality perspective as well. Owing to the semi-volatile nature of some precursor compounds and their degradation products, studies need to include landfill gas to understand degradation pathways and the overall fate of PFASs. This would also enable more realistic assessments of the release of PFASs to the environment with landfill gas.

More than 3000 PFASs are available on the market, and the identity of many are unknown (Wang et al., 2017). In addition to the maximum number (70 compounds) of PFASs studied in landfills, there must be more PFASs present in landfill leachates. Application of non-targeted methods (e.g., total oxidable precursor assay, total fluorine analysis) could provide valuable information regarding the unknown PFASs in leachates. New analytical techniques including PFAS classes never studied in leachate could provide useful information. For example, among the legacy PFASs, mixtures of C6–C12 perfluoroalkyl phosphonic and phosphinic acids (PFPAs and PFPiAs) used as wetting agents in consumer products (Wang et al., 2016); FTI, fluorotelomer acrylate and fluorotelomer methacrylate (FTMAC) used in impregnating agents (and their degradation products) (Favreau et al., 2017) should be included in landfill studies. Emerging PFAS like perfluoropolyethers (PFPEs), used as fluorinated alternatives of PFOS and PFOA in fluoropolymer manufacture (e.g., ADONA from 3M/Dyneon ($CF_3OCF_2CF_2CF_2OCHFCF_2COO\text{-}NH_4^+$), GenX from DuPont ($CF_3CF_2CF_2OCF(CF_3)COO\text{-}NH_4^+$)), surface treatment food contact material (Wang et al., 2013), are now being increasingly detected in various environmental media, as outlined in a recent review (Xiao, 2017). Research gaps regarding fate and transport of the emerging PFASs (e.g., PEPEs) in the environment (including landfills) need to be addressed.

While the fate and release of PFASs in landfill have been studied to some extent, very limited research exists on the performance of current containment practices (e.g., landfill liners). Leaching cell tests with sand/bentonite mixture barriers showed partial retention of PFASs (10 PFCAs, 4 PFSAs, 1 FOSA and 3 FTUCAs), decreasing over time, indicating limited effectiveness of sodium bentonite liners in landfills in containing PFASs (Li et al., 2015). Therefore, more research is needed to evaluate the effectiveness of current containment practices and how they can be improved to reduce PFAS emissions from landfills.

6. Conclusions

This study reviews publications which have reported the occurrence and sources of PFASs in landfills, to identify temporal and geographical trends of PFASs in landfill leachate and to identify and discuss factors affecting PFASs in landfills. The need for further research has been discussed. Research over the past decade has shown that PFAAs are routinely detected in landfill leachate, with short chain (C4-C7) PFCAs being more abundant than longer-chain ones, possibly reflecting greater mobility and increasing application of shorter-chain compounds. Despite its restricted use, PFOA remains one of the most frequently detected and abundant PFCAs in landfill leachate. This indicates that, if not managed properly, landfills could act as secondary sources of PFOAs in the environment.

Recent studies also document the presence of PFAA-precursors and their degradation products in landfill leachate, at concentrations comparable to, or higher than, the most frequently detected PFAAs (e.g., PFBA, PFOA, PFOS). Landfill ambient air also contains elevated concentrations of PFASs, primarily semi-volatile precursor compounds (such as FTOHs), compared to upwind control sites, suggesting that landfills likely act as emission sources of atmospheric PFASs. The fate and transformation of PFASs inside landfills are complex, affected by combinations of external (e.g., climate, waste input) and internal (e.g., biodegradation, sorption) factors. Release of most of the PFASs from waste to leachate occurs as a result of biodegradation, closely associated with onset of the methanogenic phase. The methane yielding stage also results in higher pH (>7) of leachate, correlated with greater mobility of PFAAs.

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Appendix A. Supplementary data

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Review

A critical review of perfluoroalkyl and polyfluoroalkyl substances (PFAS) landfill disposal in the United States

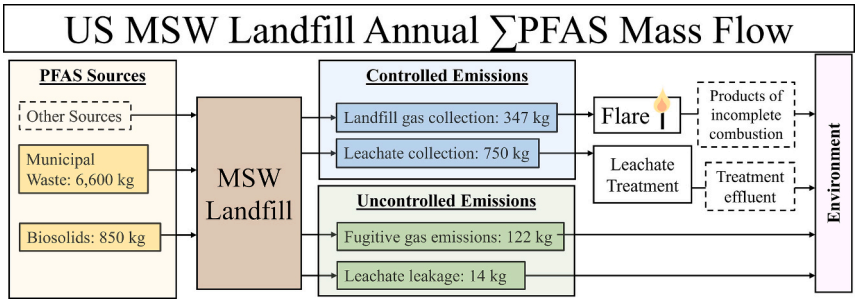
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HIGHLIGHTS

- Solid waste management strategies impact PFAS emissions.
- PFOA has the highest ratio to its respective RSL in C&D and MSW landfill leachates.
- Unlined C&D landfills present a significant source of PFAS to the environment.
- An estimated 7.5 metric tons of PFAS enter MSW landfills annually.
- Annually, 460 kg of PFAS emitted via landfill gas, 750 kg via landfill leachate.

GRAPHICAL ABSTRACT



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ABSTRACT

Landfills manage materials containing per- and polyfluoroalkyl substances (PFAS) from municipal solid waste (MSW) and other waste streams. This manuscript summarizes state and federal initiatives and critically reviews peer-reviewed literature to define best practices for managing these wastes and identify data gaps to guide future

Abbreviations: µg, Microgram; AFFF, Aqueous film-forming foams; C&D, Construction and demolition; diPAP, Polyfluoroalkyl phosphoric acid diesters; FASA, Perfluoroalkane sulfonamide; FASE, Perfluoroalkane sulfonamido ethanol; FTAc, Fluorotelomer acrylate; FTCA, Fluorotelomer carboxylic acid; FTO, Fluorotelomer olefin; FTOH, Fluorotelomer alcohol; H_2S, Hydrogen sulfide gas; HAL, Health Advisory Limit; kg, Kilogram; L, Liter; LFG, Landfill gas; MCL, Maximum Contaminent Level; MeFBSAA, Methyl-n-perfluorobutanesulfonamidoacetic acid; MeFOSAA, Methyl-n-perfluorooctanesulfonamidoacetic acid; mg, Milligram; Mg, Megagram (metric ton); MPCA, Minnesota Pollution Control Agency; MSW, Municipal solid waste; MSWI, Municipal solid waste incineration; NEBRA, North East Biosolids & Residuals Association; NWRA, National Waste & Recycling Association; PFAA, Perfluoroalkyl acids; PFAS, Per- and polyfluoroalkyl substance(s); PFBS, Perfluorobutane sulfonate; PFCA, Perfluoroalkyl carboxylic acids; PFHxA, Perfluorohexanoic acid; PFHxS, perfluorohexane sulfonate; PFOA, Perfluorooctanoic acid; PFOS, Perfluorooctane sulfonic acid; PFPeA, Perfluoropentanoic acid; PFSA, Perfluoroalkyl sulfonic acids; PIC, Product of incomplete combustion; RCRA, Resource Conservation and Recovery Act; RNG, Renewable natural gas; RO, Reverse osmosis; RSL, Regional Screening Limit; SI, Supplementary information; US EPA, United States Environmental Protection Agency; WWTP, Wastewater treatment plant.

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research. The objective is to inform stakeholders about waste-derived PFAS disposed of in landfills, PFAS emissions, and the potential for related environmental impacts. Furthermore, this document highlights data gaps and uncertainties concerning the fate of PFAS during landfill disposal. Most studies on this topic measured PFAS in liquid landfill effluent (leachate); comparatively fewer have attempted to estimate PFAS loading in landfills or other effluent streams such as landfill gas (LFG). In all media, the reported total PFAS heavily depends on waste types and the number of PFAS included in the analytical method. Early studies which only measured a small number of PFAS, predominantly perfluoroalkyl acids (PFAAs), likely report a significant underestimation of total PFAS. Major findings include relationships between PFAS effluent and landfill conditions – biodegradable waste increases PFAS transformation and leaching. Based on the results of multiple studies, it is estimated that 84% of PFAS loading to MSW landfills (7.2 T total) remains in the waste mass, while 5% leaves via LFG and 11% via leachate on an annual basis. The environmental impact of landfill-derived PFAS has been well-documented. Additional research is needed on PFAS in landfilled construction and demolition debris, hazardous, and industrial waste in the US.

1. Introduction

Per- and polyfluoroalkyl substances (PFAS) include thousands of unique manufactured chemical compounds with a hydrophobic carbon-fluorine chain and a functional group that may be hydrophilic or hydrophobic. PFAS provides beneficial properties for many consumer products and industrial applications, mostly stick- and stain-resistance and surfactant qualities. PFAS's usefulness has led to a nearly ubiquitous presence in our lives, and PFAS's stability, due to the strength of carbon-fluorine bonds, result in long half-lives and the nickname "forever chemicals."

Human exposure to PFAS has been linked to detrimental health effects which impact all systems, including reproductive effects such as decreased fertility or increased high blood pressure in pregnant women, developmental effects or delays in children, including low birth weight, accelerated puberty, bone variations, behavioral changes, increased risk of some cancers, including prostate, kidney, and testicular cancers, reduced ability of the body's immune system to fight infections, including reduced vaccine response; interference with the body's natural hormones and increased cholesterol levels and risk of obesity (reviewed by Fenton et al., 2021). In response to the growing body of evidence identifying PFAS as a significant threat to human health and the environment, the United States Environmental Protection Agency (US EPA) is undertaking research to determine the impact of PFAS via a risk paradigm approach: (1) determine toxicity, (2) understand exposure, (3) assess risk, and (4) find and innovate effective treatment and remediation techniques and strategies. Because PFAS-containing products are disposed of at the end of their useful lives, significant PFAS quantities are managed with solid waste in the US and elsewhere. Properly managing solid waste via containment, treatment, and destruction is essential to protecting our environment and reducing the risk of harmful exposures.

Recognizing the impact of PFAS on human health and the environment, the US EPA released its first provisional Health Advisory Levels (HALs) for PFAS in drinking water in 2009. As analytical capabilities and scientific understanding of PFAS health impacts have improved, the Agency has promulgated additional guidance and risk-based thresholds. For the first time, in 2023, the US EPA proposed enforceable drinking water regulatory limits to reduce human exposure to PFAS (US EPA, 2022d). In April of 2021, the US EPA released the PFAS Strategic Roadmap, which outlines the EPA's commitments to action for 2021 through 2024. Information about US EPA PFAS initiatives is summarized in ***Table S1*** of the Supplementary information (SI), and applicable limits are included in Table 1. The US EPA has also proposed designating PFOA and PFOS as hazardous substances under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) and is considering adding certain PFAS to the Resource Conservation and Recovery Act (RCRA) list of hazardous constituents (US EPA, 2022b).

At the State level, all the US states except Arkansas, Louisiana, Mississippi, Nebraska, Texas, and Wyoming have dedicated websites providing PFAS-specific information. Some states have banned PFAS-containing products, as summarized in ***Table S2*** (SI). In contrast, others have initiated their own regulatory limits and advisory guidelines, as presented in ***Table S3*** (SI). Eight states have undertaken specific actions and introduced or passed bills targeting PFAS in solid waste (see ***Table S4***, SI). Notably, PFAS regulations are rapidly evolving, and any documentation of state-level PFAS initiatives will likely be outdated quickly.

Confronted with significant quantities of PFAS managed in landfills, the solid waste community struggles to understand the best means to manage PFAS-containing waste streams. Many studies have evaluated PFAS in landfills. However, there is a need for a critical review of the literature that would define the best methodologies for managing these wastes and identify data gaps to guide future research. This manuscript aims to inform the public and stakeholders from the solid waste industry

Table 1
Average concentrations (ng L^{-1}) of select PFAS in landfill leachate and US EPA risk-based thresholds.

Leachate matrix		PFOA		PFOS		PFNA		PFBS		PFHxS		PFHxA	5:3 FTCA
		Mean (n)	DF	Mean (n)	DF	Mean (n)	DF	Mean (n)	DF	Mean (n)	DF	Mean (n)	Mean (n)
MSW		*1400 (284)*	*23*	260 (284)	6.6	69 (234)	1.2	910 (234)	0.1	540 (234)	1.4	2800 (225)	3500 (86)
CDD		*1100 (17)*	*19*	660 (17)	17	50 (17)	0.8	530 (17)	0.1	2200 (17)	5.7	1600 (17)	1400 (17)
MSWI Ash		*800 (40)*	*13*	400 (40)	10	59 (40)	1.1	1400 (40)	0.2	510 (40)	1.3	1300 (40)	700 (40)
HW (Primary)		4900 (24)	81	*4100 (24)*	*102*	530 (24)	8.7	6500 (24)	1.1	12,000 (24)	32	12,000 (24)	NM
HW (Secondary)		*100 (5)*	*1.7*	14 (5)	0.4	40 (5)	0.7	57 (5)	0.01	86 (5)	0.2	440 (5)	NM
	Tapwater RSL (HQ = 1.0)	60		40		59		6000		390		n/a	n/a
EPA limit (ng L^{-1})	Lifetime HAL	0.004		0.020		n/a		2000		n/a		n/a	n/a
	Proposed MCL	4		4		n/a		n/a		n/a		n/a	n/a

(HAL = health advisory level; MCL = maximum contaminant level; RSL = regional screening level; HQ = hazard quotient; DF = average dilution factor required to meet RSL; NM = not measured).
Italicized values represent the controlling dilution factor.

about PFAS entering the waste stream and being disposed of in landfills, potential landfill PFAS emissions, and the related environmental impacts. Furthermore, this document highlights data gaps and uncertainties concerning the fate of PFAS during landfill disposal. Data were compiled and summarized, as described in the Methods section of the SI (***Section S2*** and ***Table S5***), to provide a concise critical review of this evolving research topic.

2. Solid waste management in the United States

A detailed discussion of solid waste management in the US is included in the SI (Section S3). Residents, businesses, and industries in the United States (US) generate significant amounts of solid waste; overall municipal solid waste (MSW) generation in 2018 was 265 million metric tons (US EPA, 2020b). In addition to MSW, significant amounts of construction and demolition (C&D) waste (545 million metric tons), wastewater treatment plant (WWTP) biosolids (2.5 million metric tons), and varied amounts of industrial waste and disaster debris enter the US solid waste management system every year (US EPA, 2020b). Over time, MSW generation in the US has increased. While the fraction of MSW which is landfilled has decreased from over 90% in 1960 to 50% in recent years, the mass of MSW disposed of in landfills reached its highest recorded level at 133 million metric tons in 2018 (see Fig. S1 in the SI). The US's landfill design, monitoring, and classification are identified and regulated according to the RCRA described in the SI's RCRA section. RCRA and its regulations provide requirements for landfill engineering controls based on the type of waste the landfill receives (MSW (Subtitle D), Hazardous (Subtitle C), industrial, construction, and demolition (C&D) debris) as outlined in the SI.

2.1. Sources of PFAS in solid waste

While extensive research has been undertaken to measure PFAS in effluent from waste management activities (particularly landfill leachate), fewer studies have attempted to estimate the PFAS load entering the waste management sector. Coffin et al. (2022) estimated an extractable ∑PFAS concentration in MSW of 50 μg kg^{-1} based on concentrations in MSW screenings reported by Liu et al. (2022a). Estimating PFAS loading to landfills is not only complicated by analytical challenges and the diversity of measurable PFAS, but also by the heterogeneity of MSW and other waste streams (e.g., household products, building materials, industrial waste, and "other wastes"). The following subsections focus on waste representing suspected high PFAS load or a significant fraction of the waste stream. Fig. 1 presents PFAS concentrations measured in various products and the environment compared to those measured in landfill leachate, compost, and biosolids from WWTPs.

2.1.1. Municipal solid waste

In the US, household waste is among the most significant fractions of MSW. Few studies measured the PFAS concentration of suspected PFAS-containing consumer products in the context of direct exposure during product use (Buck et al., 2011; Favreau et al., 2017; Glüge et al., 2020; Guo et al., 2009; Herzke et al., 2012; ITRC, 2022; Kotthoff et al., 2015; OECD, 2022; US FDA, 2022; Ye et al., 2015). These findings indicate a significant load of PFAS remaining in products at the end of their useful life. Household waste consists of two main categories: the biodegradable fraction and the non-biodegradable fraction. Both types of waste streams contain PFAS, but the fate of their PFAS may differ.

2.1.1.1. Biodegradable fraction. Paper and paperboard are the most

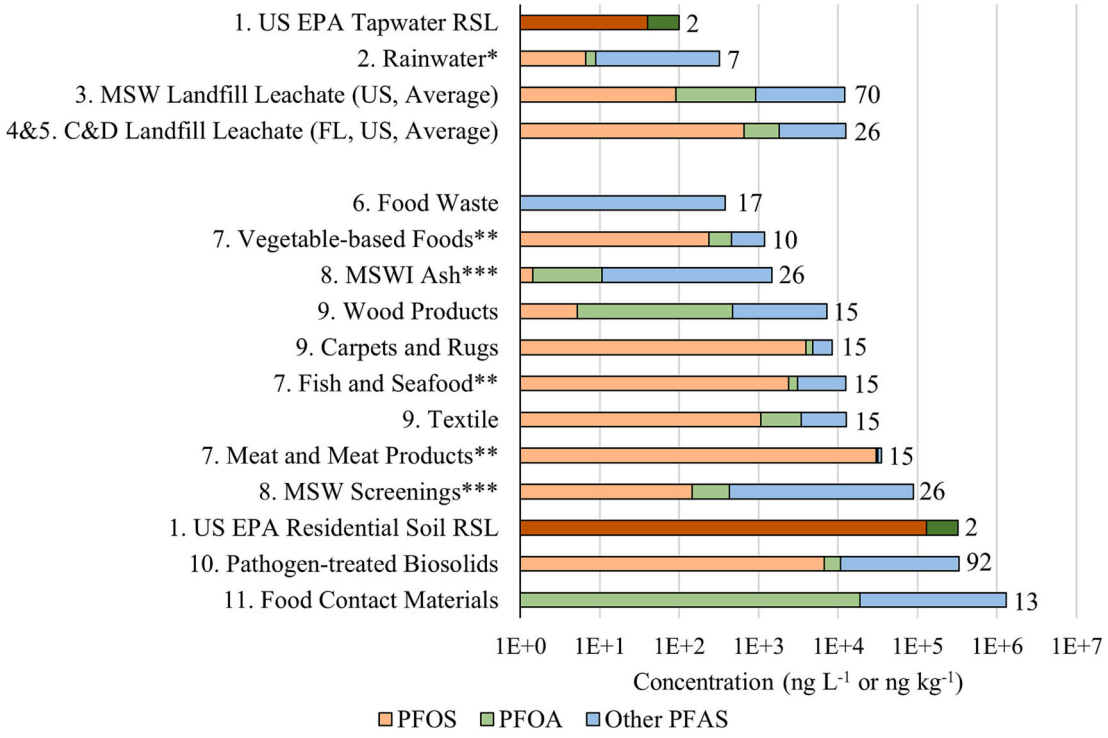


Fig. 1. PFAS concentrations and compositions measured in various products, wastes, and the environment compared to MSW landfill leachate.
* includes ultra-short chain PFAS, TFA.
** upper bound of the mean.
*** minimum total PFAS based on leachable fraction.
Note: numbers prior to matrix type refer to sources. Numbers to the right of the bars are the number of PFAS analytes. RSLs refer to risk-based screening levels, not enforceable regulatory limits. Sources: 1. US EPA (2022d) 2. Pike et al. (2021) 3. Lang et al. (2017) 4. Solo-Gabriele et al. (2020) 5. Chen et al. (2023) 6. Thakali et al. (2022) 7. European Food Safety Authority (2012) 8. Liu et al. (2022b) 9. Bečanová et al. (2016) 10. Thompson et al. (2023a, 2023b) 11. Siao et al. (2022).

abundant components of MSW, representing 23% of the US MSW generation in 2018 (US EPA, 2021a). PFAS are often added to paper products to improve stick and stain resistance, which results in paper products (including food packaging) consistently reported as a significant source of PFAS for human exposure and in the waste stream (Curtzwiler et al., 2021; D'eon et al., 2009; Ramírez Carnero et al., 2021; Robel et al., 2017; Seltenrich, 2020; Yuan et al., 2016; Zabaleta et al., 2016). In a review of studies that measured PFAS in food-contact materials, Siao et al. (2022) reported concentrations of $\sum_{13}$PFAS in food packaging as high as 8500 μg kg^{-1}; at these concentrations, paper and paper products likely contribute significantly to the overall PFAS loading in MSW, as well as contamination of food and food waste. Sapozhnikova et al. (2023) used targeted and total oxidizable precursor (TOP) assays to measure PFAS migration from food packaging into food products among 88 packaged food samples. TOP analysis identified a significant portion of total PFAS in packaging came from unknown precursor PFAS; average $\sum_{8}$PFAA was 28 μg kg^{-1} before oxidation and 380 μg kg^{-1} after oxidation. Migration from the packaging into food was found to increase over the course of the ten-day study. Unfortunately, many new products marketed as environmentally-friendly alternatives to plastic products have been found to contain PFAS (Timshina et al., 2021), and advocacy groups in the US and beyond have moved to revise compostable labeling to preclude PFAS-containing products (BioCycle, 2020). Disubstituted polyfluoroalkyl phosphates (diPAPs) have been found to represent a significant fraction of the PFAS used in paper products. However, most studies do not include diPAP as an analyte (D'eon et al., 2009; Thompson et al., 2023a). These findings suggest the concentration of $\sum$PFAS in paper products may be significantly higher than current estimates.

Another large fraction of biodegradable household waste is food waste, accounting for 22% of the MSW generated in the US in 2018 (US EPA, 2021a, 2021b, 2021c). Unlike paper products, PFAS are not intentionally added to food; contact with PFAS-containing equipment, packaging, water, feed, or soil amendments may result in residual PFAS. Several studies have been published describing the potential migration of PFAS from PFAS-impregnated food packaging (Ramírez Carnero et al., 2021). Up to 33% of extractable PFAS on the surface of food contact materials have been reported to migrate to simulated foods – the migration efficiency depends on the food type and PFAS class (Yuan et al., 2016). Additionally, some PFAS are known to bioaccumulate in the food chain. A European Food Safety Authority (2012) report lists seafood and meat as the food categories most frequently reported containing measurable concentrations of PFAS, with PFOS and PFOA quantified most commonly in 29% and 9% of samples, respectively. The same study estimated mean overall dietary exposure for PFOS and PFOA ranging from 0.07 to 32 ng kg^{-1} body weight per day, with lower exposure rates for 14 additional PFAS. Exposure was highest among toddlers and children due to higher food consumption for body size. Among 25 samples of food waste analyzed for PFAS by Thakali et al. (2022), 17 contained PFAS (mean $\sum_{17}$PFAS = 0.38 μg kg^{-1}); PFOS and PFOA were not detected in any of the samples.

Wood and yard trimmings represent approximately 18% of the US MSW generation (US EPA, 2020b). While natural wood and plant matter are unlikely to contain significant concentrations of PFAS (Thompson et al., 2023b), engineered wood building materials may be coated with PFAS to enhance performance. In a study of PFAS content in consumer and building materials, 100% of oriented strand board and wood products analyzed contained measurable PFAS concentrations, with median and maximum Σ_{15}PFAA of 5 and 18 μg kg^{-1}, respectively (Bečanová et al., 2016).

2.1.1.2. Non-biodegradable fraction. In the non-biodegradable category of household waste, carpets, and textiles have been consistently found to contain intentionally added PFAS that provide stick and stain resistance and waterproof properties (Kallee and Santen, 2012; Kim et al., 2015; Lang et al., 2016; Peaslee et al., 2020; van der Veen et al., 2022). A review of Σ_{15}PFAA in various household and consumer products found textiles, floor covering, and car interior materials represented the three highest maximum concentrations (78, 38, and 36 μg kg^{-1}, respectively); the highest non-biodegradable median PFAS concentration was from insulation (3.6 μg kg^{-1}) (Bečanová et al., 2016). PFAS and fluoropolymers are also used in non-stick cookware (Sajid and Ilyas, 2017) and electronics to provide smudge resistance, insulation, and other properties. An estimated 114 separate PFAS have been identified in electronic production (Garg et al., 2020). PFAS contamination and exposure through e-waste management have been the subject of several studies (Garg et al., 2020; Tansel, 2022; B. Zhang et al., 2020). Notably, the measurement of PFAS in e-waste itself (as opposed to through leachate, environmental contamination, or dust) is limited. A range of 0.07–0.43 μg kg^{-1} PFOS among all electronic products is provided by Garg et al. (2020). Σ_{15}PFAA reported by Bečanová et al. (2016) ranged as high as 11.7 μg kg^{-1} (median: 0.4 μg kg^{-1}) in electronic and electrical equipment (EEE) and as high as 2.2 μg kg^{-1} (median: 1.4 μg kg^{-1}) in waste EEE.

2.1.2. Industrial waste

Industrial processes generate waste and effluent in large volumes; processes that use PFAS, such as the leather tannery, chrome plating, and textile industries, represent a significant contribution of PFAS to the solid waste stream (ITRC, 2022) which are often disposed of in landfills. Other types of industrial processes which generate PFAS-containing waste involve the management of PFAS-contaminated materials, including the separation of wastewater biosolids as part of municipal wastewater treatment, the management of MSW incineration residuals (MSWI ash), and the disposal of PFAS-contaminated soils and other residuals generated as part of environmental cleanup processes.

2.1.2.1. Biosolids. WWTPs manage residential, commercial, and industrial wastewater, including landfill leachate, and have been the subject of many PFAS studies (Lenka et al., 2022). Biosolids account for a significant fraction of WWTP effluent (Fredriksson et al., 2022) and impact PFAS loading to the environment and landfills (Johnson, 2022; Thompson et al., 2023b). Reported PFAS concentrations in biosolids vary with the number of PFAS included in the analytical method. Gallen et al. (2018) reported mean $\sum_{9}$PFAS of 45 μg kg^{-1} of biosolids; Thompson et al. (2023a, 2023b) reported mean $\sum_{92}$PFAS of 500 μg kg (dry)$^{-1}$ in untreated biosludge and 330 μg kg(dry)$^{-1}$ in biosolids (biosludge treated for pathogen removal), indicating that early studies of PFAS in biosolids which measured fewer PFAS, and predominantly PFAAs, did not capture a significant portion of the total PFAS. Over 5.8 million dry metric tons of biosolids were managed in the US in 2018, of which 30% was managed in landfills, 15% was incinerated, and over 50% was used as a soil amendment (NEBRA, 2022).

2.1.2.2. MSW incineration ash. The incineration of MSW for energy recovery (MSWI) produces two solid waste streams – bottom ash, the material that does not burn, and fly ash, fine particulate matter collected in the air pollution control system. Approximately 13% of MSW in the US is managed through incineration (US EPA, 2021a, 2021b, 2021c), resulting in an estimated 7.5 million tons of MSWI ash (Liu et al., 2019). Few studies have measured PFAS in MSWI ash. Liu et al. (2021b) reported $\sum_{21}$PFAS in fly and bottom ash from three facilities in China, with concentrations ranging from 1.5 to 88 μg kg^{-1} in fly ash and from 3.1 to 77 μg kg^{-1} in bottom ash. Based on the concentrations of PFAS in a laboratory leaching study, the average minimum $\sum_{26}$PFAS in MSWI ash from a US facility was 1.5 μg kg^{-1} (Liu et al., 2022b); this represents a conservative estimate of total PFAS. These concentrations are in the same range as MSW. Incineration temperatures may not be sufficiently high to mineralize or destroy PFAS, and operational strategies likely play a significant role in the fate of PFAS during incineration. The impact

of temperatures on PFAS leaching from MSWI is discussed in greater detail in the context of MSWI ash monofill leachates.

2.1.2.3. Manufacturing wastes. There is extensive use of PFAS in some industries, as PFAS is added intentionally to products (i.e., to produce stain-resistant properties in textiles and paper products) and as part of the manufacturing process (i.e., to facilitate demolding). This results in unintentionally contaminated materials through contact. The Interstate Technology Review Committee (ITRC) thoroughly lists PFAS uses in the industrial and manufacturing sectors (ITRC, 2022). PFAS-laden manufacturing waste is often sent to landfills for disposal across industries.

Among specific industries and industrial wastes which have been the subject of PFAS analysis, high-concentration effluents from electronic industries have been described in the literature; photolithographic effluent in Taiwan contained 130,000 ng L^{-1} each of PFHxS and PFOS (Lin et al., 2009); liquid effluent from television and circuit board manufacturing contained 1600 ng L^{-1} of $\sum_{11}$PFAS (Kim et al., 2016); sludge effluent collected from an electronics industry location in South Korea contained 91 μg kg^{-1} of $\sum_{11}$PFAAs (Kim et al., 2016). PFAS are used commonly in paper processing and treatment; a case study in Norway identified PFAS impacts downstream of a landfill used for paper factory waste disposal (Langberg et al., 2021). Chrome plating industry waste sludges are designated hazardous wastes (F006), which contain high concentrations of PFAS (ITRC, 2022) and are therefore managed in Subtitle C hazardous waste landfills. A study of chrome sludge in China identified PFOS as the most predominant PFAS at concentrations as high as 2435 μg kg^{-1} (Qu et al., 2020). The chrome plating industry consumes an estimated 6500 kg of PFOS annually (Garg et al., 2020).

2.1.2.4. PFAS remediation residuals. Sites with high levels of PFAS contamination from the historical use of PFAS-containing aqueous film-forming foams (AFFF) or other releases are frequently remediated, and the contaminated media is commonly disposed of in landfills (either with the waste or used as daily cover). Remediation approaches include mobilization of PFAS and collection of the leachate, sorption of PFAS using activated carbon or other sorbents, or soil excavation for landfill disposal (Bolan et al., 2021; Ross et al., 2018). Brusseau et al. (2020) reviewed PFAS concentrations measured in soils from contaminated sites, reporting median PFOA and PFOS concentrations of 83 and 8700 μg kg^{-1}, respectively, with concentrations as high as 50,000 μg kg^{-1} for PFOA and 460,000 μg kg^{-1} for PFOS.

3. Fate of PFAS in landfills

The fate of solid waste-derived PFAS within landfills is dominated by transformation and partitioning. Many PFAS species are persistent in the environment and PFAS that are degradable can transform into more recalcitrant, typically more environmentally mobile PFAS (Bolan et al., 2021). The partitioning behavior of PFAS are related to the chemical structure of individual species, both according to PFAS class, functional groups, and chain length among homologous species. In turn, the ongoing transformation will impact partitioning behavior (Robey et al., 2020; Zhang et al., 2020a, 2020b; Smallwood et al., 2023). In landfills, PFAS may partition to the liquid phase (leachate) and gaseous phase (landfill gas; LFG), remain sorbed to the waste, and/or interact with the engineering controls of the landfills (e.g., leachate collection systems, gas collection, and control systems). PFAS that are resistant to degradation and minimally soluble or volatile, such as certain polymeric PFAS, have historically been presumed to remain immobile and sequestered in landfills, although more recent studies have called this assumption into question (Lohmann et al., 2020).

3.1. PFAS transformation

Many studies observed the transformation of PFAS precursors into terminal species under abiotic and microbially active aerobic and anaerobic conditions. While this section briefly reviews these processes to provide context to PFAS in landfills, the aim is not to conduct an exhaustive review of the topic, which is available in other reviews (Lu et al., 2023).

3.1.1. Abiotic transformation

PFAS transformation pathways under abiotic conditions include oxidation, photolysis, and thermal degradation (ITRC, 2020; Washington and Jenkins, 2015). While the bulk of PFAS transformations in organic-rich landfills are likely a result of biodegradation, these abiotic processes play an essential role in solid waste management systems. PFAS such as fluorotelomer alcohols (FTOHs) can volatilize under temperatures typical in landfills (35–55 °C). Once in the atmosphere, FTOH can transform via photolysis or other chemical reactions into perfluorocarboxylic acids (PFCAs) which are then deposited on land and waterbodies (Esfahani et al., 2022; Martin et al., 2006). Other abiotic processes include thermal degradation. An increase in the temperature of waste may facilitate PFAS transformation. Wastes within landfills may be exposed to temperatures insufficient to mineralize or defluorinate PFAS but which may cause precursor transformations. Thompson et al. (2023a, 2023b) measured higher concentrations of diPAPs in biosolids that had undergone any form of heat treatment, including heat drying as well as higher temperature vector reduction treatment, indicating the presence and transformation of unidentified precursors.

3.1.2. Aerobic transformation

Aerobic environments exist at the early stages of landfill decomposition. The waste still contains atmospheric oxygen in its void space and likely contributes to the transformation of PFAS in waste. Thompson et al. (2023a, 2023b) observed a proportional increase in PFCAs after aerobic biosolids composting, especially short-chain (perfluoropentanoic acid, PFPeA, and perfluorohexanoic acid, PFHxA). Similarly, Li et al. (2022) found significant increases among short-chain PFAAs (including PFBS and PFOS) in aerobically treated anaerobic digestor sludge. These findings are significant because short-chain PFAS are more mobile in the environment, more likely to be uptaken by plants (Ghisi et al., 2019), and more challenging to treat (Ross et al., 2018). Multiple studies have shown that aerobic decomposition facilitates the transformation of precursor PFAS to shorter-chain terminal PFAS, such as PFOA and PFOS (Hamid et al., 2020; Lee et al., 2010a; Liu et al., 2010; Lott et al., 2023; Rhoads et al., 2008; Schultz et al., 2006; Wang et al., 2009, 2011; Zhao et al., 2013).

3.1.3. Anaerobic transformations

Comparatively, fewer studies have documented PFAS transformation under anaerobic or methanogenic conditions similar to landfills. Liu et al. (2021a) compared 52 PFAS in leachate from waste collection vehicles to anaerobic MSW landfill leachate and concluded the vehicle leachate contained proportionally more precursor PFAS and short-chain PFAAs compared to the landfill leachate as a result of the transformation in the anaerobic landfill environments. Studies of anaerobic precursor transformation identified FTCAs as the predominant by-product of FTOH degradation. Allred et al. (2015) reported increased MeFBSAA and FTCA leaching over abiotic reactors in biologically active landfill microcosm reactors, indicating that methanogenic biological transformation was responsible for the increase. Zhang et al. (2013) observed the accumulation of FTCAs in landfills over time, concluding that FTCAs are indicators of FTOH transformation, while Lang et al. (2016) and Weber et al. (2022) reported PFOA accumulation in leachate as a result of precursor transformation under anaerobic experimental conditions. Lang et al. (2016) attributed this to the longer experimental duration, with PFOA appearing as a significant degradation by-product only 200+

days into the 550-day experiment.

3.2. PFAS partitioning in landfills

3.2.1. PFAS partitioning to the liquid phase

PFAS concentrations in landfill leachate are a function of multiple factors, including the PFAS profile of the incoming waste stream and conditions within the landfill, which, in turn, correspond with waste composition, stage of decomposition, and environmental factors, especially rainfall precipitation. These factors also affect the physical-chemical aspects of the leachate quality, and any discussion of PFAS in leachate should also include matrix contextualization. The number of PFAS that can be detected and quantified in landfill leachate has grown. Early methods were able to quantify 24 PFAS compounds in three classes (Huset et al., 2011), but improvements have been made; more recent studies attempted to measure 92 PFAS and detected 53, as presented in Table 2.

3.2.1.1. PFAS in landfill leachate by type

3.2.1.1.1. MSW landfills. The vast majority of PFAS landfill leachate data are measured from MSW landfills (Allred et al., 2014; California Water Boards, 2023; Chen et al., 2022, 2023; Huset et al., 2011; Lang et al., 2017; Liu et al., 2022a; Masoner et al., 2020; NWRA, 2020; Solo-Gabriele et al., 2020). The ∑PFAS content of MSW landfill leachate in published US studies ranges from BDL - 125,000 ng L^{-1} with an average of 10,500 ng L^{-1} and a weighted average of 12,600 ng L^{-1}. The weighted average is notably similar to the estimated average ∑PFAS concentration reported by Lang et al. (2017) using Monte Carlo simulation. Often, the ∑PFAS content heavily depends on the number of unique PFAS measured in the study, which ranged from two to 70 for MSW landfill leachate (see SI ***Fig. S2***). For comparison among studies, we will focus on PFAS with corresponding US EPA tapwater Regional Screening Levels (RSL) (i.e., PFOA, PFNA, PFBS, PFHxS, PFOS, and Gen-X), as presented in Table 1. Except for Gen-X, which has only been quantified in a single sample of landfill leachate from a North Carolina MSW landfill with a history of accepting PFAS manufacturing wastes (NWRA, 2020), the remaining five PFAS are reliably quantified in all published landfill leachate studies. Other PFAS which reliably contribute significantly to ∑PFAS in landfill leachates, PFHxA and 5:3 FTCA, are also included in Table 1.

PFAS concentrations have also been reported for leachates from MSW landfills in other countries, including Australia (Gallen et al., 2016, 2017), Europe (Ahrens et al., 2011; Busch et al., 2010; Eggen et al., 2010; Fuertes et al., 2017; Kallenborn et al., 2004; Knutsen et al., 2019; Perkola and Sainio, 2013; Woldegiorgis et al., 2005), and Asia (Huang et al., 2022; Liu et al., 2022b; Yan et al., 2015; Yin et al., 2017; Zhang et al., 2014). International differences in waste composition, sample collection, and analytical processes can impact reported PFAS concentrations, making a direct comparison of the overall PFAS content challenging. Concentrations of PFOS and PFOA, which have been reliably measured in most or all studies, are included for eight countries in ***Table S6*** and described by Travar et al. (2020).

Table 2
Number of PFAS measured and ∑PFAS among published landfill leachate studies.

Matrix	Number of samples	Number of PFAS detected (in Method)	Average ∑PFAS (ng L^{-1})	∑PFAS range (ng L^{-1})	Country	Reference
MSW LL	1	38 (51)	9700	9700	USA	Liu et al. (2021a)
	1	32 (51)	9400	9400	USA	Robey et al. (2020)
	78	25 (26)	12,700	300–58,000	USA	Chen et al. (2023)
	4	10 (11)	17,200	15,000–18,000	USA	Solo-Gabriele et al. (2020)
	6	24 (24)	4700	2700–7400	USA	Huset et al. (2011)
	40	30 (70)	12,200	2000–29,000	USA	Lang et al. (2017)
	11	2 (2)	840	330–2600	USA	Clarke et al. (2015)
	19	28 (28)	5400	230–29,000	USA	Helmer et al. (2022)
	39	2 (2)	1500	47–3400	USA	EGLE (2019)
	9	22 (25)	24,300	1400–125,000	USA	NWRA (2020)
	131	31 (40)	17,500	BDL – 104,000	USA	California Water Boards (2023)
	17	14 (14)	3000	33–15,000	Australia	Gallen et al. (2016)
	94	9 (9)	6100	210–46,000	Australia	Gallen et al. (2017)
	22	15 (15)	7000	Not reported	Australia	Simmons (2019)
	6	25 (43)	6100	31–13,000	Germany	Busch et al. (2010)
	11	24 (24)	9800	2500–36,000	Canada	Benskin et al. (2012)
	31	16 (18)	2700	700–6400	Canada	Li (2009)
	10	2 (2)	*	50–2300	Canada	Gewurtz et al. (2013)
	2	16 (27)	4200	2200–6100	Norway	Eggen et al. (2010)
	5	7 (8)	770	200–1500	Norway	Kallenborn et al. (2004)
	2	4 (4)	400	210–610	Finland	Perkola and Sainio (2013)
	48	7 (10)	2400	14–17,500	Ireland	Harrad et al. (2019)
	4	8 (16)	1100	640–1400	Spain	Fuertes et al. (2017)
	12	28 (30)	1700	320–11,000	Norway	Knutsen et al. (2019)
	10	17 (26)	490	0.3–1300	Sweden	Gobelius et al. (2018)
	5	11 (14)	82,100	7300–290,000	China	Yan et al. (2015)
	9	33 (57)	42,900	3040–109,000	China	Liu et al. (2022b)
	6	17 (17)	14,200	1800–43,300	China	Huang et al. (2022)
	12	18 (18)	4060	1270–7660	Singapore	Yin et al. (2017)
	5	8 (9)	6000	4200–11,000	Australia	Gallen et al. (2017)
CDD LL	13	24 (26)	9500	270–30,500	USA	Chen et al. (2023)
	2	11 (11)	15,500	14,000–16,000	USA	Solo-Gabriele et al. (2020)
MSWIA LL	2	9 (11)	3100	2800–3400	USA	Solo-Gabriele et al. (2020)
	31	26 (26)	7300	39–54,500	USA	Chen et al. (2023)
MSW GC	21	26 (26)	12,200	199–80,900	USA	Chen et al. (2023)
	12	53 (92)	19,000	3000–50,000	USA	Smallwood et al. (2023)
HW LL (Primary)	24	17 (28)	68,000	570–377,000	USA	California Water Boards (2023)
HW LL (Secondary)	5	13 (24)	1800	25–3700	USA	California Water Boards (2023)

(LL = landfill leachate; MSW = municipal solid waste; CDD = construction and demolition debris; MSWIA = MSW incineration ash; GC = gas condensate).
* Gewurtz et al. (2013) do not provide detailed data to calculate average ∑PFAS.

3.2.1.1.2. C&D landfills. PFAS were detected in all C&D landfill leachate samples analyzed across three studies with ∑PFAS ranging from 270 to 30,500 ng L^{-1} (weighted average 10,300 ng L^{-1}). Solo-Gabriele et al. (2020) and Chen et al. (2023) found no significant difference in the total measured PFAS between leachate from MSW and C&D landfills. The analytical method used by Chen et al. (2023) included 18 terminal PFAS (PFAAs) and eight precursors (FASAs, FTCAs, and FTSs). The study, however, reported a significant difference in the fraction of ∑Terminal and ∑Precursor species between MSW and C&D landfill leachates. C&D leachate contained, on average, 86% terminal PFAS, while MSW leachate contained 64% terminal PFAS (Chen et al., 2023). This could be attributed to the different types of PFAS present in each waste stream and the type of biological activity prevalent in each landfill type. Because C&D landfills contain proportionally less food waste and more concrete and gypsum drywall, the prevailing landfill conditions result in higher pH leachate and proportionally more sulfate chemical species in the leachate as opposed to ammonia, which is typically at higher concentrations in MSW landfill leachate (Townsend et al., 1999). Further, due to those differences in leachate conditions, microbial differences result from presence of different carbon sources as well as electron donors and acceptors. Generally, sulfur-reducing bacteria are found in higher concentrations at C&D landfills due to higher amounts of sulfate, while methanogens are more prevalent at conventional landfills (Meyer-Dombard et al., 2020).

Fig. 2 includes the range of concentrations for PFAS with RSLs for MSW and C&D landfill leachate; average PFHxS concentrations were higher in C&D landfill leachate than in MSW landfill leachate, and PFBS concentrations were lower in C&D landfill leachate. Waste composition is highly variable between landfills as well as over time at an individual landfill, so, while limited studies may suggest potential sources of select PFAS in C&D debris (e.g., higher concentrations of PFHxS may be attributed to their use in carpeting and other building materials (Beesoon et al., 2012; Jin et al., 2011), generalizations about specific sources may not be appropriate. Gallen et al. (2017) measured nine terminal PFAS in Australian C&D landfill leachates (n = 5), reporting average $\sum_9$PFAS concentrations of 6000 ng L^{-1} (compared to 6100 ng L^{-1} in 94 MSW leachates from the same study).

Unlike MSW landfills, at the US federal level, C&D landfills do not require a bottom liner and leachate collection systems. This contributes to the lower number of studies describing PFAS in C&D relative to MSW landfill leachate and an increase in the probability of groundwater contamination from C&D compared to MSW landfills. Average concentrations of PFOA, PFOS, PFNA, PFBS, and PFHxS, along with corresponding US EPA risk-based thresholds (HALs, MCLs, and RSLs), are included in Table 1. PFOA poses the most significant challenge as its concentration in C&D landfill leachate would have to be diluted by 19 to meet the tapwater RSL or by 287 to meet the US EPA proposed MCL.

3.2.1.1.3. MSWI Ash monofills. Solo-Gabriele et al. (2020) and Chen et al. (2023) found leachate from MSWI ash monofills to have lower ∑PFAS concentrations than leachate from MSW landfills. Solo-Gabriele et al. (2020) reported $\sum_{11}$PFAS in MSWI ash monofill leachates ranging from 2800 to 3400 ng L^{-1} and inversely correlated with incineration temperature. $\sum_{11}$PFAS in leachate from MSWI ash that underwent incineration at 800 °C was almost three times higher than after incineration at 950 °C. The decrease indicates loss of measurable PFAS via mineralization (i.e., destruction), volatilization (i.e., air emission), or transformation to PFAS species which are not measured in standard analytical methods (e.g., products of incomplete combustion or PICs). Leachates from MSWI ash which had undergone incineration at 950 °C, still contained >2000 ng L^{-1} of PFAS, indicating PFAS are not fully mineralized at these operating conditions. Liu et al. (2021b) reported substantially higher $\sum_{21}$PFAS in MSWI ash leachate from three facilities in China, with concentrations ranging from 127,000–450,000 ng L^{-1}. The study did not report incineration temperatures or other operating conditions.

However, when MSWI ash was co-disposed with other wastes, such as MSW or biosolids, ∑PFAS concentration in the leachate was on par with that in MSW landfill leachate (Solo-Gabriele et al., 2020; Liu et al., 2022a). Liu et al. (2022a) found the co-disposal of a small fraction (e.g., 4%) of MSW, including biosolids, with MSWI ash resulted in leachate concentrations that were comparable to MSW landfill leachate, suggesting liquids are preferentially flowing through and leaching PFAS from the non-incinerated waste as opposed to the ash. While MSWI ash-derived leachates have lower concentrations of PFAS, these studies suggest care should be taken to dispose of MSW and MSWI ash separately, and more research is needed to understand the fate of PFAS during MSW incineration.

3.2.1.1.4. Industrial landfills. Unlined industrial landfills that received residuals from manufacturing PFAS and PFAS-containing products have been linked to contamination of local groundwater sources. Notable examples include the House Street landfill in Belmont, Michigan which received tannery waste (US EPA, 2022e); Crown Vantage landfills in Parchment, Michigan (MPART, 2020), that were used to dispose of paper mill waste from the production of laminated paper

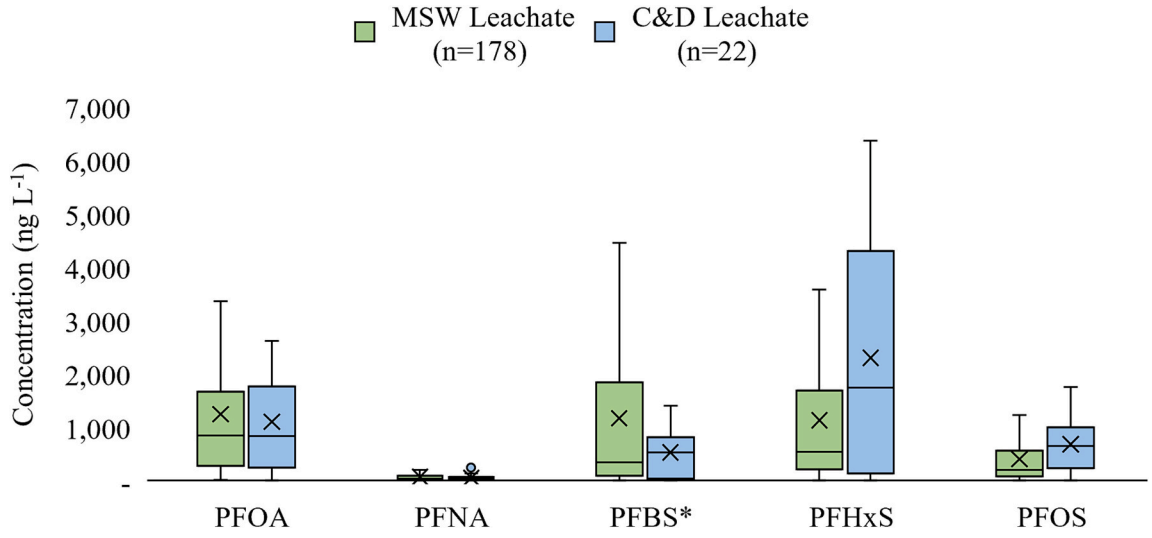


Fig. 2. Average concentrations of five PFAS with US EPA tapwater RSLs.
Data from Gallen et al. (2017), Solo-Gabriele et al. (2020), and Chen et al. (2023).
* Gallen et al. (2017) did not include PFBS analysis.

products; and the 3M Woodbury disposal site in Washington County, Minnesota, that was used to dispose of PFAS production waste. As part of this literature search, no leachate PFAS concentration data from industrial landfills in the US were located. However, Kameoka et al. (2022) measured PFAS in leachate from three industrial landfills in Japan; $\sum_{17}$PFAA concentrations averaged 45,000 ng L^{-1}.

3.2.1.1.5. Hazardous waste landfills. Although PFAS are not federally regulated as listed hazardous wastes, some solid wastes managed in Subtitle C hazardous waste landfills contain PFAS (as discussed in Section 2.1.2, e.g., chrome-plating sludge), while other PFAS-containing wastes may meet hazardous waste characteristic criteria (e.g., flammable, corrosive, etc.). Some hazardous waste landfills have also reported receipt of AFFF waste at their sites. No peer-reviewed studies have evaluated PFAS concentrations in leachate collected from hazardous waste landfills; however, California Water Boards have released PFAS concentrations for landfill leachate, including two hazardous waste landfills in California (California Water Boards, 2023). The data for these sites are included in the SI ***Table S7***. Among 29 samples from the two sites, $\sum_{24}$PFAS and $\sum_{28}$PFAS concentration was as high as 377,000 ng L^{-1} (average 68,000 ng L^{-1}), substantially higher than MSW, C&D debris, or MSWI ash landfill leachates (see Table 2). In the US, hazardous waste landfill disposal requires waste pre-treatment to minimize contaminant mobility – land disposal restrictions for hazardous waste are described in the Code of Federal Regulations (CFR; 40 CFR § 268). Leaching studies have shown minimal PFAS immobilization using traditional solidification techniques (Barth et al., 2021), which may explain elevated PFAS concentrations in the leachate.

US hazardous waste landfills must also use secondary leachate collection systems; California's database includes five samples of secondary hazardous waste landfill leachate from one site, with $\sum_{24}$PFAS averaging 1800 ng L^{-1} (see Table 2). Without exception, for all sampling locations with both primary and secondary leachate PFAS data, concentrations for individual and $\sum_{24}$PFAS were higher in the primary compared to the secondary leachate. While the absence of biological decomposition in hazardous waste landfills may minimize the microbially-mediated precursor transformation to PFAAs, waste treatment methods (e.g., lime treatment) may also impact transformation and partitioning, possibly oxidizing precursor PFAS. Hazardous waste pretreatment standards are designed to minimize traditional hazardous waste constituent leaching (e.g., lime treatment stabilizes metals and neutralizes acidic waste) and have not been optimized for PFAS stabilization; PFAS fate, transport, and transformations under hazardous waste pretreatment processes are not well understood. Because of the strict Subtitle C landfill operation requirements and the pre-treatment of wastes, leachate generation in these landfills is typically minimal, and any leachate which is produced is often managed as hazardous waste (i.e., not discharged to WWTP, as other landfill leachates often are).

3.2.1.2. Other factors impacting PFAS concentrations in leachate

3.2.1.2.1. Waste age. As waste degrades under the anaerobic conditions of biologically active landfills, the overall PFAS concentrations in the leachate and the ratio of the terminal to precursor species have been found to increase. Lang et al. (2017) reported leachate from waste older than ten years had significantly lower concentrations of PFNA, 8:2 FTCA, 5:3 FTCA, PFBS, MeFBSAA, and MeFOSAA than leachate from younger waste. These differences could be attributed to changes in the PFAS formulations in commercial products and/or the conversion of PFAA precursors. Liu et al. (2021a) measured PFAS in leachate from waste collection vehicles alongside leachate from the receiving MSW landfill. The study found significantly higher $\sum_{51}$PFAS concentrations in landfill leachate which had undergone further biological decomposition. Furthermore, Liu et al. (2021a) also reported a difference in PFAS profiles likely caused by the transformation of precursor PFAS in landfill environments.

3.2.1.2.2. Leachate quality. Although most PFAS behavior and solution chemistry studies focus on remediation technologies, generalizations regarding PFAS phase partitioning also apply to landfill leaching (Z. Du et al., 2014). Comparatively, fewer studies have explored PFAS partitioning in the context of leachate chemistry. In a landfill simulator study, Allred et al. (2015) observed increases in longer-chain PFCA and perfluoroalkyl sulfonic acids (PFSA) concentrations when biodegradation reached the methanogenic stage. At this stage, increased methanogenic and secondary fermentation and decreased volatile fatty acid concentrations from the acidogenic stage result In increased pH, more neutral pH, which is theorized to deprotonate waste surfaces, resulting in less sorption of PFAS to the degrading organic matter. This theory is supported by the results described by Solo-Gabriele et al. (2020), where a significant positive correlation was reported between PFAS concentrations and increasing leachate pH. This effect has also been observed in several previous landfill leachate sampling studies (Benskin et al., 2012; Gallen et al., 2017; Hamid et al., 2018; Yan et al., 2015).

In addition to partitioning behavior, the PFAS profile of landfill leachate is a function of PFAA precursor transformation resulting from biodegradation. Biological activity is catalyzed by landfill moisture, resulting in higher landfilled waste temperatures and more PFAS transformation. In a study of WWTP biosolids pathogen removal, precursor transformation and apparent increases in $\sum_{92}$PFAS, driven by increased PFAA content, resulted after aerobic composting and increased diPAP concentrations from heat treatment (Thompson et al., 2023b). Based on a nationwide study of 95 leachate samples collected from 18 landfills, leachate from MSW landfills in US regions with high annual precipitation showed significantly greater $\sum_{19}$PFAS than comparable landfills in arid locations (Lang et al., 2017); see Table 2 for all US-based studies included in this review. Further, leachate generation volume is significantly higher in regions that experience more precipitation. As a result, landfills in arid regions are estimated to contribute <1% of the nationwide landfill leachate PFAS mass load (Lang et al., 2017). When studies have evaluated the short-term impacts of precipitation on PFAS in landfill leachate, however, leachate PFAS concentration decreased within a day of a precipitation event due to dilution (Benskin et al., 2012; Gallen et al., 2017). Normalization of PFAS concentrations to bulk parameters such as chloride or total dissolved solids may be able to account for such dilution.

3.2.2. PFAS partitioning to the gas phase

MSW contains a proportionally more biodegradable organic matter which undergoes anaerobic decomposition in landfill environments compared to other waste streams (e.g., C&D). The decomposition of organic matter produces MSW LFG, which is, on average, about 50% methane (CH_4), and 50% carbon dioxide (CO_2), with a small fraction consisting of other gaseous and volatile constituents (Wang et al., 2021). LFG at MSW landfills is collected and managed according to the requirements of the US EPA New Source Performance Standards (US Clean Air Act, 40 CFR § 60). According to the US EPA's Landfill Methane Outreach Program (LMOP) August 2022 database, 1230 of the 2635 MSW landfills in the US have gas collection systems in place, and 1157 have flares in place (US EPA, 2022a). C&D LFG is rarely collected in the US, as C&D landfills contain less biodegradable organic matter and produce less LFG than MSW landfills. Additionally, C&D LFG contains proportionally more $H_2S_{(g)}$ produced by sulfur-reducing bacteria and the decomposition of gypsum disposed of as drywall.

Gas generation and composition at other landfill types has yet to be the subject of significant research. MSWI ash monofills are not expected to generate LFG because there is minimal biodegradable matter in the ash; however, the co-disposal of WWTP biosolids, MSW, or any degradable organic matter with MSWI ash will produce biogas as a result of decomposition. Gas generation at industrial landfill sites is primarily a function of the type of waste deposited. Organic waste like pulp and paper mill sludges will likely generate gas requiring management. In general, Subtitle C hazardous waste landfills in the US do not contain

putrescible organic waste and do not generate biogas.

3.2.2.1.1. PFAS in MSW landfill gas. PFAS volatilization and release from MSW landfills within the gaseous phase is receiving an increased focus driven by advances in volatile PFAS measurement (Riedel et al., 2019) and an improved understanding of PFAS chemistry. The partitioning coefficients (e.g., Henry's constant) for ionizable PFAS are significantly lower than neutral PFAS (Abusallout et al., 2022), making ionizable PFAS less likely to volatilize under typical MSW landfill conditions. Experimental measurement of PFAS vapor pressures similarly suggests FTOHs (i.e., neutral PFAS) are more readily volatilized than PFCAs (i.e., ionizable PFAS) and that vapor pressure decreases logarithmically with carbon chain length in homologous species (M. Zhang et al., 2020). Measurement and data of PFAS in actual MSW LFG are still minimal.

In a 2007 analysis of landfills that accepted PFAS-containing industrial wastes, the MPCA detected several PFAS (12 PFAAs and perfluorooctane sulfonamide, PFOSA) in MSW LFG with $\sum_{13}$PFAS ranging from 4.1 to 18.7 ng m^{-3} (MPCA, 2010). Titaley et al. (2023) measured neutral PFAS in LFG of three active MSW landfills (n = 12 samples) and reported concentrations of four n:2 FTOHs (n = 6, 8, 10, and 12), one fluorotelomer acrylate (6:2 FTAc), and one fluorotelomer olefins (12:2 FTO). Concentrations for individual PFAS range from 270 to 4900 ng m^{-3}, and the total measured neutral PFAS for each landfill was, on average, between 4600 and 14,000 ng m^{-3} (weighted average across all samples: 10,200 ng m^{-3}). Smallwood et al. (2023) reported FTOH in LFG condensate, which, when normalized to gas volume, was three orders of magnitude lower than the gaseous phase concentrations reported by Titaley et al. (see SI ***Table S8*** for calculations), indicating FTOHs preferentially partition to the gas phase; FTOHs may transform in the atmosphere into PFCAs, such as PFOA, which have known and suspected toxic effects.

3.2.2.1.2. PFAS in C&D landfill gas. While no data exist on the concentration of PFAS in C&D LFG, it can be conservatively assumed, based on data from MSW LFG measurements, that PFAS also leave C&D landfills via gas effluent. As previously described, PFAS-containing wastes are disposed of at C&D landfills, and it is highly likely C&D debris contains volatile PFAS, such as FTOHs, which readily transform into FTCAs and PFCAs as a result of biodegradation and environmental oxidation, respectively. Lower rates of biological activity in C&D landfills may result in slower biodegradation of PFAS like FTOH (and other volatile precursors) may persist longer in C&D compared to MSW landfills and therefore have more opportunity to volatilize and leave the landfill via LFG. This is likely offset by the lower volume of LFG generated overall at C&D landfills compared to MSW. Nonetheless, this read-across should be validated by experimental data.

3.3. Fate of PFAS in traditional landfill leachate and gas management systems

Most landfills compliant with New Source Performance Standards (Clean Air Act) and RCRA must capture gas effluent and leachate to minimize environmental impacts. Leachate is often intercepted using a low-permeability bottom liner made of high-density polyethylene, collected, and may be transported off-site to a WWTP, disposed of using deep well injection, or otherwise managed and treated on-site.

PFAS interactions with low-permeability landfill liners have been the subject of limited studies. Most landfill liners are constructed from polyethylene geomembranes. Laboratory studies of PFAS diffusion through linear low-density polyethylene report below detection diffusion rates (Di Battista et al., 2020). Diffusion through high-density polyethylene has yet to be reported but maybe even lower due to differences in material structure. Landfill liner integrity – the absence of flaws or holes – is the most critical factor in preventing PFAS transmission through geomembrane and composite liners (Di Battista et al., 2020). An analysis of landfill liner performance reported median leakage rates of 44 and 33 L ha^{-1} day^{-1} for geomembrane and composite liners, respectively, with an overall liner collection efficiency of 98% (Jain et al., 2023). Compacted clay liners, which are more common in older landfills and C&D landfills, do not adsorb PFAS, which are reported to pass through bentonite clay at the same rate as other mobile leachate constituents like chloride (Li et al., 2015). PFAS profile, leachate quality, and soil characteristics all play a role in soil interaction, and decisions should be made on a site-specific basis (Li et al., 2019; Gates et al., 2020; Mukhopadhyay et al., 2021).

Management of leachate in the US is dependent on climate – in dry regions, leachate generation is minimal, and many facilities use atmospheric evaporation. In contrast, in wet regions, leachate management presents a significant challenge (US EPA, 2021c). A nationwide survey found approximately 60% of US Subtitle D landfills conveyed their leachate to WWTPs for off-site treatment, 28% recirculated leachate or use other techniques resulting in no necessary leachate treatment, and 12% used on-site treatment (US EPA, 2021c). A breakdown of on-site leachate treatment strategies is included in SI ***Fig. S3***. Traditional leachate treatment typically targets non-PFAS leachate constituents of concern, such as ammonia and chemical oxygen demand (COD). The fate of PFAS in existing leachate treatment systems and wastewater treatment systems that manage leachate have been the subject of several studies and have been reviewed previously (Appleman et al., 2014; Lu et al., 2023; Meegoda et al., 2020; Travar et al., 2020; Zhang et al., 2022). To generalize, the treatment of ammonia and COD relies on chemical or biological oxidation, which do not effectively treat PFAS but often have the unintended effect of transforming precursor PFAS to terminal PFAS (US EPA, 2021b). Furthermore, during treatment, PFAS may partition into solids (e.g., biosolids) to a limited extent, which results in additional management challenges (Thompson et al., 2023b). Studies have recommended PFAS removal prior to such treatment (Lott et al., 2023). The targeted treatment of PFAS via removal or destruction in landfill leachate has been the subject of multiple reviews (Bandala et al., 2021; Berg et al., 2022; Lu et al., 2023; Ross et al., 2018; Travar et al., 2020) which have thoroughly discussed the effectiveness of different technologies and which are, here, summarized in Table 3.

PFAS separation technologies typically rely on adsorption over materials, such as activated carbons and ion exchange resins (US EPA, 2022c; Chow et al., 2022; Crone et al., 2019; Appleman et al., 2013), the use of high-pressure membrane separation (US EPA, 2022c; Lipp et al., 2010; Steinle-Darling and Reinhard, 2008), and newer technologies such as ozo- and foam-fractionation with the aim of concentrating the PFAS into a smaller volume of either a solid phase or concentrated liquid residual to either be disposed or destroyed via a subsequent high-energy destructive treatment method (Du et al., 2021; Labiadh et al., 2016). Several novel technologies are being investigated for the destructive treatment of landfill leachate – most require large amounts of energy in the form of chemical reactions or localized high temperatures to break the C—F bond. MSWI for energy recovery is not currently optimized to target PFAS destruction. Additional research is ongoing to define the conditions needed for PFAS destruction in MSWI and other incineration approaches, such as sewage sludge incineration.

Flaring and combustion are common LFG management techniques. Flaring is typically carried out in an open (candle) or enclosed flare. Combustion processes can generate energy on-site (e.g., a combustion engine) or off-site in a gas-fired power generation system. MSW LFG regulations target the destruction of nonmethane organic compounds (NMOCs), not PFAS. Flares generally operate at ~650 °C to 850 °C and temperatures in combustion engines or boiler systems could be lower (Wade, 2022). PFAS separation treatment has not been applied to LFG, however, laboratory-scale thermal PFAS destruction experiments indicate that temperatures higher than 1000 °C are necessary to achieve the mineralization of PFAS (Winchell et al., 2021). MSW LFG flare temperatures and the time that gaseous PFAS are in the presence of high

Table 3
Summary of treatment methods for PFAS in landfill leachate (Bandala et al., 2021; Berg et al., 2022; Lu et al., 2023; Ross et al., 2018; Travar et al., 2020; Wei et al., 2019).

	Technology	Pros	Cons	Matrix	References
Separation technologies	Activated carbon (GAC, PAC)	• High maturity level • Highly effective for long-chain PFAS	•Generates large quantities of spent sorbent that need additional treatment and disposal	GW	Busch et al. (2010); McCleaf et al. (2017), Pan et al. (2016), Ross et al. (2018); Bao et al. (2014), Pan et al. (2016); Malovanyy et al. (2023)
	Ion exchange resins	• High maturity level • Can remove compounds such as GenX	•Needs secondary treatment and disposal	GW, LL	Gao et al. (2015); Dixit et al. (2021); McCleaf et al. (2017); Ross et al. (2018); Boyer et al. (2021); Park et al. (2020); Ellis et al. (2022); Malovanyy et al. (2023)
	Membranes (RO, UF, NF)	• High maturity level and commonly practiced • 2-stage RO most effective on raw leachate	• Membrane fouling • Secondary stream with high PFAS concentrations and volume requires treatment • UF might not be effective	GW, LL	Das and Ronen (2022); Enzminger et al. (1987); Wei et al. (2019); Ross et al. (2018); Boo et al. (2018); Malovanyy et al. (2023)
	Foam/ ozofractionation	• High maturity level and commercially available pilot-scale technology • Potentially low cost	• Pretreatment of leachate might be required • Secondary treatment of concentrated PFAS required	GW, AFFF, LL	Smith et al. (2022); Robey et al. (2020); Malovanyy et al. (2023)
	Incineration	• Highly effective method • Can be used for regeneration of spent materials	•As a standalone method, not practical for large volumes of leachate		
	Chemical Oxidation	•Controllable by varying pH and temperatures	• Uses additional chemicals for treatment • Low effectiveness of removal • Needs to be paired with other methods such as UV for higher effectiveness	LL	Abu Amr et al. (2013); Lin et al. (2012)
	Electrochemical	• 98–99.7% effectiveness demonstrated • Operates at ambient temperatures • No chemicals required • Lower energy consumption compared to incineration	• Expensive electrode materials • Perchlorates could be formed	LL	Labiadh et al. (2016); Du et al. (2021); Gomez-Ruiz et al. (2017); Witt et al. (2020); Krause et al. (2021)
	Photocatalysis	• 94–99% degradation reported • Can also potentially mineralize PFAS	• Slow kinetics • Lab-scale testing only • Difficult to scale for larger volumes		Esfahani et al. (2022)
Destruction technologies	Sonolysis	• Can destroy short-chain and long-chain molecules • Effective for high concentration samples • Can be combined with chemical oxidation to lower costs • Can be used to regenerate GACs	•High capital costs		Moriwaki et al. (2005); Vecitis et al. (2008); Babu et al. (2016)
	Microwaves	• Catalytic microwave treatment could result in ~65–67% effectiveness	•Expensive for large-scale use		Gagliano et al. (2021); Lee et al. (2010b); Liu et al. (2020)
	Subcritical water oxidation	•Effective for short-chain PFAS	• Additional chemicals (e.g., zero-valent iron) needed for higher effectiveness • Slower kinetics		Hori et al. (2006)
	Supercritical water oxidation	• High maturity and close to commercialization • Low residence times required	• Full fluorine balance needed • High-pressure and temperature processes can be energy intensive	GW, AFF, LL	Pinkard et al. (2021); Hori et al. (2006); Krause et al. (2022)
	Wet Air Oxidation	•No demonstrated benefits for PFAS treatment	Converts FTOH precursors to PFCAs	LL	Travar et al. (2020)
	Biological processes	•Limited aerobic and anaerobic degradation of PFOS by bacteria and fungi reported	• Laboratory demonstrations only and thus low technology readiness level • Slow kinetics • Longer-chain PFAS converted to shorter-chain; no mineralization • Unlikely to be effective		Berhanu et al. (2023); Huang and Jaffé (2019)
	Constructed wetlands	•No demonstrated benefits for PFAS treatment	• Does not result in a concentrated PFAS stream that can be adequately managed • Environmental release of PFAS	LL	Yin et al. (2017, 2019); Awad et al. (2022); Lott et al. (2023)

(GAC = granular activated carbon; PAC = powder activated carbon; RO = reverse osmosis; UF = ultrafiltrations; NF = nanofiltration; GW = groundwater; LL = landfill leachate; AFFF = aqueous film-forming foam).

temperatures are too low to completely mineralize PFAS, but may result in the transformation of volatile PFAS into products of incomplete combustion (PICs). Notably, several PICs have been identified as significant greenhouse gases (Ahmed et al., 2020; Hong et al., 2013; Longendyke et al., 2022).

4. Environmental impact of PFAS emissions from landfills

Waste-derived PFAS may be emitted from landfills through multiple pathways, primarily in leachate or LFG effluent. While most RCRA-compliant landfills are operated to minimize environmental impacts, controls have yet to be designed to manage PFAS, and there is a subclass of small landfills in the US that are not required to install bottom liners as they are exempt from RCRA requirements (40 CFR § 258.1(f)(1)).

PFAS may be released into the atmosphere via fugitive gas emissions or gas flares. No data were found on PFAS concentrations in the ambient air surrounding C&D landfills, hazardous waste landfills, or industrial landfills; however, PFAS concentrations in the ambient air close to MSW landfills have been the subject of studies in the US, Germany, and China. Ahrens et al. (2011) reported average total FTOH concentrations of 2.6 and 26 ng m^{-3} at two US MSW landfills, representing 93% and 98% of total gas phase PFAS, with the remaining fraction consisting of perfluoroalkane sulfonamide (FASAs), perfluoroalkane sulfonamido ethanols (FASEs), and PFAAs. Weinberg et al. (2011) reported average total FTOH concentrations at two German landfill sites of 0.086 and 0.271 ng m^{-3}, representing 80% and 92% of total gas phase PFAS. Tian et al. (2018) measured PFAS in air sampled on-site at two landfills as well as downwind. The PFAS profile of the on-site air samples was more evenly split among classes. Total FTOHs were 0.61 and 2.1 ng m^{-3} at the two sites, representing 42% and 76% of $\sum_6$PFAS, with PFAAs representing the bulk of the remaining fraction. PFAS concentrations downwind of the two landfill sites were lower than on-site but elevated relative to control sites, indicating atmospheric transport of PFAS. Lower concentrations downwind may indicate dilution or deposition of volatile PFAS. Neutral PFAS readily transform in the environment – studies have shown the degradation of FTOHs into PFCAs via photooxidation (Esfahani et al., 2022; Martin et al., 2006). Tian et al. (2018) reported elevated neutral and ionizable PFAS in dry deposition samples on-site and downwind of landfills, driven primarily by PFBA and PFOA. Ahmadireskety et al. (2020) reported PFAS concentrations in landfill cover soils of approximately 8 μg kg^{-1}, similarly driven by PFCAs.

Deposition of PFAS from landfill-impacted air may also contribute to PFAS measured in surface water on landfill sites. Chen et al. (2023) reported $\sum_{26}$PFAS concentrations in stormwater at MSW landfill sites averaging 470 ng L^{-1}, significantly lower than leachate concentrations from the same study but significantly higher than groundwater samples, which averaged 140 ng L^{-1} of $\sum_{26}$PFAS. PFAS may be present in both surface and groundwater due to leachate contamination. At the same time, particulate transport from the working face or atmospheric transport and deposition of PFAS are more likely to impact surface water. The MPCA (2010) reported PFAS contamination in groundwater impacted by landfills accepting PFAS-laden industrial waste. Hepburn et al. (2019) measured PFAS and other landfill leachate indicators in groundwater impacted by legacy landfills in Australia, where PFOA represented >10% of total PFAAs, likely associated with legacy landfills.

Using the landfill liner collection efficiency reported by Jain et al. (2023) and overall leachate leakage rate of 1.9% with the leachate generation rate reported in Lang et al. (2017) (61 billion L $year^{-1}$), approximately 1.2 billion L of MSW landfill leachate enter the groundwater directly as a result of liner imperfections every year (14.3 kg of total PFAS using the average $\sum_{19}$PFAS from Lang et al. (2017)). This represents a conservative estimate, as Lang et al. (2017) note that most but not all landfills contributing to the total estimated leachate generation are lined. Although C&D leachate generation rates are not readily available, using leachate generation rates calculated for 17 MSW landfills in six US states, Jain et al. (2023) reported an average collection rate of 6900 L ha^{-1} day^{-1}. Assuming similar leachate generation rates for C&D landfills, this corresponds to approximately 2.5 million L of C&D leachate entering the groundwater per hectare of C&D landfill annually, representing a ∑PFAS mass of 26 g of PFAS per hectare of C&D landfill (see Table S9 in the SI for more calculation information). In 2012, the US EPA inventoried 1504 active C&D landfills (US EPA, 2012).

In the US, most landfill leachate generated from RCRA-permitted landfills is managed off-site (again, many C&D landfills are not required to collect leachate and thus operate without a bottom liner). This represents a significant flux of PFAS leaving the landfill. Multiple studies in the US and Australia have estimated the contribution of PFAS to municipal WWTP from landfill leachate and the environmental impact of PFAS in WWTP effluent. Masoner et al. (2020) estimated the PFAS load in landfill leachates and receiving WWTPs. They reported that landfill leachate while representing, on average, <2% of WWTP influent by volume across three sites, contributed 18% of influent PFAS. Gallen et al. (2017) reported similar contributions of PFAS to WWTPs from landfill leachates. PFAS are not effectively treated with traditional WWTP processes and are released to the environment via WWTP liquid effluent, land-applied biosolids, landfills, and possibly incineration of biosludge (Barisci and Suri, 2021; Coggan et al., 2019; Gallen et al., 2018; Helmer et al., 2022; Tavasoli et al., 2021).

5. Estimate of US MSW landfill PFAS mass balance

Estabrooks and Zemba (2019) evaluated landfill PFAS mass balance at an MSW landfill in Vermont, identifying the PFAS load from targeted waste types suspected to contain PFAS, not including residential MSW, and found that approximately 7% of the PFAS load entering landfills is emitted via leachate annually, and hypothesize the majority of PFAS remain in the waste mass within the landfill. Coffin et al. (2022) propose an estimated extractable ∑PFAS load in MSW entering landfills of 50 ng g^{-1} based on the findings in Liu et al. (2022a). This, combined with US EPA estimation of landfilled MSW in 2018 (the most recent year for which MSW generation data is available for the US), corresponds to 6600 kg of extractable PFAS entering MSW landfills in 2018 with MSW (US EPA, 2020b). Biosludge and biosolids also contribute a significant fraction of PFAS loading in MSW landfills. Using $\sum_{92}$PFAS in treated biosolids reported by Thompson et al. (2023b) and biosolids management statistics reported by NEBRA (2022), the 1.74 million dry metric tons of biosolids landfilled each year contribute an additional estimated 850 kg of PFAS to MSW landfills. Based on our calculations, a conservative estimate of 7480 kg of extractable PFAS entered US MSW landfills in 2018. This estimate does not include PFAS polymers.

As described earlier, PFAS can be emitted from landfills via the gaseous and liquid phase. MSW landfills in the US collect approximately 93.5 million m^3 of gas daily according to the US EPA LMOP database. This translates to nearly 1 kg of neutral PFAS emitted via MSW LFG per day (347 kg annually) based on the concentrations reported by Titaley et al. (2023). The US EPA estimates MSW LFG collection efficiency of approximately 75% (US EPA, 2020a), indicating an additional 31.2 million m^3 of LFG are released via fugitive emissions from MSW landfills annually. Leachate generation in the US, estimated by Lang et al. (2017), is 61.1 billion L $year^{-1}$ which corresponds to 750 kg of PFAS emitted from MSW landfills via leachate annually (using the weighted average ∑PFAS concentration of 12,300 ng L^{-1} calculated in this study). See Fig. 3 for a flowchart representing PFAS sources, controlled emissions, and uncontrolled emissions to the environment corresponding to MSW landfills, and Fig. 4 for a graphical presentation of the fraction of PFAS entering landfills from MSW and biosolids and corresponding emissions; the majority of PFAS entering landfills remain in the waste (84% annually) and significant a mass of PFAS have likely accumulated since PFAS use in consumer products began. Detailed calculations for Fig. 3 are included in the SI ***Table S10***.

One can estimate the total PFAS released via the gaseous phase per ton of MSW based on the potential methane generation capacity (L_o) of

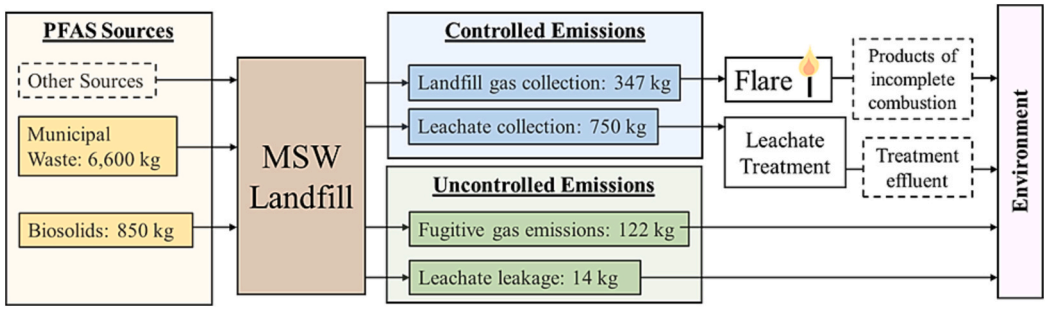


Fig. 3. Flowchart depiction of annual ∑PFAS loading and release at MSW and C&D debris landfills based on current understanding in the literature. Dashed lines represent PFAS streams which have not been quantified to any extent in the literature.

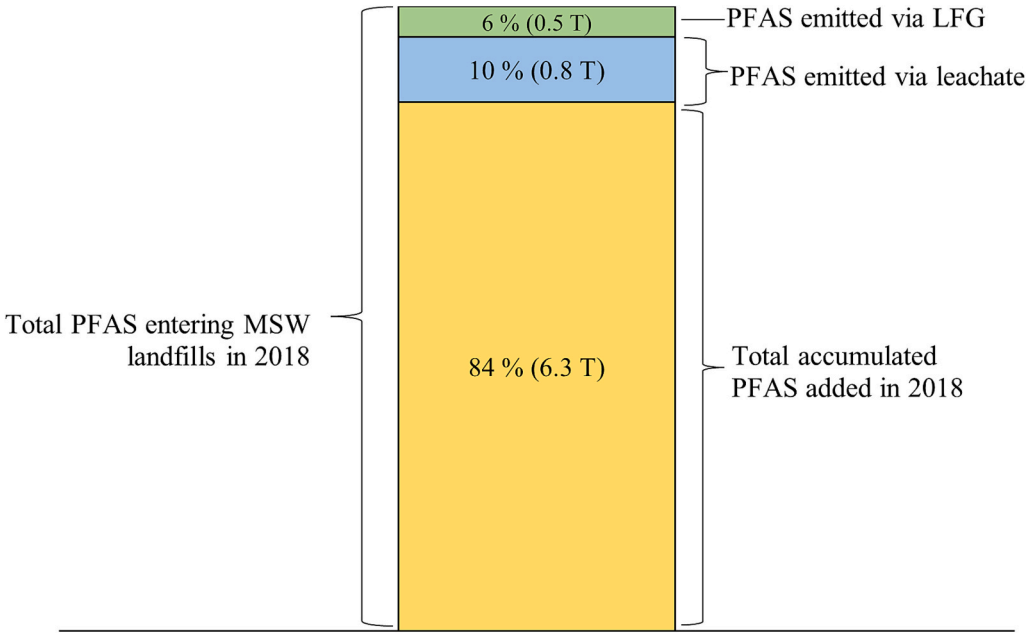


Fig. 4. Estimated PFAS mass balance for US MSW landfills.

MSW. Jain et al. (2021) estimated MSW methane emissions of 68 m^3 of methane per metric ton (Mg) of waste, or approximately 136 m^3 of LFG per Mg of waste; using these values and the Titaley et al. (2023) LFG PFAS concentrations suggest 1.38 mg of PFAS are released, cumulatively, via LFG for every Mg of MSW.

5.1. Limitations

The estimated PFAS mass loading and emissions presented here are based on multiple assumptions and, in some cases, limited data, resulting in significant uncertainty. We have not provided additional data quality assurance in this review process. A small number of studies have explored changes in landfill leachate PFAS profile over time, and no studies have looked for similar relationships in LFG; for this critical review, it was assumed that ∑PFAS reported in leachate and gas are representative of a range of waste ages and stages of decomposition and, overall, are expected to remain consistent over time. Even fewer studies have looked at C&D debris landfills in the US, and those studies are limited to Florida landfills. This critical review of previous analyses provides perspective, not precise values, which should be derived through additional empirical studies.

6. Conclusions and data gaps

The bulk of studies of PFAS in solid waste and landfills focus on MSW landfill leachate, with comparatively fewer studies estimating overall PFAS loading in other types of landfill leachate, in the solid waste itself, or gaseous effluent. Regardless of the type of landfill, in all studies across all locations, PFAS were quantified in all leachate samples. PFAS concentrations in leachates vary across studies, which may be a function of waste type, leachate qualities, climate, and the analytical method.

US MSW and C&D landfill leachates have similar ∑PFAS concentrations. However, C&D leachate contains proportionally more terminal PFAS. This is likely due to the PFAS present and the conditions within each landfill type. Concentrations of the five PFAS which have been the subject of proposed US EPA regulations (i.e., PFBS, PFHxS, PFOS, PFOA, and PFNA) consistently exceed US EPA tapwater RSLs in both MSW and C&D landfill leachates by a factor as high as 20 (PFOA) and in HW landfill leachates by a factor as high as 104 (PFOS); as presented in Table 1 and Fig. 2.

MSWI ash leachates have lower PFAS concentrations than other leachates, however, co-disposal of ash with other wastes results in disproportionately high PFAS concentrations in leachate. To minimize PFAS leaching from MSWI ash landfills, care should be taken to dispose of unburned waste which contains higher concentrations of PFAS separately from MSWI ash. No peer-reiviewed studies have reported

PFAS concentrations in effluent from hazardous waste (Subtitle C) landfills, although hazardous waste management methods are likely to reduce PFAS leaching. Since some hazardous waste landfills likely accept PFAS-laden industrial waste at relatively high levels, it would be helpful to have more information on effluent generated from these facilities.

Traditional leachate treatment methods that use oxidation (e.g., a treatment that targets ammonia, COD) are likely to increase the transformation rate of precursor PFAS to terminal PFAS, such as regulated PFAAs. Treatment that relies on volatilization, such as evaporation, likely contributes significant quantities of PFAS to the atmosphere and surrounding environment, increasing off-site transport. Separating PFAS from leachate prior to additional treatment would avoid these issues. Though there are many aqueous treatment technologies for the targeted removal or destruction of PFAS, few have been tested for effectiveness on landfill leachate. Those tested on leachate and have shown promise include supercritical water oxidation, electrochemical oxidation, reverse osmosis, and foam separation. Assuming treatment efficacy is comparable across PFAAs, reducing PFOA concentration to its limit (e.g., MCL) will reduce all other PFAS to below their respective limits. PFAS treatment of liquid wastes often produces a secondary residual waste requiring additional management.

PFAS are expected to be present in LFG as a product of volatilization and the anaerobic decomposition of biodegradable waste but have been quantified only in MSW LFG. PFAS have not been measured in C&D LFG. However, based on PFAS profiles in C&D landfill leachate, similar PFAS concentrations are likely present in C&D LFG, although LFG generation rates from C&D debris is lower. To reduce gaseous emissions of PFAS, biodegradable waste should be disposed of separately from other PFAS-containing waste. Data do not exist on the effectiveness of PFAS destruction from LFG combustion within flares and internal combustion engines or PFAS removal from LFG to RNG conversion processes. However, the temperatures reached in LFG flares are expected to transform volatile PFAS into terminal PFAS and possible PICs, with minimal mineralization.

Based on our estimate of the PFAS entering and leaving landfills, significant quantities of PFAS are emitted in both LFG and leachate; however, the bulk of PFAS remains within the waste mass on a per-year basis (see Fig. 3). This suggests landfills will be a source of PFAS emissions for the foreseeable future. Studies have demonstrated downgradient impacts on groundwater from landfills. C&D landfills pose the highest risk of environmental contamination since they are not required (at the federal level) to install liners to collect leachate. Even among lined landfills, the average liner collection efficiency is approximately 98%, corresponding to an annual flux of 14.3 kg PFAS entering groundwater via liner imperfections.

Elevated PFAS concentrations were measured in ambient air at landfills across several studies. The highest concentrations were found among FTOHs, which transform into PFAAs in the environment. Atmospheric PFAS may deposit and contribute to soil and surface water concentrations. Even if LFG collection systems were equipped to operate at temperatures and residence times sufficient to destroy PFAS, current MSW LFG collection efficiency is only 75%, and landfills not required to collect LFG will continue to emit PFAS into the atmosphere. The fate of PFAS in LFG that passes through landfill cover soil should be analyzed in future studies.

This review has identified several data gaps for PFAS emissions from US landfills. Data are needed from hazardous waste landfill sites and relevant industrial waste landfills. Furthermore, US C&D landfill leachate data are limited to Florida landfills, and additional efforts should be made to collect information from other states. C&D waste streams may vary due to regional construction requirements. The measurement of PFAS in LFG and other gaseous emissions is an area of emerging study. More research is needed on both controlled and uncontrolled landfill gaseous emissions. A closer evaluation of the fate of PFAS during leachate treatment and LFG management is needed to help decision-makers guide the solid waste community. Geomembrane liners are the most effective tools for the protection from and collection of PFAS-containing liquids, such as landfill leachate. More research is needed to understand long-term interactions between PFAS and liner systems, especially in complex matrices such as landfill leachate. More research is needed to evaluate the long-term implications of PFAS in the landfill environment since the bulk of PFAS remains within the solid waste mass. This review focused on landfilling as a management option for solid waste; evaluation of PFAS fate during other solid waste management processes (e.g., anaerobic digestion, thermal treatment, composting, and recycling) is needed.

CRediT authorship contribution statement

Thabet Tolaymat: conceptualization, approach, methodology, writing-reviewing, editing, funding, **Nicole Robey**: data curation, methodology, calculations, writing-original draft, writing and editing. **Max Krause**: writing and editing, **Judd Larson, Keith Weitz, Sameer Parvathikar**: data curation, original draft preparation, editing, **Lara Phelps, William Linak, Susan Burden, Tom Speth, Jonathan Krug:** writing and editing

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2023.167185.

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Review

Biosolids, an important route for transporting poly- and perfluoroalkyl substances from wastewater treatment plants into the environment: A systematic review

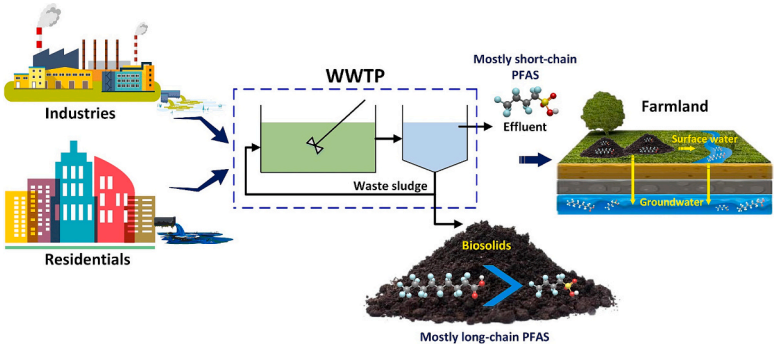
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HIGHLIGHTS

- PFAS presence in wastewater highlights WWTPs as major environmental sources.
- PFAS-laden biosolids raise concerns as WWTPs are inefficient in their removal.
- Transformation mechanisms for PFAS offer insights, requiring further exploration.
- Carbon chain length, sludge protein content, and cations influence the PFAS partitioning.

GRAPHICAL ABSTRACT



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ABSTRACT

The pervasive presence of poly- and perfluoroalkyl substances (PFAS) in diverse products has led to their introduction into wastewater systems, making wastewater treatment plants (WWTPs) significant PFAS contributors to the environment. Despite WWTPs' efforts to mitigate PFAS impact through physicochemical and biological means, concerns persist regarding PFAS retention in generated biosolids. While numerous review studies have explored the fate of these compounds within WWTPs, no study has critically reviewed their presence, transformation mechanisms, and partitioning within the sludge. Therefore, the current study has been specifically designed to investigate these aspects. Studies show variations in PFAS concentrations across WWTPs, highlighting the importance of aqueous-to-solid partitioning, with sludge from PFOS and PFOA-rich wastewater showing higher concentrations. Research suggests biological mechanisms such as cytochrome P450 monooxygenase, transamine metabolism, and beta-oxidation are involved in PFAS biotransformation, though the effects of precursor changes require further study. Carbon chain length significantly affects PFAS partitioning, with

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longer chains leading to greater adsorption in sludge. The wastewater's organic and inorganic content is crucial for PFAS adsorption; for instance, higher sludge protein content and divalent cations like calcium and magnesium promote adsorption, while monovalent cations like sodium impede it. In conclusion, these discoveries shed light on the complex interactions among factors affecting PFAS behavior in biosolids. They underscore the necessity for thorough considerations in managing PFAS presence and its impact on environmental systems.

Abbreviations

Compounds

PFAS	*Per*- and polyfloroalkyl substances
PFOS	Perfluorooctane sulfonic acid
PFOA	Perfluorooctanoic acid
PFAAs	Perfluoroalkyl acids
PFCAs	Perfluoroalkyl carboxylic acids
EtFOSA	N-ethyl perfluorooctane sulfonamide
N-EtFOSAA	2-(N-Ethylperfluorooctanesulfonamidoacetic acid)
N-EtFOSE	N-ethyl perfluorooctanesulfonamidoethanol
FOSAA	Perfluorooctane sulfonamidoacetic acid
MeFOSAA	2-(N-Methyl-perfluorooctane sulfonamidoacetic acid)
PFHpA	Perfluoroheptanoic acid
PFBA	Penta-fluorobenzoic acid
PFDS	Perfluorodecane sulfonate
PFPeA	Perfluoropentanoic acid
PFBS	Perfluorobutane sulfonic acid
PFSA	Perfluoroalkyl sulfonates
PFTrDA	Perfluorotridecanoic acid
POSF	Perfluorooctane sulfonyl fluoride
8:2 FTOH	8-2 fluorotelomer alcohol
FTSs	Fluorotelomer sulfonates
8:2 FTCA	8:2 fluorotelomer carboxylic acid
8:2 FTAL	8:2 fluorotelomer aldehyde
FTUCA	Fluorotelomer unsaturated carboxylic acid
8:2 FTAC	8:2 fluorotelomer acrylate
PFHxA	Perfluorohexanoic acid
PFODA	Perfluorooctadecanoic acid
PFECAs	Perfluoroalkyl ether carboxylic acids
PFHxS	Perfluorohexanesulfonic acid
PFDA	Perfluorodecanoic acid
PFNA	Perfluorononanoic acid
PFUnDA	Perfluoroundecanoic acid
CF_2	Perfluorinated methylene
DiPAPs	Fluorotelomer phosphate diesthers

Sets

CAS	Conventional activated sludge
DOM	Dissolved organic matter
DW	Dry weight
MBR	Membrane bioreactor
BNR	Biological nutrient removal
A_2O	Anaerobic/anoxic/oxic
SRT	Solid retention time
HRT	Hydraulic retention time
NOM	Natural organic matter
SBR	Sequencing batch reactor
WWTPs	Wastewater treatment plants

Parameters

K_d	Distribution coefficient

1. Introduction

Per- and polyfluoroalkyl substances (PFAS) are a category of industrial chemicals that have been extensively manufactured and utilized in various products in recent decades, owing to their distinctive attributes like thermal and chemical durability, as well as their surfactant and repellent qualities arising from their lipophilic and hydrophobic nature (Chiavola et al., 2020; Nguyen et al., 2022). Hence, PFAS find extensive usage across diverse sectors, such as food packaging, surface coating for cooking tools, textile protection, plating industry, photography industry, flame retardants, and insecticides (Mu et al., 2022; Nguyen et al., 2022; Szabo et al., 2018).

PFAS possess robust C—F bonds with significant energy, rendering them resistant to degradation by biotic processes, photolysis, hydrolysis, and metabolic pathways within living organisms (Barisci and Suri, 2021; Dufková et al., 2012). As a result, the extensive utilization of PFAS and their enduring nature has resulted in substantial buildup across various settings like soil, water bodies, sediment, and atmospheric components, as well as within the biological systems of plants, animals, and human beings (Guo et al., 2010; Nguyen et al., 2022).

Over 3000 commercially available PFAS are identified worldwide, but from which only <2 % are registered in various regulations. Among different PFAS, two 8-carbon PFAS, namely perfluorooctanoic acid (PFOA) and perfluorooctane sulfonic acid (PFOS), are widely used and repeatedly detected in numerous environments (Barisci and Suri, 2021) which are associated with reproductive toxicity and cancer (Venkatesan and Halden, 2014). Thus, PFOA, PFOS, and their salts have been identified as emerging long-lasting organic contaminants and are included in Annex B of the Stockholm Convention (Wanninayake, 2021).

Considering that PFAS are used in multiple products such as fire-fighting foams, surfactants, oil and water repellents, and pharmaceuticals, they can finally reach the wastewater systems. The concentration of these compounds has been measured at 0.18 to 66,190 ng/L in the wastewater entering wastewater treatment plants (WWTPs) (Chen et al., 2012; Seo et al., 2019). In WWTPs, PFAS undergo a variety of physicochemical and biological mechanisms that can degrade or separate these compounds from the wastewater. The robustness of the C—F bond within PFAS makes them recalcitrant to biological degradation processes. As a consequence, conventional wastewater treatment processes prove inadequate in effectively eliminating these compounds from wastewater, leading to their discharge into the surrounding environment (Rodríguez-Varela et al., 2021). Moreover, the generation of elevated levels of C $\geq$ 4 perfluoroalkyl acids (PFAAs) through the degradation of precursor compounds such as fluorotelomer sulfonates and fluorotelomer alcohols (FTOHs), under both anaerobic and aerobic conditions (Lenka et al., 2021) further demonstrates the ineffectiveness of conventional WWTPs in removing PFAS (Li et al., 2021). Consequently, WWTPs have been recognized as a significant contributor to the release of PFAS into the environment (Wang et al., 2020b). Hence, comprehending the fate of PFAS in WWTPs and their mass flow to the soil and receiving water bodies through treated effluent and/or biosolids is of crucial significance.

A comprehensive review by Lenka et al. (2021) indicates that PFAS concentrations in effluent wastewater samples from various WWTPs were significantly higher than those in raw wastewater samples (Lenka

et al., 2021). This is attributed to the formation of compounds like PFOS from precursor substances. Additionally, a significant portion of PFAS is not effectively eliminated during wastewater treatment processes, resulting in their discharge into surrounding environments via treated effluent or accumulation in biosolids.

Over the past decade, numerous review articles have emerged, focusing on the fate and elimination of PFAS within WWTPs (Arvaniti and Stasinakis, 2015; Barisci and Suri, 2021; Hamid and Li, 2016; Lenka et al., 2021; Phong Vo et al., 2020), yet there are still several significant challenges and gaps in our understanding that need to be addressed. For instance, Phong Vo et al. (2020) reviewed the recent literature on the origins of PFAS, their presence in the water environment, transformations, fate and removal, and approaches for remediation (Phong Vo et al., 2020). Hamid and Li (2016) conducted a thorough assessment of the contribution of WWTPs to the environmental dissemination of PFAS via aqueous, sludge, and airborne discharges (Hamid and Li, 2016). In a recent review paper, Lenka et al. (2021) investigated the occurrence, transformation, and elimination of PFAS within WWTPs (Lenka et al., 2021).

Upon evaluating the published review studies, it becomes evident that their main emphasis was on the presence and fate of PFAS within wastewater treatment systems, with consideration given to PFAS in sludge and biosolids as part of their investigations. Nevertheless, a notable gap exists in the literature regarding a comprehensive review of recent studies regarding the presence, fate, partitioning, and transformation of PFAS within the sludge line of WWTPs. Additionally, there is no comprehensive review comparing industrial, municipal, and mixed WWTPs (receiving both industrial and municipal wastewater) regarding PFAS occurrence and fate in the sludge. This paper complements prior review studies on the presence and fate of PFAS in sludge and biosolids originating from WWTPs.

Given the escalating global apprehension concerning the existence of PFAS in the environment and recognizing that both the effluent and biosolids originating from WWTPs can serve as pathways for these substances to enter the environment, this review study has been undertaken. Furthermore, with the expanding utilization of biosolids for agricultural purposes worldwide (Gallen et al., 2016; Wang et al., 2013), there is potential for PFAS to enter the food chain and water resources, adding to the urgency of this investigation. The findings of this review study can furnish valuable insights to researchers concerning the presence of PFAS in WWTP sludge, the predominant compounds present, and the influential factors affecting their sorption into the sludge.

This review provides a comprehensive exploration of various aspects related to PFAS in sludge from WWTPs. Firstly, we discuss the global distribution of studies (in Supplementary Materials) to offer an overview of existing research trends. Next, we delve into the impact of diverse wastewater streams on the PFAS content of sludge (Section 2), shedding light on how different influent sources contribute to PFAS presence in this context. Subsequently, we explore the transformation of PFAS in sludge (Section 3) and the partitioning process (Section 4), unraveling the mechanisms behind their behavior in the sludge line. Additionally, we investigate the influence of the carbon chain length on PFAS sorption onto sludge (Section 5), highlighting how molecular properties impact their interactions with this critical component of the treatment process. Moreover, we thoroughly analyze PFAS sorption into sludge across various wastewater matrices, providing valuable insights into the role of different environmental factors (Section 6). Section 8 of this review addresses challenges and future perspectives, offering crucial insights into potential areas of further research and development. Finally, we conclude the review by summarizing key findings and emphasizing the significance of understanding PFAS behavior in sludge for effective environmental management.

It should be noted that details about the search strategy, criteria for including and excluding studies, selection of studies, data extraction, and the characteristics of the reviewed studies can be found in the Supplementary Materials. A total of 594 documents were retrieved from the databases. After excluding duplicates and irrelevant studies, 76 articles that aligned with the objectives of the review were included in this study (Fig. S1). Table 1 presents a summary of these studies.

2. Influence of wastewater streams source and treatment process on PFAS content of sludge

As WWTPs receive diverse influent streams containing various PFAS compounds, their fate and transformation within the treatment processes can lead to alterations in the composition and abundance of PFAS found in sludge. Understanding the effects of various types of influent wastewater on the types of PFAS compounds present in sludge is crucial for assessing potential environmental risks and implementing effective management strategies (Thompson et al., 2023). This investigation has the potential to illuminate the fate of PFAS within wastewater treatment and its implications for both sludge handling and broader environmental safeguarding.

2.1. Effect of diverse wastewater source

Guo et al. (2008) investigated the occurrence of PFOS and PFOA in generated sludge of 16 municipal wastewater treatment systems in China. PFOS and PFOA levels ranged from 278 to 5383 ng/g dw (dry weight). The wide variation of the values reported in the investigated WWTPs is due to the matrix of wastewater reaching the plants. In general, municipal WWTPs typically exhibit lower concentrations of PFOS and PFOA compared to facilities receiving mixed wastewater streams. Because mixed wastewater combines municipal and industrial effluents, and since industrial wastewater—coming from sectors like textile, electroplating, plastic production, cosmetics, etc.—often contains elevated levels of PFAS compounds, the mixture of these sources results in higher concentrations of PFAS compounds in mixed wastewaters (Guo et al., 2010; Alder and van der Voet, 2015; Becker et al., 2008; Pan et al., 2016).

Guo et al. (2010) conducted a study in Korea exploring municipal, livestock, and industrial wastewater sources. While industrial WWTPs exhibited elevated PFAS concentrations, they were not identified as the primary source of PFAS pollution in the Korean water environment due to their relatively minor release quantities. Conversely, WWTPs in urban centers discharged higher levels of PFAS, indicating that household sewage constitutes a significant contributor to environmental contamination (Guo et al., 2010). In another study, Yu et al. (2009) examined the presence of PFOA and PFOS in sludge samples. All samples, except one from WWTP A (receiving domestic wastewater), showed detectable concentrations of PFOA and PFOS. In WWTP A, PFOS levels ranged from 13.1 to 46.0 ng/g dw, whereas WWTP B (receiving a combination of 60 % industrial wastewater and 40 % domestic wastewater) had significantly higher levels ranging from 145.1 to 702.2 ng/g dw. WWTP A showed lower levels of PFOA concentration (<5.0–44.2 ng/g dw), while WWTP B had higher concentrations (18.0–69 ng/g dw).

Comparatively, there are also some reported values with similar concentration ranges for PFOS (Alder and van der Voet, 2015; Fredriksson et al., 2022; Ulrich et al., 2016) and PFOA (Dauchy et al., 2017; Kim et al., 2022) in sludge samples. However, based on the matrix of the raw wastewater, various values have also been documented in the literature (Table 1). Accordingly, types and also concentrations of PFASs vary based on the source of raw wastewater. Briefly saying, industrial wastewater usually contain diverse and also higher concentrations of the PFAS compounds. However, depending on the amount of generated wastewater, the main contributor to environmental pollution by PFAS compound would vary.

2.2. Effects of diverse wastewater treatment processes

Yu et al. (2009) pointed out that increased levels of PFOS and PFOA in wastewater led to their elevated concentrations in sludge, primarily

Table 1
Summary of the reviewed studies.

Location	Date	NO. of WWTPs/ STPs[a]	Process type[b]	Wastewater source	Target compounds	Influent Con. ng/L	Effluent Con. ng/L	Dominant PFAS in raw influent	Sludge treatment method[c]	Treated sludge Con. μg/kg	Dominant PFAS in sludge	Sludge production (ton/y)	Total mass flow by sludge (kg/y)	Ref.
Greece	2012	1	AS	D-WWTP	$\sum_{18}$PFAS	2.6–113	0.5–33	PFUdA	DW	1.2–1042	PFUdA			(Arvaniti et al., 2014)
Switzerland	2011	34		M-WWTP	$\sum_{7}$ PFCAs/				AND	0.2–2440	PFOS	28–15,000	0.00001–1.56	(Alder and van der
		9		D-WWTP	$\sum_{3}$ PFSAs					0.3–90		94–838	0.00001–0.04	Voet, 2015)
		2		I-WWTP						0.1–18		910–6000	0.0008–0.11	
Canada	2002	7		D-WWTP	(C8-C11) PFCAs/PFOS				TH·D	0.032–460	8:2/10:2 diPAP, 8:2 diPAP			(D'eon et al., 2009)
Germany	2005	1	AS	M-WWTP	PFOA,/PFOS	12–85	48–250	PFOS	DW	11–120	PFOS		0.84	(Becker et al., 2008)
China	2004	16		D-WWTP	PFOA/PFOS				TH·D	278–5383	PFOS, PFOA			(Guo et al., 2008)
China	2008	5		M-WWTP	$\sum_{15}$PFASs					422–755	PFOS, PFOA,			(Li et al., 2010)
		2		I-WWTP						413–415	PFPrA			
		1		D-WWTP						536				
China	2008	2	AS	D-WWTP	$\sum_{19}$PFAS	1–49.9	0.7–28.8	PFOS	DW	0.2–157.9	PFOS			(Ma and Shih, 2010)
		1	CH.T							0.2–69.4				
USA	2004	1	AS+TF	D-WWTP	$\sum_{27}$PFAS	<2.6–33	0.6–34	PFOS, PFHxA, PFOA	AND	3.6–160	PFOS, PFDS			(Schultz et al., 2006)
USA	2004	1	AS	D-WWTP	$\sum_{8}$PFAS		<2.5–202	PFOA	TH·D	<10–89	PFOA,			(Sinclair and Kannan,
		1		M-WWTP			<2.5–398			<10–241	PFDA, PFUnDA			2006)
Switzerland	1993–2002	20		D-WWTP	$\sum_{12}$ FCAs/ $\sum_{6}$ PFSAs				AND	26–637	PFOS	30–8990	0.0008–0.84	(Sun et al., 2011)
Singapore	2006–07	2	AS	M-WWTP	PFOA/PFOS	7.9–638	7.3–1057		AND	17.4–702	PFOS		0.11–4.4	(Yu et al., 2009)
		1	MBR			7.3–20.5	9.3–93.8						0.021–0.038	
		1	LTM			7.3–20.5	5.3–21.8						0.06–0.14	
Republic of Korea	2019	2		D-WWTP	$\sum_{6}$ neutral PFASs /$\sum_{13}$ ionic PFASs	0.18–873	0.41–340.5	PFBS, PFHxA, PFOA		175–179	PFUnDA	485.5–1350.5	0.087–0.24	(Seo et al., 2019)
		1		M-WWTP		0.67–44	0.09–23.5	PFOS, PFHxA		268		18,009	4.83	
Greece	2010–11	1	AS	D-WWTP	$\sum_{18}$PFAS					1.7–398	PFOS, PFOA	43,800	0.035	(Stasinakis et al., 2013)
Canada	2015–16	1		M-STP	$\sum_{16}$PFAS				PLT	208–280	PFDA, PFOS	15,308	3.18–4.3	(Lakshminarasimman
		1							ALK.S	243		8377	2.03	et al., 2021)
		2							AD	2–15		207–215	0.0004–0.008	
		5							AND	2–423		277–18,622	0.0006–7.5	
Spain	2006	20		D-WWTP	$\sum_{13}$PFAS					<0.01–287	PFOS, PFDA			(Navarro et al., 2011)
Denmark	2007	6		D-WWTP	$\sum_{7}$PFAS	<0.2–32.8	<0.2–24.4	PFOA, PFHxS		0.4–74.1	PFOS, PFOA			(Bossi et al., 2008)
Australia	2017	15	AS	M-WWTP	$\sum_{21}$PFAS	9.27–412	34.1–517	PFHxS, PFBA, PFOA		53.1–126	PFOS, PFDA,			(Coggan et al., 2019)
		4	LAG							2.01–19	PFDoA, PFOA			
France	2013–14	1		M-WWTP	PFOA,/PFOS				INC	27–239	PFOS	21,000 (Before INC)	0.57–5.02	(Mailler et al., 2017)
		1							AND	10–228		55,000	0.55–12.54	
		1							TH·D	8–1742		8000	0.064–13.94	
Sweden	2005/2007	7		D-WWTP	$\sum_{16}$PFAS		17–34		TH·D	148–372	10:2 diPAPm, EtFOSAA			(Kärrman et al., 2021)
New Zealand	2020	1	AS	D-WWTP	$\sum_{38}$PFAS	0.2–12.5	0.1–10.1		DW	2.25–7.1	PFOS			(Lenka et al., 2022)

(*continued on next page*)

Table 1 (*continued*)

Location	Date	NO. of WWTPs/ STPs[a]	Process type[b]	Wastewater source	Target compounds	Influent Con. ng/L	Effluent Con. ng/L	Dominant PFAS in raw influent	Sludge treatment method[c]	Treated sludge Con. μg/kg	Dominant PFAS in sludge	Sludge production (ton/y)	Total mass flow by sludge (kg/y)	Ref.
USA	2019	1	AS	D-WWTP	$\sum_{30}$PFAS	117	167	PFOA, PFOS	INC	31.3	PFOS, PFHxA			(Seay et al., 2023)
USA	2007	2	AS	D-WWTP	$\sum_{8}$PFAS	<0.5–184	<0.5–183	PFOA	DW	<2.5–219	PFOA, PFOS	984.4	0.014–4.37	(Loganathan et al., 2007)
USA	2007	5		D-WWTP	$\sum_{10}$PFAS					<1.8–418	PFOS			(Yoo et al., 2009)
USA	2015	1		D-WWTP	$\sum_{12}$PFAS					<0.05–12	PFOS			(Johnson, 2022)
Germany	2008–13	1165		M-WWTP	$\sum_{11}$PFAS					49–7600	PFOS		2.2–17	(Ulrich et al., 2016)
China	2021	44		M-WWTP	$\sum_{13}$PFAS	59.8	11.4	PFBA, PFPeA	DW	0.61–76.1	PFOS	73–197,100	0.011–4.7	(Chen et al., 2022b)
Australia	2014	6	AS	M-WWTP	$\sum_{9}$PFAS	9.2–35	39–99	PFOS,	AND/AD	14–150	PFOS, PFOA,	2234–22,995	0.002–0.621	(Gallen et al., 2018)
		2	OD	M-WWTP		3.2–35	57–120	PFHxA,	EA/AS	46–140	PFBA	1200–4508	0.002–0.46	
		1		D-WWTP		0.98	57	PFHpA,	BR	65		1091	0.002–0.41	
		1	TF	M-WWTP		40	32	PFHxS,	AD	6.3		1007	0.001–0.003	
		1	CH.T	M-WWTP		55	21	PFOA	SP	5		3102.5	0.001–0.0015	
South Korea	2010–12	21		M-WWTP	$\sum_{13}$PFAS		33			49	PFOS, PFOA			(Kwon et al., 2017)
		60		I-WWTP			82–215			10–97				
Sweden	2018	11		M-WWTP	$\sum_{10}$PFAS	<0.4–<20	<0.4–<17	PFOA	DW	<0.22–270	PFHpA,			(Golovko et al., 2021)
		2		D-WWTP		<0.4–<17	<0.4–<17			0.54–76	PFDoDA,			
		1		I-WWTP		<0.4–<17	<0.4–<19			<0.22–21	PFNA			
China	2013–21	4	MBBR, MBR	D-WWTP	$\sum_{12}$FFCAs	30–117	25–120	PFOA;		22–37	PFOA			(Ma et al., 2022)
		1	OD	M-WWTP		100	80	PFPeA, PFBA		42		43,800	1.84	
China	2012	2	AS	D-WWTP	$\sum_{12}$PFAS	<0.25–5.91	<0.25–20.8	6:2/8:2		<0.066–130	6:2/8:2			(Loi et al., 2013)
		1	CH.T + BIO. T			<0.25–6.96	<0.25–4.83	diPAP, PFOA, PFOS		<0.066–60.7	diPAP, 8:2 diPAP, 6:2 diPAP, PFOS			
South Korea	2010	5	AS A^2/O	D-WWTP	$\sum_{15}$PFAS	110 ± 130	96 ± 120	PFOA, PFHxA		98 ± 86	PFDA, PFOS, PFOA		0.4–7.9	(Kim et al., 2012)
		5	BF AS A^2/O	I-WWTP		610 ± 420	620 ± 320	PFOA, PFHxA, PFHxS, PFOS		470 ± 450			0.09 = 7.6	
		5	AS A^2/O	M-WWTP		1100 ± 1500	2100 ± 2600	PFOA, PFHxA, PFHpA, PFOS		510 ± 600			0.11–40.8	
China	2014–15	3	ModifiedA^2/O	M-WWTP	$\sum_{18}$PFAS	232	234	PFBS, PFOA, PFOS	DW	0.09–37.5	PFOS	10,950	0.134	(Pan et al., 2016)
			A^2/O + MBR	D-WWTP		19.6	15.5			0.14–29.8		13,140	0.135	
			Unitank	D-WWTP		29.5	26.3			0.15–18		17,155	0.131	
China	2012	1		I-WWTP	$\sum_{11}$PFAS		<0.20–12,400			3.33–324	PFOA, PFPeA			(Shan et al., 2014)
China	2009	5	AS	M-WWTP	$\sum_{11}$PFAS	228–412				0.53–169	PFOS, PFOA			(Sun et al., 2012)
		1		D-WWTP		200				0.75–68.2				
Republic of Korea	2008	3	AS	D-WWTP	$\sum_{10}$PFAS	2.3–10.8	0.5–23.7	PFOA,	DW	2.9–21.9	PFOS	110,814–202,576	2.05–2.3	(Guo et al., 2010)
		1		I-WWTP		2.7–615	0.6–591	PFHxA,		4.8–24.7		NA		
		2	A^2/O	D-WWTP		0.6–20.8	0.5–48.3	PFHpA,		3.8–16.2		51,465–219,541	1.95–3.99	
		1	CNR	D-WWTP		0.7–8.5	0.6–6.7	PFOS		2.2–6.2		NA-1636	NA-0.014	
		1	BNR	D-WWTP		1–6	0.7–8.3			3.7–54.1		112,685	6.68	
		1	NPR	D-WWTP		0.6–11.5	0.9–15.2			6.4–12.2		NA-21059	NA-0.4	
		1	Bio-SAC	D-WWTP		0.8–7.3	0.5–12.1			<LOQ-23.5		NA		
		1	SBR	D-WWTP		0.7–12	0.7–6.3			<LOQ-9.9		NA		

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Table 1 (*continued*)

Location	Date	NO. of WWTPs/ STPs[a]	Process type[b]	Wastewater source	Target compounds	Influent Con. ng/L	Effluent Con. ng/L	Dominant PFAS in raw influent	Sludge treatment method[c]	Treated sludge Con. μg/kg	Dominant PFAS in sludge	Sludge production (ton/y)	Total mass flow by sludge (kg/y)	Ref.
		1	DNR	M-WWTP		0.8–37.4	0.6–49.2			<LOQ-13.7		28,387	1.19	
		1	MLE	M-WWTP		2.3–23	0.6–7.4			<LOQ-13.2		16,356	0.44	
USA	2005–13	1	AS	D-WWTP	$\sum_{12}$PFAS				DW	<LOD-218	PFNA, PFOA, PFOS			(Armstrong et al., 2016)
Nordic Countries	2017	4		M-WWTP	$\sum_{73}$PFAS		324–1460			39–210	L-PFOS,			(Aro et al., 2021)
		6		D-WWTP			183–1450			<LOD-170	PFDA, 5:3 FTCA			
Canada	2012–17	9	AS	M-WWTP	$\sum_{22}$PFAS				AND, AD, DW	13–58.4	PFOS, PFDA, PFOA			(Letcher et al., 2020)
		5	BNR						DW, AND,	4.93–67.9				
		3	PT						ALK.S	6.77–13.6				
		2	EAAS						AD	73.4–92.6				
		1	BAF						DW	12.1				
Czech Republic	2018–19	43		D-WWTP	$\sum_{32}$PFAS					5.6–963.2	PFOS			(Semerád et al., 2020)
Sweden	2004–17	2	CH.T + BIO.	M-WWTP	$\sum_{81}$PFAS				AND	49.7–1124	PFOS, PFDA,	2300–14,400	0.114–11.7	(Fredriksson et al., 2022)
		2	T	D-WWTP					AND, AD	98.3–149	PFOA	110–2400	0.011–0.36	
China	2010	1	SBR	M-WWTP	$\sum_{14}$PFAS	292	1070	PFPeA, PFOA	DW	75	PFOS, PFOA			(Zhang et al., 2015)
Nigeria	2012	9	OP	D-WWTP	$\sum_{10}$PFAS					>LQ-0.24	PFOS			(Sindiku et al., 2013)
			AS							>LQ-0.276				
			LAG							>LQ-0.138				
			LAG	I-WWTP						>LQ-0.327				
			AS							>LQ-0.596				
Finland	2009–10	1	BNR	M-WWTP	$\sum_{4}$PFAS		<0.5–640				PFOS, PFDA		PFOS 0.8–16	(Perkola and Sainio, 2013)
Australia	2018	19	AS	M-WWTP	$\sum_{44}$PFAS				DW	<LOD-303	PFOS, PFBS	N/A-30000	PFOS N/A-1.32	(Moodie et al., 2021)
			LAG	M-WWTP						LOD-57		N/A-41000	PFOS N/A-1.23	
France	2013	1	AS	I-WWTP	$\sum_{51}$PFAS	0.12–45 × 10^3	0.11–7116	PFHxA, PFBA, PFPeA	DW	0.07–72.4	PFOA			(Dauchy et al., 2017)
USA	2018–20	10		M-WWTP	$\sum_{28}$PFAS	17–314	81–1357			18–7017	PFOS			(Helmer et al., 2022)
Thailand	2009	1	AS	I-WWTP	$\sum_{10}$PFAS	847.1	661.8	PFOS, PFOA	DW	1534.5	PFOS, PENA, PFUnA,		PFOS 0.537	(Kunacheva et al., 2011)
		1	SBR	I-WWTP		673.3	1143.4	PFOS, PFNA	SD	1200.4	PFOA		PFOS 1.21	
China	2017–18	2		M-WWTP	$\sum_{16}$PFAS					478–1380	PFOS, PFOA			(Na et al., 2020)
		1		D-WWTP						2.6–9.95				
China	201–17	2	OD	D-WWTP	$\sum_{17}$PFAS	31.8–42.1	24.1–37	PFBA, PFOA		4.89–11.27	PFOA, PFOS,	12,800–14,600	0.03–0.06	(Zhou et al., 2019)
		1	A^2/O	M-WWTP		52.4	55.7			5.23	PFDA	54,800	0.11	
		1	A^2/O	DWWTP		55.1	50.1			5.89		36,000	0.08	
		2	AS	D-WWTP		43.5–53.4	24.6–39.7			2.89–5.05		2920–5480	<0.01–0.01	
Australia	2015	4		M-WWTP	$\sum_{14}$PFAS				AS	0.54–370	PFOS	423–4508	0.0002–0.37	(Gallen et al., 2016)
		1		D-WWTP						0.72–26		1091	0.0008–0.022	
		4		M-WWTP					AD	0.47–92		964–2327	0.0004–0.21	
		1		D-WWTP						0.34–53		475	0.0002–0.025	
		2		M-WWTP					AND	0.47–110		9198–22,995	0.011–2.53	
		1		D-WWTP						0.35–11		7884	0.003–0.086	
		1		M-WWTP					TH·D	0.52–94		4198	0.0013–0.4	
		1		M-WWTP					CH.T	0.22–20		3103	0.0007–0.062	
		1		M-WWTP					INC	1.7–25		NA	NA	

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Table 1 (*continued*)

Location	Date	NO. of WWTPs/ STPs[a]	Process type[b]	Wastewater source	Target compounds	Influent Con. ng/L	Effluent Con. ng/L	Dominant PFAS in raw influent	Sludge treatment method[c]	Treated sludge Con. μg/kg	Dominant PFAS in sludge	Sludge production (ton/y)	Total mass flow by sludge (kg/y)	Ref.
Sweden	2013	3		D-WWTP	$\sum_{14}$PFAS	0.22–6.9	0.056–7.9	PFHxA, PFOS, PFOA	DW	0.15–8.8	PFOS, PFOA			(Filipovic and Berger, 2015)
USA	2020–21	1		D-WWTP	$\sum_{42}$PFAS	11.1–66.1	19–76.9	PFPrA,		7.3–18	PFOS			(Kim et al., 2022)
		1				10.9–47.4	11.2–46.9	PFPeA, PFHxA, PFOA		3.4–12.5				
China	2011	3	AS	D-WWTP	PFOA/PFOS	0.33–176	1–82.6	PFOA			PFOS	4380–19,527.5	0.004–0.4	(Chen et al., 2012)
		1	AS+BF	D-WWTP		5–43.75	2.5–3.75					NA		
		1	Biofilm	D-WWTP		2.5–10	2.5–10					7300	0.007	
		4	A^2O	D-WWTP		1.81–112	1.81–531					8760–100,740	0.015–0.98	
		2		M-WWTP		2.5–66,189	2.5–164,811					8212.5–9198	0.022–1.46	
		1	CF	D-WWTP		23.59–191.2	15.41–118.8					93,075	0.365–1.02	
Canada	2009–10	1	LAG	D-WWTP	$\sum_{13}$PFAS	21–14	21.02–20.2	PFOA,	–	<1.13–17.1	PFOS, PFDA	2000	<0.0023–0.034	(Guerra et al., 2014)
		4	CH.T			<1–23.8	<1.01–21.1	PFNA,	DW	<0.88–33.7		570–12,000	<0.0016–0.115	
		6	AS			<0.99–905	<0.97–1237	PFOS,	AND + DW	0.64–3040		51–15,000	<0.0009–0.28	
		1	TF			<1.01–42.2	<0.99–39.9	PFHpA,	AND + DW	<1–13.1		12,000	<0.012–0.16	
		1	BF			<1.06–9.49	<1.01–8.01	PFHxA	AND + DW	<1.25–24.4		2300	<0.003–0.056	
		2	BNR			<1.01–4.45	<1.01–12.4		AND + DW	<1.51–55.3		2600–15,000	<0.004–0.83	
Sweden	2012–2015	2		M-WWTP	$\sum_{54}$PFAS	1.4–23	0.1–33	PFPeA,	AND	<0.02–16.8	PFBA, 6:2 FTSA	14,400	0.013–0.25	(Eriksson et al., 2017)
						0.3-20	2.1–66	PFHxA,		<0.02–68		2400	0.00007–0.19	
		1		D-WWTP		0.5–58	0.1–22	PFOA		<0.02–44.6		2300	0.0018–0.04	
Spain	2010–2011	15		D-WWTP	$\sum_{21}$PFAS	615–2157	195–567	PFBA,		503–920	L-PFOS,			(Campo et al., 2014)
		1		M-WWTP				PFDA, PFHxA, PFHpA, PFOA			PFBA			
China	2010	9		D-WWTP	$\sum_{14}$PFAS					126–571	PFOA	322,000	146	(Yan et al., 2012)
		16		M-WWTP						178–809				
USA	1998–2004	8		M-WWTP	$\sum_{12}$PFAS				AND	73–3390	PFOS, N-EtFOSAA			(Higgins et al., 2005)
China	2010–2011	56		M-WWTP	$\sum_{10}$PFAS					ND-241	PFOS, 6:2 Cl-PFAES			(Ruan et al., 2015)
USA	2001	94		D-WWTP	$\sum_{13}$PFAS				AND/AD	2–403	PFOS		10–2575	(Venkatesan and Halden, 2013)
China	2009	12	A^2O	D-WWTP	$\sum_{15}$PFAS	0.05–28.5	0.05–30.6	PFOA,		9.1–66.7	PFOA, PFOS			(Zhang et al., 2013b)
		5	OD	D-WWTP		0.05–45.7	0.05–34.4	PFPrA,		12.6–27.3				
		3	AS	D-WWTP		0.05–33	0.05–38.2	PFBA,		6.8–9.8				
		1		M-WWTP		0.05–90.6	0.05–106.6	PFHxA		9.9				
		1	AO	M-WWTP		0.05–32.1	0.05–67.3			13.2				
		1	MBR	D-WWTP		0.05–10	0.05–6.4			14.7				
		1	MBBR	D-WWTP		0.05–21.8	0.05–31.7			32.2				
		1	SBR	D-WWTP		0.05–6.3	0.05–8.5			14				
		1	LINPOR	D-WWTP		0.05–5.3	0.05–4.5			11.4				
		1	C.Ox	D-WWTP		0.05–7.1	0.05–5.3			81				
		1	AS	D-WWTP		0.05–67	0.05–68			33.6				
Greece	2009–11	1	AS	M-WWTP	$\sum_{18}$PFAS	2.4–106.1	1–209.4	PFPeA,	TH.D	1.8–45.2	PFPeA,	65,700	0.12–2.97	(Arvaniti et al., 2012)
		1		D-WWTP		1–453	1.1–160.3	PFOA, PFNA, PFOS	DW	0.58–11.4	PFOA, PFOS	401.5	0.0002–0.0046	
Spain	2012	7		D-WWTP	$\sum_{7}$PFAS				DW	<0.30–3.03	PFOS			(Martínez-Moral and Tena, 2013)
China	2014–16	2	A^2O	D-WWTP	$\sum_{6}$FTOH, $\sum_{10}$PFAA	<0.06–15.1	<0.03–17.9	8:2 FTOH	DW		8:2 FTOH, 6:2 FTOH	17,265–32,850	FTOH 0.29–1.35	(Chen et al., 2017)

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Table 1 (*continued*)

Location	Date	NO. of WWTPs/ STPs[a]	Process type[b]	Wastewater source	Target compounds	Influent Con. ng/L	Effluent Con. ng/L	Dominant PFAS in raw influent	Sludge treatment method[c]	Treated sludge Con. μg/kg	Dominant PFAS in sludge	Sludge production (ton/y)	Total mass flow by sludge (kg/y)	Ref.
		5		M-WWTP								13,140–87,600	FTOH 0.44–13.3	
		4	OD	M-WWTP								NA-3285	FTOH NA-0.15	
		1	AS	D-WWTP								2555	FTOH 0.23	
		1		M-WWTP								1898	FTOH 0.365	
USA	2021	2		D-STP	$\sum_{92}$PFAS				AC	78.8–178	6:2, 6:2/8:2, 8:2 diPAP	2700–4000	0.21–0.71	(Thompson et al., 2023)
		2							TH.D	451–501		2700–20,000	1.35–9.01	
		2							LT	112–385		4500–8200	0.92–1.73	
		2							AND	388–584		6300–11,000	3.68–4.27	
USA	2022	38		M-WWTP	$\sum_{40}$PFAS	98 ± 28	80 ± 24	4–6 PFCA	DW	160 ± 46	5:3 FTCA, 7:3 FTCA			(Schaefer et al., 2023)
China	2015–16	54		M-WWTP	$\sum_{155}$PFAS	0.001–85.7	0.002–848	PFBA, PFBS, PFOA		0.002–44.5	PFOS, 6:2 FTAB		0.02–14.6	(Jiang et al., 2023)
China	2020–21	2	AO	I-WWTP	$\sum_{34}$PFAS	86–1600		PFOA, PFBS	DW	0.75–55	PFOS			(Liu et al., 2023)
Canada	2018	1	AS	D-WWTP	$\sum_{12}$PFAS					PFOS 224–2230	PFOS			(Fournie et al., 2023)
Australia	2021–22	2	BNR	M-WWTP	$\sum_{23}$PFAS				AND	0.2–8.16	PFOS, PFOA	2300–6070	0.0002–0.06	(Kumar et al., 2023)
		1	BNR	D-WWTP					AD	6.2–29.3		9670	0.06–0.3	
		2	TF + MB	M-WWTP					AND	5.9–13		349–738	0.002–0.006	
		1	TF	M-WWTP						0.2–4.6		507	0.0001–0.0023	
		1		D-WWTP						0.1–11		6470	0.0006–0.07	
USA	2020	1		M-WWTP	$\sum_{18}$PFAS				DW	87.6	PFOS			(Pepper et al., 2021)
Netherland	2009	3		D-WWTP	$\sum_{4}$PFAS					35–48	PFOS			(Esparza et al., 2011)
Spain	2010	1		D-WWTP	$\sum_{18}$PFAS					0.3–121.1	PFOS			(Llorca et al., 2011)

Note: D-WWTP (domestic + hospital/domestic + tourist characteristics); I-WWTP (only industrial wastewater); M-WWTP (domestic + industrial/domestic + agricultural); M-STP (receives domestic and industrial WWTP sludge); D-STP (receives only domestic WWTP sludge).

[a] WWTPs (wastewater treatment plants); STPs (sludge treatment plants).

[b] AS (activated sludge); CH.T (chemical treatment); TF (trickling filter); MBR (membrane bioreactor); LTM (liquid treatment module); LAG (lagoon); OD (oxidation ditch); MBBR (moving bed biofilm reactor); BIO.T (biological treatment); A^2O (anaerobic-anoxic-oxic); BF (biofiltration); CNR (cilium nutrient removal); BNR (biological nutrient removal); NPR (nitrogen & phosphorus removal); SBR (sequencing batch reactor); DNR (Daewoo nutrient removal); MLE (modified Ludzack-Ettinger); PT (primary treatment); EAAS (extended aeration activated sludge); BAF (biological aerated filter); OP (oxidation pond); CF (chemical flocculation); AO (anaerobic-aerobic); COx (contact oxidation).

[c] DW (dewatering); AND (anaerobic digestion); TH·D (Thermal drying); PLT (Pelletization); ALk.S (alkaline stabilization); AD (aerobic digestion); INC (incineration); EA (extended aeration); BR (bioreactor); SP (stockpiling); SD (sand drying); AS (activated sludge); AC (aerobic composting); LT (lime treatment).

due to partitioning between the aqueous and solid phases. While PFOA dominated the aqueous samples of WWTP A, PFOS prevailed in the sludge samples of both WWTPs, possibly due to its superior partitioning coefficient relative to PFOA. Concerning the PFOS and PFOA levels in activated sludge samples, noteworthy distinctions were not apparent among samples gathered from distinct processes of both membrane bioreactor (MBR) and conventional activated sludge (CAS). This suggests that the sludge concentration of these compounds is primarily influenced by the aqueous concentration rather than the sludge characteristics; furthermore, the PFAS content within the biosolids may be impacted by the sludge retention time (SRT) (Yu et al., 2009).

There are limited studies comparing the impact of different wastewater treatment procedures on the concentration of these compounds in sludge. However, some studies report that the concentration of PFAS in the aqueous phase, rather than the type of treatment process, determines its concentration in sludge. Guo et al. (2010) investigated the occurrence of 10 PFASs within 15 municipal, four livestock, and three industrial WWTPs situated in Korea. There were various treatment processes including CAS process, A_2O (anaerobic/anoxic/oxic process), BNR (biological nutrient removal), and SBR (sequencing batch reactor). The results revealed no notable difference in the PFAS levels within the sludge, indicating that the selected wastewater treatment process does not significantly affect the PFAS content of the sludge (Table 1). Pan et al. (2016) also reported similar results. They investigated the concentration of PFAS in sludge from modified A_2O and A_2O + MBR processes treating mixed and domestic raw wastewater. The findings revealed that mixed wastewater entering the A_2O and A_2O + MBR processes resulted in the accumulation of PFAS in sludge, with concentrations ranging from 0.09 to 37.5 μg/kg and 0.14 to 30 μg/kg, respectively. According to the results, there was no significant difference between these two reported values. The slight difference could be attributed to the matrix of wastewater entering A_2O + MBR process, which was domestic.

In another investigation across 20 WWTPs in Canada, biosolid samples underwent analysis from facilities employing diverse treatment methods, detailed in Table 1. Results revealed PFAS concentrations ranging from 4.93 μg/kg to 92.6 μg/kg. Statistical analysis showed no significant correlations among different WWTP processes ($p > 0.05$) (Letcher et al., 2020). Conversely, Zhang et al. (2013b) examined PFAS presence in 28 WWTPs in China. They employed cluster analysis to categorize data and found that treatment processes significantly influenced PFAS profiles in both effluent and biosolids. Discrepancies between these studies may stem from various confounding factors affecting PFAS accumulation in sludge based on the type of wastewater treatment process. These factors include but are not limited to, the types of raw wastewater (Zhang et al., 2013b), the nature of PFASs (Gallen et al., 2016), PFAS precursors (Yu et al., 2009), and the SRT values of biological processes (Yu et al., 2009).

Overall, the reviewed studies showed a wide variation in PFAS concentrations, ranging from municipal WWTPs with predominantly lower PFOS and PFOA values to those receiving mixed wastewater with significantly higher concentrations. The partitioning between the liquid and solid phases has emerged as a crucial factor impacting PFAS levels in sludge, with elevated amounts detected in sludge from WWTPs where wastewater contains higher concentrations of PFOS and PFOA. Interestingly, the type of wastewater treatment process does not seem to significantly alter the PFAS content in the sludge, as evidenced by similar concentrations across different treatment methods. However, it is recommended that further research be conducted considering potential confounding variables to better understand the impact of wastewater treatment processes on PFAS content in sludge.

3. PFAS transformation in sludge

Polyfluorinated compounds have the potential to undergo transformations during treatment processes, yielding other measurable PFAS that possess perfluorinated groups of similar size. During sludge digestion processes, the changes in PFAS compounds were also linked to the transformation of the precursors existing in the wastewater (Seay et al., 2023). Particularly, there is proof indicating that 2-(N-ethyl-perfluorooctanesulfonamido) ethanol (N-EtFOSE alcohol) and 2-(N-ethyl perfluorooctane sulfonamido) acetic acid (N-EtFOSAA) undergo biotransformation into PFOS during activated sludge treatment (Sinclair and Kannan, 2006). It is also observed that the prevalence of even chain-length perfluoroalkyl carboxylic acids (PFCAs) in digested sludge aligns with the aerobic biodegradation of FTOHs (Higgins et al., 2005; Ma et al., 2022).

Gobelius et al. (2023) found some PFAS, including perfluoroheptanoic acid (PFHpA), perfluorooctanesulfonamidoacetic acid (FOSAA), and methylperfluorooctanesulfonamidoacetic acid (MeFOSAA), in digested primary sludge, while they were not present in raw primary sludge. In addition, only digested sludge contained perfluorobutanoic acid (PFBA) and PFHpA, whereas perfluorodecane sulfonic acid (PFDS) and perfluoropentanoic acid (PFPeA) were exclusively detected in the sludge before the digestion process. This provides evidence for the transformation of the aforementioned PFAS precursors in the influent wastewater.

Various transformations have been reported for the PFAS compounds within wastewater and sludge treatment processes. The increased transformation of PFAS precursors during biological treatment processes might be a factor in the increased PFAS concentrations identified in biosolids (Zhang et al., 2015). Microorganisms can obtain energy for their growth through defluorination as long as sufficient robust electron donors are accessible to accelerate these reactions (Arvaniti and Stasinakis, 2015). Additionally, reports indicate that the concentration observed in secondary clarifier sludge surpasses that in the primary clarifier sludge. Therefore, the sorption of PFAS into the primary sludge is usually weaker than its sorption into the secondary sludge. This could potentially result from the impedance caused by the background ions, as elaborated in Section 6.

Fluorotelomers (polyfluoroalkyl substances) are the other compounds of concern which have also been considered as a threat to the public health. One of the known fluorotelomers in wastewater entering the environment is Fluorotelomer phosphate diesthers (DiPAPs). 6:2 DiPAP, 8:2 DiPAP, and 6:2/8:2 DiPAP have been reported to be in various types of wastewater (Eriksson et al., 2017; Loi et al., 2013; Szabo et al., 2023). DiPAPs are generally used as greaseproofing agents in food-contact materials and leveling agents in personal care and cosmetic products (Begley et al., 2008) and also in paper and cardboard production (Langberg et al., 2021). In a study conducted by Moodie et al. (2021), 6:2 DiPAP, 8:2 DiPAP, and 6:2/8:2 DiPAP were measured in biosolids from 19 WWTPs in Australia. The diPAPs were detected in almost all of the sampled biosolids and were also generally present in the highest concentrations. Thompson et al. (2023) investigated the transformation of PFAS during biosolids treatment processes. It is reported that three investigated DiPAPs were among the most significant PFAS in all biosolid samples. At almost all of the investigated treatment facilities, the total concentration of the three investigated DiPAPs were greater than the sum of all investigated PFAAs.

Similar results have also been reported by another research team in the US and Australia in which $\sum_3$DiPAPs were greater than that of PFAAs (D'eon and Mabury, 2007; Moodie et al., 2021). It has also been reported that they can undergo various transformation routes and lead to the production of compounds such as PFCAs, PFAA, and FTCAs (Lee et al., 2010; Moodie et al., 2021; Thompson et al., 2023). Thompson et al. (2023) investigated the effect of various treatment methods of biosolids, and surprisingly, they reported that after heat treatment of the biosolids, DiPAP concentrations increased. Due to the lack of literature, it is difficult to theorize the reason for this increase. However, Thompson et al. (2023) have speculated that this increase could be due to the breakdown of triPAPs or larger homologues which are not included in the set of quantified PFAS in their study.

3.1. Transformation mechanisms

There are few studies in the literature discussing the transformation mechanisms and pathways for the generation of PFAS compounds in sludge. However, certain studies have provided insights into the transformation of a few PFAS precursors in sludge. For instance, the biodegradation of 8:2 FTOH leads to the formation of not only PFOA but also other stable metabolites like PFHxA, 2H-PFOA, and 7:3 acid (Wang et al., 2009).

According to the literature, if a precursor undergoes biodegradation, it is likely to occur in the non-fluorinated part of the substance. Defluorination has been detected in the C—F bonds of PFAS precursors, such as FTOH, but this process is limited to polyfluoroalkyl substances and not perfluoroalkyl substances (Fig. 1) (Sáez et al., 2008). Once the compound gains more fluorinated carbons, the chances of biodegradation decrease significantly. At present, there is no indication of the absolute mineralization of PFOS and PFOA compounds (Buck et al., 2011). There are various precursor compounds in wastewater that can be considered as indirect sources for PFOA and PFOS, which are usually found as dominant PFAS in biosolids (Table 1).

Fig. 2 illustrates the possible degradation pathways for fluorotelomer and perfluoroalkane sulfonamido derivatives leading to PFOA and PFOS. In a research conducted by Dinglasan et al. (2004), it was observed that while the levels of 8:2 FTOH declined in WWTP sludge, the concentrations of interim byproducts 8:2 fluorotelomer carboxylic acid (8:2 FTCA) and 8:2 fluorotelomer aldehyde (8:2 FTAL) escalated. This, in turn, led to an elevation in the levels of different metabolites called fluorotelomer unsaturated carboxylic acid (FTUCA). Notably, PFOA was detected throughout a continuous 16-day span during the experiment, aligning with the highest FTUCA concentrations. The investigation unveiled the aerobic decomposition of 8:2 FTOH, and subsequent studies have connected this biodegradation pattern to the significant existence of PFOA within the sludge (Higgins et al., 2005). Fig. 2(a) illustrates the proposed transformation pathway of fluorotelomer substances such as 8:2 FTOH and 8:2 fluorotelomer acrylate (8:2 FTAC) as indirect sources for the PFOA in biological processes (Buck et al., 2011; Dinglasan et al., 2004). Fig. 2(b) shows the proposed aerobic biotransformation of derivatives of perfluoroalkane sulfonamides, with a particular emphasis on compounds containing 8 perfluorinated C atoms such as N-EtFOSE and perfluorooctane sulfonyl fluoride (POSF), which undergoes degradation to PFOS. Researchers have identified several intermediate products that lead to the formation of PFOS (Buck et al., 2011).

Since bacteria are the main agents in biological WWTPs, studies have also focused on the bacteria dealing with the transformation of PFAS precursors. To break down PFAS, a microorganism must (a) uptake the fluorinated compound into the cell, (b) possess a proper enzyme for catalyzing the cleavage of C—F bonds, (c) detect the toxic fluoride ion produced in the process, and (d) defend against fluoride, potentially using a fluoride-proton antiporter (Wackett, 2021). Aerobic bacteria are capable of transforming the compounds such as 8:2 FTOH and EtFOSE. Biotransformation mechanisms of the PFAS compounds via aerobic bacteria have been reported to be catalyzed via cytochrome P450 monooxygenase, transamine metabolism, and beta-oxidation (Dinglasan et al., 2004; Liu et al., 2010; Wang et al., 2011; Wang et al., 2009). While a few studies have touched upon PFAS precursor transformation in sludge, there remains a critical need for further exploration into this area. Specifically, understanding the dynamics of microbial communities and their responses to various operational or environmental factors that could influence precursor degradation is essential.

Research on PFAS transformation in typical WWTPs suggests that while longer chain PFAAs are removed via sludge adsorption, shorter chain PFAS persist and are formed during treatment. Despite studies on PFAS precursors transforming into PFCAs with carbon chain lengths of 4 or more, little is known about their conversion into ultrashort-chain PFCAs (C2-C3). Investigating synthetic precursor compound degradability in laboratory settings, such as biodegradation, ozonation, could enhance understanding of transformation mechanisms and effective treatments. Further exploration into the impact of various biological treatments, particularly anaerobic and aerobic microbial activity, on PFAS precursor conversion to PFAAs is warranted.

4. PFAS partitioning into sludge

Besides the PFAS transformation that was discussed in the previous section, the partitioning of PFAS is another essential factor that impacts the fate of these compounds in WWTP (Becker et al., 2010). Partitioning pertains to the distribution of PFAS compounds between the water and solid phases (sludge). It is impacted by diverse elements, including the physicochemical attributes of PFAS, including their hydrophobicity and

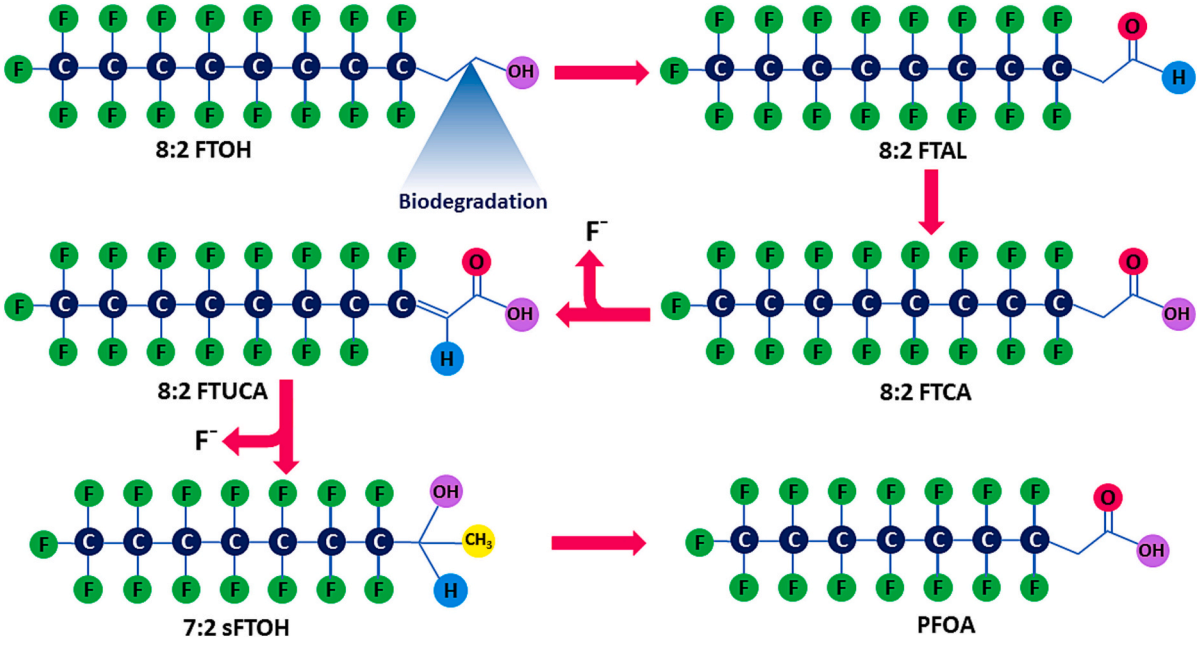


Fig. 1. Biodegradation route of PFAS precursors.

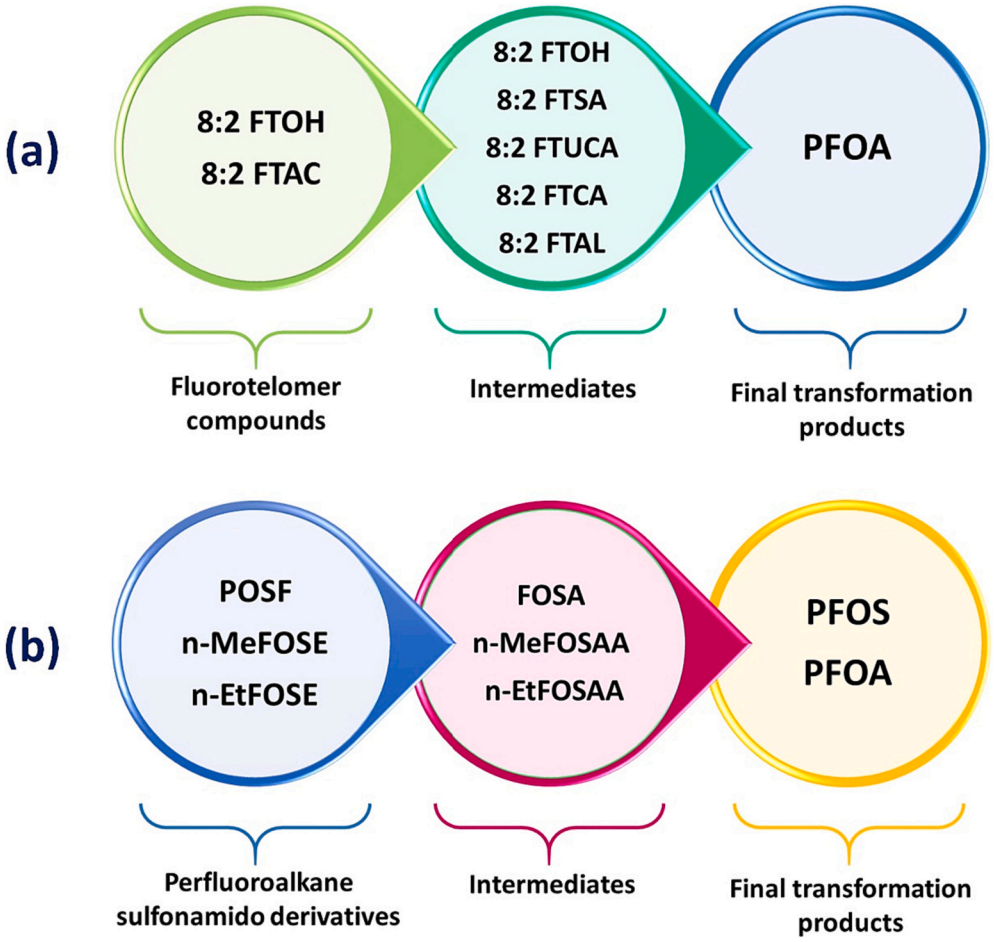


Fig. 2. Proposed degradation pathway of fluorotelomer derivatives (a) and perfluoroalkane.

charge, as well as the properties of the sludge, such as its organic composition and surface electrical charge. The partition coefficient (a ratio of solid phase to solute concentrations of PFAS), also referred to as the distribution coefficient (K_d), quantitatively illustrates how PFAS compounds distribute between water and sludge (Chiavola et al., 2020). Comprehending the partitioning pattern of PFAS and estimating the partition coefficient enables us to make predictions regarding the effectiveness of PFAS removal or partitioning to solid phase (sludge) during wastewater treatment processes. This understanding also facilitates evaluating the possible discharge of PFAS into the surroundings via both biosolids and treated effluent.

An overview of the factors influencing the partitioning of PFAS compounds will be provided in this section. The following discussion will delve into two significant factors: the impact of carbon number (Section 5) and the composition of the wastewater matrix (Section 6). Understanding these factors is crucial for comprehending the behavior of PFAS during partitioning and their subsequent destiny in wastewater treatment processes.

4.1. Impact of length of carbon chain

The properties and composition of the sewage solids are influenced by the specific treatment methods employed in the plants (Ebrahimi et al., 2021). For instance, research has shown that anaerobic digestion can decrease the volatile solids fraction and enhance the ability of hydrophobic pollutants to be sorbed (Stasinakis, 2012). Moreover, several parameters, such as type of PFAS, pH and salinity of water, can influence the K_d of PFAS in WWTP sludge. The literature widely agrees that long-chain PFAS generally have higher K_d values than their short-chain counterparts (Campo et al., 2014; Coggan et al., 2019; Ebrahimi et al., 2021; Eriksson et al., 2017; Jiang et al., 2023; Kim et al., 2022). This significant consistency is primarily attributed to the greater hydrophobicity exhibited by long-chain PFAS, which drives their preferential partitioning from water into sludge. The hydrophobic nature of long-chain PFAS enhances their affinity for organic matter present in sludge, facilitating their sorption onto sludge particles. As a result, the ratios of effluent to influent concentrations of long-chain PFAS are lower than those of short-chain PFAS in wastewater treatment processes (Elmoznino et al., 2018; Kim et al., 2022). It is important to mention that, electrostatic interactions have been proposed as being more influential than hydrophobic interactions in the sorption of PFAS with short-chain into the sludge (Eriksson et al., 2017).

While there is generally consistent evidence indicating higher partitioning of long-chain PFAS to solid phase in comparison to their short-chain counterparts, a study presented contradictory findings (Campo et al., 2014). In a research study, certain short-chain PFAS, including PFBA, perfluorohexanoic acid (PFHxA), PFPeA, and perfluorobutane sulfonate (PFBS), exhibited unexpectedly higher K_d values. Conversely, long-chain PFAS including perfluorotridecanoic acid (PFTrDA) and perfluorooctadecanoic acid (PFODA) demonstrated unusually low K_d values. These irregular observations were attributed to the formation of short-chain PFAS through precursor molecule degradation. The low K_d values observed for certain long-chain PFAS were limited to those occurring in very low concentrations, making them potentially unrepresentative and possibly influenced by inherent sampling method errors (Campo et al., 2014).

4.2. Impact of linkages

When comparing the K_d of PFAS with the same number of perfluorocarbons, it has been observed that the K_d can be influenced by the number of ether linkages and molecular size (Gomis et al., 2015; Zhang et al., 2013a). For example, in the comparison between PFCAs and perfluoroalkyl ether carboxylic acids (PFECAs) having an equivalent count of perfluorocarbons, introducing one or two ether linkages led to a decline in the K_d value of PFECAs. Nonetheless, with a rise in the quantity of ether bonds, the K_d values exhibited a notable increment, surpassing those of PFCAs (Jiang et al., 2023). These results propose a nonlinear correlation between hydrophobicity and the quantity of ether linkages. While the presence of lone electron pairs in oxygen atoms might decrease hydrophobicity, the greater molecular size of PFAS could paradoxically increase hydrophobicity. Hence, PFAS possessing greater molecular weights demonstrate an increased propensity to partition within the solid phase. (Gomis et al., 2015; Jiang et al., 2023; Zhang et al., 2013a).

4.3. Impact of ionic strength of water

Several investigations have emphasized the impact of water's ionic strength on the K_d (Hong et al., 2013; Munoz et al., 2017). For example, a notable exponential association between salinity and the K_d values of PFHxA, PFBA, and PFOS has been observed (Wang et al., 2020a). This discovery implied the occurrence of a salting-out phenomenon that facilitates the transfer of PFAS from the aqueous phase to the solid phase (Wang et al., 2020a).

Specific cations like K^+ and Na^+ form ion-pairs with anionic PFAS, altering their partitioning behavior and increasing hydrophobicity by neutralizing charged groups (Jeon et al., 2010). Another explanation is the existence of divalent cations such as Mg^{2+} and Ca^{2+} which can introduce a bridging effect. A bridging occurrence happen when a divalent cation interacts in electrostatic interactions with both the anionic head group of PFAS and the negatively charged surface of the sludge. This effect has implications for the partitioning pattern of PFAS, influencing their distribution among different phases (Arvaniti et al., 2012; Ebrahimi et al., 2021; Wang et al., 2020a).

A recent investigation observed that the K_d for PFHpA and perfluorohexanesulfonic acid (PFHxS) increased with increasing levels of Ca^{2+} and Mg^{2+} ions (ranging from 1 to 30 mM). However, beyond 30 mM, the K_d values started to decrease (Du et al., 2015). The increase in K_d values (higher partitioning to solid phase) at lower concentrations was attributed to the positive charge induced by the sorption of Ca^{2+} and Mg^{2+} on the negatively charged organic compounds present in the sludge. This favorable positive charge offered additional binding sites for the anionic PFHpA and PFHxS molecules. Nonetheless, with Ca^{2+} and Mg^{2+} concentrations surpassing 30 mM, specific organic constituents within the sludge gradually detached from the solid phase and entered the aqueous phase. This release of organic content resulted in a decrease in the quantity of the sorbent, subsequently reducing the K_d values (Du et al., 2015; Higgins and Luthy, 2006).

A comparison of the K_d for PFHpA and PFHxS with equal concentrations of Ca^{2+} and Mg^{2+} revealed that the K_d values in the existence of Ca^{2+} were much higher compared to those with Mg^{2+} (Du et al., 2015). It was hypothesized that the stronger direct hydration due to magnesium's stronger covalent character, compared to calcium, could potentially restrict its interactions with PFAS compounds and sludge, thereby reducing its affinity for sorption (Kennedy et al., 2004). In general, the presence of mono- and divalent cations can lead to higher K_d values, but further investigation with larger sampling numbers and a wider concentration range is needed to elucidate clearer trends.

4.4. Impact of solution pH

The pH of wastewater is another influential factor in the partitioning of PFAS compounds. Lower pH values have been associated with increased sorption due to the reduced negative surface charge of the sludge. This is particularly noteworthy considering that PFAS predominantly exist as anions within the evaluated pH range (Ebrahimi et al., 2021).

Some investigations have demonstrated substantial variations in PFAS K_d in sludge across pH levels ranging from 2 to 9, with increased sorption observed in more acidic environments (Zhang et al., 2013a; Zhou et al., 2010). Conversely, a rise in solution pH from 7 to 8 has been discovered to substantially amplify the K_d value of PFOS in sediment (You et al., 2010). Nevertheless, within the pH range of 4.6 to 8, the sorption of PFOS, PFBS, and PFOA onto diverse sediments remained unaffected (Milinovic et al., 2015). These findings underscore the importance of pH in influencing the K_d of PFAS and emphasize the need for further research to comprehend the role of pH for various types of PFAS compounds.

4.5. Impact of digestion and stabilization

Reports indicate that treatment processes, such as secondary treatment and stabilization, can influence the physiochemical properties of sludge and, consequently, its PFAS sorption behavior. For instance, it has been observed that anaerobic digestion decreases the fraction of volatile solids, resulting in an increased sorption capacity for hydrophobic compounds (Stasinakis, 2012). In another study, the variation in compound-specific K_d values across four different secondary treatment methods was found to be insignificant. However, sludge stabilization had a notable impact on the partitioning characteristics among biosolid samples.

Overall, K_d values for biosolids from anaerobic digestion were higher than those from composted and aerobically digested biosolids (Ebrahimi et al., 2021). Differences in partitioning coefficients among sludge stabilization processes may stem from variations in biosolid organic matter quality. For instance, aerobic microorganisms present in aerobic composting and digestion processes, such as fungi, degrade organic matter more extensively compared to anaerobic systems, potentially altering the sorption propensity of PFAS (Hubbe, 2014). Moreover, anaerobic digestion reduces the volatile solid fraction and enhances the sorption capacity of hydrophobic compounds (Stasinakis, 2012), further influencing PFAS sorption of biosolid generated in anaerobic digestion (Ebrahimi et al., 2021).

Additionally, operational parameters linked to secondary treatment methods, such as temperature and hydraulic retention time (HRT), have been shown to influence the conversion of PFAS precursors. Higher precursor conversion rates have been observed at elevated temperatures and HRTs (Guerra et al., 2014). This conversion process, involving the biotransformation of fluorinated precursors into stable PFAAs, notably affects the PFAA load in WWTP effluent. In some cases, this mechanism leads to higher effluent concentrations compared to influent concentrations (Ebrahimi et al., 2021).

In summary, the reviewed studies have explored various factors influencing the partitioning of PFAS compounds into sludge. The specific treatment methods employed in WWTPs were found to affect the properties and composition of sewage solids, ultimately impacting PFAS sorption. Long-chain PFAS displayed higher K_d (higher partitioning to solid phase) in contrast to short-chain PFAS, attributed to their greater hydrophobicity and stronger sorption onto sludge particles. However, some studies reported conflicting results, highlighting the need for further investigation. The K_d was also impacted by the quantity of ether linkages and the size of PFAS molecules, showcasing a non-linear relation between hydrophobicity and the count of ether linkages. The occurrence of monovalent and divalent ions like Ca^{2+} and Mg^{2+}, significantly affected the K_d values. pH was identified as another crucial factor, with lower pH values associated with increased sorption, while higher pH values showed varying effects depending on the PFAS compound and sediment type. Overall, these findings underscore the

importance of considering factors including length of PFAS chain, molecular structure, ionic strength, and pH in predicting and managing PFAS partitioning in sludge.

5. PFAS sorption into sludge based on the number of carbon

As elucidated in the previous section, the partitioning of PFAS compounds is significantly affected by the length of the carbon chain. According to Pan et al. (2016), PFBS and PFOS were identified as the predominant compounds within the aqueous phase of WWTP, while PFOS exhibited dominance in the WWTP dewatered sludge. The observed phenomena can be ascribed to the heightened adsorption capability of PFAS with longer carbon chains into sludge. It was discovered that in the case of sorption mechanism of the anionic perfluorochemical, perfluorinated methylene (CF_2) component contributed approximately 0.50–0.60 logarithmic units to the measured distribution coefficient value, underscoring the essential role of the carbon chain length in PFAS in shaping the sorption mechanism. On the other hand, the sulfonate moiety contributed to an additional 0.23 logarithmic units to the measured distribution coefficient (Higgins and Luthy, 2006). This point has also been approved in other studies. For instance, Guo et al. (2010) indicated that long-chain PFCAs and PFOS exhibit a greater inclination to migrate into the sludge phase, and the sorption strength of a perfluorosulfonate compound exceeded that of its perfluorocarboxylate counterpart. Furthermore, with an increase in the length of the carbon chain, the sorption of perfluorocarboxylate acid on sludge also intensified.

In another study conducted in Sweden, 26 PFAS were tracked to identify their sources, transport, and fate. Short-chain PFAS were prevalent in the wastewater, whereas long-chain PFAS were predominantly identified in the sludge. The ratio of long-chain N-EtFOSAA and perfluorosulfonic acids (PFSAs) to the total PFAS compounds were diminished within the wastewater treatment processes, since they were mainly transferred into the sludge. In the conducted analysis, PFOS exhibited the highest concentrations among all the investigated PFAS compounds. It was identified as the predominant compound in sludge, comprising 11–62 % of the total concentration of 26 PFAS compounds (Gobelius et al., 2023). This observation of PFOS dominance in sludge aligns with previous reports on the subject (Table 1). Gobelius et al. (2023) reported that the $\sum_{26}$PFAS concentrations in dewatered sludge (33 ng/g dw) and digested sludge (ranging from 12 to 36 ng/g dw) exhibited greater levels in contrast to those measured in primary clarifier sludge (ranging from 7.0 to 34 ng/g dw). Among the different sludge types, the maximum total PFAS levels were identified in sludge generated from the secondary clarifier (61 ng/g dw), while the minimum was detected in primary clarifier sludge (7.0 ng/g dw). This notable rise in PFAS concentrations between primary and secondary clarifiers signifies the considerable movement or attachment of long-chain PFAS from the aqueous phase to the solid phase throughout the biological treatment process (Gobelius et al., 2023).

Furthermore, studies have indicated that PFAS compounds with an even number of carbon atoms exhibited a stronger propensity for biosolid accumulation compared to compounds with an odd number (Bossi et al., 2008). Gobelius et al. (2023) reported that PFOA ($C_8HF_{15}O_2$) and perfluorodecanoic acid (PFDA) ($C_{10}HF_{19}O_2$) had higher sludge levels than perfluorononanoic acid (PFNA) ($C_9HF_{17}O_2$) and perfluoroundecanoic acid (PFUnDA) ($C_{11}HF_{21}O_2$). In another investigation carried out in Shanghai, China, the sludge from eight WWTPs was examined for the presence of short- and long-chain PFAS (ranging from C2 to C14). The principal identified compounds were PFOA and PFOS, comprising 2–34 % and 1–9 % of the overall PFAS, respectively. Their results indicated that PFOS level was almost close to the values of PFOA. While in similar studies, higher PFOS concentrations than PFOA were reported (Johnson, 2022; Mailler et al., 2017). The discrepancies can be attributed to the diverse influent wastewater matrices of the WWTPs.

Certain PFAS compounds, notably PFAS with long-chain such as PFOS and PFOA, have faced bans or restrictions since the late 2000s. Consequently, this phenomenon has caused changes in forming and exposure patterns, giving rise to the appearance of short-chain PFAS and other substitute PFAS compounds (Boucher et al., 2019; Gomis et al., 2015; Houtz et al., 2016). The PFAS identified in sludge represent both currently used PFAS (e.g., short chain PFAS) and legacy PFAS (e.g., PFOA and PFOS) that are still in use and are being gradually released into wastewater. Despite the major restriction of C8-based PFAS, including PFOA and PFOS, a significant proportion of PFAAs observed in municipal WWTP effluent still consisted of PFOS and PFOA even about ten years after the initiation of these restricting regulations (Guo et al., 2010). The findings presented in this section highlight the significant impact of carbon number on the partitioning of PFAS compounds, with longer-chain PFAS showing a higher tendency to adsorb into sludge during wastewater treatment processes.

6. PFAS sorption into the sludge at various wastewater matrices

The wastewater matrix plays a critical role in the sorption of PFAS into the sludge. There are limited studies investigating this fact. The studies on this case can be analyzed by examining the interactions between PFAS compounds and the organic and inorganic ions present in wastewater.

Organic ions including carboxylic acids, proteins (amino acids), natural organic matter (NOM), and dissolved organic matter (DOM), can affect the PFAS sorption into the sludge and also their elimination from the liquid phase (Li et al., 2023). The existing literature provides restricted insight into the conceivable interplays between the actual water matrix and PFAS. Few investigations have explored the impact of organic ions on PFAS sorption, predominantly concentrating on the influence of humic and fulvic acids (Qi et al., 2022). There are also limited studies regarding the sorption of PFAS onto diverse adsorbents when influenced by DOM. The existence of DOM in the background solution can play two different roles according to the types of adsorbents. Dixit et al. (2020) studied the removal of PFAS in DOM-rich water in anionic ion exchange resins. They have found that due to the blockage of the pores by DOM, they compete with PFAS compounds, reducing the resins' adsorption capacity. On the other hand, DOM were reported to help biological processes to effectively remove PFAS. This is due to the fact that DOM can act as a barrier to diminish the toxicity effect of PFAS on microorganisms (Qi et al., 2022). Furthermore, the complexation of DOM with elements such as iron and iodine can function as an electron transport mediator, enhancing electron transfer processes (Daugherty et al., 2017).

In a research conducted by Du et al. (2015), it was revealed that significant decreases in PFAS adsorption onto sludge occurred following thermal treatment, which resulted in a reduction of organic content within the surrounding matrix. Additionally, they inferred that PFAS demonstrates a heightened attraction to sludge originating from secondary treatment processes characterized by elevated biomass levels. In another study, it was observed that the higher protein content of the sludge increases the PFAS adsorption, which may be due to certain amide groups assisting the sorption of PFAS onto sludge (Yan et al., 2021). On the contrary, Pramanik et al. (2017) found that the presence of protein in the wastewater influent resulted in a reduction of PFOS and PFOA elimination.

Inorganic ions, on the other hand, are also determining the adsorption of PFAS, which is illustrated in Fig. 3 (Saeidi et al., 2020). Divalent cations like calcium and magnesium intensify PFAS sorption onto sludge by creating a linkage between negatively charged PFAS compounds and the sludge particles (Zhang et al., 2013a). The presence of monovalent cations, such as sodium in the wastewater matrix has been shown to impede PFAS adsorption compared to that of divalent cations (Arvaniti et al., 2014). There is a strong completion with anions like sulfate, phosphate, nitrate, and chloride and PFAS to occupy the adsorption sites, leading to interference with PFAS adsorption on sludge (Ateia

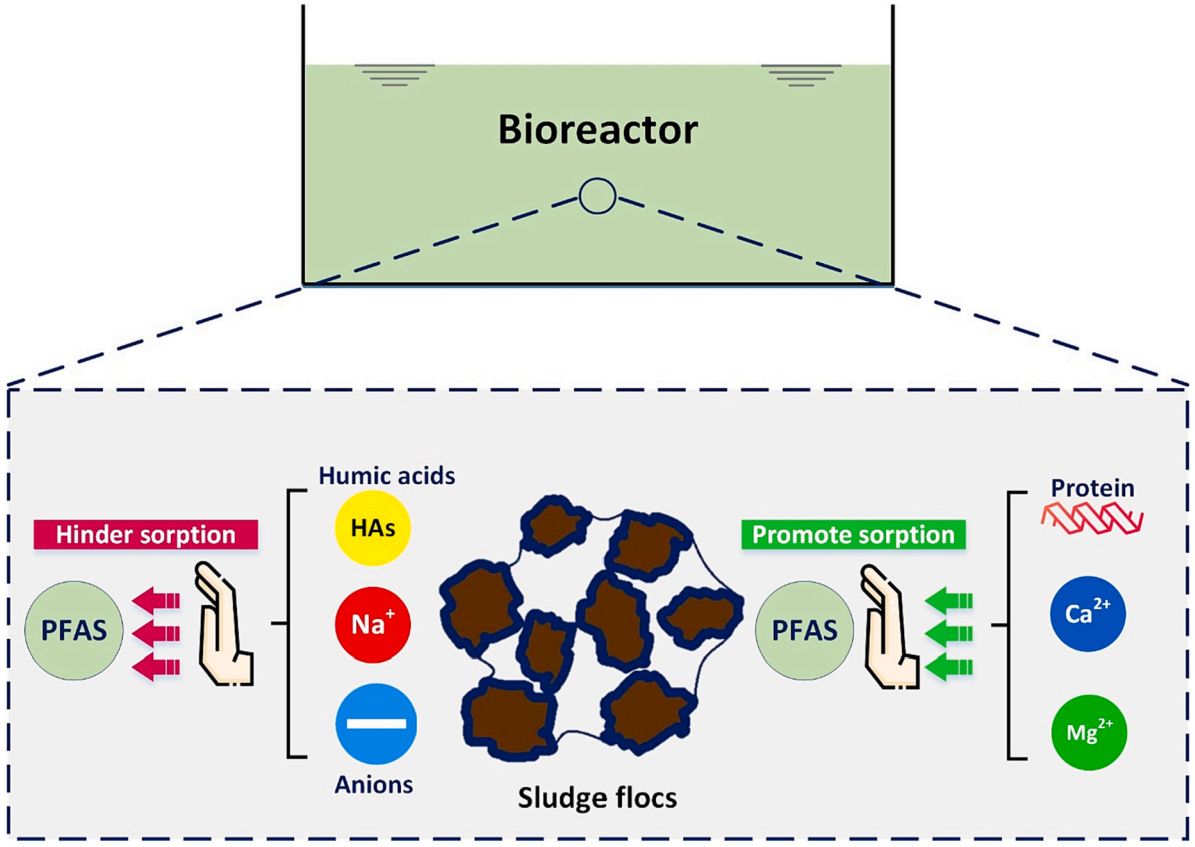


Fig. 3. The effect of various matrices of wastewater on PFAS sorption into biosolids.

et al., 2019; Dixit et al., 2021).

The diverse findings concerning the influence of organic and inorganic ions on PFAS treatment within wastewater underscore the necessity for additional investigations into the consequences of distinct organic substances and inorganic ions (cations and anions) on eliminating PFAS and their precursors.

The influence of the simultaneous presence of PFAS along with background ions on the fate of PFAS in WWTPs remains unexplored. Hence, comprehending the concurrent impacts of a range of organic and inorganic ions on eliminating coexisting ions and PFAS is paramount. Furthermore, delving into the chemical interplays between diverse ions and PFAS that result in variations in sorption patterns could yield valuable mechanistic understandings.

7. Adverse environmental impacts of accumulated PFAS in sludge

Given the intricate degradation pathways of PFAS precursors within WWTPs and the diverse behaviors of PFAS compounds during sludge bioaccumulation, it becomes evident that the presence of PFAS in more hazardous environmental components and the human food chain could potentially elevate health risks. In recent times, growing worries have emerged regarding the adverse environmental and health effects resulting from the accumulation of PFAS in sludge. WWTPs sludge is considered a final destination for the PFAS, which are usually used in various industries (Thompson et al., 2023). Because of the surfactant characteristics of PFAS, the organic carbon segment of the perfluoroalkyl chain tends to segregate, while the ionic head groups remain dissolved at the boundary between organic carbon and water (Rayne and Forest, 2009). Accordingly, the generated sludge from WWTPs, especially plants receiving industrial wastewater, could be considered as an archive of various pollutants being absorbed. On the other hand, WWTPs sludge is usually used as fertilizers in agricultural fields provided they meet certain regulations before application. Hence, the substantial volume of produced sludge globally presents a potential risk to global public health. Based on the reported studies, over 12.5 million tons of dry sludge have been generated in China, and a large amount of them is released into the environment lacking appropriate treatment processes (Chen et al., 2022a; Guo et al., 2023). In European countries, it is estimated that almost 10 million tons of dry sludge are produced, and about half of the generated sludge is applied in the farm fields as fertilizers (Domini et al., 2022). Fig. 4 illustrates distinct techniques employed for the ultimate disposal of biosolids across various countries worldwide.

Based on the gathered information, it is found that, on average,

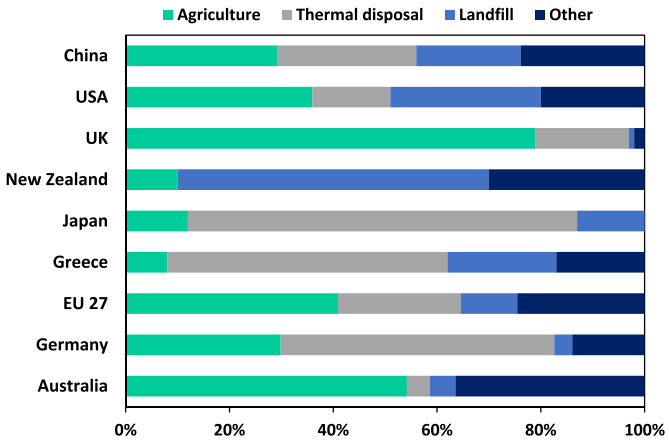


Fig. 4. Diverse approaches for managing biosolids in various countries across the globe [Data adapted from Sharma et al., 2017; Wei et al., 2020].

approximately 33.8 % of biosolids obtained from WWTPs are repurposed for agricultural use as fertilizer. However, owing to the presence of PFAS within these biosolids, there is a notable potential for these compounds to infiltrate both the food chain and the surrounding surface and underground water systems threatening public health (Johnson, 2022; Pepper et al., 2021). Accordingly, it is of great importance to routinely monitor sewage sludge, and also more conservative approaches should be considered in revising the present regulations in case of sewage sludge application in agricultural activities.

8. Future perspectives

Here are some future perspectives for studies on PFAS in sludge and biosolids:

1. **Long-term Fate and Transport:** Investigate the long-term behavior of PFAS compounds in sludge-amended soils and their potential to migrate into groundwater and surface water. This could involve monitoring and modeling studies to comprehend the persistence, movement, and potential bioaccumulation of diverse PFAS species.
2. **Transformation Pathways:** Research the potential transformation pathways of PFAS in sludge during different treatment processes, such as anaerobic digestion, composting, and incineration due to the lack of information in this area. Understanding how different PFAS compounds degrade or transform under different conditions can aid in developing effective treatment strategies.
3. **Influence of Wastewater Matrices:** Exploring techniques to diminish the uptake of PFAS within sludge, while considering the influence of diverse matrices present holds significance. Acknowledging that divalent cations enhance the sorption of PFAS into sludge by creating links between negatively charged PFAS and sludge particles, whereas monovalent cations and anions have the opposite effect.
4. **Toxicity and Ecological Impacts:** Expand studies on the toxicity of PFAS-contaminated sludge and biosolids to different organisms, including plants, aquatic life, and soil microorganisms. This research can help assess potential ecological risks and inform guidelines for safe land application.
5. **Human Health Risks:** Conduct comprehensive exposure assessments to estimate the potential human health risks related to the application of PFAS-contaminated sludge to agricultural lands. This includes assessing the pathways through which PFAS may enter the food chain and estimating potential health impacts on farmers, workers, and consumers.
6. **Regulation and Policy:** Contribute to the development of robust regulatory frameworks and guidelines for managing PFAS in sludge and biosolids. Collaborative efforts between scientists, policymakers, and regulatory agencies are essential to establish secure waste management methods and protect human well-being and the environment.
7. **Alternative Waste Management Options:** Explore alternative methods for managing PFAS-contaminated sludge and biosolids, such as thermal destruction technologies, chemical treatment, or innovative approaches like phytoremediation. Assess the feasibility, effectiveness, and potential drawbacks of these options.

By addressing these future perspectives, scientists can add to a more profound comprehension of PFAS behaviors within biosolids and sludge, facilitate informed decision-making, and ultimately mitigate the potential risks associated with these contaminants.

9. Conclusion

In conclusion, the presence of PFAS in various products has led to their eventual introduction into wastewater systems, with wastewater treatment plants being recognized as significant sources of PFAS discharge into the environment. While WWTPs attempt to mitigate the influence of PFAS through physicochemical and biological mechanisms, their presence in the generated biosolids remains an environmental concern. In the current review, the presence and dominance of PFAS in biosolids, their transformation mechanism of degradation in the biological treatment processes, and partitioning process have been comprehensively investigated in the literature, and the following conclusions can be extracted from this literature review:

- The reviewed studies showcased diverse PFAS concentrations among distinct WWTPs, underscoring the significance of partitioning between aqueous and solid phases in shaping sludge PFAS content. Higher concentrations were evident in sludge from WWTPs receiving wastewater rich in PFOS and PFOA. Notably, despite variations in treatment procedures, comparable PFAS concentrations were found in sludge. Moreover, aspects such as sludge retention time played a role in influencing PFAS accumulation.
- Aerobic bacteria have been documented to catalyze biotransformation mechanisms of PFAS compounds, involving cytochrome P450 monooxygenase, transamine metabolism, and beta-oxidation. These transformations are influenced by microorganisms utilizing defluorination for energy. However, there is a need for further research into the mechanisms and implications of PFAS precursor transformation in different sludge types.
- Treatment methods are found to affect PFAS sorption. It is also concluded that the number of ether linkages and molecular size impact K_d non-linearly. Furthermore, mono- and divalent cations influence K_d value. On the other hand, pH has a significant role in PFAS partitioning.
- The results underscore how carbon chain length significantly affects PFAS compound partitioning, indicating that longer-chain PFAS tend to adsorb more into sludge during wastewater treatment processes.
- Higher sludge protein content boosts PFAS adsorption, possibly aided by amide groups facilitating PFAS attachment. Divalent cations like calcium and magnesium bridge PFAS and sludge for increased adsorption. In contrast, monovalent cations like sodium in wastewater hinder PFAS adsorption more than divalent cations.

CRediT authorship contribution statement

Ali Behnami: Writing – original draft, Visualization, Methodology, Investigation, Conceptualization. **Khaled Zoroufchi Benis:** Writing – review & editing, Writing – original draft, Investigation. **Mojtaba Pourakbar:** Writing – review & editing, Writing – original draft, Investigation. **Mojtaba Yeganeh:** Writing – review & editing. **Ali Esrafili:** Writing – review & editing, Methodology, Conceptualization. **Mitra Gholami:** Writing – review & editing, Methodology, Investigation, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

No data was used for the research described in the article.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2024.171559.

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Review

PFAS in landfill leachate: Practical considerations for treatment and characterization

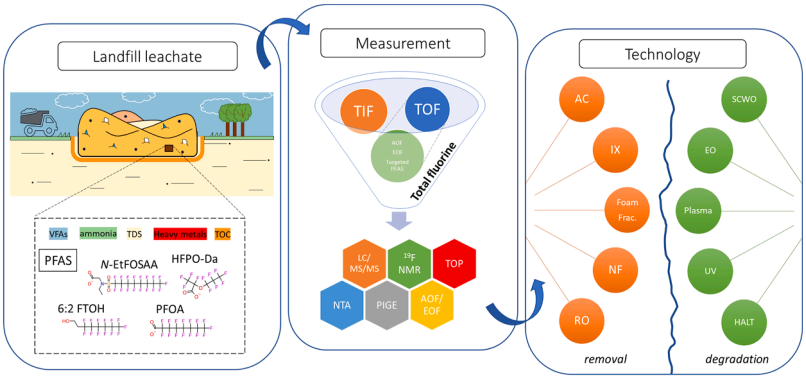
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HIGHLIGHTS

- PFAS speciation in leachate is dominated by mobile short-chain PFAS.
- Analytical techniques should be combined for complete leachate characterization.
- Foam fractionation optimization is beneficial for downstream destruction.
- Emerging destruction technologies show promising results but need further research.
- Treatment trains should be adaptable to varying flow and leachate characteristics.

GRAPHICAL ABSTRACT



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ABSTRACT

Per- and polyfluoroalkyl substances (PFAS) are widely used in consumer products and are particularly high in landfill leachate. The practice of sending leachate to wastewater treatment plants (WWTPs) is an issue for utilities that have biosolids land application limits based on PFAS concentrations. Moreover, landfills may face their own effluent limit guidelines for PFAS. The purpose of this review is to understand the most appropriate treatment technology combinations for mitigating PFAS in landfill leachate. The first objective is to understand the unique chemical characteristics of landfill leachate. The second objective is to establish the role and importance of known and emerging analytical techniques for PFAS characterization in leachate, including quantification of precursor compounds. Next, an overview of technologies that concentrate PFAS and technologies that destroy PFAS is provided, including fundamental background content and key operating parameters. Finally, practical considerations for PFAS treatment technologies are reviewed, and recommendations for PFAS treatment trains are described. Both pros and cons of treatment trains are noted. In summary, the complex matrix of leachate requires a separation treatment step first, such as foam fractionation, for example, to concentrate the PFAS into a lower-volume stream. Then, a degradation treatment step can be applied to the concentrated PFAS stream.

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1. Introduction

Per- and polyfluoroalkyl substances (PFAS) have become a major global concern because of their widespread use in everyday consumer products [1]. These substances contain carbon-fluorine (C-F) bonds, which are extremely stable bonds and exceedingly difficult to break using conventional treatment methods [2]. These chemicals have unique properties that make them repel water and oil and are resistant to heat. Therefore, PFAS are highly desirable for many applications such as non-stick cookware, make-up, and fast-food wrappers [3]. PFAS are highly persistent in the environment, meaning they do not break down easily and can accumulate over time. PFAS have also been found to bioaccumulate in human tissue [4]. This accumulation can lead to chronic diseases of select organs, including liver damage, thyroid disease, decreased fertility, high cholesterol, obesity, hormone suppression, and cancer [5,6]. These effects can be particularly severe in vulnerable populations, such as infants, pregnant women, and people with compromised immune systems [7].

The widespread use and disposal of consumer products containing PFAS, coupled with the physical and chemical stability of these chemicals, has led to increased concentrations in municipal landfill leachate [8]. Leachate is the liquid that results when rainwater passes through landfill waste and is collected by impervious liners. Typically, leachate is processed through wastewater treatment plants (WWTPs) before being released back into the environment. Unfortunately, conventional treatment methods like biological treatment have proven ineffective against PFAS compounds [9]. In some cases, measurable PFAS levels in the processed water have been higher than in the original leachate due to the breakdown of PFAS precursors during biological treatment [10]. Therefore, treated wastewater can contaminate drinking water. Treatment of municipal landfill leachate provides an opportunity to break this cycle of contamination.

The United States Environmental Protection Agency (US EPA) has developed a PFAS Strategic Roadmap that involves gathering data from landfills, to make informed decisions about future regulations to limit PFAS discharges from landfills [11] . This underscores the need to identify sustainable approaches for reducing PFAS contamination from landfills.

In recent years, researchers have developed novel approaches for treating PFAS contamination via a two-step process. The first step is a separation process that removes PFAS and reduces the volume of PFAS-contaminated matrix (e.g., spent adsorbent, brine). The second step is a degradation process that aims to mineralize PFAS. These novel approaches include separation technologies like foam fractionation (FF) and flocculation using novel flocculants, as well as destruction technologies like electrochemical oxidation, plasma treatment, supercritical water oxidation (SCWO), and hydrothermal alkaline treatment (HALT) [12]. These technologies have shown promising results in laboratory studies, pilot/demonstration testing, and field trials, suggesting their potential for effective PFAS treatment in landfill leachate. Despite their promising benefits, many of these technologies still pose significant challenges that have yet to be fully explored or addressed, prior to achieving a commercial level of readiness for full-scale implementation. Key challenges in implementing these technologies include the formation of by-products such as unfinished volatile and aqueous PFAS products [13], high energy and capital costs [14], low scalability and limited system capacity [15], operational challenges associated with frequent maintenance (e.g., scaling, corrosion, passivation) and low operational familiarity due to the novel nature of the technologies [16].

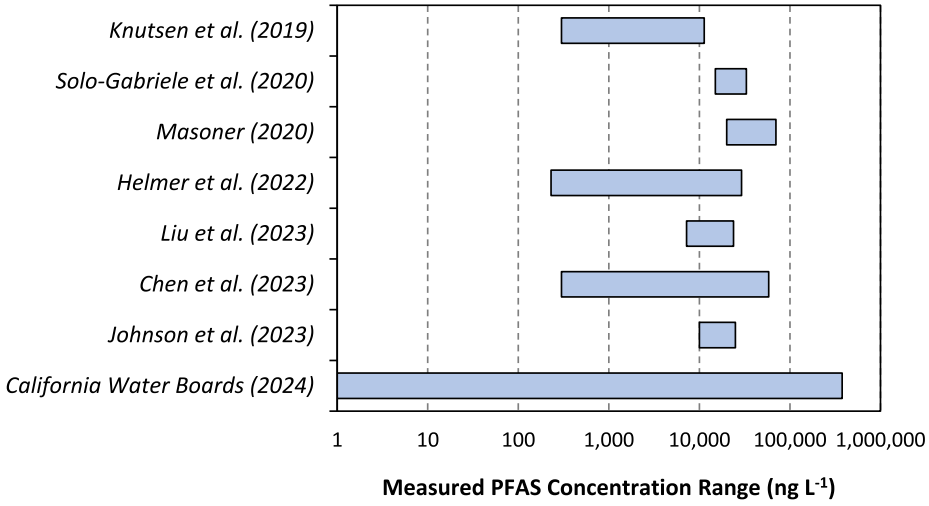
Research on emerging PFAS treatment technologies in landfill leachate has gained significant momentum in recent years, with laboratory studies progressing to pilot testing and field trials [17,18,8]. As a result, the literature on the treatment of PFAS in landfill leachate has grown rapidly, highlighting the need for a critical review and updates on the latest knowledge. While current literature mainly focuses on the theoretical aspects of emerging technologies [19] either in the early stages of development or tested on a bench-scale, there is a lack of discussion on their potential to become a viable solution in the near future. To address this gap recognized in most PFAS reviews [20-22,17,23,8], this review takes a more practical approach by assessing the commercial applicability of treatment technologies and examining state-of-the-art treatment train configurations for utilities receiving leachate streams. The specific objectives of this review are to (1) establish the unique nature of leachate composition and ramifications on treatment, (2) elucidate the complementary role of established and emerging PFAS characterization and quantification lab techniques, (3) provide an overview of commercially available treatment technologies and assess their potential applicability, (4) identify a treatment strategy for municipal landfill leachate, and (5) discuss future actions for improved management of PFAS in receiving leachate.

2. Leachate composition

Leachate contains elevated concentrations of organics, salts, heavy metals, and solids [24]. The concentration of leachate components varies widely depending on the age of the landfill and the type of waste it receives. Landfill leachates tend to have a pH of 6.5 – 8.5 with older leachates (> 10 years) having alkaline pH (> 7.5) [25]. Typical landfill leachates may have 2000 – 60,000 mg L^{-1} of chemical oxygen demand (COD) [26]. The biodegradability of leachate, as represented by the ratio of 5-day biochemical oxygen demand to COD (BOD_5/COD), decreases with the age of the landfill with the decomposition of biodegradable components over the years and can be in the range of 0.1 to 0.3 [27]. Leachate organics typically consist of 80 % volatile fatty acids (VFAs), but with time organic components mostly contain recalcitrant humic substances, e.g., humic and fulvic acids [26]. Leachate typically contains a high concentration of ammonia, and 80 % of total nitrogen in leachate can be in the form of ammonia [26]. Ammonia concentration may vary from 20 – 1200 mg L^{-1}, and older leachates (> 10 years) typically contain high concentrations of ammonia as it is more persistent in an anaerobic environment [25]. The total dissolved solids (TDS) concentration may vary from 3000 – 18,000 mg L^{-1}. A low concentration of TDS is observed in older leachates owing to a decrease in acidity resulting in low solubility of ions [28]. The total concentration of heavy metals, i.e., lead (Pb), nickel (Ni), iron (Fe), zinc (Zn), chromium (Cr), and copper (Cu), can be as high as 100 mg L^{-1} and similar to TDS, the concentration of heavy metals also decreases with the age of landfill [25]. Electric conductivity may vary from 6 – 34 mS cm^{-1} [29].

Leachate composition varies widely. An analysis of composite leachate samples from landfills in New Jersey showed that the water quality parameters were in the following ranges: ammonia: 785 – 1850 mgN L^{-1}, TDS: 9560 – 14,300 mg L^{-1}, Hardness: 324 – 735 mg L^{-1}, and total organic carbon (TOC): 495 – 2005 mg L^{-1}. A very low oxidation state was overserved in the leachates (nitrate: <0.114 mgN L^{-1}; nitrite: <0.009 mgN L^{-1}) reflecting anaerobic conditions within the landfills. In addition to nutrients being in their most reduced form (i.e., ammonia), metals were also in their dissolved form due to the low oxidation state. Accordingly, ammonia and TDS concentrations were higher in these leachate samples.

The composition of leachate is important because it can impact the treatability of PFAS. PFAS concentration is typically higher in leachate compared to other water matrices, e.g., municipal wastewater [30]. However, PFAS concentration is still typically multiple orders lower than the organics and salts in leachate. Thus, organics would compete with PFAS for adsorption sites in adsorption or ion exchange-based treatment technologies. Some studies report improved long-chain PFAS removal through aggregation of organics during adsorption [31]. However, the aggregation effect of organics on PFAS removal from leachate (which consists of a very high concentration of organics) is yet to be completely understood. Similarly, high concentration of salts would compete with PFAS in membrane-based processes. In oxidation-based technologies, organics and salts would compete for the



# leachate samples	# measured PFAS
10	28
4	11
3	25
19	28
3	17
78	26
3	13
69	43

Fig. 1. Variability of total measured PFAS concentrations in landfill leachate from recent studies (References: [44-51].

oxidants (in UV or other oxidation technologies), or electrode surface (in electrooxidation) [32,33]. Moreover, the impact of leachate composition on PFAS treatability would vary depending on the relative concentrations of organics, salts, or other constituents. For instance, leachate from elevated temperature landfills (ETLFs) typically consists of very high concentrations of organics, salts, and metals compared to typical leachate [34]. PFAS treatability in leachate from ETLFs would therefore be even more negatively impacted than in a typical leachate.

An analysis conducted by Tolaymat et al. [17] reported that about 7480 kg PFAS were disposed from municipal solid waste in 2018, of which 750 kg were emitted into leachate. Another study conducted by Lang et al. [35] analyzed 95 leachate samples from 18 landfills and estimated that annually approximately 600 kg PFAS are released in leachate. PFAS in leachate may originate from a variety of consumer products, e.g., plastics, paper, textiles, rubber, leather, electronics, cookware, cleaning liquids, etc. [36]. These PFAS compounds result in landfill leachate via direct or indirect pathways.

The concentration of PFAS may vary from 100 to greater than 100,000 ng L^{-1} in leachate [37]. For instance, as shown in Fig. 1, the variability of PFAS concentration across different studies shows that the majority of landfills contain concentrations within the tens of thousands ng L^{-1}. PFAS concentration in landfill leachate is typically higher than it is in groundwater, surface water, and wastewater due to the accumulation effect of PFAS in the landfill and the complex nature of the leachate, which highlights the need for treatment prior to processing in WWTPs [21]. For example, a study analyzing 254 groundwater samples found that PFAS concentration varied from non-detect – 1645 ng L^{-1} [38]. Another study found up to approximately 330 ng L^{-1} PFAS in groundwater near a reclaimed water recharge facility [39]. Moreover, a different study reported that groundwater samples near a PFAS manufacturing facility were contaminated with up to 4773 ng L^{-1} PFAS compounds [40]. The same study reported that the surface water near the PFAS manufacturing facility was contaminated with 426 – 3617 ng L^{-1} PFAS. However, surface water contains less PFAS compared to the aforementioned report. For example, 64 river water samples in Alabama were analyzed for PFAS and the highest reported concentration was 237 ng L^{-1} [41]. PFAS concentration in municipal wastewater influent and wastewater may vary from non-detect – 1000 ng L^{-1} and 15 to greater than 1500 ng L^{-1}, respectively [13]. Szabo et al. [42] reported up to 400 ng L^{-1} of PFAS in wastewater influent. Tavasoli et al. [43] analyzed the influent and effluent of 6 WWTPs and found that PFAS concentration ranged from 30 – 198 ng L^{-1}. Compared to the reported PFAS concentrations in groundwater, surface water, and municipal wastewater, PFAS concentration is higher in landfill leachate, and therefore leachate should be targeted as a concentrated stream of PFAS for treatment to reduce PFAS circulation.

Leachate age plays a role in PFAS profiles, although research on this topic is mixed. Huang et al. [30] reported that PFAS concentration was higher in leachates from active landfills (12,194 ng L^{-1}) than in leachates from closed landfills (2747 ng L^{-1}). The same study also suggested that PFAS concentration was not influenced by leachate properties, e.g., nitrogen, ammonia, organics, or pH level. As such, the study concluded that the age of leachate did not impact PFAS concentration in leachates [30]. However, Zhang et al. [52] reported that PFAS concentration in leachate is correlated with leachate-specific conductivity, alkalinity, TOC, ammonia, COD, TDS, and TS. However, Zhang et al. [52] emphasized that the correlation between leachate PFAS and these parameters does not indicate causation. Since leachate age, dilution, and waste types have an impact on not only leachate properties, e.g., specific conductivity, alkalinity, TOC, ammonia, COD, and TDS, but also PFAS concentration, leachate properties were correlated to PFAS concentration.

In addition, the PFAS speciation profile in landfill leachate is highly variable and does not resemble other typical water sources. Short-chain PFAS compounds may account for a major fraction of landfill leachate PFAS. For instance, an analysis of 6 municipal landfill leachates from Western China revealed that two short-chain PFAS compounds, e.g., perfluorobutane sulfonate (PFBS) and perfluorobutyric acid (PFBA) comprised 23.1 % and 20.6 % of the total PFAS mass, respectively [30].

Long-chain PFAS compounds or precursors in leachate can transform into relatively shorter-chain PFAS compounds with time and so landfill age can impact PFAS composition. Transformation of PFAS compounds can occur via multiple mechanisms, e.g., abiotic, aerobic, and anaerobic transformation [17]. For instance, thermal degradation may transform PFAS into unidentifiable precursors instead of achieving complete mineralization [53]. Some studies reported an increase in the precursors of short-chain compounds after aerobic degradation of solid wastes [54, 55,53]. In addition to the transformation of the PFAS precursors, the abundance of short-chain PFAS in leachate may also be a result of their inherent property, e.g., lack of hydrophobicity caused by a shorter nonpolar tail, which makes them more readily leachable from landfill solids into the liquid stream.

While PFAS fingerprints or profiles affect how total PFAS are reduced, equally important to treating leachate is being able to measure,

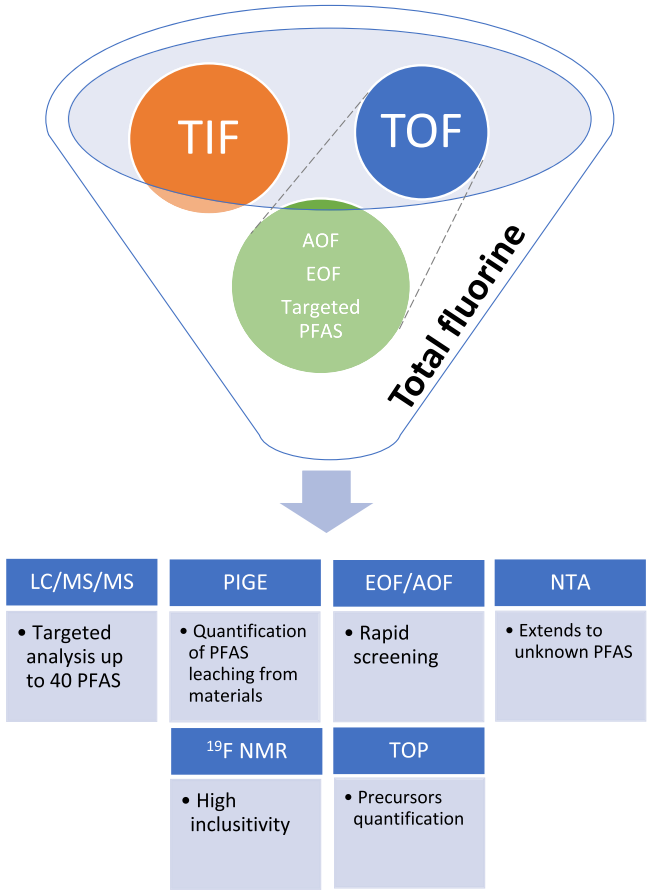


Fig. 2. Current methods for PFAS analysis. TIF: total inorganic fluorine; TOF: total organic fluorine; AOF: adsorbable organofluorine; EOF: extractable organofluorine.

or characterize the PFAS.

3. PFAS characterization in landfill leachate

To date, PFAS analysis in landfill leachate has primarily relied on the use of liquid chromatography-tandem mass spectrometry (LC-MS/MS) to identify and quantify selected subgroups of PFAS (e.g., perfluoroalkyl carboxylic [PFCAs] and sulfonic acids [PFSAs]) with reference to commercially available native and isotope-labeled standards [56-58,47,35,49,59]. Currently, LC-MS/MS methods for PFAS analysis in non-potable waters, such as those published by EPA (e.g., draft Method 1633 (U.S. Environmental Protection Agency 2023) [60]) and ASTM International (e.g., ASTM D8421–22 (ASTM International 2022) [61]), only target up to 40 analytes and fall short of capturing the chemical diversity of PFAS (i.e., over 14,700 with explicit chemical structures listed on the EPA's CompTox Chemicals Dashboard (U.S. Environmental Protection Agency 2022b) [62]). Furthermore, authentic reference standards exist for fewer than 120 PFAS, including only a handful for zwitterionic species and none for cationic species [63]. Complementary approaches capable of measuring the bulk organofluorine content in complex aqueous matrices and expanding the analytical coverage beyond routinely targeted species are therefore critical for improving the assessment of the total PFAS burden associated with landfill leachate [64,65]. As shown in Fig. 2, multiple class-specific techniques for organofluorine measurements, such as fluorine-19 nuclear magnetic resonance spectroscopy (**^{19}F NMR**) [66,67], particle-induced gamma-ray emission (**PIGE**) spectroscopy [68], the extractable or adsorbable organofluorine (**EOF** or **AOF**) assays [69,70], and the total oxidizable precursor (**TOP**) assay [71], have been increasingly used in combination with targeted LC-MS/MS methods and/or non-targeted analysis (**NTA**) [72] enabled by high-resolution mass spectrometry (HRMS) for in-depth compositional characterization of PFAS in environmental samples [73]. With a few exceptions [44,74-78], such multiplatform approaches have remained underexplored for the analysis of landfill leachate.

Given our focus on landfill leachate, surface spectroscopic techniques such as instrumental neutron activation analysis (INAA) [79,80], X-ray fluorescence (XRF) spectroscopy [81,82], fluorine *K*-edge X-ray absorption near-edge structure (XANES) spectroscopy [83,84], are excluded from further discussion despite their proven capabilities for PFAS characterization in solid materials.

3.1. Fluorine-19 nuclear magnetic resonance spectroscopy (^{19}F NMR)

^{19}F NMR has gained renewed interest in recent years as a non-destructive tool for total organofluorine (TOF) analysis since its application to determine PFAS in rainwater and aqueous Film Forming Foam (AFFF)-impacted surface water samples in the early 2000s [66,67]. Compared to ^{1}H, ^{19}F exhibits a much larger chemical shift range (e.g., over 300 ppm) [85] and is only ~17% less sensitive [86], making ^{19}F NMR a unique tool for PFAS characterization with minimal interfering background signals due to the scarcity of natural occurring fluorinated compounds [87]. Current applications of ^{19}F NMR range from quantitative measurements and structural elucidation of PFAS given its specificity for detecting the characteristic resonant frequencies of per- and polyfluorinated alkyl groups [88-92] that are distinct from those of aromatic trifluoromethyl substituents or aromatic fluorines commonly present in fluorine-containing pharmaceuticals and pesticides [93,94]. For example, with the implementation of a baseline noise-reduction technique, ^{19}F NMR not only achieved sensitivity comparable to that of LC-MS/MS for PFAS quantification in environmental water samples (albeit at the cost of analysis time) but also identified organofluorine species (e.g., pharmaceuticals with aromatic trifluoromethyl groups) beyond those targeted by LC-MS/MS [90]. Comparative evaluation of ^{19}F NMR with the TOP assay and LC-HRMS for PFAS analysis in wastewater samples further demonstrated that the former measured significantly higher PFAS concentrations and captured additional species (e.g., trifluoroacetic acid [TFA]) poorly amenable to either of the latter methods [89]. Strictly speaking, the fraction of organofluorines quantified by ^{19}F NMR is specific to the type of sample extraction method selected (e.g., SPE [90]) and cannot be defined as the total PFAS unless non-selective techniques (e.g., lyophilization [89]) are used for sample preconcentration. To our knowledge, ^{19}F NMR has not been applied for PFAS characterization in landfill leachate. Compared to LC-MS/MS, ^{19}F NMR is likely less susceptible to background interference from organic matter; however, the high salt content and presence of paramagnetic metals in landfill leachate may cause line broadening and difficulties with shimming during NMR data acquisition. Sample cleanup techniques such as solid-phase extraction may help minimize these effects but could result in the loss of species not effectively retained by the sorbents. Method development and validation are therefore needed to evaluate the trade-offs among different analytical considerations for landfill leachate in future work.

3.2. Particle induced gamma-ray emission (PIGE)

PIGE is a non-destructive, accelerator-based ion beam analysis technique that measures total fluorine in a solid sample by detecting characteristic gamma rays emitted during the de-excitation of ^{19}F nucleus [68]. Typically applied as a surface spectroscopic technique (e.g., with a beam penetration depth of up to 0.25 mm [68]), PIGE has been coupled with targeted LC-MS/MS and/or gas chromatography-MS methods to investigate PFAS release from carpet or clothing in model anaerobic landfill reactors [95] and to quantify the amount of PFAS or side-chain fluorotelomer-based polymers in papers and textiles [96,68,

97], fast food packaging [98], fabric and foam samples from children car seats [82], fire station dust [99], cosmetics [100], and facemasks [101]. Compared to bulk volume techniques such as CIC, PIGE is less sensitive and more prone to variability in sample thickness and heterogeneity; however, it can distinguish between coated and uncoated surfaces and may serve as a rapid screening tool [80]. Indeed, recent work has further demonstrated the feasibility of PIGE for PFAS monitoring in drinking water following sample filtration through activated carbon [102] as well as its promise as a field-deployable system for *in situ* screening of PFAS in groundwater at AFFF-impacted sites [103]. PIGE has not been applied for PFAS characterization in landfill leachate, but it can be integrated into a mobile platform [103] for total organofluorine screening at landfill facilities, provided that the in-line sample extraction method (e.g., solid phase extraction, SPE) can be optimized to improve PFAS recoveries while minimizing interference from inorganic fluoride.

3.3. Extractable and adsorbable organofluorine (EOF and AOF)

Combustion ion chromatography (CIC) has been widely applied for total (inorganic plus organic) and organofluorine (operationally defined as EOF or AOF) measurements since its initial development and application for fluorine mass balance analysis in seawater [69] and human blood samples [104]. Calibration of CIC can be achieved using inorganic fluoride, model organofluorine compounds, or a combination of both [105]. For the EOF assay, weak anion exchange SPE cartridges are typically used for aqueous extraction before CIC analysis, during which organofluorines (including both PFAS and other fluorinated compounds) in solvent extracts are combusted under hydropyrolytic conditions [106], and the evolved hydrogen fluoride gas is trapped and measured as fluoride ion by conductivity detection [69]. Two recent interlaboratory comparison studies confirmed the promising reproducibility and accuracy of the EOF assay for PFAS quantification in environmental and biological sample matrices, but they also highlighted several methodological considerations related to sample extraction, instrumental analysis, and quality assurance and control criteria for improving data reproducibility and comparability [107,108]. For the AOF assay, self-packed or commercially available activated carbon columns with low fluorine content are used for organofluorine extraction from aqueous samples, followed by combustion of the loaded activated carbon by CIC [70]. No interlaboratory validation study exists for the AOF assay [109], but the U.S. EPA has released a draft Method 1621 for AOF screening in wastewater (U.S. Environmental Protection Agency 2022a) [110]. Compared to the SPE sorbent (which mostly captures anionic PFAS), activated carbon (AC) also adsorbs ionic and neutral PFAS [109]; however, challenging aqueous matrices such as landfill leachate may lead to breakthrough and low recoveries of PFAS due to the competitive adsorption of matrix components onto activated carbon [74]. Consequently, results from the AOF and EOF assays are not directly comparable [111]. Concurrent optimization of the AOF and EOF assays has increased the recoveries and lowered the limits of detection for an expansive set of PFAS [112], but both assays are still limited by poor recoveries for ultrashort-chain perfluoroalkyl acids ([PFAAs]; mainly due to loss during the washing step [113]) and incomplete mineralization of PFSAs [66,86]. Overall, CIC stands as the most frequently used tool (e.g., relative to high-resolution-continuum source graphite furnace molecular absorption spectrometry [111]) for quantifying EOF or AOF in PFAS research. For the analysis of landfill leachate, the recoveries and combustion efficiencies of PFAS should be thoroughly evaluated in matrix spikes. Closing the organofluorine mass balance using EOF or AOF data should also account for potential contributions from compounds not traditionally labelled as PFAS (e.g., fluorine-containing pharmaceuticals and pesticides [74,114]) in landfill leachate.

3.4. Total oxidizable precursor (TOP)

The TOP assay converts difficult-to-analyze PFAA precursors to more readily quantifiable PFCAs and perfluoroalkyl ether carboxylic acids (PFECAs) for subsequent quantification by LC-MS/MS [71,115,116]. Originally developed as a semi-quantitative tool in 2012, the assay was first applied to estimate the level of PFAA precursors in urban runoff following oxidative digestion by hydroxyl radicals generated from persulfate thermolysis under alkaline conditions [71]. Given its compatibility with LC-MS/MS and specificity for perfluoroalkyl moieties [117], the TOP assay has been widely adapted (often with adjustments to the oxidation reaction conditions) to quantify the concentrations of PFAA precursors in various aqueous matrices, e.g., surface waters [118-120, 109,121], groundwater [119,122,123], precipitation [124], municipal and industrial wastewater [125-128,43], AFFF [129,130,63] and landfill leachate [44,76,78]), solid samples (e.g., soil, [131,122,132,133,63, 134] sediment [135], and suspended particular matter [136,137]), and consumer products (e.g., textiles [138,97,139]). Since its development, the original TOP assay has also undergone methodological modifications, most of which have sought to enhance its applicability as a quantitative tool or expand its analytical coverage to include previously overlooked precursors. For example, multiple studies have explored alternative oxidative pretreatments based on chemical oxidation (e.g., hydrogen peroxide [140] or ozone [141], photocatalysis (UV/TiO_2) [142], or UV-activated persulfate oxidation [115,143] for precursor conversion, although these approaches do not necessarily outperform heat-activated persulfate oxidation. More recent efforts have validated the recoveries of a broader suite of neutral, zwitterionic, and cationic precursors (e.g., betaines, amines, amine oxides, and quaternary ammonium compounds) [144] as well as precursors to ultrashort-chain PFCAs (e.g., TFA and perfluoropropanoic acid) [117,145,78]. With the addition of isotope-labeled surrogates (e.g., perfluoro-1-[$^{13}C_8$]octanesulfonamide) before oxidation, the recoveries of oxidizable precursors can be further corrected to enable more accurate quantification [75,115, 77]. Still, not all precursors are transformed upon oxidation (e.g., perfluoroalkyl ether carboxylic acids [PFECAs]) or form terminal products (e.g., polyfluoroalkyl ether acids with a -O-CFH- moiety) that are routinely targeted by LC-MS/MS [116]. Currently, no standardized method exists for the TOP assay despite the recognition of its potential for contaminated site investigations and regulatory monitoring [146, 120,131]. Three earlier interlaboratory comparison studies reported significant variability in TOP results [131,147,148], possibly due to inconsistencies in protocols among different laboratories; however, a recent study coordinated through the NORMAN network demonstrated satisfactory recoveries, analytical precision, and low limits of quantification for the assay when applied to the analysis of environmental water samples [149]. Considering the growing application of the TOP assay, ongoing efforts to optimize and standardize the method [146,115,77] are also expected to improve its effectiveness in environmental forensics. For example, through integration with statistical tools (e.g., Bayesian inference and principal component analysis), the TOP assay has proven useful for reconstructing the chain length and manufacturing origin of oxidizable precursors in AFFF (e.g., electrochemical fluorination versus fluorotelomerization) [130], attributing sources of PFAS in AFFF-impacted surface waters [150], and differentiating the source fingerprints of PFAS (e.g., among AFFF, municipal wastewater, and landfill leachate) [77,151]. Overall, the TOP assay has evolved as a key method for PFAS characterization and holds the potential to serve as a rapid *in situ* screening tool (e.g., when coupled with smartphone-based sensors [129]). For the analysis of landfill leachate, the recoveries of precursors should be validated against isotope-labeled surrogates in matrix spikes [77,115], and the inclusion of ultrashort-chain PFCAs in the assay is recommended considering their abundance [78].

Table 1
Overview of concentration and separation technologies applied to landfill leachate.

Technology	Water quality requirements	Pros	Cons	References
Concentration				
Foam Fractionation (FF)	Cations or cosurfactants to improve foam formation	• No generation of major byproducts • Low-cost generation of gas bubbles • Effective for PFAS-impacted wastewater and landfill leachate • No chemicals or sorbents used	• Only targets surfactants. • Short-chain PFAS are less prone to fractionation due to their higher solubility in water • Further pilot studies required	Ebersbach et al.[177] Buckley et al. [178]
Membrane separation				
Reverse Osmosis (RO)	Lower concentration of organics	• Relatively small footprint • Nonspecific • Removal of nontargeted harmful pollutants even at trace levels • Applicable to most water matrices	• Long-term decrease in PFAS rejection rate • Requires pretreatment to prevent membrane scaling and/or fouling. • High volume of water lost to reject stream (5–15%) • High energy requirements and high capital costs. More expensive than IX and GAC. • Requires chemical addition for membrane preservation such as antiscalants	Thompson et al.[179] Tow et al.[16]
Nano Filtration (NF)	Lower concentration of organics	• Lower pressure compared to RO • Less expensive than RO • Better short-chain PFAS removal compared to AC and IX • Can aid removal of heavy metals	• PFAS removal is lower compared to RO due to larger pore size • Less power intensive than RO • Retentate needs to be treated or infiltrated back to the landfill	Appleman et al.[180] Yan et al.[59] Malovanyy et al.[181]
Adsorption				
Activated Carbon (AC)	Lower concentration of competing compounds	• Ease of operation • Well-established for full-scale treatment • Can be reactivated and reused over multiple cycles	• Frequent media changeout. • PFAS could be displaced by more selective compounds resulting in concentrations higher in the effluent than in influent. • Requires higher EBCT than other adsorbents • Non selectivity • High sensitivity to TOC levels • Destruction of exhausted AC	Zeng et al. [182] Gagliano et al. [183]
Ion Exchange (IX)	Lower concentration of competing compounds Feed water needs pretreatment to prevent organic fouling.	• Ease of operation • Low empty bed contact time (EBCT) compared to conventional adsorbents • Well-established for full-scale treatment. • Targets both short- and long-chain PFAS and a wide range of hazardous contaminants even at trace levels • Significantly lower changeout frequency • Small footprint	• Competing divalent anions have high affinity reducing the capacity for PFAS • Effective regeneration of PFAS-laden resin requires proper handling and disposal of hazardous waste • PFAS could be displaced by more hydrophobic PFAS compounds resulting in concentrations higher in the effluent than in influent • Difficulties of managing exhausted media	Deng et al. [184] Boyer et al. [185] Fang et al. [186] Pannu et al. [187]

3.5. Non-targeted analysis (NTA)

With the rapid advancement in benchtop HRMS instrumentation (e.g., Orbitrap or quadrupole time-of-flight) and chemometric tools, NTA (broadly defined as a collective term for suspect and nontarget screening) has emerged as an essential tool for discovering previously unknown PFAS or their precursors (i.e., those can transform to highly persistent PFAAs) [72]. Over the past decade, the application of NTA has not only contributed to the identification of more than 830 PFAS [64] in various environmental and biological samples, consumer products, and industrial formulations [152-161] but also provided mechanistic insights into the abiotic or biotic transformations of PFAS under conditions relevant to contaminated sites and engineered treatment systems [162-167]. Current NTA data mining workflows generally integrate mass defect and CF_2-normalized Kendrick mass defect filtering, homolog series detection, diagnostic fragment and neutral loss searching, and suspect screening to establish multiple lines of evidence for the structural annotations of newly identified PFAS or their precursors at different confidence levels (e.g., ranging from level 1 confirmed by/indistinguishable from reference standard to level 5 exact mass of interest) [72,168]. Collaborative efforts by the PFAS research community have also led to the development of novel software solutions (e.g., FluoroMatch [169] and PFΔ*Screen* [170]) and reaction libraries [171,172] to improve annotation throughput and coverage. Given that landfill leachate is an analytically challenging matrix, the application of NTA for PFAS characterization may be biased towards the extractable and detectable chemical space [173] delineated by the specific sample extraction method (e.g., weak anion exchange SPE) and instrument configurations (e.g., reversed-phase LC and negative electrospray ionization). Only three studies have used NTA to identify a small number of PFCAs, PFSAs, and PFAA precursors in landfill leachates collected from facilities in Florida [169], South China [174], and Sweden [76]. These studies achieved a confidence level of 3 or higher based on proposed identification confidence levels in high resolution mass spectrometric analysis described in Schymanski et al. [175].

Complementing NTA with other class-specific techniques discussed above should enable a more thorough representation of the PFAS landscape in landfill leachate. For newly identified PFAS, semi-quantification can be performed to assess their relative abundance compared to targeted species, [154] or to close the organofluorine mass balance with EOF [176] or TOP [63]. Overall, as our analytical understanding of PFAS measurements advances, so too will our understanding of treatment efficacy.

4. Treatment strategies

4.1. Concentration and separation

Concentration and separation technologies have become increasingly important in the treatment of PFAS-contaminated landfill leachate

due to their ability to serve as a frontline treatment method for PFAS at significantly higher flow rates than destruction technologies (e.g., 22–200 $m^3 h^{-1}$ or 100–1000 gpm vs. 0.22–2.2 $m^3 h^{-1}$ or 1–10 gpm). By significantly reducing the amount of PFAS-laden waste produced, these techniques can help lower the overall cost of PFAS remediation by several orders of magnitude [8]. Additionally, concentration and separation techniques are already well-established and widely used in other applications, such as drinking water, and for previously regulated contaminants, making them familar for operational use by utilities. The three commonly used techniques for PFAS removal are adsorption, FF, and membrane separation. Each technique has unique advantages and disadvantages, making it necessary to choose the most appropriate method for a specific application (Table 1). In the next sections, the principles, advantages, and limitations of adsorption, FF, and membrane separation for the removal of PFAS are discussed.

4.1.1. Adsorption

Adsorption removal is currently the most mature technology for PFAS removal due to its ease of operation, extensive applicability, and low operational costs [188]. However, selecting the appropriate adsorption media is critical and heavily dependent on the quality of the test water. Even with highly selective media, the adsorption capacity of PFAS (μmol L^{-1}) in raw leachate can be two orders of magnitude less than that of natural water bodies [189], leading to higher frequency of changeout and material cost, and breakthrough occurring up to 28 times faster than natural water matrices (e.g., groundwater, surface water) [181]. Therefore, adsorptive media can be used as a polishing step or an "end-of-pipe" solution to reduce PFAS concentrations to meet the target goals before injecting plant effluents into groundwater wells. Even then, the life-cycle cost of the system is to be assessed if high organic loading or significant amounts of resin foulants (e.g., iron and manganese) are present in the WWTP effluent considering the renowned impact of the presence of co-existing organic matter negatively affects short-chain PFAS adsorption due to active site competition [190].

ACs, such as granular activated carbon (GAC) and powdered activated carbon (PAC), are porous carbonaceous substances derived from natural materials that are processed at high temperatures. These materials can include coal (bituminous, sub-bituminous, and lignite), coconut shells, wood (hardwood and softwood), and other organic materials such as peat, olive stones, and sawdust. ACs have been widely used as a separation technology for removing trace organic compounds, including PFAS. AC primarily interacts with PFAS through hydrophobic interactions between the non-polar functional groups on the GAC and the hydrophobic tail of PFAS. Thus, ACs are more effective for long-chain PFAS removal [191]. However, the non-selectivity of this interaction means that TOC and other hydrophobic compounds (e.g., volatile organic compounds) can also compete for adsorption sites, leading to media fouling and reducing the bed life for PFAS. For example, typical leachate contains 500 – 10,000 mg L^{-1} of TOC compared to the recommended operation value of 2 mg L^{-1} for ACs [192]. To ensure efficiency, it is essential to consider the changeout frequency through demonstration testing (i.e., rapid small-scale column tests [RSSCT] and pilot testing [193]. While ACs are a relatively inexpensive material, the fast changeout frequency can be balanced with a dedicated destruction technology, such as reactivation or incineration, which would require shipping the exhausted material to a reactivation site, preferably close to the leachate receiving plant. However, there is currently no conclusive evidence of complete destruction through mass balance analysis in the off-gas of the thermal reactivation/incineration due to a lack of information related to the methodology adopted for evaluating the process effectiveness or potential for emission of unfinished toxic products [183]. Emerging non-combustion technologies for the destruction of exhausted ACs are currently in the lowest stages of development compared to other PFAS-laden waste streams, and further exploration of these applications is required.

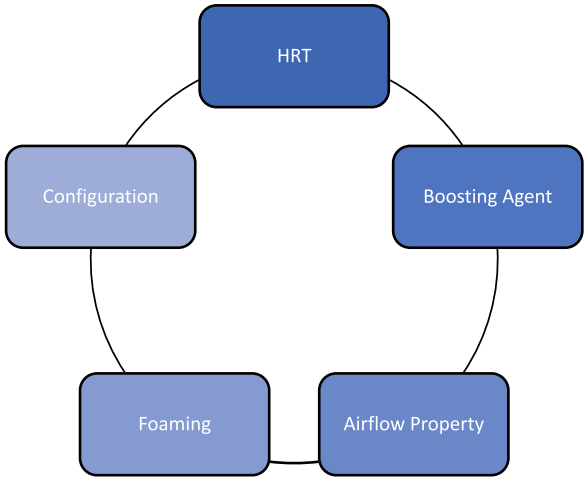


Fig. 3. Operational parameters that can be tweaked for leachate FF and have been shown to impact treatment performance.

Recent advancements in adsorption technology have resulted in the development of more selective structures using synthetic materials such as ion exchange (IX) and engineered clay-based materials [194]. However, these advancements come with challenges in handling the media. IX resins become non-regenerable and are single-use only [195,196], while synthetic regeneration of clay-based materials is not yet established [197]. The effective management of PFAS-laden media has been hindered by the limited applicability of degradation technologies. Technologies such as supercritical water oxidation exhibit low maturity levels, while others like electrochemical oxidation require an additional step to separate PFAS from the media into a "still bottom." This additional step necessitates the handling of high cosolvent content regenerants, which are not desirable at treatment facilities due to their hazardous nature [186,198].

Recent developments in adsorbent properties is driven towards a multi-mechanism removal approach (i.e., similar to IX) whereby ionic interactions are enhanced to engage with the functional head group PFAS (e.g., negatively charge carboxylic or sulfonic head group) while still utilizing hydrophobic interactions to enhance partitioning of PFAS onto the adsorbent. Ateia et al. [199] demonstrated the benefits of adsorbents interscaled with a balanced ratio of amine functional groups and a long-chain carbon to provide an interplay between electrostatic and hydrophobic interactions, thus developing the next generation of PFAS sorbents. Examples of such amine-functionalized adsorbents include surface-functionalized hexagonal mesoporous silica (HMS), mesoporous organic frameworks (COFs), and metal-organic frameworks (MOFs), among others. However, these novel adsorbents have mostly been tested in batch configuration and lack maturity in real-life applicability in column mode. Moreover, morphology plays a significant role in adsorption kinetics, whereby finer particles allow for faster diffusion of PFAS in the pores but hold a disadvantage in column configuration as that would lead to significant headloss through the column bed [199]. One novel adsorbent type, namely modified clay-based adsorbent, that shares these properties has made it to the commercial scale, which is a low-cost modified smectite clay that shows high resiliency to organic co-contaminants and specificity to PFAS adsorption [200]. Nevertheless, many of the studies elucidating the scaling of this sorbent to full-scale is yet to be illustrated. For instance, Pannu et al. [187] and Medina et al. [201] showed that RSSCT using ground clay-based media experiments significantly overpredicted pilot-scale breakthrough of PFAS, indicating that much of the scaling approach such as the impact of hydraulics, media size, and other operating conditions is yet to be fully understood.

4.1.2. Foam fractionation (FF)

Foam fractionation is a separation technique that utilizes air bubbles pumped from the bottom of the cylinder to preferentially separate

surface-active molecules. Due to their high surface activity, the hydrophobic long carbon chain moiety of PFAS contributes to migration from water to a more thermodynamically stable state where the hydrophobic tail of PFAS is pushed towards the air molecule and the hydrophilic head (e.g., sulfonic or carboxylic headgroups) remains in the water phase [202]. The PFAS compounds would then be transported along the trajectory of air molecules to the top water surface where compounds with higher surface activity are more susceptible to foaming. The foam is then skimmed off and sent for further treatment. The concentrations of PFAS in foam are reported to reach between an enrichment factor of 100 to 100,000 [203].

In general, various parameters, such as HRT, configuration, boosting agent, foaming behavior, and airflow property, can be optimized to achieve improved levels of PFAS removal efficiency in FF used for leachate (Fig. 3). From the operational standpoint, airflow is critical to achieving the volume of foaming and the desired enrichment factor. This is defined as the ratio of PFAS concentration in the foamate over the initial concentration in the feed. With an increased air flow rate, higher foaming is formed by the increased upward movement of bubbles and reduced foam drainage from capillarity [204]. This increases the removal rate of PFAS but decreases the enrichment factor due to higher volumes of foam.

Vo et al. [204] explored the effectiveness of four different types of cosurfactants (or boosting agents) for leachate FF of PFAS and their interfacial partitioning on air bubbles. The study found that all cosurfactants were effective for removing long chain PFAS, with cationic and zwitterionic surfactants being particularly effective due to their charge interactions with anionic PFAS. It was proposed that the addition of surfactants with opposing charges to PFAS provides additional means to electrostatically attract and bolster the transfer of PFAS to the interface [205,178,206]. This is especially helpful to transport PFAS with low surface activity, such as short-chain PFAS. Buckley et al. [207] showed that the addition of a limited number of individual cations to the solution significantly increased the performance of the FF process in the order of magnesium (Mg^{2+}) > potassium (K^{+}) > sodium (Na^{+}). Proprietary surfactant polymers have been developed and recently employed on a commercial level such as the use of cetylpyridinium chloride (CPC) [208,209].

In a continuous column pilot study, Smith et al. [210] evaluated various conditions in which an reduction flow rate (reduced contact time) from 16 to 6.9 L min^{-1} (13 to 30 min contact time) increased total PFAS removal from 63% to 84% but stayed asymptotic afterwards at 60 min indicating that the equilibrium has been reached after a certain contact time and that an optimal contact time could be experimentally identified based on the design conditions and feed water quality. In the same study, Smith 2023, reported an increase in overall foaming percentage (volume of foam relative to the feed) from 0.07–0.35% to 7.1% when the surfactant dosage increased by 5-fold which is suggested to be due to increased stability of the foam that leads to less drainage [211]. However, excessive foaming may become a disadvantage considering that, even though the PFAS removal could be higher, the concentration of PFAS in the foamate would be reduced due to the larger volume of foamate (lower enrichment factor). As such, piloting would help identify the optimal foaming conditions to generate the desired enrichment to volume ratio which would impact downstream destruction technologies.

Wang et al. [203] compared the efficacy of PFAS removal between single-stage leachate FF treatment and multi-stage treatment. The results showed that the removal of PFOA, 6:2 FTS, and PFOS via two-stage FF treatment was similar to that of single-stage treatment. Over 80% of PFOA and 6:2 FTS was removed from the original sample, and the removal efficiency of PFOS exceeded 90% after either treatment. However, the enrichment factors for PFOA, 6:2 FTS, and PFOS were approximately 10 times higher than those after single-stage treatment. This is because the volume of collected foam was much smaller after two-stage FF. Overall, these findings suggest that multi-stage treatment could be a viable option for removing PFAS, as it provides similar removal results to single-stage treatment, but with higher overall enrichment factors.

4.1.3. Membrane separation

High-pressure membrane technologies such as nanofiltration (NF) and reverse osmosis (RO) are considered very effective at removing PFAS from the influent stream due to the small pore size of membranes and the ionic state of PFAS [212]. NF membranes have typically higher feed recovery or are operated at a lower pressure than RO due to larger membrane pore sizes. With a denser construction and smaller pore size in RO technologies resulting in higher PFAS removal, RO has been more common for PFAS separation. The expected influent recovery using conventional RO is between 50 to 80% depending on the source water quality and up to 90% with high-recovery RO technologies [213].

Previous research has shown that NF has a success rate of more than 90% in removing PFAS from leachate [214,215]. Membrane bioreactors, followed by NF and RO, have been able to achieve more than 95% of PFAAs degradation from biologically treated or ordinary leachate, as indicated in several studies [57,59]. Aside from the superior short-chain PFAS removal ability in comparison to AC and IX resin, another benefit of NF is its ability to separate heavy metals as well. Malovanyy et al. [181] reported that NF removed up to 92% of heavy metals, whereas AC and IX demonstrated only negligible removal rates. Nevertheless, the application of NF poses a challenge in that the retentate must be treated or infiltrated back into the landfill.

Due to extremely high organic and mineral concentrations in the landfill leachate, very high fouling potential is anticipated for RO/NF, which requires significant pretreatment using at minimum, micro- or ultrafiltration membranes to protect the RO/NF membranes from suspended solids fouling. Given that typical leachate streams are considered low-flow compared to water and wastewater streams, specialty RO membranes can be effectively employed requiring wide spacers to allow for more effective chemical cleaning and more frequent removal of foulants during routine clean-in-place procedures. These specialty membranes are considered cost-competitive with conventional RO when used at higher flow streams (>4546 m^3 d^{-1} or 1 MGD) [216,217]. One of the benefits of employing RO is the non-selectivity of removal of most monovalent and divalent ions from the water but this wider implementation is hampered by membrane fouling, high energy requirements, and frequent clean-in-place requirements when applied to leachate. With the reject stream being 10–30% of the influent stream, further concentration is required to reduce the volume of PFAS-laden waste. Typical feasible concentration steps for RO brine include thermal evaporation and crystallization [16] that evaporate water and result in a concentrated solid residual. However, PFAS may partition from the aqueous brine to the vapor or gaseous stream which would require a thermal oxidizer to further ensure PFAS is reduced in emissions.

4.2. Degradation

When considering the potential value of treatments for PFAS, it is important to first distinguish terminology. Mineralization/defluorination refers to the chemical process whereby the C-F bond is broken, and inorganic fluoride remains. Parent PFAS can be partially degraded into transformation products that are i) different than the parent compound, but ii) still categorized as PFAS. Operational conditions may significantly impact the fate of influent PFAS, and it is possible that the following technologies can achieve both mineralization/defluorination and partial degradation of parent PFAS into transformation products. As with concentration and separation techniques, each of the degradation techniques has its unique advantages and disadvantages, making it necessary to choose the most appropriate method for a specific application (Table 2).

4.2.1. Plasma

Recent studies have shown that plasma technology can be an

Table 2
Overview of degradation technologies applied to landfill leachate.

Technology	Water quality requirements	Pros	Cons	References
Degradation				
Plasma	Water quality does not have an impact on process efficiency	• No chemical input • Less prone to scaling and electrode fouling • Not impacted by TOC levels • Small footprint	• Formation of aqueous and gaseous byproducts • Requires concentrated PFAS influent for higher degradation efficiency • High energy demand	Singh et al.[218] Singh et al.[219]
Supercritical Water Oxidation (SCWO)	Water quality does not have an impact on process efficiency	• No generation of major byproducts • Non-selective • Degradation of PFAS at low concentrations • Potential for energy recovery	• Can require frequent maintenance • High energy demand • Potential for scaling and corrosion • Lack of studies reporting leachate PFAS treatment	Krause et al.[12]
Hydrothermal Alkaline Treatment (HALT)	Matrices under alkaline pH conditions	• Non-selective • Effective degradation • Subcritical operation	• High scaling potential • Chemical addition • Slower reaction under neutral pH conditions	Hao et al.[220]
Ultraviolet (UV)	Acidic or alkaline condition depending on catalysts	• Degradation of SC PFAS • Higher efficiency than oxidative technologies	• Energy intensive • Pretreatment is needed for improved process efficiency and reduced fouling • Background matrix composition can interfere with PFAS degradation • Chemical or catalyst addition	Chen et al.[221] Liu et al.[222] Uwayezu et al.[223]
Electrooxidation (EO)	Lower concentration of organics can improve process efficiency	• Low relative energy requirement • Does not require additives except at low conductivity • Higher relative unit capacity • Low ecotoxicity	• Energy intensive • Expensive electrode material • Competing constituents interfere with PFAS removal • Pretreatment is needed for improved process efficiency	Garcia-Segura et al.[224] Chen et al.[225] Pierpaoli et al.[226] Urtiaga et al.[227]

effective treatment method for PFAS in liquid investigation-derived waste (IDW) and landfill leachate [228,229]. In a pilot-scale plasma reactor study, PFOA and PFOS were removed from IDW, and precursor removal increased from 36 to 99%. However, complete destruction was not achieved, as over 30 PFOS and PFOA byproducts were detected. Non-PFAS co-contaminants did not impact degradation efficiency [228].

Another lab-scale plasma reactor study treated leachate and achieved greater than 90% removal of PFOA and PFOS in 10 min. ~60% of precursors and 34% of TOC were removed [219]. The PFAS are concentrated by bubbling and degraded near the plasma-liquid interface, which explains why TOC did not interfere with degradation efficiency. Fluoride increased from 1 to 3 mg L^{-1}, indicating mineralization of PFAS occurred; the organic bound F decreased by 40% while the inorganic increased by 300% after 120 min.

The breakdown products of PFOA and PFOS during plasma treatment include PFCAs (PFBS, PFHxS), organic acids, and inorganic fluoride, as well as cyclic perfluoroalkanes in the gas phase products. While plasma is a viable technology for PFAS removal from leachate, it will not completely destroy all PFAS [218] and other byproducts can be formed during the process.

4.2.2. *Supercritical water oxidation (SCWO)*

Supercritical Water Oxidation (SCWO) is a promising treatment method for PFAS due to its ability to reduce PFAS levels to below detection limits. In this process, water enters a state referred to as "supercritical" above 374 ℃ and 22.1 MPa. Under these conditions, the solubility of organic compounds increases as does their oxidation rates. Three different companies were tested for their ability to reduce PFAS in dilute solutions of AFFF, and all three demonstration studies yielded removal greater than 99% PFAS [12]. PFOS decreased from 26 mg L^{-1} to 240 μg L^{-1} while PFOA dropped from 930 μg L^{-1} to 0.14 μg L^{-1}. The study documented removal, but the byproducts were not quantified, and a fluoride balance was not completed.

Several other studies have demonstrated the effectiveness of SCWO in removing PFAS. For example, a SCWO study employing liquid and gas analysis on PFAA in AFFF found 99.999% removal efficiency after two hours of continuous flow trials and defluorination was 60% [230]. A field demonstration of SCWO also showed greater than 99.99% PFAS removal from AFFF [231]. In another study, PFOS and PFOA were removed in SCWO, and trifluoroacetic acid was found as a main byproduct at 600 ℃. Volatile organofluorines including 1H-perfluoroalkane were detected in the gas phase. Suspected defluorination increased from 20% at 450 ℃ to over 100% at 650 ℃ [232], with temperature playing a key role in mineralization. 610 ℃ was required for 99.999% removal and more than 100% fluoride recovery. At lower temperatures, fluoride recovery was less than PFAS removal, implying incomplete destruction (mineralization) [233]. Steady-state operation resulted in 5 log reduction after 3 h, but neutralization was required before release back into the environment due to the low pH of the treated liquid [234]. In a study focused solely on PFOS treatment, it was reported that over 75% defluorination was achieved after 60 min at 50 ℃ using SCWO [235].

Matrix effects can impact SCWO. A study was conducted on SCWO treating GAC and IX waste previously used for PFAS removal. PFCAs were easier to remove than PFSAs, and long-chain PFAS removal was easier to remove than short-chain PFAS. This study only reported removal and not fluoride balance data [236]. The treatment of landfill leachate with SCWO has been studied, although not specifically with PFAS removal. Research has focused on the effectiveness of using SCWO to remove other contaminants such as ammonia. A study conducted by Scandelai et al. [237] demonstrated the effectiveness of using SCWO for the removal of ammonia from landfill leachate. Similarly, a study by Wang et al. [238] investigated the removal of ammonia, TOC, and total nitrogen from landfill leachate using SCWO. These studies highlight the potential of SCWO as a treatment method for landfill leachate and other wastewater streams contaminated with various pollutants, although additional research is necessary to determine its effectiveness in removing PFAS from landfill leachate.

4.2.3. Hydrothermal alkaline treatment (HALT)

Hydrothermal alkaline treatment (HALT) has shown promise in the defluorination of PFAS. Wu et al. [239] found that hydrothermal treatment with NaOH (200–350 ℃, 2–16.5 MPa) was capable of defluorinating PFOS, with complete conversion of C-F bonds to inorganic fluoride observed after 40 min at 1 M NaOH.

A study on AFFF found rapid degradation of PFAS at 350 ℃, 16.5 MPa using 1–5 M NaOH. Most PFAS fell below detection limits after 15 min, and even the most recalcitrant PFAS were degraded in 30 min with 5 M NaOH [240]. HALT has also shown promise in removing PFAS from soil and groundwater matrices, with cationic, anionic, and zwitterionic PFAS reduced to nondetectable levels after 90 min using 5 M NaOH [241]. Another study reported greater than 99% PFAS removal from an aqueous matrix with high total dissolved solids (10% by wt) at 350 ℃ using 16% NaOH [241]. However, complete defluorination did not occur and short-chain PFSAs were more recalcitrant. It is worth noting that reactions in matrices with base-neutralizing chemicals were slower, indicating potential leachate matrix effects with this treatment [220].

HALT can be paired with other removal technologies for even greater efficiency. For example, FF can be used to concentrate PFAS from contaminated groundwater, which can then be treated with HALT. Hao et al. [242] found that 90% defluorination was achieved within 90 min using 1 M NaOH at 350 ℃. Interestingly, elevated levels of salt (325 mg L^{-1} chlorides) and dissolved organic carbon (4.5 g L^{-1}) did not impact PFAS degradation kinetics [242].

It is also possible to augment HALT with additional materials. For instance, the addition of biochar and zero-valent iron increased PFOA reaction kinetics and drove defluorination efficiency to 99% [243]. However, it is worth noting that HALT is still a nascent technology that requires testing at pilot-scale and full-scale before widespread availability for landfill leachate treatment. Furthermore, HALT requires a substantial amount of external chemicals due to the need for chemical base addition. As mentioned earlier, the presence of organics, even at a high concentration (e.g., 4.5 g L^{-1}) might not impede PFAS removal kinetics using HALT [242]. Thus, HALT holds promise as a potentially effective treatment option for PFAS in high-strength wastewater matrices, e.g., leachate, as the process effectiveness of HALT might not be impacted by organics. However, more research is needed to better understand its potential for wider use and to optimize its efficiency.

4.2.4. Electrochemical oxidation (EO)

Electrochemical oxidation, also known as electrooxidation (EO), employs electrodes for *in situ* treatment of water without the need for external chemical addition. This process oxidizes compounds either directly on the electrode surface or through oxidants in solution that were generated from electrochemical reactions [244]. A benefit to this process is that water can be treated without additional chemicals. A drawback to this process is that the materials used for electrodes can be expensive (e.g., made of diamond) and/or composed of rare earth metals.

Much research has been conducted on electrochemical oxidation for PFAS remediation, as reviewed previously [244]. Electrochemical oxidation of PFAS primarily occurs at the electrode surface [245,246]. Importantly, leachate that contains substantial amounts of organic material blocking the pathway of PFAS to the electrode would inhibit treatment. A plethora of research has been conducted on the electrochemical treatment of PFAS in drinking water. However, a lot of this research was done using PFAS levels that were substantially higher than environmentally relevant levels and on lab-concocted waters that had an absence of oxidant scavengers [32]. Still, several papers have documented the mineralization of PFAS during electrooxidation [247-251], indicating that, with the proper conditions and advancements, electrooxidation could be a viable technology for leachate treatment. EO treatment of PFAS using boron doped diamond (BDD) can even be enhanced by the presence of sulfur compounds that promote electrochemically activated sulfate-based radicals [252,253]. Thus, leachate with a high presence of sulfur might be easier to treat for PFAS.

Some research has been conducted specifically on the treatment of PFAS in leachate at the lab scale [226]. Over 75% removal could be achieved, but only after eight hours of electrooxidation, which is rather long for EO treatment, and therefore expensive [226]. When a membrane bioreactor was used to pre-treat landfill leachate, EO was able to remove over 95% of PFAS, again suggesting that EO is an appropriate polishing treatment [227]. Similarly, FF was used to concentrate PFAS before treatment by EO for landfill leachate. This process resulted in over 80% removal of long-chain PFAS and 30% removal of short-chain PFAS [254]. Importantly though, landfill leachates contain a wide range of precursor compounds, and EO has been shown to generate PFCAs from the transformation of precursors [75].

4.2.5. UV-based treatment

UV-based treatment methods have historically been limited in their ability to effectively degrade PFAS due to the strong energy required to achieve bond fission of the C-F bonds [255]. However, recent studies have shown promising results with the use of 185 nm wavelength irradiation to generate hydrated electrons, assisted with perchloric acid to facilitate electron transfer and improve PFOA degradation under acidic conditions [221]. The effective use of UV-based technologies for PFAS degradation involves the use of a light source, oxidant, and catalyst to accelerate the degradation of PFAS [256]. Specifically, the addition of titanium dioxide (TiO_2) or titanium nanotubes (TNT) as catalysts in UV/metal-TiO_2 or UV/TNT systems has proven to be an effective method for PFAS degradation [257]. The main benefit of these methods can help degrade short- and ultra-short chain PFAS (C1-C5) and can accommodate a wide range of applications.

One of the benefits of UV-based technologies includes the degradation of short- and ultra-short chain PFAS (C1-C5) The most common method for photocatalytic reactions using UV involves the addition of sulfite, with 185 nm irradiation proving to be the most effective spectral region [222]. Under basic conditions (e.g., pH 11.2), the presence of sulfite photosensitizer enhances these reactions, where PFOS degradation dramatically increased by approximately 12 times when pH increased from 9.5 to 11.2 [222]. The major sulfite byproducts include sulfate radicals ($SO_4^{\bullet-}$), in which aqueous electrons (e_{aq-}) react with PFAS through an advanced reduction process. Another chemical used as a photosensitizer is iodide, which generates e_{aq-} but can also produce iodide-containing byproducts that have been linked to adverse health effects [258]. A recent two-step oxidation and reduction technique has been developed by Zeidabadi et al. [259] that involves anodic electrodes layered with carbon nanotubes (CNT) modified by cetyltrimethylammonium bromide (CTAB) to facilitate the adsorption of hydrophobic PFAS followed by oxidoreduction of the target compound. However, the presence of organic co-contaminants can initially be unfavorable for PFAS degradation, as they are often present in higher concentrations and can overshadow the degradation of PFAS in leachate. Therefore, UV-based treatments are better suited for leachate after pre-treatment step to reduce organic content. Nonetheless, UV-based treatment methods offer a promising solution for the remediation of PFAS-contaminated environments but require additional research for more targeted degradation.

The potential of all these technologies for PFAS removal is clear. However, they do have practical limitations that also need to be addressed.

5. Practical implications of treatment technologies for PFAS removal in leachate

5.1. Technology matrix

Table 3 presents the suitability of PFAS-specific treatment technologies reviewed in this work. The categories, including capital

Table 3
Evaluation matrix of various PFAS treatment technologies.

Technology	GAC	IX	RO	Foam Fract.	EO	SCWO	HALT	Plasma
Type	**Separation**	**Separation**	**Separation**	**Concentration**	**Degradation**	**Degradation**	**Degradation**	**Degradation**
CAPEX ($-$$$$)	$$	$$	$$$$	$$	$$$	$$$$	$$$$	$$$$
OPEX ($-$$$$)	$$$	$$$	$$$$	$	$$	$$$	$$	$$$
Applicability for Landfill Leachate[a]	1	2	3	5	4	4	4	5
Level of Technological Readiness	5	5	5	2	3	2	1	1
Energy Consumption	5	5	3	4	3	1	2	1
Chemical Addition	5	5	5	4	1	2	2	3
Treatment Capacity	5	5	4	4	3	2	2	1
Long-chain PFAS Removal	4	5	5	5	5	5	4	5
Short-chain PFAS Removal	1	5	5	1	2	5	4	1
Gas Emissions	5	5	5	5	3	1	2	2
Aqueous Byproducts	5	5	4	5	3	3	4	3

[a]*Technological capability of different processes for PFAS treatment in complex landfill leachate matrices.*

Readiness Factor	Treatment Criteria
5	The technology is highly suitable for the category
4	The technology is appropriate
3	The technology is adequate with minor improvements
2	Greater attention is required
1	The issue defined in the category should be carefully addressed before implementation

Table 4
Summary of PFAS removal efficiency, energy requirements and limitations for the four treatment trains.

Configuration	Removal efficiency	Energy Requirements*	Limitations
TT1: NF, UV-sulfite treatment	PFCAs: 95 −99% PFSAs: 88 −99%	NF: Medium UV: High	- Slow degradation of shorter PFAS - Cooling required for UV treatment - Cooling systems require optimization. - High maintenance for NF membrane
TT2: FF, EO, polishing resin	Long-chain PFAS: > 92% Short-chain PFAS: 30 −90%	FF: Low EO: Medium-High Polishing Resin: Low	- TOC enrichment hinders FF energy - EO forms oxyhalides needing polish - Issues for EO with short PFAS
TT3: IX, polishing step, regeneration, distillation, EO	Short-chain PFAS: −10 −50% Long-chain PFAS: 70 −95%	IX: Low EO: Medium-High Distillation: Medium-High	- EO produces short-chain PFAS - Optimization EO treatment required - Management of EO waste with recirculation
TT4: Ozone/UV-fractionation	Short-chain PFAS: −20 −40% Long-chain PFAS: > 90%	Ozone: Medium-High UV: High	- UV alone not effective for PFAS removal - Destructive treatment increases short-chain PFAS - Air/ozone separation unable to remove short-chain PFAS

PFCAs = Perfluoroalkyl Carboxylic Acid
PFSAs = Perfluoroalkyl Sulfonic Acids
*This rating was estimated on generic technology requirements but implications at energy requirement may vary considerably depending on the facility.

expenditure (CAPEX) and operational expenditure (OPEX), represented by a scale (e.g., $ for lower cost to $$$$ for higher costs), as well as readiness factors ranging from 1 to 5, where 1 indicates the least ready and 5 indicates the most ready. These assessments were based on information provided by manufacturers and several PFAS peer-reviewed studies. Non-quantitative parameters, such as level of technological readiness, are defined relative to each other. The technology evaluation included the following:

- Applicability for landfill leachate: Depends on the technology's ability to resist inorganic and organic fouling (e.g., scaling, headloss buildup), the degree to which PFAS removal efficiency is unaffected by competing constituents (e.g., TOC, inorganic anions), and the level of targeted removal of PFAS.
- Total Readiness Level (TRL): Determined based on the approach detailed in Berg et al. [20] and Tow et al. [16].
- Energy consumption: Estimated based on individual anticipated ranges of energy based on literature, with foam fractionation ranging from 0.6 to 0.8 kWh m^{-3} [260] to plasma treatment from 1560 to 2370 kWh m^{-3} [22].
- Chemical requirement: Evaluates the addition of chemical based on the dosage needed to optimize PFAS treatment, quench fluorine byproducts, or sustain the operation of the unit (e.g., co-fuel).
- Treatment capacity: Considers both the configuration (e.g., batch, semi-batch, and continuous) and the capability of the process unit to scale up and accommodate higher feed flow.
- Short-chain and Long-chain PFAS removal: Highlights the potential limitations of some processes to treat short-chain PFAS due to their different properties compared to long-chain PFAS (e.g., higher solubility, less surface activity, and the potential to degrade long-chain PFAS to shorter counterparts through stepwise degradation).
- Gas emissions: Consider the potential hazardous air emissions released during treatment, such as greenhouse gases (GHG), incomplete volatile PFAS, and other aerosols.
- Aqueous byproducts: Considers potential byproducts that may be released or remain in the effluent (e.g., perchlorate, disinfection byproducts, PFAS byproducts).

5.2. Treatment trains

The following four treatment trains and configurations are presented as options for successful removal of PFAS for effective management of PFAS-laden waste. While they have not all been directly tested for leachate treatment, these are believed to be applicable. Table 4 summarizes PFAS removal efficiency, energy requirements, and limitations for the treatment trains (TT), while the following list and Fig. 4a through Fig. 4d provide a more in-depth description of them.

5.2.1. TT1: NF → UV-sulfite treatment

Liu et al. [222] used the combination of NF with UV to achieve PFAS degradation (Fig. 4a). This approach had the following pros and cons:

TT1 Pros

- The hybrid NF/UV-sulfite treatment train can achieve good rejection of most PFAS at high permeate recoveries, resulting in clean and safe water.

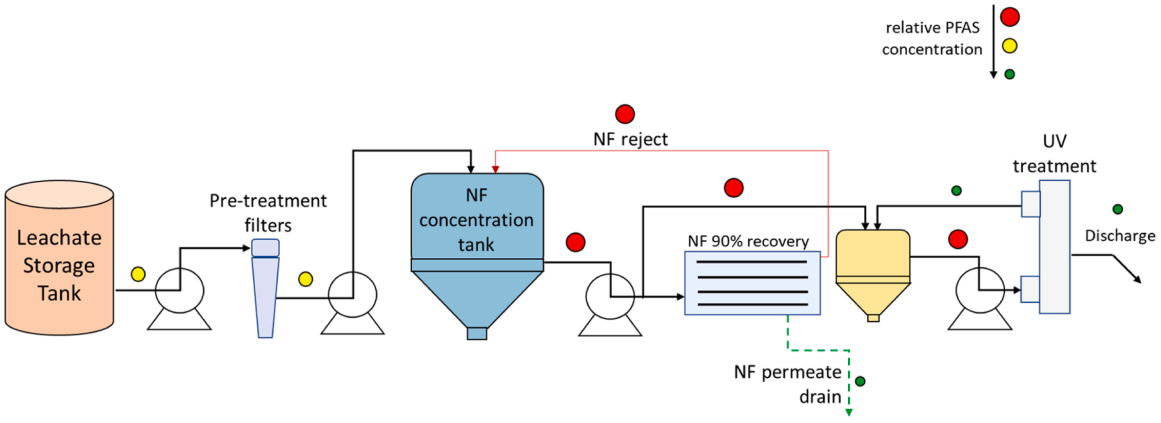


Fig. 4a. Simplified process flow diagram of NF/UV-sulfite pilot system based on Liu et al. [222].

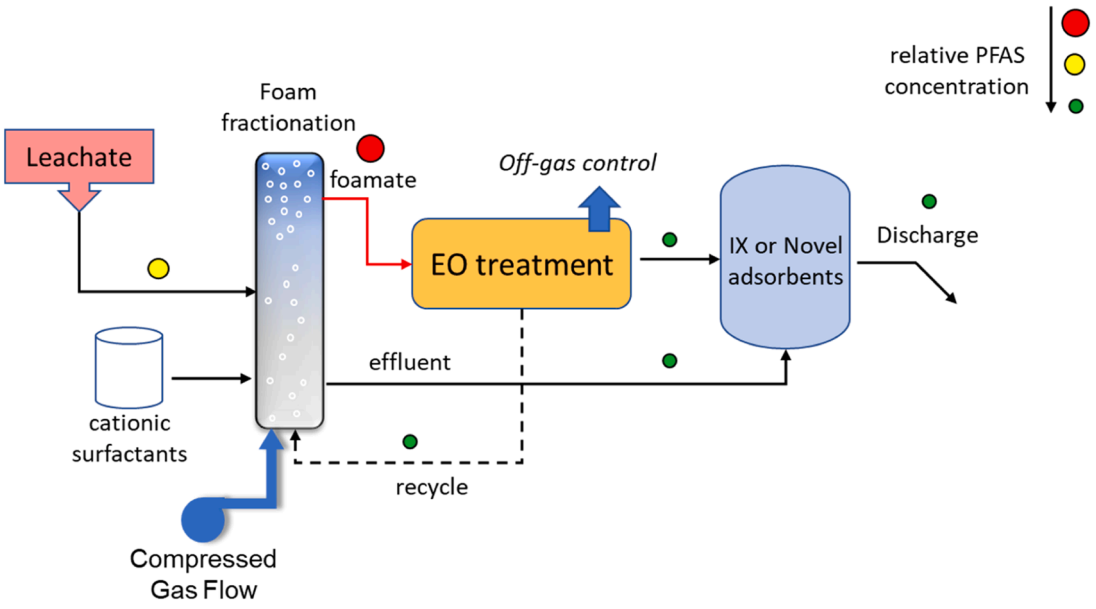


Fig. 4b. Schematic of the pilot-scale FF-EO treatment train. studied by Wang et al. [261]

- The resulting NF reject stream concentrates PFAS, reducing the volume of water that needs further treatment.
- The technology uses sulfite photosensitizer to generate e_{aq-} that degrade PFAS.

TT1 Cons

- Shorter-chain PFSAs and 6:2 fluorotelomer sulfonate (6:2 FTS) are degraded much more slowly than other PFASs.
- The process requires UV treatment, which may need cooling requirements and results in high energy inputs.
- Optimization of passive cooling systems is recommended to reduce energy requirements further.
- High maintenance requirement of NF membrane due to the potential for fouling.

5.2.2. TT2: FF → EO → polishing resin

Wang et al. [261] used FF coupled with electrooxidation to remove PFAS (Fig. 4b). This approach had the following pros and cons:

TT2 Pros

- The FF-EO treatment train is a promising technology for PFAS removal and is more energy-efficient than EO treatment alone.
- High removal of PFAS by FF leading to more concentrated foam is conducive to the subsequent EO treatment for PFAS destruction.
- The study revealed significant variables for enrichment and removal, which could be helpful in developing other separation-destruction treatment trains, such as NF-Plasma.
- The FF-EO treatment train proved overall effective and energy-efficient in treating PFAS in environmental samples.
- Concentrating the influent PFAS into foam reduced the energy requirement.

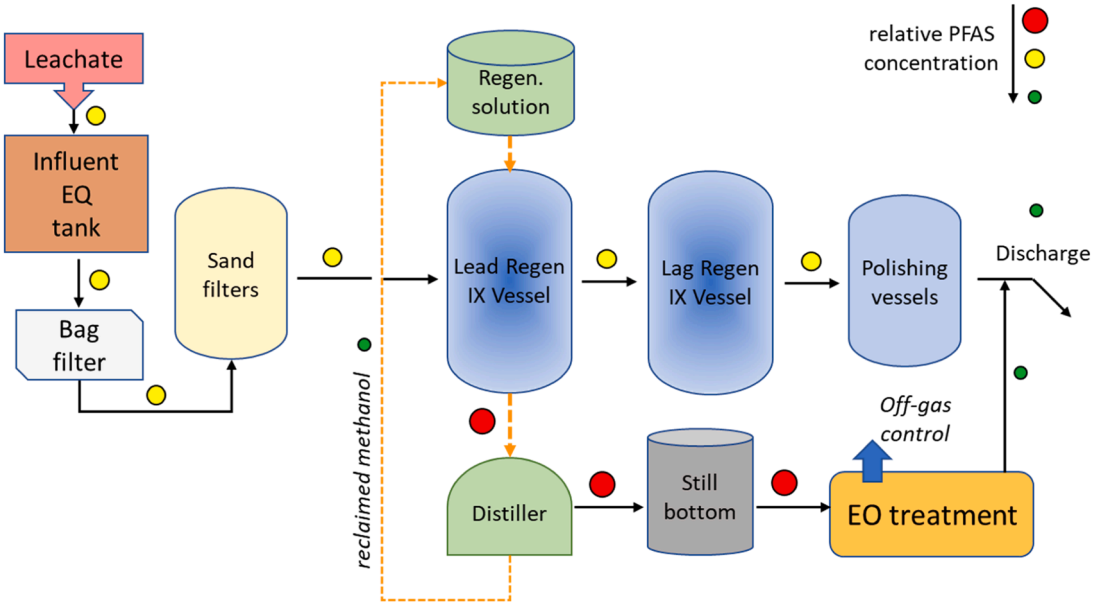


Fig. 4c. Schematic of the pilot-scale IX-EO treatment train studied by Liang et al. [262].

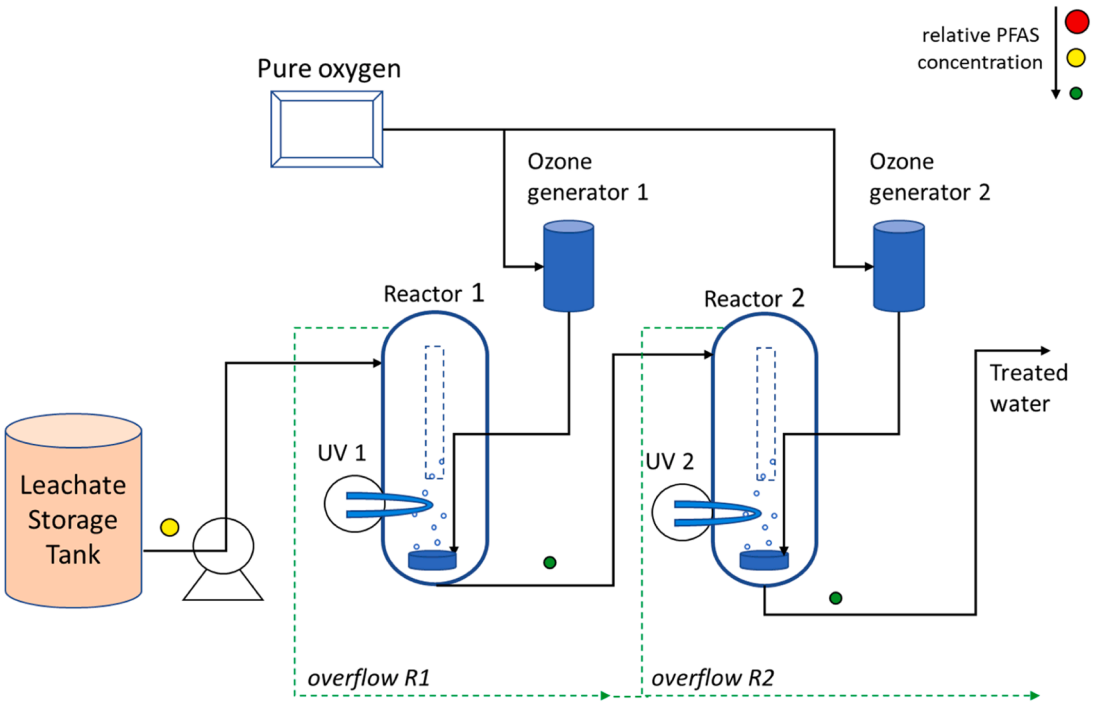


Fig. 4d. Schematic diagram of the benchtop setup used by Dai et al. [263] to study ozone-fractionation.

TT2 Cons

- The enrichment of TOC during FF treatment can hinder the energy efficiency.
- EO treatment may form oxyhalide anions (e.g., bromate, chlorate), which necessitates a polishing step in a full-scale application.
- EO treatment has less efficiency towards short-chain PFAS, and more research is needed to explore ways to improve the efficiency of treating short-chain PFAS.

5.2.3. TT3: IX → polishing step → regeneration → distillation → EO

Liang et al. [262] evaluated the use of IX and EO for PFAS remediation. This approach had the following pros and cons: (Fig. 4c)

TT3 Pros

- EO leverages the elevated salinity levels present in the still bottom to improve system conductivity. This, in turn, reduces the voltage required to effectively degrade PFAS.
- The technology has a much lower energy requirement than standalone EO treatment, making it more energy efficient.
- The study demonstrated mineralization of PFAS precursors, long-chain PFAAs, and short-chain PFAAs in a highly concentrated PFAS waste stream generated from dilute water.

TT3 Cons

- EO can produce short chain PFAS from breakdown of long chain PFAS.
- There is a need for additional research on process optimization to suppress the potential formation of oxidation byproducts during EO treatment.
- Current configuration requires proper management of EO waste. The authors propose implementing a closed-loop recirculation system for the EO effluent as a best practice management strategy.

5.2.4. TT4: Ozone/UV-fractionation

Dai et al. [263] studied and compared UV, ozonated air fractionation, air fractionation and combined UV/ozone treatment for PFAS removal (Fig. 4d). This approach had the following pros and cons:

TT4 Pros

- Ozonated air fractionation achieved the best performance, removing about 95% of PFAS from the treated water, making it an effective technology for PFAS removal.
- The air fractionation removed 81% of PFAS while UV/ozone combined treatment achieved 73% removal of PFAS, which shows that these technologies can also be effective in PFAS removal.
- The involvement of gas bubbles makes it relatively easier to remove PFSA than PFCA by both the air and ozonated air fractionations and the UV/ozone combined treatment, which is attributed to the greater lipophilicity of PFSA.

TT4 Cons

- UV alone did not show promising performance for PFAS removal.
- Short-chain PFAS concentration increases after treatment when PFAS destruction is involved, which is a limitation of the technology.
- Both the air and ozonated air fractionations could not effectively partition the short-chain PFAS from the contaminated water into the foam, due to the exponential decline of the partition factor with the reducing carbon number of the PFAS.

6. Future directions

Landfill leachate contains a complex mixture of PFAS and is a known contributor of PFAS loads to municipal WWTPs [264]. To effectively characterize PFAS in landfill leachate, a multiplatform approach is recommended. However, it is important to consider the trade-offs between sample throughput and analytical complexity. From a practical perspective, targeted LC-MS/MS remains the gold standard for quantifying currently known PFAS, and should be optimized to include ultrashort-chain PFCA given their prevalence in landfill leachate. In addition, the EOF assay and NTA (enabled by LC-HRMS) should be implemented following best practices to quantify total organofluorine

and to identify unknown PFAS when possible. Lastly, the TOP assay can be applied to estimate oxidizable PFAA precursor levels in landfill leachate, especially if the assay has been expanded to include cationic and zwitterionic species. Regardless of the techniques chosen, standardizing a matrix-specific sample extraction protocol is critical as an initial step to minimize PFAS loss or transformation, while balancing method selectivity and sensitivity Fig. 4d.

PFAS treatment technologies have advanced significantly in recent years, demonstrating promising applicability for commercial use. Despite this progress, challenges remain, including high energy consumption, high-maintenance requirements for high-strength matrices, reduced performance in the presence of background contaminants, and the production of harmful byproducts. To optimize treatment, future research should focus on testing different treatment train configurations in a side-by-side comparison of these parameters. Emerging technologies offer creative combinations that have yet to be tested on a larger scale, as most current combinations have only been utilized at lab-scale and for matrices with much lower strength under facile conditions compared to leachate (e.g., groundwater). Another practical aspect that requires exploration is the impact of operating conditions and the ability to quickly adapt treatment processes to varying influent leachate characteristics, given the wide variability of leachate. For example, influent leachate with a sudden peak in organics may increase the scavenging potential of degradation technologies [14], while more "dilute" leachate, may reduce the foaming effectiveness of FF [205]. The limiting capacity of treatment units has been a significant challenge for large treatment plants. For example, SCWO, HALT, and plasma treatment units have been restricted to a capacity of 0.6 m^3 h^{-1} (2.6 gpm), while concentration technologies like FF have been limited to 25 m^3 h^{-1} (110 gpm). Advancing the scalability of treatment technologies to handle larger flow rates is essential for progress in this field.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Data availability

Data will be made available on request.

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From:	Pamela Castle
To:	Benton Public Comment
Subject:	PFAS burden Pam Castle LU-24-027 - reference batch 4
Date:	Wednesday, June 11, 2025 10:49:02 AM
Attachments:	EPA Announces It Will Keep Maximum Contaminant Levels for PFOA, PFOS US EPA.pdf EPA fact sheet WWTP sewage sludge risk.pdf EPA fact-sheet-draft-sewage-sludge-risk-assessment-pfoa-pfos.pdf EPA landfills effluent (aka leachate).pdf Corvallis Biosolids.pdf

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This is the last batch of referenced materials.

Thank you,

Pam

EPA Announces It Will Keep Maximum Contaminant Levels for PFOA, PFOS

EPA intends to provide regulatory flexibility and holistically address these contaminants in drinking water

May 14, 2025

Contact Information
EPA Press Office (press@epa.gov)

WASHINGTON – U.S. Environmental Protection Agency (EPA) Administrator Lee Zeldin announced the agency will keep the current National Primary Drinking Water Regulations (NPDWR) for perfluorooctanoic acid (PFOA) and perfluorooctane sulfonic acid (PFOS) <https://epa.gov/sdwa/and-polyfluoroalkyl-substances-pfas>, which set nationwide limits for these "forever chemicals" in drinking water. The agency is committed to addressing Per- and Polyfluoroalkyl substances (PFAS) in drinking water while following the law and ensuring that regulatory compliance is achievable for drinking water systems.

"The work to protect Americans from PFAS in drinking water started under the first Trump Administration and will continue under my leadership," ***said EPA Administrator Zeldin****. "We are on a path to uphold the agency's nationwide standards to protect Americans from PFOA and PFOS in their water. At the same time, we will work to provide common-sense flexibility in the form of additional time for compliance. This will support water*

systems across the country, including small systems in rural communities, as they work to address these contaminants. EPA will also continue to use its regulatory and enforcement tools to hold polluters accountable."

As part of this action, EPA is announcing its intent to extend compliance deadlines for PFOA and PFOS, establish a federal exemption framework, and initiate enhanced outreach to water systems, especially in rural and small communities, through EPA's new PFAS OUTreach Initiative (PFAS OUT). This action would help address the most significant compliance challenges EPA has heard from public water systems, members of Congress, and other stakeholders, while supporting actions to protect the American people from certain PFAS in drinking water.

Paired with effluent limitations guidelines (ELGs) for PFAS and other tools to ensure that polluters are held responsible, EPA's actions are designed to reduce the burden on drinking water systems and the cost of water bills, all while continuing to protect public health and ensure that the agency is following the law in establishing impactful regulations such as these.

EPA is also announcing its intent to rescind the regulations and reconsider the regulatory determinations for PFHxS, PFNA, HFPO-DA (commonly known as GenX), and the Hazard Index mixture of these three plus PFBS to ensure that the determinations and any resulting drinking water regulation follow the legal process laid out in the Safe Drinking Water Act.

Regulatory Protection with Flexibility and Cost Savings

On April 10, 2024, EPA announced the final National Primary Drinking Water Regulation, including standards for PFOA and PFOS. At that time, EPA established legally enforceable levels for these PFAS in drinking water and gave public water systems until 2029 to comply with the Maximum Contaminant Levels (MCLs).

To allow drinking water systems more time to develop plans for addressing PFOA and PFOS where they are found and implement solutions, EPA plans to develop a rulemaking to provide additional time for compliance, including a proposal to extend the compliance date to 2031. EPA plans to issue a proposed rule this fall and finalize this rule in the Spring of 2026. Aligned with the agency's intent to provide additional compliance time for water systems, EPA encourages states seeking primacy for

implementing the PFAS drinking water regulation to request additional time from EPA to develop their applications. At the same time, EPA will support the U.S. Department of Justice in defending ongoing legal challenges to the PFAS National Primary Drinking Water Regulation with respect to PFOA and PFOS.

"EPA has done the right thing for rural and small communities by delaying implementation of the PFAS rule. This commonsense decision provides the additional time that water system managers need to identify affordable treatment technologies and make sure they are on a sustainable path to compliance. NRWA greatly appreciates this reasonable and flexible approach, and we look forward to partnering with the agency's PFAS OUTreach Initiative to help ensure water systems have the resources and support they need," ***said National Rural Water Association CEO Matthew Holmes.***

"ASDWA supports EPA's proposed approach to the PFAS regulation to extend the compliance date for systems by an additional two years. With the current compliance date of 2029, states and water systems are struggling with the timeframes to complete the pilot testing, development of construction plans, and building the necessary treatment improvements. EPA's proposed extension of the compliance date and increased technical assistance will address the number of systems that would be out of compliance in 2029 due to not being able complete all of these tasks on time," ***said Association of State Drinking Water Administrators Executive Director Alan Roberson.***

Enhancing Communication and Outreach

To enhance engagement on addressing PFAS, EPA will launch PFAS OUT to connect with every public water utility known to need capital improvements to address PFAS in their systems, including those EPA has identified as having PFOA and PFOS levels above EPA's MCL. EPA will share resources, tools, funding, and technical assistance to help utilities meet the federal drinking water standards. PFAS OUT will ensure that no community is left behind as we work to protect public health and bring utilities into compliance with federal drinking water standards. PFAS OUT will engage utilities, technical assistance providers and local, State, Tribal, and Territorial leaders to develop effective, practical solutions where they are needed most.

EPA will continue to offer free water technical assistance (WaterTA) <https://epa.gov/water-infrastructure/water-technical-assistance-waterta> that provides services to water systems to improve their drinking water and help communities access available funding. EPA's

WaterTA initiatives work with water systems nationwide to identify affordable solutions to assess and address PFAS, including PFOA and PFOS. Services offered to utilities include water quality testing, development of technical plans, operator training support, designing public engagement and outreach strategies, and support for accessing federal funding opportunities.

Holding Polluters Accountable

Drinking water systems are passive receivers of PFOA and PFOS. Polluters can contaminate the surface waters or aquifers that these systems rely on to provide the drinking water to their communities. As announced by Administrator Zeldin <https://epa.gov/newsreleases/administrator-zeldin-announces-major-epa-actions-combat-pfas-contamination>, EPA intends to take a number of actions to reduce the prevalence of PFAS in the environment, including in sources of drinking water. Progress reducing concentrations of PFAS in drinking water sources can substantially reduce the cost burden for water systems and reduce the cost of living for the communities they serve.

A Record of Leadership

Administrator Zeldin's leadership on PFAS dates back to his time in Congress, where he was a founding member of the PFAS Congressional Taskforce and a strong supporter of the PFAS Action Act, legislation to provide funding to support local communities cleaning up PFAS-contaminated water systems. He was, and remains, a staunch advocate for protecting Long Islanders and all Americans from contaminated drinking water.

In the process of developing and taking action on a number of these items, Administrator Zeldin personally heard from members of Congress on passive receiver issues where local water utilities will foot the bill for contamination and pass those costs onto consumers. This mindset and the need for a polluter pays model has guided a lot of the work to be done at EPA in the future.

Background

On April 28, 2025, Administrator Zeldin announced a long list of actions to combat PFAS contamination <https://epa.gov/newsreleases/administrator-zeldin-announces-major-epa-actions-combat-pfas-contamination> that included in part the designation of an agency lead for PFAS, the development of ELGs for certain PFAS to reduce discharges to waterways including

upstream of drinking water systems, and initiatives to engage with Congress and industry to establish a clear liability framework that ensures passive receivers and consumers are protected. This list is the first, not the last, of all decisions and actions EPA will be taking to address PFAS over the course of the Trump Administration. There will be more to come in the future across EPA's program offices to help communities impacted by PFAS contamination.

During President Trump's first term, EPA convened a two-day National Leadership Summit on PFAS <https://epa.gov/pfas/2018-pfas-national-leadership-summit> in Washington, D.C. that brought together more than 200 federal, state, and local leaders from across the country to discuss steps to address PFAS. Following the Summit, the agency hosted a series of visits during the summer of 2018 in communities <https://epa.gov/pfas/2018-pfas-community-engagement> directly impacted by PFAS. EPA interacted with more than 1,000 Americans during community engagement events in Exeter, New Hampshire, Horsham, Pennsylvania, Colorado Springs, Colorado, Fayetteville, North Carolina, and Leavenworth, Kansas, as well as through a roundtable in Kalamazoo, Michigan, and events with tribal representatives in Spokane, Washington.

In 2019, the Trump EPA announced the PFAS Action Plan <https://epa.gov/sites/default/files/2019-02/documents/pfas_action_plan_021319_508compliant_1.pdf>. This historic Plan responded to extensive public interest and input the agency received and represented the first time EPA built a multi-media, multi-program, national communication and research plan to address an emerging environmental challenge like PFAS. EPA's Action Plan identified both short-term solutions for addressing these chemicals and long-term strategies that will help provide the tools and technologies states, tribes, and local communities need to provide clean and safe drinking water to their residents and to address PFAS at the source—even before it gets into the water.

EPA supports water systems in reducing PFAS and emerging contaminants (EC) in drinking water through a range of funding resources. Federal funding opportunities include the Drinking Water State Revolving Fund (DWSRF) program, the EC Small or Disadvantaged Communities (EC-SDC) grant program, and funding resources, like EPA's Water Infrastructure Finance and Innovation Act (WIFIA) program <https://epa.gov/wifia>, that can be leveraged to provide supplemental, flexible, low-cost credit assistance to public and private borrowers.

For information about the PFAS Rule, visit Final PFAS National Primary Drinking Water Regulation <https://epa.gov/sdwa/and-polyfluoroalkyl-substances-pfas> and Per- and Polyfluoroalkyl Substances (PFAS) NPDWR Implementation <https://epa.gov/dwreginfo/pfas-rule-implementation>. For more information about PFAS Technical Assistance, visit EPA Water Technical Assistance <https://epa.gov/water-infrastructure/water-technical-assistance-waterta>. You can also Request EPA WaterTA services <https://epa.gov/water-infrastructure/forms/water-technical-assistance-request-form> for your community.

Last updated on May 14, 2025

FACT SHEET

Draft Sewage Sludge Risk Assessment for PFOA and PFOS: Information for Wastewater Treatment Plants

January 2025

This fact sheet contains information that may be useful to operators of wastewater treatment plants (WWTPs) in addressing perfluorooctanoic acid (PFOA) and perfluorooctane sulfonic acid (PFOS) contamination in sewage sludge.

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. 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 (*see* the EPA's Final Toxicity Assessment for PFOA and Final Toxicity Assessment for PFOS). PFAS manufacturers voluntarily phased out domestic manufacturing of PFOA and PFOS and their uses have been restricted by Significant New Use Rules (SNURs)

issued under the Toxic Substances Control Act (TSCA) (*see* the EPA's Risk Management for PFAS under TSCA). Though 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.

Learn more about PFAS, the EPA's PFAS Strategic Roadmap, and PFAS exposure in impacted communities.

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.[1] 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 that can contribute PFOA and PFOS to sewage sludge 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, 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 might play a role in PFAS entering the plant and contaminating 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 concentration of pollutants found in sewage sludge varies across space and time, depending on industrial and other inputs to individual WWTPs. The presence of a pollutant in sewage sludge alone does not necessarily mean that there is risk to human health or the environment from its use or disposal. 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.

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

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 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 is the recommended analytical method to measure PFOA and PFOS in sewage sludge?

The EPA recommends using EPA Method 1633 to measure 40 PFAS analytes, including PFOA and PFOS, in sewage sludge. EPA Method 1633 finished multi-laboratory validation and was finalized in January 2024. It is planned to be included in the upcoming Methods Update Rule 22, which was proposed in late 2024.

Learn more about EPA Method 1633 and Methods Update Rules.

What plans exist for PFAS monitoring in sewage sludge nationwide?

The EPA is currently planning the next National Sewage Sludge Survey (NSSS) in collaboration with the POTW Influent PFAS Study. The NSSS will focus on obtaining current national occurrence and concentration data for 40 target PFAS analytes using EPA Method 1633. The data generated by the NSSS will help inform future risk assessments and risk management actions for sewage sludge. A Voluntary Data Submission Portal also will be available throughout the duration of the POTW Influent PFAS Study and NSSS to collect more PFAS data nationwide.

Learn more about the National Sewage Sludge Survey and the POTW Influent PFAS Study.

What does this mean for WWTPs?

The draft risk assessment is not a regulation and does not compel action. The EPA's draft risk assessment indicates that each of the three common use or disposal options may result in elevated risk levels when sewage sludge with typical concentrations of PFOA or PFOS is managed. With the understanding that eliminating these risks is likely not possible at this time, the EPA recommends, in addition to pretreatment to reduce PFAS at the

source, that WWTPs consider management options or practices that can mitigate or lessen risks. The EPA recognizes that WWTPs may have constrained options for sewage sludge management and changes may not be possible, particularly in the near term. The EPA recommends working with your state and regional biosolids coordinators for support in sewage sludge management planning.

The EPA is continuing to recommend that WWTPs 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 prevent 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." 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 and the environment.

WWTPs may choose to evaluate whether additional risk mitigation actions are appropriate to reduce risk posed by certain sewage sludge use and disposal activities. To reduce potential risk associated with land application, consider land-applying in areas that may be less sensitive to PFOA and PFOS pollution, like areas far from fishable waters or with deep protected drinking water aquifers. Consider avoiding land application in fields used to graze livestock or grow feed, especially for dairy cows. Fields used to grow fruits and grain may be better alternatives to those growing hay or leafy greens like spinach or kale. To reduce potential risk associated with surface disposal of sewage sludge consider using disposal sites with composite liners and leachate collection and treatment systems (understanding how that leachate will be disposed or treated). To better understand potential risks from incineration of sewage sludge consider performance testing incinerators to gain information about potential releases of PFOA, PFOS, and other PFAS that may be generated through incomplete combustion. For example, a recently released air method, OTM-50, can help test emissions for more volatile products of incomplete combustion, in addition to using OTM-45 to monitor for PFAS emissions. For more information, please refer to the EPA's 2024 Interim Guidance on the Destruction and Disposal of PFAS and Materials Containing PFAS.

Are there innovative technologies available to remove and destroy PFAS in sewage sludge?

There are several emerging PFAS destruction technologies (*e.g.*, supercritical water oxidation, plasma gasification, pyrolysis and gasification coupled with a high-temperature thermal oxidizer) for sewage sludge. Most are still in the pilot-scale stage and further research is needed to evaluate potential products of incomplete destruction and capacity limitations. The EPA's 2024 Interim Guidance on the Destruction and Disposal of PFAS and Materials Containing PFAS discusses ORD's PFAS Innovative Treatment Team (PITT)'s research on innovative technologies, and includes a technology evaluation framework for further assessing emerging technologies.

Learn more about the EPA's ORD PITT research effort on innovative PFAS technologies.

Learn more about funding opportunities for capital projects to treat emerging contaminants through the Clean Water State Revolving Fund Emerging Contaminants.

What are the EPA's next steps after the final risk assessment is released?

After the public comment period has closed, the EPA will consider the comments received, revise the draft risk assessment as appropriate, and prepare a final risk assessment. The final risk assessment will help inform the

EPA's potential future regulatory actions under the Clean Water Act (CWA). If the final risk assessment indicates that there are risks above acceptable thresholds when using or disposing of sewage sludge, the EPA expects to propose a regulation under CWA section 405 to manage PFOA and/or PFOS in sewage sludge to protect public health and the environment. During the risk management deliberation process, the results of the final risk assessment may be integrated with other considerations, such as economic costs and treatment feasibility, to reach decisions regarding the need for and practicability of implementing various risk reduction activities.

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

Review the EPA's Frequently Asked Questions on the Draft Sewage Sludge Risk Assessment for PFOA and PFOS.

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

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

[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.

[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.

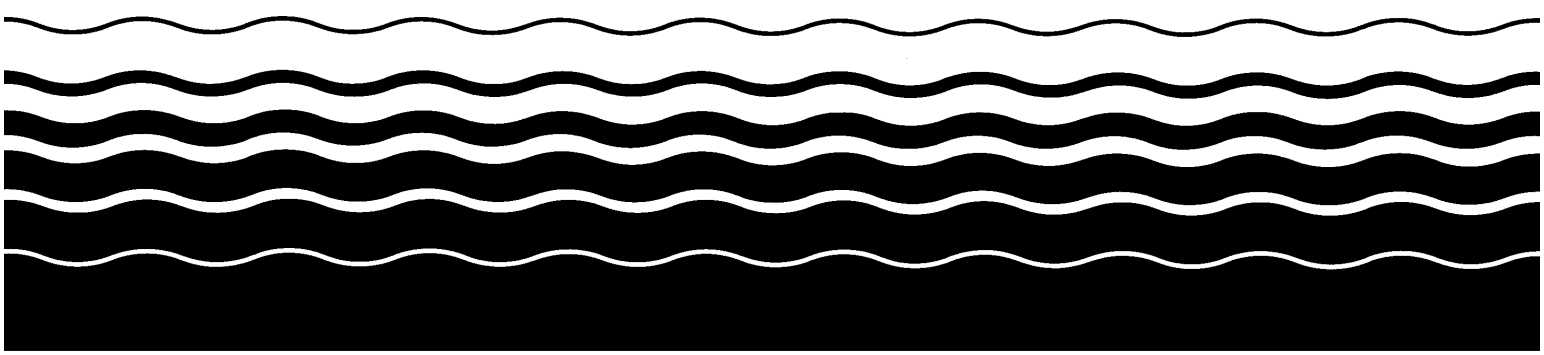
Learn more about the EPA's Draft Sewage Sludge Risk Assessment for PFOA and PFOS.

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Development Document for Final Effluent Limitations Guidelines and Standards for the Landfills Point Source Category



DEVELOPMENT DOCUMENT
FOR
FINAL EFFLUENT LIMITATIONS
GUIDELINES AND STANDARDS
FOR THE
LANDFILLS
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LANDFILLS DEVELOPMENT DOCUMENT
TABLE OF CONTENTS

1.0 LEGAL AUTHORITY 1-1
1.1 Legal Authority 1-1
1.2 Background 1-1
1.2.1 Clean Water Act (CWA) 1-1
1.2.1.1 Best Practicable Control Technology Currently Available (BPT) 1-1
1.2.1.2 Best Conventional Pollutant Control Technology (BCT) 1-2
1.2.1.3 Best Available Technology Economically Achievable (BAT) . 1-2
1.2.1.4 New Source Performance Standards (NSPS) 1-3
1.2.1.5 Pretreatment Standards for Existing Sources (PSES) 1-3
1.2.1.6 Pretreatment Standards for New Sources (PSNS) 1-4
1.2.2 Section 304(m) Requirements 1-4

2.0 SUMMARY AND SCOPE 2-1
2.1 Introduction 2-1
2.2 Subcategorization 2-1
2.3 Scope of Final Regulation 2-2
2.4 Best Practicable Control Technology Currently Available (BPT) 2-4
2.5 Best Conventional Pollutant Control Technology (BCT) 2-4
2.6 Best Available Technology Economically Achievable (BAT) 2-4
2.7 New Source Performance Standards (NSPS) 2-5
2.8 Pretreatment Standards for Existing Sources (PSES) 2-5
2.9 Pretreatment Standards for New Sources (PSNS) 2-5
2.10 Implementation of the Rule for Contaminated Ground Water Flows and Wastewater from Recovering Pumping Wells 2-5
2.11 Implementation of the Rule for Storm Water Discharges 2-7
2.12 Exclusion for Captive Landfill Facilities 2-10
2.13 Determination of Similar Wastes for Captive Landfill Facilities 2-16

3.0 INDUSTRY DESCRIPTION 3-1
3.1 Regulatory History of the Landfills Industry 3-3
3.1.1 RCRA Subtitle C 3-3
3.1.1.1 Land Disposal Restrictions 3-4
3.1.1.2 Minimum Technology Requirements 3-6
3.1.2 RCRA Subtitle D 3-6
3.1.2.1 40 CFR Part 257, Subpart A - Criteria for Classification of Solid Waste Disposal Facilities and Practices 3-7

TABLE OF CONTENTS

3.1.2.2 40 CFR Part 257, Subpart B - Conditionally Exempt Small Quantity Generator Revised Criteria 3-8
3.1.2.3 40 CFR Part 258 Revised Criteria for Municipal Solid Waste Landfills 3-8
3.1.3 Current Wastewater Regulations 3-9
3.2 Industry Profile 3-10
3.2.1 Industry Population 3-11
3.2.2 Number and Location of Facilities 3-12
3.2.2.1 Captive Landfill Facilities 3-13
3.2.3 General Information on Landfill Facilities 3-14
3.2.4 Waste Receipts and Types 3-15
3.2.5 Sources of Wastewater 3-16
3.2.5.1 Landfill Leachate 3-16
3.2.5.2 Landfill Gas Condensate 3-17
3.2.5.3 Drained Free Liquids 3-17
3.2.5.4 Truck/Equipment Washwater 3-18
3.2.5.5 Laboratory-Derived Wastewater 3-18
3.2.5.6 Storm Water 3-19
3.2.5.7 Contaminated Ground Water 3-19
3.2.5.8 Recovering Pumping Wells 3-20
3.2.6 Leachate Collection Systems 3-20
3.2.7 Pretreatment Methods 3-21
3.2.8 Baseline Treatment 3-22
3.2.9 Discharge Types 3-22

4.0 DATA COLLECTION ACTIVITIES 4-1
4.1 Introduction 4-1
4.2 Preliminary Data Summary 4-1
4.3 Clean Water Act (CWA) Section 308 Questionnaires 4-3
4.3.1 Screener Surveys 4-4
4.3.1.1 Recipient Selection and Mailing 4-4
4.3.1.2 Information Collected 4-5
4.3.1.3 Data Entry, Coding, and Analysis 4-6
4.3.1.4 Mailout Results 4-6
4.3.2 Detailed Technical Questionnaires 4-6
4.3.2.1 Recipient Selection and Mailing 4-7
4.3.2.2 Information Collected 4-7
4.3.2.3 Data Entry, Coding, and Analysis 4-8

TABLE OF CONTENTS

4.3.2.4 Mailout Results 4-9
4.4 Detailed Monitoring Questionnaire 4-9
4.4.1 Recipient Selection and Mailing 4-9
4.4.2 Information Collected 4-10
4.4.3 Data Entry, Coding, and Analysis 4-10
4.5 Engineering Site Visits 4-10
4.6 Wastewater Characterization Site Visits 4-11
4.7 EPA Week-Long Sampling Program 4-12
4.8 Other Data Sources 4-13
4.8.1 Industry Supplied Data 4-13
4.8.2 Comprehensive Environmental Response, Compensation and Liability Act (CERCLA)/Superfund Amendments and Reauthorization Act (SARA) Ground Water Data 4-13
4.8.3 POTW Study 4-14
4.8.4 National Risk Management Research Laboratory Data 4-15
4.9 QA/QC and Other Data Editing Procedures 4-15
4.9.1 QA/QC Procedures 4-16
4.9.2 Analytical Database Review 4-16
4.9.2.1 Data Review Narratives 4-16
4.9.2.2 Completeness Checks 4-16
4.9.2.3 Trip Blanks and Equipment Blanks 4-17
4.9.2.4 Field Duplicates 4-18
4.9.2.5 Grab Samples 4-19
4.9.2.6 Non-Detect Data 4-19
4.9.2.7 Bi-Phasic Samples 4-20
4.9.2.8 Conversion of Weight/Weight Data 4-20
4.9.2.9 Average Concentration Data 4-21
4.9.3 Detailed Questionnaire Database Review 4-21
4.9.4 Detailed Monitoring Questionnaire Review 4-22

5.0 INDUSTRY SUBCATEGORIZATION 5-1
5.1 Subcategorization Approach 5-1
5.2 Landfills Subcategories 5-2
5.3 Other Factors Considered for Basis of Subcategorization 5-3
5.3.1 Types of Wastes Received 5-3
5.3.2 Wastewater Characteristics 5-7
5.3.3 Facility Size 5-8
5.3.4 Ownership 5-9

TABLE OF CONTENTS

5.3.5 Geographic Location 5-9
5.3.6 Facility Age 5-11
5.3.7 Economic Characteristics 5-14
5.3.8 Treatment Technologies and Costs 5-14
5.3.9 Energy Requirements 5-15
5.3.10 Non-Water Quality Impacts 5-15

6.0 WASTEWATER GENERATION AND CHARACTERIZATION 6-1
6.1 Wastewater Generation and Sources of Wastewater 6-1
6.2 Wastewater Flow and Discharge 6-4
6.2.1 Wastewater Flow and Discharge at Subtitle D Non-Hazardous Landfills 6-5
6.2.2 Wastewater Flow and Discharge at Subtitle C Hazardous Landfills . . . 6-6
6.3 Wastewater Characterization 6-7
6.3.1 Background Information 6-8
6.3.1.1 Landfill Leachate 6-8
6.3.1.1.1 Additional Sources of Non-Hazardous Leachate Characterization Data 6-12
6.3.1.2 Landfill Gas Condensate 6-13
6.3.1.3 Drained Free Liquids 6-14
6.3.1.4 Truck and Equipment Washwater 6-14
6.3.2 Pollutant Parameters Analyzed at EPA Sampling Episodes 6-15
6.3.3 Raw Wastewater Characterization Data 6-17
6.3.4 Conventional, Toxic, and Selected Nonconventional Pollutant Parameters 6-18
6.3.5 Toxic Pollutants and Remaining Nonconventional Pollutants 6-20
6.3.6 Raw Wastewater at Subtitle D Non-Hazardous Landfills 6-21
6.3.6.1 Raw Wastewater at Subtitle D Municipal Landfills 6-21
6.3.6.2 Raw Wastewater at Subtitle D Non-Municipal Landfills 6-21
6.3.6.3 Dioxins and Furans in Raw Wastewater at Subtitle D Non-Hazardous Landfills 6-23
6.3.7 Raw Wastewater at Subtitle C Hazardous Landfills 6-24
6.3.7.1 Dioxins and Furans in Raw Wastewater at Subtitle C Hazardous Landfills 6-25

7.0 POLLUTANT PARAMETER SELECTION 7-1
7.1 Introduction 7-1
7.2 Pollutants Considered for Regulation 7-1

TABLE OF CONTENTS

7.3 Selection of Pollutants of Interest 7-2
7.4 Development of Pollutant Discharge Loadings 7-3
7.4.1 Development of Current Discharge Concentrations 7-3
7.4.1.1 Alternate Methodology for Non-Hazardous Subcategory: Subtitle D Non-Municipal 7-5
7.4.1.2 Alternate Methodology for the Hazardous Subcategory 7-5
7.4.2 Development of Pollutant Mass Loadings 7-6
7.5 Assessment of Pollutants of Interest 7-7
7.6 Selection of Pollutants to be Regulated for Direct Dischargers 7-7
7.6.1 Non-Hazardous Subcategory Pollutants to be Regulated for Direct Dischargers 7-8
7.6.2 Hazardous Subcategory Pollutants to be Regulated for Direct Dischargers 7-14
7.7 Selection of Pollutants to be Regulated for Indirect Dischargers 7-22
7.7.1 Pass-Through Analysis for Indirect Dischargers 7-22
7.7.2 Non-Hazardous Subcategory Pollutants to be Regulated for Indirect Dischargers 7-25
7.7.3 Hazardous Subcategory Pollutants to be Regulated for Indirect Dischargers 7-27

8.0 WASTEWATER TREATMENT TECHNOLOGY DESCRIPTION 8-1
8.1 Available BAT and PSES Technologies 8-1
8.1.1 Best Management Practices 8-1
8.1.2 Physical/Chemical Treatment 8-3
8.1.2.1 Equalization 8-3
8.1.2.2 Neutralization 8-4
8.1.2.3 Flocculation 8-5
8.1.2.4 Gravity Assisted Separation 8-6
8.1.2.5 Chemical Precipitation 8-8
8.1.2.5.1 Iron (Fe) Coprecipitation 8-11
8.1.2.6 Chemical Oxidation/Reduction 8-11
8.1.2.6.1 Breakpoint Chlorination 8-12
8.1.2.7 Air Stripping 8-14
8.1.2.8 Filtration 8-14
8.1.2.8.1 Sand Filtration 8-15
8.1.2.8.2 Diatomaceous Earth 8-17
8.1.2.8.3 Multimedia Filtration 8-17
8.1.2.8.4 Membrane Filtration 8-18

TABLE OF CONTENTS

8.1.2.8.4.1 Ultrafiltration 8-18
8.1.2.8.4.2 Reverse Osmosis 8-19
8.1.2.8.5 Fabric Filters 8-21
8.1.2.9 Carbon Adsorption 8-21
8.1.2.10 Ion Exchange 8-23
8.1.3 Biological Treatment 8-24
8.1.3.1 Lagoon Systems 8-26
8.1.3.2 Anaerobic Systems 8-30
8.1.3.3 Attached-Growth Biological Treatment Systems 8-31
8.1.3.4 Activated Sludge 8-34
8.1.3.5 Powder Activated Carbon Biological Treatment 8-38
8.1.3.6 Sequencing Batch Reactors (SBRs) 8-39
8.1.3.7 Nitrification Systems 8-40
8.1.3.8 Denitrification Systems 8-41
8.1.3.9 Wetlands Treatment 8-41
8.1.4 Sludge Handling 8-41
8.1.4.1 Sludge Slurrying 8-42
8.1.4.2 Gravity Thickening 8-42
8.1.4.3 Pressure Filtration 8-42
8.1.4.4 Sludge Drying Beds 8-43
8.1.5 Zero Discharge Treatment Options 8-44
8.2 Treatment Performance and Development of Regulatory Options 8-45
8.2.1 Performance of EPA Sampled Treatment Processes 8-46
8.2.1.1 Treatment Performance for Episode 4626 8-46
8.2.1.2 Treatment Performance for Episode 4667 8-48
8.2.1.3 Treatment Performance for Episode 4721 8-49
8.2.1.4 Treatment Performance for Episode 4759 8-50
8.2.1.5 Treatment Performance for Episode 4687 8-51

9.0 ENGINEERING COSTS 9-1
9.1 Evaluation of Cost-Estimation Techniques 9-1
9.1.1 Cost Models 9-1
9.1.2 Vendor Data 9-2
9.1.3 Other EPA Effluent Guideline Studies 9-3
9.1.4 Benchmark Analysis and Evaluation Criteria 9-3
9.1.5 Selection of Final Cost-Estimation Techniques 9-5
9.2 Engineering Costing Methodology 9-6
9.2.1 Treatment Costing Methodology 9-7

TABLE OF CONTENTS

9.2.1.1 Retrofit Costs 9-9
9.2.2 Land Costs 9-9
9.2.3 Residual Disposal Costs 9-9
9.2.4 Monitoring Costs 9-10
9.2.5 Off-Site Disposal Costs 9-11
9.3 Development of Cost Estimates for Individual Treatment Technologies 9-11
9.3.1 Equalization 9-12
9.3.2 Flocculation 9-13
9.3.3 Chemical Feed Systems 9-14
Sodium Hydroxide Feed Systems 9-15
Phosphoric Acid Feed Systems 9-17
Polymer Feed Systems 9-18
9.3.4 Primary Clarification 9-19
9.3.5 Activated Sludge Biological Treatment 9-20
9.3.6 Secondary Clarification 9-22
9.3.7 Multimedia Filtration 9-23
9.3.8 Reverse Osmosis 9-24
9.3.9 Sludge Dewatering 9-25
9.3.10 Granular Activated Carbon 9-26
9.3.11 Breakpoint Chlorination 9-27
9.4 Costs for Regulatory Options 9-28
9.4.1 Facility Selection 9-28
9.4.2 BPT Regulatory Costs 9-29
9.4.2.1 Subtitle D Non-Hazardous Subcategory BPT Costs 9-29
9.4.2.2 Subtitle C Hazardous Subcategory BPT Costs 9-30
9.4.3 BCT Regulatory Costs 9-30
9.4.3.1 Subtitle D Non-Hazardous Subcategory BCT Costs 9-30
9.4.3.2 Subtitle C Hazardous Subcategory BCT Costs 9-31
9.4.4 BAT Regulatory Costs 9-31
9.4.4.1 Subtitle D Non-Hazardous Subcategory BAT Costs 9-31
9.4.4.2 Subtitle C Hazardous Subcategory BAT Costs 9-32
9.4.5 NSPS Regulatory Costs 9-32
9.4.5.1 Subtitle D Non-Hazardous Subcategory NSPS Costs 9-32
9.4.5.2 Subtitle C Hazardous Subcategory NSPS Costs 9-33

10.0 NON-WATER QUALITY IMPACTS 10-1
10.1 Air Pollution 10-1
10.2 Solid and Other Aqueous Wastes 10-3

TABLE OF CONTENTS

10.3 Energy Requirements 10-5

11.0 DEVELOPMENT OF EFFLUENT LIMITATIONS GUIDELINES AND STANDARDS 11-1
11.1 Development of Long-Term Averages, Variability Factors, and Effluent Limitations 11-1
11.1.1 Calculation of Long-Term Averages 11-2
11.1.2 Calculation of Variability Factors 11-5
11.1.3 Calculation of Effluent Limitations 11-6
11.2 Best Practicable Control Technology Currently Available (BPT) 11-6
11.2.1 BPT Technology Options for the Subtitle D Non-Hazardous Subcategory 11-8
11.2.2 BPT Limits for the Subtitle D Non-Hazardous Subcategory 11-11
11.2.3 BPT Technology Options for the Subtitle C Hazardous Subcategory 11-20
11.2.4 BPT Limits for the Subtitle C Hazardous Subcategory 11-23
11.3 Best Conventional Pollutant Control Technology (BCT) 11-27
11.4 Best Available Technology Economically Achievable (BAT) 11-28
11.4.1 BAT Limits for the Subtitle D Non-Hazardous Subcategory 11-29
11.4.2 BAT Limits for the Subtitle C Hazardous Subcategory 11-31
11.5 New Source Performance Standards (NSPS) 11-31
11.6 Pretreatment Standards for Existing Sources (PSES) 11-32
11.6.1 EPA's Decision Not to Establish PSES for the Subtitle D Non-Hazardous Subcategory 11-34
11.6.1.1 EPA's Rationale for Not Establishing PSES for Ammonia 11-35
11.6.1.2 EPA's Rationale for Not Establishing PSES for Benzoic Acid 11-41
11.6.1.3 EPA's Rationale for Not Establishing PSES for P-Cresol 11-43
11.6.1.4 EPA's Rationale for Not Establishing PSES for Phenol 11-44
11.6.1.5 Public Comments to the Proposed Rule Regarding Non-Hazardous PSES 11-45
11.6.2 EPA's Decision Not to Establish PSES for the Subtitle C Hazardous Subcategory 11-48
11.7 Pretreatment Standards for New Sources (PSNS) 11-53

12.0 REFERENCES 12-1

TABLE OF CONTENTS

APPENDIX A Section 308 Survey for Landfills - Industry Population Analysis

APPENDIX B Definitions, Acronyms, and Abbreviations

LIST OF TABLES

2-1 Final Concentration Limitations for Hazardous Landfill Subcategory, Direct Discharges 2-20
2-2 Final Concentration Limitations for Hazardous Landfill Subcategory, Direct Discharges 2-21
2-3 Grouping of Subchapter N Effluent Guidelines and Standards 2-22
3-1 Number of Landfills per U.S. State 3-24
3-2 Ownership Status of Landfill Facilities 3-25
3-3 Total Landfill Facility Area 3-26
3-4 Landfill Facility Land Area Ranges 3-27
3-5 Number of Landfill Cells 3-28
3-6 Household and Non-Household Population Served 3-29
3-7 Household vs. Non-Household Customers 3-30
3-8 Wastes Received by Landfills in the United States 3-31
3-9 Total Volume of Waste Received by Landfills in 1992 by Regulatory Classification 3-32
3-10 Annual Tonnage of Waste Accepted by Landfills 3-33
3-11 Wastewater Flows Generated by Individual Landfills 3-34
3-12 Type of Leachate Collection Systems Used at Individual Landfills 3-35
3-13 Pretreatment Methods in Use at Individual Landfills 3-36
3-14 Types of Wastewater Treatment Employed by the Landfills Industry 3-37
3-15 Wastewater Treatment Facility Hours of Operation per Day 3-38
3-16 Wastewater Treatment Facility Average Hours of Operation per Day 3-39
3-17 Wastewater Treatment Facility Days of Operation per Week 3-40
3-18 Wastewater Treatment Facility Average Days of Operation per Week 3-41
3-19 Total Number of Facilities by Discharge Type 3-42
4-1 Screener Questionnaire Strata 4-5
4-2 Types of Facilities Included in EPA's Characterization and Engineering Site Visits . 4-23
4-3 Types of Facilities Included in EPA's Field Sampling Program 4-24
4-4 Episode Numbers for the Engineering Site Visits and Field Sampling Efforts 4-25
5-1 Subcategorization of the EPA Landfills Database 5-16
5-2 Raw Wastewater Median Concentrations of Pollutants of Interest Common to Both the Hazardous and Non-Hazardous Landfill Subcategories 5-20
5-3 Comparison of Subtitle D Non-Municipal and Municipal Raw Wastewater Pollutant Concentrations 5-22
5-4 Summary of EPA Sampling Data for Subtitle D Monofills, Average Raw Leachate Data for Selected Pollutants 5-24
5-5 Average Contaminated Ground Water Pollutant Concentrations at Hazardous Landfills in the EPA Database 5-25

LIST OF TABLES

5-6 Average Contaminated Ground Water Pollutant Concentrations at Non-Hazardous Landfills in the EPA Database 5-27
5-7 Age of Landfills in EPA Sampling Database 5-28
5-8 Median Raw Wastewater Characteristics at Non-Hazardous Landfills of Varying Age 5-30
6-1 Wastewater Generated in 1992: Hazardous Subcategory 6-26
6-2 Wastewater Generated in 1992: Non-Hazardous Subcategory Municipal Facilities . . 6-27
6-3 Wastewater Generated in 1992: Non-Hazardous Subcategory Non-Municipal Facilities 6-29
6-4 Quantity of In-Scope Wastewater Generated in 1992 6-30
6-5 Contaminant Concentration Ranges in Municipal Leachate as Reported in Literature Sources 6-31
6-6 Landfill Gas Condensate (from Detailed Questionnaire) 6-32
6-7 EPA Sampling Episode Pollutants Analyzed 6-33
6-8 EPA Sampling Episode List of Analytes Never Detected 6-37
6-9 Subtitle D Non-Hazardous Subcategory Median Raw Wastewater Concentration File 6-49
6-10 Subtitle C Hazardous Subcategory Median Raw Wastewater Concentration File . . . 6-50
6-11 Range of Conventional and Selected Nonconventional Pollutants Raw Wastewater Average Concentrations 6-51
6-12 Range of Metals and Toxic Pollutants Raw Wastewater Average Concentrations . . . 6-52
6-13 Range of Organic Pollutants Raw Wastewater Average Concentrations 6-53
6-14 Dioxins and Furans at Non-Hazardous EPA Sampling Episodes by Episode and Sample Point 6-54
6-15 Dioxins and Furans at Hazardous EPA Sampling Episodes by Episode and Sample Point 6-55
7-1 Non-Hazardous Subcategory Pollutants of Interest 7-29
7-2 Hazardous Subcategory Pollutants of Interest 7-30
7-3 Non-Hazardous Subcategory - POTW Percent Removals 7-31
7-4 Hazardous Subcategory - POTW Percent Removals 7-31
7-5 Non-Hazardous Subcategory - BAT Performance Data 7-32
7-6 Pass-Through Analysis for the Non-Hazardous Subcategory 7-33
7-7 Hazardous Subcategory - BAT Performance Data 7-34
7-8 Pass-Through Analysis for the Hazardous Subcategory 7-35
8-1 Wastewater Treatment Technologies Employed at In-Scope Landfill Facilities 8-53
8-2 Treatment Technology Performance for Facility 4626 - Subtitle D Municipal 8-54
8-3 Treatment Technology Performance for Facility 4667 - Subtitle D Municipal 8-55
8-4 Treatment Technology Performance for Facility 4721 - Subtitle C Hazardous 8-56
8-5 Treatment Technology Performance for Facility 4759 - Subtitle C Hazardous 8-58
8-6 Treatment Technology Performance for Facility 4687 - Subtitle D Municipal 8-60

LIST OF TABLES

9-1 Cost Comparison 9-34
9-2 Costing Source Comparison 9-35
9-3 Breakdown of Costing Method by Treatment Technology 9-36
9-4 Additional Cost Factors 9-37
9-5 Analytical Monitoring Costs 9-38
9-6 Subtitle D Non-Hazardous Facilities Costed for Off-Site Disposal 9-39
9-7 Unit Process Breakdown by Regulatory Option 9-40
9-8 Chemical Addition Design Method 9-41
9-9 Treatment Chemical Costs 9-42
9-10 Sodium Hydroxide Requirements for Chemical Precipitation 9-43
9-11 BPT/BCT/BAT Option I Subtitle D Non-Hazardous Subcategory 9-44
9-12 BPT/BCT/BAT Option II Subtitle D Non-Hazardous Subcategory 9-49
9-13 BAT Option III Subtitle D Non-Hazardous Subcategory 9-54
11-1 Removal of Pollutant of Interest Metals in the Non-Hazardous Subcategory 11-54
11-2 List of Subtitle D Municipal Solid Waste Facilities Employing Biological Treatment Considered for BPT in the Non-Hazardous Subcategory 11-55
11-3 Comparison of Raw Wastewater Mean Concentrations of Non-Hazardous Pollutants of Interest for Municipal Solid Waste Landfills and Hazardous Facility 16041 11-56
11-4 Candidate BPT Facilities for the Non-Hazardous Subcategory Eliminated from BPT Consideration Because No BOD_5 Effluent Data Was Available 11-57
11-5 Treatment Systems In Place at Landfill Facilities Considered for BPT in the Non-Hazardous Subcategory which Supplied BOD_5 Effluent Data 11-58
11-6 Landfill Facilities Considered for BPT in the Non-Hazardous Subcategory which Supplied BOD_5 Effluent Data: Flows, Concentration Data, and Reason for Elimination 11-59
11-7 Selected BPT Facilities for the Non-Hazardous Subcategory 11-60
11-8 TSS Data from Landfill Facilities Selected for BPT in the Non-Hazardous Subcategory 11-61
11-9 Facilities and Sample Points Used for the Development of BPT/BAT Effluent Limitations for the Non-Hazardous Subcategory 11-62
11-10 BPT Facility Data Excluded from the Calculation of Non-Hazardous BPT/BAT Limitations 11-64
11-11 BPT/BAT Limitations for the Non-Hazardous Subcategory 11-68
11-12 National Estimates of Pollutant of Interest Reductions for BPT/BAT Options for Municipal Solid Waste Landfills - Direct Dischargers 11-69
11-13 National Estimates of Pollutant of Interest Reductions for BPT/BAT Options for Non-Municipal Solid Waste Landfills - Direct Dischargers 11-70
11-14 Annual Pollutant Discharge Before and After the Implementation of BPT for Subtitle D Municipal Solid Waste Landfill Facilities in the Non-Hazardous Subcategory 11-71

LIST OF TABLES

11-15 Annual Pollutant Discharge Before and After the Implementation of BPT for Subtitle D Non-Municipal Solid Waste Landfill Facilities in the Non-Hazardous Subcategory 11-72
11-16 Selected BPT Facilities for the Hazardous Subcategory 11-73
11-17 Facilities and Sample Points Used for the Development of BPT/BAT Effluent Limitations for the Hazardous Subcategory 11-74
11-18 BPT Facility Data Excluded from the Calculation of Hazardous BPT/BAT Limitations 11-75
11-19 BPT/BAT Limitations for the Hazardous Subcategory 11-80
11-20 Comparison of Long-Term Averages for Nonconventional and Toxic Pollutants Regulated Under BAT for the Non-Hazardous Subcategory 11-81

LIST OF FIGURES

3-1 Development of National Estimates for the Landfills Industry 3-43
7-1 Development of Pollutants of Interest 7-36
7-2 Selection of Pollutants to be Regulated 7-37
8-1 Equalization 8-61
8-2 Neutralization 8-61
8-3 Clarification System Incorporating Coagulation and Flocculation 8-62
8-4 Calculated Solubilities of Metal Hydroxides 8-63
8-5 Chemical Precipitation System Diagram 8-64
8-6 Cyanide Destruction 8-65
8-7 Chromium Reduction 8-66
8-8 Typical Air Stripping System 8-67
8-9 Multimedia Filtration 8-68
8-10 Ultrafiltration System Diagram 8-69
8-11 Tubular Reverse Osmosis Module 8-70
8-12 Granular Activated Carbon Adsorption 8-71
8-13 Ion Exchange 8-72
8-14 Aerated Lagoon 8-73
8-15 Facultative Pond 8-74
8-16 Completely Mixed Digester System 8-75
8-17 Rotating Biological Contactor Cross-Section 8-76
8-18 Trickling Filter 8-77
8-19 Fluidized Bed Reactor 8-78
8-20 Activated Sludge System 8-79
8-21 Powdered Activated Carbon Treatment System 8-80
8-22 Sequencing Batch Reactor Process Diagram 8-81
8-23 Gravity Thickening 8-82
8-24 Plate-and-Frame Pressure Filtration System Diagram 8-83
8-25 Drying Bed 8-84
8-26 EPA Sampling Episode 4626 - Landfill Waste Treatment System Block Flow Diagram with Sampling Locations 8-85
8-27 EPA Sampling Episode 4667 - Landfill Waste Treatment System Block Flow Diagram with Sampling Locations 8-86
8-28 EPA Sampling Episode 4721 - Landfill Waste Treatment System Block Flow Diagram with Sampling Locations 8-87
8-29 EPA Sampling Episode 4759 - Landfill Waste Treatment System Block Flow Diagram with Sampling Locations 8-88
8-30 EPA Sampling Episode 4687 - Landfill Waste Treatment System Block Flow Diagram with Sampling Locations 8-89

LIST OF FIGURES

9-1 Option Specific Costing Logic Flow Diagram 9-59
9-2 Equalization Capital Cost Curve 9-60
9-3 Flocculation Capital Cost Curve 9-61
9-4 Flocculation O&M Cost Curve 9-62
9-5 Sodium Hydroxide Capital Cost Curve 9-63
9-6 Sodium Hydroxide O&M Cost Curve 9-64
9-7 Phosphoric Acid Feed Capital Cost Curve 9-65
9-8 Phosphoric Acid Feed O&M Cost Curve 9-66
9-9 Polymer Feed Capital Cost Curve 9-67
9-10 Polymer Feed O&M Cost Curve 9-68
9-11 Primary Clarifier Capital Cost Curve 9-69
9-12 Primary Clarifier O&M Cost Curve 9-70
9-13 Aeration Basin Capital Cost Curve 9-71
9-14 Air Diffusion System Capital Cost Curve 9-72
9-15 Air Diffusion System O&M Cost Curve 9-73
9-16 Secondary Clarifier Capital Cost Curve 9-74
9-17 Secondary Clarifier O&M Cost Curve 9-75
9-18 Multimedia Filtration Capital Cost Curve 9-76
9-19 Multimedia Filtration O&M Cost Curve 9-77
9-20 Reverse Osmosis Capital Cost Curve 9-78
9-21 Sludge Drying Beds Capital Cost Curve 9-79
9-22 Sludge Drying Beds O&M Cost Curve 9-80
9-23 GAC Capital Cost Curve 9-81
9-24 GAC O&M Cost Curve 9-82
9-25 Breakpoint Chlorination Capital Cost Curve 9-83
9-26 Breakpoint Chlorination O&M Cost Curve 9-84
11-1 BPT/BCT/BAT/PSES/PSNS Non-Hazardous Subcategory Option I Flow Diagram 11-82
11-2 BPT/BCT/BAT Non-Hazardous Subcategory Option II & NSPS Flow Diagram . . . 11-83
11-3 BPT/BCT/BAT Hazardous Subcategory Option II & NSPS Flow Diagram 11-84
11-4 BAT Non-Hazardous Subcategory Option III Flow Diagram 11-85

1.0 LEGAL AUTHORITY

1.1 Legal Authority

Effluent limitations guidelines and standards for the Landfills industry are promulgated under the authority of Sections 301, 304, 306, 307, 308, 402, and 501 of the Clean Water Act, 33 U.S.C. 1311, 1314, 1316, 1317, 1318, 1342, and 1361.

1.2 Background

1.2.1 Clean Water Act (CWA)

The Federal Water Pollution Control Act Amendments of 1972 established a comprehensive program to "restore and maintain the chemical, physical, and biological integrity of the Nation's waters" (Section 101(a)). To implement the Act, EPA is to issue effluent limitations guidelines, pretreatment standards, and new source performance standards for industrial dischargers. These guidelines and standards are summarized briefly in the following sections.

1.2.1.1 Best Practicable Control Technology Currently Available (BPT) (Section 304(b)(1) of the CWA)

In the guidelines for an industry category, EPA defines BPT effluent limits for conventional, priority,[1] and nonconventional pollutants. In specifying BPT, EPA looks at a number of factors. EPA first considers the cost of achieving effluent reductions in relation to the effluent reduction benefits. The Agency also considers: the age of the equipment and facilities; the processes employed and any required process changes; engineering aspects of the control technologies; non-water quality environmental impacts (including

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[1] In the initial stages of EPA CWA regulation, EPA efforts emphasized the achievement of BPT limitations for control of the "classical" pollutants (e.g., TSS, pH, BOD_5). However, nothing on the face of the statute explicitly restricted BPT limitation to such pollutants. Following passage of the Clean Water Act of 1977 with its requirement for points sources to achieve best available technology limitations to control discharges of toxic pollutants, EPA shifted its focus to address the listed priority pollutants under the guidelines program. BPT guidelines continue to include limitations to address all pollutants.

energy requirements); and such other factors as the Agency deems appropriate (CWA 304(b)(1)(B)). Traditionally, EPA establishes BPT effluent limitations based on the average of the best performances of facilities within the industry of various ages, sizes, processes or other common characteristics. Where, however, existing performance is uniformly inadequate, EPA may require higher levels of control than currently in place in an industrial category if the Agency determines that the technology can be practically applied.

1.2.1.2 Best Conventional Pollutant Control Technology (BCT) (Section 304(b)(4) of the CWA)

The 1977 amendments to the CWA required EPA to identify effluent reduction levels for conventional pollutants associated with BCT technology for discharges from existing industrial point sources. In addition to other factors specified in Section 304(b)(4)(B), the CWA requires that EPA establish BCT limitations after consideration of a two part "cost-reasonableness" test. EPA explained its methodology for the development of BCT limitations in July 1986 (51 FR 24974).

Section 304(a)(4) designates the following as conventional pollutants: biochemical oxygen demand (BOD_5), total suspended solids (TSS), fecal coliform, pH, and any additional pollutants defined by the Administrator as conventional. The Administrator designated oil and grease as an additional conventional pollutant on July 30, 1979 (44 FR 44501).

1.2.1.3 Best Available Technology Economically Achievable (BAT) (Section 304(b)(2) of the CWA)

In general, BAT effluent limitations guidelines represent the best economically achievable performance of plants in the industrial subcategory or category. The factors considered in assessing BAT include the cost of achieving BAT effluent reductions, the age of equipment and facilities involved, the process employed, potential process changes, and non-water quality environmental impacts, including energy requirements. The Agency retains considerable discretion in assigning the weight to be accorded these factors. Unlike

BPT limitations, BAT limitations may be based on effluent reductions attainable through changes in a facility's processes and operations. As with BPT, where existing performance is uniformly inadequate, BAT may require a higher level of performance than is currently being achieved based on technology transferred from a different subcategory or category. BAT may be based upon process changes or internal controls, even when these technologies are not common industry practice.

1.2.1.4 New Source Performance Standards (NSPS) (Section 306 of the CWA)

NSPS reflect effluent reductions that are achievable based on the best available demonstrated control technology. New facilities have the opportunity to install the best and most efficient production processes and wastewater treatment technologies. As a result, NSPS should represent the most stringent controls attainable through the application of the best available control technology for all pollutants (i.e., conventional, nonconventional, and priority pollutants). In establishing NSPS, EPA is directed to take into consideration the cost of achieving the effluent reduction and any non-water quality environmental impacts and energy requirements.

1.2.1.5 Pretreatment Standards for Existing Sources (PSES) (Section 307(b) of the CWA)

PSES are designed to prevent the discharge of pollutants that pass through, interfere with, or are otherwise incompatible with the operation of publicly owned treatment works (POTWs). The CWA authorizes EPA to establish pretreatment standards for pollutants that pass through POTWs or interfere with treatment processes or sludge disposal methods at POTWs. Pretreatment standards are technology-based and analogous to BAT effluent limitations guidelines.

The General Pretreatment Regulations, which set forth the framework for the implementation of categorical pretreatment standards, are found at 40 CFR Part 403. These regulations contain a definition of pass

through that addresses localized rather than national instances of pass through and establish pretreatment standards that apply to all non-domestic dischargers (see 52 FR 1586, January 14, 1987).

1.2.1.6 Pretreatment Standards for New Sources (PSNS) (Section 307(b) of the CWA)

Like PSES, PSNS are designed to prevent the discharges of pollutants that pass through, interfere-with, or are otherwise incompatible with the operation of POTWs. PSNS are to be issued at the same time as NSPS. New indirect dischargers have the opportunity to incorporate into their plants the best available demonstrated technologies. The Agency considers the same factors in promulgating PSNS as it considers in promulgating NSPS.

1.2.2 Section 304(m) Requirements

Section 304(m) of the CWA, added by the Water Quality Act of 1987, requires EPA to establish schedules for (1) reviewing and revising existing effluent limitations guidelines and standards ("effluent guidelines") and (2) promulgating new effluent guidelines. On January 2, 1990, EPA published an Effluent Guidelines Plan (55 FR 80) that established schedules for developing new and revised effluent guidelines for several industry categories. One of the industries for which the Agency established a schedule was the Hazardous Waste Treatment Industry.

The Natural Resources Defense Council (NRDC) and Public Citizen, Inc. filed suit against the Agency, alleging violation of Section 304(m) and other statutory authorities requiring promulgation of effluent guidelines (NRDC et al. v. Reilly, Civ. No. 89-2980 (D.D.C.)). Under the terms of the consent decree in that case, as amended, EPA agreed, among other things, to propose effluent guidelines for the "Landfills and Industrial Waste Combusters" category by November 1997 and final action by November 1999. Although the Consent Decree lists "Landfills and Industrial Waste Combusters" as a single entry, EPA is publishing separate regulations for Industrial Waste Combusters and for Landfills.

2.0 SUMMARY AND SCOPE

2.1 Introduction

The final regulation for the Landfills industry establishes effluent limitations guidelines and standards for the control of wastewater pollutants. This document presents the information concerning, and rationale supporting, these effluent limitations guidelines and standards. Section 2.2 discusses the subcategorization approach, Section 2.3 describes the scope of the regulation, Sections 2.4 through 2.9 summarize the final effluent limitations and pretreatment standards, and Sections 2.10 through 2.13 discuss several of the implementation issues associated with this rule.

2.2 Subcategorization

For the final rule, EPA decided that a single set of limitations and standards was not appropriate for the landfills industry and, thus, developed different limitations and standards for subcategories within the industry. These subcategories are summarized below:

RCRA Subtitle C Hazardous Waste Landfill Subcategory

Subpart A of 40 CFR Part 445, "RCRA Subtitle C Hazardous Waste Landfill Subcategory," applies to wastewater discharges from a solid waste disposal facility subject to the criteria in 40 CFR Part 264 Subpart N - Standards for Owners and Operators of Hazardous Waste Treatment, Storage, and Disposal Facilities and 40 CFR Part 265 Subpart N -Interim Standards for Owners and Operators of Hazardous Waste Treatment, Storage, and Disposal Facilities. Hazardous waste landfills are subject to requirements outlined in 40 CFR Parts 264 and 265 that include the requirement to maintain a leachate collection and removal systems during the active life and post-closure period of the landfill. For a discussion of these criteria, see the Preamble to the proposed landfill guideline at 63 FR 6426, 6430-31. (February 6, 1998).

RCRA Subtitle D Non-Hazardous Waste Landfill Subcategory

Subpart B of 40 CFR Part 445, "RCRA Subtitle D Non-Hazardous Waste Landfill Subcategory," applies

to wastewater discharges from all landfills classified as RCRA Subtitle D non-hazardous landfills subject to either of the criteria established in 40 CFR Parts 257 (Criteria for Classification of Solid Waste Disposal Facilities and Practices) or 258 (Criteria for Municipal Solid Waste Landfills). For a discussion of these criteria, see the Preamble to the proposed landfill guideline at 63 FR 6426, 6431-32. (February 6, 1998).

2.3 Scope of Final Regulation

The final limitations and standards cover pollutants in wastewater discharges associated only with the operation and maintenance of those landfills regulated under Subtitles C and D of the Resource Conservation and Recovery Act (RCRA).[1] The rule applies to wastewater generated at both active as well as closed landfills regulated under Subtitle C or Subtitle D of RCRA.

Furthermore, this rule does not apply to wastewater discharges associated with the operation and maintenance of land application or treatment units, surface impoundments, underground injection wells, waste piles, salt dome or bed formations, underground mines, caves or corrective action units.[2] Additionally, this guideline does not apply to waste transfer stations, or any wastewater not directly attributed to the operation and maintenance of Subtitle C or Subtitle D landfill units. Consequently, wastewater, such as that generated in off-site washing of vehicles used in landfill operations, is not within the scope of this guideline.

The wastewater covered by the rule includes leachate, gas collection condensate, drained free liquids, laboratory-derived wastewater, contaminated storm water, and contact washwater from truck exteriors and surface areas which have come in direct contact with solid waste at the landfill facility. However,

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[1] EPA's Subtitle C and Subtitle D regulations define "landfill". See 40 CFR 257.2, 258.2 ("municipal solid waste landfill") and 260.10. Permitted Subtitle C landfills are authorized to accept hazardous wastes as defined in 40 CFR Part 261. Subtitle D landfills are authorized to receive municipal, commercial or industrial waste that is not hazardous (as well as hazardous waste excluded from regulation under Subtitle C).

[2] These terms are defined at 40 CFR 257.2 and 260.10.

ground water and wastewater from recovery pumping well operations which have been contaminated by a landfill and are collected and discharged are excluded from this guideline. Section 2.10 discusses the exclusion from the rule for contaminated ground water flows and for wastewater from recovering pumping wells. Discharges of non-contaminated storm water, as defined by this guideline, are also not covered by the rule. EPA defines non-contaminated storm water and discusses the rationale for not covering it in this guideline at Section 2.11.

The rule does not apply to wastewater discharges generated at a landfill that is associated with an industrial or commercial operation -- so-called "captive" landfills -- in most circumstances. The following describes the applicability of the final rule to captive landfills. The final rule does not apply to discharges of landfill wastewater from captive landfills so long as one or more of the following conditions are met:

a) The captive landfill is operated in conjunction with other industrial or commercial operations, and it only receives wastes generated by the industrial or commercial operation directly associated with the landfill.

b) The landfill is operated in conjunction with other industrial or commercial operations and it receives both wastes generated by the industrial or commercial operation directly associated with the landfill as well as other wastes and the other wastes received for landfill disposal are generated by a facility that is subject to the same provisions in 40 CFR Subchapter N as the receiving facility directly associated with the landfill.

c) The landfill is operated in conjunction with other industrial or commercial operations and it receives wastes generated by the industrial or commercial operation directly associated with the landfill as well as other wastes and the other wastes are similar in nature to the wastes generated by the industrial or commercial operation directly associated with the landfill.

d) The landfill is operated in conjunction with a Centralized Waste Treatment (CWT) facility subject to 40 CFR Part 437 so long as the CWT facility commingles the landfill wastewater with other non-landfill wastewater for treatment. If a CWT facility discharges landfill wastewater separately from other CWT wastewater or commingles the wastewater from its landfill only with wastewater from other landfills, then the landfill discharge is subject to the landfill effluent guidelines.

e) The landfill is operated in conjunction with other industrial or commercial operations, and it receives wastes from public service activities (as defined in Appendix B) and the landfill does not receive a fee or other remuneration for the disposal service.

Section 2.12 discusses in detail EPA's rationale for adopting the conditions described above for the captive landfill exclusion.

2.4 Best Practicable Control Technology Currently Available (BPT)

EPA established BPT effluent limitations guidelines for conventional, priority, and nonconventional pollutants for both subcategories. For RCRA Subtitle C hazardous waste landfills, EPA promulgated effluent limitations standards based on a treatment system consisting of equalization, chemical precipitation, biological treatment, and multimedia filtration. For RCRA Subtitle D non-hazardous waste landfills, EPA promulgated effluent limitations standards based on the following treatment: equalization, biological treatment, and multimedia filtration. Table 2-1 and Table 2-2 list the final effluent limitations and standards for the Hazardous subcategory and the Non-Hazardous subcategory, respectively.

2.5 Best Conventional Pollutant Control Technology (BCT)

EPA established BCT effluent limitations guidelines equivalent to the BPT guidelines for the control of conventional pollutants (BOD_5, TSS, and pH) for both subcategories. The effluent limitations are the same as those specified for BOD_5, TSS, and pH in Table 2-1 and Table 2-2 for the Hazardous subcategory and the Non-Hazardous subcategory, respectively

2.6 Best Available Technology Economically Achievable (BAT)

EPA established BAT effluent limitations guidelines equivalent to the BPT guidelines for control of priority and nonconventional pollutants for both subcategories. Any existing hazardous landfill subject to this guideline must achieve the following effluent limitations which represent the application of BAT: Limitations

for ammonia (as N), alpha terpineol, aniline, benzoic acid, naphthalene, p-cresol, phenol, pyridine, arsenic, chromium and zinc are the same as the corresponding limitations specified in Table 2-1.

Any existing non-hazardous landfill subject to this guideline must achieve the following effluent limitations which represent the application of BAT: Limitations for ammonia (as N), alpha terpinol, benzoic acid, *p*-cresol, phenol and zinc are the same as the corresponding limitations specified in Table 2-2.

2.7 New Source Performance Standards (NSPS)

EPA established NSPS effluent limitations guidelines equivalent to the BPT, BCT, and BAT guidelines for the control of conventional, priority and nonconventional pollutants for both subcategories. Table 2-1 and Table 2-2 list the final effluent limitations and standards for the Hazardous subcategory and the Non-Hazardous subcategory, respectively.

2.8 Pretreatment Standards for Existing Sources (PSES)

EPA did not establish PSES for either subcategory. Any source subject to this rule that introduces wastewater pollutants into a publicly owned treatment works (POTW) must comply with 40 CFR Part 403.

2.9 Pretreatment Standards for New Sources (PSNS)

EPA did not establish PSNS for either subcategory. Any new source subject to this rule that introduces wastewater pollutants into a POTW must comply with 40 CFR Part 403.

2.10 Implementation of the Rule for Contaminated Ground Water Flows and Wastewater from Recovering Pumping Wells

During development of the rule, EPA considered whether it should also include contaminated ground water flows within the scope of this guideline. Historically, many landfill operations have caused the contamination

of local ground water, mostly as a result of leakage from unlined landfill units in operation prior to the minimum technology standards for landfills established by RCRA Subtitle C and D regulations. Subsequently, State and Federal action under the Comprehensive Environmental Response Compensation and Liability Act (CERCLA) has required facilities to clean up contaminated ground water. In many cases, this has resulted in the collection, treatment, and discharge of treated ground water to surface waters. In addition, in the case of RCRA Subtitle C hazardous waste landfills and municipal solid waste landfills (MSWLFs), applicable regulatory standards require ground water monitoring and post-closure care and, in the event of ground water contamination, corrective action measures. These requirements may also result in treatment of contaminated ground water by such landfill facilities.

EPA, however, has not included contaminated ground water flows within its assessment for this guideline. Several reasons support EPA's decision not to include contaminated ground water as a regulated waste stream for this rule.

EPA evaluated flows, pollutant concentrations, treatment in place, and current treatment standards for discharges of contaminated ground water from landfills. From this evaluation, EPA concluded that pollutants in contaminated ground water flows are often very dilute or are treated to very low levels prior to discharge. EPA concluded that, whether as a result of corrective action measures taken pursuant to RCRA authority or State action to clean up contaminated landfill sites, landfill discharges of treated contaminated ground water are being adequately controlled. Consequently, further regulation under this rule would be redundant and unnecessary.

EPA is aware that there are landfill facilities that collect and treat both landfill leachate and contaminated ground water flows. In the case of such facilities, EPA has concluded that decisions regarding the appropriate discharge limits should be left to the judgment of the permit writer. As indicated by data collected through the questionnaires and EPA sampling, ground water characteristics are often site specific and may contain very few contaminants or may, conversely, exhibit characteristics similar in nature to

leachate. In cases where the ground water is very dilute, the Agency is concerned that contaminated ground water may be used as a dilution flow. In these cases, the permit writer should develop "best professional judgment" (BPJ) permit limits based on separate treatment of the flows, or develop BPJ limits based on a flow-weighted building block approach, in order to prevent dilution of the regulated leachate flows. However, in cases where the ground water may exhibit characteristics similar to leachate, commingled treatment may be appropriate, cost effective, and environmentally beneficial. EPA recommends that the permit writer consider the characteristics of the contaminated ground water before making a determination if commingling ground water and leachate for treatment is appropriate. EPA recommends that the permit writer refer to the leachate characteristics data in Chapter 6 in order to determine whether contaminated ground water at a landfill has characteristics similar to leachate.

Recovering pumping well wastewater is generated as a result of the various ancillary operations associated with ground water pumping operations. These operations include construction and development, well maintenance, and well sampling (i.e. purge water). The wastewater will have very similar characteristics to contaminated ground water. Therefore, for the same reasons that EPA did not include contaminated ground water as a regulated wastewater, these regulations do not apply to wastewater from recovering pumping well operations.

2.11 Implementation of the Rule for Storm Water Discharges

EPA received extensive comments on its proposal to include contaminated storm water as a regulated waste stream under the landfills effluent guidelines. Several commenters stated that contaminated storm water (storm water that comes into contact with solid waste at the landfill site) should not be subject to the landfills effluent limitations guidelines because this is already covered by the Final National Pollutant Discharge Elimination System Storm Water Multi-sector General Permit (MSGP) for Industrial Activities (September 29, 1995; 60 FR 50803), in States where it applies, or by an equivalent general permit issued by an NPDES authorized State.

In an effort to clarify the types of storm water runoff that are subject to the landfills effluent guidelines, EPA revised the definition of contaminated and non-contaminated storm water in the final rule. EPA defines these terms as follows:

> *Contaminated storm water*: Storm water which comes in direct contact with landfill wastes, the waste handling and treatment areas, or wastewater that is subject to the limitations and standards.
>
> *Non-contaminated storm water*: Storm water which does not come in direct contact with landfill wastes, the waste handling and treatment areas, or wastewater that is subject to the limitations and standards. Non-contaminated storm water includes storm water which flows off the cap, cover, intermediate cover, daily cover, and/or final cover of the landfill.

The Storm Water Pollution Prevention Plan (SWPPP) required by the storm water MSGP or an authorized State's equivalent general permit requires landfill facilities to identify all of the sources of storm water contamination at the landfill and then implement measures and controls (such as good housekeeping for materials storage, sediment and erosion controls - particularly from intermediate and final covers) in an effort to prevent storm water contamination. EPA believes that the storm water MSGP (or an authorized State's equivalent general permit) adequately controls pollutants from storm water runoff from covered areas of the landfill. Covered areas of the landfill include the following: capped, final cover, intermediate cover, and daily cover areas. The Agency believes that the SWPPP and the monitoring requirements in the storm water MSGP provide adequate controls for reducing the level of pollutants in storm water from these areas of landfills.

EPA recognizes that there may be some incidental contact with wastes when storm water flows over a daily or intermediate cover. However, EPA concluded that such contact will not lead to any meaningful "contamination" of the storm water so long as the landfill complies with the requirements of the storm water MSGP or an authorized State's equivalent general permit. For example, the Best Management Practices (BMPs) outlined in Table L-1 and L-2 of the storm water MSGP (60 FR 50940) and the monitoring requirements in Table L-5 and L-6 for TSS and total recoverable iron (60 FR 50943) provide adequate

controls for the pollutants that would most likely be associated with runoff from covered areas of non-hazardous landfills.

Similarly, for hazardous landfills, BMPs and monitoring requirements outlined in Table K-2 (60 FR 50935) and Table K-3 (60 FR 50936), respectively, also require controls for pollutants associated with runoff from covered areas of a landfill. In EPA's view, BMPs provide a fair degree of control of these pollutants and the monitoring requirements of the MSGP provide a tool for evaluating the effectiveness of the Storm Water Pollution Prevention Plan.

As part of the Agency's continuing effort to improve its environmental and pollution control programs, EPA has concluded that, although the MSGP provides some control for contaminated storm water runoff, the landfills effluent limitations guidelines provide a more comprehensive level of control for storm water runoff that has come in direct contact with solid waste, waste handling and treatment areas, or wastewater flows that are controlled under this rule. Although the storm water MSGP considered circumstances in which untreated leachate may be incidently commingled with storm water, the Agency explicitly acknowledged in the MSGP that insufficient data were available to establish numeric limits for storm water that might be contaminated based on best available technology for MSWLFs (60 FR 50942), non-hazardous industrial landfills (60 FR 50943), and hazardous landfills (60 FR 50935).

However, EPA has now concluded that the data collected in support of the landfills effluent limitations guidelines provide the basis for establishing appropriate numeric limitations for contaminated storm water. EPA specifically noted in the Preamble for the storm water MSGP that it was developing these guidelines and that where the guidelines applied to discharges, facilities must comply with them (60 FR 50942). In addition, EPA intends to propose a reissuance of the storm water MSGP which would include the promulgated landfills effluent limitations for contaminated storm water (as defined by this landfill guideline).

2.12 Exclusion for Captive Landfill Facilities

As discussed in Section 2.3 above, the rule does not apply to captive landfills in most circumstances. In developing the proposed guidelines, an important question EPA addressed was how to treat landfill leachate generated at a landfill that is associated with an industrial or commercial operation -- so-called "captive" landfills. Currently, in the case of wastewater sources that are not subject to effluent limitations guidelines and standards, NPDES permit writers must impose limitations on discharges of these wastewater sources that are developed on a case-by-case, best professional judgment (BPJ) basis. Similarly, an indirect discharger may not introduce any pollutants to a POTW from these sources that will pass through or interfere with the POTW's operations. Generally, each POTW is required to develop a pretreatment program and enforce the prohibition on pass through and interference through specific local limits.

EPA initially considered development of effluent guidelines to address any landfill discharging directly to surface waters of the United States or introducing pollutants into a POTW. Consequently, EPA's technical evaluation for the proposal included an assessment of virtually all landfill facilities which collect wastewater as a result of landfilling operations. EPA proposed to exclude wastewater discharges from captive landfills located at industrial facilities in specific circumstances. In the proposal, a captive landfill would not have been subject to the guidelines if: 1) it commingled landfill process wastewater with non-landfill process wastewater for treatment, and 2) the landfill received only waste generated on site or waste generated from a similar activity at another facility under the same corporate structure.

For the final rule, EPA determined that these requirements are too restrictive and therefore the Agency has decided not to include captive landfills within the scope of this guideline except in a limited number of circumstances. The effect of this decision for the final rule is not to allow these wastewater sources to escape treatment. Landfill wastewater at captive facilities is and will remain subject to treatment and controls on its discharge. The Clean Water Act (CWA) requires wastewater discharges to meet technology-based effluent limitations on the discharge whether the mechanism for imposing these limitations

is EPA-established national effluent limitations guidelines or a permit writer's imposition on a case-by-case basis of BPJ limitations. In like manner, in order to prevent pass through or interference, indirect dischargers must limit their introduction of pollutants to a POTW whether EPA has established national categorical pretreatment standards for the discharge or a POTW has established local limits.

For the final rule, EPA has modified the proposal to remove the requirement that a facility must commingle its wastewater from a captive landfill with the facility's non-landfill process wastewater for treatment in order not to be subject to the landfills effluent guideline, in most circumstances. For the reasons described in detail below, EPA did not remove the commingling requirement for CWTs. In addition, EPA also changed the conditions under which captive landfills may accept off-site wastes and not be subject to this guideline.

In the proposal, EPA stated that the commingling requirement ensures that wastewater from captive landfills will undergo adequate treatment (treatment that is comparable to the level of treatment that would be required by the landfills effluent guideline) prior to discharge. EPA determined that the commingling of landfill wastewater with industrial wastewater for treatment was an unnecessary requirement to impose in nationally applicable regulations for the reasons discussed below. Permit writers are establishing appropriate limits on these discharges by either applying the effluent limitations guidelines applicable to the associated industrial activity to the discharge or developing other BPJ limitations. EPA recommends that permit writers use this guideline when developing these BPJ limitations.

From the information developed by the Agency for this rulemaking and confirmed by comments on the proposal, EPA has concluded that landfill wastewater generated by captive landfills operated in conjunction with and receiving the bulk of their waste from an industrial or commercial operation will have a similar pollutant profile to the wastewater generated in the industrial or commercial operation. EPA has further concluded that the wastewater generated by landfill operations at most of the captive facilities are already subject to effluent guidelines. In the circumstances in which the wastewater is not expressly subject to effluent guidelines, EPA has determined that permit writers generally impose BPJ limitations on the

discharge of landfill wastewater that are similar to the limitations applicable to the discharge of industrial process wastewater whether commingled or not. EPA has compared the wastewater treatment technologies employed at many of the industrial facilities operating landfills in conjunction with industrial or commercial operations to the treatment technologies that EPA used as the basis for the BPT/BAT limits in this effluent guideline. The Agency's review of such situations shows that the landfill wastewater receives treatment that is comparable or better than the level of treatment that would be required by the landfills effluent guideline.

Consequently, EPA has decided to eliminate the requirement of commingling as a condition for a captive landfill not to be subject to landfill limitations and standards (except in the case of CWTs). EPA has concluded that landfill wastewater at captive landfills is now and will continue to receive adequate treatment because the landfill wastewater generally must meet the same effluent limitations that would have been required had the waste streams been commingled. In cases where the permit writer is establishing BPJ limitations for the discharge of captive landfill wastewater that is not commingled for treatment, the permit writer should look at the effluent guidelines applicable to the associated industrial operation and the landfills effluent guidelines for potential guidance in setting those limitations.

Because of the nature of most CWTs, EPA determined that the reasons that generally supported exclusion of other captive landfills would not apply in the case of CWTs. As explained above, EPA concluded that a captive landfill which only received wastes generated in an industrial or commercial operation directly associated with the landfill or similar wastes would generate a leachate with a similar pollutant profile to the other wastewater streams produced at the industrial operation. In such circumstances, the data reviewed by EPA showed that the landfill wastewater and other industrial wastewater are generally commingled for treatment and subject to the same discharge limitations. In these circumstances, it was appropriate not to subject the landfill to this guideline.

Because a CWT, by its very nature, may generate a wide array of different solid wastes for landfill disposal, it may generate a leachate that varies significantly from other streams being treated at the CWT at the time the leachate is collected. Therefore, EPA concluded that the basis for the exclusion -- the similarity in wastewater -- would not necessarily apply in the case of CWTs. EPA decided that, in order to ensure that the CWT landfill wastewater is treated adequately, the landfill wastewater from a CWT landfill should be commingled with other CWT wastewater for treatment.

It is worth noting that the majority of industrial facilities that operate captive landfills do commingle their landfill process wastewater with other industrial wastewater for treatment. (February 6, 1998; 63 FR 6430). A review by EPA of individual NPDES permits for captive and intracompany facilities found that, for the most part, landfill waste streams are mixed with categorical wastes and subject to limitations comparable to the final limitations for landfills.

Most captive landfill facilities choose to commingle their landfill process wastewater for treatment for several reasons. First of all, wastewater flows from captive landfills are usually quite small in comparison to the wastewater flows from other industrial operations at the captive facility. EPA's data show that the landfill wastewater flows are often less than one percent and typically less than three percent of the industrial wastewater flows. Therefore, most facilities choose to commingle the relatively small volume of landfill wastewater with the larger industrial wastewater volumes rather than maintaining and operating a completely separate wastewater treatment system for the landfill wastewater. Second, as mentioned above, it is likely that leachate from landfills at industrial operations will reflect a pollutant profile similar to the facility's industrial process wastewater. Therefore, based on the similarity of the waste streams, facilities often choose to commingle these streams for treatment. In fact, most of the captive facilities identified in EPA's database commingle their leachate with other industrial process wastewater for treatment. Comments submitted in response to the proposed rule suggest that situations do exist where a captive landfill may not commingle the landfill wastewater with other process wastewater for treatment. In circumstances where a facility chooses not to commingle landfill leachate for treatment with the other

process wastewater generated, EPA has concluded, based on comments submitted, that this wastewater will still be subject to categorical or Best Professional Judgment (BPJ) limits reflecting comparable removals in most instances.

Lastly, industrial facilities with captive landfills often choose to commingle their waste streams for treatment in order to avoid additional NPDES or pretreatment requirements that would be necessary if the waste streams were treated and discharged separately. EPA concluded that the wastewater generated by landfill operations at most of the captive facilities are already subject to categorical effluent limitations (or pretreatment standards). Information gathered by EPA prior to proposal and in comments received on the proposed rule support the conclusion that these wastewater flows were either assessed and evaluated for the effluent limitations guideline applicable to the facility, or are subject to a "building block approach" (for directs) or the "combined waste stream formula" (for indirects) for developing BPJ limits or standards established by the permit writer or local control authority. This review indicates that, for the most part, these landfill waste streams are mixed with categorical wastes for treatment and subject to limitations comparable to the final landfill regulation.

Based on comments received, the Agency also determined that the requirement in the proposal that solid wastes deposited in the captive landfill must either be generated on site or from an off-site facility under the same corporate structure was too restrictive and could often prohibit a company from safely and properly disposing of solid wastes accepted from tolling, remediation, product stewardship, and public service activities.

In the proposal, EPA narrowly limited the universe of captive landfills that fall outside the scope of this rule to captive landfills that only accepted wastes from on site or from off-site facilities under the same corporate structure. The reason for this was essentially to ensure that the captive landfills were only accepting wastes that would be similar to those wastes generated on site. This in turn would provide some degree of assurance that the leachate generated from these wastes would be compatible with the on-site industrial

wastewater treatment. However, from the comments submitted on this issue, EPA determined this waste acceptance criterion for the captive exclusion was too restrictive. Those commenting on this issue identified several waste acceptance practices that are commonly used by captive landfills that would not meet the proposed exclusion criteria but are consistent with EPA's objective that landfill leachate receive treatment compatible with its expected constituents. Many of these current waste disposal practices are activities that EPA encourages and, therefore, EPA has revised the exclusion criteria pertaining to waste acceptance for captive/intracompany landfills in order to accommodate these disposal practices.

Specifically, several commenters requested that EPA broaden the criteria for determining those captive landfills that fall outside the scope of this rule to include waste acceptance from tolling and contract manufacturers, product stewardship, company partnerships, and remediation activities. EPA concluded that waste disposal at captive landfills from these types of activities will, in most cases, result in leachate that will be adequately controlled through the implementation of categorical or BPJ limitations at the facility. However, EPA remains concerned that there are circumstances in which inter-company waste products deposited in the landfill may result in contaminants in the leachate that may not be compatible with the existing industrial wastewater treatment system or may not be covered adequately by the existing industrial effluent guideline. Therefore, one of the alternative conditions for the revised applicability provisions of the guideline described above for captive landfills provides that waste accepted at the captive landfill must be of a similar nature to the wastes generated at the operation with the associated landfill. Thus, the permitting authority must determine that wastes accepted for disposal at a captive landfill are of a similar nature to the waste generated at the facility directly associated with the captive landfill. Factors that the permit writer should consider in determining whether a waste is similar are described at Section 2.13.

In addition, commenters also requested that EPA include the acceptance of wastes for disposal as a public service as a category of landfill practices that qualify for the captive exclusion. EPA agrees and has included such a provision. EPA applauds the efforts of manufacturing facilities who provide members of their communities with a cost effective and environmentally safe means for disposing of their solid waste.

Therefore, in the final rule, EPA determined that this rule shall not apply to those landfills operated in conjunction with other industrial or commercial operations which receive other wastes from public service activities so long as the company owning the landfill does not receive a fee or other remuneration for the disposal service. EPA's decision not to subject captive landfills that accept off-site wastes for disposal as a public service is not inconsistent with its decision generally to condition non-applicability on the similarity of wastes accepted for disposal. Based on its review of data collected for this guideline and comments received, EPA concluded that the quantity of wastes accepted for disposal as a public service would not in any measurable way affect the pollutant profile of the leachate generated by the landfill even if dissimilar. Of course, these wastewater flows still remain subject to treatment to achieve BPJ permit limits reflecting the landfill contribution to the facility discharge.

The Agency has determined that whether captive landfills accepting wastes from off site or from a company not within the same corporate structure on a non-commercial basis should be subject to the landfills effluent guideline should hinge on the ability of the captive landfill to handle the waste in an appropriate manner. Therefore, the Agency concluded that the waste acceptance criterion for determining those captive landfills that fall outside the scope of this rule should be based on the similarity of the waste accepted for disposal from another facility to the waste generated by the industrial or commercial operation directly associated with the landfill. In the case of captive landfills treating similar wastes, the permit writer should base permit limits on limitations for the guideline to which the industrial or commercial operation is subject or establish BPJ limitations. Again, the permit writer, if developing BPJ limitations, should consider these landfill guidelines as guidance in this effort.

2.13 Determination of Similar Wastes for Captive Landfill Facilities

As discussed at Sections 2.3 and 2.12 above, the Agency concluded that discharges from captive landfills should not be subject to the guidelines if the captive landfills only accepted waste for disposal from another facility that was similar to the waste generated by the industrial or commercial operation directly associated with the landfill. This section offers guidance to permit writers for determining whether a solid waste

received for disposal in a captive landfill is similar to those wastes generated by the facility directly associated with the landfill.

According to EPA's database, many of the industrial or commercial facilities that operate captive landfills are subject to effluent limitations guidelines in 40 CFR Subchapter N. For the most part, facilities subject to a particular industrial category effluent guideline produce similar types of wastes. Therefore, EPA decided that this rule does not apply to landfills operated in conjunction with other industrial or commercial operations when the landfill receives wastes generated by the industrial or commercial operation directly associated with the landfill and also receives other wastes generated by a facility that is subject to the same provisions in 40 CFR Subchapter N as the waste-receiving facility. However, there are cases where a captive landfill is directly associated with an industrial or commercial operation that is not subject to an effluent guideline. Or, a facility, subject to an effluent guideline, may operate a landfill in conjunction with industrial or commercial operations, but may also accept other wastes from facilities that are not subject to the same effluent guideline or not subject to an effluent guideline at all. In these cases, the permit writer must determine whether the other wastes received for disposal are of similar nature to the wastes generated by the industrial or commercial operation directly associated with the landfill. In cases where the permit writer determines that the other waste accepted by the captive landfill is not similar to the waste generated by the industrial or commercial activity directly associated with the landfill, the landfill wastewater will be subject to the landfills effluent limitations. However, if the permit writer determines that the wastes are similar, then the wastewater from the captive landfill should be subject to the same categorical effluent guideline (or BPJ limitations) as the industrial or commercial facility.

A permit writer should consider the following factors in deciding whether other wastes received by a captive landfill are similar to those wastes generated by the industrial or commercial operation directly associated with the landfill:

1. Are the other wastes received from facilities that are subject to the same provisions in 40 CFR Subchapter N as the facility directly associated with the captive landfill?

If so, then the landfills effluent guidelines do not apply to this captive landfill. If not, then the permit writer should consider the other factors listed below.

2. Are the other wastes received from facilities that are part of the same effluent guidelines "grouping" as shown in Table 2-3?

   If so, it is likely that the wastes are similar and the landfills effluent guidelines do not apply. Table 2-3 groups the industrial categories under Subchapter N into the following six groups: Organics, Metals, Inorganics and Non-Metals, Pesticides, Explosives, and Asbestos. It is likely that industries within the same industrial effluent guideline "grouping" will generate similar types of constituents in the solid wastes, and the leachate resulting from the disposal of these wastes will be controlled adequately by the effluent limitation for the industrial or commercial facility directly associated with the captive landfill. However, this may not always be the case and, therefore, EPA left to the local control authority the determination of whether the landfills effluent guideline should apply to a captive landfill that accepts wastes from other facilities that are not subject to the same provisions in 40 CFR Subchapter N. The local permitting authority will determine whether a captive landfill which accepts wastes from other industrial activities, apart from those directly associated with the landfill, is subject to the landfills effluent guidelines based on the similarity of the other wastes and the likelihood that these wastes will result in leachate that is compatible with the wastewater treatment technology used to treat the landfill leachate.

3. In the case of hazardous captive landfills, do the other wastes being received have the same hazardous waste codes as those generated at the facility directly associated with the landfill?

   If so, it is possible that the wastes are similar. However, this may not always be the case and, therefore, EPA left to the local control authority the determination of whether the landfills effluent guideline should apply to a captive landfill that accepts wastes from other facilities that are not subject to the same provisions in 40 CFR Subchapter N.

4. Is a significant portion of the waste deposited in the landfill from the industrial or commercial operation that is directly associated with the captive landfill?

   The control authority should analyze the number of customers and the amount of the off-site or inter-company waste deposited relative to the quantity of on-site or intracompany waste placed in the captive landfill. Again, the main reason for the exclusion for captive landfills is that their leachate should resemble the industrial wastewater of the operation directly associated with the landfill and, therefore, the landfill leachate will be adequately controlled by the applicable industrial effluent guidelines. However, this logic is only applicable when the bulk of the waste placed in the landfill is of similar content to that being produced by the industrial facility directly associated with the landfill. Therefore, when

applying the captive exclusion, the control authority should analyze the volume and characteristics of waste received from inter-company waste transfers in determining whether the leachate generated by the captive landfill will have similar characteristics to the industrial wastewater generated by the company owning the landfill.

5. Is the facility that is directly associated with the captive landfill deriving any revenues from waste disposal at the landfill?

   In developing the exclusion for captive landfills, EPA's intent was to exclude those non-commercial landfills that are directly associated with an industrial or commercial operation and whose leachate is currently being adequately addressed by the facility's categorical or BPJ limitations. EPA believes that where any revenues are being derived from the collection of fees for solid waste disposal at a captive landfill, the facility is accepting wastes on a commercial basis - - wastes that may well be dissimilar to that being disposed of at the landfill. The captive exception is premised on the fact that, in most cases, leachate from a landfill associated with an industrial operation will resemble the industrial process wastewater generated by the industrial operation and, therefore, the landfill leachate will be adequately controlled by the applicable industrial effluent guidelines or BPJ limitations. However, this is a reasonable assumption only in circumstances where the waste placed in the landfill is of similar content to that being produced by the industrial operation directly associated with the landfill. It is likely that a commercial landfill may accept significant volumes of waste that are not similar to the wastes generated by the industrial operation directly associated with the landfill.

6. Is the industrial or commercial facility directly associated with the captive landfill accepting wastes for disposal as part of public service activities?

   If so, and the facility does not receive a fee or other remuneration for the disposal service, the captive landfill is not subject to this rule. EPA defines public service activities in Appendix B.

Table 2-1: Final Concentration Limitations for Hazardous Landfill Subcategory, Direct Discharges

Pollutant or Pollutant Property	Maximum for 1 day (mg/L)	Monthly average shall not exceed (mg/L)
BOD_5	220	56
TSS	88	27
Ammonia	10	4.9
Arsenic (Total)	1.1	0.54
Chromium (Total)	1.1	0.46
Zinc (Total)	0.535	0.296
Alpha Terpineol	0.042	0.019
Aniline	0.024	0.015
Benzoic Acid	0.119	0.073
Naphthalene	0.059	0.022
p-Cresol	0.024	0.015
Phenol	0.048	0.029
Pyridine	0.072	0.025
pH	Shall be in the range 6.0 - 9.0 pH units.	

Table 2-2: Final Concentration Limitations for Non-Hazardous Landfill Subcategory, Direct Discharges

Pollutant or Pollutant Property	Maximum for 1 day (mg/L)	Monthly average shall not exceed (mg/L)
BOD_5	140	37
TSS	88	27
Ammonia	10	4.9
Zinc	0.20	0.11
Alpha Terpineol	0.033	0.016
Benzoic Acid	0.12	0.071
p-Cresol	0.025	0.014
Phenol	0.026	0.015
pH	Shall be in the range 6.0 - 9.0 pH units.	

Table 2-3: Grouping of Subchapter N Effluent Guidelines and Standards

Industrial Category	Part #	Characteristics					
		Organics	Metals	Inorganics Non-metal	Pesticides	Explosives	Asbestos
Dairy products and processing	405	X					
Grain mills	406	X					
Canned and preserves fruits and vegetables	407	X					
Canned and preserved seafood	408	X					
Sugar processing	409	X					
Textile mills	410	X	X				
Cement manufacturing	411		X	X			
Feedlots	412	X					
Electroplating	413		X				
Organic chemicals, plastics and synthetic fibers	414	X					
Inorganic chemicals manufacturing	415			X			
Soap and detergent manufacturing	417	X					
Fertilizer manufacturing	418			X			
Petroleum refining	419	X					
Iron and steel manufacturing	420		X				
Nonferrous metals manufacturing	421		X				
Phosphate manufacturing	422			X			
Steam electric power plants	423	X	X				
Ferroalloy manufacturing	424		X				
Leather tanning and finishing	425	X	X				
Glass manufacturing	426			X			
Asbestos manufacturing	427						X
Rubber processing	428	X					
Timber products processing	429	X					
Pulp, paper and paperboard	430	X					
Builder's paper and board mills	431	X					
Meat products	432	X					
Metal finishing	433		X				
Coal mining	434			X			
Oil and gas extraction	435	X					
Mineral mining and processing	436		X				
Pharmaceutical preparations	439	X					
Ore mining	440		X				
Paving and roofing materials (tars & asphalt)	443	X		X			
Paint formulation	446	X	X				
Ink formulation	447	X					
Gum and wood chemicals	454	X	X				
Pesticides	455				X		
Explosives manufacturing	457					X	
Carbon black manufacturing	458	X					
Photographic equipment and supplies	459		X				
Hospital	460	X					

3.0 INDUSTRY DESCRIPTION

The Landfills industry consists of facilities that receive wastes either as commercial or municipal operations or as on-site (captive) operations owned by waste generators. These landfill facilities generate wastewater and discharge it to surface waters, publicly owned treatment works (POTWs), or use some other form of zero or alternative disposal. The Resource Conservation and Recovery Act (RCRA) defines 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). RCRA classifies landfills as either Subtitle C hazardous or Subtitle D non-hazardous. Wastewater generated and discharged by landfills can include, but is not limited to, leachate, gas collection condensate, contaminated ground water, contaminated storm water, drained free liquids, truck/equipment washwater, laboratory-derived wastewater, and wastewater recovered from pumping wells.

Landfills are commonly classified by the types of wastes they accept and/or by their ownership status. Some of the terms used to describe a landfill include municipal, sanitary, chemical, industrial, RCRA, hazardous waste, Subtitle C, and Subtitle D. Although non-hazardous landfills do not knowingly accept hazardous wastes, these facilities may contain hazardous wastes due to disposal practices that occurred prior to 1980 and before the enactment of RCRA and its associated regulations. The following section provides descriptions of landfills in terms of ownership type and regulatory type.

Ownership Status

- ***Municipal:*** Municipally-owned landfills are those that are owned by local governments. Municipally-owned landfills may be designed to accept either Subtitle D or Subtitle C wastes (see "Regulatory Type").

- ***Commercial:*** Commercial landfills are privately-owned facilities and can be designed to receive either municipal, hazardous, or non-hazardous industrial wastes. Typical non-hazardous industrial wastes include packaging and shipping materials, construction and demolition debris, ash, and sludge.

- ***Captive:*** Captive landfills are operated in conjunction with other industrial or commercial operations, and receive the bulk of their wastes from the industrial or commercial operations. Captive landfills are located on, or adjacent to, the facility they service and are common at major hazardous waste generators, such as chemical and petrochemical manufacturing plants.

- ***Intra-company:*** Landfill facilities operated in conjunction with other industrial or commercial operations which only receive waste from off-site facilities under the same corporate structure, ownership, or control. These landfills are similar to captive sites but receive wastes from multiple locations of one company.

Regulatory Type

- ***Subtitle C:*** Subtitle C landfills are those disposal operations authorized by RCRA to accept hazardous wastes as defined in 40 CFR Part 261. Subtitle C landfills are subject to the criteria in 40 CFR Part 264 Subpart N - Standards for Owners and Operators of Hazardous Waste Treatment, Storage, and Disposal Facilities and 40 CFR Part 265 Subpart N - Interim Standards for Owners and Operators of Hazardous Waste Treatment, Storage, and Disposal Facilities. Hazardous waste landfills are subject to requirements outlined in 40 CFR Parts 264 and 265 that include the requirement to maintain a leachate collection and removal systems during the active life and post-closure period of the landfill. Section 3.1 presents more details on the regulatory requirements of Subtitle C.

- ***Subtitle D:*** Subtitle D landfills are those disposal operations that are subject to either of the criteria established in 40 CFR Parts 257 (Criteria for Classification of Solid Waste Disposal Facilities and Practices) or 258 (Criteria for Municipal Solid Waste Landfills). The wastes received at Subtitle D landfills include municipal refuse, ash, sludge, construction and demolition debris, and non-hazardous industrial waste. These facilities were not designed to receive hazardous wastes; however, prior to 1980 and the enactment of RCRA, older landfills may have received waste later classified as hazardous under RCRA. Any Subtitle D landfill accepting municipal refuse after October 9, 1993 is classified as a Municipal Waste Disposal Unit, and is regulated under 40 CFR 258. Any Subtitle D landfill not accepting municipal waste after October 9, 1993 continues to be regulated under 40 CFR 257. For the purposes of this document, Subtitle D landfills not accepting municipal refuse are referred to as "Subtitle D non-municipal" landfills.

The following discussions present a regulatory history of this industry and past EPA studies.

3.1 Regulatory History of the Landfills Industry

Depending on the type of wastes disposed of at a landfill, the landfill may be subject to regulation and permitting under either Subtitle C or Subtitle D of RCRA. Subtitle C facilities receive wastes that are identified or listed as hazardous wastes at 40 CFR Part 261. Subtitle D landfills can only accept wastes that are not defined as hazardous wastes at 40 CFR Part 261. The following sections outline some of the key regulations that have been developed to control the environmental impacts of Subtitle C and Subtitle D landfills.

3.1.1 RCRA Subtitle C

Subtitle C of the RCRA of 1976 directed EPA to promulgate regulations to protect human health and the environment from the improper management of hazardous wastes. Based on this statutory mandate, the goal of the RCRA program was to provide comprehensive, "cradle-to-grave" management of hazardous waste. These regulations establish a system for tracking the disposal of hazardous wastes and special design requirements for landfills depending on whether a landfill accepted hazardous or non-hazardous waste. Key statutory provisions in RCRA Subtitle C include the following:

- Section 3001: Requires the promulgation of regulations identifying the characteristics of hazardous waste and listing particular hazardous wastes.
- Section 3002: Requires the promulgation of standards, such as manifesting, record keeping, etc., applicable to generators of hazardous waste.
- Section 3003: Requires the promulgation of standards, such as manifesting, record keeping, etc., applicable to transporters of hazardous waste.
- Section 3004: Requires the promulgation of performance standards applicable to the owners and operators of facilities for the treatment, storage, or disposal of hazardous waste.
- Section 3005: Requires the promulgation of regulations requiring each person owning or operating a treatment, storage, or disposal facility to obtain a permit.

These regulations establish a system for tracking the disposal of hazardous wastes and performance and design requirements for landfills accepting hazardous waste. Under RCRA, requirements are initially triggered by a determination that a waste is hazardous as defined in 40 CFR Part 261. Any party, including the original generator, that treats, stores, or disposes of a hazardous waste must notify EPA and obtain an EPA identification number. EPA established performance regulations governing the operation of hazardous waste landfills at 40 CFR Parts 264 and 265. RCRA Subtitle C hazardous waste regulations apply to landfills that presently accept hazardous wastes or have accepted hazardous waste at any time after November 19, 1980.

3.1.1.1 Land Disposal Restrictions

The Hazardous and Solid Waste Amendments (HSWA) to RCRA, enacted on November 8, 1984, largely prohibit the land disposal of untreated hazardous wastes. Once a hazardous waste is prohibited from land disposal, the statute provides only two options for legal land disposal: 1) meet the EPA-established treatment standard for the waste prior to land disposal, or 2) dispose of the waste in a land disposal unit that has been found to satisfy the statutory no-migration test. A no- migration unit is one from which there will be no migration of hazardous constituents for as long as the waste remains hazardous. (RCRA Sections 3004 (d),(e),(g)(5)).

Under Section 3004 of RCRA, the treatment standards that EPA develops may be expressed as either constituent concentration levels or as specific methods of treatment. Under RCRA Section 3004(m)(1), the criteria for these standards is that they must substantially diminish the toxicity of the waste or substantially reduce the likelihood of migration of hazardous constituents from the waste so that short-term and long-term threats to human health and the environment are minimized. For purposes of the restrictions, the RCRA program defines land disposal to include, among other things, any placement of hazardous waste in a landfill. Land disposal restrictions are published in 40 CFR Part 268.

EPA has used hazardous waste treatability data as the basis for land disposal restrictions standards. EPA has identified Best Demonstrated Available Treatment Technology (BDAT) for each listed hazardous waste. BDAT is the treatment technology that EPA finds to be the most effective in treating a waste and that also is readily available to generators and treaters. In some cases, EPA has designated as BDAT for a particular waste stream a treatment technology shown to have successfully treated a similar but more difficult to treat waste stream. This ensured that the land disposal restrictions standards for a listed waste stream were achievable since they always reflected the actual treatability of the waste itself or of a more refractory waste.

As part of the Land Disposal Restrictions (LDRs), EPA promulgated Universal Treatment Standards (UTS) as part of the RCRA phase two final rule (July 27,1994). The UTS are a series of concentrations for wastewater and non-wastewater that provide a single treatment standard for each constituent. Previously, the LDR regulated constituents according to the identity of the original waste; thus, several numerical treatment standards existed for each constituent. The UTS simplified the standards by having only one treatment standard for each constituent in any waste residue. The LDR and the UTS restricted the concentrations of wastes that could be disposed of in landfills, thus improving the environmental quality of the leachate from landfills.

The LDR treatment standards established under RCRA may differ from the Clean Water Act effluent guidelines both in their format and in the numerical values set for each constituent. The differences result from the use of different legal criteria for developing the limits and resulting differences in the technical and economic criteria and data sets used for establishing the respective limits.

The difference in format of the LDR and effluent guidelines is that LDR establishes a single daily limit for each pollutant parameter while effluent guidelines establish monthly and daily limits. Additionally, the effluent guidelines provide for several types of discharge, including new and existing sources, and indirect and direct discharge.

The differences in numerical limits established under the Clean Water Act may differ, not only from LDR and UTS, but also from point-source category to point-source category (e.g., Electroplating, 40 CFR 413; and Metal Finishing, 40 CFR 433). The effluent guidelines limitations and standards are industry-specific, subcategory-specific, and technology-based. The numerical limits are typically based on different data sets that reflect the performance of specific wastewater management and treatment practices. Differences in the limits reflect differences in the following statutory factors that the Administrator is required to consider in developing technically and economically achievable limitations and standards: manufacturing products and processes (which for landfills involves types of waste disposed), raw materials, wastewater characteristics, treatability, facility size, geographic location, age of facility and equipment, non-water quality environmental impacts, and energy requirements. A consequence of these differing approaches is that similar or identical waste streams are regulated at different levels dependent on the receiving body of the wastewater (e.g. a POTW, a surface water, or a land disposal facility).

3.1.1.2 Minimum Technology Requirements

To further protect human health and the environment from the adverse affects of hazardous waste disposed of in landfills, the 1984 HSWA to RCRA established minimum technology requirements for landfills receiving hazardous waste. These provisions required the installation of double liners and leachate collection systems at new landfills, at replacements of existing units, and at lateral expansions of existing units. The Amendments also required all hazardous waste landfills to install ground water monitoring wells by November 8, 1987. Performance regulations governing the operation of hazardous waste landfills are included at 40 CFR Parts 264 and 265.

3.1.2 RCRA Subtitle D

Landfills managing non-hazardous wastes are currently regulated under the RCRA Subtitle D program. These landfills include municipal, private intra-company, private captive, and commercial facilities used for the management of municipal refuse, incinerator ash, sewage sludge, and a range of non-hazardous industrial wastes.

3.1.2.1 40 CFR Part 257, Subpart A - Criteria for Classification of Solid Waste Disposal Facilities and Practices

EPA promulgated the criteria on September 13, 1979 (44 FR 53460) under the authority of RCRA Sections 1008(a) and 4004(a) and Sections 405(d) and (e) of the Clean Water Act. The criteria in §257.1 through 257.4 were adopted for determining which solid waste disposal facilities and practices pose a reasonable probability of adverse effects on health and the environment, and the criteria in §257.5 through 257.30 were adopted to ensure that non-municipal non-hazardous waste disposal units that receive conditionally exempt small quantity generator (CESQG) waste do not present risks to human health and the environment taking into account the practicable capability of such units. These criteria apply to all solid waste disposal facilities and practices. However, certain facilities and practices are not covered by the criteria, such as agricultural wastes returned to the soil as fertilizers or soil conditioners, overburden resulting from mining operations intended for return to the mine site, land application of domestic sewage or treated domestic sewage, the location and operation of septic tanks, hazardous waste disposal facilities which are subject to regulations under RCRA Subtitle C (discussed above), municipal solid waste landfills that are subject to the revised criteria in 40 CFR Part 258 (discussed below), and use or disposal of sewage sludge on the land when the sewage sludge is used or disposed of in accordance with 40 CFR Part 503 (See 40 CFR Part 257.1(c)(1) - (11)).

The criteria include general environmental performance standards addressing the following eight major areas: flood plains, protection of endangered species, protection of surface water, protection of ground water, limitations on the land application of solid waste, periodic application of cover to prevent disease vectors, air quality standards (prohibition against open burning), and safety practices ensuring protection from explosive gases, fires, and bird hazards to airports. Facilities that fail to comply with any of these criteria are considered open dumps, which are prohibited by RCRA Section 4005. Those facilities that meet the criteria are considered sanitary landfills under RCRA Section 4004(a). Landfill wastewater generated at solid waste disposal facilities that are subject to the requirements of 40 CFR Part 257 Subpart A are subject to the effluent limitations for the Non-Hazardous subcategory.

3.1.2.2 40 CFR Part 257, Subpart B - Conditionally Exempt Small Quantity Generator Revised Criteria

A conditionally-exempt small-quantity generator is generally defined as one who generates no more than 100 kilograms of hazardous waste per month in a calendar year (40 CFR 261.5(a)). Such conditionally-exempt small-quantity generators (with certain exceptions) are not subject to RCRA Subtitle C requirements. However, on July 1, 1996, EPA did the following: (1) amended Part 257 to establish criteria that must be met by non-municipal, non-hazardous solid waste disposal units that receive conditionally-exempt small-quantity generator waste and (2) established separate management and disposal standards (in 40 CFR 261.5(f)(3) and (g)(3)) for those who generate conditionally-exempt small-quantity generator waste (see 61 FR 342169). The conditionally-exempt small-quantity generator revised criteria for such disposal units include location standards, ground water monitoring, and corrective action requirements. Landfill wastewater generated at solid waste disposal facilities that are subject to the requirements of 40 CFR Part 257 Subpart B are subject to the effluent limitations for the Non-Hazardous subcategory.

3.1.2.3 40 CFR Part 258 Revised Criteria for Municipal Solid Waste Landfills

On October 9, 1991, EPA promulgated revised criteria for municipal solid waste landfills in accordance with the authority provided in RCRA Sections 1008(a)(3), 4004(a), 4010 (c) and Clean Water Act (CWA) Sections 405(d) and (e) (see 56 FR 50978). Under the terms of these revised criteria, municipal solid waste landfills are defined to mean a discrete area of land or an excavation that receives household waste, and is not a land application unit, surface impoundment, injection well, or waste pile, as those terms are defined in 40 CFR 257.2 and 258.2. In addition to household waste, a municipal solid waste landfill unit also may receive other types of RCRA Subtitle D wastes, such as commercial solid waste, non-hazardous sludge, and industrial solid waste. Such a landfill may be publicly or privately owned. A municipal solid waste landfill unit may be a new unit, existing municipal solid waste landfill unit, or a lateral expansion.

The municipal solid waste landfill revised criteria include location standards (Subpart B), operating criteria (Subpart C), design criteria (Subpart D), ground water monitoring and corrective action (Subpart E), closure and post-closure care criteria (Subpart F), and financial assurance requirements (Subpart G). The design criteria specify that new municipal solid waste landfill units and lateral expansions of existing units (as defined in Section 258.2) must be constructed in accordance with either (1) a design approved by a Director of a State whose municipal solid waste landfill permit program has been approved by EPA and which satisfies a performance standard to ensure that unacceptable levels of certain chemicals do not migrate beyond a specified distance from the landfill (Sections 258.40(a)(1), (c), (d), Table 1) or (2) a composite liner and a leachate collection system (Sections 258.40(a)(2), (b)). The ground water monitoring criteria generally require owners or operators of municipal solid waste landfills to monitor ground water for contaminants and generally implement a corrective action remedy when monitoring indicates that a ground water protection standard has been exceeded. However, certain small municipal solid waste landfills located in arid or remote locations are exempt from both design and ground water monitoring requirements. The closure standards require that a final cover be installed to minimize infiltration and erosion. The post-closure provisions generally require, among other things, that ground water monitoring continue and that the leachate collection system be maintained and operated for 30 years after the municipal solid waste landfill is closed. The Director of an approved State may increase or decrease the length of the post-closure period.

Again, as is the case with solid waste disposal facilities that fail to meet the requirements in 40 CFR Part 257, Subpart A, municipal solid waste landfills that fail to satisfy the revised criteria in Part 258 constitute open dumps and are therefore prohibited by RCRA Section 4005 (40 CFR 258.1(h)). Landfill wastewater generated at solid waste disposal facilities (i.e., municipal solid waste landfills) that are subject to the requirements in 40 CFR Part 258 are subject to the effluent limitations for the Non-Hazardous subcategory.

3.1.3 Current Wastewater Regulations

Prior to this regulation, EPA had not promulgated national effluent limitations guidelines for the discharge

of wastewater from the Landfills industry. In the absence of these guidelines, permit writers have had to rely on a combination of their own best professional judgement (BPJ), water quality standards, and technology transfer from other industrial guidelines in setting permit limitations for landfills discharging to surface waters. In addition, local control authorities also have had to rely on their own best professional judgement, pass-through analyses, and other local factors in establishing pretreatment standards for the discharge of landfill wastewater to their municipal sewage systems and POTWs.

In 1989, EPA completed a preliminary study of the Landfills industry. In a report entitled "Preliminary Data Summary for the Hazardous Waste Treatment Industry," EPA concluded that wastewater discharges from landfills can be a significant source of toxic pollutants being discharged to surface waters and POTWs. In a consent decree between NRDC and EPA, dated January 31, 1992, EPA agreed, among other things, to propose effluent guidelines for the "Landfills and Industrial Waste Combusters" category by November 1997 and final action by November 1999.

3.2 Industry Profile

The growth of the Landfills industry is a direct result of RCRA and subsequent EPA and State regulations that establish the conditions under which solid waste may be disposed. The implementation of the increased control measures required by RCRA has had a number of ancillary effects on the Landfills industry.

The RCRA requirements have affected the Landfills industry in different ways. On the one hand, it has forced many landfills to close because they lacked adequate on-site controls to protect against migration of hazardous constituents from the landfill, and it was not economical to upgrade the landfill facility. As a result, a large number of landfills, especially facilities serving small populations, have closed rather than incur the significant expense of upgrading.

Conversely, large landfill operations have taken advantage of economies of scale by serving wide geographic areas and accepting an increasing portion of the nation's solid waste. For example, responses

to the EPA's Waste Treatment Industry Survey indicated that 75 percent of the nation's municipal solid waste is deposited in large landfills representing only 25 percent of the landfill population.

EPA has identified several trends in the waste disposal industry that may increase the quantity of leachate produced by landfills. More stringent RCRA regulations and the restrictions on the management of wastes have increased the amount of waste disposed at landfills as well as the number of facilities choosing to send wastes off site to commercial facilities in lieu of pursuing on-site management options. This will increase treated leachate discharges from the nation's landfills, thus, potentially putting at risk the integrity of the nation's waters. Further, as a result of the increased number of leachate collection systems, the volume of leachate requiring treatment and disposal has greatly increased.

3.2.1 Industry Population

In developing the initial landfill population to be studied for this regulation, EPA used various sources such as State environmental and solid waste departments, the National Survey of Hazardous Waste Treatment, Storage, Disposal, and Recycling Facilities respondent list, Environmental Ltd.'s "1991 Directory of Industrial and Hazardous Waste Management Firms", and other sources discussed in Chapter 4. EPA identified 10,477[1] landfill facilities as the initial landfill population in the United States in 1992. Of this group, 9,882 were Subtitle D non-hazardous landfills and 595 were Subtitle C hazardous landfills. Table 3-1 presents the total number of landfill facilities by state in EPA's mailing list database. EPA solicited technical information from a sample of this initial population via screener surveys, and the Agency sent Detailed Technical Questionnaires to a statistical sample of the screener survey respondents. A total of 252 landfill facilities received Detailed Technical Questionnaires and 220 facilities responded with sufficient technical data to be included in the questionnaire database. Chapter 4, Section 4.3 presents a detailed discussion of screener survey and Detailed Questionnaire strata.

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[1] The initial landfill population of 10,477 does not include one pre-test facility which was included as a screener survey respondent.

Because EPA only sent Detailed Technical Questionnaires to a statistical sample of the initial industry population, the Agency scaled up the information provided by questionnaire respondents to represent the entire Landfills industry. By matching up the screener survey stratum with the Detailed Technical Questionnaire stratum, EPA calculated a weighting factor for each questionnaire respondent and scaled up any data provided by the respondent by this factor. Therefore, throughout this chapter, EPA presents national estimates based on the Detailed Technical Questionnaire respondents' data scaled up by their individual weighting factors. The Agency based the national estimates presented in the tables in this chapter on all 220 facilities included in the questionnaire database. Figure 3-1 presents the logic used for the development of the national estimates. EPA presents the methodology for calculating national estimates in the Final Statistical Development Document for the Landfills Industry (EPA-821-B-99-007).

3.2.2 Number and Location of Facilities

Many of the landfill facilities presented in Table 3-1 do not generate and/or collect wastewater that is subject to this regulation. Landfill generated wastewater subject to this regulation includes leachate, gas collection condensate, truck/equipment washwater, drained free liquids, laboratory-derived wastewater, floor washings, and contaminated storm water. Non-contaminated storm water, contaminated ground water, and wastewater from recovering pumping wells are not subject to this regulation.

National estimates of the Landfills industry indicate that only 1,662 of the total population of landfill facilities collect landfill generated wastewater. EPA limited its survey of the industry to those facilities that collect landfill generated wastewater, or about 16 percent of the total number of landfills located in the U.S. Table 3-2 presents the Subtitle D and Subtitle C landfills that collect landfill generated wastewater by ownership type. The national estimates for the industry indicate that approximately 43 percent of these landfills are municipally-owned facilities, 41 percent are commercially-owned, and 13 percent are non-commercial captives. Table 3-2 also shows that the majority of non-hazardous landfills are municipally- or commercially-owned facilities, whereas hazardous landfills are primarily commercially-owned or captive facilities.

3.2.2.1 Captive Landfill Facilities

Based on EPA's survey of the Landfills industry for this guideline, the Agency identified over 200 captive and intra-company facilities that operated landfills. This rule does not apply to captive landfills in most circumstances. See Chapter 2 for EPA's rationale for not including captive landfills under this guideline.

EPA's survey showed that a majority of these landfills were at industrial facilities that are or will be subject to the following three effluent guidelines: Pulp and Paper (40 CFR Part 430), Centralized Waste Treatment (proposed 40 CFR Part 437, 64 FR 2280 January 13, 1999), or Organic Chemicals, Plastics and Synthetic Fibers (40 CFR Part 414). In addition, EPA identified approximately 30 landfills subject to one or more of the following categories: Nonferrous Metals Manufacturing (40 CFR Part 421), Petroleum Refining (40 CFR 419), Timber Products Processing (40 CFR Part 429), Iron and Steel Manufacturing (40 CFR Part 420), Transportation Equipment Cleaning (proposed 40 CFR Part 442, 63 FR 34685 June 25, 1998), and Pesticide Manufacturing (40 CFR Part 455).

Industry supplied data estimates that there are over 118 Pulp and Paper facilities with on-site landfills and that over 90 percent commingle landfill leachate with process wastewater for treatment on site. The wastewater flow originating from landfills typically represents less than one percent of the total flow through the facilities' wastewater treatment plant and, in no case, exceeds three percent of the treated flow. Approximately six percent of pulp and paper mills send landfill generated wastewater to a POTW along with process wastewater.

Based on responses to the "1992 Waste Treatment Industry: Landfills Questionnaire", EPA estimates that there are more than 30 facilities subject to the Organic Chemicals, Plastics, and Synthetic Fibers (OCPSF) guideline with on-site landfills. At OCPSF facilities with on-site landfills, landfill leachate typically represents less than one percent of the industrial flow at the facility, in no case exceeds six percent of the flow, and is typically commingled with process wastewater for treatment.

3.2.3 General Information on Landfill Facilities

EPA estimates that landfill facilities located throughout the U.S. cover approximately 726,000 acres of land area, 20 percent of which is actual disposal area (landfill), 3 percent is for wastewater treatment operations, and 63 percent is undeveloped land. Table 3-3 presents national estimates of the total landfill area covered by non-hazardous and hazardous landfill facilities. National estimates indicate that, as of 1992, hazardous facilities had, on average, used less of their total facility area for waste disposal, only about 5 percent, than non-hazardous facilities, which, on average, had used approximately 30 percent of their total facility area for waste disposal. However, since there are far more non-hazardous landfills in the U.S. than hazardous landfills, Subtitle D landfills have more future capacity than Subtitle C landfills (see Section 3.2.4). Table 3-4 presents facility land area ranges for non-hazardous and hazardous facilities, as well as totals for the industry. These frequency distributions show that a typical facility is 100 to 1,000 acres in size, and the actual landfill covers between 10 and 100 acres of that area. As of 1992, the majority of non-hazardous and hazardous landfill facilities had from 10 acres to 1,000 acres of undeveloped land available; larger facilities had as much as 1,000 to 10,000 acres of undeveloped land.

Landfills are made up of individual cells which may be dedicated to one type of waste or may accept many different types of waste. When a landfill cell reaches capacity volume, it is closed and is referred to as an "inactive" cell. "Active" cells are landfill cells that are not at capacity and continue to accept waste. Table 3-5 presents national estimates of the number of landfill cells, both active and inactive, at non-hazardous and hazardous landfills. National estimates of landfill facilities in the U.S. indicate that the average number of cells in a landfill in 1992 was approximately six. The national average of active cells in 1992 was 2.75, and the national average of inactive cells was 6.05. For hazardous facilities, the average number of cells in 1992 was 7.6, with an average of 4.2 active cells and 8.2 inactive cells. For non-hazardous facilities, the average number of cells in 1992 was 5.7, with an average of 2.5 active cells and 5.4 inactive cells. EPA's survey indicated that there were fewer active landfills in the U.S. than inactive, or closed landfills. As discussed in Section 3.2, a large number of landfills, especially facilities serving small populations, have closed rather than incur the significant expense of complying with RCRA requirements.

The number and type of customers helps to define the size of a landfill. Table 3-6 presents the national estimates of the household and non-household population served by landfills that collect landfill generated wastewater. The total population served by the Landfills industry is 46.3 million household and 5.2 million non-household customers. Non-hazardous landfills serve 99 percent of these customers. Hazardous landfills account for only 307,000 household customers and 170,000 non-household customers. Table 3-7 presents the frequency distributions of the number of household and non-household customers for the non-hazardous and hazardous subcategories as well as for both subcategories combined. Most non-hazardous facilities serve between 100 and 1,000 non-household customers and 10,000 to 100,000 household customers. EPA's survey indicates that hazardous facilities serve between zero and 10,000 non-household customers, but serve very few household customers.

3.2.4 Waste Receipts and Types

Wastes received by landfills in the United States vary from municipal solid waste to highly toxic materials. Table 3-8 presents the national estimates of the types of waste received at landfills and the percentage each waste represents of the total waste received during the following three periods: pre-1980, 1980-1985, and 1986-1992. Sixty-one percent of the waste landfilled during the pre-1980 time period was municipal solid waste and industrial wastes, while 17 percent was commercial solid waste and construction and demolition debris. Similar types of waste were disposed in landfills after 1980; however, the percentage of municipal solid waste and industrial waste decreased, and the amount of commercial solid waste, incinerator residues, PCB/TSCA wastes, and asbestos-containing wastes increased. The disposal in landfills of "other" waste types (such as contaminated soils, auto shredder scrap, and tires) also increased after 1980.

Table 3-9 presents the national estimates of wastes received by the Landfills industry in 1992 by regulatory classification. These data indicate that landfills contained approximately 6.1 billion tons of waste in 1992, and project a future capacity of 8.3 billion tons. However, the estimated future capacity of Subtitle D landfills is much larger than the future capacity of Subtitle C landfills. On average, Subtitle D landfills represent over 97 percent of the future capacity of U.S. landfills.

Table 3-10 presents the national estimates of the annual tonnage of waste accepted by landfills from 1988 through 1992. In 1988, the annual tonnage of waste accepted by Subtitle C and Subtitle D landfills was 221 million tons and, by 1992, the amount of waste accepted annually increased to 315 million tons. The annual tonnage of waste accepted by the entire landfill industry increased 20 percent from 1989 to 1990 and 14 percent from 1990 to 1991. However, when considering Subtitle C landfills alone, EPA's survey found that hazardous landfills experienced a much larger increase in the amount of waste disposed. In 1990, the amount of waste disposed in Subtitle C landfills increased 30 percent from 1989 and, in 1991, the amount of hazardous waste disposed increased 75 percent from 1990. Over the three year period from 1989 to 1991, the annual tonnage of waste landfilled in Subtitle C landfills increased 127 percent. Conversely, the annual tonnage of waste accepted by Subtitle D landfills increased 18 percent from 1989 to 1990 and then increased by only 4 percent from 1990 to 1991. Over the same three year period, from 1989 to 1991, the annual tonnage of waste landfilled in Subtitle D landfills increased by only 23 percent. The greater increase in annual waste deposited in Subtitle C landfills may be the result of more stringent RCRA regulations and stricter waste acceptance criteria (Subtitle C hazardous waste is restricted from being disposed in Subtitle D landfills).

3.2.5 Sources of Wastewater

As noted earlier, a number of landfill operations generate wastewater. In general, the types of wastewater generated by activities include leachate, landfill gas condensate, truck/equipment washwater, drained free liquids, laboratory-derived wastewater, floor washings, storm water, contaminated ground water, and wastewater from recovering pumping wells. Table 3-11 presents the national estimates of the number of landfills that generate each type of wastewater and the minimum, maximum, and median flows. Each of these wastewater sources are discussed below.

3.2.5.1 Landfill Leachate

Landfill leachate is liquid that has passed through or emerged from solid waste and contains soluble, suspended, or miscible materials removed from such waste. Over time, the potential for certain pollutants

to move into the wider environment increases. As water passes through the landfill, it may "leach" pollutants from the disposed waste, moving them deeper into the soil. This presents a potential hazard to public health and the environment through ground water contamination and other means. One measure used to prevent the movement of toxic and hazardous waste constituents from a landfill is a landfill liner operated in conjunction with a leachate collection system. Leachate is typically collected from a liner system placed at the bottom of the landfill. Leachate also may be collected through the use of slurry walls, trenches, or other containment systems. The leachate generated varies from site to site, based on a number of factors including the types of waste accepted, operating practices (including shedding, daily cover and capping), the depth of fill, compaction of wastes, annual precipitation, and landfill age. Based on EPA's survey of the industry, a total of 1,989 landfills generate leachate at flows ranging from one gallon per day to 533,000 gallons per day, with a median daily flow of approximately 5,620 gallons. Landfill leachate is subject to this regulation.

3.2.5.2 Landfill Gas Condensate

Landfill gas condensate is a liquid that has condensed in the landfill gas collection system during the extraction of gas from within the landfill. Gases such as methane and carbon dioxide are generated due to microbial activity within the landfill and must be removed to avoid hazardous, explosive conditions. In the gas collection systems, gases containing high concentrations of water vapor condense in traps staged throughout the gas collection network. The gas condensate contains volatile compounds and accounts for a relatively small percentage of flow from a landfill. The national estimates presented on Table 3-11 report a total of 158 landfills that generate landfill gas condensate at daily flows ranging from 3 gallons to 11,700 gallons. The median flow of landfill gas condensate for the Landfills industry is approximately 343 gallons per day. Landfill gas condensate is subject to the landfills effluent limitations guidelines.

3.2.5.3 Drained Free Liquids

Drained free liquids are aqueous wastes drained from waste containers (e.g., drums, trucks, etc.) or wastewater resulting from waste stabilization prior to landfilling. Landfills that accept containerized waste

may generate this type of wastewater. Wastewater generated from these waste processing activities is collected and usually combined with other landfill generated wastewater for treatment at the wastewater treatment plant. National estimates presented on Table 3-11 identify 33 landfills that generate drained free liquids at a median daily flow of 253 gallons. Daily flows range from a minimum of one gallon per day to a maximum of 82,000 gallons per day. Drained free liquids are subject to the landfills effluent limitations guidelines.

3.2.5.4 Truck and Equipment Washwater

Truck and equipment washwater is generated during either truck or equipment washes at landfills. During routine maintenance or repair operations, trucks and/or equipment used within the landfill (e.g., loaders, compactors, or dump trucks) are washed, and the resultant wastewater is collected for treatment. In addition, it is common practice for many facilities to wash the wheels, body, and undercarriage of trucks used to deliver the waste to the open landfill face upon leaving the landfill. On-site wastewater treatment equipment and storage tanks also are periodically cleaned. It is estimated that 416 landfills generate truck and equipment washwater at a median flow of 118 gallons per day and at daily flows ranging from 5 gallons per day to 15,000 gallons per day.

Floor washings are also generated during routine cleaning and maintenance of landfill facilities. National estimates presented on Table 3-11 indicate there are 70 landfills that generate and collect floor washings at flows ranging from 10 gallons per day to 5,450 gallons per day. The median flow of floor washings for the Landfills industry is approximately 743 gallons per day. Both truck and equipment washwater and floor washings are subject to this rule.

3.2.5.5 Laboratory-Derived Wastewater

Laboratory-derived wastewater is generated from on-site laboratories that characterize incoming waste streams and monitor on-site treatment performance. This source of wastewater is minimal and is usually

combined with leachate and other wastewater prior to treatment at the wastewater treatment plant. Laboratory-derived wastewater is subject to the landfills effluent limitations guidelines.

3.2.5.6 Storm Water

There are two types of storm water, contaminated and non-contaminated. Contaminated storm water is storm water which comes in direct contact with landfill wastes, the waste handling and treatment areas, or wastewater that is subject to the limitations and standards. Some specific areas of a landfill that may produce contaminated storm water include (but are not limited to) the following: the open face of an active landfill with exposed waste (no cover added), the areas around wastewater treatment operations, trucks, equipment or machinery that has been in direct contact with the waste, and waste dumping areas. Non-contaminated (non-contact) storm water is storm water that does not come in direct contact with landfill wastes, the waste handling and treatment areas, or wastewater that is subject to the limitations and standards. Non-contaminated storm water includes storm water which flows off the cap, cover, intermediate cover, daily cover, and/or final cover of the landfill. National estimates indicate that there are 1,135 landfills that generate storm water at flows ranging from 10 gallons per day to 2 million gallons per day, with a median daily flow of approximately 26,800 gallons. Storm water that does not come into contact with the wastes would not be subject to the limitations and standards, as discussed in Chapter 2 of this document.

3.2.5.7 Contaminated Ground Water

Contaminated ground water is water below the land surface in the zone of saturation that has been contaminated by landfill leachate. Contamination of ground water may occur at landfills without liners or at facilities that have released contaminants from a liner system into the surrounding ground water. Ground water also can infiltrate the landfill or the leachate collection system if the water table is high enough to penetrate the landfill area. EPA identified approximately 163 landfills that generate contaminated ground water. Daily flows ranged from 6 gallons per day to 987,000 gallons per day, with a median daily flow of

approximately 12,800 gallons. EPA excluded contaminated ground water from regulation under this guideline as discussed in Chapter 2 of this document.

3.2.5.8 Recovering Pumping Wells

In addition to the contaminated ground water generated during ground water pumping operations, there are various ancillary operations that also generate a wastewater stream. These operations include well construction and development, well maintenance, and well sampling (i.e. purge water). This wastewater will have very similar characteristics to the contaminated ground water. EPA's survey of the Landfills industry identified 50 landfills that generate wastewater from recovering pumping wells. Daily flows range from a minimum of 0.3 gallons to a maximum 80,200 gallons and a median daily flow of 136 gallons. EPA excluded wastewater recovered from pump wells from regulation under this guideline as discussed in Chapter 2 of this document.

3.2.6 Leachate Collection Systems

Most facilities subject to the landfills effluent guidelines generate and collect landfill leachate. To prevent waste material, products of waste decomposition, and free moisture from traveling beyond the limits of the disposal site, landfill facilities utilize some type of leachate collection system. The leachate collection system also reduces the depth of leachate buildup or level of saturation over the liner.

The leachate collection system usually contains several individual components. Two main leachate collection systems may be necessary, an underdrain system and a peripheral system. The underdrain system is constructed prior to landfilling and consists of a drainage system that removes the leachate from the base of the fill. The peripheral system can be installed after landfilling has occurred and, as such, is commonly used as a remedial method. The underdrain system includes a drainage layer of high permeability granular material, drainage tiles to collect the diverted flow laterally, and a low permeability liner underlying the system to retard the leachate that percolates vertically through the unsaturated zone of refuse. Where the leachate meets the low permeability layer, saturated depths of leachate develop and hydraulic gradients govern the leachate flow within the drainage layer (see reference 8).

There are several different types of leachate collection systems employed by the Landfills industry. Table 3-12 presents the different types of leachate collection systems and the national estimates of the number of landfills which employ each system. A simple gravity flow drain field is the most basic and commonly used type of collection system, employed by 50 percent of the industry. According to EPA's 1992 survey, compound leachate collection systems consisting of a liner system and collection pipes are used by 20 percent of the industry. French drains, which are gravel channels used to facilitate leachate drainage, are used by 15 percent of the landfills in the U.S. Other types of leachate collection systems utilized by 10 percent of the Landfills industry include collection sumps and risers, combined gas/leachate extraction wells, perforated toe drains to pump stations, and gravity flow in pipes to a holding pond, basin, or pump station to storage tanks.

3.2.7 Pretreatment Methods

Several types of waste accepted by landfills for disposal may require some type of pretreatment. Wastes that may require pretreatment include free liquids, containerized waste, and bulk wastes. Free liquids may be drained, removed, or stabilized. Containerized waste and bulk wastes may be shredded, stabilized, or solidified. Table 3-13 presents the types of pretreatment methods currently in use by the Landfills industry and national estimates of the number of landfills that pretreat these wastes.

Approximately 75 percent of non-hazardous landfills do not accept free liquids and, of those that do, 20 percent do not pretreat the liquids before treatment at an on-site wastewater treatment facility or treatment off site. In comparison, approximately 65 percent of hazardous landfills accept free liquids and pretreat by stabilizing, draining, or removing the liquid. Forty percent of non-hazardous landfills accept containerized waste, compared to almost 75 percent of hazardous landfills. The most common type of pretreatment for containerized waste is solidification followed by stabilization. Most landfills accept bulk wastes, although many facilities do not pretreat this type of waste. Bulk wastes are usually treated by stabilization or solidification and stabilization. Other types of pretreatment for bulk wastes include compaction, chemical treatment, flocculation, macro/microencapsulation, and recycling.

3.2.8 Baseline Treatment

Many landfills in the United States currently have wastewater treatment systems in place. The most common treatment system used to treat landfill wastewater is biological treatment. However, chemical precipitation and combinations of biological treatment, chemical precipitation, equalization, and filtration also are used widely. Table 3-14 presents the types of treatment and the national estimates of the number of landfills in the industry that employ each type of wastewater treatment. As expected, indirect and zero dischargers often do not employ on-site treatment because they either ship their wastewater off site or use alternate disposal methods such as deep well injection, incineration, evaporation, land application, or recirculation. Chapter 8 presents a detailed discussion of treatment technology and performance.

EPA's survey of the Landfills industry solicited wastewater treatment facility operating information from non-hazardous and hazardous landfills. Table 3-15 presents the national estimates of the number of landfill facilities that operate wastewater treatment systems between 1 and 24 hours per day. Direct and zero or alternative discharge facilities tend to operate treatment systems continuously, whereas many indirect discharge facilities operate less than 24 hours per day. Table 3-16 presents the average daily hours of operation of a typical on-site wastewater treatment facility. Table 3-17 presents the national estimates of the number of landfill facilities that operate wastewater treatment systems between 1 and 7 days per week. Again, direct and zero or alternative discharge facilities commonly operate their treatment systems continuously, whereas indirect dischargers do not. Table 3-18 presents the average number of days per week a typical wastewater treatment facility is in operation.

3.2.9 Discharge Types

EPA's Detailed Technical Questionnaire identified landfills that discharged wastewater directly to a surface water, indirectly to POTWs, and others that disposed of their landfill wastewater through zero or alternative discharge. Direct discharge facilities are those that discharge their wastewater directly to a receiving stream or body of water. Indirect discharging facilities discharge their wastewater indirectly to a POTW. Zero or alternative discharge facilities use treatment and disposal practices that result in no discharge of

wastewater to surface waters. Zero or alternative disposal options for landfill generated wastewater include off-site treatment at another landfill wastewater treatment system or a Centralized Waste Treatment facility, deep well injection, incineration, evaporation, land application, solidification, and recirculation.

Table 3-19 presents the national estimates of the number of landfill facilities grouped by discharge type. These estimates show that the majority of non-hazardous facilities responding to the survey were indirect dischargers, whereas the majority of hazardous facilities were zero dischargers. Although EPA identified hazardous landfills discharging directly to surface waters, none of these facilities are subject to the landfills effluent limitations guidelines.

Table 3-1: Number of Landfills per U.S. State

State	Subtitle D Landfills	Subtitle C Landfills	Total Landfills
Alabama	238	38	276
Alaska	201	1	202
Arizona	90	2	92
Arkansas	134	3	137
California	630	16	646
Colorado	216	12	228
Connecticut	125	22	147
Delaware	8	14	22
Florida	91	9	100
Georgia	277	17	294
Hawaii	15	1	16
Idaho	112	6	118
Illinois	182	14	196
Indiana	101	29	130
Iowa	118	13	131
Kansas	118	8	126
Kentucky	121	33	154
Louisiana	73	17	90
Maine	291	2	293
Maryland	50	5	55
Massachusetts	722	1	723
Michigan	762	9	771
Minnesota	257	4	261
Mississippi	97	3	100
Missouri	128	7	135
Montana	257	1	258
Nebraska	41	8	49
Nevada	127	3	130
New Hampshire	58	0	58
New Jersey	467	8	475
New Mexico	121	7	128
New York	565	10	575
North Carolina	244	39	283
North Dakota	85	1	86
Ohio	119	24	143
Oklahoma	189	7	196
Oregon	231	10	241
Pennsylvania	41	22	63
Rhode Island	12	0	12
South Carolina	127	9	136
South Dakota	193	0	193
Tennessee	112	9	121
Texas	601	70	671
Utah	92	7	99
Vermont	73	0	73
Virginia	440	8	448
Washington	72	9	81
West Virginia	57	5	62
Wisconsin	183	3	186
Wyoming	218	45	263
Puerto Rico	0	3	3
Guam	0	1	1
Total	9,882	595	10,477

Table 3-2: Ownership Status of Landfill Facilities

Ownership Status	Number of Landfill Facilities		
	Subtitle D Non-Hazardous Subcategory	Subtitle C Hazardous Subcategory	Industry Total
Commercial	506	171	677
Non-Commercial (intra-company)	5	48	53
Non-Commercial (captive)	121	94	215
Municipal	708	2	710
Federal Government	4	2	6
Government (other than Federal or Municipal)	0	0	0
Indian Tribal Interest	0	0	0
Other	1	0	1
Total	1,345	317	1,662

Table 3-3: Total Landfill Facility Area

Facility Land Type	Landfill Facility Area (acres)		
	Subtitle D Non-Hazardous Subcategory	Subtitle C Hazardous Subcategory	Industry Total
Total Facility Area	416,733	309,194	725,927
Wastewater Treatment Area	9,424	10,147	19,571
Waste Disposal Area (landfill)	119,700	16,552	136,323
Undeveloped Land	254,610	207,085	459,811

Table 3-4: Landfill Facility Land Area Ranges

Subcategory	Land Area Range (acres)	Number of Landfill Facilities			
		Total Facility Area	Wastewater Treatment Area	Waste Disposal Area (landfill)	Undeveloped Land
All Facilities	0	0	747	28	110
	>0-1	0	320	16	2
	>1-10	9	437	126	69
	>10-100	490	136	1,128	561
	>100-1,000	1,044	22	362	745
	>1,000-10,000	119	0	0	85
	Total	1,662	1,662	1,660	1,662
Subtitle C Hazardous	0	0	38	5	49
	>0-1	0	128	14	0
	>1-10	2	70	47	2
	>10-100	95	65	199	99
	>100-1,000	136	15	52	106
	>1,000-10,000	84	0	0	60
	Total	317	316	317	316
Subtitle D Non-Hazardous	0	0	708	23	61
	>0-1	0	191	2	2
	>1-10	7	366	79	67
	>10-100	395	72	930	551
	>100-1,000	909	7	310	638
	>1,000-10,000	34	0	0	25
	Total	1,345	1,344	1,344	1,344

Table 3-5: Number of Landfill Cells

Subcategory	Type of Landfill Cell	Number of Cells	
		Estimated Mean	Estimated Total
All Facilities	Total cells	6.12	13,299
	Active cells	2.75	4,608
	Inactive cells	6.05	8,690
Subtitle C Hazardous	Total cells	7.64	3,776
	Active cells	4.23	1,112
	Inactive cells	8.24	2,663
Subtitle D Non-Hazardous	Total cells	5.68	9,523
	Active cells	2.48	3,496
	Inactive cells	5.41	6,027

Table 3-6: Household and Non-Household Population Served

Population Served	Number of Customers		
	Subtitle D Non-Hazardous Subcategory	Subtitle C Hazardous Subcategory	Industry Total
Non-Household	5,043,542	170,420	5,213,962
Household	46,007,775	307,243	46,315,018

Table 3-7: Household vs. Non-Household Customers

Number of Non-Household Customers	Number of Landfill Facilities		
	Subtitle D Non-Hazardous Subcategory	Subtitle C Hazardous Subcategory	Industry Total
0	76	123	205
1	83	40	124
>1-10	33	12	45
>10-100	202	4	203
>100-1,000	544	87	628
>1,000-10,000	351	51	400
>10,000-100,000	55	0	54
>100,000-1,00,000	2	0	2
Total	1,346	317	1,661
Number of Household Customers			
0	180	313	506
1	0	0	0
>1-10	55	0	55
>10-100	29	0	28
>100-1,000	42	0	42
>1,000-10,000	195	2	195
>10,000-100,000	742	0	733
>100,000-1,00,000	102	2	103
Total	1,345	317	1,662

Table 3-8: Wastes Received by Landfills in the United States

Waste Type	Mean % for Time Period Pre-1980	Mean % for Time Period 1980-85	Mean % for Time Period 1986-92
Municipal Solid Waste	38.3	33.4	33.9
Household Hazardous Waste	0.217	0.218	0.215
Yard Waste	4.76	4.39	3.76
Commercial Solid Waste	8.56	9.92	9.94
Institutional Wastes	1.36	1.43	2.14
Industrial Wastes	22.8	19.6	17.4
Agricultural Waste	0.340	0.297	0.284
Pesticides	0.033	0.009	0.321
PCB, TSCA Wastes	0.192	1.12	0.980
Asbestos-Containing Waste	0.905	3.73	3.42
Radioactive Waste	0.019	0.002	0.001
Medical or Pathogenic Waste	0.255	0.182	0.123
Superfund Clean-Up Wastes	0.000	0.021	0.014
Mining Wastes	0.519	0.47	0.180
Incinerator Residues	1.01	1.43	3.14
Fly Ash, Not Incinerator Waste	4.49	5.82	6.30
Construction/Demolition Debris	8.40	5.91	7.95
Sewage Sludge	1.81	3.15	2.88
Dioxin Waste	0.000	0.039	0.024
Other Sludge	4.89	4.90	2.91
Other Waste Types	1.23	4.49	5.25
Industry Total	100.09	100.528	101.132

Table 3-9: Total Volume of Waste Received by Landfills in 1992 by Regulatory Classification

Time Frame	Regulatory Class	All Facilities		Subtitle C Hazardous Subcategory		Subtitle D Non-Hazardous Subcategory	
		Estimated Total Number Landfills	Total Volume Landfilled (tons)	Estimated Total Number Landfills	Total Volume Landfilled (tons)	Estimated Total Number Landfills	Total Volume Landfilled (tons)
Current	Pre 1980	561	954,273,421	190	155,418,921	370	798,854,500
	RCRA Subtitle C	333	159,252,888	323	158,994,443	10	258,445
	RCRA Subtitle D	906	1,501,319,521	115	249,656,514	791	1,251,663,007
	TSCA	108	53,167,884	102	52,654,468	6	513,416
	NRC	.	.	.	.	.	.
	Local Regulation	461	2,365,983,720	57	6,374,393	404	2,359,609,326
	CERCLA	4	10,507,627	2	72,587	2	10,435,040
	Other Regulation	560	1,018,656,724	114	36,250,349	446	982,406,374
	Total Volume Landfilled	2,146	6,063,161,789	491	659,421,679	1,655	5,403,740,110
			Future Capacity (tons)		Future Capacity (tons)		Future Capacity (tons)
Future	Pre 1980	86	101,032,485	.	.	86	101,032,485
	RCRA Subtitle C	201	66,313,422	193	65,192,737	8	1,120,685
	RCRA Subtitle D	884	6,056,763,187	33	96,321,683	851	5,960,441,504
	TSCA	34	11,202,929	28	10,897,045	6	305,884
	NRC	2	300,860	.	.	2	300,860
	Local Regulation	293	962,479,373	57	4,710,196	236	957,769,177
	CERCLA	50	4,297,618	50	4,297,618	.	.
	Other Regulation	501	1,126,823,595	127	30,749,439	374	1,096,074,156
	Total Volume Landfilled	1,706	8,329,213,474	266	212,168,721	1,441	8,117,044,753

Table 3-10: Annual Tonnage of Waste Accepted by Landfills

Year	Annual Tonnage of Waste (tons)		
	Subtitle D Non-Hazardous Subcategory	Subtitle C Hazardous Subcategory	Industry Total
1988	185,184,608	36,305,235	221,489,843
1989	196,377,576	28,867,681	225,245,257
1990	232,535,432	37,413,692	269,949,125
1991	241,454,300	65,402,768	306,857,068
1992	252,101,069	63,022,850	315,123,919

Table 3-11: Wastewater Flows Generated by Individual Landfills

Type of Wastewater Generated	Number of Landfills	Minimum Average Flow (gal/day)	Maximum Average Flow (gal/day)	Industry Median (gal/day)
Floor washing	70	10	5,450	743
Landfill leachate	1,989	1	533,000	5,620
Contaminated ground water	163	6	987,000	12,800
Storm water run-off	1,135	10	2,067,000	26,800
Landfill gas condensate	158	3	11,700	343
Recovering pumping wells	50	0.3	80,200	136
Truck/equipment washwater	416	5	15,000	118
Drained free liquids	33	1	82,000	253
Other	2	0	0	0
Total	4,016			

Table 3-12: Type of Leachate Collection Systems Used at Individual Landfills

Type of Leachate Collection	Number of Landfills		
	Subtitle D Non-Hazardous Subcategory	Subtitle C Hazardous Subcategory	Industry Total
None	46	87	132
Simple Gravity Flow Drain Field	977	266	1,242
French Drain System	341	38	379
Compound Leachate Collection	416	93	509
Suction Lysimeters	0.	2	2
Other	196	49	246
Total	1,976	535	2,510

Table 3-13: Pretreatment Methods in Use at Individual Landfills

Type of Waste	Pretreatment Method	Number of Landfills		
		Subtitle D Non-Hazardous Subcategory	Subtitle C Hazardous Subcategory	Industry Total
Free Liquids	No Pretreatment	324	113	437
	None Accepted	1,277	283	1,560
	Drained or Removed	51	115	166
	Stabilization	38	172	211
	Other	17	84	101
	Total	1,707	767	2,475
Containerized Waste	No Pretreatment	515	100	616
	None Accepted	1,008	180	1,188
	Shredded	23	70	94
	Stabilized	6	135	141
	Solidified	41	138	179
	Other	110	80	190
	Total	1,703	703	2,408
Bulk Wastes	No Pretreatment	993	216	1,209
	None Accepted	414	61	475
	Baled	33	2	35
	Shredded	82	49	131
	Stabilized	15	201	216
	Solidified	74	126	200
	Other	100	38	138
	Total	1,711	693	2,404

Table 3-14: Types of Wastewater Treatment Employed by the Landfills Industry

Type of Treatment	Number of Landfills		
	Direct Discharge	Indirect Discharge	Zero Discharge
No treatment	81	691	468
Biological treatment	119	37	19
Chemical precipitation	63	45	8
Chemical precipitation and biological treatment	32	10	0
Filtration and biological treatment	45	4	5
Equalization and biological treatment	65	28	7
Equalization, biological treatment, and filtration	37	4	5
Equalization, chemical precipitation, and biological treatment	26	8	0
Equalization, chemical precipitation, biological treatment, and filtration	26	2	0

Table 3-15: Wastewater Treatment Facility Hours of Operation per Day

Hours of Operation (hours/day)	Subtitle D Non-Hazardous Subcategory			Subtitle C Hazardous Subcategory			Industry Total		
	Direct	Indirect	Zero	Direct	Indirect	Zero	Direct	Indirect	Zero
0	0	0	0	0	0	0	0	0	0
1-23	11	295	40	11	4	6	23	275	42
24	161	508	330	122	20	153	283	552	488
Total	172	803	370	133	24	159	306	827	530

Table 3-16: Wastewater Treatment Facility Average Hours of Operation per Day

Subcategory	Average Hours of Operation/Day		
	Direct Discharge	Indirect Discharge	Zero Discharge
All Facilities	22.80	19.16	22.55
Subtitle C Hazardous	22.78	22.18	23.46
Subtitle D Non-Hazardous	22.83	18.52	21.89

Table 3-17: Wastewater Treatment Facility Days of Operation per Week

Days of Operation (days/week)	Subtitle D Non-Hazardous Subcategory			Subtitle C Hazardous Subcategory			Industry Total		
	Direct	Indirect	Zero	Direct	Indirect	Zero	Direct	Indirect	Zero
0	0	0	0	0	0	0	0	0	0
1-6	7	225	40	19	2	6	30	203	42
7	165	578	330	115	22	153	275	624	488
Total	172	803	370	134	24	159	305	827	530

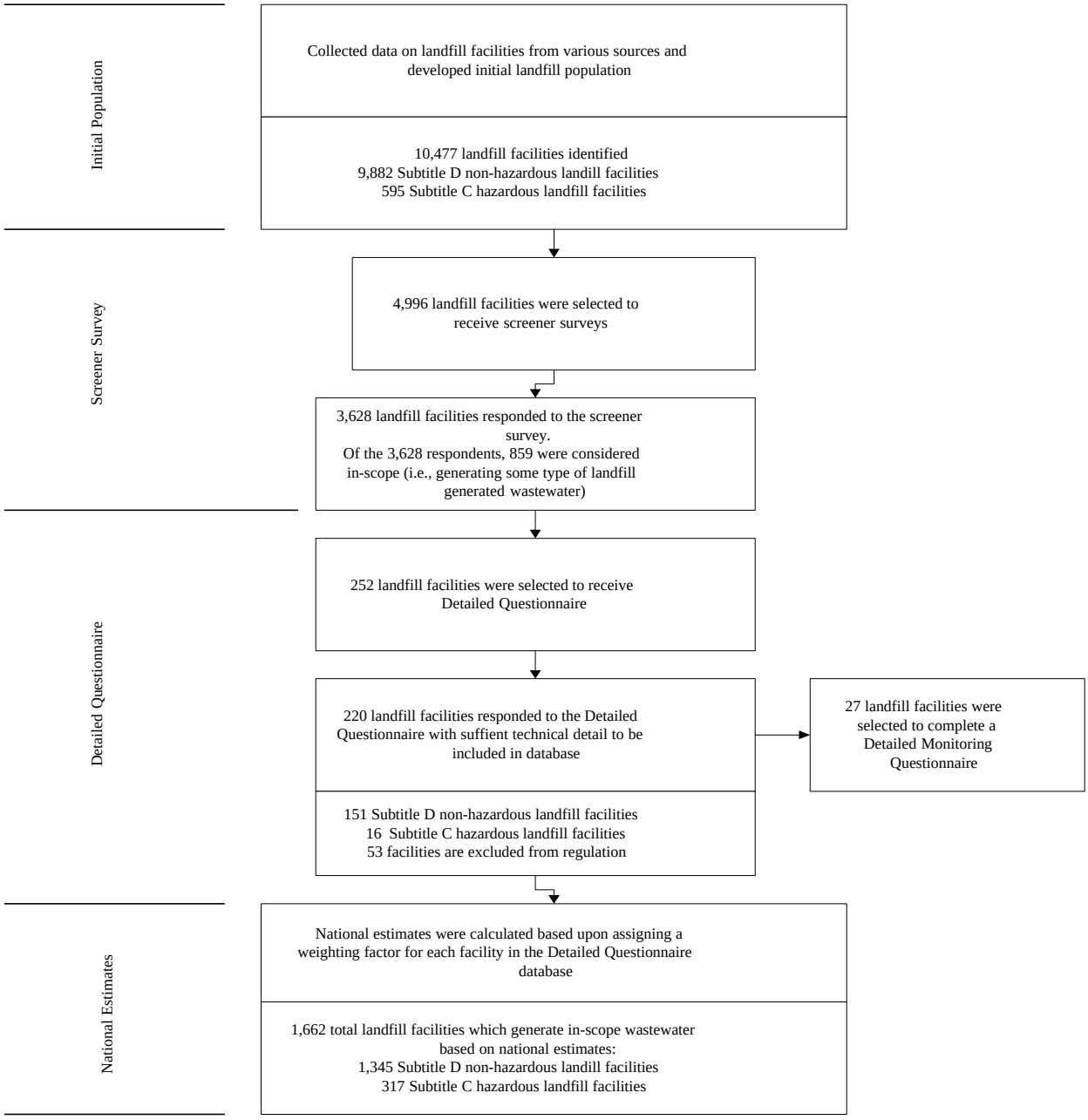
Table 3-18: Wastewater Treatment Facility Average Days of Operation per Week

Subcategory	Average Days of Operation/Week		
	Direct Discharge	Indirect Discharge	Zero Discharge
All Facilities	6.72	6.47	6.81
Subtitle C Hazardous	6.56	6.83	6.77
Subtitle D Non-Hazardous	6.94	6.39	6.84

Table 3-19: Total Number of Facilities by Discharge Type

Subcategory	Discharge Type			Total
	Direct	Indirect	Zero	
All Facilities	306	827	529	1,662
Subtitle C Hazardous	134	24	159	317
Subtitle D Non-Hazardous	172	803	370	1,345

Figure 3-1: Development of National Estimates for the Landfills Industry



4.0 DATA COLLECTION ACTIVITIES

4.1 Introduction

As part of the development of the Landfills effluent guideline, EPA collected data from a variety of different sources. These sources included existing data from previous EPA and other governmental data collection efforts, industry-provided information, new data collected from questionnaire surveys, and field sampling data. This chapter discusses each of these data sources, as well as EPA's quality assurance/quality control (QA/QC) efforts and data editing procedures. Chapters 5 through 11 present summaries and analyses of the data collected by EPA.

4.2 Preliminary Data Summary

EPA's initial effort to develop effluent limitations guidelines and pretreatment standards for the waste treatment industry began in 1986. EPA conducted a study of the hazardous waste treatment industry in which it determined the scope of the industry, the operations performed, the type of wastewater generated, and types of discharges. For this study, EPA looked at a hazardous waste treatment industry that included landfills with leachate collection and treatment facilities, incinerators with wet scrubbers, and aqueous hazardous waste treatment facilities. This study characterized the wastewater generated by facilities in the industry and the wastewater treatment technologies used to treat this wastewater. In addition, the study included industry profiles, the cost of wastewater control and treatment, and environmental assessments. EPA published the results of this study in a report entitled "Preliminary Data Summary for the Hazardous Waste Treatment Industry" (EPA 440/1-89-100), in September, 1989.

The Agency used data from the following sources in developing the preliminary data summary:

- EPA Office of Research and Development databases (includes field sampling data from 13 hazardous waste landfills in 1985).
- State Agencies (includes a Wisconsin sampling program of 20 municipal landfills in 1983).

- EPA Office of Emergency and Remedial Response Contract Laboratory Program (CLP) Statistical Database, “Most Commonly Occurring Analytes in 56 Leachate Samples.” 1980-83 data.

- National Enforcement Investigations Center (NEIC) sampling program conducted for the Hazardous Waste Groundwater Task Force during 1985.

- EPA sampling at 6 landfill facilities (1986-1987).

- Subtitle D leachate data for miscellaneous Subtitle D landfills, compiled by the EPA Office of Solid Waste.

The EPA Preliminary Data Summary identified 911 landfills that generate leachate. Of these, 173 discharged their leachate directly to surface waters, while 355 discharged indirectly through publicly owned treatment works (POTWs). The remaining 383 used other methods of leachate disposal. The most common "other" disposal method was contract hauling to a commercial aqueous waste treatment facility. However, some facilities land-applied their leachate (spraying of the leachate over the landfill) or injected it into a deep well for disposal.

The key findings of the EPA Preliminary Data Summary included:

- Some leachates were found to contain high concentrations (e.g., over 100,000 micrograms per liter (µg/l)) of toxic organic compounds.

- Raw leachates were found to contain high concentrations of BOD_5, COD, and TOC.

- Leachate flow rates varied widely due to climatic and geological conditions and landfill size. An average landfill was estimated to have a leachate generation rate of approximately 30,000 gallons per day (gpd).

- As a result of Resource Conservation and Recovery Act (RCRA) regulations, the number of leachate collection systems used at landfills was expected to increase.

- RCRA regulations also would cause solid waste generators to increase their use of commercial landfill facilities.

EPA found that a wide range of biological and physical/chemical treatment technologies were in use by landfills, capable of removing high percentages of conventional, nonconventional, and toxic pollutants. Advanced treatment technologies identified in this study include air stripping, ammonia stripping, activated carbon, and lime precipitation.

After a thorough analysis of the landfill data presented in the Preliminary Data Summary, EPA identified the need to develop an effluent guidelines regulation for the Landfills industry in order to set national guidelines and standards. EPA based its decision to develop effluent limitations guidelines on the Preliminary Data Summary's assessment of the current and future trends in the Landfills industry, its analysis of the concentrations of pollutants in the raw leachate, and the study's discussion on the treatment and control technologies available for effective pollution reduction in landfill leachate.

4.3 Clean Water Act (CWA) Section 308 Questionnaires

A major source of information and data used in developing effluent limitations guidelines and standards consisted of industry responses to detailed technical and economic questionnaires, and the subsequent detailed monitoring questionnaires, distributed by EPA under the authority of Section 308 of the CWA. These questionnaires requested information on each facility's industrial operations, ownership status, solid wastes disposed, treatment processes employed, and wastewater discharge characteristics. EPA first developed a database of various types of landfills in the United States using information collected from the following: 1) State environmental and solid waste departments, 2) other State agencies and contacts, 3) the National Survey of Hazardous Waste Treatment Storage, Disposal and Recycling Facilities respondent list, 4) Environmental Ltd.'s 1991 Directory of Industrial and Hazardous Waste Management Firms, 5) the Resource Conservation and Recovery Act (RCRA) 1992 list of Municipal Landfills, and 6) the Resource Conservation and Recovery Information System (RCRIS) National Oversight Database. Based upon these sources, EPA identified 10,477 landfill facilities in the U.S. in 1992. Of this group, 9,882 were Subtitle D landfills while 595 were Subtitle C landfills.

EPA entered all of these facilities into a database which served as the initial population for EPA to collect industry-provided data. EPA's data collection process involved the following three stages:

- Screener Surveys
- Detailed Technical Questionnaires
- Detailed Monitoring Questionnaires

The following sections discuss each of these data collection activities. A more detailed discussion of the landfills survey population can be found in Appendix A.

4.3.1 Screener Surveys

EPA developed a screener survey to collect data on all of the landfill sites in the U.S. identified by the sources above.

4.3.1.1 Recipient Selection and Mailing

EPA divided the 10,477 facilities into four strata for the purpose of determining the screener survey recipients. The Agency defined these strata as the following:

1. Subtitle C facilities.
2. Subtitle D facilities that are known wastewater generators.
3. Subtitle D facilities in states with less than 100 landfills and are not known to be wastewater generators.
4. Subtitle D facilities in states with more than 100 landfills and are not known to be wastewater generators.

The Agency decided that all of the facilities in strata 1, 2, and 3 would receive the screener survey, while only a random sample of the facilities in stratum 4 would receive the survey. Table 4-1 presents the sample frame, number of facilities sampled, and the number of respondents to receive the screener survey.

Table 4-1: Screener Questionnaire Strata

Screener Stratum (g)	Number in Frame (N_g)	Number Sampled (n_g)	Number of Responses (n'_g)
1	595	595	524
2	134	134	120
3	892	892	722
4	8,856	3,375	2,621
Total	10,477	4,996	3,987

4.3.1.2 Information Collected

Information collected by the screener surveys included the following:

- mailing address.
- landfill type, including types and amount of solid waste disposed and landfill capacity.
- wastewater generation rates as a result of landfill operations, including leachate, gas condensate, and contaminated ground water.
- regulatory classification and ownership status.
- wastewater discharge status.
- wastewater monitoring practices.
- wastewater treatment technology in use.

4.3.1.3 Data Entry, Coding, and Analysis

EPA operated a toll-free help line to assist the screener recipients with filling out the 3-page survey. The Agency responded to several thousand phone calls from facilities over a six week period. The help line answered questions regarding applicability, EPA policy, and economic and technical details.

EPA reviewed all screener surveys returned to the Agency to verify that each respondent completed the critical questions in the survey (e.g., wastewater generation and collection, number and types of landfills, discharge status, and wastewater treatment technology). The screeners were in a bubble-sheet format and were scanned directly into a computer database. Once entered, EPA checked the database for logical inconsistencies and contacted facilities to resolve any inconsistencies.

After the QA process, EPA divided the facilities in the database into the following two groups: 1) facilities that indicated they collected landfill generated wastewater; and 2) those that did not. EPA considered facilities that did not collect landfill generated wastewater to be out of the scope of this regulation and therefore did not investigate these facilities any further.

4.3.1.4 Mailout Results

Of the 4,996 screener questionnaires mailed by EPA, 3,628 responded, and of those, EPA determined that 3,581 were potentially in-scope and complete . The Agency entered these surveys into the screener database. Of these, EPA identified 859 facilities that generate and collect one or more types of landfill generated wastewater.

4.3.2 Detailed Technical Questionnaires

Once EPA analyzed the information from the screener surveys in the database, EPA developed a detailed technical and economic questionnaire to obtain more information from facilities that collect landfill generated wastewater.

4.3.2.1 Recipient Selection and Mailing

EPA used the 859 facilities that generated and collected landfill wastewater from the screener database, plus one pre-test questionnaire facility that was not in the screener database, as the frame for selection of facilities to be sent a Detailed Questionnaire. EPA divided these facilities into the following eight strata:

1. Commercial private, municipal, or government facilities that have wastewater treatment and are direct or indirect dischargers.
2. Commercial private, municipal, or government facilities that have wastewater treatment and are not direct or indirect dischargers.
3. Non-commercial private facilities with wastewater treatment
4. Facilities with no wastewater treatment
5. Commercial facilities that accept PCB wastes
6. Municipal hazardous waste facilities
7. Small businesses with no wastewater treatment
8. Pre-test facilities that were not in the screener population

The Agency decided all facilities in strata 1, 5, 6, 7, and 8 would receive the Detailed Questionnaire. EPA sent the Detailed Questionnaire to a random sample of the facilities in strata 2, 3, and 4.

These selection criteria resulted in a mailing of the Detailed Questionnaire to 252 facilities. Chapter 3, Section 3.2.1 briefly discusses the population analysis (referred to as national estimates) conducted from these questionnaire recipients.

4.3.2.2 Information Collected

The Detailed Questionnaire solicited technical and costing information regarding landfill operations at the

selected facilities. EPA divided the questionnaire into the following four main sections:

- Section A - Facility Identification and Operational Information:

    1. General facility information, including the following: ownership status, landfill type, the number of landfills on site, regulatory status, discharge status, when the landfill began accepting waste, and projected closure date.

    2. Landfill operation, including the following: types of waste accepted at the landfill, the amount of waste accepted, landfill capacity, how the waste was organized in the landfill, landfill caps, and landfill liners.

    3. Wastewater generation from landfill operations, including the following: the types of wastewater generated and the generation rates, and the ultimate disposal of the wastewater generated and collected.

- Section B - Wastewater Treatment:

    1. Description of treatment methods employed by the facility to treat the wastewater identified in Section A. This description includes a discussion of commingled wastewater, wastewater treatment technologies, residual waste disposal, and treatment plant capacities.

- Section C - Wastewater Monitoring Data:

    1. A summary of the monitoring data pertaining to the landfill generated wastewater identified in Section A that were collected in 1992 by the facility, including the following: minimum, maximum, averages, number of observations, and sampling and analytical methods.

- Section D - Detailed Wastewater Treatment Design Information:

    1. Detailed technical design, operation and costing information pertaining to the wastewater treatment technologies identified in Section B.

4.3.2.3 Data Entry, Coding, and Analysis

EPA operated a toll-free help line to assist the questionnaire recipients with filling out the Detailed Questionnaire. EPA responded to over one thousand phone calls from facilities over a three-month period.

While some calls pertained to questions of applicability, most were of a technical nature regarding specific questions in the questionnaire.

Once EPA received the completed questionnaires, the Agency thoroughly reviewed each one for technical accuracy and content. After review, the questionnaire was coded for double-key entry into the questionnaire database. EPA resolved all discrepancies between the two inputted values by referring to the original questionnaire.

EPA followed several QA/QC procedures when developing the questionnaire database, including a manual completeness and accuracy check of a random selection of 20 percent of the questionnaires and a database logic check of each completed questionnaire. These QA/QC procedures helped verify the questionnaires for completeness, resolve any internal consistencies, and identify outliers in the data. EPA checked all outliers for accuracy.

4.3.2.4 Mailout Results

Of the 252 recipients, 220 responded with sufficient technical and economic data to be included in the final EPA Detailed Questionnaire database.

4.4 Detailed Monitoring Questionnaire

In addition to the Detailed Questionnaire, EPA also requested detailed wastewater monitoring information from 27 facilities included in the Detailed Questionnaire database via a Detailed Monitoring Questionnaire.

4.4.1 Recipient Selection and Mailing

EPA selected facilities to receive Detailed Monitoring Questionnaires based upon their responses to the Detailed Questionnaire. EPA reviewed each facility's monitoring summary, discharge permit requirements, and their on-site treatment technologies. From these responses, EPA selected 27 facilities to receive a

Detailed Monitoring Questionnaire which could provide useful information on technology performance, pollutant removals, and wastewater characterization.

4.4.2 Information Collected

EPA requested recipients of the Detailed Monitoring Questionnaire to send analytical data (1992, 1993, and 1994 annual data) on daily equalized influent to their wastewater treatment system, as well as effluent data from the treatment system. The three years of analytical data assisted EPA in calculating the variability factors (discussed in Chapter 11) used in calculating the industry effluent limits. EPA also requested analytical data for intermediate waste treatment points for some facilities. In this manner, EPA was able to obtain performance information across individual treatment units in addition to the entire treatment train.

4.4.3 Data Entry, Coding, and Analysis

EPA conducted a thorough review of each Detailed Monitoring Questionnaire response to ensure that the data provided was representative of the facility's treatment system. EPA collected data from 24 semi-continuous and continuous treatment systems and 2 batch treatment systems. The Agency developed a Detailed Monitoring Questionnaire database which included all monitoring data submitted by the selected facilities.

4.5 Engineering Site Visits

EPA visited 19 facilities, including one facility outside the U.S. The purpose of these visits was to evaluate each facility as a potential week-long sampling candidate to collect treatment performance data. EPA selected these facilities based on the responses to the Detailed Questionnaire and attempted to include facilities from a broad cross section of the industry. EPA visited landfills of various ownership status (municipal, commercial, captive), landfills that accept various waste types (construction and demolition, ash, sludge, industrial, municipal, hazardous), and landfills in different geographic regions of the country. Facilities selected for engineering site visits employed various types of treatment processes, including the

following: equalization, chemical and biological treatment, filtration, air stripping, steam stripping, and membrane separation.

EPA visited each landfill for one day. During the engineering site visit, EPA obtained information on the following:

- the facility and its operations.
- the wastes accepted for treatment and the facility's acceptance criteria.
- the raw wastewater generated and its sources.
- the wastewater treatment on site.
- the location of potential sampling points.
- the site-specific sampling needs (access to facility and sample points, and required sampling safety equipment).

Table 4-2 presents a summary of the types of landfill facilities that EPA included in the engineering site visits.

4.6 Wastewater Characterization Site Visits

While conducting engineering site visits, EPA also collected samples for raw wastewater characterization at 15 landfills. EPA collected grab samples of untreated wastewater at various types of landfills and analyzed for constituents in the wastewater including conventionals, metals, organics, pesticides and herbicides, PCBs, and dioxins and furans. Chapter 6 presents the characterization data obtained by EPA.

Table 4-2 also presents a summary of the type of landfill facilities that EPA included in the characterization site visits and the number of wastewater characterization samples collected.

4.7 EPA Week-Long Sampling Program

To collect wastewater treatment performance data, EPA conducted week-long sampling efforts at six landfills. EPA selected these facilities based on the analysis of the information collected during the engineering site visits. Table 4-3 presents a summary of the types of landfills sampled and treatment technologies evaluated.

EPA prepared a detailed sampling plan for each sampling episode. The Agency collected wastewater samples at influent, intermediate, and effluent sample points throughout the entire on-site wastewater treatment system. Sampling at five of the facilities consisted of 24-hour composite samples for five consecutive days. For the sixth facility, EPA took composites of four completed batches over five days. At all facilities, the Agency collected individual grab samples for oil and grease. Volatile organic grab samples were composited in the laboratory prior to analysis.

EPA analyzed samples using EPA Office of Water approved analytical methods. The following table presents the pollutant group and the analytical method used:

Pollutant Group	Analytical Method
Conventional and Nonconventionals	Standard Methods
Metals	EPA 1620
Organics	EPA 1624, 1625
Herbicides, Pesticides, PCBs	EPA 1656, 1657,1658
Dioxins/Furans	EPA 1613

EPA used influent data to characterize raw wastewater for the industry and develop the list of pollutants of interest (see Chapter 6 for raw wastewater characterization and Chapter 7 for pollutant of interest selection). The Agency used influent, intermediate, and effluent data to evaluate performance of the wastewater treatment systems and develop current discharge concentrations, pollutant loadings, and the

best available treatment (BAT) options for the Landfills industry. EPA used effluent data to calculate long-term averages for each of the regulatory options.

Table 4-4 presents the facilities included in the engineering site visits, the raw wastewater characterization sampling effort, and the week-long sampling effort. Note that facilities utilized only for the engineering site visits do not have sampling episode numbers.

4.8 Other Data Sources

In addition to the original data collected by EPA, the Agency used other data sources to supplement the industry database. Each of these data sources is discussed below.

4.8.1 Industry Supplied Data

EPA requested the Landfills industry to provide relevant information and data. The Agency received leachate and ground water characterization and treatability studies from several facilities, including 25 discharge monitoring report (DMR) data packages. EPA used industry-supplied data to characterize the industry, develop pollutant loadings, and develop effluent limitations.

4.8.2 Comprehensive Environmental Response, Compensation and Liability Act (CERCLA)/Superfund Amendments and Reauthorization Act (SARA) Ground Water Data

EPA obtained ground water data from the "CERCLA Site Discharges To POTWs Treatability Manual" (EPA 540/2-90-007), prepared by the Industrial Technology Division of the EPA Office of Water Standards and Regulations for the EPA Office of Emergency and Remedial Response. The Agency used data from this study to supplement the ground water data collected during characterization and week-long sampling events. The purpose of the CERCLA study was to do the following:

- Identify the variety of compounds and concentration ranges present in ground water at CERCLA sites.

- Collect data on the treatability of compounds achieved by various on-site pretreatment systems.

- Evaluate the impact of CERCLA discharges to a receiving POTW.

For the CERCLA study, a total of eighteen CERCLA facilities were sampled. However, EPA only used data from facilities that received ground water contaminated as a result of landfilling activities in its analysis of contaminated ground water at landfill facilities. Based in part on this data and for the reasons discussed in Chapter 2, EPA decided not to include contaminated ground water as a regulated wastewater under this guideline. In addition, for the proposal, EPA combined the data from seven CERCLA facilities with EPA sampling data to help characterize the hazardous subcategory and to develop both the current discharge concentrations and pollutant loadings for facilities in the hazardous subcategory. However, since EPA did not include contaminated ground water as a wastewater subject to this guideline, for the final rule, EPA removed all CERCLA data from the Subtitle C raw wastewater characterization database. The data presented in subsequent chapters for hazardous wastewater characterization do not include CERCLA data.

4.8.3 POTW Study

The primary source of POTW percent removal data was the "Fate of Priority Pollutants in Publicly Owned Treatment Works" (EPA 440/1-82-303), commonly referred to as the "50-POTW Study." The 50-POTW Study presents data on 50 well-operated POTWs with secondary treatment in removing toxic pollutants. At most of these plants, a minimum of 6 days of 24-hour sampling of influent, effluent, and sludge streams was completed. Each sample was analyzed for conventional, selected non-conventional, and priority pollutants. The basic objective of the study was to generate, compile, and report data on the occurrence and fate of the 129 priority toxic pollutants in 50 POTWs. Preliminary evaluations of the data were also conducted. The report presents all of the collected data, results of the preliminary evaluations, and results of the calculations to determine the following: 1) the concentrations of priority pollutants in the

influent to POTWs, 2) the concentrations of priority pollutants discharged from the POTWs, 3) the concentrations of priority pollutants in the effluent from intermediate process streams, and 4) the concentrations of priority pollutants in the POTW sludge streams.

Some of the data collected for evaluating POTW removals in the 50-POTW Study included influent levels of pollutants that were close to the detection limit. EPA eliminated these values to reduce the possibility that low POTW removals might simply reflect low influent concentrations instead of being a true measure of treatment effectiveness. For further discussion on the editing rules EPA applied to the 50-POTW Study for use in the assessment of POTW removal, see Chapter 7, Section 7.7.1.

4.8.4 National Risk Management Research Laboratory Data

EPA's National Risk Management Research Laboratory (NRMRL) developed a treatability database (formerly called the Risk Reduction Engineering Laboratory (RREL) database). This computerized database provides information, by pollutant, on removals obtained by various treatment technologies. The database provides the user with the specific data source and the industry from which the wastewater was generated. EPA used the NRMRL database to augment the POTW database for certain pollutants which the 50-POTW Study did not evaluate. EPA edited the NRMRL data so that only treatment technologies representative of typical POTW secondary treatment operations were used. Additional edits applied by EPA are discussed in detail in Chapter 7, Section 7.7.1.

4.9 QA/QC and Other Data Editing Procedures

This section presents the quality assurance/quality control (QA/QC) procedures and editing rules used to analyze the different analytical data sets described in the previous sections (e.g., industry supplied data, Detailed Questionnaire data, Detailed Monitoring Questionnaire data, EPA field sampling, and analytical data collected by other EPA offices). For a complete discussion of all of the conventions used in calculating effluent limitations see the "Statistical Support Document for Final Effluent Limitations Guidelines and Standards for the Landfills Point Source Category" (EPA-821-B-99-007).

4.9.1 QA/QC Procedures

Each analytical data source received a QA/QC review before being included in the EPA analytical, Detailed Questionnaire, and Detailed Monitoring Questionnaire databases. The specific QA/QC activities completed for each analytical data source are discussed below.

4.9.2 Analytical Database Review

EPA's Sample Control Center (SCC) developed and maintained the analytical database, and provided a number of QA/QC functions. SCC documented the results of the QA/QC procedures in data review narratives. EPA then performed completeness checks to ensure the completeness of the analytical database. Both of these QA/QC activities are discussed below. In addition, the following paragraphs outline the editing procedures and data conventions used to finalize the landfill analytical database, to characterize each industry subcategory, and to develop current discharge information and pollutant loadings.

4.9.2.1 Data Review Narratives

The Sample Control Center performed a QA/QC data review and documented its findings in the data review narrative that accompanied each laboratory data package. The data review narrative identified missing data and any other data discrepancies encountered during the QA/QC review. EPA then checked the narratives against the data and sampling episode traffic reports to make sure SCC did not overlook any data discrepancies.

4.9.2.2 Completeness Checks

EPA performed a data completeness check of the analytical database by cross referencing the list of pollutants requested for analysis with the list of pollutants the laboratory actually analyzed at each sample point. To accomplish this, EPA prepared the following:

- a list of all requested analytical methods and method numbers.
- a list of all pollutants and CAS numbers specified under each requested analytical method.
- a schedule of analyses requested by episode for each sample point.

The purpose of the completeness check was to verify that the laboratory performed all of the analyses requested and that SCC posted the results to the database in a consistent manner. The completeness check resulted in identifying the following:

- any pollutant that was scheduled to be analyzed but was not analyzed.
- pollutants that were analyzed but were not scheduled to be analyzed.
- any pollutant for which the expected number of samples analyzed did not agree with the actual number of samples analyzed.

SCC evaluated and resolved discrepancies by subsequent QA/QC reviews. SCC documented all changes to data in the landfill analytical database in a status report entitled "Status of the Waste Treatment Industry: Landfills Database".

4.9.2.3 Trip Blanks and Equipment Blanks

SCC addressed qualifiers assigned to data as a result of trip blank and equipment blank contamination in the same way that it addressed contamination of lab method blanks, detailed below:

- Sample Results Less than Five Times Blank Results: When the sample result was less than five times the blank result, there were no means by which to ascertain whether the presence of the analyte could have attributed to blank contamination. Therefore, the result was included in the database as non-detect, with a nominal detection limit equal to the dilution-adjusted instrument detection limit.
- Sample Results Greater than Five Times but Less than Ten Times Blank Results: These data were of acceptable quality and were used to represent maximum values.

- Sample Results Greater than Ten Times Blank Results or Analyte not Detected in Sample: The presence of the analyte in the blank did not adversely affect the data in those cases where the sample results were greater than ten times the associated blank results or when the analyte was not detected in associated samples. Such data were acceptable without qualification.

4.9.2.4 Field Duplicates

EPA collected field duplicates during the EPA sampling episodes to help determine the accuracy and consistency of the sampling techniques employed in the field. In the analytical database, EPA represented field duplicate results by the letter "D" preceding the sample point number. The Agency combined duplicate samples that it considered acceptable on a daily basis using the following rules:

- If all duplicates were non-detect values, then the aggregate sample was labeled non-detect (ND), and the value of the aggregate sample was the maximum of the ND values.
- If the maximum detected value was greater than the maximum ND value, then the aggregate sample was labeled NC, and the value of the aggregate sample was the sum of the non-censored (NC) and ND values divided by the total number of duplicates for that independent sample.
- If the maximum NC value was less than or equal to the maximum ND value, then the aggregate sample was labeled ND and the value of the aggregate sample was the maximum of the ND values.
- If all duplicates were NC values, then the aggregate sample was labeled NC and the value of the aggregate sample was the average of the NC values.

In the laboratory, SCC calculated analytical precision by determining the relative percent difference of paired spiked samples. EPA considered data acceptable if the relative percent difference was within the laboratory criteria for analytical precision.

EPA considered duplicate relative percent difference values as acceptable if they were within the laboratory criteria for analytical precision plus or minus 10 percent.

4.9.2.5 Grab Samples

Most data presented in the analytical database represent composite sample results, but other types of results exist due to sampling requirements. Most grab sample results were represented by the letters "A", "B", or "C" following the sample point number in the analytical database for grabs collected on the same day. EPA collected grab samples of this nature only for oil and grease/hexane extractable material and EPA included these samples when calculating average concentrations of pollutants. The Agency averaged grab samples of any kind on a daily basis before using them in data analyses.

4.9.2.6 Non-Detect Data

EPA assigned non-detect data numeric values so that they could be used in the data analyses. In general, non-detect data can be set either at the method detection limit, at the instrument detection limit, at half of the method detection limit, or zero. Detection limits can be standardized (as in the method detection limit) or variable (as in the instrument detection limit or the sample detection limit, which may vary depending on dilution). The instrument detection limit is the lowest possible detection limit: the instrument cannot detect the contaminant below this level. In many cases, the method detection limit is significantly higher than the instrument detection limit.

For the Landfills effluent guideline, EPA defined all non-detect data collected from the EPA sampling episodes as follows: 1) the value used for non-detect data was represented by the detection limit reported in the analytical database, and 2) if the detection limit of the non-detect data was greater than the detected results, the average was calculated using all of the data, but the results were flagged for review on an individual basis. When flagged results were reviewed as a whole, the high detection limits were found to be on the same order of magnitude as the detect values; therefore, all flagged data were included in calculating averages.

4.9.2.7 Bi-Phasic Samples

In one sampling episode for a captive hazardous landfill at an industrial facility, some samples collected became bi-phasic. That is, EPA collected aqueous samples, but from the time that EPA collected the sample to the time EPA analyzed it, the sample formed a solid, organic phase. Therefore, the analyzed sample consisted of an aqueous portion and an organic portion. For these samples, EPA reported analytical results for each phase separately. The Agency calculated consolidated results for the bi-phasic samples by factoring the percent of each phase relative to the total sample volume with the results of each phase and adding the weighted results together. Pollutants were not always detected in both the aqueous and organic phases of a bi-phasic sample. In instances where EPA detected a pollutant in one phase and not in the other phase, the detection limit was set at zero, which removed the non-detect phase from the equation. When both phases were non-detect, EPA used the lowest of the two detection limits as the result.

4.9.2.8 Conversion of Weight/Weight Data

In some cases, EPA analyzed wastewater samples collected in the field as solids due to criteria specified in the analytical method. The Agency reported these results in the database in solids units of ug/kg or ng/kg. EPA converted these results to ug/L and ng/L, respectively, so that they could be used in data analysis.

The landfill analytical database contained a file called "solids" that contained percent solids values for those samples associated with a result that were reported on a weight/weight basis. This percent solids value was necessary to convert results from a weight/weight basis to a weight/volume basis.

The following formula was utilized to convert the "amount" from a weight/weight basis to a weight/volume basis. This formula assumed a density of 1:

$$\text{Amount (weight/weight)} \times [\text{Percent Solids}/100] = \text{Amount (weight/volume)}$$

where,

Amount = The result contained in the "amount" field in the "result" file.

Percent Solids = The percent solids result contained in the "percent" field in the "solids" file.

After conversion, the amount was expressed in weight/volume units as shown below:

Weight/Weight Units	Weight/Volume Units
pg/kg	pg/L
ng/kg	ng/L
ug/kg	ug/L
ug/g	ug/mL
mg/kg	mg/L

4.9.2.9 Average Concentration Data

EPA employed all data conventions discussed above when calculating the average concentration of a group of data. The Agency calculated average concentrations to develop raw waste loads, current discharge concentrations, and percent removal values. To calculate the average concentration of a pollutant at a particular sample point, the following hierarchy was used: 1) all non-detect data was set at the detection limit listed in the database, 2) all weight/weight units were converted to weight/volume units using the percent solids file, 3) all units were then converted to ug/L, 4) the bi-phasic sample results were combined into one consolidated result, 5) both duplicate pairs and grab samples were combined using the rules discussed above, and 6) the long-term average was calculated by adding all results and dividing by the number of results.

4.9.3 Detailed Questionnaire Database Review

EPA reviewed each Detailed Questionnaire for the following: 1) completeness, 2) internal consistency, and 3) outliers. Outliers refer to data values that are well outside those expected for this industry. For example,

EPA considered flow rates above 10 million gallons per day to be outliers. In cases such as this, the QA/QC reviewer would verify the accuracy and correctness of the data.

All information that EPA entered into a computer database was given a 100 percent QA/QC check to ensure that all data were inputted properly. This was accomplished by double key entry, and any discrepancies between the two inputted values compared with the original submission were corrected by the QA/QC reviewer.

Section 4.3.2 discusses additional handling procedures for Detailed Questionnaires.

4.9.4 Detailed Monitoring Questionnaire Data Review

EPA evaluated Detailed Monitoring Questionnaire data using the same procedures outlined for the Detailed Questionnaire process. The QA/QC steps included reviews for the following: 1) completeness, 2) internal consistency, and 3) outliers.

Section 4.4 discusses additional handling procedures for Detailed Monitoring Questionnaires.

Table 4-2: Types of Facilities Included in EPA's Characterization and Engineering Site Visits

Ownership Type	Characterization Site Visits	Engineering Site Visits*
Municipal	4	9
Commercial	9	8
Non-Commercial (captive, intra-company)	2	1
Waste Type	Characterization Samples Collected	
Subtitle D	13	15
Subtitle C	5	3
Landfill Type	Characterization Samples Collected	
Subtitle D Non-Hazardous	10	15
(Municipal)	(2)	(14)
(Non-Municipal)	(8)	(1)
Subtitle C Hazardous	5	3
Ground Water	3	0

*One engineering site visit was conducted outside the U.S.

Table 4-3: Types of Facilities Included in EPA's Field Sampling Program

Episode	Ownership Type			Waste Type		Landfill Subcategory		Treatment Technology
	Municipal	Commercial	Non-Commercial	Subtitle D	Subtitle C	Non-Hazardous	Hazardous	
4626	X			X		X		Equalization, chemical precipitation, biological treatment, filtration
4667	X			X		X		Equalization/stripper, chemical precipitation, biological treatment, GAC, filtration
4687	X			X		X		Equalization, filtration, reverse osmosis
4690			X		X		X	Air stripping Steam stripping
4721		X			X		X	Equalization, biological treatment
4759		X			X		X	Equalization, chemical precipitation, biological treatment

Table 4-4: Episode Numbers for the Engineering Site Visits and Field Sampling Efforts

Episode Number	Sampling/ Site Visits
4491	E, C
4503	C
4626	E, W
4630	C
4631	C
4638	C
4639	C
4644	C
4667	E, W
4683	C
4687	E, W
4738	C
4690	E, W, C
4721	E, W, C
4684	C
4685	C
4759/4682	E, W, C
4659	C
-	E
-	E
-	E
-	E
-	E
-	E
-	E
-	E
-	E
-	E
-	E

C = Raw Wastewater Characterization Sampling Episode (1-day sampling episode)
E = Engineering Site Visit
W = Five-day Sampling Episode

5.0 INDUSTRY SUBCATEGORIZATION

In developing technology-based regulations for the Landfills industry, EPA considered whether a single set of effluent limitations and standards should be established for the industry, or whether different limitations and standards were appropriate for subcategories within the industry. The Clean Water Act (CWA) requires EPA, in developing effluent limitations, to assess several factors, including manufacturing processes, products, the size and age of a site, wastewater use, and wastewater characteristics. The Landfills industry, however, is not typical of the industries regulated under the CWA. Therefore, EPA looked at additional factors that are specifically tailored to the characteristics of landfill operations in deciding appropriate limitations for landfill facilities. The factors considered for the subcategorization of the Landfills industry are listed below:

- Resource Conservation and Recovery Act (RCRA) Regulatory classification
- Types of wastes received
- Wastewater characteristics
- Facility size
- Ownership
- Geographic location
- Facility age
- Economic impacts
- Treatment technologies and costs
- Energy requirements
- Non-water quality impacts

5.1 Subcategorization Approach

Based on an evaluation of the above factors, EPA determined that there was a notable distinction between wastewater associated with Subtitle C landfills and those from Subtitle D landfills. A wider range of toxic

organic pollutants and higher concentration of metals was found at the Subtitle C landfills. Thus, the most significant differences observed in wastewater characteristics at landfills are directly correlated to the wastes received at the landfill which, in turn, are most obviously linked to the landfill's RCRA status. Therefore, EPA concluded that the most appropriate basis for subcategorization is by landfill classification under RCRA.

Additionally, the Agency believes that this subcategorization approach has the virtue of being easiest to implement because it follows the same classification previously established under RCRA and is currently in use (and widely understood) by permit writers and regulated entities. The Agency believes that any subcategorization at odds with existing RCRA classification approaches will potentially create unnecessary confusion to the regulated community. The subcategories are described below.

5.2 Landfills Subcategories

EPA is subcategorizing the Landfills industry into two subcategories as follows:

RCRA Subtitle C Hazardous Waste Landfill Subcategory

Subpart A of 40 CFR Part 445, "RCRA Subtitle C Hazardous Waste Landfill Subcategory," applies to wastewater discharges from a solid waste disposal facility subject to the criteria in 40 CFR Part 264 Subpart N - "Standards for Owners and Operators of Hazardous Waste Treatment, Storage, and Disposal Facilities" and 40 CFR Part 265 Subpart N -"Interim Standards for Owners and Operators of Hazardous Waste Treatment, Storage, and Disposal Facilities." Hazardous waste landfills are subject to requirements outlined in 40 CFR Parts 264 and 265 that include the requirement to maintain a leachate collection and removal systems during the active life and post-closure period of the landfill. For a discussion of these criteria, see Chapter 3, Section 3.1: "Regulatory History of the Landfills Industry", or see the Preamble to the proposed landfill guideline at 63 FR 6426, 6430-31. (February 6, 1998).

RCRA Subtitle D Non-Hazardous Waste Landfill Subcategory

Subpart B of 40 CFR Part 445, "RCRA Subtitle D Non-Hazardous Waste Landfill Subcategory," applies to wastewater discharges from all landfills classified as RCRA Subtitle D non-hazardous landfills subject to either of the criteria established in 40 CFR Parts 257 (Criteria for Classification of Solid Waste Disposal Facilities and Practices) or 258 (Criteria for Municipal Solid Waste Landfills). For a discussion of these criteria, see Chapter 3, Section 3.1: "Regulatory History of the Landfills Industry", or see the Preamble to the proposed landfill guideline at 63 FR 6426, 6431-32 (February 6, 1998).

Table 5-1 presents the subcategorization of all of the landfill facilities in the EPA database by questionnaire identification number. All landfill facilities included in this table completed a Detailed Questionnaire and collect wastewater; however, not all of the facilities included in this table are within the scope of the rule. Landfill facilities not covered by this rule include captive landfills, landfills that generate no in-scope wastewater, and zero or alternative discharge facilities. Chapter 2 discusses further the applicability of the rule.

5.3 Other Factors Considered for Basis of Subcategorization

EPA also evaluated the appropriateness and significance of developing subcategories based on the other factors presented earlier in this chapter. The following subsections present EPA's evaluation of each of these factors.

5.3.1 Types of Wastes Received

The type of solid waste that is deposited in a landfill often has a direct correlation to the characteristics of the leachate produced by that landfill. Wastes deposited in landfills range from municipal solid waste and non-hazardous materials to hazardous wastes containing contaminants such as pesticides. An analysis of the data collected as part of this study showed that there are differences in the wastewater generated by facilities that dispose of hazardous wastes as compared to non-hazardous wastes. These differences are reflected in both the number and types of pollutants of interest (as defined in Chapter 7) identified in each

subcategory and in the concentrations of these pollutants found in the wastewater generated. Tables presented in Chapters 6 (Tables 6-9 through 6-15) and 7 (Tables 7-1 and 7-2) of this document show these differences.

Specifically, the pollutants of interest list for the Non-Hazardous subcategory contains a total of 32 pollutants, whereas the pollutants of interest list for the Hazardous subcategory contains 63 pollutants (see Chapter 7 for discussion on pollutants of interest). In addition, there are more than twice as many pollutant-of-interest metals present in the hazardous landfill leachate (12) as in non-hazardous landfill leachate (5), and there are twice as many organic pollutants of interest present at hazardous landfills (28) than at non-hazardous landfills (14). Pollutants analyzed during EPA sampling episodes were detected approximately 47 percent of the time at hazardous facilities versus approximately 31 percent of the time at non-hazardous facilities. Tables 6-9 through 6-13 in Chapter 6 present the median, minimum, and maximum concentrations of the pollutants of interest for both subcategories and, although there are cases where the concentrations found at non-hazardous landfills are greater than the concentrations found at hazardous landfills, EPA detected higher concentrations of most pollutants of interest at hazardous landfills. In the proposed rule, EPA included data from numerous CERCLA facilities in the calculation of hazardous landfill raw wastewater pollutant characteristics. However, since these discharges consisted primarily of ground water and because the final rule does not cover ground water, EPA decided not to use the CERCLA data to characterize hazardous landfills. Table 5-2 presents the median concentrations of pollutants of interest common to both subcategories for hazardous and non-hazardous landfills.

In conclusion, EPA has determined that the most practical method of distinguishing the type of waste deposited in a landfill is achieved by utilizing the RCRA classification of landfills. As discussed in Section 5.1, the RCRA classification selected as the basis for subcategorization is based on the type of waste received by the landfill, hazardous or non-hazardous. Therefore, the type of waste disposed at a landfill is a factor that is taken into consideration because it is directly encompassed by the RCRA classification scheme -- the selected subcategorization method.

In addition to subcategorizing the Landfills industry based on RCRA classification, EPA also considered further subcategorizing the Subtitle D Non-Hazardous subcategory to account for differences between non-hazardous landfills and non-hazardous monofills. Subtitle D monofills, a class of non-hazardous landfills, accept only one type of waste that include, but are not limited to, construction and demolition debris, ash, and sludge. EPA decided not to further subcategorize Subtitle D landfill facilities. This decision is based on the following two considerations: (1) similarities in waste acceptance and leachate characteristics between monofills and other Subtitle D Non-Hazardous landfills; and (2) ease of implementation. First, EPA compared the number and type of pollutants present in Subtitle D municipal and non-municipal leachate. As shown in Table 6-9 in Chapter 6, there are nine pollutants of interest for Subtitle D non-municipal solid waste landfills whereas there are 32 pollutants of interest for Subtitle D municipal solid waste landfills. Although there were fewer pollutants of interest detected at non-municipal solid waste landfills, there were no pollutants of interest at non-municipal solid waste landfills that were not also present at municipal solid waste landfills. This is not unexpected, as the waste deposited in municipal solid waste landfills and dedicated monofills is not mutually exclusive. Although cells at a dedicated landfill may prohibit disposal of municipal refuse, a municipal solid waste landfill may also accept ash, sludge, and construction and demolition wastes. EPA also compared the median raw wastewater concentration data from Subtitle D municipal solid waste and non-municipal solid waste landfills in the EPA database in Table 6-9 and determined that the concentrations present at non-municipal solid waste landfills were equivalent to or less than the concentrations present at municipal solid waste landfills. EPA acknowledges that certain types of Subtitle D non-municipal solid waste landfills have a low organic content in their wastewater, and as a result some monofills, such as ash monofills, may not be able to operate biological treatment systems such as those selected for BPT/BAT for the Non-Hazardous subcategory. For those monofills that do not accept organic wastes, EPA found that many of the facilities could meet the subcategory limitations without treatment and, for those that could not, alternative technologies were available at costs no greater than those technologies EPA evaluated (and determined) to be economically achievable for the subcategory as a whole. EPA included the costs associated with these alternate technologies in the final cost impact analysis. See Chapter 11 for further discussion.

To further assess the differences between municipal solid waste and non-municipal solid waste landfills in the Non-Hazardous subcategory, EPA evaluated leachate characteristics from Subtitle D non-municipal solid waste landfills in published reports. Table 5-3 includes data from three reports[1] that analyzed construction and demolition monofills, ash monofills, and co-disposal sites and compares these data to the median raw wastewater data collected from non-hazardous municipal solid waste landfills as part of the Landfills industry study. The data contained in these reports indicate that the leachate characteristics at construction and demolition, co-disposal, and ash monofill facilities are comparable to the leachate characteristics from municipal solid waste landfills. Both the number and type of parameters in the leachate do not differ among these types of facilities, and concentration levels for all pollutants are comparable, with many parameters found at lower concentrations in the data from the construction and demolition, co-disposal, and ash monofill facilities. Therefore, EPA has concluded that untreated leachate characteristics at these facilities were not significantly different than at other non-hazardous landfill facilities and did not merit further subcategorization.

In addition, EPA collected data from six Subtitle D monofills during the EPA sampling program, including two sludge monofills, two ash monofills, and two construction and demolition monofills. Table 5-4 presents the average raw wastewater data for selected pollutants, along with the types of waste landfilled at each monofill. EPA evaluated its monofill data along with commenter submitted data and the data referenced in Table 5-3 and determined that there are differences in wastewater characteristics between different types of monofills. Most of these differences result from the fact that not all monofills accept the same types of waste. Some monofills accept primarily organic wastes (construction and demolition, sludge), others accept primarily inorganic wastes (ash, lime), and many monofills accept a combination of organic and inorganic wastes. As a result of the various types of monofills, EPA determined that a single subcategory for all monofills would still not address the situation where a certain class of constituents is regulated even

[1] "A Study of Leachate Generated from Construction and Demolition Landfills", Department of Environmental Engineering Sciences, University of Florida, August 1996; "Characterization of Municipal Waste Combustion Ashes and Leachates from Municipal Solid Waste Landfills, Monofills, and Co-Disposal Sites", U.S. EPA, EPA 530-SW-87-028D, October 1987; "Characterization of Municipal Waste Combustion Ash, Ash Extracts, and Leachates", U.S. EPA, EPA 530-SW-90-029A, March 1990.

though not all types of monofills contain those constituents (e.g. a utility ash monofill with low raw wastewater BOD_5 concentrations would still be in the same subcategory as a sludge monofill which may contain moderate levels of BOD_5). Thus, EPA would need to establish a separate subcategory for each type of monofill to address the differences among them. Therefore, rather than develop multiple monofill subcategories, EPA decided that, since the types of pollutants and concentrations of pollutants found at monofills are, for the most part, equivalent to or less than those found at municipal solid waste landfills, a single subcategory is appropriate for Subtitle D landfills. EPA concluded that the pollutants regulated for the Subtitle D Non-Hazardous subcategory will control the discharges from all types of Subtitle D landfills, including monofills.

The second consideration was based on ease of implementation. As discussed in Section 5.2, the RCRA classification scheme selected as the basis for subcategorization clearly defines non-hazardous, hazardous, and municipal solid waste landfill facilities. However, RCRA does not make any further distinction nor further divide the Subtitle D landfill facilities based on whether they are monofills or if they receive multiple types of waste. Therefore, by further subcategorizing the Subtitle D facilities into monofills and multiple waste landfills, a new classification scheme would be introduced to permit writers and regulated facilities. EPA concluded that the current RCRA classification scheme is widely understood by permit writers and regulated landfill facilities, making it the easiest of the subcategorization approaches to implement. Additionally, there are many facilities that operate both dedicated cells (similar to monofills) and municipal solid waste cells at the same landfill and commingle the wastewater prior to treatment. Establishing one subcategory for all non-hazardous landfills will ease implementation issues and adequately control discharges from the Landfills industry.

5.3.2 Wastewater Characteristics

EPA concluded that leachate characteristics from non-hazardous and hazardous landfills differed significantly from each other in the types of pollutants detected and the concentrations of those pollutants. The tables supporting this conclusion are presented in Chapter 6 (Tables 6-9 through 6-13) and Chapter

7 (Tables 7-1 and 7-2) of this document. As expected, EPA found that the leachate from hazardous landfills contained a greater number of contaminants at higher concentrations than leachate from non-hazardous landfills, as discussed in Section 5.3.1. This conclusion supports subcategorization based on RCRA classification of hazardous and non-hazardous landfills.

In EPA's evaluation of contaminated ground water, the wastewater characteristics of contaminated ground water from hazardous landfills differed significantly from the contaminated ground water characteristics at non-hazardous waste landfills, as shown in Table 5-5 and Table 5-6, respectively. Contaminated ground water from non-hazardous landfills contained only 16 pollutants of interest (as defined in Chapter 7) compared to the contaminated ground water from hazardous waste landfills which contained a total of 54 pollutants of interest. In addition, effluent data collected in support of this rule demonstrate that contaminated ground water flows at hazardous and non-hazardous facilities are, in general, currently adequately treated as a result of existing corrective action programs under RCRA.

Due to the site-to-site variability of contaminated ground water, EPA has decided that the treatment of these flows is best addressed through the RCRA Corrective Actions program. RCRA Corrective Action programs at the federal, state, and local level have the ability to consider the site-to-site variability of the contaminated ground water and provide the most applicable treatment necessary to control the contaminants. Therefore, EPA has decided to exclude contaminated ground water from this regulation. Chapter 2 fully describes EPA's decision not to include contaminated ground water as a landfill wastewater covered by this regulation.

5.3.3 Facility Size

EPA considered subcategorization of the Landfills industry on the basis of facility size and found that landfills of varying sizes generate similar wastewater and use similar treatment technologies. Based upon a review of the industry-provided data in the landfills' database, there was no observed correlation between waste acceptance amount or wastewater flow rate and the selection of treatment technologies. For

example, a landfill facility can add cells or increase its waste receipt rate depending on the local market need without altering or changing the characteristics of the wastewater generated. In addition, the size of a landfill was not determined to be a factor in cost-effectiveness of the regulatory options considered by EPA. Finally, EPA has determined that wastewater from landfills can be treated to the same level regardless of facility size. EPA did not promulgate a de-minimis flow exemption for this guideline; however, EPA has accounted for landfill facilities that generate small volumes of wastewater by estimating compliance costs for the BPT/BAT options based on treating their wastewater off-site at a CWT facility (see Section 9.2.5).

5.3.4 Ownership

EPA considered subcategorizing the industry by ownership. A significant number of landfills are owned by state, local, or federal governments, while others are commercially or privately owned. Landfills generally fall into the following two major categories of ownership: municipal or private. Landfills owned by municipalities are primarily designed to receive non-hazardous solid waste such as municipal solid waste, non-hazardous industrial waste, construction and demolition debris, ash, and sludge. However, municipally-owned landfills may also be designed to accept hazardous wastes.

Privately-owned landfills can also provide for the disposal of non-hazardous solid waste such as those mentioned above, and, like municipally-owned facilities, may also be designed to accept hazardous wastes. EPA found that current commercially- and municipally-owned landfills generally accept and manage wastes strictly by the RCRA classification and, although there are distinct economic differences, there is no distinction in the wastewater characteristics and wastewater treatment employed at commercially- or municipally-owned landfills. Since all landfill types could be of either ownership status, EPA determined that subcategorization based upon municipal and private ownership was not appropriate.

5.3.5 Geographic Location

EPA considered subcategorizing the industry by geographic location. Landfill sites are not limited to any

one region of the United States. A table presenting the number of landfills by state is presented in Chapter 3 (Table 3-1). While EPA included landfills from all sections of the country in the Agency's survey efforts, collection of wastewater characterization data as part of EPA's sampling episodes was limited to landfill facilities in the Northeast, South, and Midwest, where annual precipitation is either average or above average. Although wastewater generation rates appear to vary with annual precipitation, which is indirectly related to geographic location, EPA could not establish a direct correlation between leachate characteristics and geographic location due to lack of sampling data from arid parts of the United States. However, the Agency believes that seasonal variations in rainfall cause only minor fluctuations in leachate characteristics due to dilution effects and volume of leachate generated. In addition, many landfill facilities have developed site-specific best management practices to control the amount of rainwater that enters a landfill and eventually becomes part of the leachate. These practices include proper contouring of landfill cells, extensive use of daily cover, and capping of inactive landfill cells to minimize the amount of rainwater that enters the landfill. EPA's data collection efforts indicate that landfill facilities in less arid climates are more likely to use these management practices to control their wastewater generation and flows to the on-site wastewater treatment plant. The data collected by EPA did not indicate any significant variations in wastewater treatment technologies employed by facilities in colder climates versus warmer climates.

EPA notes that geographic location may have a differential impact on the costs of operating a landfill. For example, the cost of additional equipment required for the operation of the landfill or treatment system or tipping fees charged for the hauling of waste may differ from region to region. These issues were addressed in the economic impact assessment of the final rule.

Therefore, since the effect of geographic location appears to have a minimal impact on wastewater characteristics or can be easily addressed at minimal effort and cost, EPA determined that subcategorization based upon geographic location was not appropriate.

5.3.6 Facility Age

EPA considered subcategorization based on the age-related changes in leachate concentrations of pollutants for different age classes of landfills based on the evaluation of several factors. Several considerations lead to the conclusion that age-related limits are not appropriate. First, a facility's wastewater treatment system typically receives and commingles leachate from several landfills or cells of different ages. The Agency has not observed any facility which has found it advantageous or necessary to treat age-related leachates separately. The Agency did, however, sample two landfill facilities that had only one cell. One of the facilities had been receiving wastes for nine years in its landfill cell, while the other facility had only been receiving waste for one year. EPA compared the raw wastewater concentrations of the constituents in these two cells and found the concentrations to be very similar. In addition, most of the constituents in both cells were close to the median raw wastewater concentration for the Non-hazardous subcategory. Second, based on responses to the questionnaire, discussions with landfill operators and historical data, EPA understands that leachate pollutant concentrations appear to change substantially over the first two to five years of operation but then change only slowly thereafter.

These two observations imply that treatment systems must be designed to accommodate the full range of concentrations expected in influent wastewater. EPA concluded that the BPT/BAT/NSPS treatment technologies are able to treat the variations in landfill wastewater likely to occur due to age-related changes. EPA has taken into account the ability of treatment systems to accommodate age-related changes in leachate concentrations, as well as short-term fluctuations by promulgating effluent limitations which reflect the variability observed in monitoring data spanning up to three years.

Additionally, EPA addressed age-related effects on treatment technologies, costs, and pollutant loads by utilizing data collected from a variety of landfills in various stages of age and operation (e.g. closed, inactive, active). EPA sampled landfills of various ages and stages of operation (active, inactive, closed), lined and unlined, and concluded that the landfill database used to develop the effluent limitations represents leachate typically found at Subtitle D landfills. In addition, EPA received comments from several commenters stating

that the leachate characterization data presented in the proposal was consistent with their own monitoring data.

However, several commenters on the proposed rule stated that EPA's sampling data did not represent adequately the age-related differences that can exist between leachates from landfills of different ages. Table 5-7 presents the age of the landfills sampled by EPA. The table includes the sampling episode number and RCRA classification of each landfill, the number of cells in each landfill, whether the landfill is lined or not, the year the landfill began accepting waste, the year it stopped accepting waste, and the projected landfill closure date, if available. All information on landfill ages were obtained from the Detailed Questionnaire or the sampling reports from these facility's sampling episodes. All of EPA's sampling episodes occurred during a two year period from 1993 to 1995. Grouping facilities shown in Table 5-7 according to the year the facility began accepting waste and by regulatory history, there are ten pre-1980 landfills (before 1980 Section 3001 of RCRA); five landfills that fall in the 1980 to 1983 range (before the 1984 Hazardous and Solid Waste Amendment to RCRA); five landfills that fall in the 1984 to 1988 range (before Land Disposal Restrictions (LDR)); and five landfills that are post-1988 (after LDR). The landfill facilities sampled by EPA were between one and 43 years of age at the time of sampling. As seen in Table 5-7, the majority of landfill facilities sampled contained more than one cell, and often more than one landfill, and many of these landfill facilities commingled the leachate discharges from cells and landfills of various ages. As mentioned above, the Agency sampled raw wastewater at two landfill cells of different ages and found the concentrations of constituents to be very similar. EPA did not identify any facility that treated leachates separately due to differences in age.

To determine if significant differences existed between landfills of various ages, EPA compared pollutant concentration data from Subtitle D landfill facilities of different ages in the EPA database. Table 5-8 presents the median raw wastewater concentration for selected conventional, nonconventional, organic and metal pollutants for non-hazardous landfills with available raw wastewater data in the EPA database by age group. EPA determined the raw wastewater median concentrations in the table by: 1) calculating the

average concentration of a pollutant at a landfill facility using data from EPA sampling episodes, Detailed Questionnaires, and Detailed Monitoring Questionnaires, and then 2) calculating the median concentration of the landfill facility average concentrations.

As seen in Table 5-8, when landfills of various ages from EPA's landfill effluent guidelines database are compared, it is difficult to pinpoint any particular trend (i.e. organic pollutant concentrations decrease significantly with age). The absence of any particular trend associated with pollutant concentrations across landfill facilities of various ages may be due to the fact that most of the older landfill facilities in EPA's database have newer landfill cells whose leachate is commingled for treatment with the leachate from the older landfill cells. EPA acknowledges that age-related changes in landfill leachate characteristics would be expected from individual landfill cells. Most of the older landfill cells have lower concentrations of BOD_5, COD, and most organic pollutants indicating a smaller amount of degradable compounds from the aged waste (reference 13). In addition, aged leachates contain high levels of chemically reduced compounds, such as ammonia, and high chlorides because of the anaerobic environment of the landfill. These trends tend to be true for individual landfill cells. Again, however, as mentioned above, the Agency sampled raw wastewater at two landfill cells of different ages and found the concentrations of constituents to be very similar. However, when looking at a landfill facility as a whole (where leachates from several cells of various ages are commingled for treatment), the landfills effluent guidelines database does not fully support such a trend. Furthermore, the time frame of these age-related changes is not consistent in every landfill. Several factors including size of a cell, composition and disposition of refuse, precipitation levels, and the influence of leachate from older cells on newer cells, can, and do, affect how a leachate's composition changes with time. However, in general, these pollutant concentrations are within the same order of magnitude and the Agency concluded that this age-related variability in wastewater characteristics can be adequately controlled by the BPT/BAT treatment options, as demonstrated by the BAT facilities sampled by EPA.

Based on this analysis of the effects of age on wastewater characteristics, EPA determined that subcategorization based on facility age is not appropriate.

5.3.7 Economic Characteristics

EPA also considered subcategorizing the industry based on the economic characteristics of the landfill facilities. If a group of facilities with common economic characteristics, such as revenue size, was in a much better or worse financial condition than others, EPA could consider subcategorization on economics. However, based on the results of the Detailed Questionnaires, financial conditions of facilities showed no significant pattern of variation across possible subcategories, such as municipally- and commercially-owned facilities. In addition, EPA determined that the economic impacts of the compliance costs associated with the BPT/BAT regulations did not inordinately effect any particular segment of the Landfills industry. Therefore, EPA determined that subcategorization based on the economic characteristics of landfills facilities was not justified.

5.3.8 Treatment Technologies and Costs

Wastewater treatment for this industry ranges from primary systems such as equalization, screening, and settling, to advanced tertiary treatment systems such as filtration, carbon adsorption, and membrane separation. EPA found that the selected treatment technology employed at a facility was dependent on wastewater characteristics and permit requirements. Landfills with more complex mixtures of toxic pollutants in their wastewater generally had more extensive treatment systems and may utilize several treatment processes (e.g., facilities with high levels of both organic and inorganic pollutants may employ both a chemical and biological treatment system). However, subcategorizing by the waste type received by a landfill as outlined in the RCRA classification of landfills is less difficult to implement and results in addressing the same factors as using treatment processes employed. As a result, EPA did not consider treatment technologies or costs to be a basis for subcategorization.

5.3.9 Energy Requirements

The Agency did not subcategorize based on energy requirements because energy usage was not considered a significant factor in this industry and is not related to wastewater characteristics. Energy costs resulting from this regulation were accounted for in the cost section of this development document (Chapter 9) and in the economic impact assessment.

5.3.10 Non-Water Quality Impacts

The Agency evaluated the impacts of this regulation on the potential for increased generation of solid waste and air pollution. The non-water quality impacts did not constitute a basis for subcategorization. Non-water quality impacts and costs of solid waste disposal are included in the economic analysis and regulatory impact analysis for this regulation. See Chapter 10 for more information regarding non-water quality impacts.

Table 5-1: Subcategorization of the EPA Landfills Database

Hazardous Subcategory Detailed Questionnaire ID Numbers	Non-Hazardous Subcategory Detailed Questionnaire ID Numbers	
16005	16001	16128
16007	16003	16129
16017	16008	16130
16018	16009	16131
16019	16011	16132
16031	16012	16135
16032	16013	16137
16034	16014	16139
16036	16015	16148
16037	16016	16150
16040	16020	16151
16041	16023	16152
16042	16024	16153
16044	16025	16154
16045	16026	16155
16051	16027	16156
16066	16028	16158
16067	16029	16159
16068	16033	16160
16069	16035	16161
16081	16038	16162
16086	16039	16163
16087	16043	16164
16094	16046	16165
16095	16047	16166

Table 5-1: Subcategorization of the EPA Landfills Database (continued)

Hazardous Subcategory Detailed Questionnaire ID Numbers	Non-Hazardous Subcategory Detailed Questionnaire ID Numbers	
16101	16048	16170
16104	16049	16171
16105	16050	16173
16106	16052	16174
16108	16053	16175
16110	16054	16176
16134	16055	16177
16136	16056	16180
16140	16057	16184
16141	16058	16185
16142	16059	16186
16143	16060	16187
16144	16061	16189
16145	16062	16190
16146	16063	16191
16147	16064	16193
16149	16065	16196
16167	16070	16199
16168	16071	16200
16169	16072	16201
16178	16073	16202
16179	16074	16203
16182	16075	16204
16183	16076	16205
16192	16077	16206

Table 5-1: Subcategorization of the EPA Landfills Database (continued)

Hazardous Subcategory Detailed Questionnaire ID Numbers	Non-Hazardous Subcategory Detailed Questionnaire ID Numbers	
16197	16078	16208
16210	16079	16211
16218	16083	16212
16235	16084	16215
16238	16085	16217
	16088	16219
	16090	16220
	16091	16221
	16092	16222
	16093	16223
	16097	16224
	16098	16225
	16099	16228
	16102	16230
	16103	16231
	16107	16232
	16109	16233
	16111	16234
	16113	16236
	16114	16239
	16115	16240
	16116	16241
	16117	16242
	16118	16243
	16119	16245

Table 5-1: Subcategorization of the EPA Landfills Database (continued)

Hazardous Subcategory Detailed Questionnaire ID Numbers	Non-Hazardous Subcategory Detailed Questionnaire ID Numbers	
	16120	16246
	16121	16248
	16122	16249
	16123	16250
	16124	16251
	16125	16252
	16127	16253

Table 5-2: Raw Wastewater Median Concentrations of Pollutants of Interest Common to Both the Hazardous and Non-Hazardous Landfill Subcategories

Non-Hazardous Pollutants of Interest	Hazardous Median Concentration	Non-Hazardous Median Concentration*
(mg/L)		
Ammonia	268	75 - 82
BOD	621	67 - 240
COD	1,309	994 - 1,100
Nitrate/Nitrite	1.6	0.65 - 0.95
TDS	15,958	2,894 - 4,850
TOC	441	236 - 377
Total Phenols	25	0.25 - 0.57
TSS	151	21 - 137
(ug/L)		
1,4-Dioxane	466	11
2-Butanone	1,048	1,082
2-Propanone	2,889	992
4-Methyl-2-Pentanone	500	101
Alpha Terpineol	96	123
Benzoic Acid	2,482	100
Hexanoic Acid	2,703	5,818
Methylene Chloride	118	37
O-Cresol	79	15
Phenol	4,400	102
P-Cresol	144	75
Toluene	104	108
Tripropyleneglycol Methyl Ether	853	197
Chromium	36	28
Strontium	3,044	1,671 - 4,615

Table 5-2: Raw Wastewater Median Concentrations of Pollutants of Interest Common to Both the Hazardous and Non-Hazardous Landfill Subcategories (continued)

Non-Hazardous Pollutants of Interest	Hazardous Median Concentration	Non-Hazardous Median Concentration*
Titanium	33	64
Zinc	100	100

* Non-Hazardous subcategory median concentrations are presented as a range because raw wastewater data was calculated separately for municipal solid waste and non-municipal solid waste facilities.

Table 5-3: Comparison of Subtitle D Non-Municipal and Municipal Raw Wastewater Pollutant Concentrations (ug/L)

Pollutant	C & D Study		EPA Characterization Studies - Data Range			Subtitle D Municipal Raw Wastewater Median Concentration		
			1990	1987				
Metals	Mean [1]	Facilities Det/Total[2]	Ash Monofills	Co-Disposal	Monofills	Median	Mean	Max
Arsenic	12.3	12/16	ND(50) - 400	8 - 46	10 - 218	32.4	50.4	179
Barium	661	13/13	ND(2) - 9,220	270 - 890	NA	483	720	3,500
Boron	NA	NP	NA	NA	NA	3,910	3,874	16,250
Chromium	NA	NP	ND(7) - 32	ND(10) - 13	5 - 914	28	46	240
Hexavalent Chromium	NA	NP	NA	NA	NA	30	77	247
Molybdenum	NA	NP	NA	NA	NA	10	27	69
Silicon	NA	NP	470 - 15,300	NA	NA	15,759	28,817	159,000
Strontium	NA	NP	NA	NA	NA	1,671	1,569	2,146
Titanium	NA	NP	NA	NA	NA	64	66	157
Zinc	658	15/15	5.2 - 370	9 - 1,210	48 - 3,300	100	1,476	31,813
Organics								
1,4-Dioxane	49	1/5	NA	NA	NA	11	118	323
2-Butanone	NA	NP	NA	NA	ND(50)	1,082	5,119	36,544
2-Propanone	NA	NP	NA	NA	ND(50)	991	2,407	8,614
4-Methyl-2-Pentanone	130	2/8	NA	NA	ND(50)	101	3,789	46,161
Alpha Terpineol	NA	NP	NA	NA	ND(50)	123	334	1,061
Benzoic Acid	15,457	4/9	ND(50) - 73	NA	ND(50)	100	7,220	33,335
Dichloroprop	NA	NP	NA	NA	ND(50)	6	10	29
Disulfoton	3.3	2/4	NA	NA	NA	6	9	20
Hexanoic acid	NA	NP	NA	NA	ND(50)	5,818	13,148	37,256
MCPA	NA	NP	NA	NA	NA	403	816	4,370
MCPP	NA	NP	NA	NA	NA	233	432	1,900
Methylene Chloride	26.4	4/9	NA	NA	ND(50)	37	70	237
N,N-Dimethylformamide	NA	NP	NA	NA	ND(50)	10	214	1,008
O-Cresol	50	2/8	NA	NA	ND(50)	15	298	2,215
Phenol	384	3/6	ND(10) - 32	ND(50) - 2,100	ND(1.5)	102	287	1,425
P-Cresol	NA	NP	NA	NA	ND(50)	75	246	998
Toluene	61	7/9	NA	ND(50) - 120	ND(50)	108	166	598
Tripropyleneglycol Methyl Ether	NA	NP	NA	NA	ND(50)	197	568	1,235

Table 5-3: Comparison of Subtitle D Non-Municipal and Municipal Raw Wastewater Pollutant Concentrations (ug/L) (continued)

Pollutant	C & D Study		EPA Characterization Studies - Data Range			Subtitle D Municipal Raw Wastewater Median Concentration		
			1990		1987			
Conventional/Nonconventionals	Mean [1]	Facilities Det/Total[2]	Ash Monofills	Co-Disposal	Monofills	Median	Mean	Max
BOD	87,320	14/14	NA	NA	NA	240,000	1,228,534	7,609,318
COD	754,500	16/17	NA	1,300,000 -	5 - 1,200,000	994,000	2,024,932	11,881,700
Ammonia Nitrogen	20,420	16/78	4,380 - 77,400	3,900,000	1,200 - 36,000	81,717	238,163	2,900,000
TDS	2,263,100	17/18	924,000 - 41,000,000	160,000 - 410,000	NA	2,894,289	4,195,518	17,533,000
TSS	1,859,100	17/18	NA	NA	NA	137,000	735,308	14,470,000
Total Phenols	620	7/7	NA	1,930,000 -	NA	571	142,838	2,051,249
Nitrate/Nitrite	NA	NP	NA	7,970,000	NA	651	5,844	50,800
TOC	306,540	7/7	17 - 420,000	NA	59,100 - 636,000	376,521	661,477	3,446,084
				NA				
				438,000 - 1,310,000				
Dioxins/Furans								
1234678-HpCDD	NA	NP	ND(NV) - 0.222[2]	0.12 - 0.77[2]	0.009 - 172[2]	0.00014	0.0024	0.0071
OCDD	NA	NP	ND(NV) - 0.107	0.21 - 15	0.06 - 120	0.0018	0.030	0.0824

All units in ug/l unless otherwise noted
*: The number of sites that detected the parameter/the total number of sites that sampled the parameter
(1): Mean includes non-detects for metals and conventionals/nonconventionals and does not include non-detects for organics and dioxins/furans
(2): Total homolog concentration
NA: Not Analyzed
ND: Not Detected
NV: Not Available
NP: Not Applicable

Table 5-4: Summary of EPA Sampling Data for Subtitle D Monofills Average Raw Leachate Data for Selected Pollutants

Episode	Waste Type Landfilled	BOD_5	TSS	Ammonia	Zinc	Alpha Terpineol	Benzoic Acid	P-Cresol	Phenol
		(mg/L)							
4503	mill sludge (clay, lime, cellulose), fly ash, bark	120	104	53.2	0.028	ND	ND	ND	ND
4630	POTW sludge	85	292	118	0.086	ND	ND	ND	ND
4631	municipal resource recovery ash	12	11	75	0.033	ND	ND	ND	0.092
4638	C&D debris, state-regulated non-hazardous waste	67	22	0.67	0.102	ND	ND	ND	ND
4639	municipal resource recovery ash, WWTP residues	4	4	0.1	0.06	ND	ND	ND	ND
4644	C&D, yard waste, bricks, rubble, waste oil	13	4	0.85	0	ND	ND	ND	ND

ND: Non-Detect
NA: Data not provided.

Table 5-5: Average Contaminated Ground Water Pollutant Concentrations at Hazardous Landfills in the EPA Database (ug/L)

Hazardous Groundwater Pollutant of Interest	Cas #	MDL	QID 16018	QID 16031	QID 16032	QID 16034	QID 16036	QID 16094	QID 16095		QID 16136	QID 16141	QID 16144	
			Inf	Eff	Eff	Inf	Inf	Eff	Inf	Eff	Inf	Eff	Inf	Eff
1,1-Dichloroethane	75243	10		2		230	113		89	5 ND			121522	10
1,1,1-Trichloroethane	71556	10		5		180	185	1 ND	370	5 ND			37598	10
1,1,2,2-Tetrachloroethane	79345	10					0.5 ND	1 ND					218139	445
1,2,4-Trichlorobenzene	120821	10											265	19 ND
1,2-Dichlorobenzene	95501	10						1 ND					10491	19 ND
1,2-Dichloroethane	107062	10		2			4	1 ND					1376889	357
1,2,3-Trichloropropane	96184	10											1300084	138
1,3-Dichlorobenzene	541731	10											16628	19 ND
1,4-Dichlorobenzene	106467	10						1 ND					25655	19 ND
1,4-Dioxane	123911	10				46							6429	3738
2,4-Dichlorophenol	120832	10											101	109 ND
2378-TCDD	1746016	0.00001											0.00016	
2378-TCDF	51207319	0.00001											0.0066	
2,4,5-T	93765	0.2											5	
2,4,5-TP	93721	0.2											2	
2-Propanone	67641	50											25424	446
Ammonia as Nitrogen	7664417	10											27444	17760
Arsenic	7440382	10										50 ND	80	13
Benzene	71432	10	520	1							4606		37922	10
Benzoic Acid	65850	50											1330	1920
Benzyl Alcohol	100516	10											298	282
Bis(2-chloroethyl)ether	111444	10											16518	34716
Bis(2-ethylhexyl)phthalate	117817	10											1039	19 ND
BOD	C-002	2000			2700								86500	55230
Boron	7440428	100											846	770
Cadmium	7440439	5										3 ND	9	8
Chlorobenzene	108907	10	920	2				1 ND					12936	10
Chloroform	67663	10		2									132025	32
COD	C-004	5000			23600								6423889	2445850
Copper	7440508	25											53	521
Dalapon	75990	0.2											109	
Dicamba	1918009	0.2											34	
Dichlorvos	62737	5											236	

Table 5-5: Average Contaminated Ground Water Pollutant Concentrations at Hazardous Landfill in the EPA Database (ug/L) (continued)

Hazardous Groundwater Pollutant of Interest	Cas #	MDL	QID 16018	QID 16031	QID 16032	QID 16034	QID 16036	QID 16094	QID 16095		QID 16136	QID 16141	QID 16144	
			Inf	Eff	Eff	Inf	Inf	Eff	Inf	Eff	Inf	Eff	Inf	Eff
Dinoseb	88857	0.5											14	
Dioxathion	78342	5											270	
Ethyl Benzene	100414	10	372	2									14694	10
Hexane Extractable Material	C-036	5000											1700222	8750
Hexanoic Acid	142621	10											16368	28013
Lithium	7439932	100											305	219
Methylene Chloride	75092	10		2									123572	40
Molybdenum	7439987	10											13	13
Naphthalene	91203	10				54					4100		3766	19 ND
Nickel	7440020	40										10	136	1462
Nitrate/Nitrite	C-005	50										1000 ND	2136	1571
Pentachlorobenzene	608935	20											4333	38 ND
Phenol	108952	10											6029	1537
Silicon	7440213	100											6738	6602
Strontium	7440246	100											17156	12360
TOC	C-012	1000											2055028	730700
Toluene	108883	10	573	2					19	5 ND	2573		22080	10
Trans-1,2-Dichloroethene	156605	10		5				1 ND					84660	14
Trichloroethene	79016	10		5			5						272606	33
TSS	C-009	4000										37000	121639	26450
Zinc	7440666	20				120							576	3451

MDL: Method detection limit

QID: Questionnaire ID

E: Sampling episode

ND: Non-detect with respect to instrument detection limit (IDL)

*: IDL is greater than detected value

Table 5-6: Average Contaminated Ground Water Pollutant Concentrations at Non-Hazardous Landfills in the EPA Database (ug/L)

Non-Hazardous Groundwater			E4683	QID 16016	QID 16085		QID 16088		QID 16129		QID 16132	QID 16163
Pollutant of Interest	Cas #	MDL	Inf	Eff	Inf	Eff	Inf	Eff	Inf	Eff	Eff	Eff
1,1-Dichloroethane	75243	10	10 ND	0.3 ND	5.5		8.6	1	22		0.35 ND	4 ND
1,1,1-Trichloroethane	71556	10	10 ND	0.5 ND	1.4 ND		2.1	1 ND	17		0.45 ND	5 ND
1,2-Dichloroethane	107062	10	10 ND	0.3 ND	1.4 ND		2.1 ND	1 ND	15		0.35 ND	8
2,4,5-T	93765	0.2	0.2 ND		0.2 ND		2		1 ND			
2,4,5-TP	93721	0.2	0.2 ND	2000 ND	0.2 ND		5		1.9 ND			
2-Propanone	67641	50	50 ND	10.5	50 ND		50 ND		742		1.3	
Ammonia as Nitrogen	7664417	10	1340		1284	256	1300	409	80551	563		
Arsenic	7440382	10	2 ND	16	4.3		3	2	13		11	25
Benzene	71432	10	10 ND	0.3 ND	1.4 ND	5.7	2.2	1 ND	13		0.35 ND	
Benzyl Alcohol	100516	10	10 ND		10 ND		10 ND		19			
BOD	C-002	2000	14000	1000	1000 ND	751	1000 ND		213655	1835		
Boron	7440428	100	173		362		97		1091			
Cadmium	7440439	5	4 ND	18	0.4	19	4 ND	15	5 ND		3.8	2
Chlorobenzene	108907	10	10 ND	0.5 ND	1.4 ND		2.1 ND	1.5 ND	12		0.35 ND	5 ND
Chloroform	67663	10	10 ND	0.5 ND	1.7		2.1 ND	1 ND	15			
COD	C-004	5000	28000	21637	51000		14000					33300
Copper	7440508	25	12	38	10 ND		10 ND		53	121	10	2.5
Dalapon	75990	0.2	0.2 ND		0.2 ND		6					
Dicamba	1918009	0.2	0.2 ND		0.2 ND		10					
Dinoseb	88857	0.5	0.5 ND		0.5 ND		3		50 ND			
Ethyl Benzene	100414	10	10 ND	0.3 ND	1.4 ND		2.1 ND	1 ND	15		0.35 ND	5 ND
Methylene Chloride	75092	10	10 ND	1	3.3 ND		2.1 ND	3.5 ND	49	0.6	0.45	
Naphthalene	91203	10	10 ND	36 ND	10 ND		10 ND		12			
Nickel	7440020	40	14 ND	30	59		14 ND	27	45	21	16	40
Nitrate/Nitrite	C-005	50	2660		1300		1340					10000 ND
Phenol	108952	10	10 ND	54.5 ND	5718 ND		10 ND		145			
Silicon	7440213	100	3530		3880		3270					
Strontium	7440246	100	201		657		200					
TOC	C-012	1000	10000 ND		40000		10000 ND				3996	
Toluene	108883	10	10 ND	0.3 ND	1.4 ND		2.1 ND	1 ND	47		0.35 ND	5 ND
Trans-1,2-Dichloroethene	156605	10	10 ND	0.3 ND	2.8	5.7	3.6	1 ND	38	0.5 ND	0.35 ND	5 ND
Trichloroethene	79016	10	10 ND	0.35	10 ND		2.1 ND	1 ND	19	0.5	0.45	1
TSS	C-009	4000	4000 ND		24000	5593	4000 ND		43848	2651		
Zinc	7440666	20	15.2	35	70		16		82	24		

MDL: Method detection limit
QID: Questionnaire ID
E: Sampling episode
ND: Non-detect with respect to instrument detection limit (IDL)
*: IDL is greater than detected value

Table 5-7: Age of Landfills in EPA Sampling Database

Episode	RCRA Classification	Number of Cells	Year Landfill Began Accepting Waste	Year Landfill Stopped Accepting Waste	Projected Closure
4491	Subtitle D Lined (varies)	25	1970	1994	1999
4503	Subtitle D Unlined	1	1974	1990	1992-3
4626	Subtitle D Lined (comp)	1	1986	1993	2000
4630	Subtitle D Lined (clay)	5	1988	1994	2003
4631	Subtitle D Lined (comp)	5	1987	-	-
	Subtitle C Lined (clay)	-	1972	1982	1991
	Subtitle C Lined (clay)	10	1972	1982	1991
4638	Subtitle D Lined (dbl comp)	5	1990	-	-
4639	Subtitle D Lined (comp)	2	1988	-	-
4644	Subtitle D Lined (clay)	2	1989	-	-
4659	Subtitle C Unlined	-	1958	1981	1981
	Subtitle C Lined (clay)	-	1981	1988	-
4667	Subtitle D Lined (varies)	4	1974	1993	1997
	Subtitle D Unlined	1	1962	1974	1991
4683	Subtitle D - GW Lined (varies)	7	1981	-	2017

Table 5-7: Age of Landfills in EPA Sampling Database (continued)

Episode	RCRA Classification	Number of Cells	Year Landfill Began Accepting Waste	Year Landfill Stopped Accepting Waste	Projected Closure
4687	Subtitle D Lined (comp)	1	1994	-	-
4690	Subtitle C Unlined	9	1952	1973	1976
	Subtitle C Lined (comp)	2	1980	1993	2008
	Subtitle C Unlined	8	1968	1979	1980
	Subtitle D Unlined	1	1992	1993	1998
	Subtitle C Lined (clay)	1	1982	1985	1986
	Subtitle D Unlined	2	1991	1993	1998
4738	Subtitle D Lined (clay)	4	1984	1994	1998
4721	Subtitle C Lined (clay)	2	1980	1993	1997
4759	Subtitle C Lined (varies)	39	1975	1993	2000

(comp): composite liner (synthetic and clay)
(varies): cells lined with either synthetic, asphalt, clay, composite or double lined composite

Table 5-8: Median Raw Wastewater Characteristics at Non-Hazardous Landfills of Varying Age

Pollutant	Landfill Age Group (Year in which Landfill Facility Began Accepting Waste)		
	Pre-1980 Median Conc.	1980-1990 Median Conc.	1991-Present Median Conc.
Ammonia	140 mg/L (15)	95 mg/L (10)	48 mg/L (3)
BOD_5	210 mg/L (18)	125 mg/L (13)	344 mg/L (4)
COD	596 mg/L (17)	930 mg/L (11)	3,038 mg/L (4)
TOC	445 mg/L (15)	377 mg/L (8)	150 mg/L (3)
TSS	202 mg/L (17)	49 mg/L (9)	100 mg/L (4)
Alpha Terpineol	746 ug/L (2)	123 ug/L (1)	-
Benzoic Acid	75 ug/L (4)	9,308 ug/L (1)	-
P-Cresol	25 ug/L (5)	117 ug/L (2)	-
Phenol	17 ug/L (8)	242 ug/L (4)	820 ug/L (1)
Chromium	27 ug/L (16)	31 ug/L (9)	10 ug/L (3)
Zinc	145 ug/L (16)	93 ug/L (12)	139 ug/L (4)

( ): Parentheses denote number of observations (number of landfills with data).

6.0 WASTEWATER GENERATION AND CHARACTERIZATION

In 1994, under the authority of Section 308 of the Clean Water Act (CWA), the Environmental Protection Agency (EPA) distributed a questionnaire entitled "Waste Treatment Industry Questionnaire Phase II: Landfills" to 252 facilities that EPA had tentatively identified as possible generators of landfill wastewater. Some of the facilities employed on-site wastewater treatment, while others did not. EPA selected these facilities for survey purposes to represent a total of 1,024 potential generators of landfill wastewater. A total of 220 questionnaire respondents generated landfill leachate in 1992. This section presents information on wastewater generation at these facilities based on the questionnaire responses. In addition, this section also summarizes the information on wastewater characteristics for landfill facilities that EPA sampled and for those facilities that provided self-monitoring data.

6.1 Wastewater Generation and Sources of Wastewater

Landfill facilities do not generate "process wastewater" as EPA has traditionally defined it. At 40 CFR Part 122.2, EPA defines process wastewater as "any water which, during manufacturing or processing, comes into direct contact with or results from the production or use of any raw material, by-product, intermediate product, finished product or waste product". EPA typically uses this definition of process wastewater for manufacturing or processing operations. Since landfill operations do not include or result in "manufacturing processes" or "products", EPA refers to the wastewater treated at landfill facilities as landfill generated wastewater.

In general, the types of wastewater generated by activities associated with landfills and collected for treatment, discharge, or reuse are the following: leachate, landfill gas condensate, truck/equipment washwater, drained free liquids, laboratory derived wastewater, floor washings, recovering pumping wells, contaminated ground water, and storm water runoff. For the purposes of the Landfill industry study, EPA considers all of these wastewater sources "in-scope" except for contaminated ground water, recovering pumping wells, and non-contaminated storm water.

In 1992, landfill facilities in the U.S. generated approximately 22.7 billion gallons of wastewater. For the purposes of this guideline, EPA considers approximately 7.3 billion gallons of this wastewater "in-scope". The remaining 15.4 billion gallons of wastewater generated at landfills consist of contaminated ground water, wastewater recovered from pump wells, and non-contaminated storm water. The primary sources of wastewater at landfills are defined below.

Landfill leachate as defined at 40 CFR Part 258.2, is liquid that has passed through or emerged from solid waste and contains soluble, suspended, or miscible materials removed from such waste. Over time, the seepage of water through the landfill as a result of precipitation may increase the mobility of pollutants and, thereby, increase the potential for their movement into the wider environment. As water passes through the layers of waste, it may "leach" pollutants from the disposed waste, moving them deeper into the soil. This mobility may present a potential hazard to public health and the environment through ground water contamination and other means. One measure used to prevent the movement of toxic and hazardous waste constituents from a landfill is a landfill liner operated in conjunction with a leachate collection system. Leachate is typically collected from a liner system placed at the bottom of the landfill. Leachate also may be collected through the use of slurry walls, trenches, or other containment systems. The leachate generated varies from site to site based on a number of factors including the types of waste accepted, operating practices (including shedding, daily cover, and capping), the depth of fill, compaction of wastes, annual precipitation, and landfill age. Landfill leachate accounts for over 97 percent of the total volume of in-scope wastewater.

Landfill gas condensate is a liquid which has condensed in the landfill gas collection system during the extraction of gas from within the landfill. Gases such as methane and carbon dioxide are generated due to microbial activity within the landfill and must be removed to avoid hazardous and explosive conditions. In gas collection systems, gases containing high concentrations of water vapor condense in traps staged throughout the gas collection network. The gas condensate contains volatile compounds and accounts for a relatively small percentage of flow from a landfill.

Drained free liquids are aqueous wastes drained from waste containers (e.g., drums, trucks, etc.) or wastewater resulting from waste stabilization prior to landfilling. Landfills that accept containerized waste may generate this type of wastewater. Wastewater generated from these waste processing activities is collected and usually combined with other landfill generated wastewater for treatment at the wastewater treatment plant.

Truck/equipment washwater is generated during either truck or equipment washes at landfills. During routine maintenance or repair operations, trucks and/or equipment used within the landfill (e.g., loaders, compactors, or dump trucks) are washed and the resultant wastewater is collected for treatment. In addition, it is common practice for many facilities to wash the wheels, body, and undercarriage of trucks used to deliver the waste to the open landfill face upon leaving the landfill. On-site wastewater treatment equipment and storage tanks are also periodically cleaned and their associated washwaters are collected. Floor washings generated during routine cleaning and maintenance of the facility also are collected for treatment.

Laboratory-derived wastewater is generated from on-site laboratories which characterize incoming waste streams and monitor on-site treatment performance. Landfill facilities usually combine the very small amounts of wastewater from this source with leachate and other wastewater for treatment at the wastewater treatment plant.

Contaminated storm water is storm water which comes in direct contact with landfill wastes, the waste handling and treatment areas, or wastewater that is subject to the limitations and standards. Some specific areas of a landfill that may produce contaminated storm water include (but are not limited to) the following: the open face of an active landfill with exposed waste (no cover added); the areas around wastewater treatment operations; trucks, equipment or machinery that has been in direct contact with the waste; and waste dumping areas. Storm water that does not come into contact with these areas was not considered to be within the scope of this study.

Landfill operations also generate and discharge wastewater that is not covered by this regulation. These sources include non-contaminated storm water, contaminated ground water, and wastewater from recovering pumping wells. Chapter 2: "Scope of the Regulation" discusses the exclusion of these flows. A brief description of this wastewater is presented below.

Non-contaminated (non-contact) storm water is storm water that does not come in direct contact with landfill wastes, the waste handling and treatment areas, or wastewater that is subject to the limitations and standards. Non-contaminated storm water includes storm water which flows off the cap, cover, intermediate cover, daily cover, and/or final cover of the landfill.

Contaminated ground water is water below the land surface in the zone of saturation which has been contaminated by landfill leachate. Contaminated ground water occurs at landfills without liners or at facilities that have released contaminants from a liner system into the surrounding ground water. Ground water can also infiltrate the landfill or the leachate collection system if the water table is high enough to penetrate the landfill area.

Recovering pumping wells generate wastewater as a result of the various ancillary operations associated with ground water pumping operations. These operations include construction and development, well maintenance, and well sampling (i.e. purge water). The wastewater will have very similar characteristics to contaminated ground water.

6.2 Wastewater Flow and Discharge

Tables 6-1 through 6-4 present national estimates of the flows for primary wastwater sources found at landfills reported in "Section A" of the "Waste Treatment Industry Questionnaire Phase II: Landfills". Chapter 3, Section 3.2.1 discusses how EPA calculated national estimates. The Agency based the national estimates presented in Tables 6-1 through 6-4 on 167 of the 220 facilities that generate and treat landfill leachate. EPA excluded the remaining 53 facilities from this guideline as discussed in Chapter 2. EPA

considers these 167 landfill facilities as "in-scope" facilities, or within the scope of the regulation. The tables report the flows by subcategory, as follows: Non-Hazardous subcategory (broken down into Subtitle D municipal solid waste and non-municipal solid waste facilities) and Hazardous subcategory. The tables also show the amount of wastewater flow from landfills by discharge status, as follows: direct, indirect, and zero.

Direct discharge facilities are those that discharge their wastewater directly into a receiving stream or body of water. Based on national estimates, there were no direct discharging hazardous landfills identified in the Landfills industry study. Indirect discharging facilities discharge their wastewater indirectly to a publicly-owned treatment works (POTW). Zero or alternative discharge facilities use treatment and disposal practices that result in no discharge of wastewater to surface waters or POTWs. Alternative disposal options for landfill generated wastewater include off-site treatment at another landfill wastewater treatment system or a Centralized Waste Treatment facility, deep well injection, incineration, evaporation, land application, and recirculation back to the landfill.

Tables 6-1, 6-2, and 6-3 present wastewater flows by subcategory (Hazardous and Non-Hazardous, which is divided into Municipal and Non-Municipal) and discharge type for the different types of wastewater generated by landfills in 1992. Total flows are reported for wastewater treated on site and off site, discharged untreated to a POTW or surface water, and recycled flows that are put back into the landfill. Wastewater flows identified as "Other" treatment include evaporation, incineration, or deep well injection.

Table 6-4 combines the in-scope wastewater flows from Tables 6-1, 6-2, and 6-3. Table 6-4 does not include out-of-scope flows from contaminated ground water, recovering pumping wells, or storm water. The table presents the national estimates of facilities subject to this guideline and the estimated wastewater flows from these facilities.

6.2.1 Wastewater Flow and Discharge at Subtitle D Non-Hazardous Landfills

Landfill facilities generated approximately 7 billion gallons of in-scope wastewater at non-hazardous landfills

in 1992. Flows collected from leachate collection systems are the primary source of wastewater, accounting for over 98 percent of the in-scope wastewater generated at non-hazardous landfills.

Landfill facilities subject to this guideline have several options for the discharge of their wastewater. EPA estimates that there are 143 Subtitle D non-hazardous facilities discharging wastewater directly into a receiving stream or body of water, accounting for 1.1 billion gallons per year. In addition, there are 756 facilities discharging wastewater indirectly to a POTW, accounting for 4.7 billion gallons per year.

Also, there are a number of facilities which use treatment and disposal practices that result in no discharge of wastewater to surface waters. The Agency estimates that there are 338 of these zero or alternative discharge facilities. Several zero or alternative discharge facilities in the Non-Hazardous subcategory recycle wastewater flows back into the landfill. The recirculation of leachate is generally believed to encourage the biological activity occurring in the landfill and accelerates the stabilization of the waste. The recirculation of landfill leachate is not prohibited by federal regulations, although many states have prohibited the practice. EPA estimates that 348 million gallons of landfill wastewater are recirculated back to Subtitle D non-hazardous landfill units each year.

6.2.2 Wastewater Flow and Discharge at Subtitle C Hazardous Landfills

Hazardous landfill facilities generated approximately 342 million gallons of in-scope wastewater in 1992. Flows collected from leachate collection systems are the primary source of wastewater, accounting for approximately 72 percent of the in-scope wastewater generated at hazardous landfills, and routine maintenance activities such as truck/equipment washing and floor washing account for 26 percent of the flows.

Landfill facilities have several options for the discharge of their wastewater. EPA's survey of the Landfills industry did not identify any hazardous landfills subject to the guideline that discharge in-scope wastewater

directly to surface waters. EPA estimates that there are 6 facilities discharging wastewater indirectly to POTWs, accounting for 40 million gallons per year.

The Agency estimates that 139 hazardous landfill facilities use zero or alternative discharge disposal options which account for over 302 million gallons per year. EPA estimates that 102 facilities ship wastewater off site for treatment, often to a treatment plant located at another landfill or to a Centralized Waste Treatment facility. Shipping off site accounts for 9 million gallons per year of wastewater. Another 36 facilities use underground injection for disposal of their wastewater, accounting for 312 million gallons per year, while 1 facility solidifies less than 0.1 million gallons per year of landfill wastewater.

6.3 Wastewater Characterization

The Agency collected the information reported in this section through its sampling program and data supplied by the Landfills industry via technical questionnaires. EPA sampling programs consisted of five-day events at landfills with selected BAT treatment systems (EPA sampled both raw leachate and treated effluent at these facilities) as well as one-day events to characterize raw leachate quality at landfill facilities. The Agency also used industry-provided data, as supplied in the Detailed Questionnaire and in the Detailed Monitoring Questionnaire responses, to characterize landfill generated wastewater. In addition, for the proposal, EPA used data collected as part of the Centralized Waste Treatment Industry study (see reference 31) and Comprehensive Environmental Response, Compensation and Liability Act (CERCLA) ground water study (see reference 25) in the characterization of the wastewater from hazardous landfill facilities. However, after proposal, EPA decided not to include CERCLA data in characterizing hazardous landfill leachate because CERCLA discharges consisted primarily of ground water, which is not a wastewater flow covered by this regulation. Chapter 4 discusses these data sources in detail as well as the QA/QC procedures and editing rules used to evaluate these data. EPA characterized the raw wastewater for each subcategory by taking the median influent concentration from all data sources for each pollutant detected in that subcategory. This pollutant concentration is referred to as the Median Raw Wastewater Concentration File.

This section presents background information on the types of wastewater generated at landfill facilities and the factors that affect the wastewater characteristics. It also discusses the pollutant parameters analyzed and detected at EPA sampling episodes and the methodology for developing the Median Raw Wastewater Concentration File. This section also presents available literature data on the wastewater characteristics of Non-Hazardous subcategory landfill generated wastewater.

6.3.1 Background Information

Landfill generated wastewater is comprised of several wastewater sources that EPA discussed in Section 6.1. Wastewater that is subject to the landfill regulation includes landfill leachate, landfill gas condensate, truck/equipment washwater, drained free liquids, laboratory-derived wastewater, floor washings, and contaminated storm water runoff. Wastewater sources at landfills which are not subject to the landfill regulation include contaminated ground water, wastewater from recovering pumping wells, and non-contaminated storm water. The following section discusses the primary sources of in-scope landfill generated wastewater.

6.3.1.1 Landfill Leachate

Leachate is the liquid which passes through or emerges from solid waste, and contains soluble, suspended, or miscible materials removed from such waste. Several factors affect leachate quality, including the following:

- types of waste accepted/deposited
- operating practices (shredding, cover, and capping)
- amount of infiltration
- depth of fill
- compaction
- age

Waste types received for disposal are the most representative characteristic of a landfill and, therefore, of the wastewater generated, since the main contaminants in the wastewater are derived from the materials deposited into the fill (see Chapter 5: Industry Subcategorization). The amount of infiltration and the age of a landfill primarily affect the concentration of contaminants in the leachate. The remaining factors mainly influence the rate of infiltration.

EPA considered the following two factors when characterizing landfill leachate: the concentration of contaminants in the leachate and the volume of leachate generated. On a relative basis, the highest concentrations of contaminants are typically present in leachate of new or very young landfills. However, the overall loads (i.e., the mass) of pollutants are generally not very large because new landfills typically generate low volumes of leachate. As the volume of waste approaches the capacity of the landfill and the production of leachate increases, both the pollutant loadings (flow x concentration) and the concentrations of certain contaminants (mainly organic pollutants) increase. The increase of pollutant concentration is attributed to the onset of decomposition activities within the landfill and to the leachate traversing the entire depth of refuse. Therefore, large pollutant loadings from a typical landfill occur during a period of high leachate production and high contaminant levels (see reference 13). The exact periods of varying leachate production cannot be quantified readily but are site specific and dependent on each of the above variables.

Over a period of time (as the landfill ages and leaching continues), the concentration of contaminants in the leachate decreases (see reference 13). The landfill may continue to generate substantial quantities of leachate; however, pollutant loadings are lower due to the lower concentrations of soluble, suspended, or miscible contaminants remaining in the landfill. As decomposition of the landfill continues, the landfill attains a stabilized state of equilibrium where further leaching produces leachate with lower loadings than during the period of peak leachate production. This stabilized state is presumably the result of decomposition of landfill waste by indigenous microorganisms, which will remove many of the contaminants usually susceptible to further leaching.

Biological decomposition of landfill municipal refuse is often modeled after the anaerobic breakdown of other organic wastes. The following discussion of the decomposition process has been adapted from a report on the characteristics of landfill leachate prepared by the Wisconsin Department of Natural Resources (see reference 13).

Biological activity occurs in a landfill shortly after deposition of organic material. At first, wastes high in moisture content decompose rapidly under aerobic conditions, creating large amounts of heat. As oxygen is depleted, the intermediate anaerobic stage of decomposition begins. This change from aerobic to anaerobic conditions occurs unevenly through the landfill and depends upon the rate of oxygen diffusion in the fill layers. In the first stage of anaerobic decomposition, extra-cellular enzymes convert complex organic wastes to soluble organic molecules. Once the organic wastes are solubilized, the second stage of anaerobic decomposition converts them to simple organic molecules, such as acetic, propionic, and butyric acids, and other organic acids. These soluble organic acids enter the leachate percolating through a landfill, resulting in decreased pH of the leachate and increasing oxygen demand. Anaerobic activity in the landfill can also lower the reduction oxidation (redox) potential of the wastes which, under low pH conditions, can cause an increase in inorganic contaminants. Eventually, bacteria within the landfill begin converting the organic acids to methane. The absence of organic acids in the landfill increases the pH of the leachate which can lead to a decrease in the solubility of inorganic contaminants, lowering inorganic concentrations in the leachate (see reference 13).

The age or degree of decomposition of a landfill may, in certain circumstances, be ascertained by observing the concentration of various leachate indicator parameters, such as BOD_5, TDS, or the organic nitrogen concentration (see reference 13). The concentrations of these leachate indicator parameters can vary over the decomposition life of a landfill. Typically, older landfills have lower concentrations of BOD_5, COD, and most organic pollutants, indicating a smaller amount of degradable compounds from the aged waste. In addition, aged leachates can contain high levels of chemically reduced compounds, such as ammonia, and high chlorides because of the anaerobic environment of the landfill. However, using these indicator

parameters alone does not take into account any refuse-filling variables, such as processing of wastes prior to disposal and fill depth. To compensate for these additional variables, researchers examined ratios of leachate parameters over time (see reference 13). One such ratio is the ratio of BOD_5 to COD in the leachate. Leachates from younger landfills typically exhibit BOD_5 to COD ratios of approximately 0.8, while older landfills exhibit a ratio as low as 0.1. The decline in the BOD_5 to COD ratio with age is due primarily to the readily decomposable material (phenols, alcohols) degrading faster than the more recalcitrant compounds (heavy molecular weight organic compounds). As a result, as the landfill ages the BOD_5 of the leachate will decrease faster than the COD. Other ratios examined that reportedly decrease over time include the following: volatile solids to fixed solids, volatile acids to TOC, and sulfate to chloride (see reference 13).

It is common to find that the sum of individual organic contaminants does not always match the measured TOC and/or COD value. Compounds that comprise this difference are not always readily identified due to the complex nature of leachate and due to the presence of other organic compounds found in leachate. Myriad organic compounds exist in decomposing refuse and most of the organics in leachate are soluble. Reportedly, free volatile acids constitute the main organic fraction in leachate (see reference 13). However, other organic compounds have been identified in landfill leachates including carbohydrates, proteins, and humic and fulvic-like substances. Gaps in mass balance results are typically attributed to these compounds.

Responses to EPA's Detailed Questionnaire indicate that 1,625 in-scope landfills collect leachate at a median daily flow of 6,000 gallons per day. In 1992, in-scope landfills in the U.S. generated approximately 7.2 billion gallons of landfill leachate. Of this, approximately 1.6 billion gallons were treated on site, 719 million gallons were treated off site, 3.7 billion gallons were sent untreated to POTWs, 417 million gallons were sent untreated to a surface water, 348 million gallons were recycled back to the landfill, and 358 million gallons were treated or disposed by other methods, such as off-site treatment at another landfill wastewater treatment system or a Centralized Waste Treatment facility, deep well injection, incineration, evaporation, or land application.

6.3.1.1.1 Additional Sources of Non-Hazardous Leachate Characterization Data

Most of the existing literature regarding non-hazardous landfill leachate characteristics resulted from studies taken at an isolated range of municipal landfills in the 1970s and 1980s. Data presented in these reports on pollutant concentrations found in leachate are typically expressed in ranges due to the variability of leachate from various landfills. The range of pollutant concentration values, as well as the lack of specific information on factors affecting leachate results (e.g., sampling methods, analytical methods, landfill waste types, etc.) limit the usefulness of these data. However, these data are mentioned as additional background information in support of EPA's characterization activities. Table 6-5 presents a summary of available municipal leachate characteristic data from the following sources:

- Five published papers: George, 1972; Chian and DeWalle, 1977; Metry and Cross, 1977; Cameron, 1978; and Shams-Korzani and Henson, 1993.
- McGinley, Paul M. and Kmet, P. "Formation, Characteristics, Treatment and Disposal of Leachate from Municipal Solid Waste Landfills." Wisconsin Department of Natural Resources Special Report, August 1984, and
- Sobotka & Co., Inc. Case history data compiled and reported to U.S. EPA's Economic Analysis Branch, Office of Solid Waste, July 1986.

The variability and high pollutant concentrations in older landfill leachate characterization data can be attributed to landfills that accepted waste prior to the enactment of the Resource Conservation and Recovery Act (RCRA) in 1980. Landfills in operation prior to this date may have disposed of a multitude of different industrial and/or toxic wastes in addition to municipal solid waste. The disposal of these high-strength wastes could account for the large variability observed in leachate characteristics data collected from municipal landfills in this period. After the promulgation of RCRA, EPA established controls that specified the type and characteristics of wastes that may be received by either a hazardous (Subtitle C) or non-hazardous (Subtitle D) facility (see Chapter 3: Section 3.1 for the discussion on regulatory history). EPA has also mandated other control measures, such as leachate collection systems, under RCRA for both types of landfills. By instituting the acceptance criteria and leachate control standards under RCRA, the

characteristics of the leachate from both hazardous and non-hazardous landfills do not vary as greatly as observed in landfills prior to 1980. EPA's data shows that RCRA regulations have resulted in smaller concentration ranges for pollutants from landfills. EPA did observe pollutant variability in the data it collected; however, the variability was not as great as found in the data from older literature sources.

6.3.1.2 Landfill Gas Condensate

Landfill gas condensate forms in the collection lines used to extract and vent landfill gas. Condensate collects at low points in the gas collection lines and landfill facilities usually pump it to the on-site wastewater holding tank or treatment system. Responses to EPA's Detailed Questionnaire indicate that 158 in-scope landfills collect landfill gas condensate at a median daily flow of 343 gallons per day. In 1992, in-scope landfills in the U.S. generated approximately 23 million gallons of landfill gas condensate. Of this, approximately 20 million gallons were treated on site, 1.7 million gallons were treated off site, and 0.8 million gallons were sent untreated to POTWs. Of the 155 facilities collecting gas condensate, 66 commingle condensate with leachate for treatment on site, 79 facilities do not treat the condensate on site, and 10 facilities treat landfill gas condensate separately from other landfill generated wastewater.

Landfill gas condensate represents a small amount of the total wastewater flow for the industry. Hazardous waste landfills produce 9 million gallons/year of gas condensate, or about 4 percent of the leachate flow volume. Municipal solid waste landfills produce 14 million gallons/year of gas condensate, or about 0.2 percent of the leachate flow volume.

Of the 37 respondents to the Detailed Questionnaire that collect landfill gas condensate, five facilities treat the condensate separately from leachate. These facilities treated landfill gas condensate with one or more of the following technologies: equalization, neutralization, oil-water separation, granular activated carbon, and air stripping. All five facilities discharged the treated waste stream indirectly to a POTW. Table 6-6 presents landfill gas condensate monitoring data provided in the Detailed Questionnaire from two facilities that collect and treat landfill gas condensate separately from other landfill generated wastewater. Facility

16012 presented landfill gas condensate monitoring data after treatment by hydrocarbon/aqueous phase separation and caustic neutralization, and facility 16015 presented monitoring data after treatment by equalization, caustic neutralization, and carbon adsorption.

6.3.1.3 Drained Free Liquids

Drained free liquids are liquids drained from containerized waste prior to landfilling. Wastewater characteristics and volume of drained free liquids vary greatly depending upon the contents and origin of the waste. However, they will have similar characteristics to the containerized waste and, therefore, similar characteristics to landfill leachate. Drained free liquids include other wastewater generated by waste processing activities, such as waste stabilization. Waste stabilization includes the chemical fixation or solidification of the solid waste. Wastewater generated from these activities includes decant from the waste treated and any associated rinse waters. This waste processing wastewater is collected separately and then combined with leachate and other landfill operation wastewater for treatment at the wastewater treatment facility.

Responses to EPA's Detailed Questionnaire indicate that 25 in-scope landfills collect drained free liquids at a median daily flow of 3 gallons per day. In 1992, in-scope landfills in the U.S. generated approximately 0.5 million gallons of drained free liquids. Of this, approximately 715 gallons were treated on site and 47,000 gallons were treated or disposed by other methods such as treatment by a CWT or deep well injection.

6.3.1.4 Truck and Equipment Washwater

Landfill facilities generate truck and equipment washwater during either truck or equipment washes at the landfill. Depending on the type and usage of the vehicle/equipment cleaned and the type of landfill, the washwater volume and characteristics can vary greatly. For hazardous and non-hazardous landfill facilities, washwater will typically be more dilute in strength in comparison to typical leachate characteristics and contain mostly solids. Insoluble solids, consisting of mostly inorganics, metals, and low concentrations of

organic compounds are the primary source of contaminants in the washwater. Since truck and equipment washwater tends to contain the same constituents as the waste being landfilled and are similar in characteristic to the landfill leachate, they are typically combined for treatment with leachate and other landfill generated wastewater.

Responses to EPA's Detailed Questionnaire indicate that 356 in-scope landfills collect truck and equipment washwater at a median daily flow of 141 gallons per day. In 1992, in-scope landfills in the U.S. generated approximately 101 million gallons of truck and equipment washwater. Of this, approximately 38 million gallons were treated on site, 9 million gallons were sent untreated to POTWs, 1.3 million gallons were either treated off site, recycled back to the landfill, or sent untreated to a surface water, and 53 million gallons were treated or disposed by other methods, such as off-site treatment at another landfill wastewater treatment system or a Centralized Waste Treatment facility, deep well injection, incineration, evaporation, or land application.

Floor washings are also generated during routine cleaning and maintenance of landfill facilities. Responses to EPA's Detailed Questionnaire indicate that 68 in-scope landfills collect floor washings at a median daily flow of 985 gallons per day. In 1992, in-scope landfills in the U.S. generated approximately 45 million gallons of floor washings. Of this, approximately 6.4 million gallons were treated on site, 3.3 million gallons were sent untreated to POTWs, and 35 million gallons were treated or disposed by other methods, as discussed above.

6.3.2 Pollutant Parameters Analyzed at EPA Sampling Episodes

EPA conducted 19 sampling episodes at 18 landfill facilities. The Agency conducted five episodes at hazardous landfill facilities and 13 at non-hazardous facilities. EPA conducted one-day sampling episodes for the purpose of collecting raw wastewater samples to characterize landfill generated wastewater. Samples collected during the week-long sampling episodes included raw wastewater samples as well as

intermediate and effluent samples to evaluate the entire wastewater treatment system. Chapter 4 discusses these data collection activities in further detail.

Table 6-7 presents the pollutants analyzed at the one-day and week-long sampling episodes. EPA analyzed for a total of 470 pollutants in the raw wastewater, intermediate, and treated effluent waste stream samples, including 232 toxic and nonconventional organic compounds, 69 toxic and nonconventional metals, 4 conventional pollutants, and 165 toxic and nonconventional pollutants including pesticides, herbicides, dioxins, and furans. The list of pollutants analyzed are included under the following analytical methods: method 1613 for dioxins/furans, method 1620 for metals, method 1624 for volatile organics, method 1625 for semivolatile organics, and methods 1656, 1657, and 1658 for pesticides/herbicides, as well as classical wet chemistry methods.

Table 6-8 presents the list of pollutants analyzed at EPA sampling episodes by subcategory and episode number and whether EPA detected the pollutant in the facility's raw wastewater. If EPA did not detect a pollutant at a facility, Table 6-8 lists an ND (non-detect) in the appropriate row. If EPA did detect a pollutant at a facility, Table 6-8 lists a blank, and in cases where EPA did not sample for a pollutant at a facility, Table 6-8 lists a dash.

EPA collected composite samples at the week-long sampling events at episodes 4626, 4667, 4687, 4690, 4721, and 4759, while EPA collected grab samples at the remaining 12 one-day sampling events. The Agency developed a preliminary list of pollutants of interest by eliminating those pollutants that EPA never detected at any facility in a subcategory from the initial list of 470 pollutants. For the Non-Hazardous subcategory, EPA sampling never detected 316 pollutants in the raw wastewater at Subtitle D municipal facilities and 324 pollutants in the raw wastewater at Subtitle D non-municipal facilities. For the Hazardous subcategory, EPA sampling never detected 250 pollutants in the raw wastewater. Therefore, out of the 470 pollutants initially analyzed for, EPA detected 154 pollutants at least once at Subtitle D municipal facilities and 146 pollutants at least once at Subtitle D non-municipal facilities. For the Hazardous

subcategory, EPA detected 220 pollutants at least once at hazardous facilities. Using the editing criteria presented in Chapter 7, the Agency reduced this preliminary list of pollutants of interest to the final list of 32 pollutants of interest for the Non-Hazardous subcategory (32 pollutants of interest for Subtitle D municipal facilities and 9 pollutants of interest for Subtitle D non-municipal facilities) and 63 pollutants of interest for the Hazardous subcategory. Tables 6-9 and 6-10 present the median concentration for the pollutants of interest for both subcategories.

6.3.3 Raw Wastewater Characterization Data

In order to characterize wastewater from the Landfills industry, EPA compiled raw wastewater data from EPA sampling, the Detailed Questionnaire, the Detailed Monitoring Questionnaire, and the Centralized Waste Treatment Industry (CWT) database.

EPA reviewed each data source to determine if the data was representative of landfill generated wastewater. First, EPA selected only those sample points corresponding to raw wastewater. Second, EPA used several criteria to eliminate sampling data not considered representative of raw landfill wastewater. In characterizing landfill raw wastewater, EPA included only sampled wastewater containing at least 85 percent leachate and/or gas condensate. Therefore, EPA eliminated raw wastewater data that consisted mainly of wastewater that is not subject to this rule (e.g., storm water, ground water, or sanitary wastewater). Also, EPA eliminated wastewater data containing industrial process wastewater. This eliminated the possibility of finding pollutants that may not have originated in a landfill.

Next, EPA grouped all data points according to the classification of the landfill, e.g. municipal solid waste, hazardous waste, or Subtitle D non-municipal solid waste. Many facilities provided data from both technical questionnaires (the Detailed Questionnaire and the Detailed Monitoring Questionnaire), and in several instances, EPA conducted sampling at a facility that also provided data in the technical questionnaires. In these cases, EPA combined all data from the facility to obtain a facility average concentration for each pollutant. For each subcategory, EPA gathered the facility averages for all pollutants

into a file called the Raw Wastewater Source File. EPA then calculated the median of the facility average concentrations in the Source File to determine the median raw wastewater concentration for each pollutant in the subcategory. Tables 6-9 and 6-10 present the median values for the Non-Hazardous and Hazardous subcategories, respectively. EPA refers to this file as the Median Raw Wastewater Concentration File. Tables 6-11 through 6-13 present, by subcategory, the minimum and maximum of the facility average concentrations in the Raw Wastewater Source File, along with the number of observations and number of non-detect values. Note that although EPA included CERCLA data in the characterization of hazardous landfill leachate for the proposal, EPA did not include CERCLA data for raw wastewater characterization for the final rule. The CERCLA data consists primarily of contaminated ground water and, since contaminated ground water is not subject to the regulations, EPA determined that CERCLA data should not be used for hazardous landfill wastewater characterization. Therefore, the raw wastewater characterization data for the Hazardous subcategory presented in Tables 6-11 through 6-13 do not include CERCLA data.

6.3.4 Conventional, Toxic, and Selected Nonconventional Pollutant Parameters

The Clean Water Act defines different types of pollutant parameters used to characterize raw wastewater. These parameters include conventional, nonconventional, and toxic pollutants. Conventional pollutants found in landfill generated wastewater include the following:

- Total Suspended Solids (TSS)
- 5-day Biochemical Oxygen Demand (BOD_5)
- pH
- Oil and Grease (measured as Hexane Extractable Material)

Total solids in wastewater are defined as the residue remaining upon evaporation of the liquid at just above its boiling point. TSS is the portion of the total solids that can be filtered out of solution using a 1 micron filter. Raw wastewater TSS in leachate is a function of the type and form of wastes accepted for disposal

at landfill facilities. Landfill design and operational parameters such as depth of fill, compaction, and capping also influence the concentration of TSS. BOD_5 is one of the most important gauges of pollution potential of a wastewater and varies with the amount of biodegradable matter that can be assimilated by biological organisms under aerobic conditions. The nature of the chemicals contained in landfill generated wastewater affects the BOD_5 due to the differences in susceptibility of different molecular structures to microbiological degradation. Landfill generated wastewater containing compounds with lower susceptibility to decomposition by microorganisms tends to exhibit lower BOD_5 values, even though the total organic loading may be much higher when compared to wastewater exhibiting substantially higher BOD_5 values. For example, a landfill generated wastewater may have a low BOD_5 value while, at the same time, exhibiting a high TOC or COD concentration. Raw wastewater BOD_5 values can vary depending on the waste deposited in the landfill and the landfill age, as noted previously in Section 6.3.1.1.

The pH of a solution is a unitless measurement which represents the acidity or alkalinity of a wastewater stream (or aqueous solution) based on the disassociation of the acid or base in the solution into hydrogen (H^+) or hydroxide (OH^-) ions, respectively. Raw wastewater pH can be a function of the waste deposited in a landfill but can vary depending on the conditions within the landfill, as noted previously in Section 6.3.1.1. Fluctuations in pH are controlled readily by equalization followed by neutralization. Control of pH is necessary to achieve proper removal of pollutants in treatment systems such as metals precipitation and biological treatment systems.

Oil and grease also may be present in selected landfill generated wastewater. Proper control of oil and grease is important because it can interfere with the operation of certain wastewater treatment system processes such as chemical precipitation and the settling operations in biological systems. If it is not removed prior to discharge, excessive levels of oil and grease can interfere with the operation of POTWs and can create a film along surface waters, disrupting the biological activities in those waterways.

Table 6-11 presents the minimum and maximum facility average concentration data for TSS, BOD_5, and oil and grease for each landfill subcategory and the minimum and maximum facility average values for pH. EPA obtained the minimum and maximum values presented for each pollutant in the table from the Raw Wastewater Source File for both subcategories. The Source File contains many pollutants which EPA detected at least once in a subcategory but were not necessarily selected as pollutants of interest. EPA discusses the selection of pollutants of interest in Chapter 7.

EPA also used certain classical nonconventional pollutants for the purposes of raw wastewater characterization. These pollutant parameters include the following: ammonia as nitrogen, nitrate/nitrite, total dissolved solids, total organic carbon, total phenols, chemical oxygen demand, amenable cyanide, and total phosphorus. All of these pollutants are pollutants of interest for either the Non-Hazardous or Hazardous subcategory, with the exception of total phosphorus. For the purposes of presenting raw wastewater characterization data, EPA included these nonconventional pollutants with the conventional pollutants for each landfill subcategory in Table 6-11.

6.3.5 Toxic Pollutants and Remaining Nonconventional Pollutants

Table 6-12 presents the minimum and maximum facility-average concentration data for metals and toxic pollutants for the Non-Hazardous and Hazardous subcategories. EPA obtained the minimum and maximum values presented for each pollutant in the table from the Raw Wastewater Source File for both subcategories. Most of the pollutants included in Table 6-12 are pollutants of interest for either the Non-Hazardous or Hazardous subcategory. EPA detected a wide range of metals in raw wastewater from landfill facilities in both subcategories including both toxic pollutant and nonconventional pollutant metals.

Table 6-13 presents the minimum and maximum facility average concentration data for organic toxic and nonconventional pollutants for the two subcategories. EPA obtained the minimum and maximum values presented for each pollutant in the table from the Raw Wastewater Source File for both subcategories. All pollutants included in Table 6-13 are pollutants of interest for either the Non-Hazardous or Hazardous

subcategory. EPA detected a wide range of organic pollutants in raw wastewater at landfill facilities in both subcategories. Many of these are common organic pollutants found in municipal or commercial waste.

6.3.6 Raw Wastewater at Subtitle D Non-Hazardous Landfills

6.3.6.1 Raw Wastewater at Subtitle D Municipal Landfills

Raw wastewater generated at Subtitle D municipal landfills contained a range of conventional, toxic, and nonconventional pollutants. This wastewater also contained significant concentrations of common nonconventional metals such as iron, magnesium, and manganese. These metals are naturally occurring elements found in raw water, and the presence of these metals in landfill raw wastewater can be attributed to background levels in the water source used at the facility. Generally, toxic heavy metals were found at relatively low concentrations. EPA did not find toxic metals such as arsenic, cadmium, mercury, and lead at treatable levels in any of EPA's sampling episodes. Typical organic pollutants found in leachate included 2-butanone (methyl ethyl ketone) and 2-propanone (acetone), which are common solvents used in household products (such as paints and nail polish), and common industrial solvents such 4-methyl-2-pentanone and 1,4-dioxane. EPA detected only trace concentrations of only two pesticides (dichloroprop and disulfoton) in wastewater from municipal landfills. Additionally, EPA's data showed high loads of organic acids such as benzoic acid and hexanoic acid resulting from anaerobic decomposition of solid waste.

EPA identified 32 pollutants of interest for Subtitle D municipal landfills, including the following: eight conventional/nonconventional pollutants, six metals, 16 organics and pesticides/herbicides, and two dioxins/furans. In the Agency's sampling episodes, EPA never detected 316 pollutants, while approximately 122 pollutants were detected but were not present above the minimum level.

6.3.6.2 Raw Wastewater at Subtitle D Non-Municipal Landfills

A subset of the Subtitle D Non-Hazardous landfill subcategory is the Subtitle D non-municipal landfill.

These types of landfills do not accept municipal solid waste or household refuse. Rather, these facilities accept a number of different types of non-hazardous, non-municipal solid wastes. Waste types accepted at Subtitle D non-municipal facilities include, but are not limited to, municipal incinerator ash, industrial non-hazardous wastes and sludges, wastewater treatment plant sludge, yard waste, and construction and demolition wastes.

EPA identified 9 pollutants of interest for Subtitle D non-municipal landfills, including the following: eight conventional/nonconventional pollutants and one metal. In the Agency's sampling episodes, EPA never detected 324 pollutants, while 136 pollutants were detected but were not present above the minimum level.

Many Subtitle D non-municipal facilities accept two or more of the non-municipal waste types discussed above. Certain facilities accept only one type of waste and are referred to as "monofills". EPA performed an analysis to determine if significant differences existed in raw wastewater characteristics from Subtitle D municipal landfills and these monofill facilities. As discussed in Chapter 5, Section 5.3.1, EPA analyzed characterization data collected at municipal solid waste landfills and monofills as part of EPA's sampling program and analyzed data from several published reports, including prior EPA studies, analyzing construction and demolition monofills, ash monofills, and co-disposal sites. EPA evaluated these data to identify any pollutants found at significant concentrations in monofills that were not found in Subtitle D municipal landfills.

Based on a review of these data sources, EPA observed that the pollutants present in raw wastewater from monofills were not significantly different from those found in Subtitle D municipal landfills, and, in fact, pollutants present in monofills were a subset of those pollutants found at municipal solid waste landfills. In addition, concentrations of virtually all pollutants found in ash, sludge, and construction and demolition waste monofills were significantly lower than those found in raw wastewater from Subtitle D municipal landfills (see Chapter 5, Tables 5-3 and 5-4). EPA acknowledges that there were no organic pollutants of interest detected at Subtitle D non-municipal landfills, and that some monofills, such as ash monofills, may

have a low organic content and, therefore, may not be able to use the selected BPT/BAT treatment technology (biological treatment) to treat the wastewater. However, EPA concluded that these Subtitle D non-municipal facilities can meet the BPT/BAT limitations using available technologies. These treatment systems may be installed at costs comparable to those for biological treatment. As discussed in Chapter 11, EPA established equivalent effluent limitations for all Subtitle D non-hazardous landfills.

6.3.6.3 Dioxins and Furans in Raw Wastewater at Subtitle D Non-Hazardous Landfills

There are 210 isomers of chlorinated dibenzo-p-dioxins (CDD) and chlorinated dibenzofurans (CDF). EPA is primarily concerned with the 2,3,7,8-substituted congeners, of which EPA considers 2,3,7,8-TCDD to be the most toxic and is the only one that is a toxic pollutant. EPA considers non- 2,3,7,8-substituted congeners to be less toxic, in part, because they are not readily absorbed by living organisms. Dioxins and furans may be formed as by-products in certain industrial unit operations related to petroleum refining, pesticide and herbicide production, paper bleaching, and production of materials involving chlorinated compounds. Dioxins and furans are not water-soluble and are not expected to leach out of non-hazardous landfills in significant quantities.

As part of EPA sampling episodes at 13 non-hazardous landfills, EPA analyzed raw wastewater samples for 17 congeners of dioxins and furans. Table 6-14 presents the results of the data analyses. EPA also used additional raw leachate data from ash monofills from previous EPA studies, as discussed in Chapter 5, Section 5.3.1. EPA found low levels of OCDD, HpCDD, and HxCDD in raw wastewater at several landfills. The Agency did not detect the most toxic dioxin congener, 2,3,7,8-TCDD, in raw wastewater at a Subtitle D landfills. All concentrations of dioxins and furans in raw, untreated wastewater were well below the Universal Treatment Standards for F039 wastes (multi-source leachate) in 40 CFR 268.48, which establish minimum concentration standards based on based on the Best Demonstrated Available Treatment Technology (BDAT)[1]. At the concentrations found in raw landfill wastewater, EPA expects

[1] EPA bases UTS on the BDAT for each listed hazardous waste. BDAT represents the treatment technology that EPA concludes is the most effective for treating a particular waste that is also readily available to generators and treaters.

dioxins and furans to partition to the biological sludge as part of the BPT/BAT treatment technologies. EPA included the partitioning of dioxins and furans to the sludge in the evaluation of treatment benefits and water quality impacts. EPA sampling data and calculations conclude that the concentrations of dioxins and furans present in the wastewater would not prevent the sludge from being redeposited in a non-hazardous landfill.

6.3.7 Raw Wastewater at Subtitle C Hazardous Landfills

The Agency used data from EPA sampling episodes and industry supplied data obtained through the technical questionnaires to characterize raw wastewater from Subtitle C hazardous landfills. Wastewater generated at Subtitle C landfills contained a wide range of conventional, toxic, and nonconventional pollutants at treatable levels. There were a significantly greater number of pollutants found in hazardous landfill raw wastewater in comparison to non-hazardous landfills. Pollutants which were common to both untreated non-hazardous and hazardous wastewater were generally an order of magnitude higher in concentration in hazardous landfill wastewater. The list of pollutants of interest for the Hazardous subcategory (presented in Table 6-10), which includes 63 parameters, reflects the more toxic nature of hazardous landfill wastewater and the wide range of industrial waste sources. Chapter 7 discusses the methodology for determining pollutants of interest. For further discussion on the differences between hazardous and non-hazardous landfill leachate, see Chapter 5, Section 5.3.1.

Pollutants typical of raw leachate from hazardous facilities and found at higher median concentrations than at Subtitle D facilities included arsenic, chromium, copper, nickel, and zinc. EPA did not detect cadmium, lead, and mercury at treatable concentrations in the raw wastewater for any of the hazardous landfills sampled during EPA sampling episodes.

EPA identified a total of 63 pollutants of interest for Subtitle C hazardous landfills, including the following: 11 conventional/nonconventional pollutants, 11 metals, 37 organics and pesticides/herbicides, and 4 dioxins/furans. EPA sampling episodes never detected 250 pollutants, while approximately 157 pollutants were detected but were not present above the minimum level.

6.3.7.1 Dioxins and Furans in Raw Wastewater at Subtitle C Hazardous Landfills

As part of EPA sampling episodes at two in-scope Subtitle C landfills and two in-scope pre-1980 industrial landfills, EPA analyzed raw leachate samples for 17 congeners of dioxins and furans. Table 6-15 presents the results of these analyses. As in the Non-Hazardous subcategory, EPA did not detect the most toxic dioxin congener, 2,3,7,8-TCDD, at any in-scope hazardous/industrial landfill. EPA found low levels of several congeners in raw wastewater at many of the sampled landfills. Low levels of OCDD, OCDF, HpCDD, and HpCDF were detected in over half of the landfills sampled. However, all concentrations of dioxins and furans in raw, untreated wastewater were well below the Universal Treatment Standards (UTS) for F039 wastes (multi-source leachate) in 40 CFR 268.48, which establish minimum concentration standards based on BDAT. At the concentrations found in raw landfill wastewater, EPA expects dioxins and furans to partition to the biological sludge as part of the BPT/BAT treatment technologies.

Table 6-1: Wastewater Generated in 1992: Hazardous Subcategory (gallons)

Discharge Type	Wastewater Type	Treated On-Site	Treated Off-Site	Untreated to POTW	Untreated to Surface Water	Recycled Flow	Other
Indirect	Leachate	37,600,000	0	0	0	0	0
	Gas Condensate	772,000	0	0	0	0	0
	Truck/Equipment Washwater	1,220,000	0	101,000	0	0	0
	Floor Washings	706,000	0	0	0	0	0
	Storm Water	0	0	4,740,000	294,000,000	0	0
	Total Indirect	40,298,000	0	4,841,000	294,000,000	0	0
Zero	Leachate	18,100,000	20,600,000	0	0	0	169,000,000
	Gas Condensate	8,390,000	0	0	0	0	0
	Drained Free Liquids	0	0	0	0	0	47,000
	Truck/Equipment Washwater	28,400	513,000	0	0	0	50,300,000
	Floor Washings	0	0	0	0	0	35,000,000
	Contaminated Ground Water	28,700,000	0	0	0	0	0
	Storm Water	0	2,300,000	30,700,000	662,000,000	0	0
	Total Zero	55,218,400	23,413,000	30,700,000	662,000,000	0	254,347,000
Subcategory Total		95,516,400	23,413,000	35,541,000	956,000,000	0	254,347,000

Table 6-2: Wastewater Generated in 1992: Non-Hazardous Subcategory Municipal Facilities (gallons)

Discharge Type	Wastewater Type	Treated On-Site	Treated Off-Site	Untreated to POTW	Untreated to Surface Water	Recycled Flow	Other
Direct	Leachate	565,000,000	782,000	804,000	167,000,000	49,000	94,400,000
	Gas Condensate	1,570,000	0	0	0	0	0
	Drained Free Liquids	715	0	0	0	0	0
	Truck/Equipment Washwater	15,300,000	0	0	0	0	0
	Floor Washings	4,890,000	0	0	0	0	0
	Contaminated Ground Water	163,000,000	0	0	0	0	0
	Storm Water	348,000,000	0	0	3,430,000,000	0	0
	Total Direct	1,097,760,715	782,000	804,000	3,597,000,000	49,000	94,400,000
Indirect	Leachate	777,000,000	7,640,000	3,640,000,000	0	29,800,000	5,870,000
	Gas Condensate	9,700,000	65,900	793,000	0	0	19,700
	Truck/Equipment Washwater	20,700,000	0	9,060,000	594,000	0	0
	Floor Washings	794,000	0	3,320,000	0	0	0
	Contaminated Ground Water	226,000,000	0	0	0	0	0
	Storm Water	3,710,000,000	0	677,000,000	3,890,000,000	85,400,000	1,060,000,000

Table 6-2: Wastewater Generated in 1992: Non-Hazardous Subcategory Municipal Facilities (gallons) (cont'd)

Discharge Type	Wastewater Type	Treated On-Site	Treated Off-Site	Untreated to POTW	Untreated to Surface Water	Recycled Flow	Other
Indirect	Total Indirect	4,744,194,000	7,705,900	4,330,173,000	3,890,594,000	115,200,000	1,065,889,700
Zero	Leachate	170,000,000	561,000,000	0	0	233,000,000	88,600,000
	Gas Condensate	0	1,610,000	0	0	0	0
	Truck/Equipment Washwater	425,000	0	0	0	177,000	2,990,000
	Contaminated Ground Water	296,000,000	0	0	0	0	0
	Storm Water	3,930,000	0	0	137,000,000	212,000,000	24,700,000
	Total Zero	470,355,000	562,610,000	0	137,000,000	445,177,000	116,290,000
Subcategory Total		6,312,309,715	571,097,900	4,330,977,000	7,624,594,000	560,426,000	1,276,579,700

Table 6-3: Wastewater Generated in 1992: Non-Hazardous Subcategory Non-Municipal Facilities (gallons)

Discharge Type	Wastewater Type	Treated On-Site	Treated Off-Site	Untreated to POTW	Untreated to Surface Water	Recycled Flow	Other
Direct	Leachate	0	0	0	250,000,000	0	0
	Storm Water	0	0	0	4,900,000	0	0
	Total Direct	0	0	0	254,900,000	0	0
Indirect	Leachate	47,400,000	0	57,000,000	0	85,100,000	0
	Contaminated Ground Water	0	0	4,120,000	0	0	0
	Storm Water	19,800,000	0	0	0	0	43,100,000
	Total Indirect	67,200,000	0	61,120,000	0	85,100,000	43,100,000
Zero	Leachate	56,700	129,000,000	0	0	0	0
	Truck/Equipment Washwater	2,000	0	0	0	0	0
	Total Zero	58,700	129,000,000	0	0	0	0
Subcategory Total		67,258,700	129,000,000	61,120,000	254,900,000	85,100,000	43,100,000

Table 6-4: Quantity of In-Scope Wastewater Generated in 1992 (gallons)

Discharge Status	Subcategory					Total Wastewater Generated	Total Number of Facilities
	Non-Hazardous			Hazardous			
	Subtitle D Municipal	Subtitle D Non-Municipal	Subtitle D Facilities	Subtitle C	Subtitle C Facilities		
Direct	849,679,000	249,659,000	143	0	0	1,099,338,000	143
Indirect	4,509,255,000	189,511,000	756	40,361,000	6	4,739,127,000	762
Zero	1,058,156,000	128,633,000	338	302,112,000	139	1,488,901,000	477
Total	6,417,090,000	567,803,000	1,237	342,473,000	145	7,327,366,000	1,382

Table 6-5: Contaminant Concentration Ranges in Municipal Leachate as Reported in Literature Sources

Pollutant Parameter	George (1972)	Chain/DeWalle (1977)	Metry/Cross (1977)	Cameron (1978)	Wisconsin Report (20 Sites)	Sobotka Report (44 Sites)
Conventional						
BOD	9 - 54,610	81 - 33,360	2,200 - 720,000	9 - 55,000	ND - 195,000	7 - 21,600
pH	3.7 - 8.5	3.7 - 8.5	3.7 - 8.5	3.7 - 8.5	5 - 8.9	5.4 - 8.0
TSS	6 - 2,685	10 - 700	13 - 26,500		2 - 140,900	28 - 2,835
Non-Conventional						
Alkalinity	0 - 20,850	0 - 20,850	310 - 9,500	0 - 20,900	ND - 15,050	0 - 7,375
Bicarbonate			3,260 - 5,730			
Chlorides	34 - 2,800	4.7 - 2,467	47 - 2,350	34 - 2,800	2 - 11,375	120 - 5,475
COD	0 - 89,520	40 - 89,520	800 - 750,000	0 - 9,000	6.6 - 97,900	440 - 50,450
Fluorides				0 - 2.13	0 - 0.74	0.12 - 0.790
Hardness	0 - 22,800	0 - 22,800	35 - 8,700	0 - 22,800	52 - 225,000	0.8 - 9,380
NH3-Nitrogen	0 - 1,106	0 - 1,106	0.2 - 845	0 - 1,106		11.3 - 1,200
NO3-Nitrogen	0 - 1,300	0.2 - 1,0.29	4.5 - 18			0 - 5,0.95
Organic Nitrogen			2.4 - 550			4.5 - 78.2
Ortho-Phosphorus		6.5 - 85	0.3 - 136	0 - 154		
Sulfates	1 - 1,826	1 - 1,558	20 - 1,370	0 - 1,826	ND - 1,850	8 - 500
Sulfide				0 - 0.13		
TOC		256 - 28,000			ND - 30,500	5 - 6,884
TDS	0 - 42,276	584 - 44,900	100 - 51,000	0 - 42,300	584 - 50,430	1,400 - 16,120
Total-K-Nitrogen	0 - 1,416				2 - 3,320	47.3 - 938
Total Phosphorus	1 - 154	0 - 130			ND - 234	
Total Solids		0 - 59,200				1,900 - 25,873
Metals						
Aluminum				0 - 122	ND - 85	0.010 - 5.07
Arsenic				0 - 11.6	ND - 70.2	0 - 0.08
Barium				0 - 5.4	ND - 12.5	0.01 - 10
Beryllium				0 - 0.3	ND - 0.36	0.001 - 0.01
Boron				0.3 - 73	0.867 - 13	
Cadmium		0.03 - 17		0 - 0.19	ND - 0.04	0 - 0.1
Calcium	5 - 4,080	60 - 7,200	240 - 2,570	5 - 4,000	200 - 2,500	95.5 - 2,100
Total Chromium				0 - 33.4	ND - 5.6	0.001 - 1.0
Copper	0 - 9.9	0 - 9.9		0 - 10	ND - 4.06	0.003 - 0.32
Cyanide				0 - 0.11	ND - 6	0 - 4.0
Iron	0.2 - 5,500	0 - 2,820	0.12 - 1,700	0.2 - 5,500	ND - 1,500	0.22 - 1,400
Lead	0 - 0.5	<0.10 - 2.0		0 - 5.0	0 - 14.2	0.001 - 1.11
Magnesium	16.5 - 15,600	17 - 15,600	64 - 547	16.5 - 15,600	ND - 780	76 - 927
Manganese	0.06 - 1,400	0.09 - 125	13	0.06 - 1,400	ND - 31.1	0.03 - 43
Mercury				0 - 0.064	ND - 0.01	0 - 0.02
Molybendum				0 - 0.52	0.01 - 1.43	
Nickel				0.01 - 0.8	ND - 7.5	0.01 - 1.25
Potassium	2.8 - 3,770	28 - 3,770	28 - 3,800	2.8 - 3,770	ND - 2,800	30 - 1,375
Sodium	0 - 7,700	0 - 7,700	85 - 3,800	0 - 7,700	12 - 6,010	
Titanium				0 - 5.0	<0.01	
Vanadium				0 - 1.4	0.01	
Zinc	0 - 1,000	0 - 370	0.03 - 135	0 - 1,000	ND - 731	0.01 - 67

All concentrations in mg/L, except pH (std units).
ND = Non-detect

Table 6-6: Landfill Gas Condensate (from Detailed Questionnaire)

QID	Pollutant	# Obs	# ND	Avg. Conc.	Unit
16012	**Conventional**				
	Oil & Grease	1	0	422	mg/L
	Metals				
	Arsenic	1	0	570	ug/L
16015	**Organics**				
	1,2-Benzenedicarboxylic Acid, Diethyl Ester	3	1	2.0	mg/L
	1,3-Butadiene, 1,1,2,3,4,4-Hexachloro-	3	1	2.2	mg/L
	1,3-Dichlorobenzene	3	1	1.2	mg/L
	1,4-Dichlorobenzene	3	1	2.0	mg/L
	2,4,6-Trichlorophenol	3	2	15.0	mg/L
	2,4-Dichlorophenol	3	2	15.0	mg/L
	2,4-Dimethylphenol	3	2	17.3	mg/L
	2,6-Dinitrotoluene	3	2	5.83	mg/L
	2-Methyl-4,6-Dinitrophenol	3	0	100	mg/L
	2-Nitrophenol	3	2	17.5	mg/L
	3,4-Benzopyrene	3	2	2.0	mg/L
	3-Methyl-4-Chlorophenol	3	1	20.0	mg/L
	Benz(E)Acephenenthrylene	3	2	2.33	mg/L
	Benzenamine, 4-Nitro-	3	1	2.2	mg/L
	Benzene, Nitro-	3	2	4.3	mg/L
	Benzene Hexachloride	3	1	2.3	mg/L
	Benzene, Ethyl-	3	2	3.4	mg/L
	Benzene, Methyl-	3	2	2.6	mg/L
	Benzo(Def)Phenanthrene	3	1	2.2	mg/L
	Bis(2-Chloroethoxy)Methane	3	2	2.8	mg/L
	Chloroform	3	2	3.9	mg/L
	Di-n-propyl Nitrosamine	3	0	3.3	mg/L
	Ethene, Trichloro	3	2	2.5	mg/L
	Ethene, Tetrachloro-	3	1	10.6	mg/L
	O-Chlorophenol	3	2	8.7	mg/L
	Residue, Non-flammable	3	0	27.2	mg/L
	Metals				
	Gold	3	1	0.04	mg/L
	Lead	3	2	0.13	mg/L
	Zinc	3	0	0.14	mg/L

16012: Treated effluent after hydrocarbon/aqueous phase separation and caustic neutralization.
16015: Treated effluent after equalization, caustic neutralization, and carbon adsorption.
QID: Questionnaire ID number
Obs: Number of observations
ND: Number of non-detects

Table 6-7: EPA Sampling Episode Pollutants Analyzed

POLLUTANT	CAS NUM
CLASSSICAL WET CHEMISTRY	
AMENABLE CYANIDE	C-025
AMMONIA NITROGEN	7664-41-7
BOD	C-002
CHLORIDE	16887-00-6
COD	C-004
FLUORIDE	16984-48-8
HEXANE EXTRACTABLE MATERIAL	C-036
HEXAVALENT CHROMIUM	18540-29-9
NITRATE/NITRITE	C-005
PH	C-006
RECOVERABLE OIL AND GREASE	C-007
TDS	C-010
TOC	C-012
TOTAL CYANIDE	57-12-5
TOTAL PHENOLS	C-020
TOTAL PHOSPHORUS	14265-44-2
TOTAL SOLIDS	C-008
TOTAL SULFIDE	18496-25-8
TSS	C-009
1613: DIOXINS/FURANS	
2378-TCDD	1746-01-6
2378-TCDF	51207-31-9
12378-PECDD	40321-76-4
12378-PECDF	57117-41-6
23478-PECDF	57117-31-4
123478-HXCDD	39227-28-6
123678-HXCDD	57653-85-7
123789-HXCDD	19408-74-3
123478-HXCDF	70648-26-9
123678-HXCDF	57117-44-9
123789-HXCDF	72918-21-9
234678-HXCDF	60851-34-5
1234678-HPCDD	35822-46-9
1234678-HPCDF	67562-39-4
1234789-HPCDF	55673-89-7
OCDD	3268-87-9
OCDF	39001-02-0
1657: PESTICIDES/HERBICIDES	
AZINPHOS ETHYL	2642-71-9
AZINPHOS METHYL	86-50-0
CHLORFEVINPHOS	470-90-6
CHLORPYRIFOS	2921-88-2
COUMAPHOS	56-72-4
CROTOXYPHOS	7700-17-6
DEF	78-48-8
DEMETON A	8065-48-3A
DEMETON B	8065-48-3B
DIAZINON	333-41-5
DICHLORFENTHION	97-17-6
DICHLORVOS	62-73-7
DICROTOPHOS	141-66-2
DIMETHOATE	60-51-5
DIOXATHION	78-34-2
DISULFOTON	298-04-4
EPN	2104-64-5
ETHION	563-12-2
ETHOPROP	13194-48-8
FAMPHUR	52-85-7
FENSULFOTHION	115-90-2
FENTHION	55-38-9
HEXAMETHYLPHOSPHORAMIDE	680-31-9
LEPTOPHOS	21609-90-5
MALATHION	121-75-5
MERPHOS	150-50-5

POLLUTANT	CAS NUM
1657: PESTICIDES/HERBICIDES	
METHAMIDOPHOS	10265-92-6
METHYL CHLORPYRIFOS	5598-13-0
METHYL PARATHION	298-00-0
METHYL TRITHION	953-17-3
MEVINPHOS	7786-34-7
MONOCROTOPHOS	6923-22-4
NALED	300-76-5
PARATHION (ETHYL)	56-38-2
PHORATE	298-02-2
PHOSMET	732-11-6
PHOSPHAMIDON E	297-99-4
PHOSPHAMIDON Z	23783-98-4
RONNEL	299-84-3
SULFOTEPP	3689-24-5
SULPROFOS	35400-43-2
TEPP	107-49-3
TERBUFOS	13071-79-9
TETRACHLORVINPHOS	22248-79-9
TOKUTHION	34643-46-4
TRICHLORFON	52-68-6
TRICHLORONATE	327-98-0
TRICRESYLPHOSPHATE	78-30-8
TRIMETHYLPHOSPHATE	512-56-1
1656: PESTICIDES/HERBICIDES	
ACEPHATE	30560-19-1
ACIFLUORFEN	50594-66-6
ALACHLOR	15972-60-8
ALDRIN	309-00-2
ATRAZINE	1912-24-9
BENFLURALIN	1861-40-1
ALPHA-BHC	319-84-6
BETA-BHC	319-85-7
GAMMA-BHC	58-89-9
DELTA-BHC	319-86-8
BROMACIL	314-40-9
BROMOXYNILOCTANOATE	1689-99-2
BUTACHLOR	23184-66-9
CAPTAFOL	2425-06-1
CAPTAN	133-06-2
CARBOPHENOTHION	786-19-6
ALPHA-CHLORDANE	5103-71-9
GAMMA-CHLORDANE	5103-74-2
CHLOROBENZILATE	510-15-6
CHLORONEB	2675-77-6
CHLOROPROPYLATE	5836-10-2
CHLOROTHALONIL	1897-45-6
DIBROMOCHLOROPROPANE	96-12-8
DACTHAL (DCPA)	1861-32-1
4,4'-DDD	72-54-8
4,4'-DDE	72-55-9
4,4'-DDT	50-29-3
DIALLATE A	2303-16-4A
DIALLATE B	2303-16-4B
DICHLONE	117-80-6
DICOFOL	115-32-2
DIELDRIN	60-57-1
ENDOSULFAN I	959-98-8
ENDOSULFAN II	33213-65-9
ENDOSULFAN SULFATE	1031-07-8
ENDRIN	72-20-8
ENDRIN ALDEHYDE	7421-93-4
ENDRIN KETONE	53494-70-5
ETHALFLURALIN	55283-68-6
ETRADIAZOLE	2593-15-9

Table 6-7: EPA Sampling Episode Pollutants Analyzed (continued)

POLLUTANT	CAS NUM
1656: PESTICIDES/HERBICIDES	
FENARIMOL	60168-88-9
HEPTACHLOR	76-44-8
HEPTACHLOR EPOXIDE	1024-57-3
ISODRIN	465-73-6
ISOPROPALIN	33820-53-0
KEPONE	143-50-0
METHOXYCHLOR	72-43-5
METRIBUZIN	21087-64-9
MIREX	2385-85-5
NITROFEN	1836-75-5
NORFLUORAZON	27314-13-2
PCB-1016	12674-11-2
PCB-1221	11104-28-2
PCB-1232	11141-16-5
PCB-1242	53469-21-9
PCB-1248	12672-29-6
PCB-1254	11097-69-1
PCB-1260	11096-82-5
PENTACHLORONITROBENZENE	82-68-8
PENDAMETHALIN	40487-42-1
CIS-PERMETHRIN	61949-76-6
TRANS-PERMETHRIN	61949-77-7
PERTHANE	72-56-0
PROPACHLOR	1918-16-7
PROPANIL	709-98-8
PROPAZINE	139-40-2
SIMAZINE	122-34-9
STROBANE	8001-50-1
TERBACIL	5902-51-2
TERBUTHYLAZINE	5915-41-3
TOXAPHENE	8001-35-2
TRIADIMEFON	43121-43-3
TRIFLURALIN	1582-09-8
1658: PESTICIDES/HERBICIDES	
DALAPON	75-99-0
DICAMBA	1918-00-9
DICHLOROPROP	120-36-5
DINOSEB	88-85-7
MCPA	94-74-6
MCPP	7085-19-0
PICLORAM	1918-02-1
2,4-D	94-75-7
2,4-DB	94-82-6
2,4,5-T	93-76-5
2,4,5-TP	93-72-1
1620: METALS	
ALUMINUM	7429-90-5
ANTIMONY	7440-36-0
ARSENIC	7440-38-2
BARIUM	7440-39-3
BERYLLIUM	7440-41-7
BISMUTH	7440-69-9
BORON	7440-42-8
CADMIUM	7440-43-9
CALCIUM	7440-70-2
CERIUM	7440-45-1
CHROMIUM	7440-47-3
COBALT	7440-48-4
COPPER	7440-50-8
DYSPROSIUM	7429-91-6
ERBIUM	7440-52-0
EUROPIUM	7440-53-1
GADOLINIUM	7440-54-2
GALLIUM	7440-55-3

POLLUTANT	CAS NUM
1620: METALS	
GERMANIUM	7440-56-4
GOLD	7440-57-5
HAFNIUM	7440-58-6
HOLMIUM	7440-60-0
INDIUM	7440-74-6
IODINE	7553-56-2
IRIDIUM	7439-88-5
IRON	7439-89-6
LANTHANUM	7439-91-0
LEAD	7439-92-1
LITHIUM	7439-93-2
LUTETIUM	7439-94-3
MAGNESIUM	7439-95-4
MANGANESE	7439-96-5
MERCURY	7439-97-6
MOLYBDENUM	7439-98-7
NEODYMIUM	7440-00-8
NICKEL	7440-02-0
NIOBIUM	7440-03-1
OSMIUM	7440-04-2
PALLADIUM	7440-05-3
PHOSPHORUS	7723-14-0
PLATINUM	7440-06-4
POTASSIUM	7440-09-7
PRASEODYMIUM	7440-10-0
RHENIUM	7440-15-5
RHODIUM	7440-16-6
RUTHENIUM	7440-18-8
SAMARIUM	7440-19-9
SCANDIUM	7440-20-2
SELENIUM	7782-49-2
SILICON	7440-21-3
SILVER	7440-22-4
SODIUM	7440-23-5
STRONTIUM	7440-24-6
SULFUR	7704-34-9
TANTALUM	7440-25-7
TELLURIUM	13494-80-9
TERBIUM	7440-27-9
THALLIUM	7440-28-0
THORIUM	7440-29-1
THULIUM	7440-30-4
TIN	7440-31-5
TITANIUM	7440-32-6
TUNGSTEN	7440-33-7
URANIUM	7440-61-1
VANADIUM	7440-62-2
YTTERBIUM	7440-64-4
YTTRIUM	7440-65-5
ZINC	7440-66-6
ZIRCONIUM	7440-67-7
1624: VOLATILE ORGANICS	
1,1-DICHLOROETHANE	75-34-3
1,1-DICHLOROETHENE	75-35-4
1,1,1-TRICHLOROETHANE	71-55-6
1,1,1,2-TETRACHLOROETHANE	630-20-6
1,1,2-TRICHLOROETHANE	79-00-5
1,1,2,2-TETRACHLOROETHANE	79-34-5
1,2-DIBROMOETHANE	106-93-4
1,2-DICHLOROETHANE	107-06-2
1,2-DICHLOROPROPANE	78-87-5
1,2,3-TRICHLOROPROPANE	96-18-4
1,3-DICHLOROPROPANE	142-28-9
1,4-DIOXANE	123-91-1

Table 6-7: EPA Sampling Episode Pollutants Analyzed (continued)

POLLUTANT	CAS NUM
1624: VOLATILE ORGANICS	
2-BUTANONE (MEK)	78-93-3
2-CHLORO-1,3-BUTADIENE	126-99-8
2-CHLOROETHYLVINYL ETHER	110-75-8
2-HEXANONE	591-78-6
2-METHYL-2-PROPENENITRILE	126-98-7
2-PROPANONE (ACETONE)	67-64-1
2-PROPENAL (ACROLEIN)	107-02-8
2-PROPEN-1-OL (ALLYL ALCOHOL)	107-18-6
3-CHLOROPROPENE	107-05-1
4-METHYL-2-PENTANONE	108-10-1
ACRYLONITRILE	107-13-1
BENZENE	71-43-2
BROMODICHLOROMETHANE	75-27-4
BROMOFORM	75-25-2
BROMOMETHANE	74-83-9
CARBON DISULFIDE	75-15-0
CHLOROACETONITRILE	107-14-2
CHLOROBENZENE	108-90-7
CHLOROETHANE	75-00-3
CHLOROFORM	67-66-3
CHLOROMETHANE	74-87-3
CIS-1,3-DICHLOROPROPENE	10061-01-5
CROTONALDEHYDE	4170-30-3
DIBROMOCHLOROMETHANE	124-48-1
DIBROMOMETHANE	74-95-3
DIETHYL ETHER	60-29-7
ETHYL BENZENE	100-41-4
ETHYL CYANIDE	107-12-0
ETHYL METHACRYLATE	97-63-2
IODOMETHANE	74-88-4
ISOBUTYL ALCOHOL	78-83-1
METHYLENE CHLORIDE	75-09-2
M-XYLENE	108-38-3
O+P XYLENE	136777-61-2
TETRACHLOROETHENE	127-18-4
TETRACHLOROMETHANE	56-23-5
TOLUENE	108-88-3
TRANS-1,2-DICHLOROETHENE	156-60-5
TRANS-1,3-DICHLOROPROPENE	10061-02-6
TRANS-1,4-DICHLORO-2-BUTENE	110-57-6
TRICHLOROETHENE	79-01-6
TRICHLOROFLUOROMETHANE	75-69-4
VINYL ACETATE	108-05-4
VINYL CHLORIDE	75-01-4
1625: SEMIVOLATILE ORGANICS	
1-METHYLFLUORENE	1730-37-6
1-METHYLPHENANTHRENE	832-69-9
1-PHENYLNAPHTHALENE	605-02-7
1,2-DIBROMO-3-CHLOROPROPANE	96-12-8
1,2-DICHLOROBENZENE	95-50-1
1,2-DIPHENYLHYDRAZINE	122-66-7
1,2,3-TRICHLOROBENZENE	87-61-6
1,2,3-TRIMETHOXYBENZENE	634-36-6
1,2,4-TRICHLOROBENZENE	120-82-1
1,2,4,5-TETRACHLOROBENZENE	95-94-3
1,2:3,4-DIEPOXYBUTANE	1464-53-5
1,3-BENZENEDIOL (RESORCINOL)	108-46-3
1,3-DICHLORO-2-PROPANOL	96-23-1
1,3-DICHLOROBENZENE	541-73-1
1,3,5-TRITHIANE	291-21-4
1,4-DICHLOROBENZENE	106-46-7
1,4-DINITROBENZENE	100-25-4
1,4-NAPHTHOQUINONE	130-15-4
1,5-NAPHTHALENEDIAMINE	2243-62-1

POLLUTANT	CAS NUM
1625: SEMIVOLATILE ORGANICS	
2-BROMOCHLOROBENZENE	694-80-4
2-CHLORONAPHTHALENE	91-58-7
2-CHLOROPHENOL	95-57-8
2-ISOPROPYLNAPHTHALENE	2027-17-0
2-METHYL-4,6-DINITROPHENOL	534-52-1
2-METHYLBENZOTHIOAZOLE	120-75-2
2-METHYLNAPHTHALENE	91-57-6
2-NITROANILINE	88-74-4
2-NITROPHENOL	88-75-5
2-PHENYLNAPHTHALENE	612-94-2
2-PICOLINE	109-06-8
2-(METHYLTHIO)BENZOTHIAZOLE	615-22-5
2,3-BENZOFLUORENE	243-17-4
2,3-DICHLOROANILINE	608-27-5
2,3-DICHLORONITROBENZENE	3209-22-1
2,3,4,6-TETRACHLOROPHENOL	58-90-2
2,3,6-TRICHLOROPHENOL	933-75-5
2,4-DIAMINOTOLUENE	95-80-7
2,4-DICHLOROPHENOL	120-83-2
2,4-DIMETHYLPHENOL	105-67-9
2,4-DINITROPHENOL	51-28-5
2,4-DINITROTOLUENE	121-14-2
2,4,5-TRICHLOROPHENOL	95-95-4
2,4,5-TRIMETHYLANILINE	137-17-7
2,4,6-TRICHLOROPHENOL	88-06-2
2,6-DICHLORO-4-NITROANILINE	99-30-9
2,6-DICHLOROPHENOL	87-65-0
2,6-DINITROTOLUENE	606-20-2
2,6-DI-TERT-BUTYL-P-BENZOQUINONE	719-22-2
3-BROMOCHLOROBENZENE	108-37-2
3-CHLORONITROBENZENE	121-73-3
3-METHYLCHOLANTHRENE	56-49-5
3-NITROANILINE	99-09-2
3,3-DICHLOROBENZIDINE	91-94-1
3,3'-DIMETHOXYBENZIDINE	119-90-4
3,5-DIBROMO-4-HYDROXYBENZONITRILE	1689-84-5
3,6-DIMETHYLPHENANTHRENE	1576-67-6
4-AMINOBIPHENYL	92-67-1
4-BROMOPHENYLPHENYLETHER	101-55-3
4-CHLORO-2-NITROANILINE	89-63-4
4-CHLORO-3-METHYLPHENOL	59-50-7
4-CHLOROANILINE	106-47-8
4-CHLOROPHENYLPHENYLETHER	7005-72-3
4-NITROANILINE	100-01-6
4-NITROBIPHENYL	92-93-3
4-NITROPHENOL	100-02-7
4,4-METHYLENE-BIS(2-CHLOROANILINE)	101-14-4
4,5-METHYLENE-PHENANTHRENE	203-64-5
5-CHLORO-O-TOLUIDINE	95-79-4
5-NITRO-O-TOLUIDINE	99-55-8
7,12-DIMETHYLBENZ(A)ANTHRACENE	57-97-6
ACENAPHTHENE	83-32-9
ACENAPHTHYLENE	208-96-8
ACETOPHENONE	98-86-2
ALPHA-NAPHTHYLAMINE	134-32-7
ALPHA-TERPINEOL	98-55-5
ANILINE	62-53-3
ANTHRACENE	120-12-7
ARAMITE	140-57-8
BENZANTHRONE	82-05-3
BENZENETHIOL	108-98-5
BENZIDINE	92-87-5
BENZOIC ACID	65-85-0
BENZO(A)ANTHRACENE	56-55-3

Table 6-7: EPA Sampling Episode Pollutants Analyzed (continued)

POLLUTANT	CAS NUM	POLLUTANT	CAS NUM
1625: SEMIVOLATILE ORGANICS		1625: SEMIVOLATILE ORGANICS	
BENZO(A)PYRENE	50-32-8	N-NITROSOMORPHOLINE	59-89-2
BENZO(B)FLUORANTHENE	205-99-2	N-NITROSOPIPERIDINE	100-75-4
BENZO(GHI)PERYLENE	191-24-2	N,N-DIMETHYLFORMAMIDE	68-12-2
BENZO(K)FLUORANTHENE	207-08-9	O-ANISIDINE	90-04-0
BENZYL ALCOHOL	100-51-6	O-CRESOL	95-48-7
BETA-NAPHTHYLAMINE	91-59-8	O-TOLUIDINE	95-53-4
BIPHENYL	92-52-4	P-CRESOL	106-44-5
BIS(2-CHLOROETHOXY)METHANE	111-91-1	P-CYMENE	99-87-6
BIS(2-CHLOROETHYL)ETHER	111-44-4	P-DIMETHYLAMINO-AZOBENZENE	60-11-7
BIS(2-CHLOROISOPROPYL)ETHER	108-60-1	PENTACHLOROBENZENE	608-93-5
BIS(2-ETHYLHEXYL)PHTHALATE	117-81-7	PENTACHLOROETHANE	76-01-7
BUTYLBENZYLPHTHALATE	85-68-7	PENTACHLOROPHENOL	87-86-5
CARBAZOLE	86-74-8	PENTAMETHYLBENZENE	700-12-9
CHRYSENE	218-01-9	PERYLENE	198-55-0
CROTOXYPHOS	7700-17-6	PHENACETIN	62-44-2
DIBENZOFURAN	132-64-9	PHENANTHRENE	85-01-8
DIBENZOTHIOPHENE	132-65-0	PHENOL	108-95-2
DIBENZO(A,H)ANTHRACENE	53-70-3	PHENOTHIAZINE	92-84-2
DIETHYLPHTHALATE	84-66-2	PRONAMIDE	23950-58-5
DIMETHYLPHTHALATE	131-11-3	PYRENE	129-00-0
DIMETHYLSULFONE	67-71-0	PYRIDINE	110-86-1
DI-N-BUTYLPHTHALATE	84-74-2	SAFROLE	94-59-7
DI-N-OCTYLPHTHALATE	117-84-0	SQUALENE	7683-64-9
DIPHENYLETHER	101-84-8	STYRENE	100-42-5
DIPHENYLAMINE	122-39-4	THIANAPHTHENE (2,3-BENZOTHIOPHENE)	95-15-8
DIPHENYLDISULFIDE	882-33-7	THIOACETAMIDE	62-55-5
ETHYLMETHANESULFONATE	62-50-0	THIOXANTHONE	492-22-8
ETHYLENETHIOUREA	96-45-7	TRIPHENYLENE	217-59-4
ETHYNYLESTRADIOL-3-METHYLETHER	72-33-3	TRIPROPYLENEGLYCOLMETHYLETHER	20324-33-8
FLUORANTHENE	206-44-0		
FLUORENE	86-73-7		
HEXACHLOROBENZENE	118-74-1		
HEXACHLOROBUTADIENE	87-68-3		
HEXACHLOROCYCLOPENTADIENE	77-47-4		
HEXACHLOROETHANE	67-72-1		
HEXACHLOROPROPENE	1888-71-7		
HEXANOIC ACID	142-62-1		
INDENO(1,2,3-CD)PYRENE	193-39-5		
ISOPHORONE	78-59-1		
ISOSAFROLE	120-58-1		
LONGIFOLENE	475-20-7		
MALACHITE GREEN	569-64-2		
METHAPYRILENE	91-80-5		
METHYLMETHANESULFONATE	66-27-3		
NAPHTHALENE	91-20-3		
N-C10 (N-DECANE)	124-18-5		
N-C12 (N-DODECANE)	112-40-3		
N-C14 (N-TETRADECANE)	629-59-4		
N-C16 (N-HEXADECANE)	544-76-3		
N-C18 (N-OCTADECANE)	593-45-3		
N-C20 (N-EICOSANE)	112-95-8		
N-C22 (N-DOCOSANE)	629-97-0		
N-C24 (N-TETRACOSANE)	646-31-1		
N-C26 (N-HEXACOSANE)	630-01-3		
N-C28 (N-OCTACOSANE)	630-02-4		
N-C30 (N-TRIACONTANE)	638-68-6		
NITROBENZENE	98-95-3		
N-NITROSODIETHYLAMINE	55-18-5		
N-NITROSODIMETHYLAMINE	62-75-9		
N-NITROSODI-N-BUTYLAMINE	924-16-3		
N-NITROSODI-N-PROPYLAMINE	621-64-7		
N-NITROSODIPHENYLAMINE	86-30-6		
N-NITROSOMETHYL-ETHYLAMINE	10595-95-6		
N-NITROSOMETHYL-PHENYLAMINE	614-00-6		

Table 6-8: EPA Sampling Episode List of Analytes Never Detected

		Non-Hazardous Subcategory														Hazardous Subcategory					
		Subtitle D Municipal					**Subtitle D Non-Municipal**														
POLLUTANT	**CAS NUM**	E4491	E4626	E4667	E4687	E4738	E4503	E4630	E4631	E4638	E4639	E4644	E4683	E4690	E4721	E4631	E4659	E4682	E4690	E4721	E4759
1613: DIOXINS/FURANS																					
2378-TCDD	1746-01-6	ND	-	-	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
2378-TCDF	51207-31-9	ND	-	-	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND		ND	ND		ND	ND
12378-PECDD	40321-76-4	ND	-	-	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
12378-PECDF	57117-41-6	ND	-	-	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND		ND	ND		ND	ND
23478-PECDF	57117-31-4	ND	-	-	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND	ND	ND
123478-HXCDD	39227-28-6	ND	-	-	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND	ND	ND
123678-HXCDD	57653-85-7	ND	-	-	ND		ND	ND	ND	ND	ND	ND	ND		ND		ND	ND	ND	ND	ND
123789-HXCDD	19408-74-3	ND	-	-	ND		ND	ND	ND	ND	ND	ND	ND		ND		ND	ND	ND	ND	ND
123478-HXCDF	70648-26-9	ND	-	-	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND		ND	ND		ND	ND
123678-HXCDF	57117-44-9	ND	-	-	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND		ND	ND		ND	ND
123789-HXCDF	72918-21-9	ND	-	-	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND	ND	ND
234678-HXCDF	60851-34-5	ND	-	-	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND		ND	ND
1234678-HPCDD	35822-46-9		-	-	ND		ND		ND	ND	ND	ND	ND		ND		ND	ND			ND
1234678-HPCDF	67562-39-4	ND	-	-	ND		ND	ND	ND	ND	ND	ND	ND		ND		ND	ND			ND
1234789-HPCDF	55673-89-7	ND	-	-	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND		ND	ND		ND	ND
OCDD	3268-87-9		-	-	ND		ND		ND	ND	ND	ND	ND				ND	ND			ND
OCDF	39001-02-0	ND	-	-	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND		ND	ND			ND
1620: METALS																					
ALUMINUM	7429-90-5									ND	ND	ND	ND				ND				
ANTIMONY	7440-36-0				ND	ND	ND	ND	ND			ND		ND		ND	ND				
ARSENIC	7440-38-2								ND		ND		ND								
BARIUM	7440-39-3																				
BERYLLIUM	7440-41-7	ND	ND	ND	ND		ND	ND	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
BISMUTH	7440-69-9	ND		ND	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
BORON	7440-42-8			ND																	
CADMIUM	7440-43-9						ND	ND	ND	ND			ND		ND	ND	ND		ND	ND	
CALCIUM	7440-70-2																				
CERIUM	7440-45-1	ND		ND	ND		ND	ND	ND	ND	ND		ND	ND	ND	ND		ND	ND	ND	ND
CHROMIUM	7440-47-3				ND		ND	ND	ND		ND	ND	ND				ND		ND		
COBALT	7440-48-4		ND		ND				ND	ND	ND	ND	ND		ND						
COPPER	7440-50-8				ND		ND					ND			ND	ND	ND				
DYSPROSIUM	7429-91-6	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
ERBIUM	7440-52-0	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	
EUROPIUM	7440-53-1	ND			ND	ND	ND	ND	ND				ND	ND	ND	ND	ND	ND	ND	ND	
GADOLINIUM	7440-54-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND
GALLIUM	7440-55-3	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
GERMANIUM	7440-56-4	ND	ND		ND	ND	ND	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
GOLD	7440-57-5	ND		ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
HAFNIUM	7440-58-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND

Table 6-8: EPA Sampling Episode List of Analytes Never Detected (continued)

		Non-Hazardous Subcategory														Hazardous Subcategory					
		Subtitle D Municipal					Subtitle D Non-Municipal														
POLLUTANT	**CAS NUM**	E4491	E4626	E4667	E4687	E4738	E4503	E4630	E4631	E4638	E4639	E4644	E4683	E4690	E4721	E4631	E4659	E4682	E4690	E4721	E4759
HOLMIUM	7440-60-0	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
INDIUM	7440-74-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND
IODINE	7553-56-2	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND
IRIDIUM	7439-88-5	ND	ND		ND	ND	ND	ND	ND				ND	ND	ND	ND	ND		ND	ND	ND
IRON	7439-89-6																				
LANTHANUM	7439-91-0	ND	ND	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	
LEAD	7439-92-1	ND	ND				ND	ND	ND			ND	ND		ND	ND	ND		ND	ND	
LITHIUM	7439-93-2	ND			ND	ND		ND		ND	ND		ND								
LUTETIUM	7439-94-3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND
MAGNESIUM	7439-95-4																				
MANGANESE	7439-96-5																				
MERCURY	7439-97-6				ND		ND	ND	ND	ND	ND		ND	ND	ND	ND	ND	ND	ND		
MOLYBDENUM	7439-98-7				ND	ND	ND	ND		ND			ND		ND	ND					
NEODYMIUM	7440-00-8	ND	ND	ND	ND		ND	ND	ND		ND		ND	ND	ND	ND	ND	ND	ND	ND	
NICKEL	7440-02-0									ND	ND	ND	ND								
NIOBIUM	7440-03-1	ND			ND	ND	ND	ND	ND				ND	ND	ND	ND		ND	ND	ND	ND
OSMIUM	7440-04-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PALLADIUM	7440-05-3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	
PHOSPHORUS	7723-14-0	ND			ND						ND	ND	ND				ND				
PLATINUM	7440-06-4	ND	ND		ND	ND	ND	ND	ND	ND			ND	ND	ND	ND	ND	ND	ND	ND	ND
POTASSIUM	7440-09-7	ND										ND									
PRASEODYMIUM	7440-10-0	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	
RHENIUM	7440-15-5	ND	ND		ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND		ND	ND	ND	
RHODIUM	7440-16-6	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	
RUTHENIUM	7440-18-8	ND	ND		ND	ND	ND	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	
SAMARIUM	7440-19-9	ND	ND	ND	ND	ND	ND	ND	ND			ND	ND	ND	ND	ND	ND	ND	ND	ND	
SCANDIUM	7440-20-2	ND			ND	ND	ND	ND	ND				ND	ND	ND	ND	ND	ND	ND	ND	
SELENIUM	7782-49-2	ND			ND	ND	ND	ND			ND	ND	ND	ND	ND		ND				
SILICON	7440-21-3	ND																			
SILVER	7440-22-4	ND	ND		ND	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
SODIUM	7440-23-5																				
STRONTIUM	7440-24-6	ND																			
SULFUR	7704-34-9	ND										ND									
TANTALUM	7440-25-7	ND			ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND		ND	ND	ND	
TELLURIUM	13494-80-9	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
TERBIUM	7440-27-9	ND	ND		ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND
THALLIUM	7440-28-0	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND			ND
THORIUM	7440-29-1	ND	ND		ND	ND	ND	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
THULIUM	7440-30-4	ND	ND	ND	ND	ND	ND	ND	ND		ND		ND	ND	ND	ND	ND	ND	ND	ND	ND
TIN	7440-31-5	ND			ND	ND	ND	ND	ND	ND	ND	ND	ND		ND		ND			ND	

Table 6-8: EPA Sampling Episode List of Analytes Never Detected (continued)

		Non-Hazardous Subcategory														Hazardous Subcategory					
		Subtitle D Municipal					Subtitle D Non-Municipal														
POLLUTANT	**CAS NUM**	E4491	E4626	E4667	E4687	E4738	E4503	E4630	E4631	E4638	E4639	E4644	E4683	E4690	E4721	E4631	E4659	E4682	E4690	E4721	E4759
TITANIUM	7440-32-6				ND						ND	ND	ND				ND		ND		
TUNGSTEN	7440-33-7	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	
URANIUM	7440-61-1	ND	ND		ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND
VANADIUM	7440-62-2				ND		ND		ND	ND	ND	ND	ND				ND				
YTTERBIUM	7440-64-4	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND
YTTRIUM	7440-65-5			ND			ND		ND	ND	ND	ND	ND		ND	ND	ND	ND	ND	ND	ND
ZINC	7440-66-6											ND									
ZIRCONIUM	7440-67-7	ND	ND		ND	ND	ND	ND	ND	ND	ND		ND	ND	ND		ND	ND	ND	ND	
1624: VOLATILE ORGANICS																					
1,1-DICHLOROETHANE	75-34-3			ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND			ND				
1,1-DICHLOROETHENE	75-35-4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND
1,1,1-TRICHLOROETHANE	71-55-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND			ND
1,1,1,2-TETRACHLOROETHANE	630-20-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND
1,1,2-TRICHLOROETHANE	79-00-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND
1,1,2,2-TETRACHLOROETHANE	79-34-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND
1,2-DIBROMOETHANE	106-93-4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
1,2-DICHLOROETHANE	107-06-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND				
1,2-DICHLOROPROPANE	78-87-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND		ND	ND		ND	ND
1,2,3-TRICHLOROPROPANE	96-18-4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND
1,3-DICHLOROPROPANE	142-28-9	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
1,4-DIOXANE	123-91-1	ND				ND	ND	ND	ND		ND	ND	ND		ND		ND			ND	
2-BUTANONE (MEK)	78-93-3						ND	ND	ND	ND	ND	ND	ND				ND				
2-CHLORO-1,3-BUTADIENE	126-99-8	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND
2-CHLOROETHYLVINYL ETHER	110-75-8	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
2-HEXANONE	591-78-6	ND			ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND				ND
2-METHYL-2-PROPENENITRILE	126-98-7	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND
2-PROPANONE (ACETONE)	67-64-1							ND		ND	ND	ND	ND								
2-PROPENAL (ACROLEIN)	107-02-8	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
2-PROPEN-1-OL (ALLYL ALCOHOL)	107-18-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND		ND	ND
3-CHLOROPROPENE	107-05-1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4-METHYL-2-PENTANONE	108-10-1	ND				ND	ND	ND	ND	ND	ND	ND	ND	ND			ND				
ACRYLONITRILE	107-13-1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
BENZENE	71-43-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND			ND				
BROMODICHLOROMETHANE	75-27-4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
BROMOFORM	75-25-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
BROMOMETHANE	74-83-9	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
CARBON DISULFIDE	75-15-0	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND		ND	
CHLOROACETONITRILE	107-14-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND
CHLOROBENZENE	108-90-7	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND			ND
CHLOROETHANE	75-00-3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND			ND	ND		ND	ND

Table 6-8: EPA Sampling Episode List of Analytes Never Detected (continued)

		Non-Hazardous Subcategory													Hazardous Subcategory					
		Subtitle D Municipal					Subtitle D Non-Municipal													
POLLUTANT	**CAS NUM**	E4491	E4626	E4667	E4687	E4738	E4503	E4630	E4631	E4638	E4639	E4644	E4683	E4690	E4721	E4631	E4659	E4682	E4690	E4721	E4759
CHLOROFORM	67-66-3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND
CHLOROMETHANE	74-87-3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
CIS-1,3-DICHLOROPROPENE	10061-01-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND
CROTONALDEHYDE	4170-30-3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
DIBROMOCHLOROMETHANE	124-48-1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
DIBROMOMETHANE	74-95-3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
DIETHYL ETHER	60-29-7	ND		ND		ND	ND	ND	ND	ND	ND	ND	ND	ND			ND		ND		ND
ETHYL BENZENE	100-41-4						ND	ND	ND	ND	ND	ND	ND	ND			ND				
ETHYL CYANIDE	107-12-0	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
ETHYL METHACRYLATE	97-63-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
IODOMETHANE	74-88-4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
ISOBUTYL ALCOHOL	78-83-1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND			ND	
METHYLENE CHLORIDE	75-09-2				ND		ND	ND	ND	ND	ND	ND	ND		ND		ND				
M-XYLENE	108-38-3				ND		ND	ND	ND	ND	ND	ND	ND	ND			ND		ND		
O+P XYLENE	136777-61-2				ND	ND	ND	ND	ND	ND	ND	ND	ND	ND			ND		ND		ND
TETRACHLOROETHENE	127-18-4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND				ND
TETRACHLOROMETHANE	56-23-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND			ND
TOLUENE	108-88-3								ND	ND	ND	ND	ND	ND			ND				
TRANS-1,2-DICHLOROETHENE	156-60-5	ND			ND	ND	ND	ND	ND	ND	ND	ND	ND	ND			ND				ND
TRANS-1,3-DICHLOROPROPENE	10061-02-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
TRANS-1,4-DICHLORO-2-BUTENE	110-57-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
TRICHLOROETHENE	79-01-6	ND	ND		ND	ND	ND	ND	ND		ND	ND	ND	ND	ND		ND				ND
TRICHLOROFLUOROMETHANE	75-69-4	ND	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
VINYL ACETATE	108-05-4	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
VINYL CHLORIDE	75-01-4	ND				ND	ND	ND	ND	ND	ND	ND	ND	ND			ND				ND
1625: SEMIVOLATILE ORGANICS																					
1-METHYLFLUORENE	1730-37-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
1-METHYLPHENANTHRENE	832-69-9	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
1-PHENYLNAPHTHALENE	605-02-7	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
1,2-DIBROMO-3-CHLOROPROPANE	96-12-8	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
1,2-DICHLOROBENZENE	95-50-1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND		ND
1,2-DIPHENYLHYDRAZINE	122-66-7	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		
1,2,3-TRICHLOROBENZENE	87-61-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
1,2,3-TRIMETHOXYBENZENE	634-36-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
1,2,4-TRICHLOROBENZENE	120-82-1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
1,2,4,5-TETRACHLOROBENZENE	95-94-3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
1,2:3,4-DIEPOXYBUTANE	1464-53-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
1,3-BENZENEDIOL (RESORCINOL)	108-46-3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
1,3-DICHLORO-2-PROPANOL	96-23-1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
1,3-DICHLOROBENZENE	541-73-1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND

Table 6-8: EPA Sampling Episode List of Analytes Never Detected (continued)

		Non-Hazardous Subcategory														Hazardous Subcategory					
		Subtitle D Municipal					Subtitle D Non-Municipal														
POLLUTANT	**CAS NUM**	E4491	E4626	E4667	E4687	E4738	E4503	E4630	E4631	E4638	E4639	E4644	E4683	E4690	E4721	E4631	E4659	E4682	E4690	E4721	E4759
1,3,5-TRITHIANE	291-21-4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND		ND
1,4-DICHLOROBENZENE	106-46-7		ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND		ND
1,4-DINITROBENZENE	100-25-4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
1,4-NAPHTHOQUINONE	130-15-4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
1,5-NAPHTHALENEDIAMINE	2243-62-1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
2-BROMOCHLOROBENZENE	694-80-4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
2-CHLORONAPHTHALENE	91-58-7	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
2-CHLOROPHENOL	95-57-8	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
2-ISOPROPYLNAPHTHALENE	2027-17-0	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
2-METHYL-4,6-DINITROPHENOL	534-52-1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND
2-METHYLBENZOTHIOAZOLE	120-75-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
2-METHYLNAPHTHALENE	91-57-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND
2-NITROANILINE	88-74-4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
2-NITROPHENOL	88-75-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND
2-PHENYLNAPHTHALENE	612-94-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
2-PICOLINE	109-06-8			ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		
2-(METHYLTHIO)BENZOTHIAZOLE	615-22-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND
2,3-BENZOFLUORENE	243-17-4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
2,3-DICHLOROANILINE	608-27-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
2,3-DICHLORONITROBENZENE	3209-22-1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
2,3,4,6-TETRACHLOROPHENOL	58-90-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
2,3,6-TRICHLOROPHENOL	933-75-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
2,4-DIAMINOTOLUENE	95-80-7	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
2,4-DICHLOROPHENOL	120-83-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
2,4-DIMETHYLPHENOL	105-67-9	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND			ND		
2,4-DINITROPHENOL	51-28-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND
2,4-DINITROTOLUENE	121-14-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
2,4,5-TRICHLOROPHENOL	95-95-4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
2,4,5-TRIMETHYLANILINE	137-17-7	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
2,4,6-TRICHLOROPHENOL	88-06-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
2,6-DICHLORO-4-NITROANILINE	99-30-9	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
2,6-DICHLOROPHENOL	87-65-0	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
2,6-DINITROTOLUENE	606-20-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND
2,6-DI-TERT-BUTYL-P-BENZOQUINONE	719-22-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
3-BROMOCHLOROBENZENE	108-37-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
3-CHLORONITROBENZENE	121-73-3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
3-METHYLCHOLANTHRENE	56-49-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
3-NITROANILINE	99-09-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
3,3-DICHLOROBENZIDINE	91-94-1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
3,3'-DIMETHOXYBENZIDINE	119-90-4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND

Table 6-8: EPA Sampling Episode List of Analytes Never Detected (continued)

		Non-Hazardous Subcategory														Hazardous Subcategory					
		Subtitle D Municipal					Subtitle D Non-Municipal														
POLLUTANT	**CAS NUM**	E4491	E4626	E4667	E4687	E4738	E4503	E4630	E4631	E4638	E4639	E4644	E4683	E4690	E4721	E4631	E4659	E4682	E4690	E4721	E4759
3,5-DIBROMO-4-HYDROXYBENZONITRILE	1689-84-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
3,6-DIMETHYLPHENANTHRENE	1576-67-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4-AMINOBIPHENYL	92-67-1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4-BROMOPHENYL PHENYL ETHER	101-55-3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4-CHLORO-2-NITROANILINE	89-63-4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4-CHLORO-3-METHYLPHENOL	59-50-7	ND	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND
4-CHLOROANILINE	106-47-8	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4-CHLOROPHENYL PHENYL ETHER	7005-72-3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4-NITROANILINE	100-01-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4-NITROBIPHENYL	92-93-3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4-NITROPHENOL	100-02-7	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND
4,4-METHYLENE-BIS(2-CHLOROANILINE)	101-14-4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4,5-METHYLENE-PHENANTHRENE	203-64-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
5-CHLORO-O-TOLUIDINE	95-79-4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
5-NITRO-O-TOLUIDINE	99-55-8	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
7,12-DIMETHYLBENZ(A)ANTHRACENE	57-97-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
ACENAPHTHENE	83-32-9	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND		ND
ACENAPHTHYLENE	208-96-8	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
ACETOPHENONE	98-86-2	ND		ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND		ND	ND
ALPHA-NAPHTHYLAMINE	134-32-7	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND	ND
ALPHA-TERPINEOL	98-55-5					ND	ND	ND	ND	ND	ND	ND	ND	ND				ND	ND		ND
ANILINE	62-53-3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND			ND		ND		
ANTHRACENE	120-12-7	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
ARAMITE	140-57-8	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
BENZANTHRONE	82-05-3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
BENZENETHIOL	108-98-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
BENZIDINE	92-87-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
BENZOIC ACID	65-85-0	ND				ND	ND	ND	ND	ND	ND	ND	ND		ND						
BENZO(A)ANTHRACENE	56-55-3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
BENZO(A)PYRENE	50-32-8	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
BENZO(B)FLUORANTHENE	205-99-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
BENZO(GHI)PERYLENE	191-24-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
BENZO(K)FLUORANTHENE	207-08-9	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
BENZYL ALCOHOL	100-51-6	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND		
BETA-NAPHTHYLAMINE	91-59-8		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND	ND
BIPHENYL	92-52-4	ND	ND	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
BIS(2-CHLOROETHOXY) METHANE	111-91-1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
BIS(2-CHLOROETHYL) ETHER	111-44-4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND
BIS(2-CHLOROISOPROPYL) ETHER	108-60-1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND
BIS(2-ETHYLHEXYL) PHTHALATE	117-81-7	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND

Table 6-8: EPA Sampling Episode List of Analytes Never Detected (continued)

		Non-Hazardous Subcategory														Hazardous Subcategory					
		Subtitle D Municipal					Subtitle D Non-Municipal														
POLLUTANT	**CAS NUM**	E4491	E4626	E4667	E4687	E4738	E4503	E4630	E4631	E4638	E4639	E4644	E4683	E4690	E4721	E4631	E4659	E4682	E4690	E4721	E4759
BUTYL BENZYL PHTHALATE	85-68-7	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
CARBAZOLE	86-74-8	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
CHRYSENE	218-01-9	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
CROTOXYPHOS	7700-17-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
DIBENZOFURAN	132-64-9	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND
DIBENZOTHIOPHENE	132-65-0	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
DIBENZO(A,H)ANTHRACENE	53-70-3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
DIETHYL PHTHALATE	84-66-2	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
DIMETHYL PHTHALATE	131-11-3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
DIMETHYL SULFONE	67-71-0	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
DI-N-BUTYL PHTHALATE	84-74-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
DI-N-OCTYL PHTHALATE	117-84-0	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
DIPHENYL ETHER	101-84-8	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND
DIPHENYLAMINE	122-39-4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
DIPHENYLDISULFIDE	882-33-7	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
ETHYL METHANESULFONATE	62-50-0	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
ETHYLENETHIOUREA	96-45-7	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
ETHYNYLESTRADIOL-3-METHYL ETHER	72-33-3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
FLUORANTHENE	206-44-0	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND
FLUORENE	86-73-7	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND		ND
HEXACHLOROBENZENE	118-74-1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
HEXACHLOROBUTADIENE	87-68-3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
HEXACHLOROCYCLOPENTADIENE	77-47-4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
HEXACHLOROETHANE	67-72-1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
HEXACHLOROPROPENE	1888-71-7	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
HEXANOIC ACID	142-62-1	ND					ND		ND	ND	ND	ND	ND		ND				ND		
INDENO(1,2,3-CD)PYRENE	193-39-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
ISOPHORONE	78-59-1			ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
ISOSAFROLE	120-58-1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
LONGIFOLENE	475-20-7	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
MALACHITE GREEN	569-64-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
METHAPYRILENE	91-80-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
METHYL METHANESULFONATE	66-27-3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
NAPHTHALENE	91-20-3		ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND			ND		ND		
N-C10 (N-DECANE)	124-18-5		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND
N-C12 (N-DODECANE)	112-40-3	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND
N-C14 (N-TETRADECANE)	629-59-4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND		ND
N-C16 (N-HEXADECANE)	544-76-3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND
N-C18 (N-OCTADECANE)	593-45-3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND
N-C20 (N-EICOSANE)	112-95-8	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND

Table 6-8: EPA Sampling Episode List of Analytes Never Detected (continued)

		Non-Hazardous Subcategory														Hazardous Subcategory					
		Subtitle D Municipal					Subtitle D Non-Municipal														
POLLUTANT	**CAS NUM**	E4491	E4626	E4667	E4687	E4738	E4503	E4630	E4631	E4638	E4639	E4644	E4683	E4690	E4721	E4631	E4659	E4682	E4690	E4721	E4759
N-C22 (N-DOCOSANE)	629-97-0	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND
N-C24 (N-TETRACOSANE)	646-31-1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND
N-C26 (N-HEXACOSANE)	630-01-3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND
N-C28 (N-OCTACOSANE)	630-02-4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND
N-C30 (N-TRIACONTANE)	638-68-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
NITROBENZENE	98-95-3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
N-NITROSODIETHYLAMINE	55-18-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
N-NITROSODIMETHYLAMINE	62-75-9	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
N-NITROSODI-N-BUTYLAMINE	924-16-3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
N-NITROSODI-N-PROPYLAMINE	621-64-7	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
N-NITROSODIPHENYLAMINE	86-30-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
N-NITROSOMETHYL -ETHYLAMINE	10595-95-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
N-NITROSOMETHYL-PHENYLAMINE	614-00-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
N-NITROSOMORPHOLINE	59-89-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
N-NITROSOPIPERIDINE	100-75-4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
N,N-DIMETHYLFORMAMIDE	68-12-2	ND			ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND
O-ANISIDINE	90-04-0	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
O-CRESOL	95-48-7	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND			ND			
O-TOLUIDINE	95-53-4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
P-CRESOL	106-44-5	ND		ND			ND	ND	ND	ND	ND	ND	ND				ND		ND		
P-CYMENE	99-87-6	ND			ND	ND		ND	ND	ND	ND	ND	ND	ND			ND	ND	ND		ND
P-DIMETHYLAMINO-AZOBENZENE	60-11-7	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PENTACHLOROBENZENE	608-93-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PENTACHLOROETHANE	76-01-7	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PENTACHLOROPHENOL	87-86-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PENTAMETHYLBENZENE	700-12-9	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PERYLENE	198-55-0	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PHENACETIN	62-44-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PHENANTHRENE	85-01-8	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND		ND
PHENOL	108-95-2					ND	ND	ND		ND	ND	ND	ND								
PHENOTHIAZINE	92-84-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PRONAMIDE	23950-58-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PYRENE	129-00-0	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND
PYRIDINE	110-86-1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		
SAFROLE	94-59-7	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
SQUALENE	7683-64-9	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
STYRENE	100-42-5	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
THIANAPHTHENE (2,3-BENZOTHIOPHENE)	95-15-8	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
THIOACETAMIDE	62-55-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
THIOXANTHONE	492-22-8	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND

Table 6-8: EPA Sampling Episode List of Analytes Never Detected (continued)

		Non-Hazardous Subcategory														Hazardous Subcategory					
		Subtitle D Municipal					Subtitle D Non-Municipal														
POLLUTANT	**CAS NUM**	E4491	E4626	E4667	E4687	E4738	E4503	E4630	E4631	E4638	E4639	E4644	E4683	E4690	E4721	E4631	E4659	E4682	E4690	E4721	E4759
TRIPHENYLENE	217-59-4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
TRIPROPYLENEGLYCOLMETHYL ETHER	20324-33-8	ND		ND			ND	ND	ND	ND	ND	ND	ND	ND			ND	ND	ND		
1656: PESTICIDES/HERBICIDES																					
ACEPHATE	30560-19-1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
ACIFLUORFEN	50594-66-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	
ALACHLOR	15972-60-8	ND	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND
ALDRIN	309-00-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
ATRAZINE	1912-24-9	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND			
BENFLURALIN	1861-40-1	ND	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	
ALPHA-BHC	319-84-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND		ND	
BETA-BHC	319-85-7	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		
GAMMA-BHC	58-89-9		ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		
DELTA-BHC	319-86-8	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND		
BROMACIL	314-40-9	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND
BROMOXYNIL OCTANOATE	1689-99-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	
BUTACHLOR	23184-66-9	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	
CAPTAFOL	2425-06-1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
CAPTAN	133-06-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
CARBOPHENOTHION	786-19-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
ALPHA-CHLORDANE	5103-71-9	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	
GAMMA-CHLORDANE	5103-74-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
CHLOROBENZILATE	510-15-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND		
CHLORONEB	2675-77-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
CHLOROPROPYLATE	5836-10-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	
CHLOROTHALONIL	1897-45-6	ND	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	
DIBROMOCHLOROPROPANE (DBCP)	96-12-8	ND		ND		ND	ND		ND	ND	ND	ND	ND	ND	ND		ND		ND	ND	
DACTHAL (DCPA)	1861-32-1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	
4,4'-DDD	72-54-8	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4,4'-DDE	72-55-9	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	
4,4'-DDT	50-29-3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND			ND	ND	ND		
DIALLATE A	2303-16-4A	ND	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND		ND
DIALLATE B	2303-16-4B	ND	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND		
DICHLONE	117-80-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
DICOFOL	115-32-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
DIELDRIN	60-57-1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	
ENDOSULFAN I	959-98-8	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND	ND
ENDOSULFAN II	33213-65-9	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
ENDOSULFAN SULFATE	1031-07-8	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND		ND
ENDRIN	72-20-8	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND
ENDRIN ALDEHYDE	7421-93-4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND

Table 6-8: EPA Sampling Episode List of Analytes Never Detected (continued)

		Non-Hazardous Subcategory														Hazardous Subcategory					
		Subtitle D Municipal					Subtitle D Non-Municipal														
POLLUTANT	**CAS NUM**	E4491	E4626	E4667	E4687	E4738	E4503	E4630	E4631	E4638	E4639	E4644	E4683	E4690	E4721	E4631	E4659	E4682	E4690	E4721	E4759
ENDRIN KETONE	53494-70-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	
ETHALFLURALIN	55283-68-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	
ETRADIAZOLE	2593-15-9	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	
FENARIMOL	60168-88-9	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
HEPTACHLOR	76-44-8	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND		
HEPTACHLOR EPOXIDE	1024-57-3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		
ISODRIN	465-73-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND
ISOPROPALIN	33820-53-0	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND
KEPONE	143-50-0	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
METHOXYCHLOR	72-43-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	
METRIBUZIN	21087-64-9	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	
MIREX	2385-85-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
NITROFEN	1836-75-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	
NORFLUORAZON	27314-13-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PCB-1016	12674-11-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PCB-1221	11104-28-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PCB-1232	11141-16-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PCB-1242	53469-21-9	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PCB-1248	12672-29-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PCB-1254	11097-69-1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PCB-1260	11096-82-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PENTACHLORONITROBENZENE (PCNB)	82-68-8	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	
PENDAMETHALIN	40487-42-1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	
CIS-PERMETHRIN	61949-76-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
TRANS-PERMETHRIN	61949-77-7	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PERTHANE	72-56-0	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	
PROPACHLOR	1918-16-7	ND	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND	
PROPANIL	709-98-8	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PROPAZINE	139-40-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND		
SIMAZINE	122-34-9	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND			
STROBANE	8001-50-1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
TERBACIL	5902-51-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
TERBUTHYLAZINE	5915-41-3	ND	ND	ND			ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND				
TOXAPHENE	8001-35-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
TRIADIMEFON	43121-43-3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
TRIFLURALIN	1582-09-8	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND	
1657: PESTICIDES/HERBICIDES																					
AZINPHOS ETHYL	2642-71-9	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
AZINPHOS METHYL	86-50-0	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
CHLORFEVINPHOS	470-90-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND

Table 6-8: EPA Sampling Episode List of Analytes Never Detected (continued)

		Non-Hazardous Subcategory														Hazardous Subcategory					
		Subtitle D Municipal					**Subtitle D Non-Municipal**														
POLLUTANT	**CAS NUM**	E4491	E4626	E4667	E4687	E4738	E4503	E4630	E4631	E4638	E4639	E4644	E4683	E4690	E4721	E4631	E4659	E4682	E4690	E4721	E4759
CHLORPYRIFOS	2921-88-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
COUMAPHOS	56-72-4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
CROTOXYPHOS	7700-17-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
DEF	78-48-8	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
DEMETON A	8065-48-3A	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND		
DEMETON B	8065-48-3B	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
DIAZINON	333-41-5		ND			ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND		
DICHLORFENTHION	97-17-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND
DICHLORVOS	62-73-7	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	
DICROTOPHOS	141-66-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	
DIMETHOATE	60-51-5	ND			ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
DIOXATHION	78-34-2	ND	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
DISULFOTON	298-04-4	ND	ND				ND	ND	ND		ND		ND	ND	ND	ND	ND		ND	ND	ND
EPN	2104-64-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
ETHION	563-12-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
ETHOPROP	13194-48-8	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
FAMPHUR	52-85-7	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
FENSULFOTHION	115-90-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
FENTHION	55-38-9	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
HEXAMETHYLPHOSPHORAMIDE	680-31-9	ND	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	
LEPTOPHOS	21609-90-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
MALATHION	121-75-5		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND
MERPHOS	150-50-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND
METHAMIDOPHOS	10265-92-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND
METHYL CHLORPYRIFOS	5598-13-0	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
METHYL PARATHION	298-00-0	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND
METHYL TRITHION	953-17-3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
MEVINPHOS	7786-34-7	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
MONOCROTOPHOS	6923-22-4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
NALED	300-76-5	ND	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND
PARATHION (ETHYL)	56-38-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PHORATE	298-02-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PHOSMET	732-11-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PHOSPHAMIDON E	297-99-4	ND	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	
PHOSPHAMIDON Z	23783-98-4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
RONNEL	299-84-3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
SULFOTEPP	3689-24-5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
SULPROFOS	35400-43-2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
TEPP	107-49-3		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND
TERBUFOS	13071-79-9	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND

Table 6-8: EPA Sampling Episode List of Analytes Never Detected (continued)

		Non-Hazardous Subcategory														Hazardous Subcategory					
		Subtitle D Municipal					Subtitle D Non-Municipal														
POLLUTANT	**CAS NUM**	E4491	E4626	E4667	E4687	E4738	E4503	E4630	E4631	E4638	E4639	E4644	E4683	E4690	E4721	E4631	E4659	E4682	E4690	E4721	E4759
TETRACHLORVINPHOS	22248-79-9	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
TOKUTHION	34643-46-4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND
TRICHLORFON	52-68-6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
TRICHLORONATE	327-98-0	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
TRICRESYLPHOSPHATE	78-30-8	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
TRIMETHYLPHOSPHATE	512-56-1		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	
1658: PESTICIDES/HERBICIDES																					
DALAPON	75-99-0	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	ND	ND		ND		ND	ND
DICAMBA	1918-00-9		ND					ND	ND	ND	ND	ND	ND								
DICHLOROPROP	120-36-5	ND	ND					ND	ND	ND	ND	ND	ND								
DINOSEB	88-85-7	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND	ND		ND		ND	ND			
MCPA	94-74-6							ND	ND		ND		ND								
MCPP	7085-19-0					ND	ND	ND	ND		ND		ND			ND			ND		
PICLORAM	1918-02-1	ND	ND	ND	ND	ND		ND	ND	ND	ND	ND	ND		ND	ND	ND			ND	
2,4-D	94-75-7		ND			ND		ND	ND	ND	ND	ND	ND				ND				
2,4-DB	94-82-6		ND	ND		ND		ND	ND	ND	ND	ND	ND		ND						
2,4,5-T	93-76-5	ND	ND					ND	ND	ND	ND	ND	ND			ND	ND				
2,4,5-TP	93-72-1		ND			ND		ND	ND	ND	ND		ND								
CLASSSICAL WET CHEMISTRY																					
AMENABLE CYANIDE	C-025	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND		ND	ND	
AMMONIA NITROGEN	7664-41-7										ND										
BOD	C-002													ND							
CHLORIDE	16887-00-6	ND																			
COD	C-004																				
FLUORIDE	16984-48-8																				
HEXANE EXTRACTABLE MATERIAL	C-036	-					-		ND	ND	ND	ND	ND				ND				
HEXAVALENT CHROMIUM	18540-29-9					ND		ND	ND	ND	ND	ND	ND	ND		ND		ND	ND		
NITRATE/NITRITE	C-005																				
PH	C-006																				
RECOVERABLE OIL AND GREASE	C-007		-	-	-	-		-	-	-	-	ND	-	-	-	-	ND	-	-	-	-
TDS	C-010																				
TOC	C-012										ND	ND	ND				ND				
TOTAL CYANIDE	57-12-5			ND	ND	ND	ND				ND	ND	ND	ND					ND		
TOTAL PHENOLS	C-020							ND					ND								
TOTAL PHOSPHORUS	14265-44-2								ND	ND	ND		ND				ND				
TOTAL SOLIDS	C-008	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	ND	-	-	-	-
TOTAL SULFIDE	18496-25-8		-										ND	ND							
TSS	C-009										ND	ND	ND								

Table 6-9: Subtitle D Non-Hazardous Subcategory Median Raw Wastewater Concentration File

Subtitle D Non-Hazardous Pollutant of Interest	Subtitle D Municipal Median Concentration (ug/L)	Subtitle D Non-Municipal Median Concentration (ug/L)
Conventional		
BOD	240,000	67,000
TSS	137,000	20,500
Classical (Non-Conventional)		
Ammonia as Nitrogen	81,717	75,000
COD	994,000	1,100,000
Hexavalent Chromium	30	
Nitrate/Nitrite	651	950
TDS	2,894,289	4,850,000
TOC	376,521	236,000
Total Phenols	571	251
Organic (Toxic & Non-Conventional)		
1,4-Dioxane	10.8	
2-Butanone	1,082	
2-Propanone	992	
4-Methyl-2-Pentanone	101	
Alpha Terpineol	123	
Benzoic Acid	100	
Hexanoic Acid	5,818	
Methylene Chloride	36.8	
N,N-Dimethylformamide	10	
O-Cresol	15	
P-Cresol	75	
Phenol	102	
Toluene	108	
Tripropyleneglycol Methyl Ether	197	
Metals (Toxic & Non-Conventional)		
Barium	483	
Chromium	28	
Strontium	1,671	4,615
Titanium	63.8	
Zinc	100	
Pesticides/Herbicides (Non-Conventional)		
Dichloroprop	6.1	
Disulfoton	6.1	
Dioxins/Furans (Non-Conventional)		
1234678-HpCDD	0.00014	
OCDD	0.0018	

Table 6-10: Subtitle C Hazardous Subcategory Median Raw Wastewater Concentration File

Subtitle C Hazardous Pollutant of Interest	Median Conc. (ug/L)	Subtitle C Hazardous Pollutant of Interest	Median Conc. (ug/L)
Conventional		**Organics (cont.)**	
BOD	620,500	Toluene	104
Hexane Extractable Material	29,360	Trans-1,2-Dichloroethene	74.3
TSS	151,000	Trichloroethene	44.6
Classical (Non-Conventional)		Tripropyleneglycol Methyl Ether	853
Amenable Cyanide	1,638	Vinyl Chloride	42.7
Ammonia as Nitrogen	268,000	**Metals (Toxic & Non-Coventional)**	
COD	1,308,833	Arsenic	214
Nitrate/Nitrite	1,580	Chromium	47.8
TDS	15,958,333	Copper	36
TOC	440,902	Lithium	450
Total Phenols	25,004	Molybdenum	913
Organics (Toxic & Non-Conventional)		Nickel	240
1,1-Dichloroethane	45.7	Selenium	20
1,4-Dioxane	466	Strontium	3,044
2,4-Dimethylphenol	70	Tin	146
2-Butanone	1,048	Titanium	32.6
2-Propanone	2,889	Total Cyanide	82.5
4-Methyl-2-Pentanone	500	Zinc	100
Alpha Terpineol	95.7	**Pesticides/Herbicides (Non-Coventional)**	
Aniline	237	2,4,5-TP	4.1
Benzene	36.9	2,4-D	5
Benzoic Acid	2,482	2,4-DB	7.9
Benzyl Alcohol	43.6	Dicamba	4
Diethyl Ether	50	Dichloroprop	7.3
Ethylbenzene	44.8	MCPA	209
Hexanoic Acid	2,703	MCPP	870
Isobutyl Alcohol	39.7	Picloram	2
Methylene Chloride	118	Terbuthylazine	14.5
M-Xylene	29.4	**Dioxins/Furans (Non-Conventional)**	
Naphthalene	48.9	1234678-HpCDD	0.00018
O+P Xylene	17.1	1234678-HpCDF	0.00013
O-Cresol	78.8	OCDD	0.00035
Phenol	4,400	OCDF	0.0019
Pyridine	70		
P-Cresol	144		

Table 6-11: Range of Conventional and Selected Nonconventional Pollutants Raw Wastewater Average Concentrations (ug/L)

		Non-Hazardous Subcategory								Hazardous Subcategory			
		Subtitle D Municipal				Subtitle D Non-Municipal							
Pollutant	Cas No.	Min	Max	#Obs	#ND	Min	Max	#Obs	#ND	Min	Max	#Obs	#ND
Amenable Cyanide	C-025	-	-			-	-			0.01	29,895	4	2
Biochemical Oxygen Demand (BOD)	C-002	10,500	7,609,318	31	0	1,000	3,799,333	9	1	22,000	2,962,535	8	0
Total Suspended Solids (TSS)	C-009	6,500	14,470,000	26	0	4,000	16,500,000	8	2	31,667	568,233	9	0
pH	C-006	6.7	9.8	5	0	6.6	9.2	9	0	5.8	11	6	0
Hexane Extractable Material	C-036	5,000	26,000	4	0	5,000	64,000	9	4	5,000	64,800	5	1
Ammonia as Nitrogen	7664417	1,782	2,900,000	24	0	100	5,860,000	9	1	9,767	613,620	6	0
Chemical Oxygen Demand (COD)	C-004	35,000	11,881,700	28	0	80,000	16,700,000	9	0	270,000	6,872,579	8	0
Nitrate/Nitrite	C-005	20	50,800	17	3	50	36,000	9	1	380	192,516	6	0
Total Dissolved Solids (TDS)	C-010	752,000	17,533,000	22	0	936,000	33,900,000	9	0	4,594,917	31,000,000	6	0
Total Organic Carbon (TOC)	C-012	9,400	3,446,084	22	0	10,000	4,820,000	9	2	2,000	3,824,286	8	2
Total Phenols	C-020	50	2,051,249	15	1	50	39,200	9	1	280	192,367	5	0
Total Phosphorus	14265442	17	6,500	17	6	10	22,700	7	2	10	15,900	5	1

#Obs: Number of observations
#ND: Number of non-detects
(-): Not detected in any sample

Table 6-12: Range of Metals and Toxic Pollutants Raw Wastewater Average Concentrations (ug/L)

		Non-Hazardous Subcategory								Hazardous Subcategory			
		Subtitle D Municipal				Subtitle D Non-Municipal							
Pollutant	Cas No.	Min	Max	#Obs	#ND	Min	Max	#Obs	#ND	Min	Max	#Obs	#ND
Aluminum	7429905	60.5	111,100	7	0	21.5	712,000	8	3	-	-	-	-
Arsenic	7440382	-	-	-	-	2	18,300	10	3	17	1,370	9	1
Barium	7440393	43	3,500	19	1	140	3,570	10	0	-	-	-	-
Boron	7440428	36	5,704	7	0	76	16,250	8	0	511	8,175	7	0
Chromium	7440473	2	240	27	9	-	-	-	-	10	720	9	3
Chromium (Hexavalent)	18540299	2	247	9	3	-	-	-	-	-	-	-	-
Copper	7440508	-	-	-	-	-	-	-	-	9	610	9	4
Iron	7439896	2,494	1,667,600	27	0	556	100,000	9	0	3,585	36,758	7	0
Lithium	7439932	-	-	-	-	-	-	-	-	101	1,166	6	0
Magnesium	7439954	24,100	212,480	14	0	8,139	388,000	9	0	8,307	440,767	6	0
Manganese	7439965	149	78,820	20	0	471	7,151	9	0	81	9,045	6	0
Molybdenum	7439987	-	-	-	-	4.2	69	8	4	9	18,757	6	1
Nickel	7440020	-	-	-	-	-	-	-	-	60	2,871	9	0
Phosphorus	7723140	-	-	-	-	-	-	-	-	551	24,650	7	1
Selenium	7782492	-	-	-	-	-	-	-	-	14	173	9	3
Silicon	7440213	1,034	91,100	4	0	2,498	159,000	8	0	2,520	17,911	6	0
Strontium	7440246	787	2,146	4	0	277	30,100	8	0	369	30,839	6	0
Sulfur	7704349	3,969	107,999	4	0	13,700	386,573	7	0	10,360	786,857	6	0
Tin	7440315	-	-	-	-	-	-	-	-	30	1,118	6	1
Titanium	7440326	4	157	6	1	4.4	1,740	8	2	3	764	6	2
Total Cyanide	57125	-	-	-	-	-	-	-	-	10	13,317	10	1
Zinc	7440666	11.5	31,813	27	1	2	1,240	10	1	45.5	846	9	0

#Obs: Number of observations
#ND: Number of non-detects
(-): Not detected in any sample

Table 6-13: Range of Organic Pollutants Raw Wastewater Average Concentrations (ug/L)

		Non-Hazardous Subcategory								Hazardous Subcategory			
		Subtitle D Municipal				Subtitle D Non-Municipal							
Pollutant	Cas No.	Min	Max	#Obs	#ND	Min	Max	#Obs	#ND	Min	Max	#Obs	#ND
1,1-Dichloroethane	75343	-	-	-	-	-	-	-	-	0.5	250	10	4
1,4-Dioxane	123911	10	323	5	2	-	-	-	-	10	7,611	9	5
1234678-HpCDD	35822469	0.00005	0.007	3	1	-	-	-	-	0.00005	0.007	6	2
1234678-HpCDF	67562394	-	-	-	-	-	-	-	-	0.00005	0.001	6	2
2,4-D	94757	-	-	-	-	-	-	-	-	0.5	310	9	4
2,4-DB	94826	-	-	-	-	-	-	-	-	2.9	120	6	1
2,4-Dimethylphenol	105679	-	-	-	-	-	-	-	-	10	2,546	9	5
2,4,5-TP	93721	-	-	-	-	-	-	-	-	0.1	13.2	9	4
2-Butanone	78933	19.3	36,544	14	3	-	-	-	-	50	15,252	10	3
2-Propanone	67641	50	8,614	12	4	50	780	10	6	73	8,166	10	1
4-Methyl-2-Pentanone	108101	35	46,161	13	4	-	-	-	-	50	3,168	9	3
Alpha Terpineol	98555	10	1,061	5	1	-	-	-	-	10	654	6	3
Aniline	62533	-	-	-	-	-	-	-	-	10	2,500	9	5
Benzene	71432	-	-	-	-	-	-	-	-	0.3	229	10	5
Benzoic Acid	65850	0.55	33,335	7	3	-	-	-	-	50	306,194	6	1
Benzyl Alcohol	100516	-	-	-	-	-	-	-	-	10	5,690	6	4
Dicamba	1918009	-	-	-	-	-	-	-	-	0.49	31	6	0
Dichloroprop	120365	1	29	5	2	-	-	-	-	2.2	44	6	1
Diethyl Ether	60297	-	-	-	-	-	-	-	-	10	159	9	5
Disulfoton	298044	2.3	20	5	2	-	-	-	-	-	-	-	-
Ethyl Benzene	100414	-	-	-	-	-	-	-	-	0.5	1,072	10	4
Hexanoic Acid	142621	10	37,256	5	1	-	-	-	-	13	31,086	6	1
Isobutyl Alcohol	78831	-	-	-	-	-	-	-	-	10	10,000	9	6
MCPA	94746	-	-	-	-	50	4370	8	2	15	7,071	6	1
MCPP	7085190	-	-	-	-	50	1900	8	4	13	12,887	6	3
Methylene Chloride	75092	1.6	237	20	6	-	-	-	-	1	19,112	10	4
M-Xylene	108383	-	-	-	-	-	-	-	-	10	650	6	2
Naphthalene	91203	-	-	-	-	-	-	-	-	10	7,799	9	5
N,N-Dimethylformamide	68122	10	1,008	5	3	-	-	-	-	-	-	-	-
OCDD	3268879	0.0001	0.082	3	1	0.0001	0.0176	8	5	0.0001	0.062	6	2
OCDF	39001020	-	-	-	-	-	-	-	-	0.0001	0.012	6	2
O-Cresol	95487	1	2,215	8	6	-	-	-	-	10	626	9	2
O+P Xylene	136777612	-	-	-	-	-	-	-	-	10	230	6	2
P-Cresol	106445	1	998	9	3	-	-	-	-	10	17,396	7	2
Phenol	108952	2	1,425	14	5	-	-	-	-	10	99,947	9	1
Picloram	1918021	-	-	-	-	-	-	-	-	0.5	7.3	5	2
Pyridine	110861	-	-	-	-	-	-	-	-	10	10,000	9	6
Terbuthylazine	5915413	-	-	-	-	-	-	-	-	5	97	5	2
Toluene	108883	3	598	23	5	-	-	-	-	5	2,541	10	3
Trans-1,2-Dichloroethene	156605	-	-	-	-	-	-	-	-	0.4	6,237	10	4
Trichloroethene	79016	-	-	-	-	-	-	-	-	0.5	27,083	10	4
Tripropyleneglycol Methyl Ether	20324338	99	1,235	5	2	-	-	-	-	99	3,182	6	3
Vinyl Chloride	75014	-	-	-	-	-	-	-	-	0.2	1,429	10	5

#Obs: Number of observations
#ND: Number of non-detects
(-): Not detected in any sample

Table 6-14: Dioxins and Furans at Non-Hazardous EPA Sampling Episodes by Episode and Sample Point

Subtitle D Episode/SP	Sample Type	1234678-HpCDD	1234678-HpCDF	OCDD	OCDF	123478-HxCDD	123478-HxCDF	1234789-HpCDF	123678-HxCDD	123678-HxCDF	12378-PeCDD	12378-PeCDF	123789-HxCDD	123789-HxCDF	234678-HxCDF	23478-PeCDF	2378-TCDD	2378-TCDF
Municipal																		
4491 sp01 - inf	grab	140 pg/L	ND	1800 pg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4626 sp01 - inf	-	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
4626 sp02 - inf	-	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
4626 sp03 - inf	-	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
4626 sp08 - eff	-	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
4626 sp09 - FC	grab	32.9 ng/kg	ND	803 ng/kg	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4626 sp09 - FC	grab	41.2 ng/kg	ND	1100 ng/kg	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4667 sp01 - inf	-	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
4667 sp06 - eff	-	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
4667 sp07 - FC	grab	29 ng/kg	ND	279 ng/kg	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4667 sp07 - FC	grab	32 ng/kg	ND	271 ng/kg	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4667 sp07 - FC	grab	44 ng/kg	ND	308 ng/kg	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4667 sp07 - FC	grab	43 ng/kg	ND	338 ng/kg	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4667 sp07 - FC	grab	39 ng/kg	ND	290 ng/kg	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4687 sp01 - inf	comp	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4687 sp03 - eff	comp	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
4738 sp01 - inf	grab	240 pg/L	56 pg/L	11,000 pg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4738 sp02 - inf	grab	480 pg/L	ND	5,300 ng/kg	ND	ND	ND	ND	6 ng/kg	ND	ND	ND	16 ng/kg	ND	ND	ND	ND	ND
Non-Municipal																		
4503 sp01 - inf	grab	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4630 sp01 - inf	grab	103 pg/L	ND	5380 pg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4631 sp03 - inf	grab	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4638 sp01 - inf	grab	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4639 sp01 - inf	grab	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4644 sp01 - inf	grab	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4721 sp04 - inf	grab	ND	ND	503 pg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND

Note: Only filter cake was analyzed for dioxins and furans in Municipal episodes 4626 and 4667

sp: sample point
inf: influent
eff: effluent

comp: composite sample
grab: grab sample

NS: Not sampled
ND: Non-detect
FC: Filter cake

mg/L = 1000 ug/L
ug/L = 1000 ng/L
ng/L = 1000 pg/L

Table 6-15: Dioxins and Furans at Hazardous EPA Sampling Episodes by Episode and Sample Point

Episode Sample Point	Sample Type	1234678-HpCDD	1234678-HpCDF	OCDD	OCDF	123478-HxCDD	123478-HxCDF	1234789-HpCDF	123678-HxCDD	123678-HxCDF	12378-PeCDD	12378-PeCDF	123789-HxCDD	123789-HxCDF	234678-HxCDF	23478-PeCDF	2378-TCDD	2378-TCDF
4631 sp01 - inf	grab	13,600 pg/L	1,180 pg/L	116,000 pg/L	6,600 pg/L	ND	95.4 pg/L	162 pg/L	798 pg/L	202 pg/L	ND	79.1 pg/L	196 pg/L	ND	ND	ND	ND	31.1 pg/L
4631 sp02 - inf	grab	479 pg/L	88 pg/L	7,920 pg/L	573 pg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4659 sp01 - inf	grab	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4682 sp01 - inf	grab	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4682 sp02 - inf	grab	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4721 spD01 - inf	comp	446 pg/L	ND	4,160 pg/L	135 pg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4721 sp01 - inf	comp	752 pg/L	86 pg/L	9,070 pg/L	357 pg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4721 sp01 - inf	comp	593 pg/L	55 pg/L	6,290 pg/L	243 pg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4721 sp01 - inf	comp	576 pg/L	ND	5,040 pg/L	136 pg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4721 sp01 - inf	comp	496 pg/L	62 pg/L	4,630 pg/L	212 pg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4721 sp02 - eff	-	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
4721 sp03 - inf	grab	551 pg/L	70 pg/L	5,080 pg/L	162 pg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4721 sp05 - inf	grab	698 pg/L	ND	5,080 pg/L	290 pg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4721 sp06 - inf	grab	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4759 sp01 - inf	comp	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4759 sp03 - eff	comp	ND	ND	100 pg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND

sp: sample point
inf: influent
eff: effluent

comp: composite sample
grab: grab sample

D: Duplicate
ND: Non-detect
NS: Not sampled

mg/L = 1000 ug/L
ug/L = 1000 ng/L
ng/L = 1000 pg/L

7.0 POLLUTANT PARAMETER SELECTION

7.1 Introduction

EPA reviewed wastewater characterization data presented in Chapter 6 to identify which pollutant parameters present in landfills wastewater should be considered for regulation. EPA classifies pollutants into the following three categories: conventional, nonconventional, and toxic pollutants. Conventional pollutants include BOD_5, TSS, oil and grease, and pH. Toxic pollutants -- EPA also refers to them as priority pollutants -- include selected metals, pesticides and herbicides, and over 100 organic parameters that represent a comprehensive list of volatile and semi-volatile compounds. Nonconventional pollutants are any pollutants that do not fall within the specific conventional and toxic pollutant lists and include, for example, TOC, COD, chloride, fluoride, ammonia-nitrogen, nitrate/nitrite, total phenol, and total phosphorous.

This chapter presents the criteria used for the selection of pollutant parameters EPA evaluated for regulation and the selection of pollutants for which EPA has established effluent limitations and standards.

7.2 Pollutants Considered for Regulation

To characterize landfill wastewater and to determine the pollutants that it should evaluate for potential limitations and standards, EPA collected wastewater characterization samples at 15 landfill facilities, in addition to influent data collected at six, week-long sampling episodes. EPA analyzed wastewater samples for 470 conventional, toxic, and nonconventional pollutants including metals, organics, pesticides, herbicides, and dioxins and furans. Chapter 6 presents this wastewater characterization data.

From the original list of 470 analytes, EPA developed a list of "pollutants of interest" for each subcategory that it would further evaluate for possible regulation. This list reflects the types of pollutants typically found in landfill wastewater. From this list of pollutants, EPA calculated the current pollutant mass loadings for

the industry and estimated the pollutant loading associated with compliance with the final limitations and standards. The list of pollutants of interest also served as the basis for selecting pollutants for regulation.

7.3 Selection of Pollutants of Interest

EPA determined pollutants of interest for each subcategory using the raw wastewater data collected during the EPA sampling program. Chapter 6 presents the landfill facilities sampled in each subcategory in Table 6-8 and whether EPA detected the pollutants analyzed in the facility's raw wastewater. EPA only included the sampled facilities that were within the scope of the rule to determine the pollutants of interest. Therefore, EPA did not include sampling data from captive exempt facilities nor contaminated ground water data in the analysis. Figure 7-1 presents a diagram of the procedures used to select pollutants of interest.

EPA applied the following criteria to develop a list of pollutants for further evaluation for each subcategory:

1. EPA determined any pollutant detected three or more times in the influent at a concentration at or above 5 times the minimum level at more than one facility to be a pollutant of interest.

2. For dioxins/furans, EPA determined any dioxin or furan detected three or more times in the influent at a concentration above the minimum level at more than one facility to be a pollutant of interest.

3. EPA excluded pollutants that are naturally occurring compounds in soil or ground water at landfill facilities or pollutants that are used as treatment chemicals in this industry from the pollutants of interest list. These compounds include aluminum, boron, calcium, chloride, fluoride, iron, manganese, magnesium, potassium, silicon, sodium, sulfur, total phosphorus, and total sulfide.

Tables 7-1 and 7-2 list the final pollutants of interest for the Non-Hazardous and Hazardous subcategories that EPA has selected for further evaluation after applying these criteria. As shown in Table 7-1, EPA identified separate lists of pollutants of interest for Subtitle D municipal solid waste landfills and Subtitle D non-municipal solid waste landfills. However, EPA combined these two lists for the entire Non-Hazardous landfill subcategory. At proposal, one Non-Hazardous subcategory pollutant of interest, MCPA, was present at non-municipal solid waste landfills and was not present at municipal solid waste landfills. However, after proposal, EPA re-evaluated the status of several facilities in the landfills database

and now classifies an additional nine facilities as captive landfills not included within the scope of this guideline. With the removal of pollutants associated with these facilities from the analysis, EPA determined that, after application of the criteria, MCPA was no longer a pollutant of interest for non-municipal facilities because it was detected only twice in the influent at a concentration at or above 5 times the minimum level at two non-captive facilities. Therefore, EPA did not include MCPA as a pollutant of interest for the Non-Hazardous subcategory for the final rule. Pollutants of interest in both subcategories include conventional, nonconventional, and toxic pollutants and include metals, organics, pesticides, herbicides, and dioxins and furans.

7.4 Development of Pollutant Discharge Loadings

EPA estimated mass loadings of pollutant discharges for the pollutants of interest on a facility-by-facility basis. The Agency calculated pollutant loadings for current discharges and estimated projected discharges based on each of the regulatory options using the procedures described below.

7.4.1 Development of Current Discharge Concentrations

The current discharge concentration database contains the discharge concentration for each pollutant of interest at each facility in each subcategory. The Agency determined mass loadings by multiplying the pollutant concentration by the facility-specific regulated wastewater flow. EPA used all available data including Detailed Questionnaire and Detailed Monitoring Questionnaire data and EPA sampling data to determine mass loadings.

In the Detailed Questionnaire and Detailed Monitoring Questionnaires, EPA requested facilities to provide information on wastewater treatment-in-place and to provide concentration data on treated wastewater effluent. The Agency compiled all effluent wastewater data for each facility after screening the data using the conventions discussed in Chapter 4 for raw wastewater. For facilities with multiple effluent sample points, EPA determined the final effluent concentration by taking a flow weighted average of the samples. From the effluent wastewater data from each facility, the Agency created a data file that contained one

average concentration value for each pollutant of interest at each facility. The amount of data in the file varied significantly from facility to facility. EPA based several of the current discharge concentrations on hundreds of sampling data points obtained through the Detailed Monitoring Questionnaire, while it based others on as few as one sampling data point. The Detailed Monitoring Questionnaire data reflect up to three years of data and are unique to each facility in terms of numbers of parameters analyzed and monitoring frequency. Additionally, monitoring may have been performed weekly, monthly, or quarterly. For facilities sampled by EPA, there was information available for all 470 analytes, and sampling typically reflected the daily performance of a system over a five-day period.

For facilities with wastewater treatment-in-place, but with either no available effluent data or incomplete effluent data, EPA generated a treated effluent average concentration. To develop the treated effluent average concentration, EPA grouped facilities by subcategory and then placed them in treatment-in-place groups, depending on the type of treatment employed on site. Within a treatment-in-place group, the Agency calculated the treated effluent average concentration for a pollutant of interest by taking the median of all weighted source averages for all facilities within the treatment-in-place group. If there were no data for a particular pollutant within a treatment-in-place group, EPA calculated the treated effluent average concentration for a pollutant of interest in a subcategory by taking the median of all weighted source averages for all facilities within the entire subcategory.

For facilities with no treatment-in-place, the Agency used raw wastewater concentrations to represent current effluent discharge values. EPA calculated facility averages using all available data sources and using the procedures outlined above. For facilities with no treatment-in-place and with either no influent data or incomplete influent data, the Agency used the subcategory median raw wastewater concentration (see Section 6.3.3 for details on developing the Median Raw Wastewater Concentration File) to represent the current discharge for each pollutant of interest.

For the Hazardous subcategory and for Subtitle D non-municipal solid waste facilities in the Non-Hazardous subcategory, there were insufficient effluent data to calculate a representative treatment-in-place or subcategory treated effluent average concentration result for several pollutants of interest. The alternate methodologies developed to calculate representative current discharge concentration values for both the Hazardous subcategory and for Subtitle D non-municipal facilities in the Non-Hazardous subcategory are discussed below.

7.4.1.1 Alternate Methodology for Non-Hazardous Subcategory: Subtitle D Non-Municipal

For Subtitle D non-municipal solid waste facilities in the Non-Hazardous subcategory, EPA used the effluent data from municipal solid waste landfills to supplement insufficient non-municipal data. EPA concluded this was appropriate in the circumstances because of the similarities in the median raw wastewater concentrations from Subtitle D municipal and non-municipal facilities. Table 6-7 in Chapter 6 presents the Subtitle D municipal and non-municipal median raw wastewater concentration data.

EPA employed the following procedure to calculate current discharge concentrations for Subtitle D non-municipal solid waste facilities. First, EPA used all available non-municipal landfill effluent data. Next, EPA placed non-municipal facilities in municipal facility treatment-in-place groups according to treatment employed on site. Then, EPA used municipal landfills treatment-in-place treated effluent average concentrations for each non-municipal facility with insufficient data.

7.4.1.2 Alternate Methodology for the Hazardous Subcategory

EPA estimated current discharge concentrations for the facilities in the Hazardous subcategory using the long-term averages developed for the subcategory (see Chapter 11: Development of Effluent Limitations and Standards). EPA's data collection efforts did not identify any direct discharging hazardous landfills, and EPA obtained detailed information from only three indirect discharging landfills. Therefore, the Agency modeled the current discharge concentrations on the small number of indirect discharging facilities in the

EPA database as a function of the expected discharge concentrations after treatment using the long-term averages. EPA used industry-provided effluent data whenever available.

The Agency developed an approach based upon the installed treatment system at the facility. EPA estimated the current discharge concentration as twice the long-term average (LTA) for a facility without any biological or chemical treatment-in-place. The modeling approach used to develop the current discharge concentration (CDC) for the indirect dischargers in the Hazardous subcategory is presented below.

QID	Treatment-In-Place	Modeling Scheme
16017	Separation and neutralization	2 x LTA^{med}
16041	Sequencing batch reactors	LTA
16087	Equalization, chemical precipitation, primary sedimentation, activated sludge, and secondary sedimentation	LTA

For facility 16017, the current discharge concentration value was based upon a function of the LTA^{med}. The LTA^{med} is the median of the long-term averages in the Hazardous subcategory. The long-term averages used in this subcategory are from BAT facilities 16041 and 16087. Therefore, the corresponding long-term averages were used for both of these BAT facilities.

7.4.2 Development of Pollutant Mass Loadings

Using the current discharge concentrations discussed above, EPA generated mass loading estimates for each pollutant of interest at each facility by multiplying the current discharge concentration value by the facility's average daily discharge flow rate. This resulted in mass loadings, reported in pounds per day, for each facility in the database. EPA calculated mass loadings to determine the amount of pollution discharged directly or indirectly to surface waters by landfill facilities and to estimate the amount of pollutant reduction after implementation of each regulatory technology option. Summaries of pollutant mass loadings for the selected regulatory options are presented in Chapter 11.

7.5 Assessment of Pollutants of Interest

As indicated above, EPA developed extensive lists of pollutants of interest for this industry. EPA used the full list of pollutants of interest to develop pollutant loadings and pollutant reductions as a result of treatment. However, the Agency only selected certain pollutants for regulation, since specific regulation of every pollutant is not always the most cost-effective approach to developing effluent limitations guidelines.

The treatment technologies evaluated as the basis of the regulation remove classes of compounds with similar treatability characteristics. Several of the pollutants of interest in the Landfills industry are similar in terms of their chemical structure and treatability. As a result, the regulation of a set of pollutants within a chemical class ensures that the treatment technologies will provide adequate control of other pollutants of interest within that class of compounds.

Based upon this analysis, EPA decided not to regulate certain pollutants of interest in the Non-Hazardous and Hazardous subcategories because their removals are represented adequately by another regulated pollutant, as discussed in the sections below. In addition, the Agency did not select several other pollutants of interest for regulation because EPA found these pollutants at concentrations below treatable levels in the Landfills industry. EPA also did not select pollutants for regulation if the Agency determined that these pollutants were found at only trace amounts in the industry, and therefore were not likely to cause toxic effects. The Agency also excluded several pollutants of interest from regulation because the selected BPT treatment technology would not remove these pollutants.

7.6 Selection of Pollutants To Be Regulated for Direct Dischargers

Based upon the data analyses outlined above, EPA developed a list of pollutants to be regulated for the Hazardous and Non-Hazardous subcategories. Figure 7-2 presents a diagram that illustrates the procedures used to select the regulated pollutants. EPA is not establishing effluent limitations and standards for all conventional, toxic, and nonconventional pollutants. There may be pollutants present in a specific landfill or type of landfill for which EPA did not establish limitations under this guideline but which may be

of concern to a receiving stream or POTW. Due to the specific nature of landfill waste at various sites, permit writers and local authorities may need to consider case-by-case limitations or standards for these pollutants. EPA's regulations require the permit writer or local authority to include technology-based limits for any toxic pollutant which is or may be discharged at a level greater than the level which can be achieved by treatment requirements appropriate to the permittee or which may pass through or interfere with POTW operations. (40 CFR § 122.44(e), 125.3. *See also* 40 CFR § 403.5(c) which requires the establishment of local limits in a POTW pretreatment program for any pollutant which may cause pass through or interference). The following sections discuss EPA's reasons for not establishing effluent limitations for selected pollutants.

7.6.1 Non-Hazardous Subcategory Pollutants to be Regulated for Direct Dischargers

EPA developed the list of pollutants to be regulated for the Non-Hazardous subcategory from the pollutants of interest list for the Non-Hazardous subcategory. The non-hazardous pollutants of interest list combines the pollutants of interest from Subtitle D municipal and non-municipal solid waste facilities for a total of 32 pollutants of interest. The BPT/BAT facilities selected by EPA demonstrate removal of the regulated pollutants. These facilities employed equalization, biological treatment, and for some, multimedia filtration. Initially, EPA considered regulating all 32 pollutants of interest. After a thorough analysis, EPA, however, chose not to set limitations for 24 pollutants of interest under BPT/BAT/NSPS for one of the following reasons:

- The pollutant (or pollutant parameter) is controlled through the regulation of other pollutants (or pollutant parameters).
- The pollutant (or pollutant parameter) is present in only trace amounts in the subcategory and/or is not likely to cause toxic effects.
- C The pollutant (or pollutant parameter) is not controlled by the selected BPT technology.

The following seven Non-Hazardous subcategory pollutants of interest are pollutants that are controlled through the regulation of other pollutants**:**

Seven Pollutants Not Selected for Regulation in the Non-Hazardous Subcategory Because They Are Controlled Through the Regulation of Other Pollutants
COD
TOC
Total Phenols
Hexanoic Acid
O-Cresol
Tripropyleneglycol Methyl Ether
Titanium

COD is an alternative method of estimating the oxygen demand of the wastewater. However, EPA selected BOD_5 for regulation because it is more appropriately controlled by a biological treatment system. TOC measures all oxidizable organic material in a waste stream, including the organic chemicals not oxidized (and, therefore, not detected) in BOD_5 and COD tests. TOC is a rapid test for estimating the total organic carbon in a waste stream. For reasons similar to those used for not selecting COD for regulation, EPA did not select TOC for regulation. Total phenols is a general wet chemistry indicator measurement for phenolic compounds. Regulation of phenol will control other phenolic compounds. Similarly, hexanoic acid is relatively biodegradable and should be controlled by regulating benzoic acid. O-cresol is structurally similar to p-cresol and should be controlled by regulating p-cresol. Tripropyleneglycol methyl ether has treatability characteristics similar to alpha terpineol in a biological treatment system and should be controlled by regulating alpha terpineol. EPA determined that titanium will be removed incidentally by biological treatment in the same manner as zinc, through sorption into the biomass. Therefore, titanium should be controlled by regulating zinc.

In the proposal, EPA chose not to regulate 2-butanone, 2-propanone, and 4-methyl-2-pentanone because they were controlled through the regulation of toluene. After proposal EPA decided not to regulate toluene. The reasons these pollutants were not selected for regulation in the final rule are discussed below.

The following thirteen Non-Hazardous subcategory pollutants of interest are present in only trace amounts and/or are not likely to cause toxic effects:

Thirteen Pollutants Not Selected for Regulation in the Non-Hazardous Subcategory Because They Are Present in Only Trace Amounts and/or Are Not Likely to Cause Toxic Effects
Nitrate/Nitrite
TDS
1,4-Dioxane
4-Methyl-2-Pentanone
Methylene Chloride
N,N-Dimethylformamide
Toluene
Barium
Chromium
Dichloroprop
Disulfoton
1,2,3,4,6,7,8-HpCDD
OCDD

EPA presents the Non-Hazardous subcategory median raw wastewater concentration data for the pollutants of interest in Chapter 6, Table 6-9, and the minimum and maximum concentrations for conventional and nonconventional pollutants, metals, organic pollutants, and dioxins/furans in Tables 6-11 through 6-14.

For this industry, nitrate/nitrite is used primarily as a measure of the extent of nitrification that occurs during the biodegradation process. Typically, levels of nitrate/nitrite found in landfill wastewater do not require removal. Removal of nitrate/nitrite can be obtained by specially designed biological treatment systems (such as nitrification/denitrification systems) that are able to complete the conversion of nitrate/nitrite to nitrogen gas. Often, removal of nitrate/nitrite is required to address specific water quality concerns for an individual receiving water (i.e., nutrient problems in the Great Lakes). EPA has determined that the levels of nitrate/nitrite in landfill wastewater do not justify regulation on a national level and individual permit writers can address specific water quality considerations.

TDS is used primarily as a water quality measurement and not as a pollutant that can be controlled through biological treatment. It often is used as a measurement of the salinity of an ambient water or a wastewater and often indicates the presence of naturally occurring salts of metals such as sodium, iron, and magnesium. While it can inhibit biological treatment processes at levels above 10,000 mg/L, acclimated biological treatment systems can operate successfully with influent TDS concentrations as high as 76,000 mg/L (reference 55). The median concentration of total dissolved solids in the Non-Hazardous subcategory was only 4,850 mg/L for non-municipal solid waste landfills and 2,890 mg/L for municipal solid waste landfills. Therefore, EPA has determined that concentrations of total dissolved solids found in landfills in the Non-Hazardous subcategory do not justify regulation. EPA's sampling data showed levels of n,n-dimethylformamide in landfill wastewater generally near the analytical detection limit (median concentration for non-hazardous municipal solid waste landfills was 10 ug/L) and, because of this low concentration throughout the subcategory, regulation was not warranted.

EPA classifies four pollutants, 1,4-dioxane, 4-methyl-2-pentanone, methylene chloride, and toluene as "volatile organics" under analytical test method 1624. In the proposed rule, EPA established direct discharge limitations for toluene for landfills in the Non-Hazardous subcategory. However, after proposal, EPA decided not to regulate toluene because it is not treated by the biological treatment technology selected as the basis for the landfills effluent limitations. Furthermore, based on the concentration of toluene in untreated municipal leachate (108 ug/L), the Agency concluded that the loading of toluene to the atmosphere will not cause toxic effects.

While EPA acknowledges that a small portion of the removal of these pollutants is due to biological degradation, these pollutants are highly volatile and the primary mechanism for their removal from wastewater is through volatilization to the atmosphere. EPA based these final regulations on the performance of an aerated biological system. Wastewater aeration may increase the volatilization of certain organic compounds, a potential environmental concern. While EPA does not recognize the transfer of pollutants from one media to another as effective treatment, based on the concentrations of these pollutants

in untreated wastewater (below treatable levels (10 times the method detection limit)), indications are that the potential increase in air emissions due to this regulation will be minimal.

Volatile organic compound (VOC) levels in historic landfill leachate (from both hazardous and non-hazardous waste landfills dating from the 1930s to the mid-1990s) are also at levels which are low enough as not to call into question EPA's determination to base these rules on the performance of aerated biological systems. Tables 6-9, 6-10, and 6-13 show the concentrations of VOCs found in landfill wastewater.

Furthermore, EPA's Office of Air and Radiation is currently evaluating the air emissions from wastewater generated at municipal solid waste landfills, and intends to take the landfills effluent limitations guidelines into account in determining whether further controls under Section 112 of the Clean Air Act (which requires technology-based standards for hazardous air pollutants emitted by major sources of emissions of those pollutants) are justified. (Preliminary indications are that hazardous air pollutant emissions from aeration would be a minor fraction of those from other landfill emission sources such as landfill gas emissions.)

EPA's sampling detected two metals, barium and chromium, below treatable levels at non-hazardous landfills in the EPA database. The median raw wastewater concentrations of barium and chromium found at municipal landfills is 0.48 mg/L and 0.03 mg/L, respectively, less than 5 times the method detection limit. EPA is excluding these two metals from regulation because, at the concentrations found at non-hazardous landfills, these pollutants are not likely to cause toxic effects.

EPA found low levels of dichloroprop, disulfoton, 1,2,3,4,6,7,8-HpCDD, and OCDD in raw wastewater at several Non-Hazardous subcategory landfills. At the concentrations found, EPA expects these pollutants to partition to the biological sludge created as a result of the use of the BPT/BAT treatment technologies. EPA sampling data and calculations conclude that the concentrations of these pollutants present in the wastewater will not prevent the sludge from being redeposited in a non-hazardous landfill.

The following four pollutants were not selected for regulation in the Non-Hazardous subcategory because they are not controlled by the selected BPT/BAT technology:

Four Pollutants Not Selected for Regulation in the Non-Hazardous Subcategory Because They Are Not Controlled by the Selected BPT/BAT Technology
2-Butanone
2-Propanone
Hexavalent Chromium
Strontium

EPA classifies 2-butanone and 2-propanone as "volatile organics" under analytical test method 1624. Because the selected BPT/BAT technology for the Non-Hazardous subcategory is aerated equalization followed by biological treatment and then multimedia filtration, EPA determined that the majority of the removal of volatile organic compounds is due to volatilization to the atmosphere in either the aerated equalization tanks or in the activated sludge aeration basin. Therefore, EPA did not regulate volatile organic pollutants because the BPT/BAT technology does not provide controls for the removal of these pollutants.

EPA detected hexavalent chromium and strontium in wastewater at the facilities selected as the basis for BPT/BAT/NSPS, but EPA did not have adequate removal data at the BPT/BAT/NSPS facilities employing biological treatment and, therefore, these pollutants could not be regulated. For both pollutants, EPA had removal data from one BPT/BAT facility. In both cases, the BPT facilities demonstrated negative percent removals of these pollutants. In addition to the lack of adequate data, EPA determined that for this subcategory, these metals are not present in concentrations that are likely to cause toxic effects. Therefore, these two metals were excluded from regulation in the Non-Hazardous subcategory.

In conclusion, the following eight pollutants of interest are regulated under BPT/BAT/NSPS in the Non-Hazardous subcategory:

Eight Pollutants Selected for Regulation in the Non-Hazardous Subcategory
Ammonia as Nitrogen
BOD_5
TSS
Alpha Terpineol
Benzoic Acid
P-Cresol
Phenol
Zinc

The Agency wishes to note that zinc was selected for regulation in spite of the fact that exclusion criteria used to eliminate other pollutants of interest apply, at least partially. Zinc has been selected for regulation in spite of its relatively low untreated wastewater concentration. The median concentration of zinc found in raw wastewater at municipal solid waste landfills and at non-municipal solid waste landfills is 0.10 mg/L and 0.09 mg/L, respectively. EPA selected zinc for regulation because EPA observed incidental removals ranging from 58 percent to 90 percent at the treatment systems selected for BPT. Additionally, EPA's sampling did not find raw wastewater concentrations of zinc at levels that would inhibit biological treatment systems (see Chapter 11, Section 11.2.1).

Chapter 11 describes in detail the development of the effluent limitations for each of these pollutants.

7.6.2 Hazardous Subcategory Pollutants to be Regulated for Direct Dischargers

EPA developed the list of pollutants to be regulated for the Hazardous subcategory from the Hazardous subcategory pollutants of interest list. The two BPT/BAT facilities selected by EPA demonstrate removal of the regulated pollutants through the use of chemical precipitation and biological treatment. Initially, EPA considered regulating all 63 pollutants of interest; EPA chose, however, not to set limitations for 50 pollutants of interest under BPT/BAT/NSPS for one of the following reasons:

- The pollutant (or pollutant parameter) is controlled through the regulation of other pollutants (or pollutant parameters).

- The pollutant (or pollutant parameter) is present in only trace amounts in the subcategory and/or is not likely to cause toxic effects.

C The pollutant (or pollutant parameter) is not controlled by the selected BPT technology.

As discussed in Chapter 6, after proposal, EPA analyzed the raw wastewater characterization data for hazardous landfills without CERCLA ground water data. As a result, raw wastewater concentrations for several pollutants of interest have changed since proposal and, therefore, in some cases, EPA's reasons for not selecting these pollutants for regulation also have changed.

EPA did not select the following thirteen Hazardous subcategory pollutants of interest for regulation because they are controlled through the regulation of other pollutants:

Thirteen Pollutants Not Selected for Regulation in the Hazardous Subcategory Because They Are Controlled Through the Regulation of Other Pollutants
COD
TOC
Total Phenols
2,4-Dimethylphenol
Benzyl Alcohol
Diethyl Ether
Isobutyl Alcohol
Hexanoic Acid
O-Cresol
Tripropyleneglycol Methyl Ether
Molybdenum
Nickel
Strontium

COD is an alternative method of estimating the oxygen demand of the wastewater. EPA, however, selected BOD_5 for regulation because it is more appropriately controlled by a biological treatment system. TOC measures all oxidizable organic material in a waste stream, including the organic chemicals not oxidized (and, therefore, not detected) in BOD_5 and COD tests. TOC is a rapid test for estimating the total

organic carbon in a waste stream. For similar reasons to the rationale for not selecting COD for regulation, EPA did not select TOC for regulation.

While present in treatable concentrations, EPA did not have adequate removal data for molybdenum, nickel, and strontium at the Hazardous subcategory BPT/BAT facilities. However, these metals should be controlled adequately through the regulation of both chromium and zinc. Total phenols is a general, wet chemistry indicator measurement for phenolic compounds and should be controlled by regulating phenol. Similarly, 2,4-dimethylphenol has chemical and treatability characteristics similar to phenol and, therefore, should also be controlled through the regulation of phenol. Hexanoic acid, benzyl alcohol, and isobutyl alcohol are relatively biodegradable and should be controlled by regulating benzoic acid. O-cresol is structurally similar to p-cresol and should be controlled by regulating p-cresol. Tripropyleneglycol methyl ether and diethyl ether have treatability characteristics similar to alpha terpineol in a biological treatment system and should be controlled by regulating alpha terpineol.

In the proposal, EPA chose not to regulate 2-butanone, 2-propanone, 4-methyl-2-pentanone, ethylbenzene, m-xylene, and o+p xylene because they were controlled through the regulation of toluene. After proposal EPA decided not to regulate toluene. The reasons these pollutants were not selected for regulation in the final rule are discussed below.

EPA did not select the following sixteen pollutants of interest for regulation in the Hazardous subcategory because they are present in only trace amounts and/or are not likely to cause toxic effects:

Sixteen Pollutants Not Selected for Regulation in the Hazardous Subcategory Because They Are Present in Only Trace Amounts and/or Are Not Likely to Cause Toxic Effects
Hexane Extractable Material
Nitrate/Nitrite
TDS
2,4-D
2,4-DB

2,4,5-TP
Dicamba
Dichloroprop
MCPA
MCPP
Picloram
Terbutylazine
1,2,3,4,6,7,8-HpCDD
1,2,3,4,6,7,8-HpCDF
OCDD
OCDF

EPA presents the Hazardous subcategory median raw wastewater concentration data for the pollutants of interest in Chapter 6, Table 6-10, and the minimum and maximum concentrations for conventional and nonconventional pollutants, metals, organic pollutants, and dioxins/furans in Tables 6-11 through 6-13, and Table 6-15.

For this industry, nitrate/nitrite is used primarily as a measure of the extent of nitrification that occurs during the biodegradation process. Typically, levels of nitrate/nitrite found in landfill wastewater do not require removal. Removal of nitrate/nitrite can be obtained by specially designed biological treatment systems (such as nitrification/denitrification systems) that are able to complete the conversion of nitrate/nitrite to nitrogen gas. Often, removal of nitrate/nitrite is required to address specific water quality concerns for an individual receiving water (i.e., nutrient problems in the Great Lakes). EPA has, however, determined that the levels of nitrate/nitrite in landfill wastewater do not justify regulation on a national level and individual permit writers can address specific water quality considerations.

TDS is used primarily as a water quality measurement and not as a pollutant that can be controlled through biological treatment. It often is used as a measurement of the salinity of an ambient water or a wastewater and often indicates the presence of naturally occurring salts of metals such as sodium, iron, and magnesium. While it can inhibit biological treatment processes at levels above 10,000 mg/L, acclimated biological

treatment systems can operate successfully with influent TDS concentrations as high as 76,000 mg/L (reference 55). The median concentration of total dissolved solids was 16,000 mg/L for landfills in the Hazardous subcategory. Therefore, EPA has determined that concentrations of total dissolved solids found in landfills in the Hazardous subcategory do not justify regulation. Similarly, hexane extractable material is a general, wet chemistry indicator measurement for oil and grease compounds that generally can be controlled through source reduction and good housekeeping. Therefore, EPA did not select hexane extractable material for regulation.

EPA detected low levels of 2,4-D, 2,4-DB, 2,4,5-TP, dicamba, dichloroprop, MCPA, MCPP, picloram, terbutylazine, 1,2,3,4,6,7,8-HpCDD, 1,2,3,4,6,7,8-HpCDF, OCDD, and OCDF in three out of five of the Hazardous subcategory landfills sampled during EPA's sampling program. At the concentrations found in raw landfill wastewater, EPA expects these pollutants to partition to the biological sludge created as a result of the use of the BPT/BAT treatment technologies. EPA sampling data and calculations conclude that the concentrations of these pollutants present in the untreated wastewater will not prevent the sludge from being redeposited in a hazardous landfill.

EPA did not select the following twenty-one pollutants for regulation in the Hazardous subcategory because they are not controlled by the selected BPT/BAT technology:

Twenty-One Pollutants Not Selected for Regulation in the Hazardous Subcategory Because They Are Not Controlled by the Selected BPT/BAT Technology
Amenable Cyanide
Total Cyanide
1,1-Dichloroethane
1,4-Dioxane
2-Butanone
2-Propanone
4-Methyl-2-Pentanone
Benzene
Ethylbenzene

M-Xylene
Methylene Chloride
O+P Xylene
Toluene
Trans-1,2-Dichloroethene
Trichloroethene
Vinyl Chloride
Copper
Lithium
Selenium
Tin
Titanium

EPA classifies 1,1-dichloroethane, 1,4-dioxane, 2-butanone, 2-propanone, 4-methyl-2-pentanone, benzene, ethylbenzene, m-xylene, methylene chloride, o+p xylene, toluene, trans-1,2-dichloroethene, trichloroethene, and vinyl chloride as "volatile organics" under analytical test method 1624. Because the selected BPT/BAT technology for the Hazardous subcategory is aerated equalization followed by chemical precipitation, biological treatment, and multimedia filtration, EPA determined that the majority of the removal of volatile organic compounds is due to volatilization to the atmosphere in either the aerated equalization tanks or in the activated sludge aeration basin. Therefore, EPA did not regulate volatile organic pollutants because the BPT/BAT technology does not provide controls for removal of these pollutants.

While EPA does not recognize the transfer of pollutants from one media to another as effective treatment, based on the concentrations of these pollutants in untreated wastewater (below treatable levels (10 times the method detection limit)), indications are that the potential increase in air emissions due to this regulation will be minimal.

Volatile organic compounds (VOCs) in hazardous waste landfill leachate are being steadily minimized due to the Resource Conservation and Recovery Act (RCRA) land disposal restriction rules, which typically require aggressive destructive treatment of organics in hazardous wastes before the waste can be landfilled

(see 40 CFR 268.40 and 268.48).[1] VOC levels in historic landfill leachate (from both hazardous and non-hazardous waste landfills dating from the 1930s to the mid-1990s) are also at levels which are low enough as not to call into question EPA's determination to base these rules on the performance of aerated biological systems. Tables 6-9, 6-10, and 6-13 show the concentrations of VOCs found in landfill wastewater.

For the proposed rule, EPA established direct discharge limitations for benzene and toluene for landfills in the Hazardous subcategory. However, after proposal, EPA decided not to regulate benzene and toluene because they are not treated by the chemical or biological treatment technology selected as the basis for the landfills effluent limitations. Furthermore, based on the concentration of benzene (37 ug/L) and toluene (104 ug/L) in untreated leachate, the Agency concluded that the loading of benzene and toluene to the atmosphere will not cause toxic effects.

The Hazardous subcategory median untreated wastewater concentrations for copper, lithium, selenium, tin, and titanium were well below treatable concentrations (10 times the method detection limit). Median untreated wastewater concentrations ranged from 0.02 mg/L to 0.03 mg/L for selenium, copper, and titanium, 0.15 mg/L for tin, and 0.45 mg/L for lithium. While the metals are incidentally removed by the BPT/BAT technology, these concentrations are well below treatable concentrations for conventional metals precipitation technologies.

For total cyanide, the median untreated wastewater concentration for the Hazardous subcategory is 0.08 mg/L, which is well below treatable concentrations for conventional cyanide destruction technologies. While the median raw wastewater concentration of amenable cyanide at hazardous landfills is 1.6 mg/L, EPA concluded that the median untreated wastewater concentration data for total cyanide is more

[1] There are certain exceptions to these treatment requirements for hazardous wastewater which is disposed in surface impoundments. RCRA section 3005 (j) (11). However, if this wastewater contains VOCs above a designated concentration level, then the impoundments are subject to rules requiring control of the resulting air emissions. 40 CFR 264.1085 and 263.1086.

representative than amenable cyanide data of cyanide concentrations in hazardous landfill wastewater because the Agency collected data from ten facilities on total cyanide (one of which was non-detect) and only four facilities (two of which were non-detect) on amenable cyanide.

Based on these factors, the Agency concluded that the five metals plus amenable and total cyanide were present in untreated landfill wastewater at concentrations that were too low to be treated effectively by conventional metals and cyanide treatment technologies (chemical precipitation and chemical oxidation, respectively). Because EPA's BPT/BAT technology does not control these small concentrations of pollutants, the Agency has decided to exclude them from regulation.

In conclusion, the following 13 pollutants of interest will be regulated under BPT/BAT/NSPS in the Hazardous subcategory:

Thirteen Pollutants Selected for Regulation in the Hazardous Subcategory
Ammonia as Nitrogen
BOD_5
TSS
Alpha Terpineol
Aniline
Benzoic Acid
Naphthalene
P-Cresol
Phenol
Pyridine
Arsenic
Chromium
Zinc

Chapter 11 describes in detail the development of the effluent limitations for each of these pollutants.

7.7 Selection of Pollutants to be Regulated for Indirect Dischargers

Section 307(b) of the Clean Water Act (CWA) requires the Agency to promulgate pretreatment standards for existing sources (PSES) and new sources (PSNS). To establish pretreatment standards, EPA must first determine whether each BAT pollutant under consideration is not susceptible to treatment by a POTW, or interferes with the POTW's operation or sludge disposal practices.

7.7.1 Pass-Through Analysis for Indirect Dischargers

The Agency evaluated whether a pollutant is susceptible to treatment at a POTW by comparing removals between direct dischargers and well-operated POTWs for pollutants of interest for both subcategories, listed in Tables 7-1 and 7-2. In comparing removals, the Agency compares the percentage of a pollutant removed by POTWs with the percentage of the pollutant removed by direct discharging facilities applying BAT.

EPA compares removals for two reasons: 1) to ensure that wastewater treatment performance for indirect dischargers is equivalent to that for direct dischargers, and 2) to recognize and take into account the treatment capability and performance of the POTW in regulating the discharge of pollutants from indirect dischargers. Rather than compare the mass or concentration of pollutants discharged by the POTW with the mass or concentration of pollutants discharged by a BAT facility, EPA compares the percentage of the pollutants removed by the BAT treatment system with the POTW removal. EPA takes this approach because a comparison of mass or concentration of pollutants in a POTW effluent to pollutants in a BAT facility's effluent would not take into account the mass of pollutants discharged to the POTW from non-industrial sources, nor the dilution of the pollutants in the POTW effluent to lower concentrations from the addition of large amounts of non-industrial wastewater.

To establish the performance of well-operated POTWs, EPA used the information provided from “Fate of Priority Pollutants in Publicly Owned Treatment Works” (commonly referred to as the 50-POTW Study), supplemented by EPA's National Risk Management Research Laboratory's (NRMRL) treatability

database. EPA used NRMRL's database for those pollutants not found in the 50-POTW study. Chapter 4 discusses these studies in detail.

The 50-POTW Study presents data on 50 well-operated POTWs achieving secondary treatment. For this rulemaking, EPA edited the data in the 50-POTW Study and the data collected for this rule. Because the 50-POTW Study data included influent levels that were close to the detection limit, EPA eliminated these values, thereby minimizing the possibility that low POTW removals might simply reflect low influent concentrations instead of being a true measure of treatment effectiveness. EPA applied the following hierarchal data editing rules to the 50-POTW Study data:

1) Include only detected pollutants having at least three pairs (influent/effluent) of data points.

2) Eliminate average pollutant influent values less than 10 times the minimum analytical detection limit, along with the corresponding effluent values.

3) For analytes where no average influent concentrations were greater than 10 times the minimum level[2], eliminate all average influent values less than five times the minimum level, along with the corresponding effluent values;

4) For analytes where no average influent concentration was greater than five times the minimum level, eliminate all average influent concentrations less than 20 ug/L, along with the corresponding effluent values.

After editing the database, EPA then calculated POTW-specific percent removals for each pollutant based on its average influent and average effluent values. The POTW percent removal used for each pollutant in the pass-through test is the median value of all the POTW specific percent removals for that pollutant. EPA then compared the median POTW percent removal to the median percent removal for the BAT option treatment technology to determine pass through.

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[2] In applying the data editing rules for the 50-POTW Study for the final rule, the minimum level assigned to the non-detect values was the minimum level at the time of the 50-POTW Study (circa 1978-1980). For the proposal, the minimum level assigned to the non-detect values for 50-POTW removals was the Landfills study minimum levels (circa 1994).

The 50-POTW Study did not contain data for all pollutants for which the pass-through analysis was required. Therefore, EPA obtained additional data from EPA's NRMRL Treatability Database . The database provides the user with the specific source and the industry from which the wastewater was generated. EPA used the NRMRL database to augment the POTW database for the pollutants for which the 50-POTW Study did not cover. EPA applied the following data editing rules to the data in the NRMRL database:

1) Only use treatment technologies representative of typical POTW secondary treatment operations (aerobic lagoons, activated sludge, activated sludge with sedimentation and/or filtration).

2) Only use domestic or industrial wastewater data.

3) Use pilot-scale and full-scale data; eliminate bench-scale data.

4) Use data from a paper in a peer-reviewed journal or government report; edit out lesser quality references.

5) Eliminate zero or negative percent removals.

6) For each of the NRMRL sources, EPA first selected data having at least three pairs (influent/effluent) of data points. If no data source contained three pairs of data points, then EPA selected only those facilities having at least two pairs of data points. If none of the data sources contained two pairs of data points, then EPA selected those with one pair (influent/effluent) of data points. EPA applied the paired data editing criteria explained above to the following hierarchy of NRMRL data sources:

    a. NRMRL Treatability data at > 10xMDL – Domestic wastewater.
    b. NRMRL Treatability data at > 5xMDL – Domestic wastewater.
    c. NRMRL Treatability data at >20 ug/L - Domestic wastewater.
    d. NRMRL Treatability data at > 10xMDL – Industrial wastewater.
    e. NRMRL Treatability data at > 5xMDL – Industrial wastewater.
    f. NRMRL Treatability data at >20 ug/L – Industrial wastewater.
    g. NRMRL Treatability data - any available Domestic and/or Industrial data.
    h. Generic pollutant group removal data.

From the NRMRL facilities remaining after applying the above editing criteria, EPA determined the median percent removal for a particular pollutant. The Agency used this median percent removal to represent

POTW removal and compared it to the median percent removal for the BAT option treatment technology in order to determine pass through.

Tables 7-3 and 7-4 present the POTW percent removals for each regulated pollutant in the Non-Hazardous and Hazardous subcategory, respectively. These tables indicate the source of the percent removal and which editing criteria applied.

7.7.2 Non-Hazardous Subcategory Pollutants to be Regulated for Indirect Dischargers

EPA conducted a removal comparison on the priority and nonconventional pollutants regulated under BAT for non-hazardous landfills. EPA did not perform this assessment for the regulated conventional pollutants, namely BOD_5 and TSS, since the conventional pollutants are generally not regulated under PSES and PSNS. For the proposal, EPA evaluated the seven nonconventional and toxic pollutants proposed for regulation under BAT for the Non-Hazardous subcategory, and concluded that ammonia removals were greater at the BAT facilities. Following the proposal, EPA reviewed the data used for the BAT percent removal calculations. In the proposal, EPA calculated the BAT percent removals using data from well-operated biological treatment facilities in EPA's database. However, some of these facilities did not pass the editing criteria for selection as a BPT/BAT facility. In the revised analysis, EPA calculated percent removals using data from only those seven facilities that passed the BPT/BAT editing criteria. In addition, in the proposal, EPA inadvertently failed to use selected BAT facilities in the calculation of percent removals for several pollutants even though the data that met the editing criteria for the facility were available. As a result of this review, the BAT facility removals for the analysis have changed for the Non-Hazardous subcategory since the proposal. Finally, after proposal, EPA decided not to set BPT limits for toluene. Therefore, this pollutant is not considered in the analysis, see Section 7.6.1.

In determining BAT percent removals, EPA used data from selected BAT facilities only if they met the following criteria:

1) The influent concentration for a particular pollutant was greater than 10xMDL,

2) The facility had demonstrated removal of the pollutant (EPA did not use facilities showing zero or negative percent removal), and

3) The facility did not employ treatment technologies in addition to the selected BAT that may contribute to further reduction of the pollutant.

Applying the editing criteria outlined above to those facilities selected as BAT resulted in a different set of facilities being used in the calculation of the percent removals than in proposal for each of the pollutants to be regulated. Table 7-5 lists the BAT facilities used in the calculation of percent removals for the non-hazardous regulated pollutants.

The Agency used EPA sampling episode data, Detailed Questionnaire Section C data and Detailed Monitoring Questionnaire data to calculate the non-hazardous BAT facility percent removals. However, if a particular facility had applicable Detailed Questionnaire Section C and Detailed Monitoring Questionnaire data, EPA used only the Detailed Monitoring Questionnaire data in calculating the BAT percent removals because of a potential overlap of the concentration data submitted for these two questionnaires. EPA used only data with matching influent and effluent data points. The Agency calculated a percent removal for each data source, and then determined an overall median percent removal for each regulated pollutant. Table 7-5 presents the summary of BAT performance data used in calculating the percent removals for the Non-Hazardous subcategory. Table 7-6 presents the results of the removal comparison for the Non-Hazardous subcategory. This table shows the median BAT percent removal and the median POTW percent removal. Although the removal comparison suggests that, at the time of proposal, only ammonia would pass through, as a result of further review of the applicable data contained in the Public Record, the comparison for the final rule suggests that three other pollutants (benzoic acid, p-cresol, and phenol) would pass through in the Non-Hazardous subcategory. However, for the reasons discussed in Chapter 11, EPA is not establishing pretreatment limits for any pollutant in the Non-Hazardous subcategory because it concluded the pollutants which might pass through were, in fact, in most cases susceptible to treatment and that national regulation was not required.

7.7.3 Hazardous Subcategory Pollutants to be Regulated for Indirect Dischargers

EPA conducted removal comparisons for the priority and nonconventional pollutants regulated under BAT for hazardous landfills. EPA did not perform the analysis for the regulated conventional pollutants, namely BOD_5 and TSS, since the conventional pollutants are generally not regulated under PSES and PSNS. For the proposal, EPA performed the analysis on the thirteen nonconventional and toxic pollutants proposed for regulation under BAT for the Hazardous subcategory and determined that seven pollutants appeared to pass through. EPA proposed pretreatment standards for the following six of these pollutants: ammonia as nitrogen, benzoic acid, toluene, alpha terpineol, p-cresol, and aniline. For the proposed rule, EPA used both of the BAT facilities in the calculation of percent removals. However, upon review of the data editing procedures, EPA determined that some of the facility data should not have been used in the calculation of percent removals. As a result of this review, the BAT facility removals for the removal comparison have changed for the Hazardous subcategory since the proposal. Finally, after proposal, EPA decided not to set BPT limits for toluene and benzene; therefore, these pollutants are not considered in the comparison (see Section 7.6.2).

In determining BAT percent removals, EPA used data from selected BAT facilities only if they met the following criteria:

1) The influent concentration for a particular pollutant was greater than 10xMDL,

2) The facility had demonstrated removal of the pollutant (EPA did not use facilities showing zero or negative percent removal), and

3) The facility did not employ treatment technologies in addition to the selected BAT that may contribute to further reduction of the pollutant.

Applying the editing criteria outlined above to those facilities selected as BAT resulted in a different set of facilities being used in the calculation of the percent removals for each of the pollutants to be regulated. Table 7-7 lists the BAT facilities used in the calculation of percent removals for the hazardous regulated pollutants.

The Agency used EPA sampling episode data Detailed Questionnaire Section C data and Detailed Monitoring Questionnaire data to calculate the hazardous BAT facility percent removals. However, if a particular facility had applicable Detailed Questionnaire Section C and Detailed Monitoring Questionnaire data, EPA used only the Detailed Monitoring Questionnaire data in calculating the BAT percent removals because of a potential overlap of the concentration data submitted for these two questionnaires. EPA used only data with matching influent and effluent data points. The Agency calculated a percent removal for each data source, and then determined an overall median percent removal for each regulated pollutant. Table 7-7 presents the summary of BAT performance data used in calculating the percent removals for the Hazardous subcategory. Table 7-8 presents the results of the removal comparison for the Hazardous subcategory. This table shows the median BAT percent removal and the median POTW percent removal. At the time of proposal, the removal comparison suggested better removals at BAT facilities than at POTWs for seven pollutants (ammonia, alpha terpineol, aniline, benzoic acid, p-cresol, phenol, and toluene). As a result of EPA's assessment, the comparison now suggests greater BAT removals for the following eight pollutants: ammonia, alpha terpineol, aniline, benzoic acid, naphthalene, p-cresol, phenol, and pyridine. However, for the reasons discussed in Chapter 11, EPA is not establishing pretreatment limits for any pollutant in the Hazardous subcategory.

Table 7-1: Non-Hazardous Subcategory Pollutants of Interest

Non-Hazardous Pollutant of Interest	Cas #	Subtitle D Municipal Pollutant of Interest	Subtitle D Non-Municipal Pollutant of Interest
Conventional			
BOD	C-002	X	X
TSS	C-009	X	X
Nonconventional			
Ammonia as Nitrogen	7664417	X	X
COD	C-004	X	X
Nitrate/Nitrite	C-005	X	X
TDS	C-010	X	X
TOC	C-012	X	X
Total Phenols	C-020	X	X
Organic			
1,4-Dioxane	123911	X	
2-Butanone	78933	X	
2-Propanone	67641	X	
4-Methyl-2-Pentanone	108101	X	
Alpha Terpineol	98555	X	
Benzoic Acid	65850	X	
Hexanoic Acid	142621	X	
Methylene Chloride	75092	X	
N,N-Dimethylformamide	68122	X	
O-Cresol	95487	X	
P-Cresol	106445	X	
Phenol	108952	X	
Toluene	108883	X	
Tripropyleneglycol Methyl Ether	20324338	X	
Metals			
Barium	7440393	X	
Chromium	7440473	X	
Hexavalent Chromium	18540299	X	
Strontium	7440246	X	X
Titanium	7440326	X	
Zinc	7440666	X	
Pesticides/Herbicides			
Dichloroprop	120365	X	
Disulfoton	298044	X	
Dioxins/Furans			
1234678-HpCDD	35822469	X	
OCDD	3268879	X	

Table 7-2: Hazardous Subcategory Pollutants of Interest

Pollutant of Interest	Cas #	Pollutant of Interest	Cas #
Conventional		**Organics (cont.)**	
BOD	C-002	P-Cresol	106445
Hexane Extractable Material	C-036	Toluene	108883
TSS	C-009	Trans-1,2-Dichloroethene	156605
Nonconventional		Trichloroethene	79016
Amenable Cyanide	C-025	Tripropyleneglycol Methyl Ether	20324338
Ammonia as Nitrogen	7664417	Vinyl Chloride	75014
COD	C-004	**Metals**	
Nitrate/Nitrite	C-005	Arsenic	7440382
TDS	C-010	Chromium	7440473
TOC	C-012	Copper	7440508
Total Phenols	C-020	Lithium	7439932
Organics		Molybdenum	7439987
1,1-Dichloroethane	75343	Nickel	7440020
1,4-Dioxane	123911	Selenium	7782492
2,4-Dimethylphenol	105679	Strontium	7440246
2-Butanone	78933	Tin	7440315
2-Propanone	67641	Titanium	7440326
4-Methyl-2-Pentanone	108101	Total Cyanide	57125
Alpha Terpineol	98555	Zinc	7440666
Aniline	62533	**Pesticides/Herbicides**	
Benzene	71432	2,4,5-TP	93721
Benzoic Acid	65850	2,4-D	94757
Benzyl Alcohol	100516	2,4-DB	94826
Diethyl Ether	60297	Dicamba	1918009
Ethylbenzene	100414	Dichloroprop	120365
Hexanoic Acid	142621	MCPA	94746
Isobutyl Alcohol	78831	MCPP	7085190
Methylene Chloride	75092	Picloram	1918021
M-Xylene	108383	Terbuthylazine	5915413
Naphthalene	91203	**Dioxins/Furans**	
O+P Xylene	136777612	1234678-HpCDD	35822469
O-Cresol	95487	1234678-HpCDF	67562394
Phenol	108952	OCDD	3268879
Pyridine	110861	OCDF	39001020

Table 7-3: Non-Hazardous Subcategory - POTW Percent Removals

Pollutant	MDL (ug/L)	Median % Removal	POTW Percent Removal Source
Ammonia as Nitrogen	10	39	50 POTW 10xMDL
Alpha-Terpineol	10	95	NRMRL 10xMDL - Industrial
Benzoic Acid	50	81	NRMRL 10xMDL - Industrial
P-Cresol	10	68	NRMRL 10xMDL - Domestic & Industrial Sources
Phenol	10	95	50 POTW 10xMDL
Zinc	20	81	50 POTW 10xMDL

Table 7-4: Hazardous Subcategory - POTW Percent Removals

Pollutant	MDL (ug/L)	Median % Removal	POTW Percent Removal Source
Ammonia as Nitrogen	10	39	50 POTW 10xMDL
Alpha-Terpineol	10	95	NRMRL 10xMDL - Industrial
Aniline	10	98	NRMRL 10xMDL - Industrial
Benzoic Acid	50	81	NRMRL 10xMDL - Industrial
Napthalene	10	95	50 POTW 10xMDL
Phenol	10	95	50 POTW 10xMDL
Pyridine	10	95	NRMRL 10xMDL - Industrial
P-Cresol	10	75	NRMRL 10xMDL - Domestic & Industrial Sources
Arsenic	10	66	50 POTW >20 ppb
Chromium	10	82	50 POTW 10xMDL
Zinc	20	81	50 POTW 10xMDL

Table 7-5: Non-Hazardous Subcategory - BAT Performance Data

Pollutants of Interest	Facility /Episode	Avg Inf	Avg Eff	% Removal
Ammonia	16041 (DMQ)	679	5.39	99.21
	16041 (ANL)	475	1.4	99.71
	16122 (ANL)	181	1.14	99.37
	16132 (DMQ)	206	5.9	97.14
				99.29 Median
Alpha Terpineol	16041 (ANL)	653	10	98.47
	16122 (ANL)	123	10	91.87
				95.17 Median
Benzoic Acid	16041 (ANL)	15400	50	99.68
	16122 (ANL)	9300	50	99.46
				99.57 Median
P-Cresol	16041 (ANL)	1360	10	**99.26 Median**
Phenol	16041 (ANL)	5120	10	99.80
	16118 (DET)	350	10	97.14
	16122 (ANL)	395	10	97.47
				97.47 Median
Zinc	16041 (DMQ)	505	214	57.62
	16041 (ANL)	310	87	71.94
	16132 (DMQ)	490	50	89.80
				71.94 Median

All units in ug/L, except ammonia in mg/L.
ANL: EPA sampling episode data
DMQ: Detailed Monitoring Questionnaire data
DET: Detailed Questionnaire data

Table 7-6: Pass-Through Analysis for the Non-Hazardous Subcategory

Pollutant	Average BAT Percent Removal	Average POTW Percent Removal
Ammonia	99%	39%
Alpha Terpineol	95%	95%
Benzoic Acid	99%	81%
P-Cresol	99%	68%
Phenol	97%	95%
Zinc	72%	81%

Table 7-7: Hazardous Subcategory - BAT Performance Data

Pollutants of Interest	Facility /Episode	Avg Inf	Avg Eff	% Removal
Ammonia	16041 (DMQ)	679	5.39	99.21
	16041 (ANL)	475	1.4	99.71
	16122 (ANL)	181	1.14	99.37
	16132 (DMQ)	206	5.9	97.14
				99.29 Median
Alpha-Terpineol	16041 (ANL)	653	10	**98.47 Median**
Aniline	16041 (ANL)	1060	10	99.06
	16087 (ANL)	533	10	98.12
				98.59 Median
Benzoic Acid	16041 (ANL)	15400	50	99.68
	16087 (ANL)	64957	50	99.92
				99.80 Median
Naphthalene	16041 (ANL)	645	10	**98.45 Median**
P-Cresol	16041 (ANL)	1360	10	99.26
	16087 (ANL)	5022	10	99.80
				99.53 Median
Phenol	16041 (ANL)	5120	10	99.80
	16087 (DET)	98500	814	99.17
	16087 (ANL)	65417	31	99.95
				99.80 Median
Pyridine	16087 (ANL)	301	10	**96.68 Median**
Arsenic	16087 (DMQ)	1400	325	76.79
	16087 (ANL)	584	308	47.26
				62.02 Median
Chromium	16041 (DET)	210	120	42.86
	16087 (DMQ)	730	312	57.26
	16087 (ANL)	415	82	80.24
				57.26 Median
Zinc	16041 (DMQ)	505	214	57.62
	16041 (ANL)	310	87	71.94
	16087 (DMQ)	550	380	30.91
				57.62 Median

All units in ug/L, except ammonia in mg/L.
ANL: EPA sampling episode data
DMQ: Detailed Monitoring Questionnaire data
DET: Detailed Questionnaire data

Table 7-8: Pass-Through Analysis for the Hazardous Subcategory

Pollutant	Average BAT Percent Removal	Average POTW Percent Removal
Ammonia	99%	39%
Alpha Terpineol	98%	95%
Aniline	99%	98%
Benzoic Acid	99%	81%
Naphthalene	98%	95%
P-Cresol	99%	68%
Phenol	99%	95%
Pyridine	97%	95%
Arsenic	62%	66%
Chromium	57%	82%
Zinc	58%	81%

Figure 7-1: Development of Pollutants of Interest

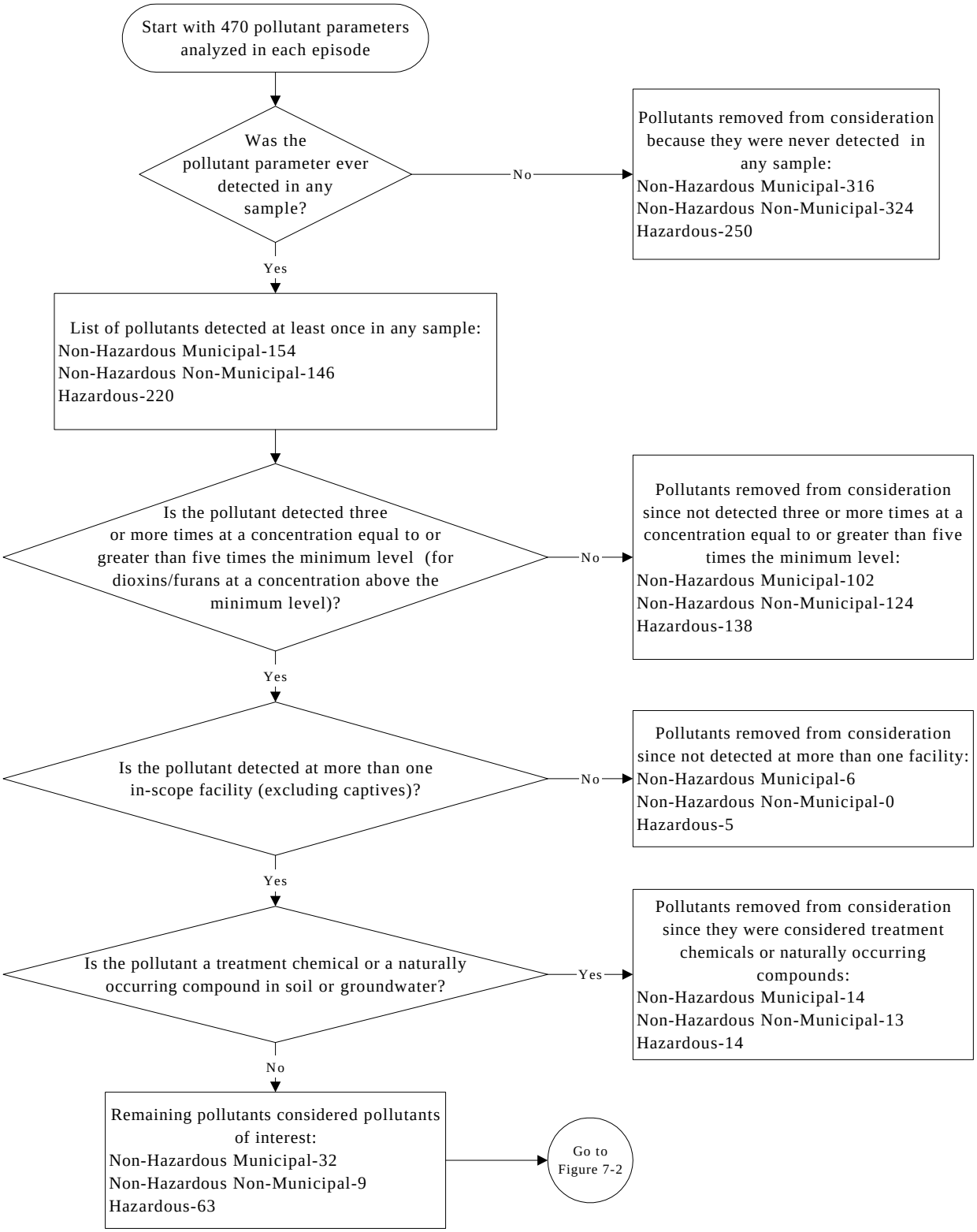
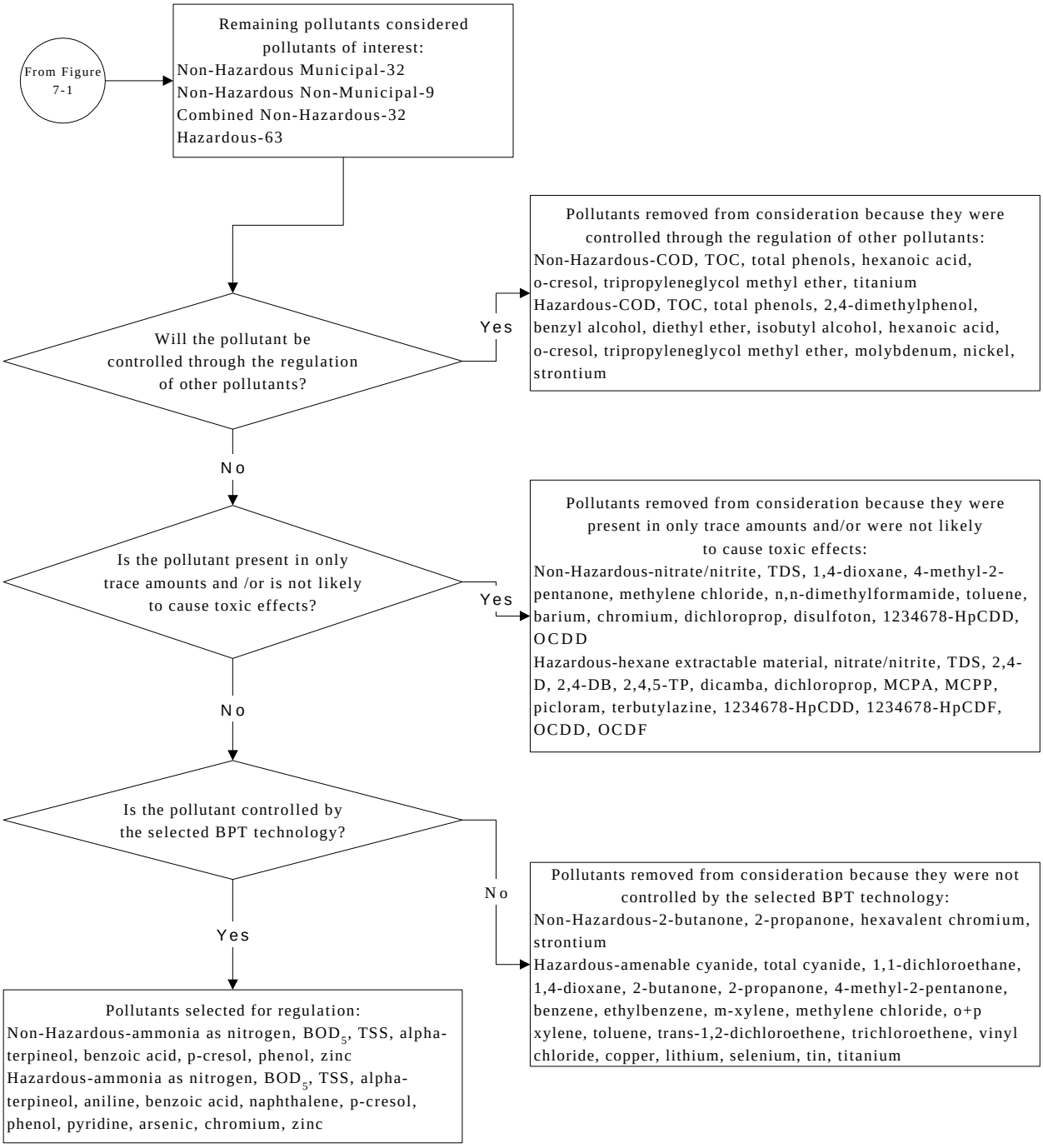


Figure 7-2: Selection of Pollutants to be Regulated



8.0 WASTEWATER TREATMENT TECHNOLOGY DESCRIPTION

This chapter consists of the following two main parts: Section 8.1, describing the wastewater treatment and sludge handling methods currently in use in the Landfills industry and Section 8.2, presenting a discussion on the performance of treatment systems evaluated by EPA using data collected during engineering site visits and field sampling programs.

8.1 Available BAT and PSES Technologies

The Landfills industry uses a wide variety of technologies for treating wastewater discharges. These technologies can be classified into the following five areas:

		Section
C	Best Management Practices	8.1.1
C	Physical/Chemical Treatment	8.1.2
C	Biological Treatment	8.1.3
C	Sludge Handling	8.1.4
C	Zero Discharge options	8.1.5

The EPA's Detailed Questionnaire obtained information on 14 treatment technologies currently in use in the Landfills industry. Table 8-1 presents the technologies most commonly used by in-scope Subtitle D non-hazardous and Subtitle C hazardous landfill facilities by discharge type. The table reports the percent of landfill facilities which use each treatment technology. In addition, EPA collected detailed information on available technologies from engineering site visits to a number of landfill facilities. The data presented below are based on these data collection efforts.

8.1.1 Best Management Practices

Best management practices with regard to wastewater generation at landfills can be designed to do one of

two things: reduce the volume of leachate produced by the landfill or reduce the toxicity of the leachate produced by the landfill. The volume of leachate generated by a landfill is largely dependent on the annual precipitation that falls within the landfill area, percolates through the landfilled waste, and collects in the leachate collection system. State and Resource Conservation and Recovery Act (RCRA) regulations require closed landfills to install an impermeable cap over the landfill to prevent infiltration of rainwater, which will eventually reduce the volume of wastewater produced by the landfill. Open landfills, however, can similarly use methods to reduce rainwater infiltration to the landfill and, hence, reduce wastewater generation. The open face of the landfill is the active area where solid waste is deposited, compacted, and covered with daily fill. This area can act as a collection point for rainwater. By maintaining a small open face on the landfill, along with using impermeable materials on the closed or inactive sections, a landfill operator can reduce the volume of wastewater collected and produced by an open landfill.

The criteria outlined by the Office of Solid Waste and Emergency Response in 40 CFR § 257, 258, 264, and 265 provide additional controls to reduce the volume and/or toxicity of landfill leachate. 40 CFR Part 257 ("Criteria for Classification of Solid Waste Disposal Facilities and Practices") establishes disposal practices for non-municipal, non-hazardous waste disposal units (including waste disposal units that receive conditionally-exempt small quantity generator waste). In Part 257.3-3(c), the regulations state that a facility shall not cause non-point source pollution of waters of the United States that violates the applicable legal requirements implementing an area or Statewide water quality management plan. 40 CFR Part 258 ("Criteria for Municipal Solid Waste Landfills") requires municipal solid waste landfills to design, construct and maintain run-on/run-off control systems (40 CFR 258.26), cover the disposed solid waste with six inches of earthen material at the end of each operating day (40 CFR 258.21), and subject these facilities to closure criteria, which require a final cover to be applied to cover the wastes (40 CFR 258.60). These requirements greatly reduce the risk of storm water becoming contaminated as a result of direct contact with the deposited solid waste. Subpart N of 40 CFR Part 264 ("Standards for Owners and Operators of Hazardous Waste Treatment, Storage, and Disposal Facilities") establishes design and operating requirements for hazardous waste landfills. Hazardous waste landfills must design, construct, operate, and

maintain run-on/run-off control systems (264.301(g)) and, if the landfill contains particulate matter which may be subject to wind dispersal, the operator must cover or otherwise manage the landfill to control wind dispersal (264.301(j)). Subpart N of 40 CFR Part 265, "Interim Status Standards for Owners and Operators of Hazardous Waste Treatment, Storage, and Disposal Facilities," defines similar controls to those identified above for Part 264 for the control of storm water contamination.

In addition, many municipal solid waste landfills and communities have developed programs to prevent toxic materials from being deposited in landfills. Solid waste generated by households may contain many types of waste which may present an environmental hazard, including paints, pesticides, and batteries. Many communities have developed household hazardous waste collection programs which collect and dispose of these hazardous wastes in an appropriate manner, thus avoiding deposition of hazardous wastes in the municipal landfill and reducing the risks associated with the leachate produced by the landfill.

8.1.2 Physical/Chemical Treatment

8.1.2.1 Equalization

Wastewater and leachate generation rates at landfills vary due to their direct relationship to rainfall, storm water run-on and run-off, ground water entering the waste-containing zone, and the moisture content and absorption capability of the wastes. To allow for the equalization of pollutant loadings and flow rates, leachate and other landfill generated wastewater is often collected prior to treatment in tanks or ponds with sufficient capacity to hold the peak flows generated at the facility. A constant flow is delivered to the treatment system from these holding tanks in order to dampen the variation in hydraulic and pollutant loadings to the wastewater treatment system. This reduction in hydraulic and pollutant variability increases the performance and reliability of down stream treatment systems and can reduce the size of subsequent treatment tanks and chemical or polymer feed rates by reducing the maximum flow rates and concentrations of pollutants. Equalization also lowers the operating costs associated with treatment units by reducing instantaneous treatment capacity demand and by optimizing the amount of treatment chemicals required for a less erratic set of treatment variables. National estimates based on EPA's Detailed Questionnaire data

show that 21 percent of direct and 12 percent of indirect non-hazardous landfill facilities use some form of equalization as part of wastewater treatment systems.

Equalization systems consist of steel or fiberglass holding tanks or lined ponds that provide sufficient capacity to contain peak flow conditions. Detention times are determined using a mass balance equation and are dependent on site-specific generation rates and treatment design criteria. According to data collected by EPA's Detailed Questionnaire, detention times can range from less than a day to 90 days, with a median value of about two days. Equalization systems contain either mechanical mixing systems or aeration systems to enhance the equalization process by keeping the tank contents well mixed and prohibiting the settling of solids.

A breakdown of equalization systems used in the Landfills industry based on the responses to the Detailed Questionnaire is as follows:

Equalization Type	% Non-Hazardous Facilities		% Hazardous Facilities
	Direct	Indirect	Indirect
Unstirred	13	7	0
Mechanically Stirred	>1	<1	0
Aerated	11	6	0

A typical equalization system is shown in Figure 8-1.

8.1.2.2 Neutralization

Wastewater generated by landfills may have a wide range of pH depending on the types of waste deposited in the landfill. In many instances, raw wastewater may require neutralization to eliminate either high or low pH values that may upset a treatment system, such as activated sludge biological treatment. However, landfill facilities also use neutralization systems in conjunction with certain chemical treatment processes, such as chemical precipitation, to adjust the pH of the wastewater to optimize process control. Acids, such as sulfuric acid or hydrochloric acid, are added to reduce pH, and alkalies, such as sodium hydroxide,

are added to raise pH values. Neutralization may be performed in a holding tank, rapid mix tank, or an equalization tank. Typically, neutralization systems at the end of a treatment system are designed to control the pH of the discharge to between 6 and 9. National estimates based on EPA's Detailed Questionnaire data show that 33 percent of indirect hazardous landfills, 6 percent of indirect non-hazardous landfills, and 7 percent of direct non-hazardous landfill facilities employ neutralization as part of wastewater treatment systems using a variety of chemical additives to control pH.

Figure 8-2 presents a flow diagram for a typical neutralization system.

8.1.2.3 Flocculation

Flocculation is a treatment technology used to enhance sedimentation or filtration treatment system performance. Flocculation precedes these processes and usually consists of a rapid mix tank, or in-line mixer, and a flocculation tank. The waste stream is initially mixed while a flocculation chemical is added. Flocculants adhere readily to suspended solids and each other to facilitate gravity sedimentation or filtration. Coagulants can be added to reduce the electrostatic surface charges and enhance the formation of complex hydrous oxides. Coagulation allows for the formation of larger, heavier particles, or flocculants (which usually form in a flocculation chamber), that can settle faster. There are three different types of flocculants commonly used: inorganic electrolytes, natural organic polymers, and synthetic polyelectrolytes. The selection of the specific treatment chemical is highly dependent upon the characteristics and chemical properties of the contaminants. A rapid mix tank is usually designed for a detention time from 15 seconds to several minutes (see reference 3). After mixing, the coagulated wastewater flows to a flocculation basin where slow mixing of the waste occurs. The slow mixing allows for the particles to agglomerate into heavier, more settleable solids. Mixing is provided either by mechanical paddle mixers or by diffused air. Flocculation basins are typically designed for a detention time of 15 to 60 minutes (see reference 3). Since many landfill facilities employ gravity-assisted separation and chemical precipitation as part of wastewater treatment systems, EPA assumes that many of these facilities employ flocculation to enhance system performance.

8.1.2.4 Gravity Assisted Separation

Gravity-assisted separation or sedimentation is a simple, economical, and widely used method for the treatment of landfill wastewater. Clarification systems remove suspended matter, flocculated impurities, and precipitates from wastewater. By allowing the wastewater to become quiescent, the suspended matter, which is heavier than water, can settle to the bottom of the clarifier, forming a sludge blanket which can be removed. This process can occur in specially designed tanks, or in earthen ponds and basins. Clarification systems can also be equipped to allow for the removal of materials lighter than water, such as oils, which are skimmed from the surface and collected for disposal. Sedimentation units at landfills are used as either primary treatment options to remove suspended solids or as a secondary treatment option following a biological or chemical precipitation process. Sedimentation processes are highly sensitive to flow fluctuations and, therefore, usually require equalization at facilities with large flow variations.

Clarifiers can be rectangular, square, or circular in shape. In rectangular or square tanks, wastewater flows from one end of the tank to the other with settled sludge collected into a hopper located at one end of the tank. In circular tanks, flow enters from the center and flows towards the outside edge with sludge collected in a center hopper. Treated wastewater exits the clarifier by flowing over a weir located at the top of the clarifier. Sludge which accumulates at the bottom of the clarifier is periodically removed and is typically stabilized and/or dewatered prior to disposal. National estimates based on EPA's Detailed Questionnaire data suggest that 67 percent of indirect hazardous landfills, 9 percent of indirect non-hazardous landfills, and 27 percent of direct non-hazardous landfill facilities employ some form of gravity-assisted separation as part of wastewater treatment systems.

Flocculation systems are commonly used in conjunction with gravity-assisted clarification systems to improve their solids removal efficiency. Some clarifiers are designed with a center well to introduce flocculants and allow for coagulation in order to improve removal efficiencies. A schematic of a typical clarification system using coagulation and flocculation is shown in Figure 8-3. The main design parameters used in designing a clarifier are the overflow rate, detention time, and the side water depth. Overflow rate

is the measure of the flow as a function of the surface area of the clarifier. Typical design parameters used for both primary and secondary clarifiers are presented below (see reference 7):

Design Parameter	Primary	Secondary
Overflow rate, gpd/sq ft	600-1,000	500-700
Detention time, min	90-150	90-150
Minimum Side water depth, ft	8	10

A variation of conventional clarification process is the chemically-assisted clarification process. Coagulants are added to clarifiers to enhance liquid-solid separation, permitting solids denser than water to settle to the bottom and materials less dense than water (including oil and grease) to flow to the surface. Settled solids form a sludge at the bottom of the clarifier which can be pumped out continuously or intermittently. Oil and grease and other floating materials may be skimmed.

Chemically assisted clarification may be used alone or as part of a more complex treatment process. It also may be used in the following capacities:

- The first process applied to wastewater containing high levels of settleable suspended solids.
- The second stage of most biological treatment processes to remove the settleable materials, including microorganisms, from the wastewater; the microorganisms then can be either recycled to the biological reactor or sent to the facility's sludge handling system.
- The final stage of most chemical precipitation (coagulation/flocculation) processes to remove the inorganic flocs from the wastewater.

As discussed in Chapter 9, chemically-assisted clarification was a component of the model wastewater treatment technology for estimating the BPT engineering costs of compliance and applied in certain cases. In developing regulatory compliance costs, EPA used chemically-assisted clarification processes as an additional polishing process after biological treatment. Chemically- assisted clarification processes consist of both a clarifier and a polymer feed system. For facilities currently with sedimentation following biological

treatment, EPA provided additional costs only for a polymer feed system. EPA included chemically-assisted clarification systems in the BPT option to aid in the settling process following biological treatment to enhance both TSS and BOD_5 removals through the wastewater treatment process. Higher BOD_5 removals can be obtained by the additional removal of microbial floc in the clarifier. EPA costed facilities for a chemically-assisted clarification system when their current performance for TSS and/or BOD_5 was slightly out of compliance with regulatory levels (up to 10 mg/L for BOD_5 and 50 mg/L for TSS). For instance, if a facility had an aerobic lagoon treatment system and exceeded the regulatory level for TSS by 20 mg/L, EPA costed the facility for a chemically-assisted clarification system.

Although no landfill facilities in EPA's database reported using chemical addition, chemically-assisted clarification is a proven technology for the removal of BOD_5 and TSS in a variety of industrial categories (see reference 19).

National estimates indicate that less than one percent of direct and indirect non-hazardous landfills use an alternative clarification system design based on corrugated plate interceptor (CPI) technology. These systems include a series of small (approximately two inch square) inclined tubes in the clarification settling zone. The suspended matter must only travel a short distance, when settling or floating, before they reach a surface of the tube. At the tubes' surface, the suspended matter further coagulate. Because of the increased surface area provided by the inclined tubes, CPI units can have much smaller settling chambers than standard clarifiers.

8.1.2.5 Chemical Precipitation

Chemical precipitation is used for the removal of metal compounds from wastewater. In the chemical precipitation process, soluble metallic ions and certain anions found in landfill wastewater are converted to insoluble forms, which precipitate from solution. Most metals are relatively insoluble as hydroxides, sulfides, or carbonates. Coagulation processes are used in conjunction with precipitation to facilitate removal by agglomeration of suspended and colloidal materials. The precipitated metals are subsequently

removed from the wastewater stream by liquid filtration or clarification (or some other form of gravity assisted separation). Other treatment processes such as equalization, chemical oxidation, or reduction (e.g., hexavalent chromium reduction) usually precede the chemical precipitation process. The performance of the chemical precipitation process is affected by chemical interactions, temperature, pH, solubility of waste contaminants, and mixing effects.

Common precipitates used at landfills facilities include lime, sodium hydroxide, soda ash, sodium sulfide, and alum. Other chemicals used in the precipitation process for pH adjustment and/or coagulation include sulfuric and phosphoric acid, ferric chloride, and polyelectrolytes. Often, facilities use a combination of these chemicals. Precipitation using sodium hydroxide or lime is the conventional method of removing metals from wastewater at landfill facilities. Hydroxide precipitation is effective in removing metals such as antimony, arsenic, chromium, copper, lead, mercury, nickel, and zinc. However, sulfide precipitation is used, instead of hydroxide precipitation, to remove specific metal ions such as mercury, lead, and silver. Carbonate precipitation is another method of chemical precipitation and is used primarily to remove antimony and lead. Use of alum as a precipitant/coagulant agent results in the formation of aluminum hydroxides in wastewater containing calcium or magnesium bicarbonate. Aluminum hydroxide is an insoluble gelatinous floc which settles slowly and entraps suspended materials. It is effective for removing metals such as arsenic and cadmium.

Since lime is less expensive than caustic (sodium hydroxide), it is more frequently used at landfill facilities employing hydroxide precipitation. However, lime is more difficult to handle and feed, as it must be slaked, slurried, and mixed and can often plug feed system lines. Lime precipitation also produces a larger volume of sludge. The reaction mechanism for precipitation of a divalent metal using lime is shown below:

$$M^{++} + Ca(OH)_2 \rightarrow M(OH)_2 + Ca^{++}$$

And, the reaction mechanism for precipitation of a divalent metal using sodium hydroxide is:

$$M^{++} + 2NaOH \rightarrow M(OH)_2 + 2Na^{++}$$

In addition to the type of treatment chemical chosen, an important design factor in the chemical precipitation operation is pH. Metal hydroxides are amphoteric, meaning they can react chemically as acids or bases. As such, their solubilities increase toward both lower and higher pH levels. Therefore, there is an optimum pH for precipitation for each metal, which corresponds to its point of minimum solubility. Figure 8-4 presents calculated solubilities of metal hydroxides. For example, as demonstrated on this figure, the optimum pH range where zinc is least soluble is 8 to 10.

Another key consideration in a chemical precipitation application is the detention time in the sedimentation phase of the process. The optimal detention time is dependent on the wastewater being treated and the desired effluent quality.

The first step of a chemical precipitation process is pH adjustment and the addition of coagulants. This process usually takes place in separate mixing and flocculation tanks. After mixing the wastewater with treatment chemicals, the resultant mixture agglomerates in the flocculation tank, and is mixed slowly by either mechanical means, such as mixers or recirculation pumping. The wastewater then undergoes a separation/dewatering process, such as clarification or filtration, where the precipitated metals are removed from solution. In a clarification system, a flocculant, such as a polymer, is sometimes added to aid in the settling process. The resulting sludge from the clarifier or filter must be further treated, disposed, or recycled.

National estimates based on EPA's database indicate that 33 percent of indirect hazardous landfills, 5 percent of indirect non-hazardous landfills, and 9 percent of direct non-hazardous landfill facilities employ chemical precipitation as part of wastewater treatment systems. A typical chemical precipitation system is presented in Figure 8-5.

8.1.2.5.1 Iron (Fe) Coprecipitation

One cost-effective approach to remove metals is the iron adsorption and coprecipitation process. This process involves adding an iron salt, such as ferric chloride or ferric sulfate, to wastewater (unless it already contains sufficient quantities of dissolved iron) to form iron hydroxide precipitate [$Fe(OH)_3(s)$]. Above a pH of 4, the formation of this amorphous precipitate occurs rapidly, causing entrapment of many dissolved and suspended forms of various metals. This "sweep floc" results in the formation of a large quantity of solids (sludge) that can be gravity separated in a conventional clarifier (see reference 57).

8.1.2.6 Chemical Oxidation/Reduction

Chemical oxidation treatment processes can be used to remove ammonia, to oxidize cyanide, to reduce the concentration of residual organics, and to reduce the bacterial and viral content of wastewater. Both chlorine and ozone are two chemicals that are commonly used to destroy residual organics in wastewater. When these chemicals are used for this purpose, disinfection of the wastewater is usually an added benefit. A further benefit of using ozone is the removal of color. Ozone can also be combined with hydrogen peroxide to remove organic compounds in contaminated ground water. Another use of oxidation is for the conversion of pollutants to end products or to intermediate products that are more readily biodegradable or removed more readily by adsorption. National estimates based on the Detailed Questionnaire data show that 33 percent of indirect hazardous landfills, 11 percent of direct non-hazardous landfills, and less than one percent of indirect non-hazardous landfill facilities use chemical oxidation units as part of wastewater treatment systems.

Chemical oxidation is a chemical reaction process in which one or more electrons are transferred from the chemical being oxidized to the chemical initiating the transfer (the oxidizing agent). The electron acceptor may be another element, including an oxygen molecule, or it may be a chemical species containing oxygen, such as hydrogen peroxide, chlorine dioxide (see Section 8.1.2.6.1), permanganate, or ozone. This process is also effective in destroying cyanide and toxic organic compounds. Figure 8-6 presents a process schematic for a chemical oxidation system that uses an alkaline chlorination process.

Chemical oxidation is a potential treatment option for the removal of certain organic pollutants from leachate or ground water. The amount of oxidant required in practice is generally greater than the theoretical mass calculated. The reasons for this are numerous and include incomplete oxidant consumption and oxidant demand caused by other species in solution. Oxidation reactions are catalysts and pH dependent; hence, pH control is an important design variable. For many facilities utilizing chemical oxidation, partial oxidation of organics, followed by additional treatment options, may be more efficient and cost effective than using a complete oxidation treatment scheme alone.

According to the Detailed Questionnaire data, landfill facilities use chemical oxidation processes to treat cyanide-bearing wastes and organic pollutants and as a disinfectant. When treating cyanide or organic wastes, these processes use strong oxidizing chemicals, such as chlorine in elemental or hypochlorite salt form. As a disinfection process, an oxidant (usually chlorine) is added to the wastewater in the form of either chlorine dioxide or sodium hypochlorite (see Section 8.1.2.6.1). Other disinfectant chemicals include ozone, hydrogen peroxide, sulfur dioxide, and calcium hypochlorite. Once the oxidant is mixed with the wastewater, sufficient detention time (usually 30 minutes) is allowed for the disinfecting reactions to occur (see reference 7).

Chemical reduction processes involve a chemical reaction in which electrons are transferred from one chemical to another to reduce the chemical state of a contaminant. The main application of chemical reduction in leachate treatment is the reduction of hexavalent chromium to trivalent chromium. Chromium reduction is necessary due to the inability of hexavalent chromium to form a hydroxide, and enables the trivalent chromium to be precipitated from solution in conjunction with other metallic salts. Figure 8-7 presents a flow diagram of a chromium reduction system. Sulfur dioxide, sodium bisulfate, sodium metabisulfate, and ferrous sulfate are typical reducing agents used at landfill facilities.

8.1.2.6.1 Breakpoint Chlorination

Breakpoint chlorination, in wide use as a wastewater treatment technology, is a physical-chemical means

of removing ammonia from wastewater. Chlorine is added to process waters until the chlorine demand of the wastewater has been satisfied. At this point, the total dissolved residual chlorine has reached a minimum (the breakpoint) and the ammonia has been oxidized to form nitrogen gas and hydrochloric acid. EPA evaluated breakpoint chlorination as an alternative to biological treatment for removing ammonia at landfill facilities with low BOD concentrations. EPA concluded that these facilities may have difficulty operating biological treatment systems due to the low organic content of the wastewater.

The most common chlorine compounds used in wastewater treatment plants are chlorine gas (Cl_2), calcium hypochlorite [$Ca(OCl)_2$], sodium hypochlorite (NaOCl), and chlorine dioxide (ClO_2). Calcium and sodium hypochlorite are most often used in very small treatment plants, such as package plants, where simplicity and safety are far more important than cost. Sodium hypochlorite is often used at large facilities, primarily for reasons of safety as influenced by local conditions. Because chlorine dioxide does not react with ammonia, it is also used in a number of treatment facilities where interferences with ammonia are a concern.

The maintenance of a chlorine residual for the purpose of wastewater disinfection is complicated by the fact that free chlorine not only reacts with ammonia, but is also a strong oxidizing agent. As chlorine is added, readily oxidizable substances, such as Fe^{+2}, Mn^{+2}, H_2S, and organic matter, react with the chlorine and reduce most of it to the chloride ion. After meeting this immediate demand, the chlorine continues to react with the ammonia to form chloramines. Additional chlorine will cause some of the chloramines to be converted to nitrogen trichloride (NCl_3), the remaining will be oxidized to nitrous oxide (N_2O) and nitrogen (N_2), and the chlorine will be reduced to the chloride ion. With continued addition of chlorine, most of the chloramines will be oxidized at the breakpoint. Continued addition of chlorine past the breakpoint will result in a directly proportional increase in the free available chlorine (unreacted hypochlorite). The main reason for adding enough chlorine to obtain a free chlorine residual is that usually disinfection can then be ensured (see reference 56).

8.1.2.7 Air Stripping

Stripping is an effective treatment method for removing dissolved volatile organic compounds from wastewater. The removal is accomplished by passing air or steam through the agitated waste stream. The process results in a contaminated off-gas stream which, depending upon the air emissions standards, usually requires air pollution control equipment. National estimates based on EPA's Detailed Questionnaire data indicate that 4 percent of direct and approximately one percent of indirect non-hazardous landfill facilities use air stripping as part of wastewater treatment systems.

The driving force of air stripping mass-transfer operation is the difference in concentrations between the air and liquid streams. Pollutants are transferred from the more concentrated wastewater stream to the less concentrated air stream until equilibrium is reached. This equilibrium relationship is defined by Henry's Law. The strippability of a pollutant is expressed as its Henry's Law Constant, which is a function of its volatility and solubility.

Air stripping (or steam stripping) can be performed in tanks or in spray or packed towers. Treatment in packed towers is the most efficient application. The packing typically consists of plastic rings or saddles. The two types of towers that are commonly used, cross-flow and countercurrent, differ in design only in the location of the air inlets. In the cross-flow tower, the air is drawn through the sides for the total length of the packing. The countercurrent tower draws its entire air flow from the bottom. The cross-flow towers have been found to be more susceptible to scaling problems and are less efficient than countercurrent towers.

Figure 8-8 presents a flow diagram of a countercurrent air stripper.

8.1.2.8 Filtration

Filtration is a method for separating solid particles from a fluid through the use of a porous medium. The driving force in filtration is a pressure gradient caused by gravity, centrifugal force, or a vacuum. Filtration

treatment processes can be used at landfills to remove solids from wastewater after physical/chemical or biological treatment or as the primary source of leachate treatment. Filtration processes include a broad range of media and membrane separation technologies from ultrafiltration to reverse osmosis. To aid in removal, the filter medium may be precoated with a filtration aid such as ground cellulose or diatomaceous earth.

National estimates based on the Detailed Questionnaire data indicate that 11 percent of direct and two percent of indirect non-hazardous landfill facilities have some form of filtration as part of wastewater treatment systems, including the following:

Type of Filtration System	% Non-Hazardous Facilities	
	Direct	Indirect
Sand	6	<1
Diatomaceous earth	0	<1
Granular multimedia	6	<1
Membrane	0	1
Fabric	0	<1

Dissolved compounds in landfill wastewater are sometimes pretreated to convert the compound to an insoluble solid particle prior to filtration. Polymers are sometimes injected into the filter feed piping downstream of feed pumps to enhance flocculation of smaller flocs that may escape an upstream clarifier. Pretreatment for iron and calcium is sometimes necessary to prevent fouling and scaling.

The following sections discuss the various types of filtration in use at landfills facilities.

8.1.2.8.1 Sand Filtration

Sand filtration processes consist of either a fixed or moving bed of media that traps and removes suspended solids from water passing through the media. There are two types of fixed sand bed filters: pressure and gravity. Pressure filters contain media in an enclosed, watertight pressure vessel and require a feed pump

to force the water through the media. A gravity filter operates on the basis of differential pressure of a static head of water above the media, which causes flow through the filter. Filter loading rates for sand filters are typically between 2 to 6 gpm/sq ft (see reference 7).

All fixed media filters have influent and effluent distribution systems consisting of pipes and fittings. Strainers in the tank bottom are usually stainless steel screens. Layers of uniformly sized gravel also serve as bottom strainers and as a support for the sand. For both types of filters, the bed builds up head loss over time. Head loss is a measure of solids trapped in the filter. As the filter becomes filled with trapped solids, the efficiency of the filtration process falls off, and the filter must be backwashed. Filters are backwashed by reversing the flow so that the solids in the media are dislodged and can exit the filter; sometimes air is dispersed into the sand bed to scour the media.

Fixed-bed filters can be automatically backwashed when the differential pressure exceeds a preset limit or when a timer starts the backwash cycle. Powered valves and a backwash pump are activated and controlled by adjustable cam timers or electronic programmable-logic controllers to perform the backwash function. A supply of clean backwash water is required. Backwash water and trapped particles are commonly discharged to an equalization tank upstream of the wastewater treatment system's primary clarifier or screen for removal.

Moving bed filters use an air lift pump and draft tube to recirculate sand from the filter bottom to the top of the filter vessel, which is usually open at the top. Dirty water entering the filter at the bottom must travel upward, countercurrently, through the downward moving fluidized sand bed. Particles are strained from the rising water and carried downward with the sand. Due to the difference in specific gravity, the lighter particles are removed from the filter when the sand is recycled through a separation box at the top of the filter or in a remote location. The heavier sand falls back into the filter, while the lighter particles flow over a weir to waste. Moving bed filters are continuously backwashed and have a constant rate of effluent flow.

8.1.2.8.2 Diatomaceous Earth

These filtration systems use diatomaceous earth, a natural substance, as a precoat on either a vacuum or pressure filter arrangement to enhance removal efficiencies. In these instances, the diatomaceous earth is placed as a thin layer over a screen. The wastewater then is passed through the layer of earth and screen, with the suspended particles being filtered. A vacuum can be drawn across the screen, or pressure applied to the wastewater to help the liquid pass through the filter medium.

8.1.2.8.3 Multimedia Filtration

Multimedia, or granular bed, filtration is used for achieving supplemental removal of residual suspended solids from the effluent of chemical or biological treatment processes. These filters can be operated either by gravity or under pressure in a vessel. In granular-bed filtration, the wastewater stream is sent through a bed containing one or more layers of different granular materials. The solids are retained in the voids between the media particles while the wastewater passes through the bed. Typical media used in granular-bed filters include anthracite coal, sand, and garnet. These media can be used alone, such as in sand filtration, or in a multimedia combination. Multimedia filters are designed such that the individual layers of media remain fairly discrete. This is accomplished by selecting appropriate filter loading rates, media grain size, and bed density. Hydraulic loading rates for a multimedia filter are between 4 to 10 gpm/sq ft (see reference 7).

A multimedia filter operates with the finer, denser media at the bottom and the coarser, less dense media at the top. A common arrangement is garnet at the bottom of the bed, sand in the middle, and anthracite coal at the top. Some mixing of these layers occurs. During filtration, the removal of the suspended solids is accomplished by a complex process involving one or more mechanisms, such as straining, sedimentation, interception, impaction, and adsorption. The medium size is the principal characteristic that affects the filtration operation. If the medium is too small, much of the driving force will be wasted in overcoming the frictional resistance of the filter bed. If the medium is too large, small particles will travel through the bed, preventing optimum filtration.

The flow pattern of multimedia filters is usually top-to-bottom. Upflow filters, horizontal filters, and biflow filters are also used. A top-to-bottom multimedia filter is represented in Figure 8-9.

8.1.2.8.4 Membrane Filtration

Membrane filtration systems employ a semi-permeable membrane and a pressure differential. Both ultrafiltration and reverse osmosis are commonly used membrane filtration processes.

8.1.2.8.4.1 Ultrafiltration

Ultrafiltration uses a semipermeable microporous membrane, through which the wastewater is passed under pressure. Water and low molecular weight solutes, such as salts and surfactants, pass through the membrane and are removed as permeate. Emulsified oils and suspended solids are rejected by the membrane and removed with some of the wastewater as a concentrated liquid. The concentrate is recirculated through the membrane unit until the flow of permeate drops. The permeate can either be discharged or passed along to another treatment unit. The concentrate is contained and held for further treatment or disposal. Several types of ultrafiltration membranes configurations are available: tubular, spiral wound, hollow fiber, and plate-and-frame. A typical ultrafiltration system is presented in Figure 8-10.

Ultrafiltration is commonly used for the treatment of metal-bearing and oily wastewater. It can remove substances with molecular weights greater than 500, including suspended solids, oil and grease, large organic molecules, and complexed heavy metals (see reference 8). Ultrafiltration is used when the solute molecules are greater than ten times the size of the solvent molecules and less than one-half micron. The primary design consideration in ultrafiltration is the membrane selection. A membrane pore size is chosen based on the size of the contaminant particles targeted for removal. Other design parameters to be considered are the solids concentration, viscosity, and temperature of the feed stream, and the membrane permeability and thickness.

8.1.2.8.4.2 Reverse Osmosis

Reverse osmosis is a separation process that uses selective semipermeable membranes to remove dissolved solids, such as metal salts, from water. The membranes are more permeable to water than to contaminants or impurities. The wastewater is forced through the membrane at an applied pressure that exceeds the osmotic pressure caused by the dissolved solids. Molecules of water pass through the membrane as permeate while contaminants are flushed along the surface of the membrane and exit as concentrate. The concentrate flow from a reverse osmosis system ranges from 10 to 50 percent of the feed flow, with concentrations of dissolved solids and contaminants approaching 10 times that of the feed water (see reference 6). The percentage of permeate that passes through the membranes is a function of operating pressure, membrane type, and concentration of the contaminants in the feed.

Cellulose acetate, aromatic polyamide, and thin-film composites are commonly used membrane materials. Reverse osmosis membranes are configured into tubular, spiral wound, hollow fiber, or plate-and-frame modules. Modules are inserted into long pressure vessels that can hold one or more modules. Reverse osmosis systems consist of a pretreatment pump, a high pressure feed pump, one or more pressure vessels, controls, and instrumentation. A tubular reverse osmosis module is shown in Figure 8-11.

Membranes have a limited life depending upon application and are replaced when cleaning is no longer effective. Membranes can be cleaned manually or chemically by recirculating the cleaning solution through the membranes to restore performance. Membranes can also be removed from the reverse osmosis system and sent off site for flushing and rejuvenation. Membranes are replaced when cleaning is no longer effective.

Membrane pore sizes for a typical reverse osmosis system range from 0.0005 to 0.002 microns, while pressures of 300 to 400 psi are usually required (see reference 39). Therefore, reverse osmosis feed-water needs to be very low in turbidity. Pretreatment of landfill wastewater prior to reverse osmosis treatment may be necessary, including chemical addition and clarification, or cartridge filtration using 5 micron filters

to remove suspended particulates from the influent in order to protect pumps and membranes. Carbon adsorption is recommended as pretreatment for membranes sensitive to chlorine. Biofouling can be prevented by chlorination and dechlorination of the feed water. To maintain the solubility of metals such as calcium, magnesium, and iron, the pH can be adjusted with acid. Aside from pH adjustment, chemical requirements include the following: bactericide, dechlorination, and chelating agents.

One variation of conventional reverse osmosis technology used at landfill facilities is an innovative membrane separation technology using disc tube modules. This innovative process is designed to treat liquid waste that is higher in dissolved solids content, turbidity, and contaminant levels than waste treated by conventional membrane separation processes. This process also reduces the potential for membrane fouling and scaling, allowing it to be the primary treatment for waste streams such as landfill leachate.

The disc tube membrane module features larger feed-flow channels and a higher feed-flow velocity than typical membrane separation systems (see reference 48). These characteristics allow the disc tube module greater tolerance for dissolved solids and turbidity and a greater resistance to membrane fouling and scaling. The high flow velocity, short feed-water path across each membrane, and the circuitous flow path create turbulent mixing reducing boundary layer effects, and minimizing membrane fouling and scaling.

Membrane material for the disc tube module is formed into a cushion with a porous spacer material on the inside. The membrane cushions are alternately stacked with hydraulic discs on a tension rod. The hydraulic disks support the membranes and provide the flow channels for the feed liquid to pass over the membranes. After passing through the membrane material, permeate flows through collection channels to a product recovery tank. A stack of cushions and disks is housed in a pressure vessel. The number of disks per module, number of modules, and the membrane materials can be varied to suit the application. Modules are typically combined in a treatment unit or stage. Disc tube module units can be connected in series to improve permeate water quality or in parallel to increase system treatment capacity (see reference 48).

Like all membrane separation processes, reverse osmosis technology reduces the volume of the waste. The degree of volume reduction is dependent on the waste characteristics and the system design. Reverse osmosis technology can treat liquid waste streams containing low molecular weight volatile and semivolatile organics, metals, and other inorganic compounds.

8.1.2.8.5 Fabric Filters

Fabric filters consist of a vessel that contains a cloth or paper barrier through which the wastewater must pass. The suspended matter is screened by the fabric and the effectiveness of the filter depends on the mesh size of the fabric. Fabric filters can either be backwashed or built as disposable units.

For waters having less than 10 mg/L suspended solids, cartridge fabric filters may be cost effective. Cartridge filters have very low capital cost and can remove particles of 1 micron or larger (see reference 39). Using two-stage cartridge filters (coarse and fine) in series extends the life of the fine cartridge. Disposable or backwashable bag filters also are available and may be quite cost effective for certain applications. Typically, these fabric filters are used to remove suspended solids prior to other filtration systems to protect membranes and equipment and reduce solids fouling.

8.1.2.9 Carbon Adsorption

Activated-carbon adsorption is a physical separation process in which organic and inorganic materials are removed from wastewater by sorption, or attraction, and accumulation of the compounds on the surface of the carbon granules. This process is commonly referred to as granular activated carbon adsorption. While the primary removal mechanism is adsorption, biological degradation and filtration are additional pollutant removal mechanisms provided by the activated- carbon filter. Adsorption capacities of 0.5 to 10 percent by weight are typical in industrial applications (see reference 5). Spent carbon can either be regenerated on site, by processes such as wet-air oxidation or steam stripping, or, for smaller operations, be regenerated off site or sent directly for disposal. Vendors of carbon can exchange spent carbon with fresh carbon under contract.

Activated-carbon systems consist of a vessel containing a bed of carbon (usually 4 to 12 feet in depth), whereby the wastewater is either passed upflow or downflow through the filter bed (see reference 6). Carbon vessels are typically operated under pressure, though some designs use gravity beds. For smaller applications, granular activated carbon systems also are available in canister systems, which can be readily changed-out and sent for off-site regeneration.

Often more than one carbon vessel is used in series, such that the first column can be used until the carbon is "exhausted" before it is regenerated. The partially-exhausted second column is then used as the first column and another column is rotated behind it to provide polishing. Up to three columns are typically used in a rotating fashion. When all of the available adsorption sites on the granular activated carbon are occupied, a rise in organic concentrations is observed in the effluent leaving the vessel. At this point the granular activated carbon in the vessel is saturated and is said to have reached break-through.

The key design parameter is the adsorption capacity of the granular activated carbon. This is a measure of the mass of contaminant adsorbed per unit mass of carbon and is a function of the chemical compounds being removed, type of carbon used, and process and operating conditions. The volume of carbon required is based upon the COD and/or pollutant-specific concentrations in the wastewater to be treated and desired frequency of carbon change-outs. The vessel is typically designed for an empty bed contact time of 15 to 60 minutes (see reference 5). Non-polar, high molecular weight organics with low solubility are readily adsorbed using GAC. Certain organic compounds have a competitive advantage for adsorption onto GAC, which results in compounds being preferentially adsorbed or causing other less competitive compounds to be desorbed from the GAC. Most organic compounds and some metals typically found in landfill leachate are effectively removed using GAC.

National estimates based on EPA's Detailed Questionnaire data indicate that greater than one percent of indirect and greater than one percent of direct non-hazardous landfill facilities employ carbon adsorption

as part of wastewater treatment systems. Figure 8-12 presents a flow diagram of a typical carbon adsorption vessel.

8.1.2.10 Ion Exchange

Ion exchange is an adsorption process that uses a resin media to remove contaminants from wastewater. Ion exchange is commonly used for the removal of heavy metals from relatively low-concentration waste streams. A key advantage of the ion exchange process is that it allows for the recovery and reuse of the metals in a wastewater. Ion exchange also can be designed to be selective to certain metals and can provide effective removal from wastewater having high concentrations of background compounds such as iron, magnesium, and calcium. A disadvantage is that the resins can be fouled by oils and heavy polymers. Pretreatment for ground water or leachate treated by an ion exchange system typically includes a cartridge filtration unit. Additional tanks and pumps are required for regeneration, chemical feed, and collection of spent solution.

In an ion exchange system, the wastewater stream is passed through a bed of resin. The resin contains bound groups of ionic charge on its surface, which are exchanged for ions of the same charge in the wastewater. Resins are classified by type, either cationic or anionic. The selection of a resin is dependent upon the wastewater contaminant to be removed. Cation resins adsorb metals, while anion resins adsorb such contaminants as nitrate and sulfate. A commonly-used resin is polystyrene copolymerized with divinylbenzene. Key parameters for designing an ion-exchange system include a resin bed loading rate of 2 to 4 gallons per minute per cubic foot, and a pressure vessel diameter providing for a cross-sectional area loading rate of 5 to 8 gallons per minute per square foot (see reference 5).

The ion exchange process involves the following four steps: treatment, backwash, regeneration, and rinse. During the treatment step, wastewater is passed through the resin bed. The ion exchange process continues until pollutant breakthrough occurs. The resin is then backwashed to clean the bed and to remove suspended solids. During the regeneration step, the resin is contacted with either an acidic or alkaline

solution containing the ion originally present in the resin. This "reverses" the ion exchange process and removes the ions that were originally present in the wastewater and were retained by the resin. The bed is then rinsed to remove residual regenerating solution. The resulting contaminated regenerating solution must be further processed for reuse or disposal. Depending upon system size and economics, some facilities choose to remove the spent resin and replace it with resin regenerated off-site instead of regenerating the resin in-place.

Ion exchange equipment ranges from simple, inexpensive systems such as domestic water softeners, to large, continuous industrial applications. A common industrial setup is fixed-bed resin in a vertical column, where the resin is regenerated in-place. Other operating modes include batch and fluidized bed. These systems can be designed so that the regenerant flow is concurrent or countercurrent to the treatment flow. A countercurrent design, although more complex to operate, provides a higher treatment efficiency. The beds can contain a single type of resin for selective treatment, or the beds can be mixed to provide for more complete deionization of the waste stream. Often, individual beds containing different resins are arranged in series, which makes regeneration easier than in the mixed bed system.

National estimates based on the Detailed Questionnaire data show that less than one percent of indirect non-hazardous landfills employ some form of ion exchange as part of wastewater treatment systems. Figure 8-13 presents a flow diagram of a typical ion exchange setup, fixed-bed resin in a vertical column.

8.1.3 Biological Treatment

Biological treatment uses microbes which consume, and thereby destroy, organic compounds as a food source. Leachate from landfills can contain large quantities of organic materials that can be readily stabilized using biological treatment processes. In addition to the carbon food source supplied by the organic pollutants, the microbes also require energy and supplemental nutrients for growth, such as nitrogen and phosphorus. There are several different classes of microbes that are commonly used in the biological treatment of organic bearing wastes. Aerobic microbes require oxygen to grow, whereas anaerobic

microbes grow in the absence of oxygen. An adaptive type of anaerobic microbe, called a facultative anaerobe, can grow with or without oxygen.

The success of biological treatment in treating wastewater is dependent on several factors, such as the pH and temperature of the wastewater, the nature of the pollutants, the nutrient requirements of the microbes, the presence of other inhibiting pollutants (such as toxic heavy metals), and variations in the feed stream loading.

Aerobic biological treatment systems utilize an acclimated community of microorganisms to degrade, coagulate, and remove organic and other contaminants from wastewater. Organic contaminants in the wastewater are used by the treatment organisms for biological synthesis and growth, with a small portion for cellular maintenance. Resulting products from biological treatment include cellular biomass, carbon dioxide, water and, sometimes, the nondegradable fraction of the organic material.

In the biological treatment process, wastewater is mixed or introduced to the biomass. The microorganisms responsible for stabilization can be maintained in suspended form or can be attached to a solid media. Examples of the suspended growth biological treatment systems include various activated sludge treatment processes and aerobic lagoons. Biological treatment processes which employ the use of fixed film media include trickling filtration, biotowers, and rotating biological contactors.

Anaerobic biological treatment systems can degrade organic matter in wastewater and ultimately convert carbonaceous material into methane and carbon dioxide. Anaerobic systems have been shown to be most effective for high strength leachate (COD over 4,000 mg/L) and for wastewater containing refractory contaminants because of effectiveness of methanotropic microorganisms in metabolizing these compounds. A disadvantage to anaerobic treatment systems is the sensitivity of the methanotropic microorganisms to certain toxic substances.

Initially, in an anaerobic treatment process, the complex organic matter in the raw waste stream is converted to soluble organics by extra-cellular enzymes. This step facilitates the later conversion of soluble organic matter into simple organic acids. The final step involves the conversion of organic acids into methane and carbon dioxide. The bacteria responsible for the conversions have very slow growth rates. In addition, methanotropic bacteria are very sensitive to environmental conditions, require the complete absence of oxygen, a narrow pH range (6.5 to 7.5), and can be readily inhibited by the presence of toxic compounds such as certain heavy metals.

The table below presents EPA's estimated number of landfill facilities that use variations of biological treatment as part of landfill wastewater treatment systems:

Type of Biological Treatment	% Non-Hazardous Facilities		% Hazardous Facilities
	Direct	Indirect	Indirect
Activated Sludge	8	1	33
Aerobic Lagoon Systems	7	3	0
Facultative Lagoons	7	<1	0
Trickling Filters	0	0	0
Anaerobic Systems	2	<1	0
Powdered Activated Carbon Treatment (PACT)*	>1	<1	0
* with Activated Sludge			
Nitrification Systems	2	<1	0
Rotating Biological Contactors (RBCs)	0	0	0
Sequencing Batch Reactors (SBRs)	>1	0	33
Denitrification Systems	>1	0	0
Other[+]	13	0	0
[+] includes aerated submerged fixed film and wetlands			

The following sections present a discussion of biological treatment systems in use at landfill facilities.

8.1.3.1 Lagoon Systems

A lagoon, stabilization pond, or oxidation pond is a body of water contained in an earthen dike and designed for biological treatment. While in the lagoon, wastewater is treated to reduce degradable organics through biodegradation and reduce suspended solids through sedimentation. The biological process taking

place in the lagoon can be aerobic, anaerobic, or both (facultative), depending on the design. Because of the low construction and operating costs, lagoons offer a financial advantage over other treatment methods and are popular where sufficient land is available at reasonable cost.

Lagoons are used in wastewater treatment for stabilization of suspended, dissolved, and colloidal organics either as a main biological treatment process or as a polishing treatment process following other biological treatment systems. Aerobic, facultative, and aerated lagoons are generally used for wastewater of low and medium organic strength. High-strength wastewater and wastewater of variable strength often are treated by a series of lagoons. A common configuration is an anaerobic lagoon, followed by a facultative lagoon and an aerobic lagoon.

The performance of lagoons in removing degradable organics depends on detention time, temperature, and the nature of the waste. Aerated lagoons generally provide a high degree of BOD_5 reduction more consistently than aerobic or facultative lagoons. Typical problems associated with lagoons are excessive algae growth, offensive odors from anaerobic lagoons if sulfates are present and the lagoon is not covered, and seasonal variations in effluent quality. The major classes of lagoons that are based on the nature of biological activities are discussed below.

Aerobic lagoons depend on algae photosynthesis and natural aeration to assist in the biological activity. These shallow lagoons (3 to 4 feet in depth) rely on both the natural oxygen transfer occurring through the surface area of the lagoon and the production of oxygen from photosynthetic algae. Aerobic lagoons are generally suitable for treating low- to medium-strength landfill leachates due to the recommended smaller food to mass ratios. Because of this design limitation, aerobic lagoons are used in combination with other lagoons to treat higher-strength landfill leachates to achieve additional organic removal following conventional wastewater treatment processes. The typical hydraulic detention time for an aerobic lagoon is 10 to 40 days, with an organic loading of 60 to 120 pounds of BOD_5 per day per acre (see reference 7).

A variation of the aerobic lagoon is the aerated lagoon. These lagoons do not depend on algae and sunlight to furnish dissolved oxygen, but require additional oxygen to be introduced to prevent anaerobic conditions. In these systems, mechanical or diffused aeration devices are used in the lagoons for oxygen transfer and to create some degree of mixing (see Figure 8-14). Due to this mixing, additional suspended solids removal in the effluent from the lagoon may be required. The recommended hydraulic detention time is 3 to 20 days, with an organic loading of 20 to 400 pounds of BOD_5 per day per acre (see reference 7). Based on these higher design loading rates, aerated lagoons are well suited for treatment of medium-strength landfill leachates.

Aerated lagoons are relatively simple to operate. The influent is fed into the basin where it is mixed and aerated with the lagoon contents. Settled sludge is not routinely withdrawn from the lagoon. Lagoons require only periodic cleanings when the settled solids significantly reduce lagoon volume. Since operation requires no sludge recycle, the hydraulic detention time is equal to the sludge retention time. Contaminant reduction in a lagoon system is typically less than other biological treatment systems. As a result, aerobic lagoons are commonly used together with other physical/chemical treatment processes, such as lime addition and settling, to ensure sufficient pollutant removal efficiencies.

Anaerobic lagoons are relatively deep ponds (up to 6 meters) with steep sidewalls in which anaerobic conditions are maintained by keeping organic loading so high that complete deoxygenation is prevalent. Some oxygenation is possible in a shallow surface zone. If floating materials in the waste form an impervious surface layer, complete anaerobic conditions will develop. Treatment or stabilization results from anaerobic digestion of organic wastes by acid-forming bacteria that break down organics. The resultant acids are then converted to carbon dioxide, methane, and other end products. Anaerobic lagoons are capable of providing treatment of high-strength wastewater and are resistant to shock loads.

In the typical anaerobic lagoon, raw wastewater enters near the bottom of the pond (often at the center) and mixes with the active microbial mass in the sludge blanket, which can be as much as 2 meters (6 feet)

deep. The discharge is located near one of the sides of the pond, submerged below the liquid surface. Excess sludge is washed out with the effluent and recirculation of waste sludge is not required.

Anaerobic lagoons are customarily contained within earthen dikes. Depending on soil and wastewater characteristics, lining with various impervious materials, such as rubber, plastic, or clay may be necessary. Pond geometry may vary, but surface area-to-volume ratios are minimized to enhance heat retention.

Waste stabilization in a facultative lagoon treatment system is accomplished by a combination of anaerobic microorganisms, aerobic microorganisms, and a preponderance of facultative microorganisms that thrive under anaerobic as well as aerobic conditions. Facultative systems consist of lagoons of intermediate depth (3 to 8 feet) in which the wastewater is stratified into three zones (see Figure 8-15). These zones consist of an anaerobic bottom layer, an aerobic surface layer, and an intermediate zone dominated by the facultative microorganisms. Stratification is a result of solids settling and temperature-water density variations. Oxygen in the surface zone is provided by natural oxygen transfer and photosynthesis or, as in the case of an aerated facultative lagoon, by mechanical aerators or diffusers. Facultative lagoons usually consist of earthen dikes, but some are lined with various impervious materials, such as synthetic geomembranes or clay.

A facultative lagoon is designed to permit the accumulation of settleable solids on the basin bottom. This sludge at the bottom of the facultative lagoon will undergo anaerobic digestion, producing carbon dioxide and methane. The liquid and gaseous intermediate products from the accumulated solids, together with the dissolved solids furnished in the influent, provide the food for the aerobic and facultative bacteria in the upper layers of the liquid in the lagoon. Recommended hydraulic detention time for a facultative lagoon without aeration is 7 to 30 days, with an organic loading of 15 to 50 pounds of BOD_5 per day per acre (see reference 7).

8.1.3.2 Anaerobic Systems

Types of anaerobic biological treatment systems include complex mix anaerobic digestors (see Figure 8-16), contact reactors with sludge recycle, and anaerobic filters. A digestor uses an air tight reactor where wastes are mixed with digestor contents that contain the suspended anaerobic microorganisms. A digestor operated in a complete mix mode without sludge recycling has a hydraulic detention time equal to the solids retention time. Anaerobic digestion in a reactor can also occur with sludge recycling. This permits a much larger solids retention time (SRT) than the hydraulic detention time. System stability is greater at increased SRTs, and since the hydraulic detention time can be decreased, the reactor volume can also be reduced. The anaerobic filter or biotower microbes are maintained in a film on packed solid media within an air-tight column. A variation of the anaerobic fixed-film process is a fluidized bed process. The basic tower design is similar to that of an aerobic reactor in that the influent is fed into the reactor at countercurrent flow. This process provides for very high SRTs and variable hydraulic detention times.

Stabilization of leachate in an anaerobic treatment unit requires the maintenance of a viable community of anaerobic microbes. Treatment efficiency is dependent on many interrelated factors such as hydraulic detention time, SRT, temperature, and, to a lesser extent, organic loading, nutrients, and toxics. Microorganisms responsible for degrading the organic waste must remain in the reactor long enough to reproduce. When the microbes spend less time in the system than they require to reproduce, the solids are eventually washed out of the system. Anaerobic treatment facilities are typically designed with an SRT of 2 to 10 times the washout time (typical washout time reported for organic acids is about 3.5 days). For degradation of organic acids in leachate, this washout time would yield an SRT of 7 to 35 days (see reference 7). The most common temperature regime for an anaerobic reactor is in the range of 25 to 38 degrees C (see reference 7). Typical loadings for anaerobic systems are from 30 to 100 pounds of COD per 1,000 cubic feet of reactor volume (see reference 7). Since the synthesis of new cellular material is slow in anaerobic systems, nutrient requirements are not as large as in aerobic systems. Nutrient addition needs to be evaluated and, in the case of leachate with low phosphorus concentrations, will require phosphorus addition.

8.1.3.3 Attached-Growth Biological Treatment Systems

Attached-growth biological treatment systems are used to biodegrade the organic components of a wastewater. In these systems, the biomass adheres to the surfaces of rigid supporting media. As wastewater contacts the supporting medium, a thin-film biological slime develops and coats the surfaces. As this film (consisting primarily of bacteria, protozoa, and fungi) grows, the slime periodically breaks off the medium and is replaced by new growth. This phenomenon of losing the slime layer is called sloughing and is primarily a function of organic and hydraulic loadings on the system. The effluent from the system is usually discharged to a clarifier to settle and remove the agglomerated solids.

Attached-growth biological systems are applicable to industrial wastewater amenable to aerobic biological treatment in conjunction with suitable pre- and post-treatment units. These systems are effective for the removal of suspended or colloidal materials.

The three major types of attached-growth systems used at landfills facilities are rotating biological contactors, trickling filters, and fluidized-bed biological reactors. These processes are described below.

Rotating biological contactors are a form of aerobic attached-growth biological treatment system where the biomass adheres to the surface of a rigid media. In a rotating biological contactor, the rigid media usually consists of a plastic disk or corrugated plastic medium mounted on a horizontal shaft (see Figure 8-17). The medium slowly rotates in wastewater (with 40 to 50 percent of its surface immersed) as the wastewater flows past. During the rotation, the medium picks up a thin layer of wastewater, which flows over its surface absorbing oxygen from the air. The biological mass growing on the medium surface absorbs organic pollutants, which then are biodegraded. Excess microorganisms and other solids are continuously removed from the film on the disk by shearing forces created by the rotation of the disk in the wastewater. The sloughed solids are carried with the effluent to a clarifier, where they are separated from the treated effluent.

Rotating biological contactors provide a greater degree of flexibility for landfills with changing leachate characteristics. Modular construction of rotating biological contactors permit their multiple staging to meet increases or decreases in treatment demand. Staging, which employs a number of rotating biological contactors operated in series, enhances biological treatment efficiency, improves shock-handling ability, and also may aid in achieving nitrification.

Typical rotating biological contactor design parameters include a hydraulic loading of 2.0 to 4.0 gallons per square feet per day and an organic loading of 2.0 to 3.5 pounds BOD_5 per 1,000 square feet per day (see reference 12).

Factors which affect the efficiency of rotating biological contactor systems include the type and concentration of organic matter, hydraulic detention time, rotational speed, media surface area submergence, and pre- and post-treatment activities. Variations of the basic rotating biological contactor process design include the addition of air to the tanks, chemicals for pH control, use of molded covers or housing for temperature control, and sludge recycle to enhance nitrification. Rotating biological contactors are typically well suited for the treatment of soluble organics and adequate for nitrification. They are low-rate systems capable of handling limited loadings capacity and are not efficient for degrading refractory compounds or removing metals (see reference 7).

Trickling filtration is another aerobic fixed-film biological treatment process that consists of a suitable structure, packed with inert medium, such as rock, wood, or plastic. The wastewater is distributed over the upper surface of the medium by either a fixed spray nozzle system or a rotating distribution system (see Figure 8-18). The inert medium develops a biological slime that absorbs and biodegrades organic pollutants. Air flows through the filter by convection, thereby providing the oxygen needed to maintain aerobic conditions.

Trickling filters are classified as low-rate or high-rate, depending on the organic loading. Typical design organic loading values range from 5 to 25 pounds and 25 to 45 pounds BOD_5 per 1,000 cubic feet per day for low-rate and high-rate, respectively (see reference 11). A low-rate filter generally has a media bed depth of 1.5 to 3 meters and does not use recirculation. A high-rate filter can have a bed depth from 1 to 9 meters and recirculates a portion of the effluent for further treatment (see reference 7).

A variation of a trickling filtration process is the aerobic biotower which can be operated in a continuous or semi-continuous manner. Influent is pumped to the top of a tower, where it flows by gravity through the tower. The tower is packed with media, plastic or redwood, containing the microbial growth. Biological degradation occurs as the wastewater passes over the media. Treated wastewater collects into the bottom of the tower. If needed, additional oxygen is provided via air blowers countercurrent to the wastewater flow. Alternative variations of this treatment process involve the inoculation of the raw influent with bacteria, adding nutrients, and using upflow biotowers. Wastewater collected in the biotowers is delivered to a clarifier to separate the biological solids from the treated effluent.

An aerobic fluidized-bed biological reactor is a variation of a fixed-film biological treatment process. Microorganisms are grown on either granular activated carbon or sand media. Influent wastewater enters the reactor through a distributor which is designed to provide for fluidization of the media (see Figure 8-19). As the biofilm grows, the media bed expands, thereby reducing the density of the media. The rising bed is intercepted at a given height with the bulk of the biomass removed from the media. The media then is returned to the reactor. Additional oxygen can be predissolved in the influent to enhance performance. The use of granular activated carbon as a medium integrates biological treatment and carbon adsorption processes, which has the advantage of handling loading fluctuations, as well as greater removals of organic contaminants.

Due to a short hydraulic detention time, this process is favorable for low to moderate levels of contamination. The vertical installation of the reactor and high loading capability reduces conventional land

requirements. The maximum design loading is 400 pounds of BOD per 1,000 square feet of reactor area per day with a minimum hydraulic detention time of 5 to 10 minutes (see reference 7).

8.1.3.4 Activated Sludge

The activated sludge process is a specific continuous-flow, aerobic biological treatment process that employs suspended-growth aerobic microorganisms to biodegrade organic contaminants. In this process (shown in Figure 8-20), a suspension of aerobic microorganisms is maintained in a relatively homogeneous state by mechanical mixing or turbulence induced by diffused aerators in an aeration basin. This suspension of microorganisms is called the mixed liquor.

Wastewater is introduced into the basin and mixed with the tank contents. The biological process often is preceded by gravity settling to remove larger and heavier suspended solids. A series of biochemical reactions take place in the aeration tank. These reactions degrade organics and generate new biomass. Microorganisms oxidize the soluble and suspended organic pollutants to carbon dioxide and water using the available supplied oxygen. These organisms also agglomerate colloidal and particulate solids. After a specific contact period in the aeration basin, the mixture is passed to a settling tank where the microorganisms are separated from the treated water. A portion of the settled solids in the clarifier is recycled back to the aeration system to maintain the desired concentration of microorganisms in the reactor. The remainder of the settled solids is wasted and sent to sludge handling facilities.

To ensure biological stabilization of organic compounds in activated sludge systems, adequate nutrient levels must be available to the biomass. The primary nutrients are nitrogen and phosphorus. Lack of these nutrients can impair biological activity and result in reduced removal efficiencies. Certain leachates can have low concentrations of nitrogen and phosphorus relative to the oxygen demand. As a result, nutrient supplements (e.g., phosphoric acid addition for additional phosphorus) have been used in activated sludge systems at landfill facilities.

The effectiveness of the activated sludge process is governed by several design and operation variables. The key variables are organic loading, sludge retention time, hydraulic or aeration detention time, oxygen requirements, and the biokinetic rate constant (K). The organic loading is described as the food-to-microorganism (F/M) ratio, or kilograms of BOD_5 applied daily to the system per kilogram of mixed liquor suspended solids (MLSS). The MLSS in the aeration tank is determined by the rate and concentration of activated sludge returned to the tank. The organic loading (F/M ratio) affects the BOD_5 removal, oxygen requirements, biomass production, and the settleability of the biomass. The sludge (or solids) retention time (SRT) or sludge age is a measure of the average retention time of solids in the activated sludge system. Sludge retention time is important in the operating of an activated sludge system because it must be maintained at a level that is greater than the maximum generation time of microorganisms in the system. If adequate sludge retention time is not maintained, the bacteria are washed from the system faster than they can reproduce and the process fails. The SRT also affects the degree of treatment and production of waste sludge. A high SRT results in carrying a high quantity of solids in the system, obtaining a higher degree of treatment, and producing less waste sludge. The hydraulic detention time is used to determine the size of the aeration tank and should be determined by use of F/M ratio, SRT, and MLSS. The biokinetic rate constant (or K-rate) determines the speed of the biochemical oxygen demand reaction and generally ranges from 0.1 to 0.5 $days^{-1}$ for municipal wastewater (see reference 11). The value of K for any given organic compound is temperature-dependent. Because microorganisms are more active at higher temperatures, the value of K increases with increasing temperature. Oxygen requirements are based on the amount of oxygen required for BOD_5 synthesis and the amount required for endogenous respiration. The design parameters will also vary with the type of wastewater to be treated. The oxygen requirement to satisfy BOD_5 synthesis is established by the characteristics of the wastewater. The oxygen requirement to satisfy endogenous respiration is determined by the total solids maintained in the system and their characteristics.

Modifications of the activated sludge process are common, as the process is extremely versatile and can be adapted for a wide variety of organically contaminated wastewater. The typical modification may represent a variation in one or more of the key design parameters, including the F/M loading, aeration

location and type, sludge return, and contact basin configuration. The modifications in practice have been identified by the major characteristics that distinguish the particular configuration. The characteristic types and modifications are briefly described as follows:

- Conventional. The aeration tanks are long and narrow, with plug flow (i.e., little forward or backwards mixing).

- Complete Mix. The aeration tanks are shorter and wider, and the aerators, diffusers, and entry points of the influent and return sludge are arranged so that the wastewater mixes completely.

- Tapered Aeration. A modification of the conventional process in which the diffusers are arranged to supply more air to the influent end of the tank, where the oxygen demand is highest.

- Step Aeration. A modification of the conventional process in which the wastewater is introduced to the aeration tank at several points, lowering the peak oxygen demand.

- High Rate Activated Sludge. A modification of conventional or tapered aeration in which the aeration times are shorter, the pollutants loadings are higher per unit mass of microorganisms in the tank. The rate of BOD_5 removal for this process is higher than that of conventional activated sludge processes, but the total BOD_5 removals are lower.

- Pure Oxygen. An activated sludge variation in which pure oxygen instead of air is added to the aeration tanks. The tanks are covered, and the oxygen-containing off-gas is recycled. Compared to normal air aeration, pure oxygen aeration requires a smaller aeration tank volume and treats high-strength wastewater and widely fluctuating organic loadings more efficiently.

- Extended Aeration. A variation of complete mix in which low organic loadings and long aeration times permit more complete wastewater degradation and partial aerobic digestion of the microorganisms.

- Contact Stabilization. An activated sludge modification using two aeration stages. In the first stage, wastewater is aerated with the return sludge in the contact tank for 30 to 90 minutes, allowing finely suspended colloidal and dissolved organics to absorb to the activated sludge. The solids are settled out in a clarifier and then aerated in the sludge aeration (stabilization) tank for 3 to 6 hours before flowing into the first aeration tank (see reference 11).

- Oxidation Ditch Activated Sludge. An extended aeration process in which aeration and mixing are provided by brush rotors placed across a race-track-shaped basin. Waste enters the ditch at one end, is aerated by the rotors, and circulates.

Activated sludge systems are effective in the removal of soluble (dissolved) organics by biosorption as well as suspended and colloidal matter typically found in landfill leachate. Suspended matter is removed by entrapment in the biological floc while colloidal matter is removed by physiochemical adsorption to the biological floc. For example, inorganic contaminants, such as heavy metals, that are common in low concentrations in landfill wastewater are often precipitated and concentrated in the biological sludges generated from activated sludge systems at landfill facilities. Halogenated organic compounds may be driven off to a certain extent in the aeration process while other less volatile compounds are removed by a combination of biodegradation and air stripping in the aeration basin. Finally, activated sludge systems treating landfill leachates with an excess loading of certain nutrients (i.e. amounts of nitrogen that exceed the requirements of the biomass in the activated sludge system) can be operated so that nitrification of ammonia can occur in the activated sludge system. For higher concentrations, stand-alone nitrification systems may be required; these systems are discussed later in this chapter.

Conventional, plug-flow activated sludge systems can adequately treat the organic loadings found in low- to medium-strength landfill leachates. Higher-strength leachates often are treated at landfill facilities using extended aeration mode of activated sludge treatment. This process allows for a large hydraulic detention time of up to 29 hours and for a sludge detention time of 20 to 30 days (see reference 7). Aerator loading for the complete-mix extended-aeration process is between 10 to 15 pounds of BOD_5 per 1,000 cubic feet of aerator tank volume (see reference 7). Extended aeration also provides for minimal operator supervision in comparison to other activated sludge processes and occasional sludge wasting. EPA sampled a facility (EPA sampling episode 4759) in the Hazardous subcategory that employed a complete-mix extended-aeration treatment process for high-strength leachate. Design parameters for this system include influent BOD_5 loading of 3520 mg/L with a hydraulic detention time of 28 hours. Higher-strength leachates are also occasionally treated with a combination of biological processes, sometimes using a lagoon or attached growth system prior to the activated sludge system to reduce organic loading. Since activated sludge systems are sensitive to the loading and flow variations typically found at landfill facilities, equalization is often required prior to treatment using activated sludge systems. Also, activated sludge

systems treating landfill leachates typically generate excess amounts of secondary sludge that may require additional stabilization, dewatering, and disposal.

8.1.3.5 Powdered Activated Carbon Biological Treatment

In this biophysical treatment process, powdered activated carbon is added to a biological treatment system (usually an activated sludge system). The adsorbent qualities of the powdered carbon aid in the removal of organic compounds, particularly those that may be difficult to biodegrade. Powdered activated carbon also enhances color removal and the settling characteristics of the biological floc.

The mixture of influent, activated sludge biomass, and powdered activated carbon is held in the aeration basin for a sufficient detention time adequate for the desired treatment efficiencies (see Figure 8-21). After contact in the aeration basin, the mixture flows to a clarifier, where settled solids are fed back to the aeration basin to maintain adequate concentrations of microorganisms and carbon. Clear overflow from the clarifiers is either further processed or discharged. Fresh carbon is periodically added to the aeration basin as required and is dependent on desired removal efficiencies. Excess solids are removed directly from the recycled sludge stream. Wasted solids can be processed by conventional dewatering means or by wet-air oxidation for the destruction of organics and regeneration of activated carbon. Regeneration also can be handled off site for smaller applications.

Powdered activated carbon activated sludge treatment combines physical adsorption properties of carbon with biological treatment, achieving a higher degree of treatment than possible by either mode alone. Powdered activated carbon removes the more difficult to degrade refractory organics, enhances solids removal, and buffers the system against loading fluctuations and shock loads. Variations of the powdered activated carbon biological process includes operation in a batch fill and draw mode (similar to a sequencing batch reactor), multiple-stage powdered activated carbon units, and combinations of aerobic and anaerobic powdered activated carbon biological systems. Operation in a batch mode provides for flexibility in the system, by readily allowing for adjustments to the time and aeration mode in each process

stage. This mode of operation is particularly applicable to the treatment of leachate with variable composition and strength. The powdered activated carbon biological treatment process is well suited for the treatment of leachate containing high concentrations of soluble organics (particularly with low BOD_5 to COD ratios). It can obtain better color and refractive organics removal than conventional biological processes and can provide for treatment of leachates contaminated with various trace organic compounds.

8.1.3.6 Sequencing Batch Reactors (SBRs)

A sequencing batch reactor is a suspended-growth biological system in which the wastewater is mixed with existing biological floc in an aeration basin. SBRs are unique in that a single tank acts as an equalization tank, an aeration tank, and a clarifier (see Figure 8-22). A SBR is operated on a batch basis where the wastewater is mixed and aerated with the biological floc for a specific period of time. The contents of the basin then are allowed to settle and the liquid (or supernatant) is decanted. The batch operation of a sequencing batch reactor makes it applicable to wastewater that is highly variable because each batch can be treated differently, depending on its waste characteristics.

A sequencing batch reactor system has four cycles: fill, react, settle, and decant. The fill cycle has three phases. The first phase, called static fill, introduces the wastewater to the system under static conditions. During this phase, anaerobic conditions can exist. During the second phase, the wastewater is mixed to eliminate the scum layer and to initiate the oxygenation process. The third phase consists of aeration and biological degradation. The react cycle is a time-dependent process that continually mixes and aerates the wastewater while allowing the biological degradation process to complete. Because the reaction is a batch process, the period of time of aeration can vary to match the characteristics and loadings of the wastewater. The settling cycle utilizes a large surface area (entire reactor area) and a lower settling rate than used in conventional sedimentation processes, to allow for settling under quiescent conditions. Next, during the decant cycle, approximately one-third of the tank volume is removed by subsurface withdrawal. This treated effluent then can be further treated or disposed. The period of time that the reactor waits prior to the commencement of another batch processing is the idle period. Excess biomass is periodically removed

from the sequencing batch reactor when the quantity exceeds that needed for operation and is usually dewatered prior to disposal.

A sequencing batch reactor carries out all of the functions of a conventional continuous-flow activated sludge process, such as equalization, biological treatment, and sedimentation, in a time sequence rather than a space sequence. Detention times and loadings vary with each batch and are highly dependent on the loadings in the raw wastewater at that time. Typically, a sequencing batch reactor operates with a hydraulic detention time of 1 to 10 days with an SRT of 10 to 30 days. The MLSS is maintained at 3,500 to 10,000 mg/L (see reference 7). The overall control of the system can be accomplished automatically by using level sensors or timing devices. By using a single tank to perform all of the required functions associated with biological treatment, a sequencing batch reactor saves on land requirements. It also provides for greater operation flexibility for treating leachate with viable waste characteristics by being able to readily vary detention time and mode of aeration in each stage. Sequencing batch reactors also can be used to achieve complete nitrification/denitrification and phosphorus removal.

8.1.3.7 Nitrification Systems

In this process, nitrifying bacteria are used in an aerobic biological treatment system to convert ammonia compounds to nitrate compounds. Nitrification is usually followed by denitrification (see next section) which converts nitrates to nitrogen gas. Nitrifying bacteria, such as *nitrosomonas* and *nitrobacter*, derive their energy for growth from the oxidation of inorganic nitrogen compounds. *Nitrosomonas* converts ammonia to nitrites, and *nitrobacter* converts nitrites to nitrates.

The nitrification process usually follows a standard biological process that has already greatly reduced the organic content of the wastewater; however, there are some biological systems that can provide organic (BOD_5) removal concurrently with ammonia destruction. The nitrification process can be oriented as either a suspended growth process (e.g. activated sludge system) or an attached-growth process (e.g. trickling filter).

8.1.3.8 Denitrification Systems

Denitrification is an anoxic process whereby nitrate nitrogen is converted to gaseous nitrogen, and possibly nitrous oxide and nitric oxide. Denitrification is a two step process in which the first step converts nitrates to nitrites, and the second step converts nitrite to nitrogen gas. The bacteria use nitrogen as an electron source rather than oxygen in digesting a carbon food source. Since the waste stream reaching the denitrification process has low levels of organic material, a carbon source (usually methanol) must be added.

The denitrification process can occur as a suspended-growth process or as an attached-growth process. Attached growth systems can be designed as either fixed-bed or fluidized-bed reactor systems. Effluents from denitrification processes may need to be re-aerated to meet dissolved oxygen discharge requirements.

8.1.3.9 Wetlands Treatment

An alternative and innovative biological treatment technology for treating landfill wastewater is wetland treatment. Wetlands can either be natural or man-made (artificial) systems and contain vegetation that allow for the natural attenuation of contaminants. Wetlands are designed to provide for a contact time of usually 10 to 30 days. Vegetation in the wetlands transforms nutrients and naturally degrades organics. Certain metals also can be absorbed by vegetation through root systems. Key design variables include loading rates, climatic constraints, and site characteristics. Wetland systems are still mainly experimental and are not a widely accepted or proven treatment technology for the treatment of landfill leachate.

8.1.4 Sludge Handling

Sludges are generated by a number of treatment technologies, including equalization, gravity-assisted separation, chemical precipitation, and biological treatment. These sludges are further processed at landfill sites using various methods. The following sections describe each type of sludge-handling system used within the Landfills industry.

8.1.4.1 Sludge Slurrying

Sludge slurrying is the process of transporting sludge from one treatment process to another. It only can be applied to liquid sludges that can be pumped through a pipe under pressure. National estimates based on EPA's Detailed Questionnaire data indicate that 33 percent of indirect hazardous landfills and less than one percent of indirect non-hazardous landfills use sludge-slurrying systems as part of their wastewater treatment systems.

8.1.4.2 Gravity Thickening

Gravity thickening, as shown in Figure 8-23, consists of placing the sludge in a unit similar to a gravity-assisted separator, where the sludge is allowed to settle, with the liquid supernatant remaining at the top. The thickened sludge is then removed, and the separated liquid is returned to the wastewater treatment system for further treatment. Usually sludges that contain two to three percent solids can be thickened to approximately five to ten percent solids using gravity thickening. National estimates based on the Detailed Questionnaire responses show that 67 percent of indirect hazardous landfills, 4 percent of indirect non-hazardous landfills, and 7 percent of direct non-hazardous landfill facilities employ gravity thickening as part of their wastewater treatment systems.

8.1.4.3 Pressure Filtration

Plate-and-frame pressure-filtration systems are used at landfill facilities to dewater sludges from physical/chemical and biological treatment processes. Sludges generated at a total solids concentration of two to five percent by weight are dewatered to a 30 to 50 percent solids mass using plate-and-frame filtration (see reference 3). Sludges from treatment systems can be thickened by gravity or stabilized prior to dewatering by pressure filtration or may be processed directly with the plate-and-frame filtration unit.

A pressure filter consists of a series of screens (see Figure 8-24) upon which the sludge is applied under pressure. A precoat material may be applied to the screens to aid in solids removal. The applied pressure

forces the liquid through the screen, leaving the solids to accumulate behind the screen. Filtrate which passes through the screen media is recirculated back to the head of the on-site wastewater treatment plant. Screens (also referred to as plates) are held by frames placed side-by-side and held together with a vice-type mechanism. The unit processes sludge until all of the plates are filled with dry sludge as indicated by a marked rise in the application pressure. Afterwards, the vice holding the plates is loosened and the frames separated. Dried sludge is manually scraped from the plates and collected in a hopper for final disposal. The size of the filter and the number of plates utilized depends not only on the amount of solids produced by treatment processes, but also is highly dependent on the desired operational requirements for the filter. A plate-and-frame filter can produce a drier sludge than possible with most other methods of sludge dewatering. It is usually not operated continuously, but offers operational flexibility since it can be operated in a batch mode.

Pressure filtration is the most common method of sludge dewatering used at landfill facilities. National estimates indicate that 67 percent of indirect hazardous landfills, 5 percent of indirect non-hazardous landfills, and 8 percent of direct non-hazardous landfill facilities use pressure filtration systems as part of their wastewater treatment systems.

8.1.4.4 Sludge Drying Beds

Sludge-drying beds are an economical and effective means of dewatering sludge when land is available. Sludge may be conditioned by thickening or stabilization prior to application on the drying beds, which are typically made up of sand and gravel. Sludge is placed on the beds in an 8 to 12 inch layer and allowed to dry. The drying area is partitioned into individual beds, approximately 20 feet wide by 20 to 100 foot long (see reference 13), or a convenient size so that one or two beds will be filled by the sludge discharge from other sludge-handling units or sludge- storage facilities. The outer boundaries may be constructed with concrete or earthen embankments for open beds. Open beds are used where adequate area is available and sufficiently isolated to avoid complaints caused by odors. Covered beds with greenhouse-

type enclosures are used when it is necessary to dewater sludge continuously throughout the year, regardless of the weather, and where sufficient isolation does not exist for the installation of open beds.

Sludge is dried by drainage through the sludge mass and supporting sand and by evaporation from the surface exposed to the air. Most of the water leaves the sludge by drainage; thus, the provision of an adequate underdrainage system is essential. Drying beds are equipped with lateral drainage tiles that should be adequately supported and covered with coarse gravel or crushed stone. The sand layer should be from 9 to 12 inches deep (see reference 13) with an allowance for some loss from cleaning operations. Water drained from the sludge is collected and typically recirculated back to the on-site wastewater treatment system. Sludge can be removed from the drying bed after it has drained and dried sufficiently. The moisture content is approximately 60 percent after 10 to 15 days under favorable conditions (see reference 13). Dried sludge is manually removed from the beds and sent for on-site or off-site disposal. Figure 8-25 depicts the cross section of a typical drying bed.

8.1.5 Zero Discharge Treatment Options

In this section, additional treatment processes and disposal methods associated with zero or alternative discharge at landfill facilities are described. Based on the responses to the Detailed Questionnaire, national estimates indicate that 27 percent of all non-hazardous landfill facilities and 96 percent of all hazardous landfill facilities use zero-discharge treatment options. The most commonly used zero-discharge treatment method employed by these facilities is land application and recirculation. This section describes land application, recirculation, deep-well disposal, evaporation, solidification, and off-site disposal.

Land application involves the spreading of the wastewater over an area of land that is capped, closed, or an unused portion of a landfill. The land generally has sufficient percolation characteristics to allow the water to drain adequately into the soil. The area is assessed to insure that the soil can provide adequate biological activity to cause the degradation of organic pollutants and also to provide sufficient binding of any metals present.

Recirculation involves the spraying of recycled landfill leachate over areas of a landfill. Although this process promotes biodegradation and evaporation of the leachate volume, recirculation is primarily used as a means of dust control.

Deep well disposal consists of pumping the wastewater into a disposal well, which then discharges the liquid into a deep aquifer. Normally, these aquifers are thoroughly characterized to insure that they are not hydrogeologically connected to a drinking-water supply. The characterization requires the confirmation of the existence of impervious layers of rock above and below the aquifer.

Traditionally used as a method of sludge dewatering, evaporation, or solar evaporation, can also involve the discharge and ultimate storage of wastewater into a shallow, lined, on-site ditch. Since the system is open to the atmosphere, the degree of evaporation is greatly dependent upon climatic conditions.

Solidification is a process in which materials, such as fly ash, cements, and lime, are added to the waste to produce a solid. Depending on both the contaminant and binding material, the solidified waste may be disposed of in a landfill.

Some facilities that have a low leachate generation rate (either because of arid conditions or capping), transport their wastewater off site to either another landfill facility's wastewater treatment system or to a Centralized Wastewater Treatment (CWT) facility for ultimate disposal.

8.2 Treatment Performance and Development of Regulatory Options

This section presents an evaluation of performance data on treatment systems collected by EPA during field sampling programs. The results of these EPA sampling episodes assisted the Agency in evaluating the various types of treatment technologies. For those facilities employing the selected technologies, the sampling data were used to develop the effluent limitations. A more detailed discussion of the development of effluent limitations can be found in Chapter 11.

8.2.1 Performance of EPA Sampled Treatment Processes

To collect data on potential BAT treatment technologies, EPA reviewed responses to the Detailed Questionnaire to identify candidate facilities that had well-operated and designed wastewater treatment systems. EPA conducted 19 site visits to 18 facilities to evaluate treatment systems. Based on these site visits, EPA selected a total of six facilities for sampling which consisted of five consecutive days of sampling raw influent wastewater and intermediate and effluent points in the wastewater treatment system. EPA conducted one of these 5-day sampling episodes (4690) at a facility that was eventually excluded from the regulation because it is a captive landfill. In addition, the only technology sampled at this facility primarily treated contaminated ground water. For the reasons discussed in Chapter 2, EPA decided to exclude contaminated ground water flows from this regulation. EPA did not use the data collected during this sampling episode in selection of pollutants of interest or in the calculation of effluent limitations. Therefore, EPA does not discuss this facility further in this section. For the remaining five sampling facilities, EPA collected data on a variety of biological and chemical treatment processes. Technologies evaluated at the selected sampling facilities include hydroxide precipitation, activated sludge, sequencing batch reactors, multimedia filtration, and reverse osmosis. Table 4-2 in Chapter 4**,** presents a summary of the treatment technologies sampled during each EPA sampling episode. Presented below are the summaries of the treatment system performance data for each of the sampling episodes that EPA evaluated in the development of the effluent limitations guidelines and standards.

8.2.1.1 Treatment Performance for Episode 4626

EPA performed a 5-day sampling program during episode 4626 to obtain performance data on several treatment technologies including hydroxide precipitation, biological treatment using anaerobic and aerobic biotowers, and multimedia filtration. A flow diagram of the landfill wastewater treatment system sampled during episode 4626 is presented in Figure 8-26. The wastewater treatment system used at this Subtitle D municipal facility treats predominately landfill generated wastewater, including leachate and gas condensate. Table 8-2 presents a summary of percent removal data collected at episode 4626 for the performance of the biological treatment system and for the entire treatment system, excluding the multimedia

filtration system used to polish the discharge from the effluent holding tank. EPA calculated percent removal efficiencies for the processes by first obtaining an average concentration based upon the daily sampling results for each sample collection location (influent and effluent point to the treatment process). EPA calculated the percent removal efficiency of the system using the following equation:

$$\text{Percent Removal} = \frac{[\text{Influent Concentration} - \text{Effluent Concentration}]}{\text{Influent Concentration}} \times 100$$

EPA reported negative and zero percent removals for a treatment process on the table as 0.0 percent.

EPA determined the treatment efficiency of the biological treatment unit operation using the data obtained from sampling points 04 and 07 (see Figure 8-26). As demonstrated on the Table 8-2, the biological treatment unit experienced good overall removals for TOC (93.0 percent), COD (90.85 percent), and ammonia as nitrogen (99.14 percent). The biological unit operation alone did not demonstrate high removals for BOD_5 (10.2 percent), TSS (9.32 percent), or for various metals (generally less than 10 percent removals) because the pollutants were generally not present in the biological treatment unit influent at treatable levels. The unit's influent BOD_5 was 39.2 mg/L, TSS was 11.8 mg/L, and most metals were not at detectable levels even though the raw wastewater at this facility exhibited a BOD_5 concentration of 991 mg/L, TSS of 532 mg/L, and several metals at treatable levels. The biological treatment unit influent was low because this facility employed large aerated equalization tanks and a chemical precipitation system prior to biological treatment. The equalization tanks had a retention time of approximately 15 days and were followed by a chemical precipitation system using sodium hydroxide. Due to the long retention time and wastewater aeration, significant biological activity occurred in these tanks. The resulting insoluble pollutants were removed in the primary clarifier prior to entering the biological towers. EPA did not detect organic pollutant parameters in the effluent from the biological treatment process with the exception of 1,4-dioxane at a concentration of 13.8 ug/L.

To determine the treatment efficiency of the entire treatment system, EPA determined the influent concentration by taking a flow-weighted average of the two influent sampling points, sampling points 01 and 02. EPA represented the effluent from the treatment system by sample point 07. The entire treatment system experienced good removals for the following conventional and nonconventional pollutants parameters: BOD_5, TSS, ammonia as nitrogen, COD, TOC, and total phenols. Each of the organic pollutant parameters identified in the influent to the treatment system was removed to non-detectable levels, with the exception of 1,4-dioxane, which still experienced a high percent removal (94.2 percent). Most metals had good percent removals or were removed to non-detectable levels.

8.2.1.2 Treatment Performance for Episode 4667

EPA performed a 5-day sampling program during episode 4667 to obtain performance data on various treatment units, including ammonia removal, hydroxide precipitation, biological treatment using a sequencing batch reactor, granular activated-carbon adsorption, and multimedia filtration. A flow diagram of the landfill wastewater treatment system sampled during episode 4667 is presented in Figure 8-27. The wastewater treatment process used at this Subtitle D non-hazardous facility primarily treats landfill generated wastewater and a small amount of sanitary wastewater flow from the on-site maintenance facility. Table 8-3 presents a summary of percent removal data collected during episode 4667 for the biological treatment unit operation (SBR) and for the entire treatment system.

EPA determined the treatment efficiency of the biological treatment unit using the data obtained from sampling points 03 and 04 (see Figure 8-27). As demonstrated on Table 8-3, the SBR treatment unit experienced moderate overall removals for TOC (43.4 percent), COD (24.7 percent), and BOD_5 (48.7 percent). The Agency observed improved removal efficiencies for TSS (82.9 percent), total phenols (74.2 percent), and ammonia as nitrogen (80.7 percent). Metals, such as barium, chromium, and zinc, had low removal efficiencies. However, as also noted for facility 4626, the Agency observed these metals in the influent to the biological system at low concentrations, often close to the detection limit. Other metals also

had poor removal efficiencies including boron and silicon. EPA did not detect organic parameters in the effluent from the SBR treatment unit.

EPA determined the treatment efficiency of the entire treatment system at the facility using the data obtained from sampling points 01 and 06 (see Figure 8-27). Overall the treatment system experienced good removals for BOD_5, TSS, ammonia as nitrogen, COD, TOC and total phenols. Each of the organic pollutants detected in the influent was removed to non-detect levels in the effluent. Also, each of the metal parameters experienced a good removal rate through the treatment system.

8.2.1.3 Treatment Performance for Episode 4721

EPA performed a 5-day sampling program during episode 4721 to obtain performance data on the sequencing batch reactor (SBR) treatment unit operation installed at this Subtitle C hazardous facility. A flow diagram of the landfill wastewater treatment system sampled during episode 4721 is presented in Figure 8-28. The wastewater treatment process used at this facility treats predominately landfill generated wastewater. The majority of the landfill wastewater was generated by Subtitle D non-hazardous landfills. However, the facility also commingled wastewater generated by an on-site hazardous waste landfill for treatment. The facility also treats limited amounts of off-site generated wastewater at the on-site treatment plant, primarily from another landfill facility operated by the same entity. Table 8-4 presents a summary of percent removal data collected during episode 4721 for the SBR treatment unit.

EPA determined the treatment efficiency of the biological treatment unit using the data obtained from sampling points 01 and 02 (see Figure 8-28). As demonstrated on the Table 8-4, the SBR treatment unit experienced good overall removals for a number of convention/nonconventional and organic parameters, including total phenols, BOD_5, aniline, benzoic acid, 2-propanone, 2-butanone, naphthalene, alpha terpineol, ethylbenzene, p-cresol, m-xylene, 4-methyl-2-pentanone, toluene, phenol, hexanoic acid, and ammonia as nitrogen. EPA observed removal of all of the organic parameters detected in the influent to non-detect levels in the effluent. COD and TOC percent removals were observed at 72.2 and 66.3

percent, respectively. The percent removal for TSS was 72.1 percent. Metals with quantitative percent removals include arsenic (61.9 percent), chromium (46.3 percent), copper (61.2 percent), and zinc (66.3 percent).

8.2.1.4 Treatment Performance for Episode 4759

EPA performed a 5-day sampling program during episode 4759 to obtain performance data on various treatment processes installed at this Subtitle C hazardous facility, including chemical precipitation using ferric chloride and sodium hydroxide and biological treatment using an activated sludge process. A flow diagram of the landfill wastewater treatment system sampled during episode 4759 is presented in Figure 8-29. The wastewater treatment process used at this facility treats predominately landfill generated wastewater, but also handles limited amounts of contaminated storm water from storage containment systems. Table 8-5 presents a summary of percent removal data collected at episode 4759 for the biological treatment units only and for the entire treatment system (combined chemical precipitation and biological treatment processes).

EPA determined the treatment efficiency of the biological treatment unit operations using the data obtained from sampling points 02 and 03 (see Figure 8-29). As demonstrated on the Table 8-5, the biological treatment units experienced good overall removals for a number of conventional/nonconventional and organic parameters, including BOD_5, COD, TOC, total phenols, aniline, benzoic acid, 2,4-dimethylphenol, 2-propanone, methylene chloride, 2-butanone, benzyl alcohol, isobutyl alcohol, o-cresol, p-cresol, 4-methyl-2-pentanone, phenol, pyridine, toluene, and hexanoic acid. Most of the organic parameters detected in the influent were removed to non-detect levels in the effluent from the biological treatment units. Most of the metal parameters, such as chromium, copper, selenium, titanium, and zinc, were observed at low concentrations in the influent to the biological treatment units and, therefore, did not demonstrate good removal rates.

EPA determined the treatment efficiency of the entire treatment system at the facility using the data obtained from sampling points 01 and 03 (see Figure 8-29). As demonstrated on Table 8-5, the entire treatment system experienced good overall removals for a number of convention/nonconventional and organic parameters, including total phenols, BOD_5, 2,4-dimethylphenol, aniline, benzene, benzoic acid, 2-propanone, methylene chloride, 2-butanone, benzyl alcohol, isobutyl alcohol, o-cresol, p-cresol, 4-methyl-2-pentanone, phenol, pyridine, toluene, tripropyleneglycol methyl ether, and hexanoic acid. Most of the organic parameters detected in the influent were removed to non-detectable levels in the effluent. COD and TOC percent removals were observed at 76.4 percent and 84.2 percent, respectively. Ammonia as nitrogen and TSS had poor removal rates of 25.7 percent and 26.6 percent, respectively. Metals with quantitative percent removals include arsenic (46.6 percent), chromium (80.2 percent), copper (45.2 percent), strontium (66.8 percent), titanium (89.6 percent), and zinc (62.5 percent). Pesticide/herbicide parameters such as 2,4-DB, dicamba and dichloroprop had good removal efficiencies through the treatment system. Dioxin/furan parameters were not detected in either the influent or effluent samples.

8.2.1.5 Treatment Performance for Episode 4687

EPA performed a 5-day sampling program during episode 4687 to obtain performance data on the reverse osmosis treatment process installed at this Non-Hazardous Subtitle D facility. A flow diagram of the landfill wastewater treatment system sampled during episode 4687 is presented in Figure 8-30. The wastewater treatment process used at this facility treats on-site landfill generated wastewater. Table 8-6 presents a summary of percent removal data collected at episode 4687 for a single-pass reverse osmosis unit including the multimedia filtration unit and the entire treatment system consisting of a second pass reverse osmosis unit.

EPA determined the treatment efficiency of the single-pass reverse osmosis treatment system at the facility using the data obtained from sampling points 01 and 02 (see Figure 8-30). As demonstrated on Table 8-6, the single-pass reverse osmosis treatment system demonstrated good overall removals for a number of conventional/nonconventional and organic parameters, including TSS, TOC, BOD_5, TDS, COD, 4-methyl-

2-pentanone, alpha terpineol, benzoic acid, tripropyleneglycol methyl ether, and hexanoic acid. A number of other organic parameters also were observed to have been removed by the treatment process at various levels lower than 95 percent. Total phenols and ammonia as nitrogen percent removals were observed at 75.1 and 76.7 percent, respectively. Metals with quantitative percent removals include arsenic (87.4 percent), boron (54.1 percent), silicon (88.3 percent), and strontium (92.9 percent). All of the pesticide/herbicide parameters detected in the influent, including 2,4,5-TP, 2,4-D, 2,4-DB, dicamba, dichlorprop, MCPA and MCPP, were removed to non-detect levels.

EPA determined the treatment efficiency of the entire treatment system at the facility using the data obtained from sampling points 01 and 03 (see Figure 8-30). The additional polishing reverse osmosis unit caused the removal efficiency of most of the conventional and nonconventional parameters to increase. These parameters include BOD_5, ammonia as nitrogen, COD, TDS, TOC, and total phenols. The removal efficiency of several organic parameters were observed to increase from the single-pass treatment system including 2-butanone, 2-propanone, phenol, p-cresol, and toluene. The percent removal for boron also increased from 54.1 percent in the single-pass reverse osmosis system to 94.4 percent in the two-stage reverse osmosis treatment system.

Table 8-1: Wastewater Treatment Technologies Employed at In-Scope Landfill Facilities
(Percent of Landfills Industry)

Treatment Technology	Subtitle D Non-Hazardous		Subtitle C Hazardous
	Direct Discharge	Indirect Discharge	Indirect Discharge
Equalization	21.0	11.2	0.0
Neutralization	6.3	6.7	33.3
Chemical oxidation	11.2	0.5	33.3
Chemical precipitation	9.1	5.4	33.3
Adsorption	1.4	1.3	0.0
Filtration	10.5	1.5	0.0
Stripping	4.2	1.3	0.0
Biological treatment	32.2	3.8	66.7
Gravity assisted separation	27.3	9.0	66.7
Sludge preparation	3.5	0.5	33.3
Sludge dewatering	12.6	5.2	66.7

Table 8-2: Treatment Technology Performance for Facility 4626 - Subtitle D Municipal

Pollutant of Interest Subtitle D Municipal	CAS #		Biological Treatment Unit Operation Only: Sample Points 4 to 7					Entire Treatment System: Sample Points 1 & 2 (flow weighted) to 7					
				Influent		Effluent	%			Influent		Effluent	%
		DL	SP	Conc. (ug/L)	SP	Conc. (ug/L)	Removal	DL	SP	Conc. (ug/L)	SP	Conc. (ug/L)	Removal
Conventional													
BOD	C-002	2,000	04	39,200	07	35,200	10.2	2,000	1+2	991,067	07	35,200	96.5
TSS	C-009	4,000	04	11,800	07	10,700	9.3	4,000	1+2	532,800	07	10,700	98.0
Nonconventional													
Ammonia as Nitrogen	7664417	10.0	04	135,000	07	1,156	99.1	10.0	1+2	193,333	07	1,156	99.4
COD	C-004	5,000	04	1,742,600	07	159,400	90.9	5,000	1+2	4,028,000	07	159,400	96.0
Hexavalent Chromium	18540299	10.0	04	ND	07	ND		10.0	1+2	68.7	07	ND	85.4
Nitrate/Nitrite	C-005	50.0	04	1,535	07	130,500	0.0	50.0	1+2	693	07	130,500	0.0
TDS	C-010		04	5,960,000	07	5,181,000	13.1		1+2	5,012,667	07	5,181,000	0.0
TOC	C-012	1,000	04	758,000	07	52,800	93.0	1,000	1+2	1,316,200	07	52,800	96.0
Total Phenols	C-020	50.0	04	182	07	50.0	72.5	50.0	1+2	1,204	07	50.0	95.9
Organics													
1,4-Dioxane	123911	10.0	04	NS	07	13.8	NS	10.0	1+2	240	07	13.8	94.2
2-Butanone	78933	50.0	04	NS	07	ND	NS	50.0	1+2	227,893	07	ND	100
2-Propanone	67641	50.0	04	NS	07	ND	NS	50.0	1+2	27,655	07	ND	99.8
4-Methyl-2-Pentanone	108101	50.0	04	NS	07	ND	NS	50.0	1+2	598	07	ND	91.6
Alpha Terpineol	98555	10.0	04	NS	07	ND	NS	10.0	1+2	134	07	ND	92.6
Benzoic Acid	65850	50.0	04	NS	07	ND	NS	50.0	1+2	14,657	07	ND	99.7
Hexanoic Acid	142621	10.0	04	NS	07	ND	NS	10.0	1+2	36,256	07	ND	100
Methylene Chloride	75092	10.0	04	NS	07	ND	NS	10.0	1+2	50.3	07	ND	80.1
N,N-Dimethylformamide	68122	10.0	04	NS	07	ND	NS	10.0	1+2	39.3	07	ND	74.5
O-Cresol	95487	10.0	04	NS	07	ND	NS	10.0	1+2	86.4	07	ND	88.4
P-Cresol	106445	10.0	04	NS	07	ND	NS	10.6 /10.0	1+2	ND	07	ND	
Phenol	108952	10.0	04	NS	07	ND	NS	10.0	1+2	685	07	ND	98.5
Toluene	108883	10.0	04	NS	07	ND	NS	10.0	1+2	1,095	07	ND	99.1
Tripropyleneglycol Methyl Ether	20324338	99.0	04	NS	07	ND	NS	105 /99.0	1+2	ND	07	ND	
Metals													
Barium	7440393	200	04	10.3	07	21.8	0.0	200	1+2	2427	07	21.8	99.1
Boron	7440428	100	04	3,211	07	2,925	8.9	100	1+2	4330	07	2,925	32.5
Chromium	7440473	10.9	04	11.6	07	ND	6.5	10.9	1+2	36.6	07	ND	70.3
Silicon	7440213	100	04	784	07	648	17.4	100	1+2	768	07	648	15.7
Strontium	7440246	80.3	04	ND	07	82.5	0.0	100	1+2	2,912	07	82.5	97.2
Titanium	7440326	4.2	04	4.2	07	ND	1.0	4.2	1+2	13.0	07	ND	67.9
Zinc	7440666	10.6	04	ND	07	12.0	0.0	20.0	1+2	144	07	12.0	91.6
Pesticides/Herbicides													
Dichloroprop	120365	1.0	04	NS	07	NS	NS	1.0	1+2	NS	07	NS	NS
Disulfoton	298044	2.0	04	NS	07	NS	NS	2.0	1+2	NS	07	NS	NS
Dioxins/Furans													
1234678-HpCDD	35822469	50.0 pg/L	04	NS	07	NS	NS	50.0 pg/L	1+2	NS	07	NS	NS
OCDD	3268879	100 pg/L	04	NS	07	NS	NS	100 pg/L	1+2	NS	07	NS	NS

Negative percent removal are recorded as 0.0.

NS: Not Sampled SP: Sample point.

ND: Non-detect DL: Specific detection limits of sample when there is a non-detect, otherwise it is the method detection limit

Table 8-3: Treatment Technology Performance for Facility 4667 - Subtitle D Municipal

Pollutant of Interest Subtitle D Municipal	CAS #	Biological Treatment Unit Operation Only: Sample Points 3 to 4						Entire Treatment System: Sample Points 1 to 6					
		DL	SP	Influent Conc. (ug/L)	SP	Effluent Conc. (ug/L)	% Removal	DL	SP	Influent Conc. (ug/L)	SP	Effluent Conc. (ug/L)	% Removal
Conventional													
BOD	C-002	2,000	03	232,600	04	119,300	48.7	2,000	01	1,088,000	06	201,000	81.5
TSS	C-009	4,000	03	59,600	04	10,200	82.9	4,000	01	93,400	06	ND	95.7
Nonconventional													
Ammonia as Nitrogen	7664417	10.0	03	134,800	04	26,040	80.7	10.0	01	295,900	06	12,060	95.9
COD	C-004	5,000	03	635,000	04	478,200	24.7	5,000	01	2,932,000	06	251,000	91.4
Hexavalent Chromium	18540299	10.0	03	ND	04	ND		10.0	01	26.0	06	ND	61.5
Nitrate/Nitrite	C-005	50.0	03	14,400	04	87,800	0.0	50.0	01	494	06	87,000	0.0
TDS	C-010		03	4,024,000	04	3,987,000	0.9		01	6,232,000	06	3,834,000	38.5
TOC	C-012	1,000	03	212,600	04	120,400	43.4	1,000	01	1,098,600	06	82,000	92.5
Total Phenols	C-020	50.0	03	204	04	52.6	74.2	50.0	01	940	06	ND	94.7
Organics													
1,4-Dioxane	123911	10.0	03	NS	04	ND	NS	10.0	01	323	06	ND	96.9
2-Butanone	78933	50.0	03	NS	04	ND	NS	50.0	01	8,767	06	ND	99.4
2-Propanone	67641	50.0	03	NS	04	ND	NS	50.0	01	13,021	06	ND	99.6
4-Methyl-2-Pentanone	108101	50.0	03	NS	04	ND	NS	50.0	01	1,239	06	ND	96.0
Alpha Terpineol	98555	10.0	03	NS	04	ND	NS	10.0	01	430	06	ND	97.7
Benzoic Acid	65850	50.0	03	NS	04	ND	NS	50.0	01	33,335	06	ND	99.9
Hexanoic Acid	142621	10.0	03	NS	04	ND	NS	10.0	01	37,256	06	ND	100
Methylene Chloride	75092	10.0	03	NS	04	ND	NS	208 /10.0	01	ND	06	ND	
N,N-Dimethylformamide	68122	10.0	03	NS	04	ND	NS	10.0	01	1,008	06	ND	99.0
O-Cresol	95487	10.0	03	NS	04	ND	NS	10.0	01	2,215	06	ND	99.6
P-Cresol	106445	10.0	03	NS	04	ND	NS	10.0	01	ND	06	ND	
Phenol	108952	10.0	03	NS	04	ND	NS	10.0	01	387	06	ND	97.4
Toluene	108883	10.0	03	NS	04	ND	NS	10.0	01	668	06	ND	98.5
Tripropyleneglycol Methyl Ether	20324338	99.0	03	NS	04	ND	NS	99.0	01	ND	06	ND	
Metals													
Barium	7440393	200	03	19.4	04	32.4	0.0	200	01	283	06	42.6	85.0
Boron	7440428	100	03	2,842	04	2,483	12.6	100	01	6,700	06	2,334	65.2
Chromium	7440473	10.0	03	10.5	04	11.3	0.0	11.1	01	90.6	06	ND	87.7
Silicon	7440213	100	03	5,284	04	6,766	0.0	100	01	27,158	06	6,859	74.7
Strontium	7440246	100	03	193	04	237	0.0	100	01	1,935	06	249	87.1
Titanium	7440326	2.5	03	4.8	04	ND	48.1	2.5	01	69.9	06	ND	96.4
Zinc	7440666	20.0	03	25.2	04	58.6	0.0	20.0	01	494	06	27.1	94.5
Pesticides/Herbicides													
Dichloroprop	120365	1.0	03	NS	04	ND	NS	11.8 /1.0	01	ND	06	ND	
Disulfoton	298044	2.0	03	NS	04	ND	NS	2.0	01	6.1	06	ND	67.2
Dioxins/Furans													
1234678-HpCDD	35822469	50.0 pg/L	03	NS	04	NS	NS	50.0 pg/L	01	NS	06	NS	NS
OCDD	3268879	100 pg/L	03	NS	04	NS	NS	100 pg/L	01	NS	06	NS	NS

Negative percent removal are recorded as 0.0.

NS: Not Sampled DL: Specific detection limits of sample when there is a non-detect, otherwise it is the method detection limit

ND: Non-detect SP: Sample point

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Table 8-4: Treatment Technology Performance for Facility 4721 - Subtitle C Hazardous

Pollutant of Interest Subtitle C Hazardous	CAS #	Biological Treatment Unit: Sample Points 1 to 2					
		DL	SP	Influent Conc. (ug/L)	SP	Effluent Conc. (ug/L)	% Removal
Conventional							
BOD	C-002	2,000	01	877,875	02	47,000	94.7
Oil and Grease	C-036	5,000	01	45,442	02	6,792	85.1
TSS	C-009	4,000	01	191,375	02	53,375	72.1
Nonconventional							
Amenable Cyanide	C-025	10.0	01	ND	02	ND	
Ammonia as Nitrogen	7664417	10.0	01	382,250	02	1,433	99.6
COD	C-004	5,000	01	2,033,750	02	565,750	72.2
Nitrate/Nitrite	C-005	50.0	01	1,770	02	333,375	0.0
TDS	C-010		01	12,275,000	02	12,075,000	1.6
TOC	C-012	1,000	01	562,250	02	189,625	66.3
Total Cyanide	57125	20.0	01	54.1	02	46.1	14.8
Total Phenols	C-020	50.0	01	3,195	02	67.6	97.9
Organics							
1,1-Dichloroethane	75343	10.0	01	31.5	02	ND	68.2
1,4-Dioxane	123911	10.0	01	ND	02	ND	
2-Butanone	78933	50.0	01	6,398	02	ND	99.2
2-Propanone	67641	50.0	01	4,398	02	ND	98.9
2,4-Dimethylphenol	105679	10.0	01	79.0	02	ND	87.4
4-Methyl-2-Pentanone	108101	50.0	01	2,175	02	ND	97.7
Alpha Terpineol	98555	10.0	01	691	02	ND	98.6
Aniline	62533	10.0	01	685	02	ND	98.5
Benzene	71432	10.0	01	127	02	ND	92.2
Benzoic Acid	65850	50.0	01	5,294	02	ND	99.1
Benzyl Alcohol	100516	10.0	01	23.7	02	ND	57.9
Diethyl Ether	60297	50.0	01	104	02	ND	51.8
Ethylbenzene	100414	10.0	01	545	02	ND	98.2
Hexanoic Acid	142621	10.0	01	1,632	02	ND	99.4
Isobutyl Alcohol	78831	10.0	01	ND	02	ND	
M-Xylene	108383	10.0	01	412	02	ND	97.6
Methylene Chloride	75092	10.0	01	49.2	02	ND	79.7
Naphthalene	91203	10.0	01	486	02	ND	97.9
O+P Xylene	136777612	10.0	01	155	02	ND	93.6
O-Cresol	95487	10.0	01	ND	02	ND	
P-Cresol	106445	10.0	01	218	02	ND	95.4
Phenol	108952	10.0	01	1,553	02	ND	99.4
Pyridine	110861	10.0	01	12.0	02	ND	16.5
Toluene	108883	10.0	01	1,468	02	ND	99.3
Trans-1,2-Dichloroethene	156605	10.0	01	52.7	02	ND	81.0
Trichloroethene	79016	10.0	01	ND	02	ND	
Tripropyleneglycol Methyl Ether	20324338	99.0	01	1,756	02	ND	94.4
Vinyl Chloride	75014	10.0	01	15.6	02	ND	36.0
Metals							
Arsenic	7440382	10.0	01	1,492	02	569	61.9
Boron	7440428	100	01	8,839	02	8,449	4.4
Chromium	7440473	10.0	01	86.7	02	46.5	46.4
Copper	7440508	8.0	01	20.6	02	ND	61.2
Lithium	7439932	100	01	277	02	316	0.0
Metals (Cont'd)							
Molybdenum	7439987	10.0	01	227	02	266	0.0
Nickel	7440020	40.0	01	131	02	125	4.1

Table 8-4: Treatment Technology Performance for Facility 4721 - Subtitle C Hazardous (continued)

Pollutant of Interest Subtitle C Hazardous	CAS #	Biological Treatment Unit: Sample Points 1 to 2					
		DL	SP	Influent Conc. (ug/L)	SP	Effluent Conc. (ug/L)	% Removal
Selenium	7782492	15.5	01	20.0	02	ND	22.5
Silicon	7440213	100	01	5,518	02	5,024	9.0
Strontium	7440246	100	01	2,846	02	2,494	12.4
Tin	7440315	30.0	01	30.7	02	ND	2.4
Titanium	7440326	5.0	01	64.5	02	5.3	91.7
Zinc	7440666	20.0	01	253	02	85.3	66.3
Pesticides/Herbicides							
2,4-D	94757	1.0	01	1.2	02	ND	14.0
2,4-DB	94826	2.0	01	3.9	02	ND	48.4
2,4,5-TP	93721	0.2	01	0.5	02	ND	55.1
Dicamba	1918009	0.2	01	1.1	02	0.4	64.2
Dichloroprop	120365	1.0	01	2.1	02	1.3	37.7
MCPA	94746	50.0	01	59.1	02	ND	15.3
MCPP	7085190	50.0	01	153	02	51.9	66.1
Picloram	1918021	0.5	01	0.5	02	ND	2.0
Terbuthylazine	5915413	5.0	01	6.0	02	ND	16.8
Dioxins/Furans							
1234678-HpCDD	35822469	50.0 pg/L	01	588 pg/L	02	NS	NS
1234678-HpCDF	67562394	50.0 pg/L	01	63.3 pg/L	02	NS	NS
OCDD	3268879	100.0 pg/L	01	6,148 pg/L	02	NS	NS
OCDF	39001020	100.0 pg/L	01	237 pg/L	02	NS	NS

Negative percent removal are recorded as 0.0.
NS: Not Sampled
ND: Non-detect
DL: Specific detection limits of sample when there is a non-detect, otherwise it is the method detection limit
SP: Sample point.

Table 8-5: Treatment Technology Performance for Facility 4759 - Subtitle C Hazardous

Pollutant of Interest Subtitle C Hazardous	CAS #	Bioliogical Treatment Unit Only: Sample Points 2 to 3						Entire Treatment System Sample Points 1 to 3					
		DL	SP	Influent Conc. (ug/L)	SP	Effluent Conc. (ug/L)	% Removal	DL	SP	Influent Conc. (ug/L)	SP	Effluent Conc. (ug/L)	% Removal
Conventional													
BOD	C-002	2,000	02	2,650,000	03	62,800	97.6	2,000	01	2,664,000	03	62,800	97.6
Oil and Grease	C-036	5,000	02	30,167	03	9,333	69.1	5,000	01	37,333	03	9,333	75.0
TSS	C-009	4,000	02	47,300	03	90,000	0.0	4,000	01	122,600	03	90,000	26.6
Nonconventional													
Amenable Cyanide	C-025	20.0	02	NS	03	271	NS	20.0	01	3,990	03	271	93.2
Ammonia as Nitrogen	7664417	10.0	02	194,400	03	155,500	20.0	10.0	01	209,400	03	155,500	25.7
COD	C-004	5,000	02	5,200,000	03	1,180,000	77.3	5,000	01	5,006,000	03	1,180,000	76.4
Nitrate/Nitrite	C-005	50.0	02	263,196	03	240,423	8.7	50.0	01	259,242	03	240,423	7.3
TDS	C-010		02	17,230,000	03	15,680,000	9.0		01	16,360,000	03	15,680,000	4.2
TOC	C-012	1,000	02	1,800,000	03	284,700	84.2	1,000	01	1,804,000	03	284,700	84.2
Total Cyanide	57125	20.0	02	869	03	796	8.5	20.0	01	9,756	03	796	91.9
Total Phenols	C-020	50.0	02	97,340	03	155	99.8	50.0	01	97,860	03	155	99.8
Organics													
1,1-Dichloroethane	75343	10.0	02	23.8	03	ND	58.0	10.0	01	26.7	03	ND	62.5
1,4-Dioxane	123911	10.0	02	1,935	03	702	63.7	10.0	01	2,003	03	702	65.0
2-Butanone	78933	50.0	02	1,633	03	ND	96.9	50.0	01	1,724	03	ND	97.1
2-Propanone	67641	50.0	02	3,254	03	65.0	98.0	50.0	01	3,634	03	65.0	98.2
2,4-Dimethylphenol	105679	10.0	02	1,798	03	201	88.8	10.0	01	1,550	03	201	87.0
4-Methyl-2-Pentanone	108101	50.0	02	1,009	03	ND	95.1	50.0	01	1,027	03	ND	95.1
Alpha Terpineol	98555	10.0	02	ND	03	ND		10.0	01	ND	03	ND	
Aniline	62533	10.0	02	577	03	ND	98.3	10.0	01	533	03	ND	98.1
Benzene	71432	10.0	02	32.0	03	ND	68.7	10.0	01	36.2	03	ND	72.4
Benzoic Acid	65850	50.0	02	70,690	03	ND	99.9	50.0	01	64,957	03	ND	99.9
Benzyl Alcohol	100516	10.0	02	859	03	ND	98.8	10.0	01	878	03	ND	98.9
Diethyl Ether	60297	50.0	02	ND	03	ND		50.0	01	ND	03	ND	
Ethylbenzene	100414	10.0	02	13.8	03	ND	27.3	10.0	01	15.8	03	ND	36.5
Hexanoic Acid	142621	10.0	02	5,266	03	ND	99.8	10.0	01	3,640	03	ND	99.7
Isobutyl Alcohol	78831	10.0	02	127	03	ND	92.1	10.0	01	138	03	ND	92.8
M-Xylene	108383	10.0	02	10.6	03	ND	5.3	10.0	01	10.7	03	ND	6.2
Methylene Chloride	75092	10.0	02	604	03	10.3	98.3	10.0	01	661	03	10.3	98.4
Naphthalene	91203	10.0	02	22.0	03	ND	54.6	10.0	01	24.8	03	ND	59.6
O+P Xylene	136777612	10.0	02	ND	03	ND		10.0	01	ND	03	ND	
O-Cresol	95487	10.0	02	61.2	03	ND	83.7	10.0	01	188	03	ND	94.7
P-Cresol	106445	10.0	02	5,119	03	ND	99.8	10.0	01	5,022	03	ND	99.8
Phenol	108952	10.0	02	54,808	03	29.7	100	10.0	01	65,417	03	29.7	100
Pyridine	110861	10.0	02	309	03	ND	96.8	10.0	01	301	03	ND	96.7
Toluene	108883	10.0	02	120	03	ND	91.7	10.0	01	136	03	ND	92.6
Trans-1,2-Dichloroethene	156605	10.0	02	ND	03	ND		10.0	01	ND	03	ND	
Trichloroethene	79016	10.0	02	ND	03	ND		10.0	01	ND	03	ND	
Tripropyleneglycol Methyl Ether	20324338	99.0	02	ND	03	ND		99.0	01	1,021	03	ND	90.3
Vinyl Chloride	75014	10.0	02	ND	03	ND		10.0	01	ND	03	ND	
Metals													
Arsenic	7440382	10.0	02	389	03	312	19.9	10.0	01	584	03	312	46.6
Boron	7440428	100	02	2,706	03	2,486	8.1	100	01	2,918	03	2,486	14.8
Chromium	7440473	10.0	02	158	03	82.4	47.8	10.0	01	415	03	82.4	80.2

Table 8-5: Treatment Technology Performance for Facility 4759- Subtitle C Hazardous (continued)

Pollutant of Interest Subtitle C Hazardous	CAS #	Bioliogical Treatment Unit Only: Sample Points 2 to 3						Entire Treatment System Sample Points 1 to 3					
		DL	SP	Influent Conc. (ug/L)	SP	Effluent Conc. (ug/L)	% Removal	DL	SP	Influent Conc. (ug/L)	SP	Effluent Conc. (ug/L)	% Removal
Metals (cont.)													
Copper	7440508	25.0	02	61.1	03	76.4	0.0	25.0	01	139	03	76.4	45.2
Lithium	7439932	100	02	253	03	239	5.5	100	01	266	03	239	10.2
Molybdenum	7439987	10.0	02	13,710	03	13,130	4.2	10.0	01	13,260	03	13,130	1.0
Nickel	7440020	40.0	02	2,014	03	1,878	6.8	40.0	01	2,060	03	1,878	8.8
Selenium	7782492	5.0	02	191	03	190	0.2	5.0	01	178	03	190	0.0
Silicon	7440213	100	02	6,924	03	6,153	11.1	100	01	6,036	03	6,153	0.0
Strontium	7440246	100	02	105	03	94.4	9.9	100	01	284	03	94.4	66.8
Tin	7440315	30.0	02	800	03	723	9.5	30.0	01	908	03	723	20.4
Titanium	7440326	5.0	02	5.1	03	2.4	52.1	5.0	01	23.3	03	2.4	89.6
Zinc	7440666	20.0	02	26.7	03	47.2	0.0	20.0	01	126	03	47.2	62.5
Pesticides/Herbicides													
2,4-D	94757	1.0	02	NS	03	11.8	NS	1.0	01	11.2	03	11.8	0.0
2,4-DB	94826	2.0	02	NS	03	4.3	NS	2.0	01	43.8	03	4.3	90.2
2,4,5-TP	93721	0.2	02	NS	03	0.4	NS	0.2	01	0.5	03	0.4	18.3
Dicamba	1918009	0.2	02	NS	03	0.9	NS	0.2	01	41.6	03	0.9	97.9
Dichloroprop	120365	1.0	02	NS	03	4.7	NS	1.0	01	18.3	03	4.7	74.3
MCPA	94746	50.0	02	NS	03	182	NS	50.0	01	332	03	182	45.3
MCPP	7085190	50.0	02	NS	03	288	NS	50.0	01	662	03	288	56.5
Picloram	1918021	0.5	02	NS	03	2.5	NS	0.5	01	4.5	03	2.5	45.2
Terbuthylazine	5915413	5.0	02	NS	03	28.4	NS	5.0	01	97.6	03	28.4	70.9
Dioxins/Furans													
1234678-HpCDD	35822469	50.0 pg/L	02	NS	03	ND	NS	50.0 pg/L	01	ND	03	ND	
1234678-HpCDF	67562394	50.0 pg/L	02	NS	03	ND	NS	50.0 pg/L	01	ND	03	ND	
OCDD	3268879	100 pg/L	02	NS	03	ND	NS	100 pg/L	01	ND	03	100 pg/L	
OCDF	39001020	100 pg/L	02	NS	03	ND	NS	100 pg/L	01	ND	03	ND	

Negative percent removal are recorded as 0.0.
NS: Not Sampled
ND: Non-detect
DL: Specific detection limits of sample when there is a non-detect, otherwise it is the method detection limit
SP: Sample point.

Table 8-6: Treatment Technology Performance for Facility 4687 - Subtitle D Municipal

Pollutant of Interest Subtitle D Municipal	CAS #	Single-Stage Reverse Osmosis Treatment System Only: Sample Point 1 to 2						Entire Treatment System: Sample Point 1 to 3					
		DL	SP	Influent Conc. (ug/L)	SP	Effluent Conc. (ug/L)	% Removal	DL	SP	Influent Conc. (ug/L)	SP	Effluent Conc. (ug/L)	% Removal
Conventional													
BOD	C-002	2,000	01	1,182,000	02	54,000	95.4	2,000	01	1,182,000	03	5,400	99.5
TSS	C-009	4,000	01	171,800	02	ND	97.7	4,000	01	171,800	03	ND	97.7
Noconventional													
Ammonia as Nitrogen	7664417	10.0	01	58,480	02	13,600	76.7	10.0	01	58,480	03	608	99.0
COD	C-004	5,000	01	1,526,000	02	72,200	95.3	5,000	01	1,526,000	03	11,400	99.3
Hexavalent Chromium	18540299	10.0	01	28.0	02	ND	64.3	10.0	01	28.0	03	ND	64.3
Nitrate/Nitrite	C-005	50.0	01	1,300	02	666	48.8	50.0	01	1,300	03	502	61.4
TDS	C-010		01	2,478,000	02	116,600	95.3	10,000	01	2,478,000	03	ND	99.6
TOC	C-012	1,000	01	642,600	02	25,000	96.1	10,000	01	642,600	03	ND	98.4
Total Phenols	C-020	50.0	01	1,262	02	316	75.0	50.0	01	1,262	03	62.8	95.0
Organics													
1,4-Dioxane	123911	10.8 /14.9	01	ND	02	ND		10.8 /10.0	01	ND	03	ND	
2-Butanone	78933	50.0	01	3,250	02	1,774	45.4	50.0	01	3,250	03	372	88.6
2-Propanone	67641	50.0	01	1,580	02	1,842	0.0	50.0	01	1,580	03	470	70.3
4-Methyl-2-Pentanone	108101	50.5	01	382	02	ND	86.8	50.0	01	382	03	ND	86.9
Alpha Terpineol	98555	10.0	01	44.5	02	ND	77.5	10.0	01	44.5	03	ND	77.5
Benzoic Acid	65850	50.0	01	7,685	02	96.3	98.8	50.0	01	7,685	03	ND	99.4
Hexanoic Acid	142621	10.0	01	5,818	02	118	98.0	10.0	01	5,818	03	ND	99.8
Methylene Chloride	75092	10.0	01	ND	02	ND		10.0	01	ND	03	ND	
N,N-Dimethylformamide	68122	10.0	01	ND	02	ND		10.0	01	ND	03	ND	
O-Cresol	95487	10.0	01	ND	02	ND		10.0	01	ND	03	ND	
P-Cresol	106445	10.0	01	797	02	253	68.3	10.0	01	797	03	22.3	97.2
Phenol	108952	10.0	01	702	02	185	73.6	10.0	01	702	03	29.3	95.8
Toluene	108883	10.0	01	376	02	112	70.2	10.0	01	376	03	15.1	96.0
Tripropyleneglycol Methyl Ether	20324338	99.0	01	1,207	02	ND	91.8	99.0	01	1,207	03	ND	91.8
Metals													
Barium	7440393	200	01	280	02	5.6	98.0	200	01	280	03	1.4	99.5
Boron	7440428	100	01	1,808	02	830	54.1	100	01	1,808	03	101	94.4
Chromium	7440473	9.0	01	ND	02	ND		9.0	01	ND	03	ND	
Silicon	7440213	100	01	4,362	02	511	88.3	100	01	4,362	03	355	91.9
Strontium	7440246	100	01	1,406	02	ND	92.9	100	01	1,406	03	ND	92.9
Titanium	7440326	4.0	01	ND	02	ND		4.0	01	ND	03	ND	
Zinc	7440666	10.9 /9.0	01	ND	02	ND		10.9 /10.0	01	ND	03	ND	
Pesticides/Herbicides													
Dichloroprop	120365	1.0	01	6.1	02	ND	83.6	1.0	01	6.1	03	ND	83.6
Disulfoton	298044	2.0	01	14.3	02	ND	86.1	2.0	01	14.3	03	ND	86.1
Dioxins/Furans													
1234678-HpCDD	35822469	49.8 pg/L	01	ND	02	NS	NS	49.8 pg/L	01	ND	03	NS	NS
OCDD	3268879	99.5 pg/L	01	ND	02	NS	NS	99.5 pg/L	01	ND	03	NS	NS

Negative percent removal are recorded as 0.0.

NS: Not Sampled DL: Specific detection limits of sample when there is a non-detect, otherwise it is the method detection limit

ND: Non-detect SP: Sample point.

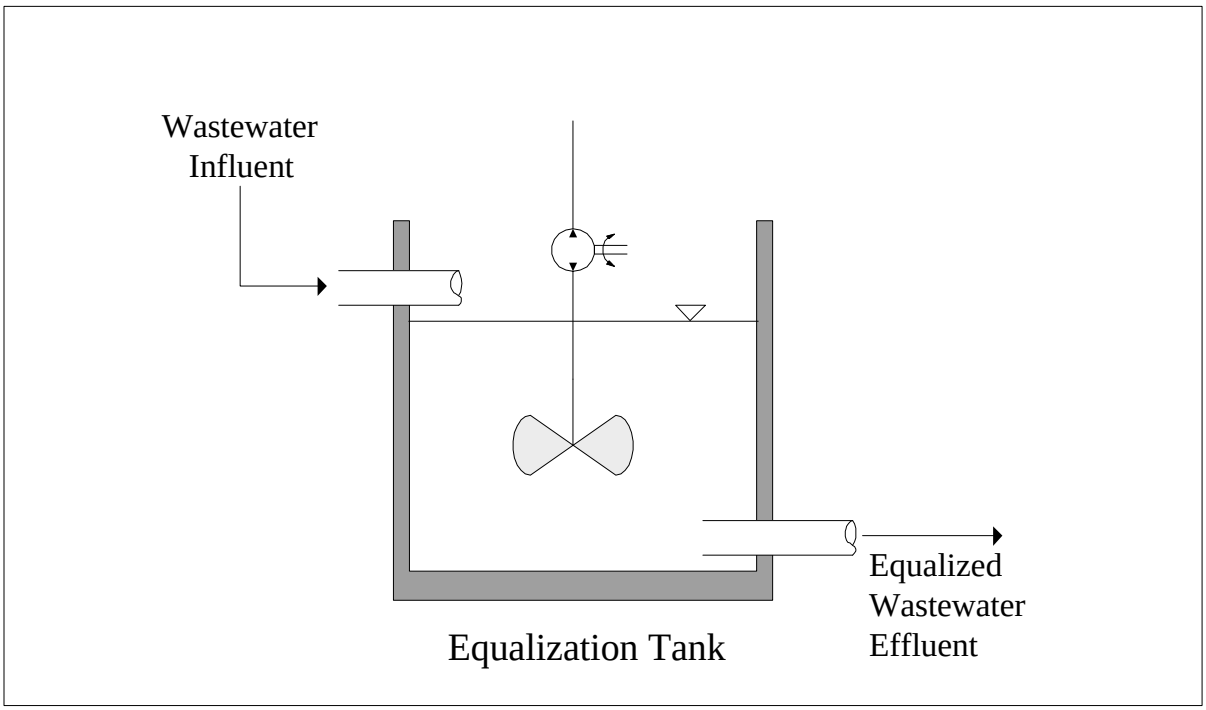


Figure 8-1: Equalization

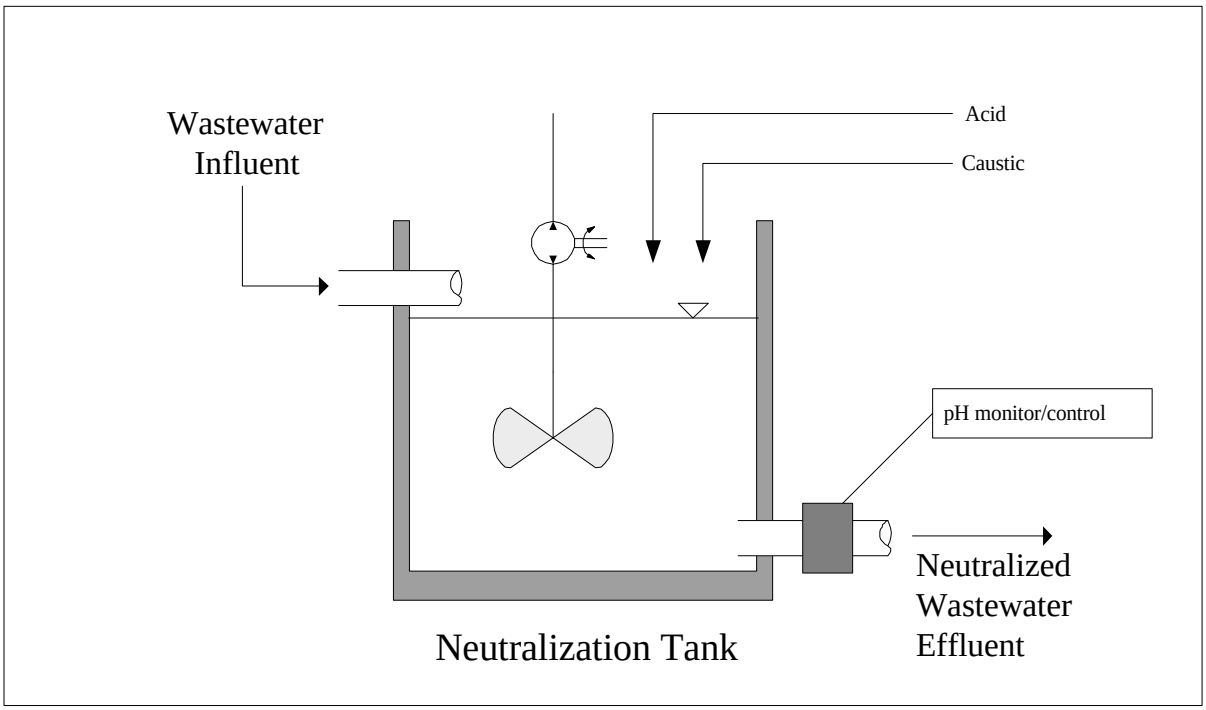


Figure 8-2: Neutralization

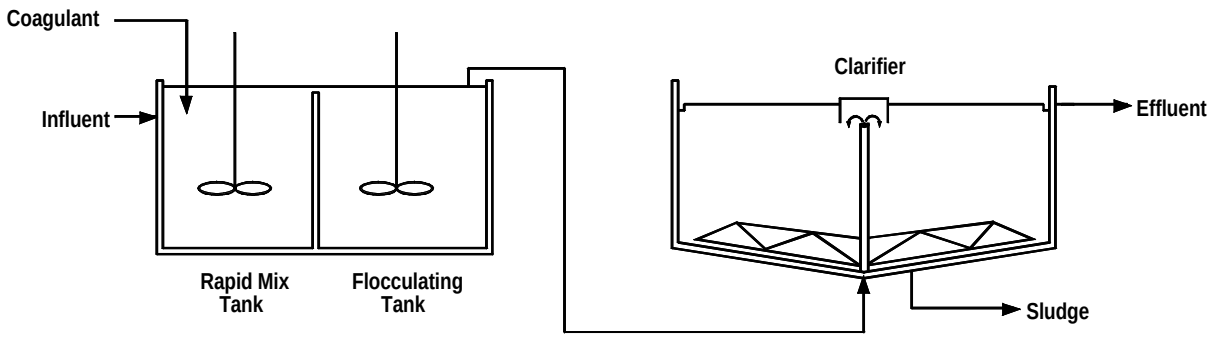


Figure 8-3: Clarification System Incorporating Coagulation and Flocculation

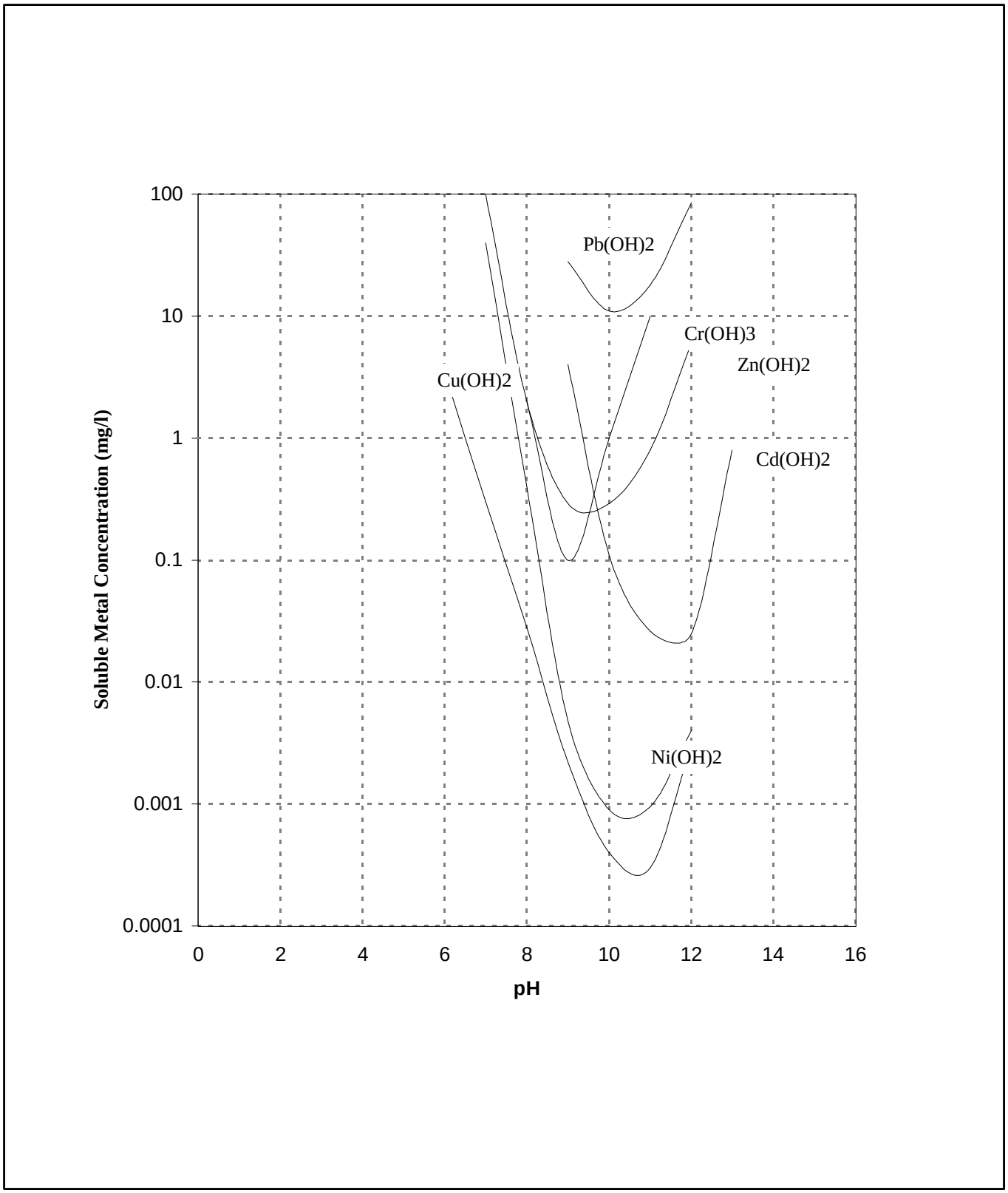


Figure 8-4: Calculated Solubilities of Metal Hydroxides

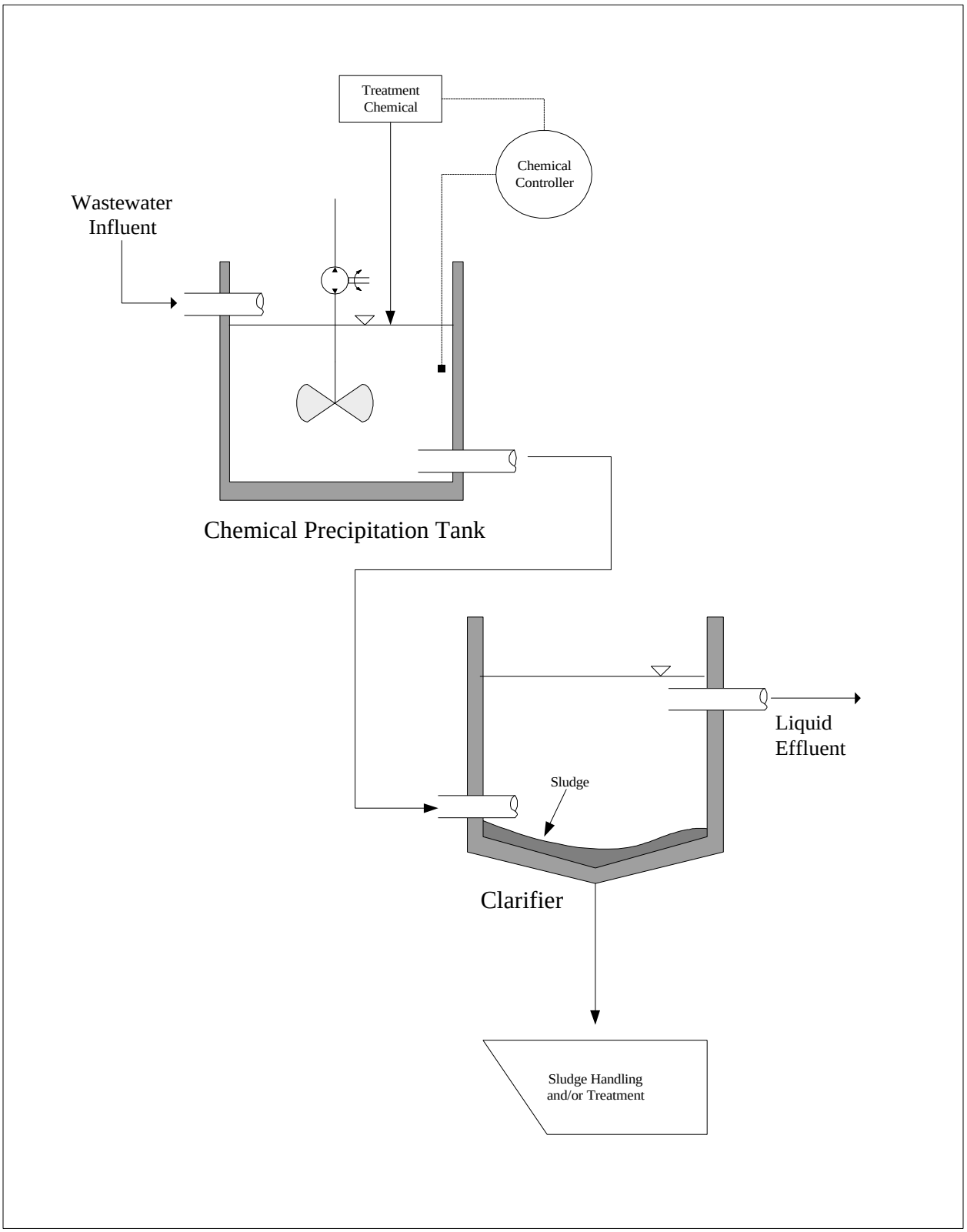


Figure 8-5: Chemical Precipitation System Design

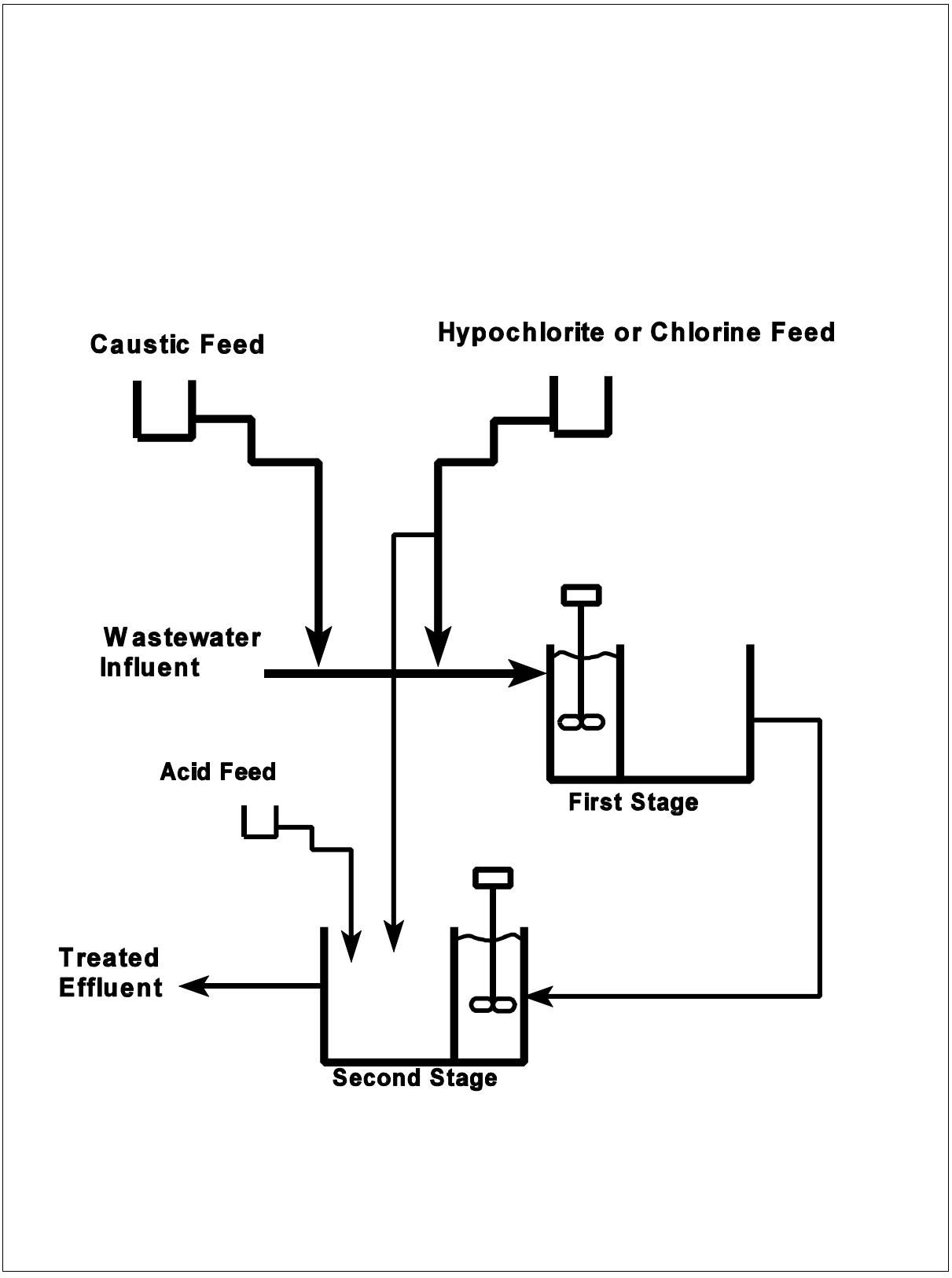


Figure 8-6: Cyanide Destruction

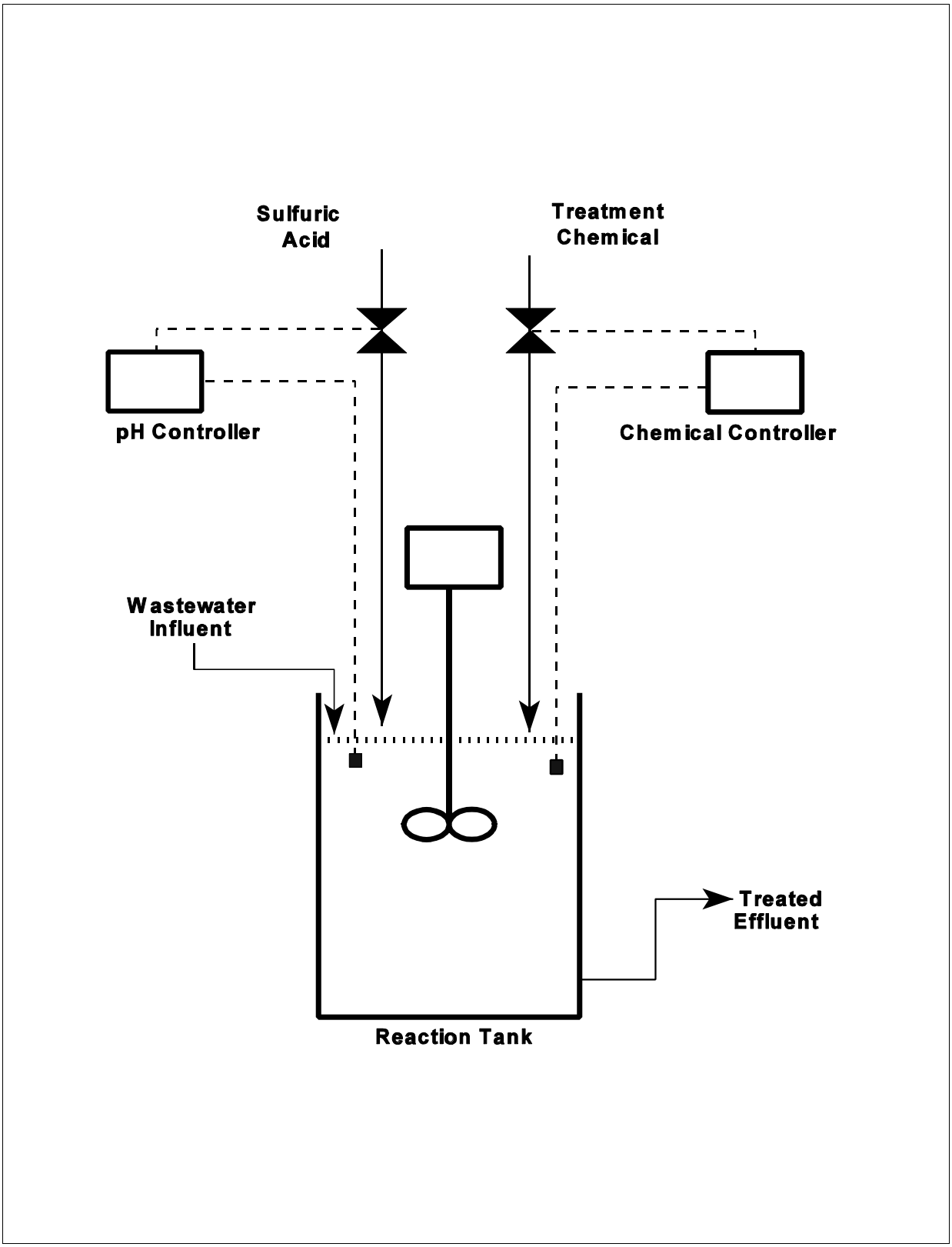


Figure 8-7: Chromium Reduction

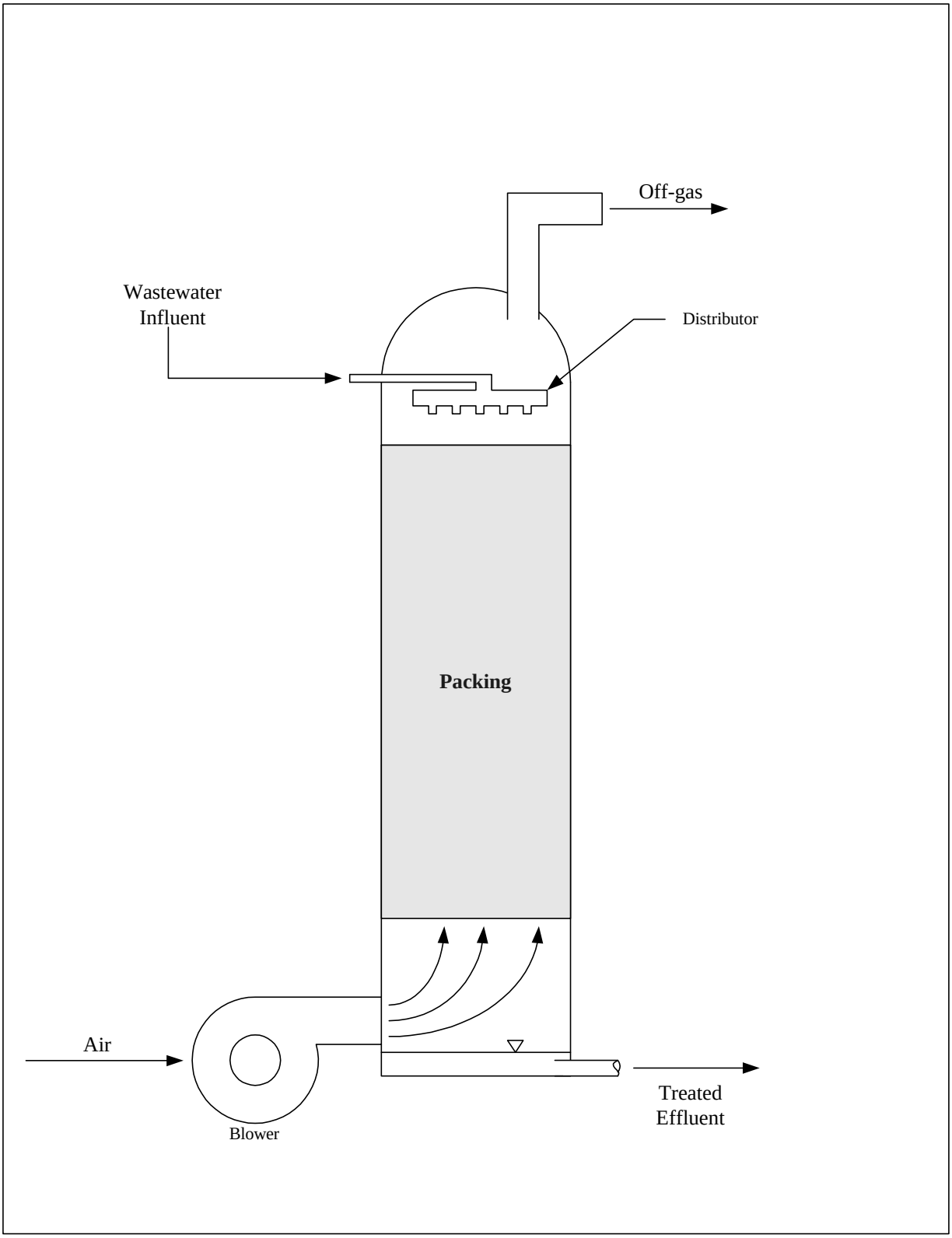


Figure 8-8: Typical Air Stripping System

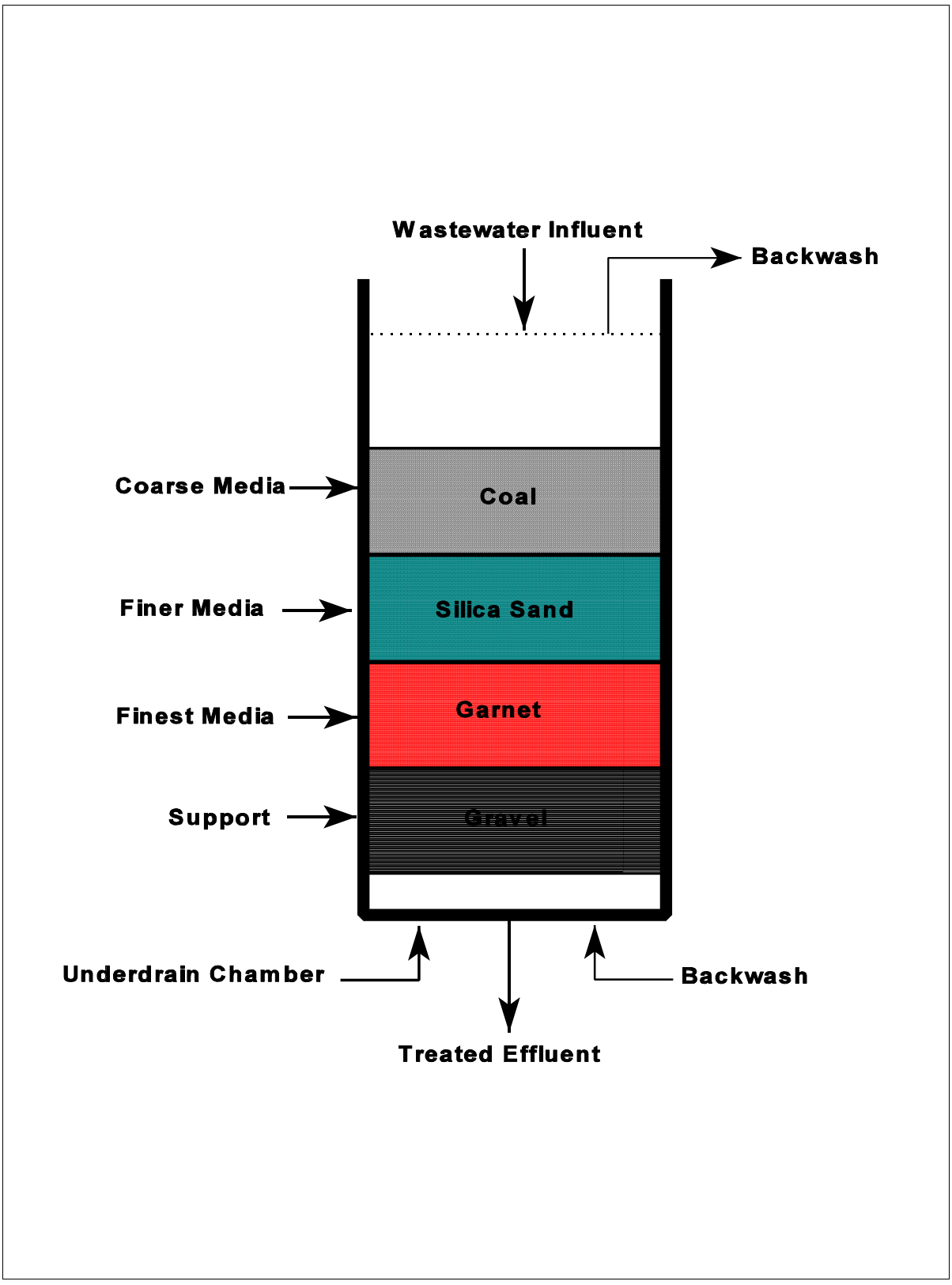


Figure 8-9: Multimedia Filtration

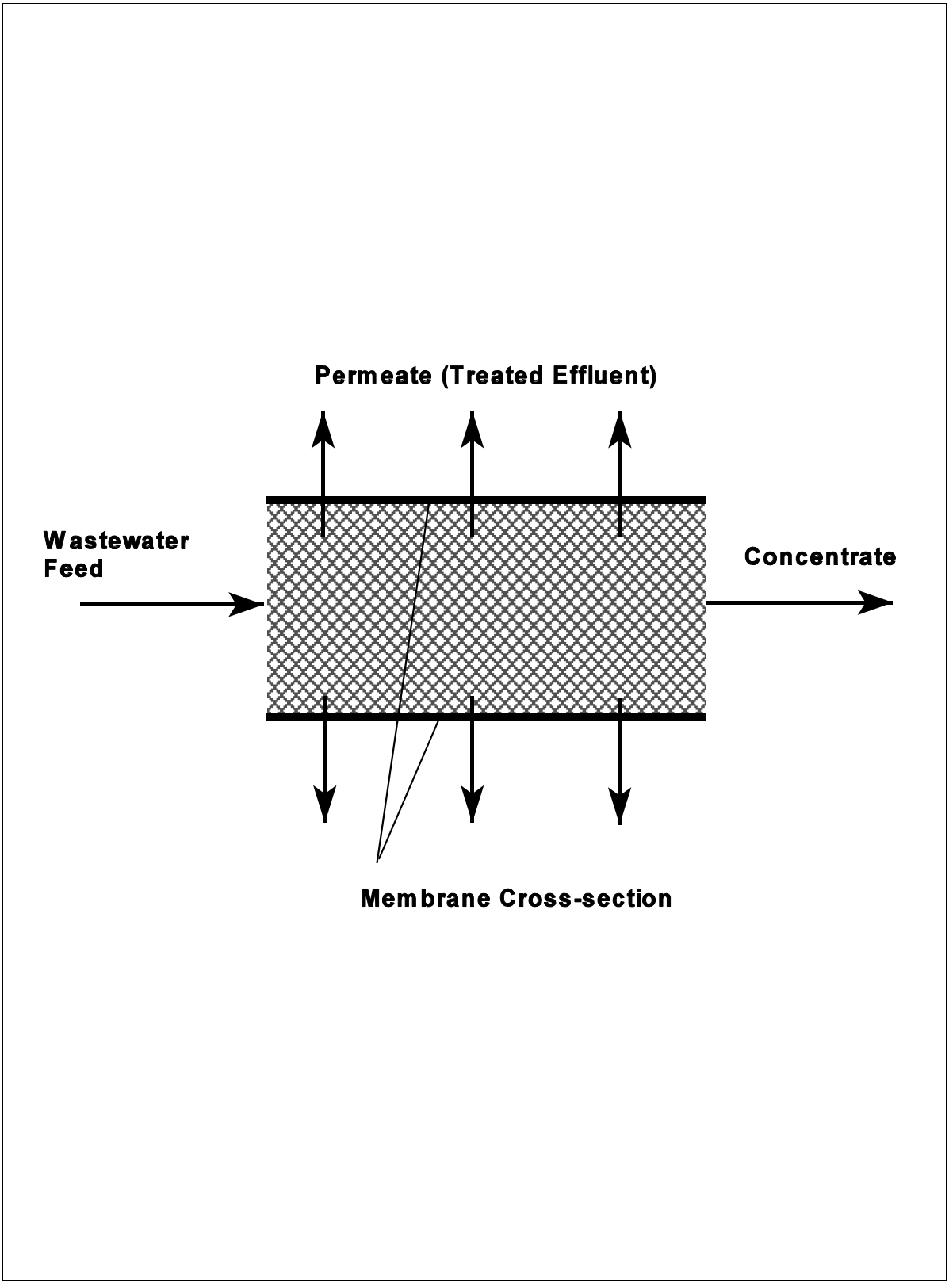


Figure 8-10: Ultrafiltration System Diagram

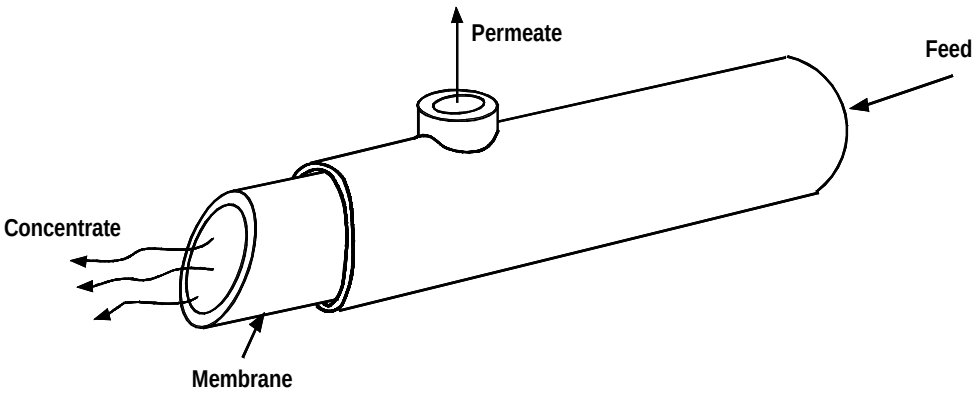


Figure 8-11: Tubular Reverse Osmosis Module

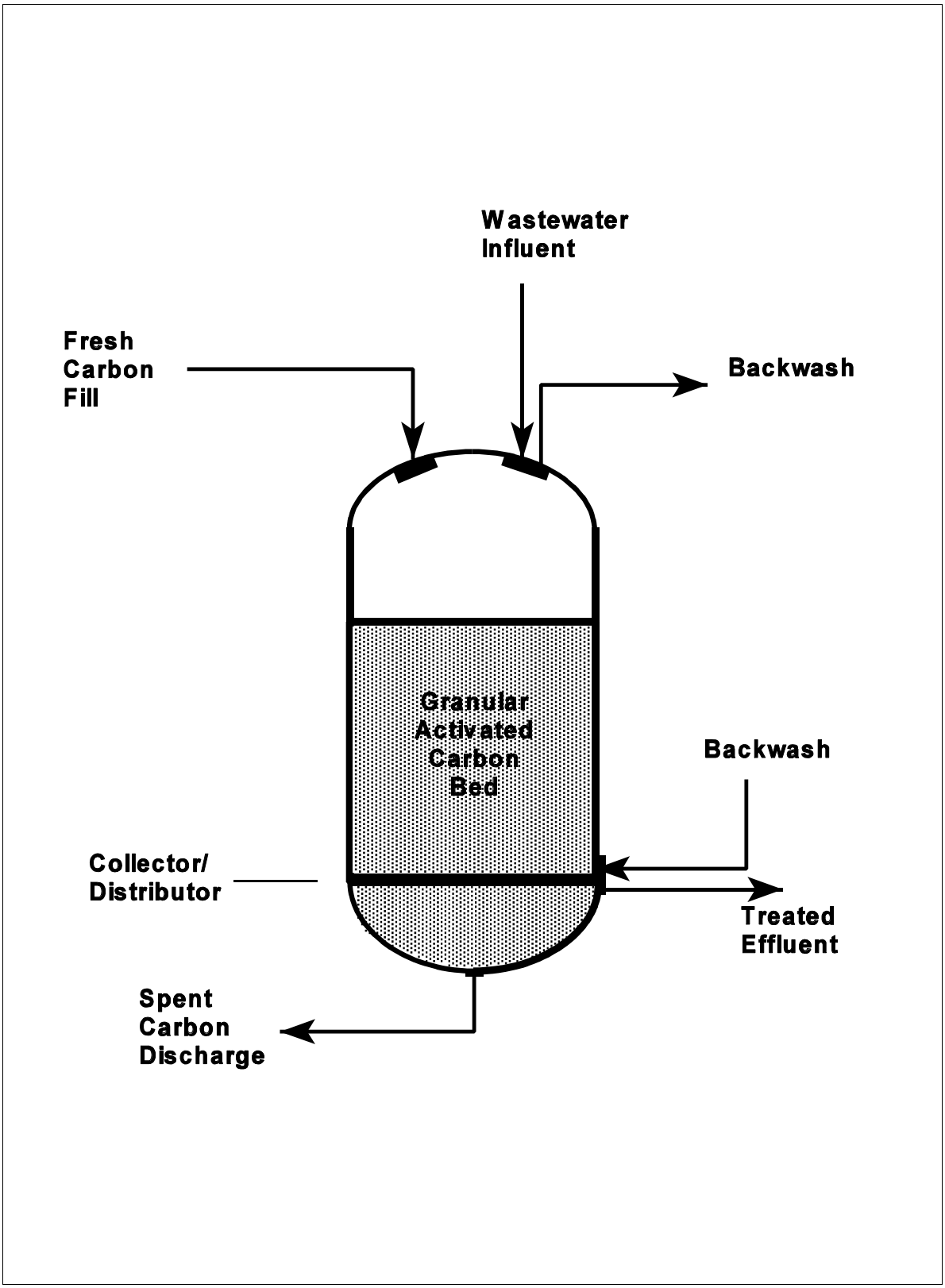


Figure 8-12: Granular Activated Carbon Adsorption

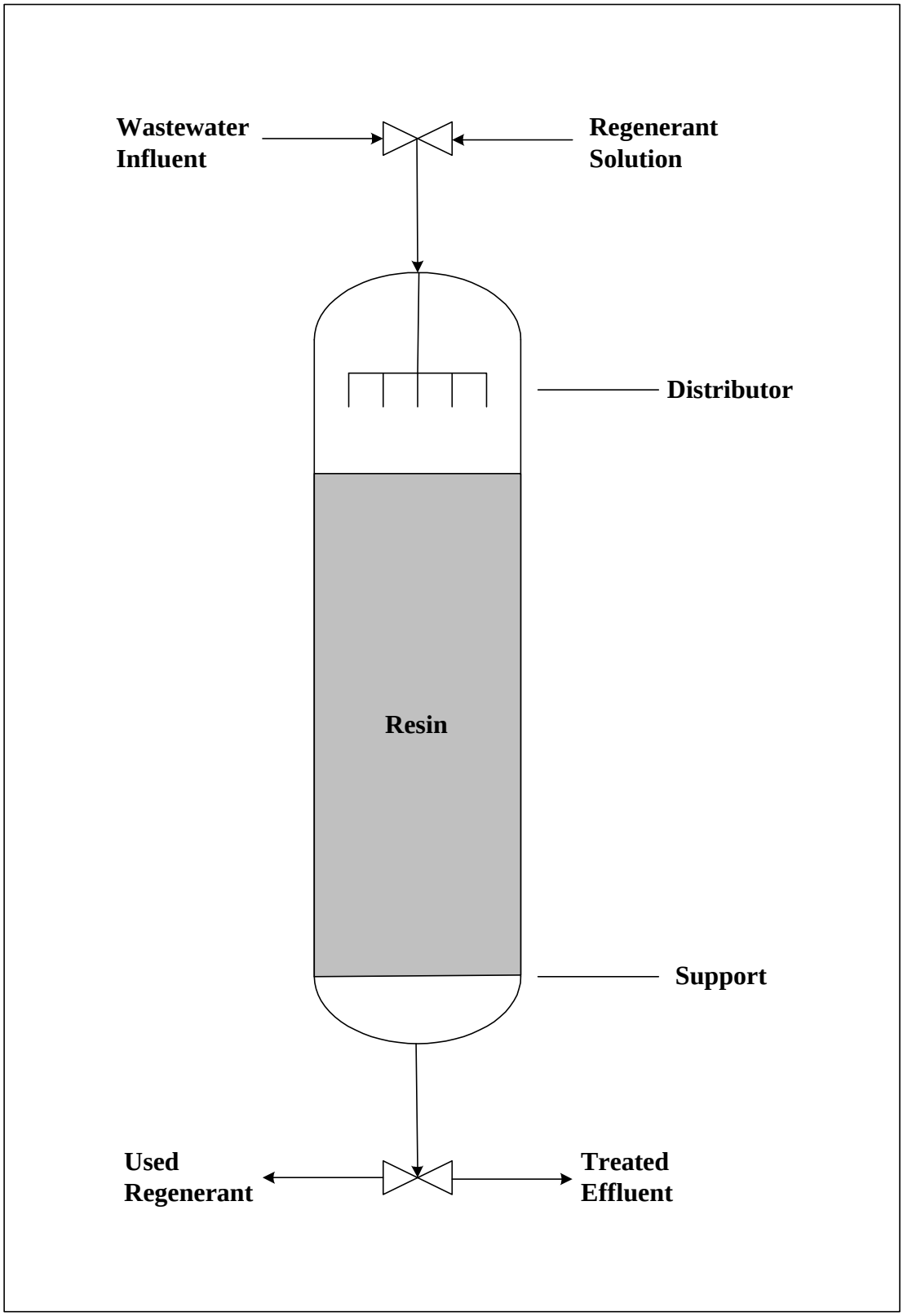


Figure 8-13: Ion Exchange

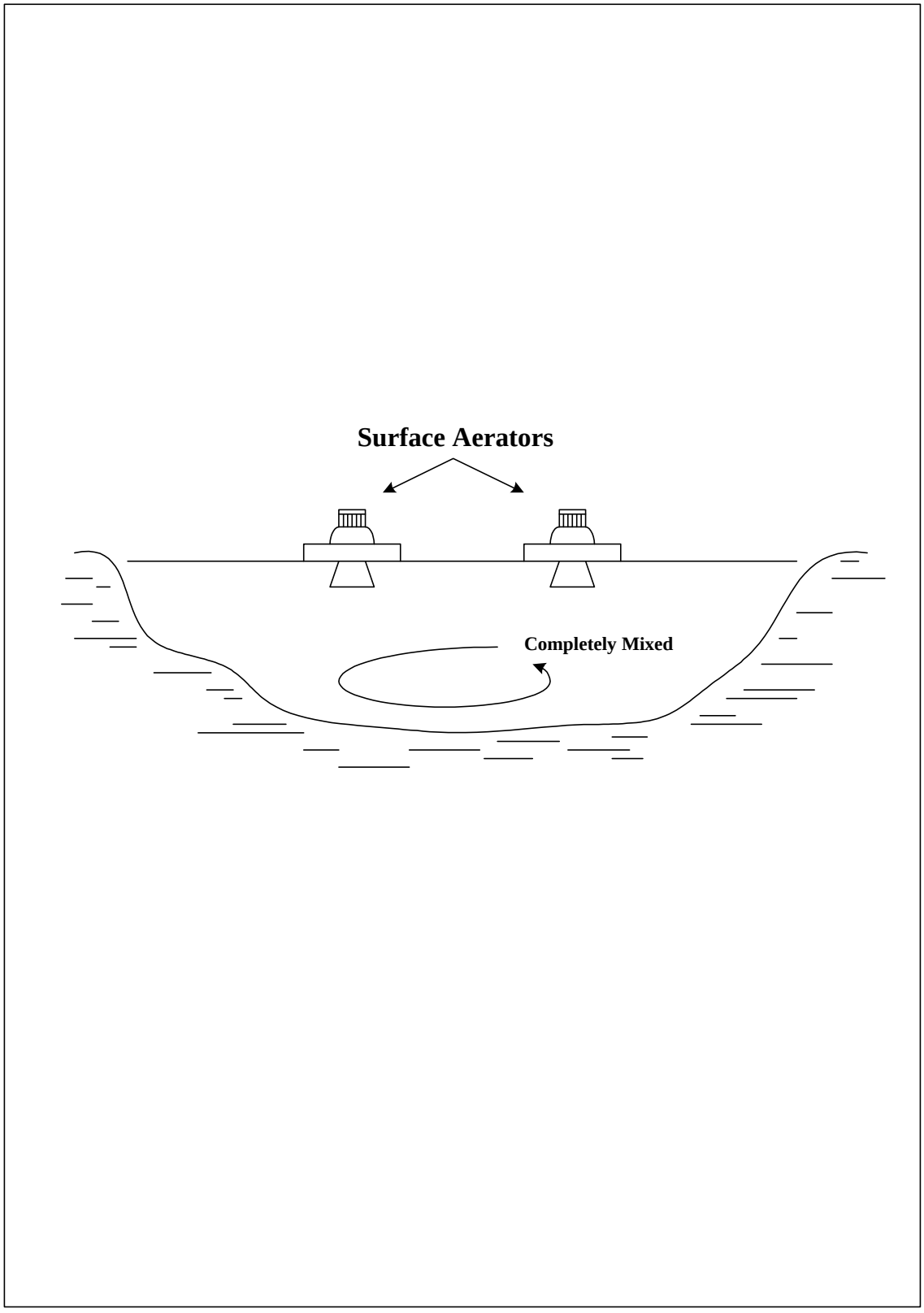


Figure 8-14: Aerated Lagoon

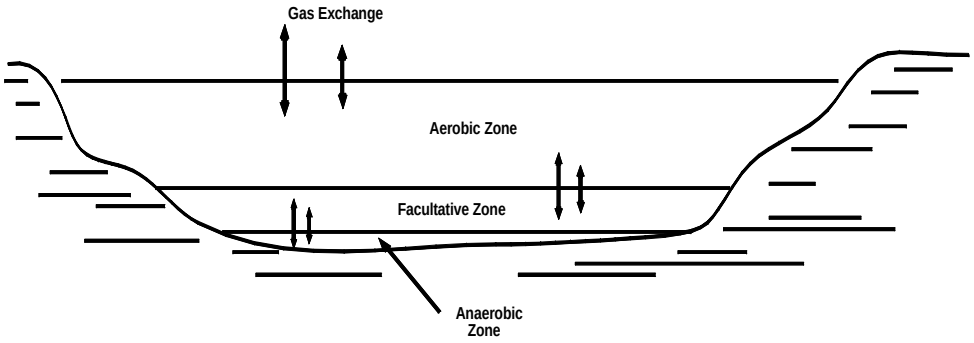


Figure 8-15: Facultative Pond

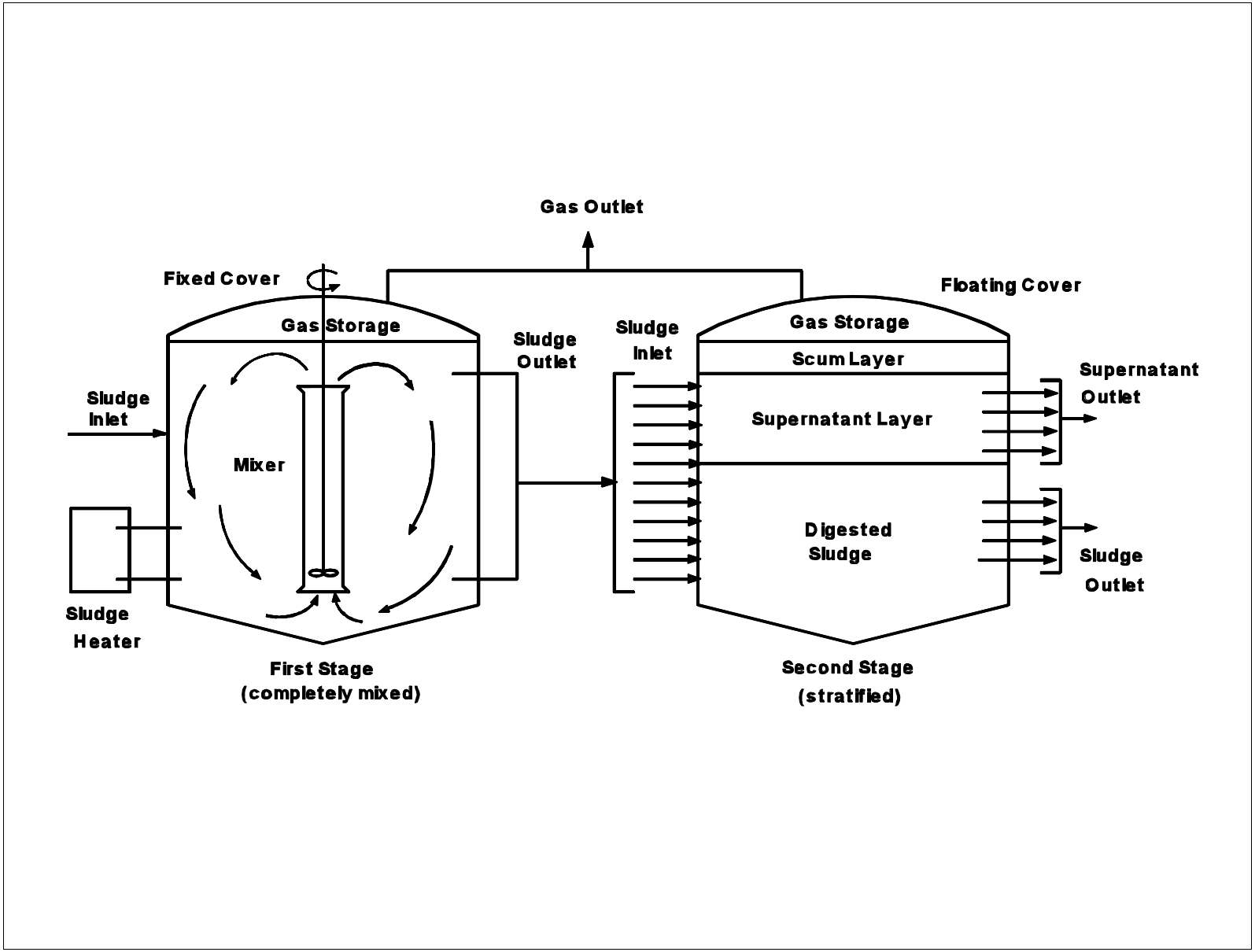


Figure 8-16: Completely Mixed Digestor System

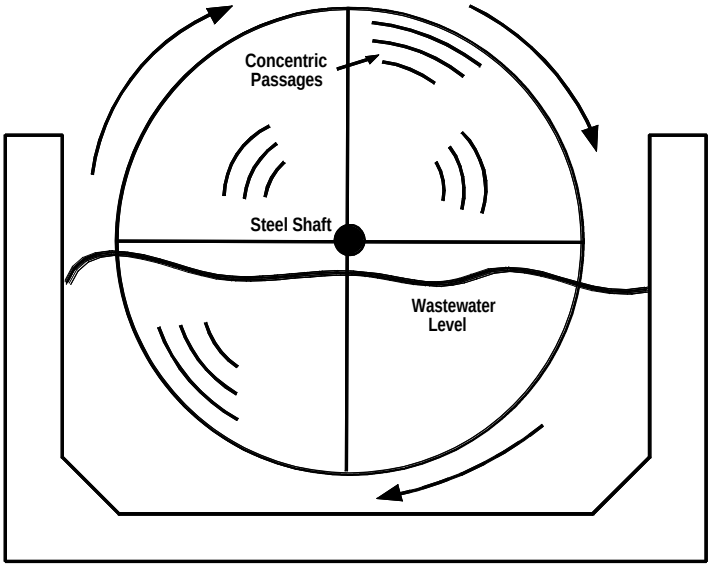


Figure 8-17: Rotating Biological Contactor Cross-Section

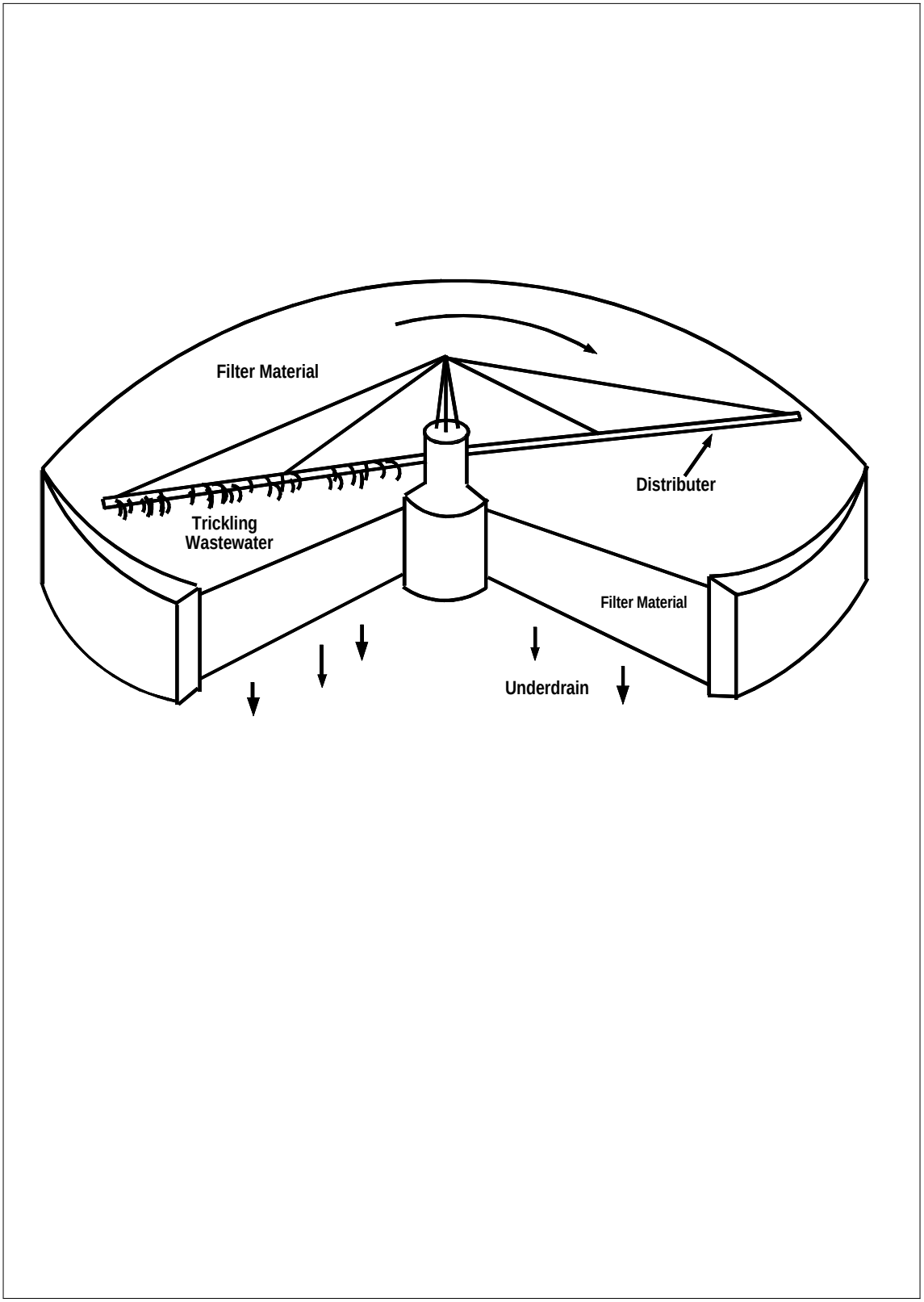


Figure 8-18: Trickling Filter

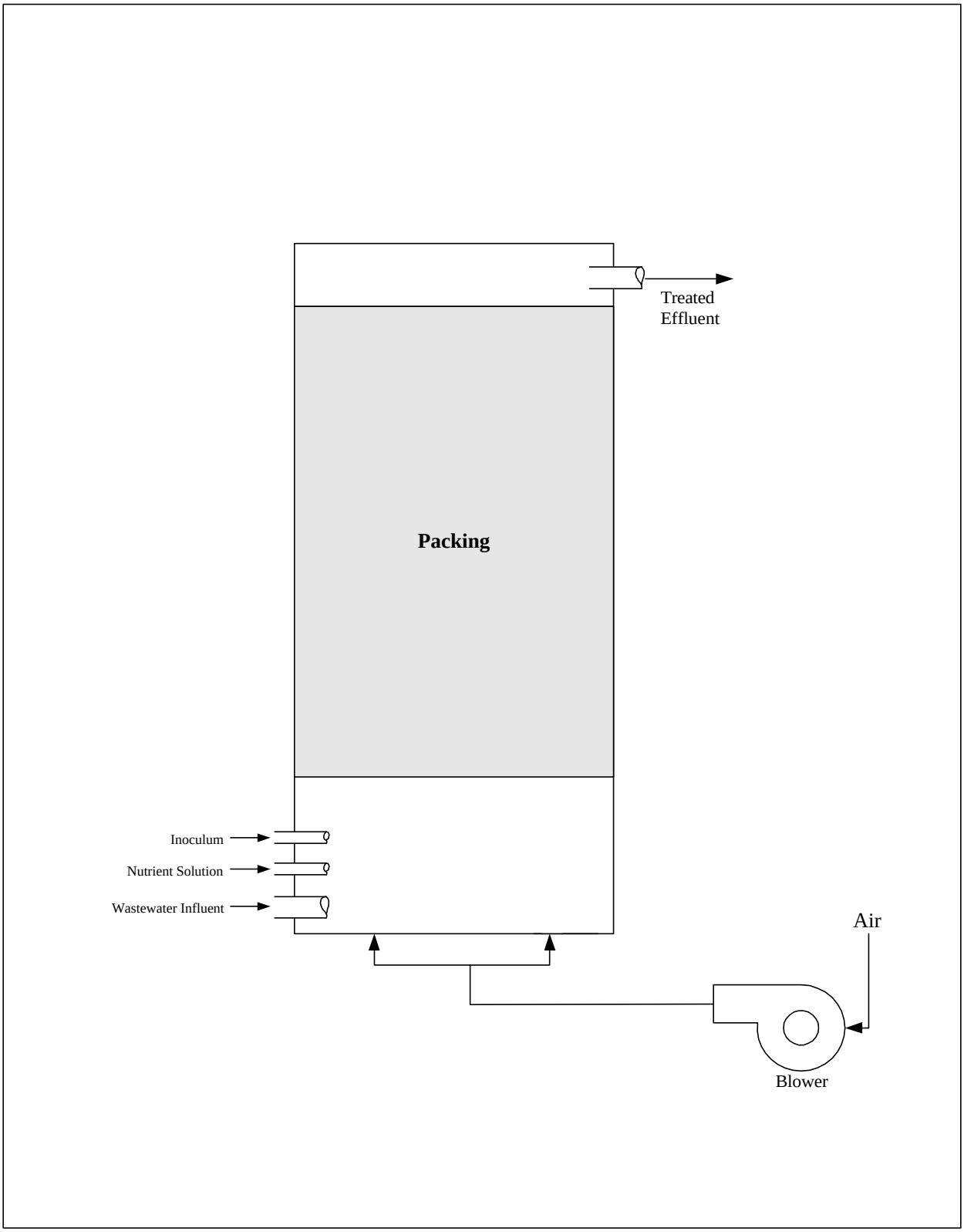


Figure 8-19: Fluidized Bed Reactor

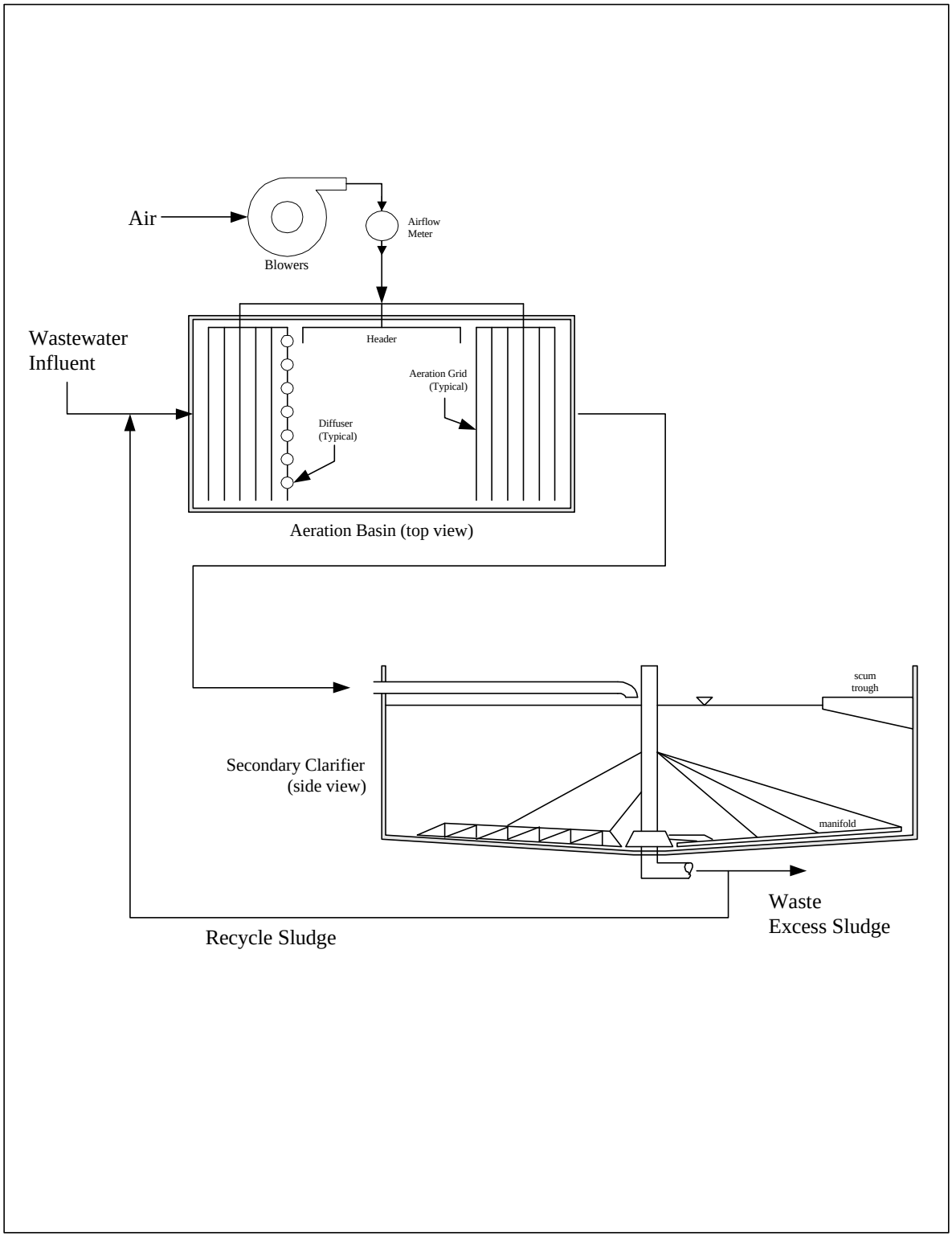


Figure 8-20: Activated Sludge System

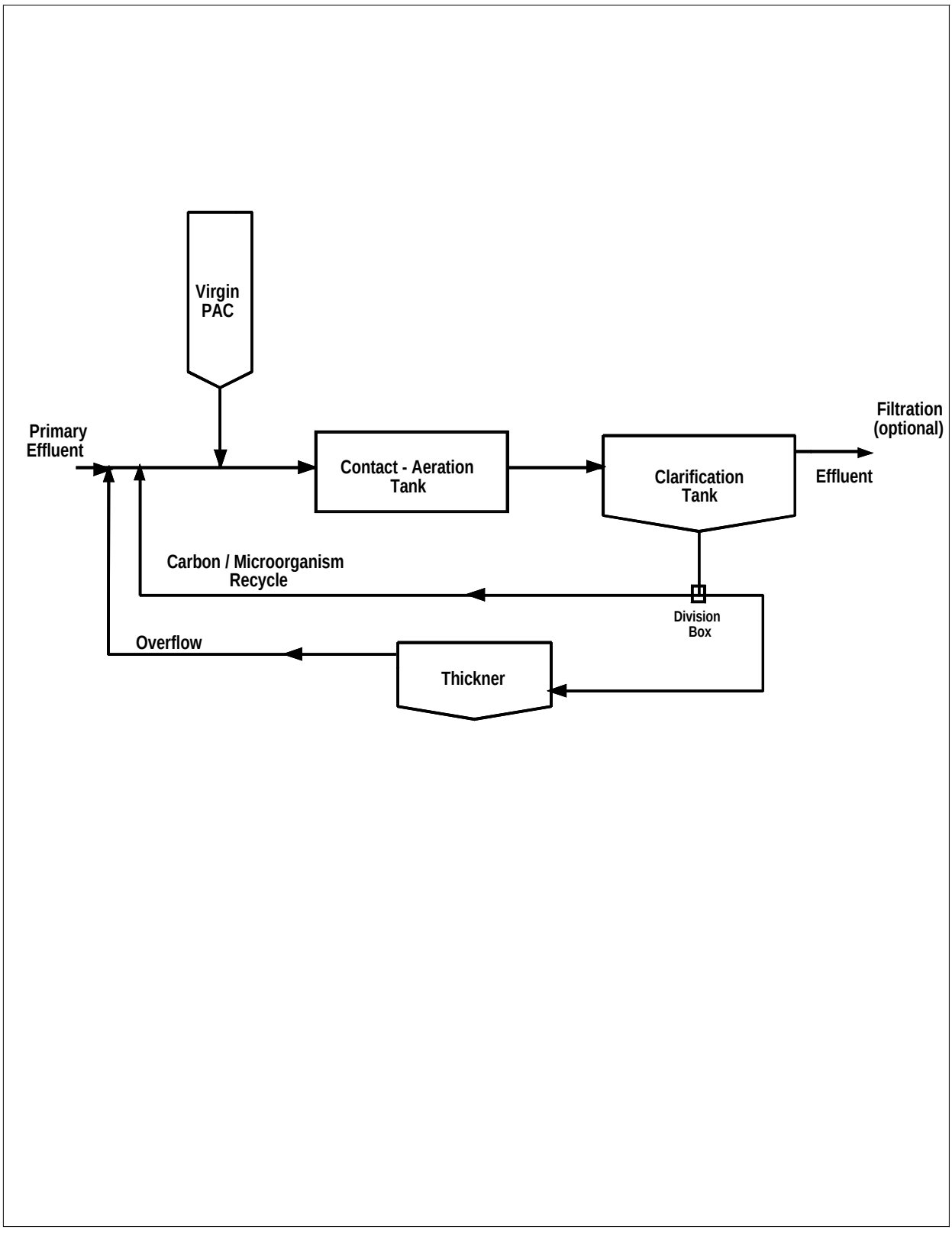


Figure 8-21: Powder Activated Carbon Treatment System

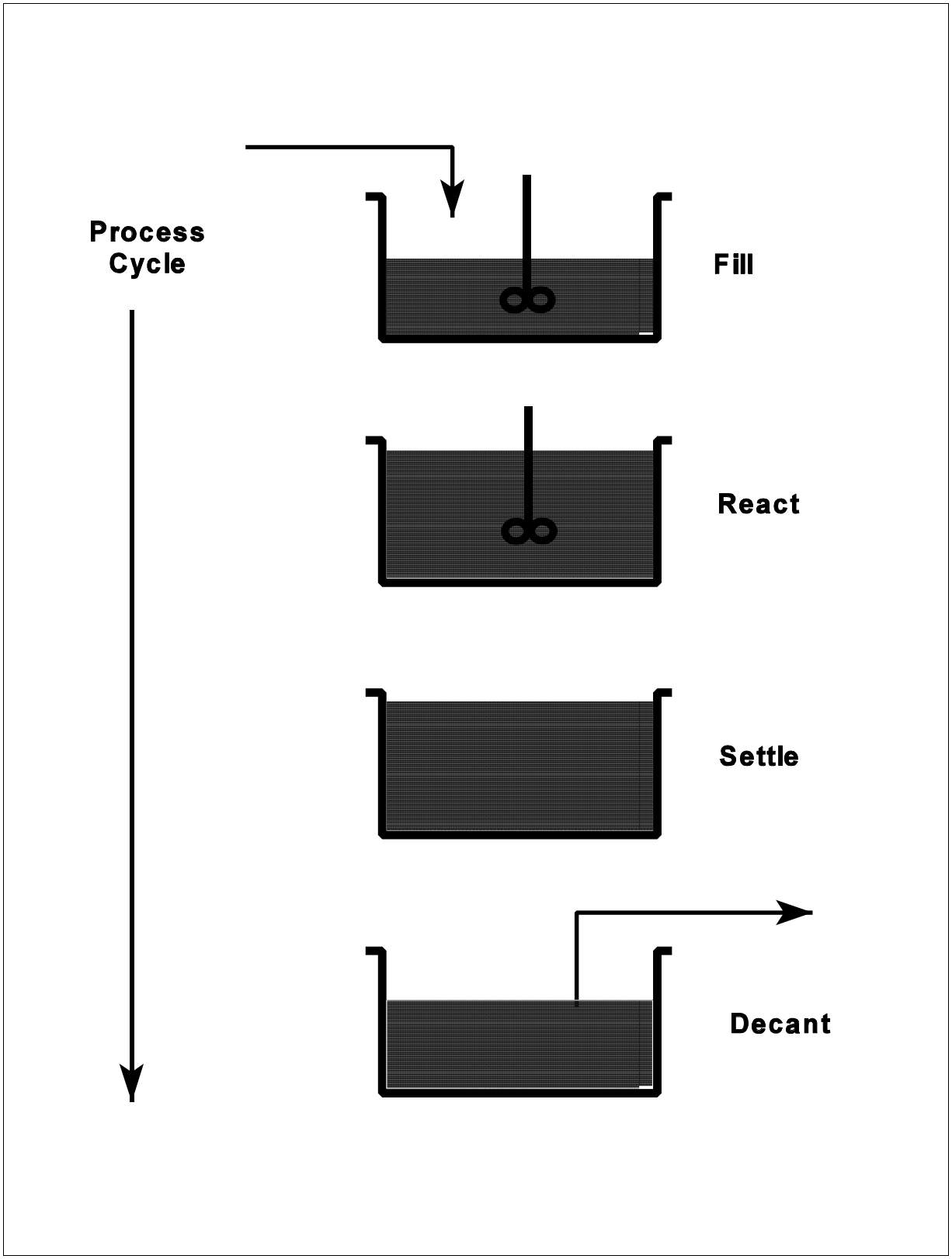


Figure 8-22: Sequencing Batch Reactor Process Diagram

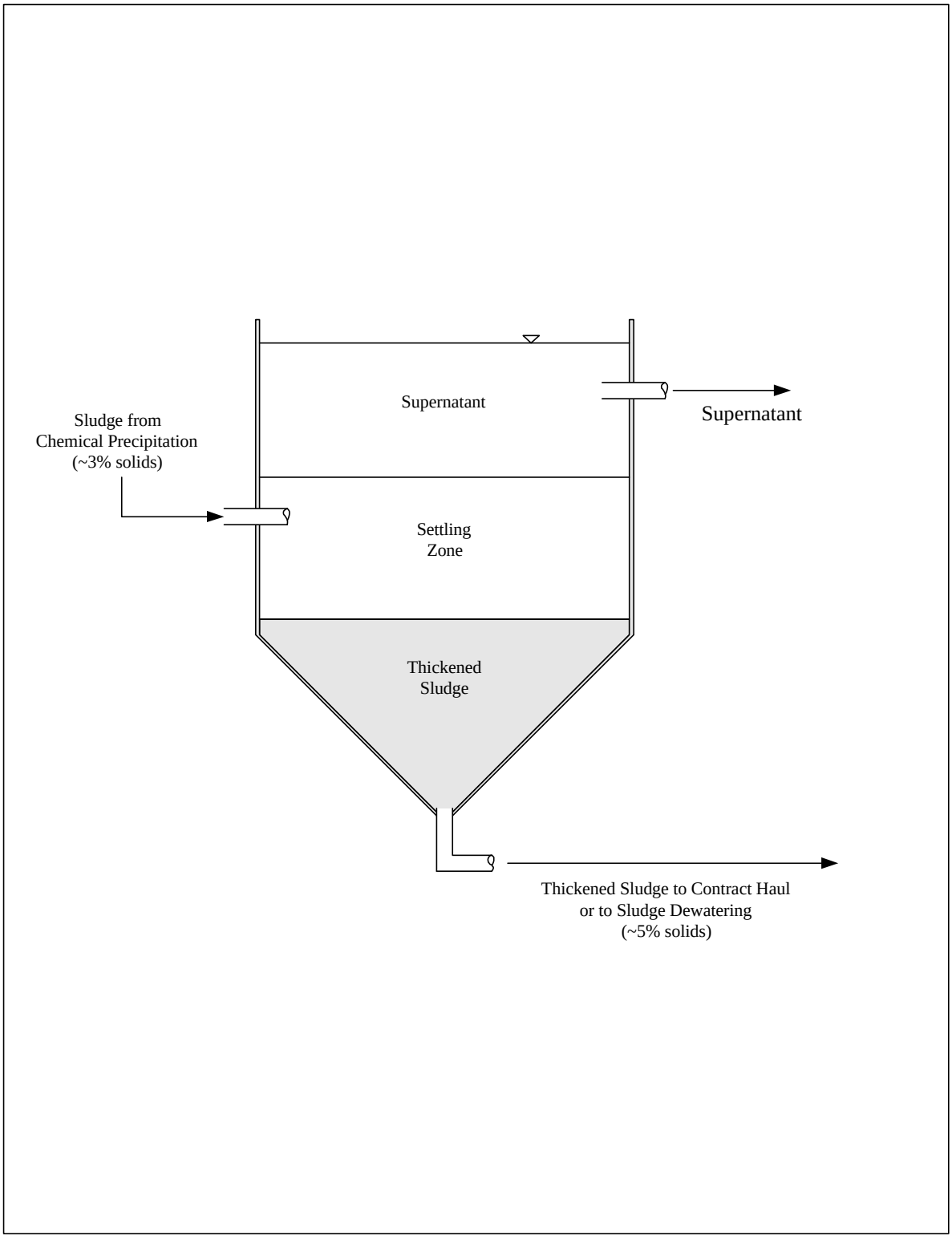


Figure 8-23: Gravity Thickening

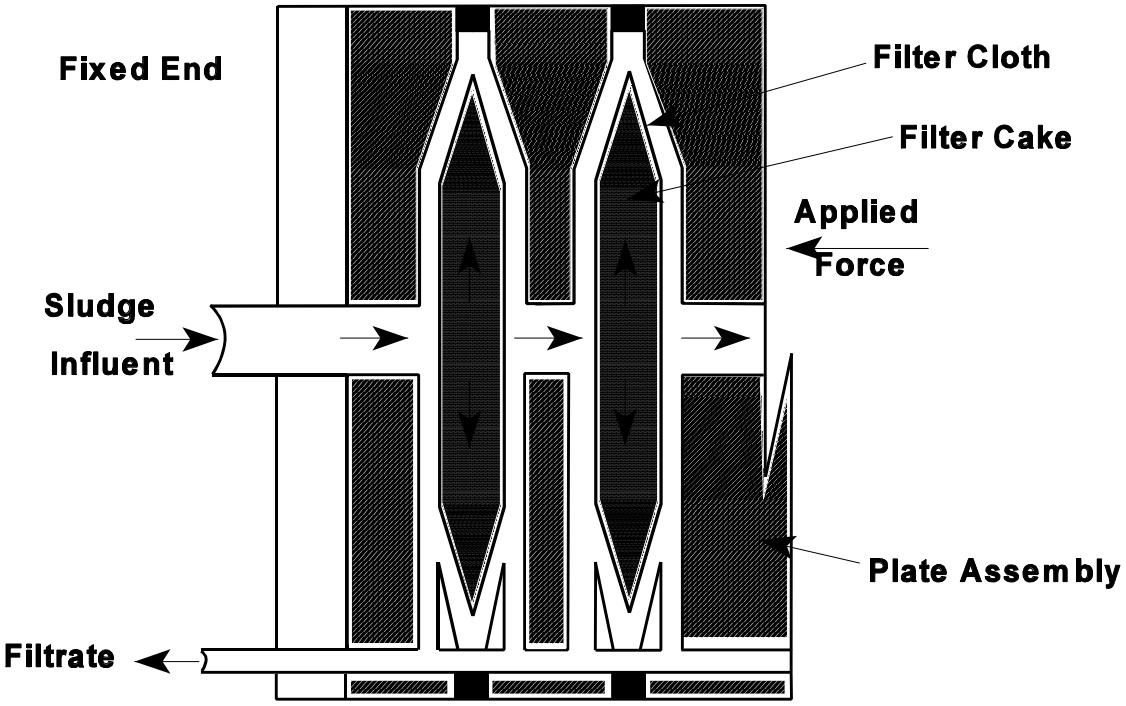


Figure 8-24: Plate-and-Frame Pressure Filtration System Diagram

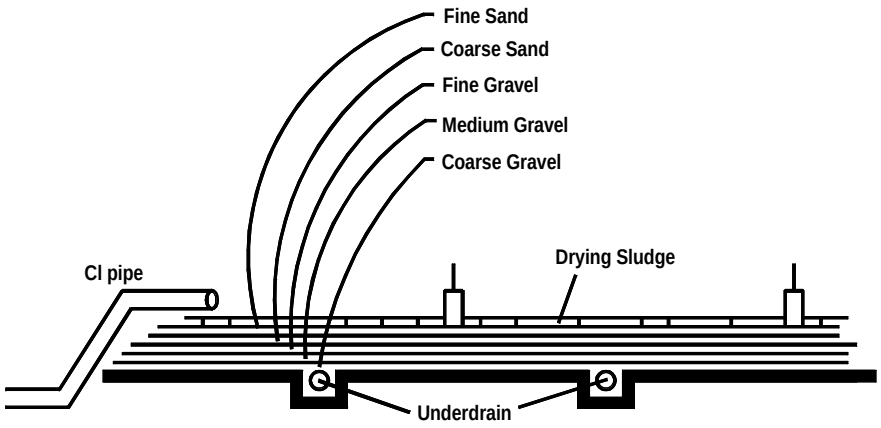


Figure 8-25: Drying Bed

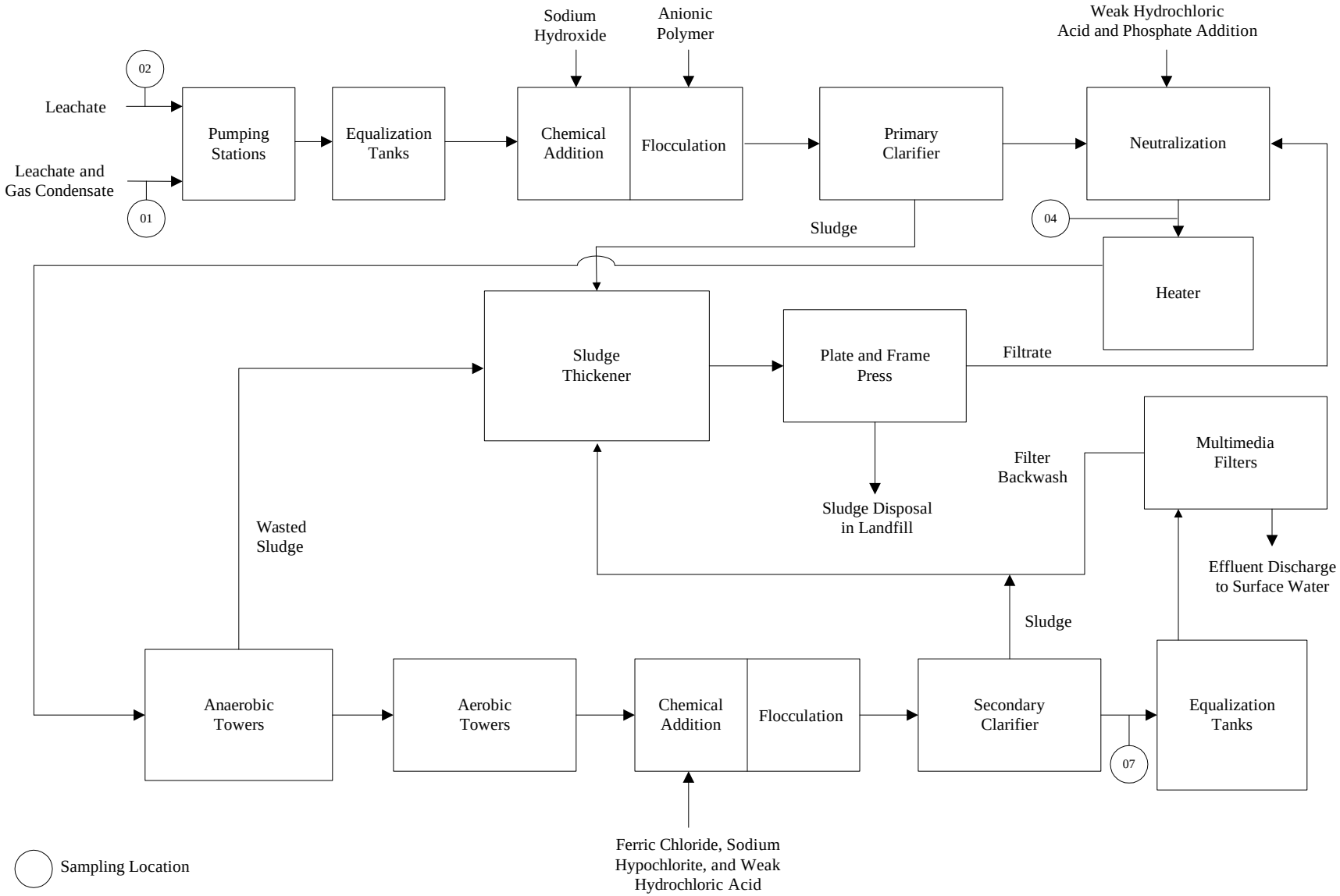


Figure 8-26: EPA Sampling Episode 4626 - Landfill Waste Treatment System Block Flow Diagram with Sampling Locations

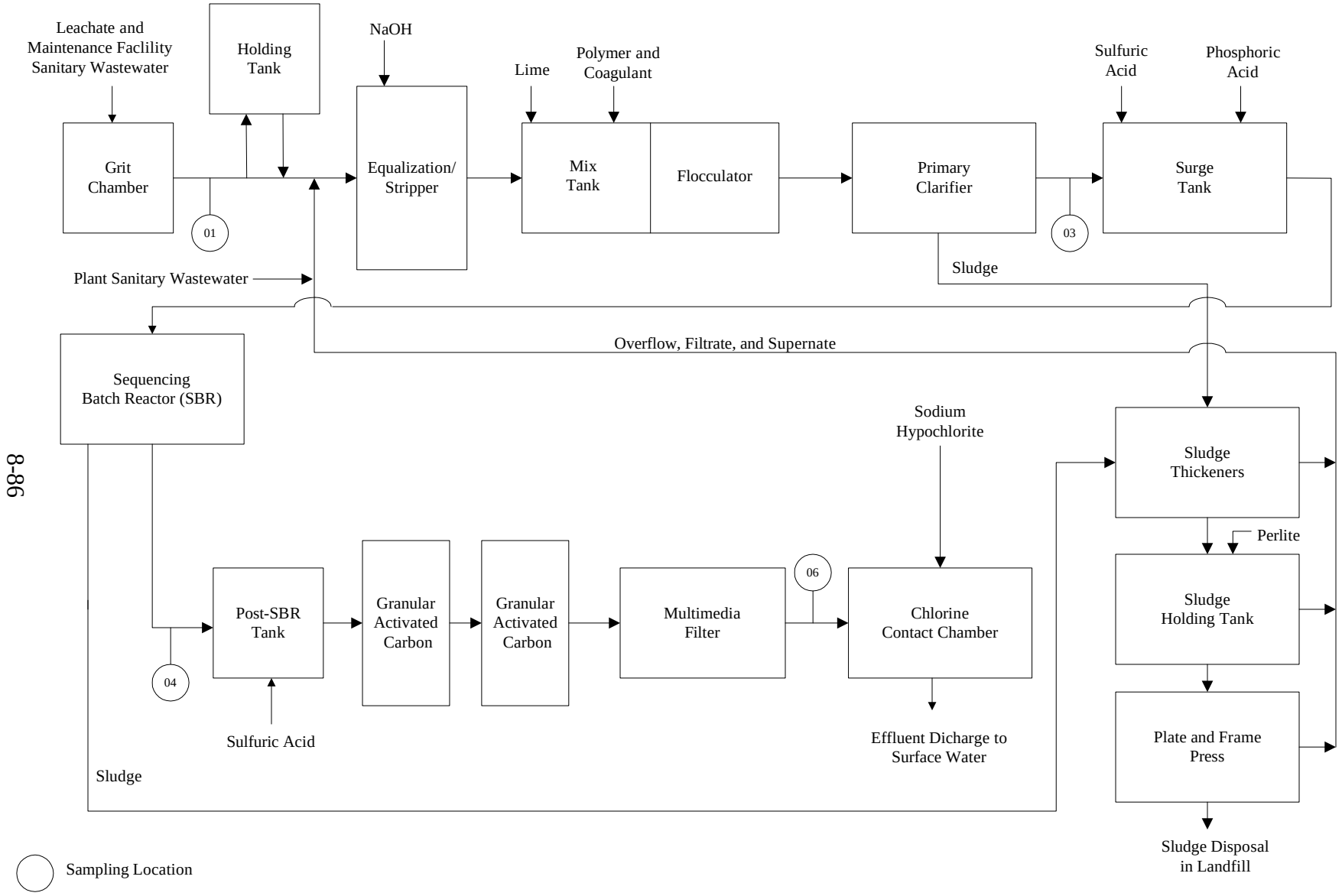


Figure 8-27: EPA Sampling Episode 4667 - Landfill Waste Treatment System Block Flow Diagram with Sampling Locations

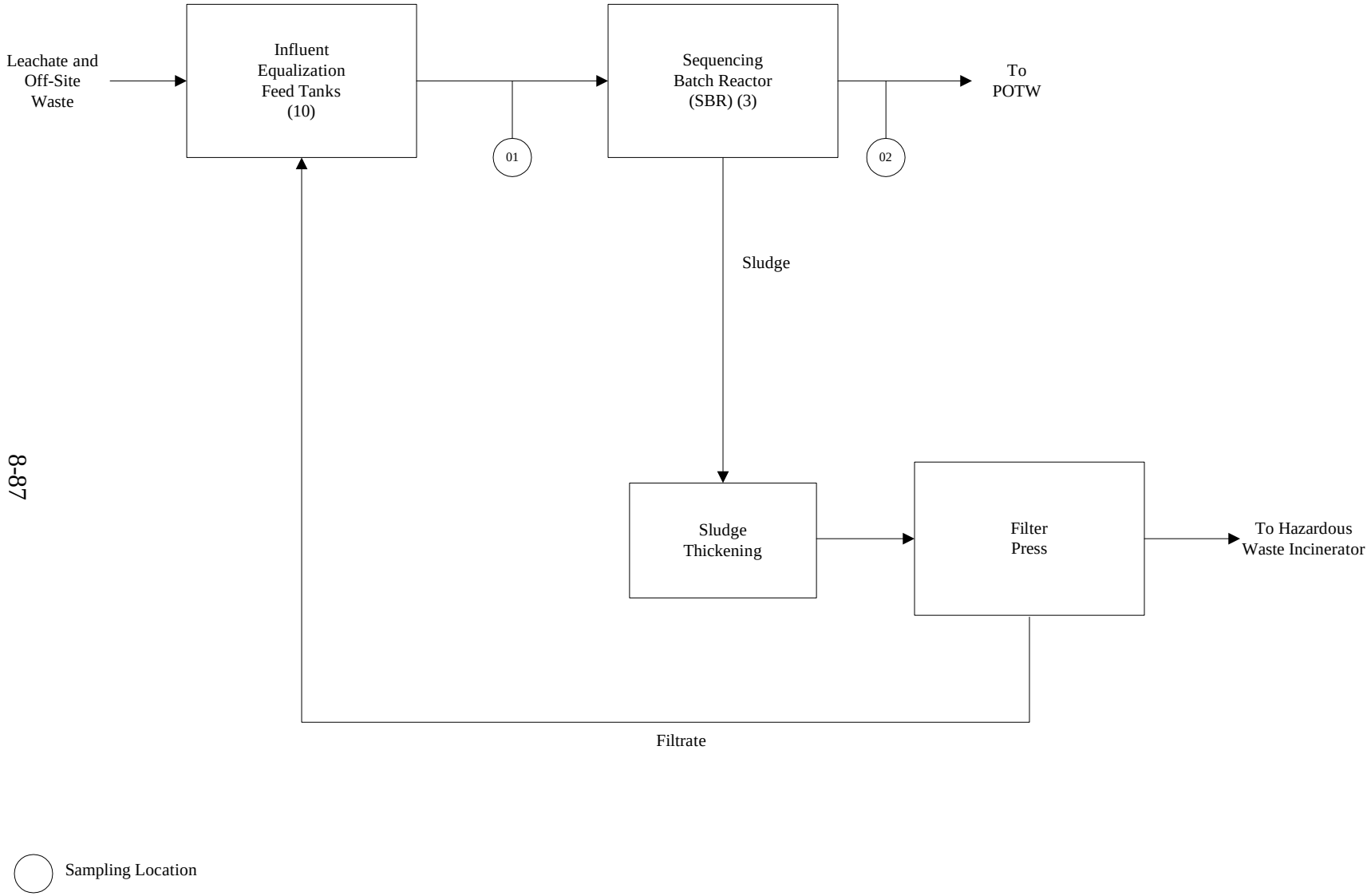


Figure 8-28: EPA Sampling Episode 4721 - Landfill Waste Treatment System Block Flow Diagram with Sampling Locations

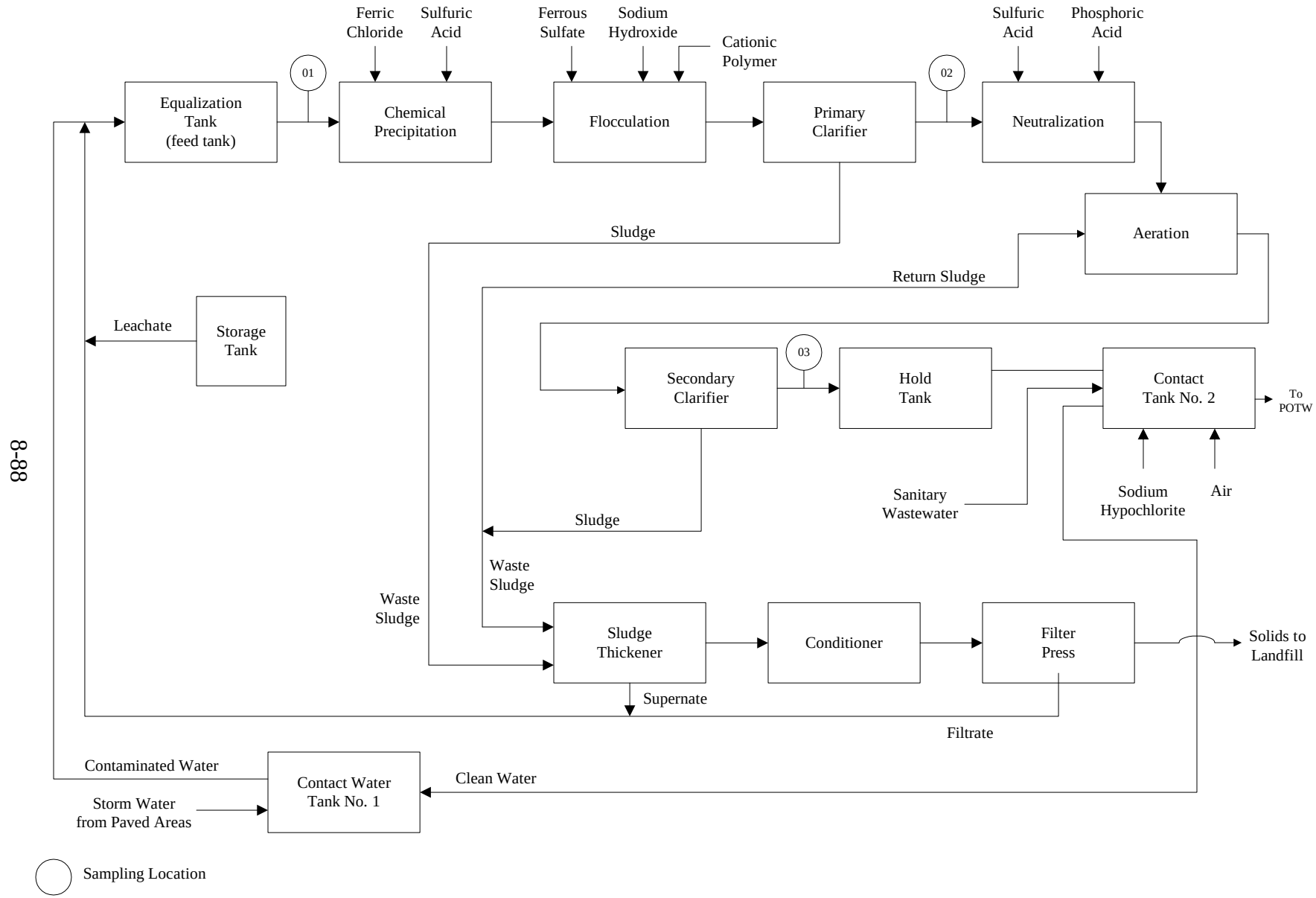


Figure 8-29: EPA Sampling Episode 4759 - Landfill Waste Treatment System Block Flow Diagram with Sampling Locations

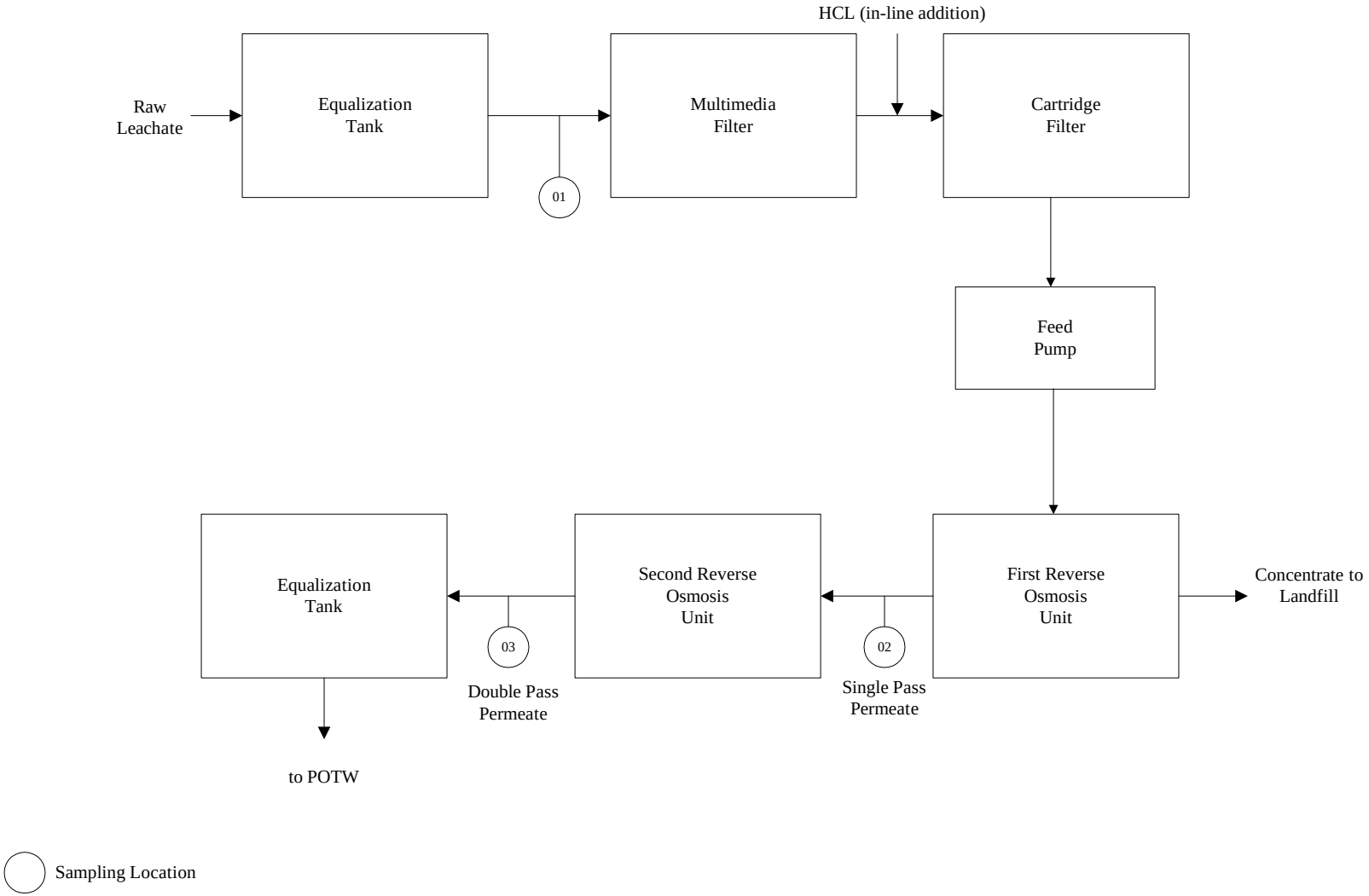


Figure 8-30: EPA Sampling Episode 4687 - Landfill Waste Treatment System Block Flow Diagram with Sampling Locations

9.0 ENGINEERING COSTS

This chapter presents the costs estimated for compliance with the effluent limitations guidelines and standards for the Landfills industry. Section 9.1 provides a discussion of the cost-estimation methodologies considered by EPA including evaluation of two cost-estimation models. Section 9.2 presents a discussion of the types of cost estimates developed, while in Section 9.3, the development of capital costs, operating and maintenance (O&M) costs, and other related costs is described in detail. Section 9.4 summarizes the compliance costs for each regulatory option considered by EPA.

9.1 Evaluation of Cost-Estimation Techniques

This section presents a discussion of the cost-estimation techniques considered by EPA, including evaluation of two cost-estimation models. In this section, the Agency presents the criteria used to evaluate these techniques as well as the results of a benchmark analysis to compare the accuracy of these techniques. This section also presents the selected cost-estimation techniques.

9.1.1 Cost Models

EPA developed compliance-cost estimates for leachate treatment systems to determine the economic impact of the regulation. EPA has identified existing cost-estimation models to facilitate the development of compliance-cost estimates. In a mathematical cost model, various design and vendor data on a variety of treatment technologies are combined and cost equations that describe costs as a function of system parameters, such as flow, are developed for each treatment technology. Using these types of models allows for the generation of compliance-cost estimates for several regulatory options that are based on the iterative addition of treatment technologies and can assist EPA in the selection of options as the basis for the regulations.

EPA evaluated the following two well-known cost models for use in developing costs:

- Computer-Assisted Procedure for the Design and Evaluation of Wastewater Treatment Systems (CAPDET), developed by the U.S. Army Corps of Engineers.

- W/W Costs Program (WWC), Version 2.0, developed by CWC Engineering Software.

CAPDET is intended to provide planning level cost estimates to analyze alternatives in the design of wastewater treatment systems. Modules are used to develop cost estimates for a variety of physical, chemical, and biological treatment unit processes and can be linked together to represent entire treatment trains. Equations in each of these modules are based upon common engineering principles used for wastewater treatment system design. The CAPDET algorithm generates a design based on input parameters selected by the user, calculates cost estimates for various treatment trains, and ranks them based on present worth, capital, operating, or energy costs.

The WWC cost model was developed by Culp/Wesner/Culp from a variety of engineering sources, including vendor supplied data, actual plant construction data, unit takeoffs from actual and conceptual designs, and published data. The model calculates cost estimates for a variety of individual treatment technology units that can be combined together to develop compliance-cost estimates for the complete treatment systems. The WWC model does not design each treatment technology unit but rather prompts the user to provide design-input parameters that form the basis for the cost estimate. The WWC model includes a separate spreadsheet program that provides design criteria guidelines to assist in developing the input parameters to the cost-estimating program. The spreadsheet includes treatment component design equations and is supplied with default parameters that are based upon accepted design criteria used in wastewater treatment, to assist in the design of particular treatment units. The spreadsheet also is flexible enough to allow selected design parameters to be modified to estimate industry-specific factors accurately. Once design inputs are entered into the program, the WWC model calculates both construction and operation and maintenance (O&M) costs for the selected wastewater treatment system.

9.1.2 Vendor Data

For certain wastewater treatment technology units, the cost model was not considered the most accurate

estimate of costs. For these instances, EPA determined that reported equipment and operation and maintenance costs obtained directly from equipment vendors often can provide accurate cost estimates.

EPA provided information on landfill wastewater characteristics to vendors to determine the appropriate treatment unit and accurate sizing. Quotes obtained from vendors included equipment costs that EPA factored up to total capital costs to account for site preparation, mobilization costs, and engineering contingencies. EPA also obtained vendor quotes for operation and maintenance costs, including utility usage and cost. The Agency used vendor quotes to determine cost curves for equalization, multi-media filtration, granular activated carbon, breakpoint chlorination, and reverse osmosis. EPA based the cost curves used for these treatment technologies on direct vendor quotes, commercial costing guides, or cost information developed from vendor quotes as part of the Centralized Waste Treatment (CWT) effluent guidelines effort.

9.1.3 Other EPA Effluent Guideline Studies

EPA reviewed other EPA effluent studies, such as the Organic Chemicals and Plastics and Synthetic Fibers (OCPSF) industry effluent guidelines, to obtain additional costing background and supportive information. However, EPA did not use costs developed as part of other industrial effluent guidelines in costing for this industry, with the exception of the CWT effluent guideline data referenced in Section 9.1.2.

9.1.4 Benchmark Analysis and Evaluation Criteria

EPA performed benchmark analyses to evaluate the accuracy of each cost-estimation technique. This benchmark analysis used reported costs provided in the 308 Questionnaires and compared them to costs generated using each cost-estimation technique. EPA selected four landfill facilities (Questionnaire Identification numbers (QIDs) 16122, 16125, 16041, and 16087) with wastewater treatment systems for the benchmark analysis. The agency developed cost estimates for wastewater treatment units that make up the treatment systems at these landfill facilities using the WWC and CAPDET models and vendor quotes. Next, EPA compared these cost estimates to the reported component costs provided in the 308 Questionnaires to evaluate the accuracy of each methodology in estimating capital and operation and

maintenance costs. This cost comparison is presented in Table 9-1. Treatment technologies that EPA used in this benchmark analysis include the following:

- equalization,
- chemical precipitation,
- activated sludge,
- sedimentation, and
- multi-media filtration.

EPA also benchmarked cost estimates developed using these techniques against reported costs for wastewater treatment systems that included equalization, chemical precipitation, and multimedia filtration and were obtained from industrial waste combustor facilities as part of that effluent guidelines effort. EPA believes that the wastewater characteristics being treated by these treatment systems, i.e., inorganic contaminants and solids in an uncomplexed matrix, are similar for both landfills and industrial waste combustor facilities and that this additional comparison provides a more thorough evaluation of the Agency's cost-estimation methodologies. Table 9-2 presents a comparison of the capital and O&M costs obtained for the wastewater treatment systems at four industrial waste combustor facilities to the cost estimates obtained using each technique, i.e., the WWC and CAPDET models, and vendor quotes.

As shown in Tables 9-1 and 9-2, EPA has determined that, based on the results of the benchmark analyses for both data sources, the WWC model generated cost estimates that are considered more accurate than the CAPDET model when compared to reported treatment technology costs as provided in 308 Questionnaire responses. In all instances, the WWC model estimated the more accurate treatment system capital and O&M costs as compared to CAPDET and vendor costs. For several facilities, such as QIDs 16087, 16122, and 16125, the WWC model generated capital costs to within 32 percent of costs provided in the questionnaires. EPA estimated O&M costs for several facilities, including QIDs 16041, 16087, and 16122, to within 18 percent of costs provided in the 308 Questionnaires.

EPA used the following criteria to evaluate each cost-estimation technique and to select the appropriate option for developing a methodology for estimating compliance costs for the Landfills industry:

- Does the model contain costing modules representative of the various wastewater technologies in use or planned for use in the Landfills industry?
- Can the model produce costs in the expected flow range experienced in this industry?
- Can the model be adapted to cost entire treatment trains used in the Landfills industry?
- Is sufficient documentation available regarding the assumptions and sources of data so that costs are credible and defensible?
- Is the model capable of providing detailed capital and operation and maintenance costs with unit-costing breakdowns?
- Is the model capable of altering the default design criteria in order to accurately represent reported design criteria indicative of the Landfills industry?

9.1.5 Selection of Final Cost-Estimation Techniques

Based upon the results of the benchmark analysis, EPA selected the WWC model for estimating costs for the majority of the treatment technologies that form the basis for BPT/BAT/NSPS effluent limitations and standards. The Agency determined that the WWC model is capable of producing accurate capital and O&M costs for a wide range of treatment technologies. EPA found that the CAPDET model was not capable of generating cost estimates for many of the technologies that form the basis for BPT/BAT/NSPS effluent limitations and standards for the Landfills industry, and the Agency determined that it was not accurate in estimating technology costs for landfill facilities. Therefore, EPA decided not to use the CAPDET model for estimating compliance costs.

EPA has determined that the WWC model best satisfies the selection criteria. The program can estimate costs for a wide range of typical and innovative treatment technologies and can combine these costs of each technology to develop system costs. Since the WWC model is a computer based program, it readily allows for the iterative development of costs for a number of facilities and regulatory options. The program utilizes cost modules that can accommodate the range of flows and design-input parameters needed to develop cost estimates for landfill facilities. Cost estimates generated by this model are based upon a

number of sources, including actual construction and operation costs, along with published data, and are presented in a breakdown summary table that contains unit costs and totals. Finally, the WWC model can be adapted to estimate costs based upon specified design criteria and wastewater flow rates.

EPA notes that there were particular technologies for which the WWC model did not produce accurate cost estimates. These technologies included equalization, multimedia filtration, granular activated carbon, breakpoint chlorination, and reverse osmosis. In some low-flow situations, costs developed for these treatment technologies were excessively high as compared to industry provided costs in 308 Questionnaire responses. For these technologies, EPA determined that vendor quotes provided a more accurate estimate of compliance costs and would be used in the final engineering costing methodology for these technologies.

In addition, in a select few cases, EPA determined that it would be more economically feasible for some facilities to truck/pipe their wastewater off-site for treatment than to construct and maintain their own wastewater treatment system. These facilities had extremely low average daily flow rates (50 gallons or less); therefore, EPA substituted an off-site disposal cost for CWT treatment for BPT/BAT capital and O & M costs (see also 9.2.6).

9.2 Engineering Costing Methodology

This section presents the costing methodology used to develop treatment costs for BPT, BCT, and BAT options for the Landfills industry. This section also presents a description of additional costs, such as monitoring costs, that EPA developed. The following discussion presents a detailed summary of the technical approach used to estimate the compliance costs for each landfill facility. The Agency developed total capital and annual operation and maintenance costs for each facility in its database to upgrade its existing wastewater treatment system, or to install new treatment technologies, to comply with the long term averages for each regulatory option. Development of the long-term averages is discussed in Chapter 11 of this document and in the Statistical Support documents. EPA costed facilities primarily using the WWC model and, on occasion, from cost curves developed from vendor quotes. Table 9-3 presents a breakdown

of the cost-estimation method used for each treatment technology. EPA developed additional costs for monitoring, Resource Conservation and Recovery Act (RCRA) permit modifications, and residual disposal. The Agency developed total facility compliance costs under each BPT, BCT, and BAT option by adding treatment costs with these additional costs. EPA did not develop cost estimates for zero or alternative discharge facilities for any of the regulatory options (with the exception of some low flow facilities, see 9.2.5).

9.2.1 Treatment Costing Methodology

The methodology used to develop facility-specific BPT, BCT, and BAT option-compliance costs is presented graphically on the flow diagram in Figure 9-1. EPA costed facilities for an entire new treatment system, whether or not they had existing treatment at the facility, if the collected flow subject to this guideline was less than 85 percent of the total facility flow rate.

For each regulatory option, EPA evaluated each landfill facility in the Detailed Questionnaire database to determine if the facility would incur costs in order to comply with the regulations. EPA compared the current discharge concentrations of the facility's effluent with the long-term averages from each regulatory option. If the facility's current discharge concentration was less than the long-term average, EPA considered it to be in compliance. A facility considered to be in compliance was projected to incur costs only for additional monitoring requirements. If a facility was not in compliance but had treatment unit operations in-place capable of complying with the long-term averages, EPA costed the facility for system upgrades that would bring the facility into compliance.

For facilities that did not have BPT/BCT/BAT treatment systems or the equivalent, the Agency developed cost estimates for the additional unit operations and/or system upgrades necessary to meet each long term average. Facilities that were already close to compliance with the long-term averages only required an upgrade to achieve compliance with limitations for a regulatory option. EPA developed upgrade costs using the WWC model whenever possible and included either additional equipment to be installed as part

of an existing wastewater treatment system, expansion of existing equipment, or operational changes. Examples of upgrade costs include such items as new or expanded chemical feed systems and improved or expanded aeration systems. If a facility had no treatment system (or one that could not achieve desired levels with upgrades or minor additions) the Agency developed cost estimates for an entire BPT/BCT/BAT treatment system for that facility.

The first step in using the WWC model was to use the design-criteria guidelines spreadsheet to develop input parameters for the computer program. EPA used reported pollutant loadings from the facility whenever possible. If pollutant loadings were not available for a particular parameter, EPA used the estimates of pollutant concentrations in untreated landfill wastewater (see Chapter 6). The Agency also used the facility's baseline flow rate and the regulatory option long-term averages in the design of the unit operation. Certain parameters such as BOD_5, TSS, and ammonia are used directly in the WWC model and the design-criteria guideline spreadsheet to design the various treatment unit operations. EPA selected metals that were included as pollutants of interest to assist in the design of chemical precipitation systems. The metals to be treated typically control the type and amount of precipitating agents, which govern the chemical feed system design. A more detailed discussion of the design parameters and costs associated with individual treatment technologies is presented in Section 9.3.

The design parameters from the design-criteria spreadsheet then were input in the WWC model to generate installed capital and O&M costs. O&M costs for treatment chemicals, labor, materials, electricity, and fuel are included in the WWC model O&M costs. Treatment costs developed using the WWC model were corrected to 1992 dollars using the Engineering News Record published indexes. After EPA developed the installed capital and annual O&M costs for each facility, it applied selected cost factors, as shown in Table 9-4, to the results to develop total capital and O&M costs.

To complete the estimation of compliance costs for each regulatory option, EPA developed cost estimates for other than treatment component costs. The assessment must take into account other costs associated

with compliance with the effluent limitations guidelines and standards, including the following:

- land,
- residual disposal,
- RCRA permit modifications, and
- monitoring.

Each of these additional costs are further discussed and defined in the following sections.

The Agency developed final capital costs for each facility and then amortized them using a seven percent interest rate over 15 years. EPA the added this annualized capital cost to the annual O&M cost to develop a total annual cost for each regulatory option.

9.2.1.1 Retrofit Costs

EPA applied a retrofit cost factor when additional equipment or processes were required for existing systems. Retrofit costs cover the need for system modifications and components, such as piping, valves, controls, etc., that are necessary to connect new treatment units and processes to an existing treatment facility. EPA estimated retrofit costs at 20 percent of the installed capital cost of the equipment.

9.2.2 Land Costs

EPA did not include land costs in this analysis because it determined that landfills have adequate land to accommodate additional treatment systems. Typically, the size of the required treatment system is small when compared to the land area occupied by landfills. Landfills, as required by regulation and permit, have buffer zones around the fill areas. New treatment systems, or upgrades to an existing system, can be installed readily in this buffer zone or elsewhere at the landfill without the need to acquire new land.

9.2.3 Residual Disposal Costs

For each of the proposed treatment system additions or upgrades, EPA estimated a cost for residual disposal. The Agency used two approaches: the first addressed facilities with current sludge-handling

capabilities, while the second addressed facilities without current sludge handling capabilities. EPA prepared residual disposal costs on an annualized basis and added to the total O&M costs.

For facilities with sludge-handling capabilities, EPA evaluated the present solids treatment/dewatering system to determine if it was capable of handling the additional sludge expected to be produced under a particular regulatory option. For facilities with insufficient capacity to handle the additional solids loadings, EPA developed upgrade costs for sludge conditioning and dewatering to account for the additional solids. For facilities with sufficient solids treatment capability, the Agency did not provide additional sludge-treatment costs. For facilities without installed sludge conditioning and dewatering facilities, EPA developed cost estimates for a sludge conditioning and dewatering systems.

Dewatered sludge is assumed to be disposed of on-site in the landfill. EPA's cost estimate also includes the costs associated with the handling and transportation of the sludge to the on-site landfill.

9.2.4 Monitoring Costs

EPA developed costs for the monitoring of treatment system effluent for direct dischargers. The Agency based the costs upon the following assumptions:

- Monitoring costs are based on the number of outfalls through which leachate/ground water is discharged. The costs associated with a single outfall is multiplied by the total number of outfalls to arrive at the total cost for a facility. Monitoring costs estimated by EPA are incremental to the costs already incurred by the facility.
- The capital costs for flow-monitoring equipment are included in EPA's estimates.
- Sample-collection costs (equipment and labor) and sample shipment costs are not included in EPA's estimates because EPA assumes that the facility is already conducting these activities as part of its current permit requirements.

Based upon a review of current monitoring practices at landfills, many conventional and nonconventional parameters, as well as several metals, are already being monitored on a routine basis. EPA developed monitoring costs based upon BOD_5 and TSS monitoring 20 times per month and weekly monitoring of ammonia and other toxic and nonconventional pollutants. In general, these frequencies are higher than currently required. Table 9-5 presents the monitoring cost per sample for the landfill facilities.

9.2.5 Off-Site Disposal Costs

EPA evaluated whether it would be more cost effective for small-flow facilities to have their landfill wastewater hauled off site and treated at a centralized waste treatment facility, as opposed to on-site treatment. EPA compared total annual costs for new or upgraded wastewater treatment facilities to the costs for off-site treatment at a centralized waste treatment facility. Off-site disposal costs were estimated at $0.25 per gallon of wastewater treated. EPA added transportation costs to the off-site treatment costs at a rate of $3.00 per loaded mile using an average distance of 250 miles to the treatment facility. The Agency based transportation costs upon the use of a 5,000-gallon tanker truck load. Facilities that treat their wastewater off site are considered zero or alternative dischargers and, hence, do not incur ancillary costs such as residual disposal, monitoring and permit modifications. EPA then used the lower of the two costs for either on-site or off-site treatment. Table 9-6 presents the facilities that EPA costed using off-site treatment.

9.3 Development of Cost Estimates for Individual Treatment Technologies

In Chapter 8, EPA identified and described the wastewater control and treatment technologies used in the Landfills industry. The following sections describe how EPA developed cost estimates for each of the treatment technologies used in the regulatory options. Specific assumptions regarding the equipment used, flow ranges, input and design parameters, design, and cost calculations are discussed for each treatment technology. Table 9-3, previously referenced, presented the method used to estimate costs for each of treatment technologies used in the BPT, BCT, and BAT options. Table 9-7 presents a summary of the

cost-estimation techniques for each treatment technology for the BPT, BCT, and BAT regulatory options, including the WWC treatment module numbers.

To facilitate the costing of many facilities, EPA developed capital and O&M cost curves for specific technologies and system components. The Agency developed these curves, which represent cost as a function of flow rate or other system design parameters, using a commercial statistical software package (Slidewrite Plus Version 2.1). First, EPA developed costs using the WWC model for each technology or component using, as a design basis, five different flow rates or other system design parameters (depending upon the governing design-parameter). For instance, a technology costed on the basis of flow would have costs estimated using the WWC model at 0.01 million gallons per day (MGD), 0.05 MGD, 0.1 MGD, 0.5 MGD, and 1.0 MGD. EPA based the ranges for the five selected points upon a review of the flow- or technology-design parameters for landfill facilities and selected them to represent the range from low to high. Next, EPA entered these five data points (flow/design parameter and associated cost) into a commercial statistical software program. EPA developed cost curves to model the total capital and O&M costs by the program using curve fitting routines. EPA used a second-order natural-log equation format to develop all curves. All cost curves yielded total capital and O&M costs, unless otherwise noted.

9.3.1 Equalization

EPA conducted a review of questionnaire responses to determine the typical hydraulic detention time for equalization. Based upon of review of industry-furnished data, EPA selected a detention time of 48 hours.

EPA based equalization costs developed for each regulatory option on published price quotes for storage tanks. These costs were taken from the 1996 Environmental Restoration Unit Cost Book published by R.S. Means, Inc. EPA developed a cost curve as a function of flow from these tank quotes. The Agency based construction costs upon published data for an above-ground circular steel tank. EPA also included additional costs associated with a wastewater pumping system and diffused aeration to provide sufficient

mixing of tank contents to prohibit settling. The capital cost curve developed for equalization is presented as Equation 9-1 and is graphically presented in Figure 9-2.

Capital Costs

$$\ln(Y) = 15.177382 + 1.981547\ln(X) + 0.15768\ln(X)^2 \quad (9\text{-}1)$$

where:

X = Flow Rate (MGD), and

Y = Capital Cost (1992 $)

The O&M cost for the equation was taken as a function of the capital cost and is based upon 10 percent of the total capital cost per year.

9.3.2 Flocculation

EPA developed a cost curve for flocculation using WWC unit process 72. Costs for flocculation were a function of flow at a hydraulic detention time of 20 minutes. The capital and O&M cost curves developed for flocculation are presented below as Equations 9-2 and 9-3:

Capital Costs

$$\ln(Y) = 11.744579 + 0.633178\ln(X) - 0.015585\ln(X)^2 \quad (9\text{-}2)$$

O&M Costs

$$\ln(Y) = 8.817304 + 0.533382\ln(X) + 0.002427\ln(X)^2 \quad (9\text{-}3)$$

where:

X = Flow Rate (MGD), and

Y = Cost (1992 $)

Figures 9-3 and 9-4 graphically present the flocculation capital and O&M cost curves, respectively.

EPA based cost estimates for flocculation basins on rectangular-shaped, reinforced concrete structures with a depth of 12 feet and length-to-width ratio of 4:1. The Agency used common wall construction where the total basin volume exceeded 12,500 cubic feet. Vertical-turbine flocculators have higher structural costs than horizontal paddle flocculators because they require structural support above the basin. Horizontal paddles are less expensive and more efficient for use in larger basins, particularly when tapered flocculation is practiced. EPA based manufactured equipment costs on a G value 80 (G is the mean temporal velocity gradient that describes the degree of mixing; i.e., the greater the value of G the greater the degree of mixing). EPA based cost estimates for drive units on variable speed drives for maximum flexibility and, although common drives for two or more parallel basins are often utilized, EPA based the costs on individual drives for each basin.

Energy requirements are based on a G value 80 and an overall motor/mechanism efficiency of 60 percent. The Agency based labor requirements on routine operation and maintenance of 15 minutes/day/basin (maximum basin volume 12,500 cubic ft.) and a 4-hour oil change every 6 months.

9.3.3 Chemical Feed Systems

The following section presents the methodology used to calculate the chemical-addition feed rates used with each applicable regulatory option. Table 9-8 is a breakdown of the design process used for each type of chemical feed. Chemical costs were taken from the September 1992 Chemical Marketing Reporter and are presented in Table 9-9.

For facilities with existing chemical precipitation systems, EPA evaluated the system to determine if it was achieving the regulatory option long-term averages. If the existing system was achieving long-term averages, no additional chemical costs were necessary. However, if the facility was not achieving the long-term averages for an option, EPA estimated costs for an upgrade to the chemical precipitation system.

First, EPA determined the stoichiometric requirements to remove each metal pollutant of interest to the long-term average level. If the current feed rates were within the calculated feed rates, no additional costs were calculated. For facilities currently feeding less than the calculated amounts, EPA estimated costs for an upgrade to add additional precipitation chemicals, such as a coagulant, or expand their existing chemical feed system to accommodate larger dosage rates.

EPA costed facilities without an installed chemical precipitation system for an entire metals precipitation system. The Agency based the chemical feed rates used at a particular facility for either an upgrade or a new system upon stoichiometric requirements, pH adjustments, and the buffering ability of the raw influent.

In the CWT industry guideline, EPA determined that the stoichiometric requirements for chemical addition far outweighed the pH and buffer requirements. EPA determined that 150 percent of the stoichiometric requirement would sufficiently account for pH adjustment and buffering of the solution. The Agency included an additional 50 percent of the stoichiometric requirement to react with metals not on the pollutant of interest list. Finally, EPA added an additional 10 percent increase from the stoichiometric amount as excess. A total of 210 percent of the stoichiometric requirement was estimated when calculating costs for chemical addition systems.

Sodium Hydroxide Feed Systems

The stoichiometric requirement for either lime or hydroxide to remove a particular metal is based upon the following generic equation:

$$lb_{treatment\ chemical} = \left(\frac{lb_{M\ removed}}{year}\right)\left(\frac{valence_{M}}{MW_{M}}\right)\left(\frac{MW_{treatment\ chemical}}{valence_{Na/Ca}}\right)$$

where, M is the target metal and MW is the molecular weight.

The calculated amounts of sodium hydroxide to remove a pound of each of the selected metal pollutants of concern are presented in Table 9-10.

EPA developed sodium hydroxide chemical feed system costs for many facilities using the WWC model. The Agency used reported facility loadings to establish the sodium hydroxide dosage requirement. WWC unit process 45 was used to develop capital and O&M costs for sodium hydroxide feed systems. The capital and O&M cost curves developed for sodium hydroxide feed systems based upon the calculated dosage are presented as Equations 9-4 and 9-5, respectively.

Capital Costs

$$\ln(Y) = 10.653 - 0.184\ln(X) + 0.040\ln(X)^2 \quad (9\text{-}4)$$

O&M Costs

$$\ln(Y) = 8.508 - 0.0464\ln(X) + 0.014\ln(X)^2 \quad (9\text{-}5)$$

where:

X = Dosage Rate (lb/day), and

Y = Cost (1992 $)

Figures 9-5 and 9-6 graphically present the sodium hydroxide feed system capital and O&M cost curves, respectively.

EPA based cost estimates for a sodium hydroxide feed system on WWC unit process 45 for a sodium hydroxide feed rate of between 10 to 10,000 lb/day. EPA based costs on dry sodium hydroxide addition when rates were less than 200 lb/day and on liquid sodium hydroxide when feed rates were higher.

The WWC model assumes that dry sodium hydroxide (98.9 percent pure) is delivered in drums and mixed to a 10 percent solution on site. A volumetric feeder is used to feed sodium hydroxide to one of two tanks: one for mixing the 10 percent solution and one for feeding. Two tanks are necessary for this process because of the slow rate of sodium hydroxide addition due to the high heat of solution. Each tank is equipped with a mixer and a dual-head metering pump, used to convey the 10 percent solution to the point of application. Pipe and valving is required to convey water to the dry sodium hydroxide solution mixing tanks and between the metering pumps and the point of application.

A 50 percent sodium hydroxide solution is purchased premixed and delivered by bulk transport for feed rates greater than 200 lb/day. The 50 percent solution contains 6.38 pounds of sodium hydroxide per gallon and is stored for 15 days in fiberglass reinforced polyester (FRP) tanks. Dual-head metering pumps are used to convey the liquid solution to the point of application, and a standby metering pump is provided in all systems. The storage tanks are located indoors, since 50 percent sodium hydroxide begins to crystallize at temperatures below 54EF.

Phosphoric Acid Feed Systems

In the Subtitle C Hazardous subcategory, phosphoric acid is necessary to neutralize the waste stream and to provide phosphorus to biological treatment systems.

EPA costed the phosphoric acid feed system using the WWC unit process 46. EPA determined that the amount of phosphoric acid necessary to provide nutrient phosphorus was the controlling factor over the amount required for pH adjustment. EPA used a ratio of BOD_5 removed to the amount of phosphorus present in the influent waste stream (100 pounds BOD_5 removed to one pound phosphorus) to determine the amount of phosphoric acid to be added as a nutrient feed to a biological treatment system. To allow for solution buffering, 10 percent excess phosphoric acid was added. The capital and O&M cost curves developed for phosphoric acid feed systems based upon the calculated dosage are presented as Equations 9-6 and 9-7, respectively.

Capital Costs

$$\ln(Y) = 10.042 - 0.155\ln(X) + 0.049\ln(X)^2 \quad (9\text{-}6)$$

O&M Costs

$$\ln(Y) = 7.772 - 0.086\ln(X) + 0.041\ln(X)^2 \quad (9\text{-}7)$$

where:

X = Dosage Rate (gpd), and

Y = Cost (1992 $)

Figures 9-7 and 9-8 graphically present the phosphoric acid feed system capital and O&M cost curves, respectively.

EPA based costs on systems capable of metering 93 percent concentrated acid from a storage tank directly to the point of application. For feed rates up to 200 gpd, the concentrated acid is delivered in drums and stored indoors. At higher flow rates, the acid is delivered in bulk and stored outdoors in FRP tanks. Phosphoric acid is stored for 15 days and a standby metering pump is included for all installations.

Polymer Feed Systems

EPA used WWC unit process 34 to cost for polymer feed systems based upon a dosage rate of 2 mg/L. Although this module estimates costs for a liquid alum feed system, EPA determined that the costs generated by this module were more reasonable and accurate in developing polymer system costs than the WWC unit process 43 for polymer feed systems. The capital and O&M unloaded cost curves developed for polymer feed systems are presented as Equations 9-8 and 9-9, respectively.

Capital Costs

$$\ln(Y) = 10.539595 - 0.13771\ln(X) + 0.052403\ln(X)^2 \quad (9\text{-}8)$$

O&M Costs

$$\ln(Y) = 9.900596 + 0.99703\ln(X) + 0.00019\ln(X)^2 \quad (9\text{-}9)$$

where:

X = Dosage Rate (lb/hr), and

Y = Cost (1992 $)

Figures 9-9 and 9-10 graphically present the polymer feed system capital and O&M cost curves, respectively.

Polymer is stored for 15 days in fiberglass-reinforced polyester tanks. For smaller installations, the tanks are located indoors and left uncovered and, for larger installations, the tanks are covered and vented, with insulation and heating provided. Dual-head metering pumps deliver the polymer from the storage tank and meters the flow to the point of application. Feed costs include 150 feet of 316 stainless steel pipe, along with fittings and valves for each metering pump. A standby metering pump is included for each installation.

9.3.4 Primary Clarification

EPA developed cost curves for primary clarification using WWC unit process 118 for a rectangular basin with a 12 foot side wall depth. EPA based costs for primary clarification upon a function of flow at an overflow rate of 900 gallons per day per square feet tank size. The capital and O&M cost curves developed for primary clarification are presented as Equations 9-10 and 9-11, respectively.

Capital Costs

$$\ln(Y) = 12.517967 + 0.575652\ln(X) + 0.009396\ln(X)^2 \quad (9\text{-}10)$$

O&M Costs

$$\ln(Y) = 10.011664 + 0.268272\ln(X) + 0.00241\ln(X)^2 \quad (9\text{-}11)$$

where:

X = Flow Rate (MGD), and

Y = Cost (1992 $)

Figures 9-11 and 9-12 graphically present the primary clarification capital and O&M cost curves, respectively.

EPA based estimated costs on rectangular basins with a 12 feet side water depth (SWD) and chain-and-flight sludge collectors. Costs for the structure assumed multiple units with common wall construction and include the chain-and-flight collector, collector drive mechanism, weirs, the reinforced concrete structure complete with inlet and outlet troughs, a sludge sump, and sludge-withdrawal piping. Yard piping to and from the clarifier is not included in the cost estimates.

9.3.5 Activated Sludge Biological Treatment

EPA based costs for biological treatment systems using the activated sludge process using the WWC unit process 18 for a rectangular aeration basin with an 10 foot SWD. EPA determined basin size using a 24 hour hydraulic detention time using Equation 9-12.

$$X = ((24 \text{ Hours} \times 3600) \times (Z))/1{,}000 \qquad (9\text{-}12)$$

where:

X = Basin Volume (1,000 cu ft)

Z = Flow Rate (cfs)

The WWC model assumes zero O&M costs for the aeration basins only. The unloaded (without engineering cost factors applied) capital cost curve developed for aeration basins with an 10 foot SWD is presented as Equation 9-13.

$$\ln(Y) = -1.033901 + 3.722693\ln(X) - 0.197016\ln(X)^2 \quad (9\text{-}13)$$

where:

X = Basin Volume (in thousands of cubic feet), and

Y = Capital Cost (1992 $)

Figure 9-13 graphically presents the aeration basin capital cost curve.

Aeration using diffused air was costed for the basin using WWC unit process 26 and reported facility loading conditions. EPA calculated aeration requirements using the facility BOD_5 and ammonia loadings using Equation 9-14.

$$X = ((A + B)/0.075 \times C \times 0.232 \times 1440)/1{,}000 \quad (9\text{-}14)$$

where:

X = Air Requirement (1,000 standard cubic feet per minute [scfm])

A = BOD_5 to Aeration Basin (lb/day) based on 1.8 lb O_2/lb BOD_5 influent

B = Ammonia to Aeration Basin (lb/day) based on 4.6 lb O_2/lb ammonia influent

C = Transfer Efficiency at 9 percent

The unloaded capital and O&M cost curves developed for air diffusion systems are presented as Equations 9-15 and 9-16, respectively.

Capital Costs

$$\ln(Y) = 11.034417 + 0.992985\ln(X) - 0.002521\ln(X)^2 \quad (9\text{-}15)$$

O&M Costs

$$\ln(Y) = 9.497546 + 0.549715\ln(X) - 0.004216\ln(X)^2 \quad (9\text{-}16)$$

where:

X = Air Requirement (1,000 scfm), and

Y = Cost (1992 $)

Figures 9-14 and 9-15 graphically present the air diffusion system capital and O&M cost curves, respectively.

The costs for aeration basins include all equipment, piping, electrical, and labor for installation. The air-supply system costs include piping from air source to aeration basin, blowers, controls, and housing. Aeration-basin cost estimates include excavation, concrete walkways, in-basin process piping, and handrails and attendant costs, but excludes the cost of aeration equipment, electrical and instrumentation work. EPA considered providing for heated aeration basins for facilities located in cold-weather climates. Based upon data collected by EPA, biological treatment of landfill generated wastewater was not adversely affected by climate conditions.

9.3.6 Secondary Clarification

EPA developed cost curves for secondary clarification using WWC unit process 118 for a rectangular basin with a 12 foot side wall depth with chain-and-flight collectors. EPA based costs for secondary clarification upon a function of flow, at an overflow rate of 900 gallons per day per square feet tank size. The capital and O&M cost curves developed for secondary clarification are presented as Equations 9-17 and 9-18, respectively.

Capital Costs

$$\ln(Y) = 12.834601 + 0.688675\ln(X) + 0.035432\ln(X)^2 \quad (9\text{-}17)$$

O&M Costs

$$\ln(Y) = 10.197762 + 0.339952\ln(X) + 0.015822\ln(X)^2 \quad (9\text{-}18)$$

where:

X = Flow Rate (MGD), and

Y = Cost (1992 $)

Figures 9-16 and 9-17 graphically present the secondary clarification capital and O&M cost curves, respectively.

Costs for the structure assumed multiple units with common wall construction, and include the chain-and-flight collector, collector drive mechanism, weirs, the reinforced concrete structure, complete with inlet and outlet troughs, a sludge sump, and sludge-withdrawal piping. Yard piping to and from the clarifier is not included in the cost estimates.

9.3.7 Multimedia Filtration

EPA developed cost curves as a function of flow rate for a multimedia filtration system using vendor-supplied quotes. The Agency developed cost curves as part of the CWT effluent guidelines effort. The capital and O&M cost curves developed for multimedia filtration are presented as Equations 9-19 and 9-20, respectively.

Capital Costs

$$\ln(Y) = 12.265 + 0.658\ln(X) + 0.036\ln(X)^2 \quad (9\text{-}19)$$

O&M Costs

$$\ln(Y) = 10.851 + 0.168\ln(X) + 0.018\ln(X)^2 \quad (9\text{-}20)$$

where:

X = Flow Rate (MGD), and

Y = Cost (1992 $)

Figures 9-18 and 9-19 graphically present the multimedia filtration capital and O&M cost curves, respectively.

The total capital costs for the multimedia filtration systems represent equipment and installation costs. The total construction cost includes the costs of the filter, instrumentation and controls, pumps, piping, and installation. The operation and maintenance costs include energy usage, maintenance, labor, and taxes and insurance. Energy costs include electricity to run the pumps, lighting, and instrumentation and controls. The labor requirement for the multimedia filtration system was four hours per day.

9.3.8 Reverse Osmosis

EPA developed capital and O&M cost curves as a function of flow rate for reverse osmosis treatment using vendor supplied quotes. EPA based costs on one single-pass system using disk tube module technology. The capital cost curve developed for reverse osmosis is presented as Equation 9-21.

$$\ln(Y) = 14.904 - 0.0142\ln(X) - 0.0687\ln(X)^2 \quad (9\text{-}21)$$

where:

X = Flow Rate (MGD), and

Y = Capital Cost (1992 $)

Figure 9-20 graphically presents the reverse osmosis capital-cost curves. Based upon vendor supplied costs, O&M costs were taken at $0.02/gallon.

Costs for a standard reverse osmosis system generally include the following components: filter booster pump, sand or carbon filter, cartridge filter, high-pressure pump and control system, reverse osmosis module permeators, pure water deacidification filter, in-built closed circuit cleaning system, automatic pure water membrane flushing system, power and control system with microprocessor, full instrumentation and measurement equipment, comprehensive fail-safe system, fault indication, and modular skid frame construction. The costs did not take into account the following optional equipment: main raw-water supply pump, pure water tank and distribution pump, chlorine dosing system, ultra-violet disinfection system, containerized/mobile systems, self-contained power supply, and anti-magnetic systems.

9.3.9 Sludge Dewatering

EPA based costs estimated for sludge dewatering upon sludge-drying beds. EPA costed each facility separately using the WWC unit process 128. EPA based the required bed area upon influent characteristics at a loading of 15 gallons per day of sludge per square foot bed area. EPA calculated drying bed area using Equation 9-22.

$$X = (A \times 365)/B \qquad (9\text{-}22)$$

where:

X = Area (sq ft)

A = Total Dry Solids (lb/day) based on 0.8 lb solids/lb BOD_5 influent

B = 15 lb per year sludge/sq ft

The unloaded capital and O&M cost curves developed for sludge-drying beds are presented as Equations 9-23 and 9-24, respectively.

Capital Costs

$$\ln(Y) = 4.488639 + 0.716471\ln(X) + 0.000005311\ln(X)^2 \qquad (9\text{-}23)$$

O&M Costs

$$\ln(Y) = 6.95049 + 0.33155\ln(X) + 0.002882\ln(X)^2 \quad (9\text{-}24)$$

where:

X = Area (sq ft), and

Y = Cost (1992 $)

Figures 9-21 and 9-22 graphically present the sludge-drying bed capital and O&M cost curves, respectively.

Included in the costs are sludge-distribution piping, nine inches of sand media overlying nine inches of gravel media, two foot concrete dividers between beds, and an underdrain system to remove percolating water. EPA excluded land costs from the cost estimates.

Energy requirements are based on the following: a front-end loader to remove dried sludge from the beds and prepare the bed for the next sludge application, cleaning and preparation time of 3 hours for a 4,000 square foot bed, diesel fuel consumption of 4 gallons per hour, and 20 cleanings/bed/year.

9.3.10 Granular Activated Carbon

EPA developed cost curves as a function of flow rate for a granular activated carbon (GAC) system using vendor-supplied quotes. EPA estimated the capital and O&M costs for GAC using the "Power Plant Wastewater Treatment Technology Review Report", Electric Power Research Institute (EPRI), November 1996, Exhibits A3-1 and D3-1, respectively, and supplemented using "Technologies and Costs for Removal of Arsenic from Drinking Water", Office of Ground Water and Drinking Water, EPA, Draft July 1998. The capital and O&M cost curves developed for GAC adsorption are presented as Equations 9-25 and 9-26, respectively.

Capital Costs

$$\ln(Y) = 12.772 + 0.457\ln(X) - 0.025\ln(X)^2 \quad (9\text{-}25)$$

O&M Costs

$$\ln(Y) = 9.691 - 0.224\ln(X) - 0.041\ln(X)^2 \quad (9\text{-}26)$$

where:

X = Flow Rate (MGD), and

Y = Cost (1992 $)

Figures 9-23 and 9-24 graphically present the GAC adsorption capital and O&M cost curves, respectively.

The total capital costs for the GAC systems represent equipment and installation costs. The total construction cost includes the costs of the GAC, instrumentation and controls, pumps, piping, and installation. The operation and maintenance costs include carbon replacement/disposal, energy usage, maintenance, labor, and taxes and insurance. Energy costs include electricity to run the pumps, lighting, and instrumentation and controls. The labor requirement for the GAC system was four hours per day.

9.3.11 Breakpoint Chlorination

EPA developed cost curves as a function of flow rate for a breakpoint chlorination system using vendor-supplied quotes. EPA extrapolated cost estimates for breakpoint chlorination from data supplied by the EPA Office of Ground Water and Drinking Water report. The capital and O&M cost curves developed for a breakpoint chlorination system are presented as Equations 9-27 and 9-28, respectively.

Capital Costs

$$\ln(Y) = 12.219 + 0.051\ln(X) - 0.045\ln(X)^2 \quad (9\text{-}27)$$

O&M Costs

$$\ln(Y) = 12.881 + 0.923\ln(X) + 0.053\ln(X)^2 \quad (9\text{-}28)$$

where:

X = Flow Rate (MGD), and

Y = Cost (1992 $)

Figures 9-25 and 9-26 graphically present the breakpoint chlorination capital and O&M cost curves, respectively.

The total capital costs for the breakpoint chlorination systems represent equipment and installation costs. The total construction cost includes the costs of the chlorine addition unit, instrumentation and controls, pumps, piping, and installation. The operation and maintenance costs include chemical usage, energy usage, maintenance, labor, and taxes and insurance. Energy costs include electricity to run the pumps, lighting, and instrumentation and controls. The labor requirement for the breakpoint chlorination system was four hours per day.

9.4 Costs for Regulatory Options

The following sections present the costs estimated for compliance with the BPT/ BCT/BAT and NSPS effluent limitations guidelines and standards for the Subtitle D Non-Hazardous and Subtitle C Hazardous subcategories. Costs for each of the regulatory options are presented below for only the facilities in the 308 Questionnaire database, as well as for all of the facilities in the Landfills industry based on national estimates (see Chapter 3, Section 3.2.1 for an explanation of national estimates). All costs estimates in this section are expressed in terms of 1992 dollars, unless otherwise noted.

9.4.1 Facility Selection

EPA evaluated each of the 220 Detailed Questionnaires that were returned with sufficient technical and

economic data to determine if the facility would be subject to the final limitations and standards and would, therefore, incur costs as a result of the regulation. EPA determined that 94 of the 220 facilities would not incur costs because of the following reasons:

- 49 facilities indicated that they were zero or alternative discharge
- 40 facilities were operated in conjunction with other industrial or commercial operations and EPA determined that the rule was not applicable to these facilities
- 5 respondents did not generate in-scope wastewater.

EPA calculated costs for each of the remaining 126 facilities and then modeled the national population by using statistically-calculated survey weights. EPA projected the landfill industry costs (presented below) for several technology options based on costs developed for 123 Subtitle D and 3 Subtitle C facilities.

9.4.2 BPT Regulatory Costs

EPA developed preliminary cost-effectiveness analyses using interim costing-rounds to select BPT regulatory options. The BPT costs for each subcategory are presented below.

9.4.2.1 Subtitle D Non-Hazardous Subcategory BPT Costs

Once EPA developed current discharge and untreated landfill wastewater pollutant concentrations for facilities in the Subtitle D Non-Hazardous subcategory, EPA evaluated two options,BPT Options I and II.

BPT Option I: Equalization and activated sludge biological treatment with secondary clarification, and sludge-dewatering. For the facilities in the 308 Questionnaire database, Table 9-11 presents the total capital ($2,737,104) and annual O&M costs ($838,579) for this option, as well as the total amortized annual cost for each facility. Based on national estimates, BPT Option I for the Subtitle D Non-Hazardous subcategory is estimated to have total annualized pre-tax costs of $7.30 million (based on 1998 dollars).

BPT Option II: Equalization, activated sludge biological treatment with secondary clarification, multimedia filtration, and sludge-dewatering. For the facilities in the 308 Questionnaire database, Table 9-12 presents the total capital ($3,252,453) and annual O&M ($1,027,788) costs for this option, as well as the total amortized annual cost for each facility. Based on national estimates, BPT Option II for the Subtitle D Non-Hazardous subcategory is estimated to have total annualized pre-tax and post-tax costs of $8.57 and $7.64 million (based on 1998 dollars), respectively.

9.4.2.2 Subtitle C Hazardous Subcategory BPT Costs

Once EPA developed current discharge and untreated landfill wastewater pollutant concentrations for facilities in the Subtitle C Hazardous subcategory, EPA evaluated one BPT option, BPT Option I.

BPT Option I: Equalization, chemical precipitation, activated sludge biological treatment with secondary clarification, multimedia filtration, and sludge-dewatering. Since EPA did not identify any direct discharge facilities in the Subtitle C Hazardous subcategory database, there are no costs associated with this option.

9.4.3 BCT Regulatory Costs

EPA developed preliminary cost-effectiveness analyses using interim costing-rounds to select BCT regulatory options. The BCT costs for each subcategory are presented below.

9.4.3.1 Subtitle D Non-Hazardous Subcategory BCT Costs

Once EPA developed current discharge and untreated landfill wastewater pollutant concentrations for facilities in the Subtitle D Non-Hazardous subcategory, EPA evaluated two options, BCT Option I and II.

BCT Option I: Equalization and activated sludge biological treatment with secondary clarification, and sludge-dewatering. This option is equivalent to BPT Option I for the Non-Hazardous subcategory with costs previously provided in Section 9.4.2.1 above.

BCT Option II: Equalization, activated sludge biological treatment with secondary clarification, multimedia filtration, and sludge-dewatering. This option is equivalent to BPT Option II for the Non-Hazardous subcategory with costs previously provided in Section 9.4.2.1 above.

9.4.3.2 Subtitle C Hazardous Subcategory BCT Costs

Once EPA developed current discharge and untreated landfill wastewater pollutant concentrations for facilities in the Subtitle C Hazardous subcategory, EPA evaluated one option, BCT Option I.

BCT Option I: Equalization, chemical precipitation, activated sludge biological treatment with secondary clarification, multimedia filtration, and sludge-dewatering. This option is equivalent to BPT Option I for the Subtitle C Hazardous subcategory and, therefore, has no associated costs.

9.4.4 BAT Regulatory Costs

EPA developed preliminary cost-effectiveness analyses using interim costing-rounds to select BAT regulatory options. The BAT costs for each subcategory are presented below.

9.4.4.1 Subtitle D Non-Hazardous Subcategory BAT Costs

EPA costed three BAT options for the Subtitle D Non-Hazardous subcategory: BAT Options I, II and III.

BAT Option I: Equalization and activated sludge biological treatment with secondary clarification, and sludge-dewatering. This option is equivalent to BPT Option I for the Non-Hazardous subcategory with costs previously provided in Section 9.4.2.1 above.

BAT Option II: Equalization, activated sludge biological treatment with secondary clarification, multimedia filtration, and sludge-dewatering. This option is equivalent to BPT Option II for the Non-Hazardous subcategory with costs previously provided in Section 9.4.2.1 above.

BAT Option III: Equalization, activated sludge biological treatment, multimedia filtration, and reverse osmosis with sludge-dewatering. For facilities in the 308 Questionnaire database, Table 9-13 presents the total capital ($34,518,089) and annual O&M ($5,896,531) costs for this option as well as the total amortized annual cost for each facility. Based on national estimates, BAT Option III for the Subtitle D Non-Hazardous subcategory is estimated to have a total annualized pre-tax cost of $45.95 million (based on 1998 dollars).

9.4.4.2 Subtitle C Hazardous Subcategory BAT Costs

Once EPA developed current discharge and untreated landfill wastewater pollutant concentrations for facilities in the Subtitle C Hazardous subcategory, EPA evaluated one BAT option, BPT Option I.

BAT Option I: Equalization, chemical precipitation, activated sludge biological treatment with secondary clarification, multimedia filtration, and sludge-dewatering. This option is equivalent to BPT Option I for the Hazardous subcategory and, therefore, has no associated costs.

9.4.5 NSPS Regulatory Costs

EPA developed preliminary cost-effectiveness analyses using interim costing-rounds to select NSPS regulatory options. The NSPS costs for each subcategory are presented below.

9.4.5.1 Subtitle D Non-Hazardous Subcategory NSPS Costs

EPA is establishing NSPS for the Subtitle D Non-Hazardous subcategory to be equivalent to the limitations established for BPT Option II for this subcategory, which also is the basis for BCT and BAT.

NSPS: Equalization, activated sludge biological treatment with secondary clarification, multimedia filtration, and sludge-dewatering. The total NSPS annual cost for the Non-Hazardous subcategory is $52,755 assuming an average facility flow of 10,000 gpd.

9.4.5.2 Subtitle C Hazardous Subcategory NSPS Costs

EPA is establishing NSPS for the Subtitle C Hazardous subcategory to be equivalent to the limitations established for BPT Option I for this subcategory, which also is the basis for BCT and BAT.

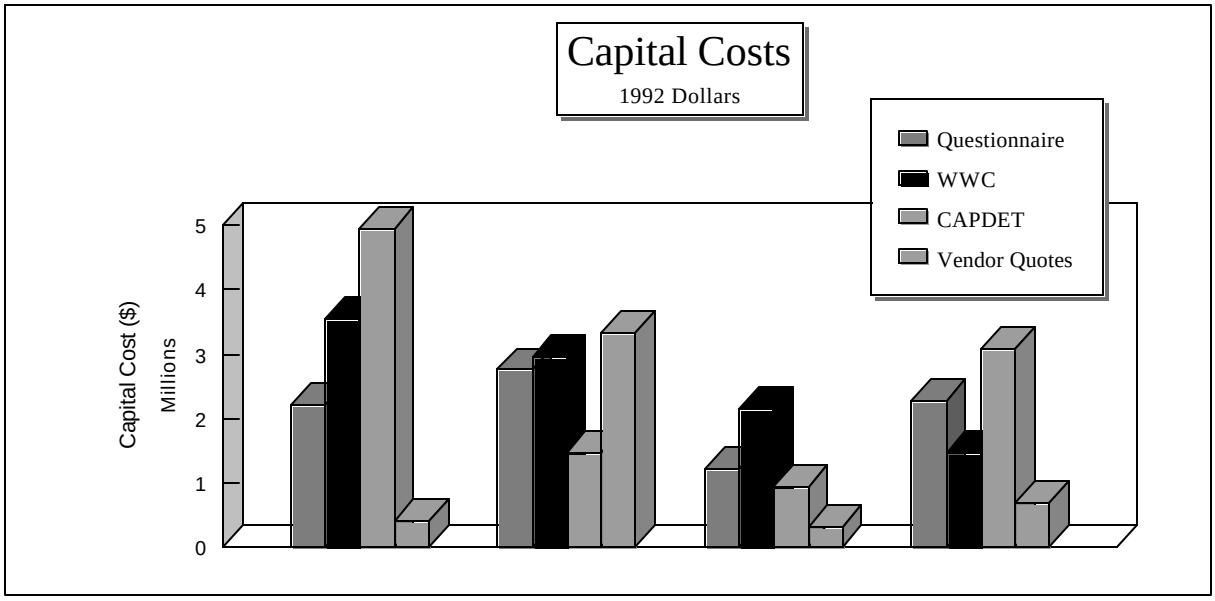
NSPS: Equalization, chemical precipitation, activated sludge biological treatment with secondary clarification, multimedia filtration, and sludge-dewatering. The total NSPS annual cost for the Hazardous subcategory is $132,031 assuming an average facility flow of 10,000 gpd.

Table 9-1: Cost Comparison

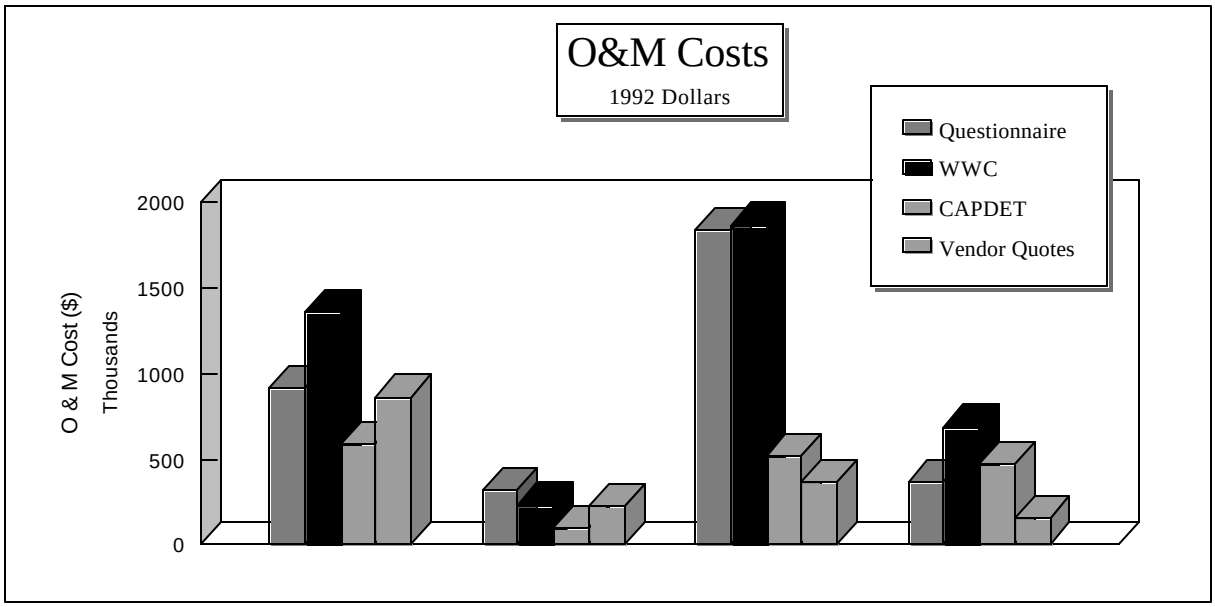
Facility QID	Treatment Train	CAPDET Computer Run		WWC Engineering Software		Vendor Quotes		Questionnaire Responses	
		Capital Cost 1992	O&M Costs	Capital Cost 1992	O&M Costs	Capital Cost 1992	O&M Costs	Capital Cost 1992	O&M Costs
16122	Chemical Precipitation	$232,366	$178,773	$190,308	$41,883	$177,504	$163,397	NA	$22,858
	Above+Anaerobic&Aerobic Bio	$1,217,370	$353,181	$836,433	$79,898	$794,343	$305,669	NA	$133,314
	Above+2nd Chemical Precipitation	$1,449,732	$587,637	$908,201	$91,295	$971,847	$469,066	NA	$133,872
	Above+Equalization+Multimedia Filter	$1,517,811	$715,088	$1,573,621	$91,295	$1,553,010	$543,840	NA	$133,872
	Equalization	$58,478	$69,475	$692,252	$1,997	$526,532	$36,442	NA	$3,388
	Entire Treatment Train	$1,576,289	$784,563	$2,782,188	$317,747	$2,154,117	$586,240	$4,113,628	$311,400
16125	Equalization+Air Stripping	$57,717	$61,556	$394,570	$20,718	$243,800	$54,147	$588,714	$8,247
	Chemical Precipitation+SBR	$282,073	$255,294	$1,928,245	$103,100	(a)	(a)	$2,067,188	$31,534
	Above+Carbon+Multimedia Filter	$478,266	$460,622	$2,492,431	$145,949	(b)	(b)	$2,534,242	$34,883
16087	Entire Treatment Train	NA	NA	$2,519,307	$816,351	(c)	(c)	$2,423,057	$992,578
16041	SBR+Sludge Equipment	$159,908	$115,066	$2,378,898	$436,879	NA	NA	$6,293,919	$460,050

NA: Not Available
(a): Capital O&M costs without the SBR are $82,675 and $56,972, respectively
(b): Capital O&M costs without the SBR are $140,078 and $106,642, respectively
(c): Capital O&M costs without the activated sludge system and chlorine addition are $189,120 and $100,849, respectively

Table 9-2: Costing Source Comparison



	Chem Precip	Chem Precip and Filtration	Chem Precip	2-stage Chem Precip
Questionnaire	2,206,980	2,751,204	1,214,563	2,265,009
WWC	3,543,264	2,950,035	2,144,446	1,476,821
CAPDET	4,948,779	1,475,480	942,216	3,072,253
Vendor Quotes	399,878	3,314,930	319,206	670,158



	Chem Precip	Chem Precip and Filtration	Chem Precip	2-stage Chem Precip
Questionnaire	910,000	315,000	1,837,000	363,000
WWC	1,355,505	231,728	1,864,219	686,360
CAPDET	585,855	99,036	515,859	466,848
Vendor Quotes	860,867	222,135	361,623	151,889

Table 9-3: Breakdown of Costing Method by Treatment Technology

Treatment Technology	Cost Using WWC Program	Cost Using Vendor Quotes	Key Design Parameter(s)
Equalization		X(a)	Flow rate
Flocculation	X		Flow rate
Chemical Feed System	X		Flow rate & Pollutant of Interest Metals
Primary & Secondary Clarification	X		Flow rate
Activated Sludge	X		Flow rate, BOD_5, & Ammonia
Reverse Osmosis		X	Flow rate
Multimedia Filtration		X(b)	Flow rate
Activated Carbon		X(c)	Flow rate
Breakpoint Chlor.		X(d)	Flow rate
Sludge-Drying Beds	X		Flow rate, TSS & BOD_5

(a) Based upon costs provided in Environmental Restoration Unit Cost Book

(b) Cost curves developed using vendor quotes in the CWT guideline effort

(c) Based upon costs provided in "Power Plant Wastewater Treatment Technology Review Report", Electric Power Research Institute (EPRI), November 1996, Exhibits A3-1 and D3-1, respectively, and supplemented using "Technologies and Costs for Removal of Arsenic from Drinking Water", Office of Ground Water and Drinking Water, EPA, Draft July 1998

(d) Costs were extrapolated from data supplied by the EPA Office of Ground Water and Drinking Water report

Table 9-4: Additional Cost Factors

Type	Factor	Percent of Capital Cost
Capital	Site Work & Interface Piping	18
	General Contractor Overhead	10
	Engineering	12
	Instrumentation & Controls	13
	Buildings	6
	Site Improvements	10
	Legal, Fiscal, & Administrative	2
	Interest During Construction	9
	Contingency	8
	Retrofit (if necessary)	20
O&M	Taxes & Insurance	2[1]

(1) 2 percent of total capital costs, which includes WWC costs and capital costs listed above.

Table 9-5: Analytical Monitoring Costs

Pollutants	Cost/Sample ($)[1]
Subtitle D Non-Hazardous	
Ammonia as N	18.00
BOD_5	15.00
TSS	6.00
Metals & Organics	105.00
Subtitle C Hazardous	
Ammonia as N	18.00
BOD_5	15.00
TSS	6.00
Metals & Volatile/Semi-Volatile Organics	1600.00

(1) Cost based on 1995 analytical laboratory costs adjusted to 1992 dollars.

Table 9-6: Subtitle D Non-Hazardous Facilities Costed for Off-Site Disposal

Facility QID	Flow (gpd)	Off-Site Disposal Cost ($/yr)
16048	5	730
16055	8	1168
16062	50	7300
16139	50	7300
16148	77	11242
16160	137	20002
16250	200	29200

Table 9-7: Unit Process Breakdown by Regulatory Option

Treatment Technology Description	Subcategory		WWC Unit Process #*	WWC Unit Process # Description
	Non-Hazardous	Hazardous		
Equalization & activated sludge	BPT/BCT/BAT Option I		NA	equalization
			18	aeration basin
			26	aeration system
			118	secondary clarification
			128	sludge dewatering
Equalization, activated sludge & multimedia filtration	BPT/BCT/BAT Option II NSPS		NA	equalization
			18	aeration basin
			26	aeration system
			118	secondary clarification
			NA	multimedia filtration
			128	sludge dewatering
Equalization, activated sludge, multimedia filtration & single-stage reverse osmosis	BAT Option III		NA	equalization
			18	aeration basin
			26	aeration system
			118	secondary clarification
			NA	multimedia filtration
			NA	single-stage reverse osmosis
			128	sludge dewatering
Equalization, chemical precipitation, activated sludge & multimedia filtration		BPT/BCT/BAT Option I NSPS	NA	equalization
			72	flocculation tank
			45	sodium hydroxide feed system
			34	polymer feed system
			118	primary clarification
			46	phosphoric acid feed system
			18	aeration basin
			26	aeration system
			118	secondary clarification
			NA	multimedia filtration
			128	sludge dewatering

*NA=Not Applicable-Vendor Quotes Used

Table 9-8: Chemical Addition Design Method

Chemical	Basis for Design	
	Stoichiometry	Reference[1] (mg/L)
Sodium Hydroxide	X	
Polymer		2.0
Phosphoric Acid	X	

(1) From: Industrial Water Pollution Control, 2nd Edition.

Table 9-9: Treatment Chemical Costs

Treatment Chemical	Cost
Sodium Hydroxide	$350/ton
Polymer	$2.25/lb
Phosphoric Acid	$300/ton

Table 9-10: Sodium Hydroxide Requirements for Chemical Precipitation

Pollutant	Dosage Rate Sodium Hydroxide (lb/lb metal removed)
Cadmium	0.71
Chromium, total	2.31
Iron	2.15
Nickel	2.04
Zinc	1.22
Phosphorus	6.46

Table 9-11: BPT/BCT/BAT Option I Subtitle D Non-Hazardous Subcategory

		CAPITAL COSTS ($)						AMORTIZED	O&M COSTS ($/YR)				TOTAL
ID#	Flow (MGD)	Equipment	Sludge Handling	Retrofit	Permit Modification	Land	Total Capital	TOTAL CAPITAL(a) ($/YR)	Equipment	Solids Handling	Monitoring	Total O&M	ANNUAL COST ($/YR)(b)
16001	0.0793	153,015	2,004	31,004	0	0	186,023	20,424	19,637	4,078	11,540	35,255	55,679
16003	0.00472	0	0	0	0	0	0	0	0	0	0	0	0
16008	0	0	0	0	0	0	0	0	0	0	0	0	0
16009	0.01613	0	0	0	0	0	0	0	0	0	0	0	0
16011	0	0	0	0	0	0	0	0	0	0	0	0	0
16012	0.00221	0	0	0	0	0	0	0	0	0	0	0	0
16013	0.015	0	0	0	0	0	0	0	0	0	0	0	0
16014	0	0	0	0	0	0	0	0	0	0	0	0	0
16015	0.0005	0	0	0	0	0	0	0	0	0	0	0	0
16016	0.0023	0	0	0	0	0	0	0	0	0	0	0	0
16020	0.04581	0	0	0	0	0	0	0	0	0	0	0	0
16023	0.05734	0	0	0	0	0	0	0	0	0	0	0	0
16024	0.00592	0	0	0	0	0	0	0	0	0	0	0	0
16025	0	0	0	0	0	0	0	0	0	0	0	0	0
16026	0	0	0	0	0	0	0	0	0	0	0	0	0
16027	0	0	0	0	0	0	0	0	0	0	0	0	0
16028	0.01985	0	0	0	0	0	0	0	0	0	0	0	0
16029	0.025	0	0	0	0	0	0	0	0	0	0	0	0
16033	0.0091	0	0	0	0	0	0	0	0	0	0	0	0
16035	0	0	0	0	0	0	0	0	0	0	0	0	0
16038	0.00822	0	0	0	0	0	0	0	0	0	0	0	0
16039	0.00178	0	0	0	0	0	0	0	0	0	0	0	0
16043	0.00218	0	0	0	0	0	0	0	0	0	0	0	0
16044	0	0	0	0	0	0	0	0	0	0	0	0	0
16046	0	0	0	0	0	0	0	0	0	0	0	0	0
16047	0.00115	38,175	2,004	0	0	0	40,179	4,411	8,760	1,917	11,540	22,217	26,628
16048	5E-06	0	0	0	0	0	0	0	0	0	0	0	730
16049	0.0017	35,037	2,004	7,408	0	0	44,449	4,880	8,302	2,208	11,540	22,050	26,930
16050	0.01	58,533	2,004	0	0	0	60,537	6,647	11,672	1,917	11,540	25,129	31,776
16052	0.0546	217,678	5,563	44,648	0	0	267,889	29,413	17,799	6,897	11,072	35,768	65,180
16053	0.00124	39,625	2,004	0	0	0	41,629	4,571	9,002	1,917	11,540	22,459	27,030
16054	0.00075	16,544	2,004	3,710	0	0	22,258	2,444	5,276	1,917	11,357	18,550	20,994
16055	8E-06	0	0	0	0	0	0	0	0	0	0	0	1,168
16056	0.00137	40,636	2,004	0	0	0	42,640	4,682	8,921	1,917	11,540	22,378	27,060
16058	0.003	44,348	2,004	9,270	0	0	55,622	6,107	8,936	1,917	0	10,853	16,960
16059	0.0011	38,017	2,004	0	0	0	40,021	4,394	8,730	1,917	11,540	22,187	26,581
16060	0.0018	43,919	2,004	0	0	0	45,923	5,042	9,178	2,208	11,540	22,926	27,968

Table 9-11: BPT/BCT/BAT Option I Subtitle D Non-Hazardous Subcategory (continued)

		CAPITAL COSTS ($)						AMORTIZED	O&M COSTS ($/YR)				TOTAL
ID#	Flow (MGD)	Equipment	Sludge Handling	Retrofit	Permit Modification	Land	Total Capital	TOTAL CAPITAL(a) ($/YR)	Equipment	Solids Handling	Monitoring	Total O&M	ANNUAL COST ($/YR)(b)
16061	0	0	0	0	0	0	0	0	0	0	0	0	0
16062	0.00005	0	0	0	0	0	0	0	0	0	0	0	7,300
16063	0.0067	75,309	2,004	0	0	0	77,313	8,489	11,152	3,562	11,540	26,254	34,742
16064	0.01197	62,083	2,004	0	0	0	64,087	7,036	12,127	3,931	11,540	27,598	34,634
16065	0.008	71,448	2,004	14,690	0	0	88,143	9,678	10,481	3,231	11,090	24,802	34,480
16068	0	0	0	0	0	0	0	0	0	0	0	0	0
16070	0.00133	0	0	0	0	0	0	0	0	0	0	0	0
16071	0.006	0	0	0	0	0	0	0	0	0	0	0	0
16072	0	0	0	0	0	0	0	0	0	0	0	0	0
16073	0.0182	0	0	0	0	0	0	0	0	0	0	0	0
16074	0	0	0	0	0	0	0	0	0	0	0	0	0
16075	0.01021	0	0	0	0	0	0	0	0	0	0	0	0
16076	0	0	0	0	0	0	0	0	0	0	0	0	0
16077	0.00816	0	0	0	0	0	0	0	0	0	0	0	0
16078	0.00499	0	0	0	0	0	0	0	0	0	0	0	0
16079	0.11247	344,770	0	68,954	0	0	413,724	45,425	23,219	0	11,180	34,399	79,824
16083	0.001	29,000	2,004	6,201	0	0	37,205	4,085	7,835	1,735	11,540	21,110	25,195
16084	0.00643	0	0	0	0	0	0	0	0	0	0	0	0
16085	0.03	0	0	0	0	0	0	0	0	0	0	0	0
16088	0.03621	0	0	0	0	0	0	0	0	0	0	0	0
16090	0.00393	0	0	0	0	0	0	0	0	0	0	0	0
16091	0.2321	0	0	0	0	0	0	0	0	0	0	0	0
16092	0.00668	0	0	0	0	0	0	0	0	0	0	0	0
16093	0.08158	222,598	0	44,520	0	0	267,118	29,328	30,361	0	11,180	41,541	70,869
16097	0.019	0	0	0	0	0	0	0	0	0	10,520	10,520	10,520
16098	0	0	0	0	0	0	0	0	0	0	0	0	0
16099	0.01533	0	0	0	0	0	0	0	0	0	0	0	0
16102	0.01394	110,824	0	22,165	0	0	132,989	14,602	13,163	0	11,540	24,703	39,304
16103	0.03756	0	0	0	0	0	0	0	0	0	0	0	0
16107	0.00129	0	0	0	0	0	0	0	0	0	0	0	0
16109	0.05056	0	0	0	0	0	0	0	0	0	0	0	0
16111	0.0072	0	0	0	0	0	0	0	0	0	0	0	0
16113	0	0	0	0	0	0	0	0	0	0	0	0	0
16114	0.00864	0	0	0	0	0	0	0	0	0	0	0	0
16115	0.00407	0	0	0	0	0	0	0	0	0	0	0	0
16116	0.0042	0	0	0	0	0	0	0	0	0	0	0	0
16117	0.04	0	0	0	0	0	0	0	0	0	9,908	9,908	9,908

Table 9-11: BPT/BCT/BAT Option I Subtitle D Non-Hazardous Subcategory (continued)

		CAPITAL COSTS ($)						AMORTIZED	O & M COSTS ($/YR)				TOTAL
ID#	Flow (MGD)	Equipment	Sludge Handling	Retrofit	Permit Modification	Land	Total Capital	TOTAL CAPITAL(a) ($/YR)	Equipment	Solids Handling	Monitoring	Total O & M	ANNUAL COST ($/YR)(b)
16118	0.0288	0	0	0	0	0	0	0	0	0	0	0	0
16119	0.00729	13,151	2,004	3,031	0	0	18,186	1,997	2,577	1,948	11,117	15,642	17,639
16120	0.04278	0	0	0	0	0	0	0	0	0	9,200	9,200	9,200
16121	0.08028	0	0	0	0	0	0	0	0	0	0	0	0
16122	0.0255	0	0	0	0	0	0	0	0	0	9,948	9,948	9,948
16123	0.04608	206,903	8,080	42,997	0	0	257,980	28,325	19,430	8,365	11,540	39,335	67,660
16124	0.01666	0	0	0	0	0	0	0	0	0	0	0	0
16125	0.01419	0	0	0	0	0	0	0	0	0	10,712	10,712	10,712
16127	0.00363	48,545	2,004	10,110	0	0	60,659	6,660	9,190	2,756	11,540	23,486	30,146
16128	0.00396	0	0	0	0	0	0	0	0	0	0	0	0
16129	0.00469	0	0	0	0	0	0	0	0	0	11,540	11,540	11,540
16130	0.0003	4,400	2,004	1,281	0	0	7,685	844	10,400	4,078	11,540	26,018	26,862
16131	0.03	0	0	0	0	0	0	0	0	0	0	0	0
16132	0.03	0	0	0	0	0	0	0	0	0	0	0	0
16135	0.01149	0	0	0	0	0	0	0	0	0	0	0	0
16137	0	0	0	0	0	0	0	0	0	0	0	0	0
16139	0.00005	0	0	0	0	0	0	0	0	0	0	0	0
16140	0	0	0	0	0	0	0	0	0	0	0	0	0
16143	0	0	0	0	0	0	0	0	0	0	0	0	0
16144	0	0	0	0	0	0	0	0	0	0	0	0	0
16146	0	0	0	0	0	0	0	0	0	0	0	0	0
16148	0.00008	0	0	0	0	0	0	0	0	0	0	0	0
16149	0	0	0	0	0	0	0	0	0	0	0	0	0
16150	0.04578	0	0	0	0	0	0	0	0	0	0	0	0
16151	0.00205	0	0	0	0	0	0	0	0	0	0	0	0
16152	0	0	0	0	0	0	0	0	0	0	0	0	0
16153	0.008	0	0	0	0	0	0	0	0	0	0	0	0
16154	0.01022	0	0	0	0	0	0	0	0	0	0	0	0
16155	0.00831	0	0	0	0	0	0	0	0	0	0	0	0
16156	0.173	0	0	0	0	0	0	0	0	0	0	0	0
16158	0.01428	0	0	0	0	0	0	0	0	0	0	0	0
16159	0.225	0	0	0	0	0	0	0	0	0	0	0	0
16160	0.00014	0	0	0	0	0	0	0	0	0	0	0	0
16161	0.053	0	0	0	0	0	0	0	0	0	0	0	0
16162	0.0009	0	0	0	0	0	0	0	0	0	0	0	0
16163	0	0	0	0	0	0	0	0	0	0	0	0	0
16164	0.01	0	0	0	0	0	0	0	0	0	0	0	0

Table 9-11: BPT/BCT/BAT Option I Subtitle D Non-Hazardous Subcategory (continued)

		CAPITAL COSTS ($)						AMORTIZED	O & M COSTS ($/YR)				TOTAL
ID#	Flow (MGD)	Equipment	Sludge Handling	Retrofit	Permit Modification	Land	Total Capital	TOTAL CAPITAL(a) ($/YR)	Equipment	Solids Handling	Monitoring	Total O & M	ANNUAL COST ($/YR)(b)
16165	0.03022	0	0	0	0	0	0	0	0	0	0	0	0
16166	0.00342	0	0	0	0	0	0	0	0	0	0	0	0
16169	0	0	0	0	0	0	0	0	0	0	0	0	0
16170	0.0048	55,201	2,004	11,441	0	0	68,647	7,537	9,594	4,078	11,235	24,907	32,444
16171	0.024	0	0	0	0	0	0	0	0	0	0	0	0
16173	0.025	0	0	0	0	0	0	0	0	0	0	0	0
16174	0.0072	0	0	0	0	0	0	0	0	0	0	0	0
16175	0	0	0	0	0	0	0	0	0	0	0	0	0
16176	0.03727	0	0	0	0	0	0	0	0	0	0	0	0
16177	0	0	0	0	0	0	0	0	0	0	0	0	0
16180	0	0	0	0	0	0	0	0	0	0	0	0	0
16184	0	0	0	0	0	0	0	0	0	0	0	0	0
16185	0	0	0	0	0	0	0	0	0	0	0	0	0
16186	0.00304	0	0	0	0	0	0	0	0	0	0	0	0
16187	0.003	0	0	0	0	0	0	0	0	0	0	0	0
16189	0	0	0	0	0	0	0	0	0	0	0	0	0
16190	0	0	0	0	0	0	0	0	0	0	0	0	0
16191	0	0	0	0	0	0	0	0	0	0	0	0	0
16193	0.0023	0	0	0	0	0	0	0	0	0	0	0	0
16196	0.01223	108,110	11,645	0	0	0	119,755	13,148	14,487	10,115	11,540	36,142	49,290
16197	0	0	0	0	0	0	0	0	0	0	0	0	0
16199	0.0008	0	0	0	0	0	0	0	0	0	0	0	0
16200	0.01142	0	0	0	0	0	0	0	0	0	0	0	0
16201	0.00188	0	0	0	0	0	0	0	0	0	0	0	0
16202	0.01301	0	0	0	0	0	0	0	0	0	0	0	0
16203	0.02	0	0	0	0	0	0	0	0	0	0	0	0
16204	0	0	0	0	0	0	0	0	0	0	0	0	0
16204	0	0	0	0	0	0	0	0	0	0	0	0	0
16205	0	0	0	0	0	0	0	0	0	0	0	0	0
16206	0.05739	0	0	0	0	0	0	0	0	0	0	0	0
16208	0.00334	0	0	0	0	0	0	0	0	0	0	0	0
16211	0.15	0	0	0	0	0	0	0	0	0	0	0	0
16212	0.0007	15,300	2,004	0	0	0	17,304	1,900	18,800	1,516	10,500	30,816	32,716
16215	0	0	0	0	0	0	0	0	0	0	0	0	0
16217	0	0	0	0	0	0	0	0	0	0	0	0	0
16219	0.02544	0	0	0	0	0	0	0	0	0	0	0	0
16220	0.03041	0	0	0	0	0	0	0	0	0	0	0	0

Table 9-11: BPT/BCT/BAT Option I Subtitle D Non-Hazardous Subcategory (continued)

		CAPITAL COSTS ($)						AMORTIZED	O & M COSTS ($/YR)				TOTAL
ID#	Flow (MGD)	Equipment	Sludge Handling	Retrofit	Permit Modification	Land	Total Capital	TOTAL CAPITAL(a) ($/YR)	Equipment	Solids Handling	Monitoring	Total O & M	ANNUAL COST ($/YR)(b)
16221	0.00662	0	0	0	0	0	0	0	0	0	0	0	0
16222	0.01548	0	0	0	0	0	0	0	0	0	0	0	0
16222	0	0	0	0	0	0	0	0	0	0	0	0	0
16223	0.02904	153,000	2,004	0	0	0	155,004	17,019	51,200	4,078	10,500	65,778	82,797
16224	0	0	0	0	0	0	0	0	0	0	0	0	0
16225	0.031	0	0	0	0	0	0	0	0	0	0	0	0
16228	0.00072	0	0	0	0	0	0	0	0	0	0	0	0
16230	0	0	0	0	0	0	0	0	0	0	0	0	0
16231	0	0	0	0	0	0	0	0	0	0	0	0	0
16232	0	0	0	0	0	0	0	0	0	0	0	0	0
16233	0.0097	94,269	9,868	0	0	0	104,137	11,434	13,366	9,277	11,540	34,183	45,617
16234	0.03083	0	0	0	0	0	0	0	0	0	0	0	0
16236	0.00595	0	0	0	0	0	0	0	0	0	0	0	0
16239	0	0	0	0	0	0	0	0	0	0	0	0	0
16240	0	0	0	0	0	0	0	0	0	0	0	0	0
16240	0.0056	0	0	0	0	0	0	0	0	0	0	0	0
16241	0	0	0	0	0	0	0	0	0	0	0	0	0
16242	0.0005	0	0	0	0	0	0	0	0	0	0	0	0
16243	0	0	0	0	0	0	0	0	0	0	0	0	0
16245	0	0	0	0	0	0	0	0	0	0	0	0	0
16246	0.00135	0	0	0	0	0	0	0	0	0	0	0	0
16248	0.01	0	0	0	0	0	0	0	0	0	0	0	0
16249	0	0	0	0	0	0	0	0	0	0	0	0	0
16250	0.0002	0	0	0	0	0	0	0	0	0	0	0	0
16251	0.0007	0	0	0	0	0	0	0	0	0	0	0	0
16252	0.005	0	0	0	0	0	0	0	0	0	0	0	0
16253	0.01776	0	0	0	0	0	0	0	0	0	11,068	11,068	11,068
TOTALS	2.694	2,340,439	75,236	321,429	0	0	2,737,104	300,519	373,594	87,480	368,307	829,381	1,139,098

(a) Amortization assuming 7% interest over 15 year period.

(b) Off-site disposal costs used for low flow facilities 16048, 16055, and 16062

Table 9-12: BPT/BCT/BAT Option II Subtitle D Non-Hazardous Subcategory

ID#	Flow (MGD)	CAPITAL COSTS ($)						AMORTIZED TOTAL CAPITAL(a) ($/YR)	O & M COSTS ($/YR)				TOTAL ANNUAL COST ($/YR)(b)
		Equipment	Sludge Handling	Retrofit	Permit Modification	Land	Total Capital		Equipment	Solids Handling	Monitoring	Total O & M	
16001	0.0793	203,456	2,004	41,092	0	0	246,552	27,070	44,857	4,078	11,540	60,475	87,545
16003	0.00472	0	0	0	0	0	0	0	0	0	0	0	0
16008	0	0	0	0	0	0	0	0	0	0	0	0	0
16009	0.01613	0	0	0	0	0	0	0	0	0	0	0	0
16011	0	0	0	0	0	0	0	0	0	0	0	0	0
16012	0.00221	0	0	0	0	0	0	0	0	0	0	0	0
16013	0.015	0	0	0	0	0	0	0	0	0	0	0	0
16014	0	0	0	0	0	0	0	0	0	0	0	0	0
16015	0.0005	0	0	0	0	0	0	0	0	0	0	0	0
16016	0.0023	0	0	0	0	0	0	0	0	0	0	0	0
16020	0.04581	0	0	0	0	0	0	0	0	0	0	0	0
16023	0.05734	0	0	0	0	0	0	0	0	0	0	0	0
16024	0.00592	0	0	0	0	0	0	0	0	0	0	0	0
16025	0	0	0	0	0	0	0	0	0	0	0	0	0
16026	0	0	0	0	0	0	0	0	0	0	0	0	0
16027	0	0	0	0	0	0	0	0	0	0	0	0	0
16028	0.01985	0	0	0	0	0	0	0	0	0	0	0	0
16029	0.025	0	0	0	0	0	0	0	0	0	0	0	0
16033	0.0091	0	0	0	0	0	0	0	0	0	0	0	0
16035	0	0	0	0	0	0	0	0	0	0	0	0	0
16038	0.00822	0	0	0	0	0	0	0	0	0	0	0	0
16039	0.00178	0	0	0	0	0	0	0	0	0	0	0	0
16043	0.00218	0	0	0	0	0	0	0	0	0	0	0	0
16044	0	0	0	0	0	0	0	0	0	0	0	0	0
16046	0	0	0	0	0	0	0	0	0	0	0	0	0
16047	0.00115	51,650	2,004	0	0	0	53,654	5,891	15,497	1,917	11,540	28,954	34,845
16048	5E-06	0	0	0	0	0	0	0	0	0	0	0	730
16049	0.0017	48,843	2,004	10,169	0	0	61,017	6,699	15,205	2,208	11,540	28,953	35,653
16050	0.01	58,533	2,004	0	0	0	60,537	6,647	11,672	1,917	11,540	25,129	31,776
16052	0.0546	217,678	5,563	44,648	0	0	267,889	29,413	17,799	6,897	11,072	35,768	65,180
16053	0.00124	39,625	2,004	0	0	0	41,629	4,571	9,002	1,917	11,540	22,459	27,030
16054	0.00075	30,019	2,004	6,405	0	0	38,427	4,219	12,013	1,917	11,357	25,287	29,506
16055	8E-06	0	0	0	0	0	0	0	0	0	0	0	1,168
16056	0.00137	54,111	2,004	0	0	0	56,115	6,161	15,659	1,917	11,540	29,116	35,277
16058	0.003	44,348	2,004	9,270	0	0	55,622	6,107	8,936	1,917	0	10,853	16,960
16059	0.0011	51,492	2,004	0	0	0	53,496	5,874	15,468	1,917	11,540	28,925	34,798
16060	0.0018	57,885	2,004	0	0	0	59,889	6,575	16,161	2,208	11,540	29,909	36,484
16061	0	0	0	0	0	0	0	0	0	0	0	0	0
16062	0.00005	0	0	0	0	0	0	0	0	0	0	0	7,300
16063	0.0067	94,714	2,004	0	0	0	96,718	10,619	20,855	3,562	11,540	35,957	46,576
16064	0.01197	62,083	2,004	0	0	0	64,087	7,036	12,127	3,931	11,540	27,598	34,634

Table 9-12: BPT/BCT/BAT Option II Subtitle D Non-Hazardous Subcategory (continued)

ID#	Flow (MGD)	CAPITAL COSTS ($)						AMORTIZED TOTAL CAPITAL(a) ($/YR)	O & M COSTS ($/YR)				TOTAL ANNUAL COST ($/YR)(b)
		Equipment	Sludge Handling	Retrofit	Permit Modification	Land	Total Capital		Equipment	Solids Handling	Monitoring	Total O & M	
16065	0.008	91,929	2,004	18,787	0	0	112,719	12,376	20,721	3,231	11,090	35,042	47,418
16068	0	0	0	0	0	0	0	0	0	0	0	0	0
16070	0.00133	0	0	0	0	0	0	0	0	0	0	0	0
16071	0.006	0	0	0	0	0	0	0	0	0	0	0	0
16072	0	0	0	0	0	0	0	0	0	0	0	0	0
16073	0.0182	0	0	0	0	0	0	0	0	0	0	0	0
16074	0	0	0	0	0	0	0	0	0	0	0	0	0
16075	0.01021	0	0	0	0	0	0	0	0	0	0	0	0
16076	0	0	0	0	0	0	0	0	0	0	0	0	0
16077	0.00816	0	0	0	0	0	0	0	0	0	0	0	0
16078	0.00499	0	0	0	0	0	0	0	0	0	0	0	0
16079	0.11247	356,066	0	71,213	0	0	427,279	46,913	27,018	0	11,180	38,198	85,111
16083	0.001	42,475	2,004	8,896	0	0	53,374	5,860	14,573	1,735	11,540	27,848	33,708
16084	0.00643	0	0	0	0	0	0	0	0	0	0	0	0
16085	0.03	0	0	0	0	0	0	0	0	0	0	0	0
16088	0.03621	0	0	0	0	0	0	0	0	0	0	0	0
16090	0.00393	0	0	0	0	0	0	0	0	0	0	0	0
16091	0.2321	0	0	0	0	0	0	0	0	0	0	0	0
16092	0.00668	0	0	0	0	0	0	0	0	0	0	0	0
16093	0.08158	222,598	0	44,520	0	0	267,118	29,328	30,361	0	11,180	41,541	70,869
16097	0.019	72,380	0	14,476	0	0	86,856	9,536	3,597	0	10,520	14,117	23,653
16098	0	0	0	0	0	0	0	0	0	0	0	0	0
16099	0.01533	0	0	0	0	0	0	0	0	0	0	0	0
16102	0.01394	135,429	0	27,086	0	0	162,514	17,843	25,465	0	11,540	37,005	54,848
16103	0.03756	0	0	0	0	0	0	0	0	0	0	0	0
16107	0.00129	0	0	0	0	0	0	0	0	0	0	0	0
16109	0.05056	0	0	0	0	0	0	0	0	0	0	0	0
16111	0.0072	0	0	0	0	0	0	0	0	0	0	0	0
16113	0	0	0	0	0	0	0	0	0	0	0	0	0
16114	0.00864	0	0	0	0	0	0	0	0	0	0	0	0
16115	0.00407	0	0	0	0	0	0	0	0	0	0	0	0
16116	0.0042	0	0	0	0	0	0	0	0	0	0	0	0
16117	0.04	37,048	0	7,410	0	0	44,458	4,881	18,524	0	9,908	28,432	33,313
16118	0.0288	0	0	0	0	0	0	0	0	0	0	0	0
16119	0.00729	13,151	2,004	3,031	0	0	18,186	1,997	2,577	1,948	11,117	15,642	17,639
16120	0.04278	0	0	0	0	0	0	0	0	0	9,200	9,200	9,200
16121	0.08028	0	0	0	0	0	0	0	0	0	0	0	0
16122	0.0255	0	0	0	0	0	0	0	0	0	9,948	9,948	9,948
16123	0.04608	246,283	8,080	50,873	0	0	305,236	33,513	39,120	8,365	11,540	59,025	92,538
16124	0.01666	0	0	0	0	0	0	0	0	0	0	0	0
16125	0.01419	0	0	0	0	0	0	0	0	0	10,712	10,712	10,712

Table 9-12: BPT/BCT/BAT Option II Subtitle D Non-Hazardous Subcategory (continued)

ID#	Flow (MGD)	CAPITAL COSTS ($)						AMORTIZED TOTAL CAPITAL(a) ($/YR)	O&M COSTS ($/YR)				TOTAL ANNUAL COST ($/YR)(b)
		Equipment	Sludge Handling	Retrofit	Permit Modification	Land	Total Capital		Equipment	Solids Handling	Monitoring	Total O&M	
16127	0.00363	55,540	2,004	11,509	0	0	69,053	7,582	11,684	2,756	11,540	25,980	33,562
16128	0.00396	0	0	0	0	0	0	0	0	0	0	0	0
16129	0.00469	0	0	0	0	0	0	0	0	0	11,540	11,540	11,540
16130	0.0003	4,400	2,004	1,281	0	0	7,685	844	10,400	4,078	11,540	26,018	26,862
16131	0.03	0	0	0	0	0	0	0	0	0	0	0	0
16132	0.03	0	0	0	0	0	0	0	0	0	0	0	0
16135	0.01149	0	0	0	0	0	0	0	0	0	0	0	0
16137	0	0	0	0	0	0	0	0	0	0	0	0	0
16139	0.00005	0	0	0	0	0	0	0	0	0	0	0	0
16140	0	0	0	0	0	0	0	0	0	0	0	0	0
16143	0	0	0	0	0	0	0	0	0	0	0	0	0
16144	0	0	0	0	0	0	0	0	0	0	0	0	0
16146	0	0	0	0	0	0	0	0	0	0	0	0	0
16148	0.00008	0	0	0	0	0	0	0	0	0	0	0	0
16149	0	0	0	0	0	0	0	0	0	0	0	0	0
16150	0.04578	0	0	0	0	0	0	0	0	0	0	0	0
16151	0.00205	0	0	0	0	0	0	0	0	0	0	0	0
16152	0	0	0	0	0	0	0	0	0	0	0	0	0
16153	0.008	0	0	0	0	0	0	0	0	0	0	0	0
16154	0.01022	0	0	0	0	0	0	0	0	0	0	0	0
16155	0.00831	0	0	0	0	0	0	0	0	0	0	0	0
16156	0.173	0	0	0	0	0	0	0	0	0	0	0	0
16158	0.01428	0	0	0	0	0	0	0	0	0	0	0	0
16159	0.225	0	0	0	0	0	0	0	0	0	0	0	0
16160	0.00014	0	0	0	0	0	0	0	0	0	0	0	0
16161	0.053	0	0	0	0	0	0	0	0	0	0	0	0
16162	0.0009	0	0	0	0	0	0	0	0	0	0	0	0
16163	0	0	0	0	0	0	0	0	0	0	0	0	0
16164	0.01	0	0	0	0	0	0	0	0	0	0	0	0
16165	0.03022	0	0	0	0	0	0	0	0	0	0	0	0
16166	0.00342	0	0	0	0	0	0	0	0	0	0	0	0
16169	0	0	0	0	0	0	0	0	0	0	0	0	0
16170	0.0048	55,201	2,004	11,441	0	0	68,647	7,537	9,594	4,078	11,235	24,907	32,444
16171	0.024	0	0	0	0	0	0	0	0	0	0	0	0
16173	0.025	0	0	0	0	0	0	0	0	0	0	0	0
16174	0.0072	0	0	0	0	0	0	0	0	0	0	0	0
16175	0	0	0	0	0	0	0	0	0	0	0	0	0
16176	0.03727	0	0	0	0	0	0	0	0	0	0	0	0
16177	0	0	0	0	0	0	0	0	0	0	0	0	0
16180	0	0	0	0	0	0	0	0	0	0	0	0	0
16184	0	0	0	0	0	0	0	0	0	0	0	0	0

Table 9-12: BCT/BPT/BAT Option II Subtitle D Non-Hazardous Subcategory (continued)

ID#	Flow (MGD)	CAPITAL COSTS ($)						AMORTIZED TOTAL CAPITAL(a) ($/YR)	O&M COSTS ($/YR)				TOTAL ANNUAL COST ($/YR)(b)
		Equipment	Sludge Handling	Retrofit	Permit Modification	Land	Total Capital		Equipment	Solids Handling	Monitoring	Total O&M	
16185	0	0	0	0	0	0	0	0	0	0	0	0	0
16186	0.00304	0	0	0	0	0	0	0	0	0	0	0	0
16187	0.003	0	0	0	0	0	0	0	0	0	0	0	0
16189	0	0	0	0	0	0	0	0	0	0	0	0	0
16190	0	0	0	0	0	0	0	0	0	0	0	0	0
16191	0	0	0	0	0	0	0	0	0	0	0	0	0
16193	0.0023	0	0	0	0	0	0	0	0	0	0	0	0
16196	0.01223	131,628	11,645	0	0	0	143,273	15,731	26,246	10,115	11,540	47,901	63,632
16197	0	0	0	0	0	0	0	0	0	0	0	0	0
16199	0.0008	0	0	0	0	0	0	0	0	0	0	0	0
16200	0.01142	0	0	0	0	0	0	0	0	0	0	0	0
16201	0.00188	0	0	0	0	0	0	0	0	0	0	0	0
16202	0.01301	0	0	0	0	0	0	0	0	0	0	0	0
16203	0.02	0	0	0	0	0	0	0	0	0	0	0	0
16204	0	0	0	0	0	0	0	0	0	0	0	0	0
16204	0	0	0	0	0	0	0	0	0	0	0	0	0
16205	0	0	0	0	0	0	0	0	0	0	0	0	0
16206	0.05739	0	0	0	0	0	0	0	0	0	0	0	0
16208	0.00334	0	0	0	0	0	0	0	0	0	0	0	0
16211	0.15	0	0	0	0	0	0	0	0	0	0	0	0
16212	0.0007	15,300	2,004	0	0	0	17,304	1,900	18,800	1,516	10,500	30,816	32,716
16215	0	0	0	0	0	0	0	0	0	0	0	0	0
16217	0	0	0	0	0	0	0	0	0	0	0	0	0
16219	0.02544	0	0	0	0	0	0	0	0	0	0	0	0
16220	0.03041	0	0	0	0	0	0	0	0	0	0	0	0
16221	0.00662	0	0	0	0	0	0	0	0	0	0	0	0
16222	0.01548	0	0	0	0	0	0	0	0	0	0	0	0
16222	0	0	0	0	0	0	0	0	0	0	0	0	0
16223	0.02904	153,000	2,004	0	0	0	155,004	17,019	51,200	4,078	10,500	65,778	82,797
16224	0	0	0	0	0	0	0	0	0	0	0	0	0
16225	0.031	0	0	0	0	0	0	0	0	0	0	0	0
16228	0.00072	0	0	0	0	0	0	0	0	0	0	0	0
16230	0	0	0	0	0	0	0	0	0	0	0	0	0
16231	0	0	0	0	0	0	0	0	0	0	0	0	0
16232	0	0	0	0	0	0	0	0	0	0	0	0	0
16233	0.0097	116,040	9,868	0	0	0	125,908	13,824	24,252	9,277	11,540	45,069	58,893
16234	0.03083	0	0	0	0	0	0	0	0	0	0	0	0
16236	0.00595	0	0	0	0	0	0	0	0	0	0	0	0
16239	0	0	0	0	0	0	0	0	0	0	0	0	0
16240	0	0	0	0	0	0	0	0	0	0	0	0	0
16240	0.0056	0	0	0	0	0	0	0	0	0	0	0	0

Table 9-12: BCT/BPT/BAT Option II Subtitle D Non-Hazardous Subcategory (continued)

ID#	Flow (MGD)	CAPITAL COSTS ($) Equipment	Sludge Handling	Retrofit	Permit Modification	Land	Total Capital	AMORTIZED TOTAL CAPITAL(a) ($/YR)	O&M COSTS ($/YR) Equipment	Solids Handling	Monitoring	Total O&M	TOTAL ANNUAL COST ($/YR)(b)
16241	0	0	0	0	0	0	0	0	0	0	0	0	0
16242	0.0005	0	0	0	0	0	0	0	0	0	0	0	0
16243	0	0	0	0	0	0	0	0	0	0	0	0	0
16245	0	0	0	0	0	0	0	0	0	0	0	0	0
16246	0.00135	0	0	0	0	0	0	0	0	0	0	0	0
16248	0.01	0	0	0	0	0	0	0	0	0	0	0	0
16249	0	0	0	0	0	0	0	0	0	0	0	0	0
16250	0.0002	0	0	0	0	0	0	0	0	0	0	0	0
16251	0.0007	0	0	0	0	0	0	0	0	0	0	0	0
16252	0.005	0	0	0	0	0	0	0	0	0	0	0	0
16253	0.01776	26,840	0	5,368	0	0	32,208	3,536	13,420	0	11,068	24,488	28,024
TOTALS	2.694	2,789,743	75,236	387,473	0	0	3,252,453	357,102	562,803	87,480	368,307	1,018,590	1,384,890

(a) Amortization assuming 7% interest over 15 year period.

(b) Off-site disposal costs used for low flow facilities 16048, 16055, and 16062

Table 9-13: BAT Option III Subtitle D Non-Hazardous Subcategory

ID#	Flow (MGD)	CAPITAL COSTS ($)						AMORTIZED TOTAL CAPITAL(a) ($/YR)	O & M COSTS ($/YR)				TOTAL ANNUAL COST ($/YR)(b)
		Equipment	Sludge Handling	Retrofit	Permit Modification	Land	Total Capital		Equipment	Solids Handling	Monitoring	Total O & M	
16001	0.0793	2,183,593	2,004	437,119	0	0	2,622,716	287,960	623,747	4,078	11,540	639,365	927,325
16003	0.00472	0	0	0	0	0	0	0	0	0	0	0	0
16008	0	0	0	0	0	0	0	0	0	0	0	0	0
16009	0.01613	0	0	0	0	0	0	0	0	0	0	0	0
16011	0	0	0	0	0	0	0	0	0	0	0	0	0
16012	0.00221	0	0	0	0	0	0	0	0	0	0	0	0
16013	0.015	0	0	0	0	0	0	0	0	0	0	0	0
16014	0	0	0	0	0	0	0	0	0	0	0	0	0
16015	0.0005	0	0	0	0	0	0	0	0	0	0	0	0
16016	0.0023	0	0	0	0	0	0	0	0	0	0	0	0
16020	0.04581	0	0	0	0	0	0	0	0	0	0	0	0
16023	0.05734	0	0	0	0	0	0	0	0	0	0	0	0
16024	0.00592	0	0	0	0	0	0	0	0	0	0	0	0
16025	0	0	0	0	0	0	0	0	0	0	0	0	0
16026	0	0	0	0	0	0	0	0	0	0	0	0	0
16027	0	0	0	0	0	0	0	0	0	0	0	0	0
16028	0.01985	0	0	0	0	0	0	0	0	0	0	0	0
16029	0.025	0	0	0	0	0	0	0	0	0	0	0	0
16033	0.0091	0	0	0	0	0	0	0	0	0	0	0	0
16035	0	0	0	0	0	0	0	0	0	0	0	0	0
16038	0.00822	0	0	0	0	0	0	0	0	0	0	0	0
16039	0.00178	0	0	0	0	0	0	0	0	0	0	0	0
16043	0.00218	0	0	0	0	0	0	0	0	0	0	0	0
16044	0	0	0	0	0	0	0	0	0	0	0	0	0
16046	0	0	0	0	0	0	0	0	0	0	0	0	0
16047	0.00115	191,967	2,004	0	0	0	193,971	21,297	23,878	1,917	11,540	37,335	58,632
16048	5E-06	46,193	0	0	0	0	46,193	5,072	14,452	0	0	14,452	20,254
16049	0.0017	247,768	2,004	49,954	0	0	299,726	32,908	27,615	2,208	11,540	41,363	74,272
16050	0.01	797,074	2,004	0	0	0	799,078	87,734	84,672	1,917	11,540	98,129	185,864
16052	0.0546	1,949,079	5,563	390,928	0	0	2,345,571	257,531	416,379	6,897	11,072	434,348	691,879
16053	0.00124	190,146	2,004	0	0	0	192,150	21,097	18,054	1,917	11,540	31,511	52,609
16054	0.00075	123,852	2,004	25,171	0	0	151,028	16,582	17,488	1,917	11,357	30,762	47,344
16055	8E-06	32,864	0	0	0	0	32,864	3,608	7,737	0	0	7,737	12,513
16056	0.00137	218,417	2,004	0	0	0	220,421	24,201	25,638	1,917	11,540	39,095	63,296
16058	0.003	361,815	2,004	72,764	0	0	436,583	47,934	30,836	1,917	0	32,753	80,688
16059	0.0011	186,408	2,004	0	0	0	188,412	20,687	23,498	1,917	11,540	36,955	57,641
16060	0.0018	266,809	2,004	0	0	0	268,813	29,514	29,301	2,208	11,540	43,049	72,563
16061	0	0	0	0	0	0	0	0	0	0	0	0	0
16062	0.00005	36,642	0	0	0	0	36,642	4,023	8,043	0	0	8,043	19,366
16063	0.0067	664,889	2,004	0	0	0	666,893	73,221	69,765	3,562	11,540	84,867	158,088

Table 9-13: BAT Option III Subtitle D Non-Hazardous Subcategory (continued)

ID#	Flow (MGD)	CAPITAL COSTS ($)						AMORTIZED TOTAL CAPITAL(a) ($/YR)	O&M COSTS ($/YR)				TOTAL ANNUAL COST ($/YR)(b)
		Equipment	Sludge Handling	Retrofit	Permit Modification	Land	Total Capital		Equipment	Solids Handling	Monitoring	Total O&M	
16064	0.01197	885,558	2,004	0	0	0	887,562	97,450	99,486	3,931	11,540	114,957	212,406
16065	0.008	733,057	2,004	147,012	0	0	882,073	96,847	79,121	3,231	11,090	93,442	190,289
16068	0	0	0	0	0	0	0	0	0	0	0	0	0
16070	0.00133	0	0	0	0	0	0	0	0	0	0	0	0
16071	0.006	0	0	0	0	0	0	0	0	0	0	0	0
16072	0	0	0	0	0	0	0	0	0	0	0	0	0
16073	0.0182	0	0	0	0	0	0	0	0	0	0	0	0
16074	0	0	0	0	0	0	0	0	0	0	0	0	0
16075	0.01021	0	0	0	0	0	0	0	0	0	0	0	0
16076	0	0	0	0	0	0	0	0	0	0	0	0	0
16077	0.00816	0	0	0	0	0	0	0	0	0	0	0	0
16078	0.00499	0	0	0	0	0	0	0	0	0	0	0	0
16079	0.11247	2,562,809	0	512,562	0	0	3,075,371	337,659	848,079	0	11,180	859,259	1,196,918
16083	0.001	165,966	2,004	33,594	0	0	201,564	22,131	21,873	1,735	11,540	35,148	57,279
16084	0.00643	0	0	0	0	0	0	0	0	0	0	0	0
16085	0.03	0	0	0	0	0	0	0	0	0	0	0	0
16088	0.03621	0	0	0	0	0	0	0	0	0	0	0	0
16090	0.00393	0	0	0	0	0	0	0	0	0	0	0	0
16091	0.2321	0	0	0	0	0	0	0	0	0	0	0	0
16092	0.00668	0	0	0	0	0	0	0	0	0	0	0	0
16093	0.08158	2,221,423	0	444,285	0	0	2,665,708	292,680	625,858	0	11,180	637,038	929,719
16097	0.019	1,067,839	0	213,568	0	0	1,281,407	140,692	138,700	0	10,520	149,220	289,912
16098	0	0	0	0	0	0	0	0	0	0	0	0	0
16099	0.01533	0	0	0	0	0	0	0	0	0	0	0	0
16102	0.01394	1,035,581	0	207,116	0	0	1,242,698	136,442	127,227	0	11,540	138,767	275,208
16103	0.03756	0	0	0	0	0	0	0	0	0	0	0	0
16107	0.00129	0	0	0	0	0	0	0	0	0	0	0	0
16109	0.05056	0	0	0	0	0	0	0	0	0	0	0	0
16111	0.0072	0	0	0	0	0	0	0	0	0	0	0	0
16113	0	0	0	0	0	0	0	0	0	0	0	0	0
16114	0.00864	0	0	0	0	0	0	0	0	0	0	0	0
16115	0.00407	0	0	0	0	0	0	0	0	0	0	0	0
16116	0.0042	0	0	0	0	0	0	0	0	0	0	0	0
16117	0.04	1,562,645	0	312,529	0	0	1,875,174	205,884	310,524	0	9,908	320,432	526,316
16118	0.0288	0	0	0	0	0	0	0	0	0	0	0	0
16119	0.00729	603,122	0	120,624	0	0	723,746	79,463	53,202	0	11,117	64,319	143,783
16120	0.04278	1,569,551	0	313,910	0	0	1,883,461	206,794	312,258	0	9,200	321,458	528,251
16121	0.08028	0	0	0	0	0	0	0	0	0	0	0	0
16122	0.0255	1,240,783	0	248,157	0	0	1,488,939	163,478	186,150	0	9,948	196,098	359,576
16123	0.04608	1,864,917	8,080	374,599	0	0	2,247,596	246,774	375,504	8,365	11,540	395,409	642,183

Table 9-13: BAT Option III Subtitle D Non-Hazardous Subcategory (continued)

		CAPITAL COSTS ($)						AMORTIZED	O & M COSTS ($/YR)				TOTAL
ID#	Flow (MGD)	Equipment	Sludge Handling	Retrofit	Permit Modification	Land	Total Capital	TOTAL CAPITAL(a) ($/YR)	Equipment	Solids Handling	Monitoring	Total O & M	ANNUAL COST ($/YR)(b)
16124	0.01666	0	0	0	0	0	0	0	0	0	0	0	0
16125	0.01419	909,456	0	181,891	0	0	1,091,347	119,824	103,609	0	10,712	114,321	234,145
16127	0.00363	423,029	2,004	85,007	0	0	510,040	56,000	38,161	2,756	11,540	52,457	108,457
16128	0.00396	0	0	0	0	0	0	0	0	0	0	0	0
16129	0.00469	444,502	0	88,900	0	0	533,403	58,565	34,237	0	11,540	45,777	104,342
16130	0.0003	36,269	0	7,254	0	0	43,523	4,779	2,190	0	11,540	13,730	18,509
16131	0.03	0	0	0	0	0	0	0	0	0	0	0	0
16132	0.03	0	0	0	0	0	0	0	0	0	0	0	0
16135	0.01149	0	0	0	0	0	0	0	0	0	0	0	0
16137	0	0	0	0	0	0	0	0	0	0	0	0	0
16139	0.00005	0	0	0	0	0	0	0	0	0	0	0	0
16140	0	0	0	0	0	0	0	0	0	0	0	0	0
16143	0	0	0	0	0	0	0	0	0	0	0	0	0
16144	0	0	0	0	0	0	0	0	0	0	0	0	0
16146	0	0	0	0	0	0	0	0	0	0	0	0	0
16148	0.00008	0	0	0	0	0	0	0	0	0	0	0	0
16149	0	0	0	0	0	0	0	0	0	0	0	0	0
16150	0.04578	0	0	0	0	0	0	0	0	0	0	0	0
16151	0.00205	0	0	0	0	0	0	0	0	0	0	0	0
16152	0	0	0	0	0	0	0	0	0	0	0	0	0
16153	0.008	0	0	0	0	0	0	0	0	0	0	0	0
16154	0.01022	0	0	0	0	0	0	0	0	0	0	0	0
16155	0.00831	0	0	0	0	0	0	0	0	0	0	0	0
16156	0.173	0	0	0	0	0	0	0	0	0	0	0	0
16158	0.01428	0	0	0	0	0	0	0	0	0	0	0	0
16159	0.225	0	0	0	0	0	0	0	0	0	0	0	0
16160	0.00014	0	0	0	0	0	0	0	0	0	0	0	0
16161	0.053	0	0	0	0	0	0	0	0	0	0	0	0
16162	0.0009	0	0	0	0	0	0	0	0	0	0	0	0
16163	0	0	0	0	0	0	0	0	0	0	0	0	0
16164	0.01	0	0	0	0	0	0	0	0	0	0	0	0
16165	0.03022	0	0	0	0	0	0	0	0	0	0	0	0
16166	0.00342	0	0	0	0	0	0	0	0	0	0	0	0
16169	0	0	0	0	0	0	0	0	0	0	0	0	0
16170	0.0048	507,196	2,004	101,840	0	0	611,040	67,089	44,634	4,078	11,235	59,947	127,036
16171	0.024	0	0	0	0	0	0	0	0	0	0	0	0
16173	0.025	0	0	0	0	0	0	0	0	0	0	0	0
16174	0.0072	0	0	0	0	0	0	0	0	0	0	0	0
16175	0	0	0	0	0	0	0	0	0	0	0	0	0
16176	0.03727	0	0	0	0	0	0	0	0	0	0	0	0

Table 9-13: BAT Option III Subtitle D Non-Hazardous Subcategory (continued)

		CAPITAL COSTS ($)						AMORTIZED	O & M COSTS ($/YR)				TOTAL
ID#	Flow (MGD)	Equipment	Sludge Handling	Retrofit	Permit Modification	Land	Total Capital	TOTAL CAPITAL(a) ($/YR)	Equipment	Solids Handling	Monitoring	Total O & M	ANNUAL COST ($/YR)(b)
16177	0	0	0	0	0	0	0	0	0	0	0	0	0
16180	0	0	0	0	0	0	0	0	0	0	0	0	0
16184	0	0	0	0	0	0	0	0	0	0	0	0	0
16185	0	0	0	0	0	0	0	0	0	0	0	0	0
16186	0.00304	0	0	0	0	0	0	0	0	0	0	0	0
16187	0.003	0	0	0	0	0	0	0	0	0	0	0	0
16189	0	0	0	0	0	0	0	0	0	0	0	0	0
16190	0	0	0	0	0	0	0	0	0	0	0	0	0
16191	0	0	0	0	0	0	0	0	0	0	0	0	0
16193	0.0023	0	0	0	0	0	0	0	0	0	0	0	0
16196	0.01223	965,897	11,645	0	0	0	977,542	107,329	115,547	10,115	11,540	137,202	244,531
16197	0	0	0	0	0	0	0	0	0	0	0	0	0
16199	0.0008	0	0	0	0	0	0	0	0	0	0	0	0
16200	0.01142	0	0	0	0	0	0	0	0	0	0	0	0
16201	0.00188	0	0	0	0	0	0	0	0	0	0	0	0
16202	0.01301	0	0	0	0	0	0	0	0	0	0	0	0
16203	0.02	0	0	0	0	0	0	0	0	0	0	0	0
16204	0	0	0	0	0	0	0	0	0	0	0	0	0
16204	0	0	0	0	0	0	0	0	0	0	0	0	0
16205	0	0	0	0	0	0	0	0	0	0	0	0	0
16206	0.05739	0	0	0	0	0	0	0	0	0	0	0	0
16208	0.00334	0	0	0	0	0	0	0	0	0	0	0	0
16211	0.15	0	0	0	0	0	0	0	0	0	0	0	0
16212	0.0007	134,753	2,004	0	0	0	136,757	15,015	20,233	1,516	10,500	32,249	47,264
16215	0	0	0	0	0	0	0	0	0	0	0	0	0
16217	0	0	0	0	0	0	0	0	0	0	0	0	0
16219	0.02544	0	0	0	0	0	0	0	0	0	0	0	0
16220	0.03041	0	0	0	0	0	0	0	0	0	0	0	0
16221	0.00662	0	0	0	0	0	0	0	0	0	0	0	0
16222	0.01548	0	0	0	0	0	0	0	0	0	0	0	0
16222	0	0	0	0	0	0	0	0	0	0	0	0	0
16223	0.02904	1,531,517	7,768	0	0	0	1,539,285	169,005	246,811	8,212	10,500	265,523	434,528
16224	0	0	0	0	0	0	0	0	0	0	0	0	0
16225	0.031	0	0	0	0	0	0	0	0	0	0	0	0
16228	0.00072	0	0	0	0	0	0	0	0	0	0	0	0
16230	0	0	0	0	0	0	0	0	0	0	0	0	0
16231	0	0	0	0	0	0	0	0	0	0	0	0	0
16232	0	0	0	0	0	0	0	0	0	0	0	0	0
16233	0.0097	840,751	9,868	0	0	0	850,619	93,393	95,062	9,277	11,540	115,879	209,272
16234	0.03083	0	0	0	0	0	0	0	0	0	0	0	0

Table 9-13: BAT Option III Subtitle D Non-Hazardous Subcategory (continued)

ID#	Flow (MGD)	CAPITAL COSTS ($) Equipment	Sludge Handling	Retrofit	Permit Modification	Land	Total Capital	AMORTIZED TOTAL CAPITAL(a) ($/YR)	O&M COSTS ($/YR) Equipment	Solids Handling	Monitoring	Total O&M	TOTAL ANNUAL COST ($/YR)(b)
16236	0.00595	0	0	0	0	0	0	0	0	0	0	0	0
16239	0	0	0	0	0	0	0	0	0	0	0	0	0
16240	0	0	0	0	0	0	0	0	0	0	0	0	0
16240	0.0056	0	0	0	0	0	0	0	0	0	0	0	0
16241	0	0	0	0	0	0	0	0	0	0	0	0	0
16242	0.0005	0	0	0	0	0	0	0	0	0	0	0	0
16243	0	0	0	0	0	0	0	0	0	0	0	0	0
16245	0	0	0	0	0	0	0	0	0	0	0	0	0
16246	0.00135	0	0	0	0	0	0	0	0	0	0	0	0
16248	0.01	0	0	0	0	0	0	0	0	0	0	0	0
16249	0	0	0	0	0	0	0	0	0	0	0	0	0
16250	0.0002	0	0	0	0	0	0	0	0	0	0	0	0
16251	0.0007	0	0	0	0	0	0	0	0	0	0	0	0
16252	0.005	0	0	0	0	0	0	0	0	0	0	0	0
16253	0.01776	1,056,810	0	211,362	0	0	1,268,173	139,239	143,068	0	11,068	154,136	293,374
TOTALS	2.694	29,860,948	76,992	4,580,148	0	0	34,518,089	3,789,901	5,442,636	85,588	368,307	5,896,531	9,695,630

(a) Amortization assuming 7% interest over 15 year period.

(b) Off-site disposal costs used for low flow facilities 16048, 16055, and 16062

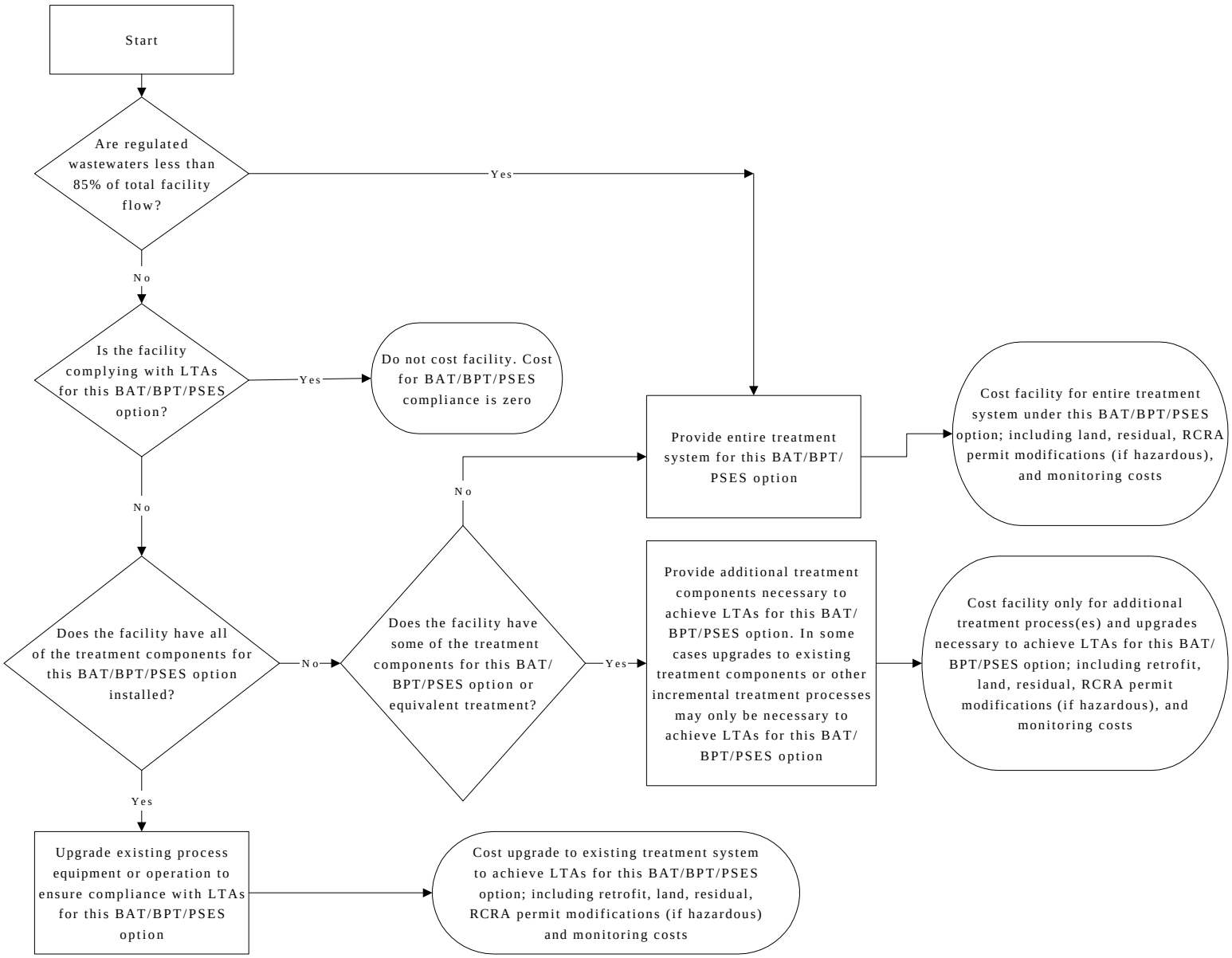


Figure 9-1: Option-Specific Costing Logic Flow Diagram

Figure 9-2
Equalization Capital Cost Curve

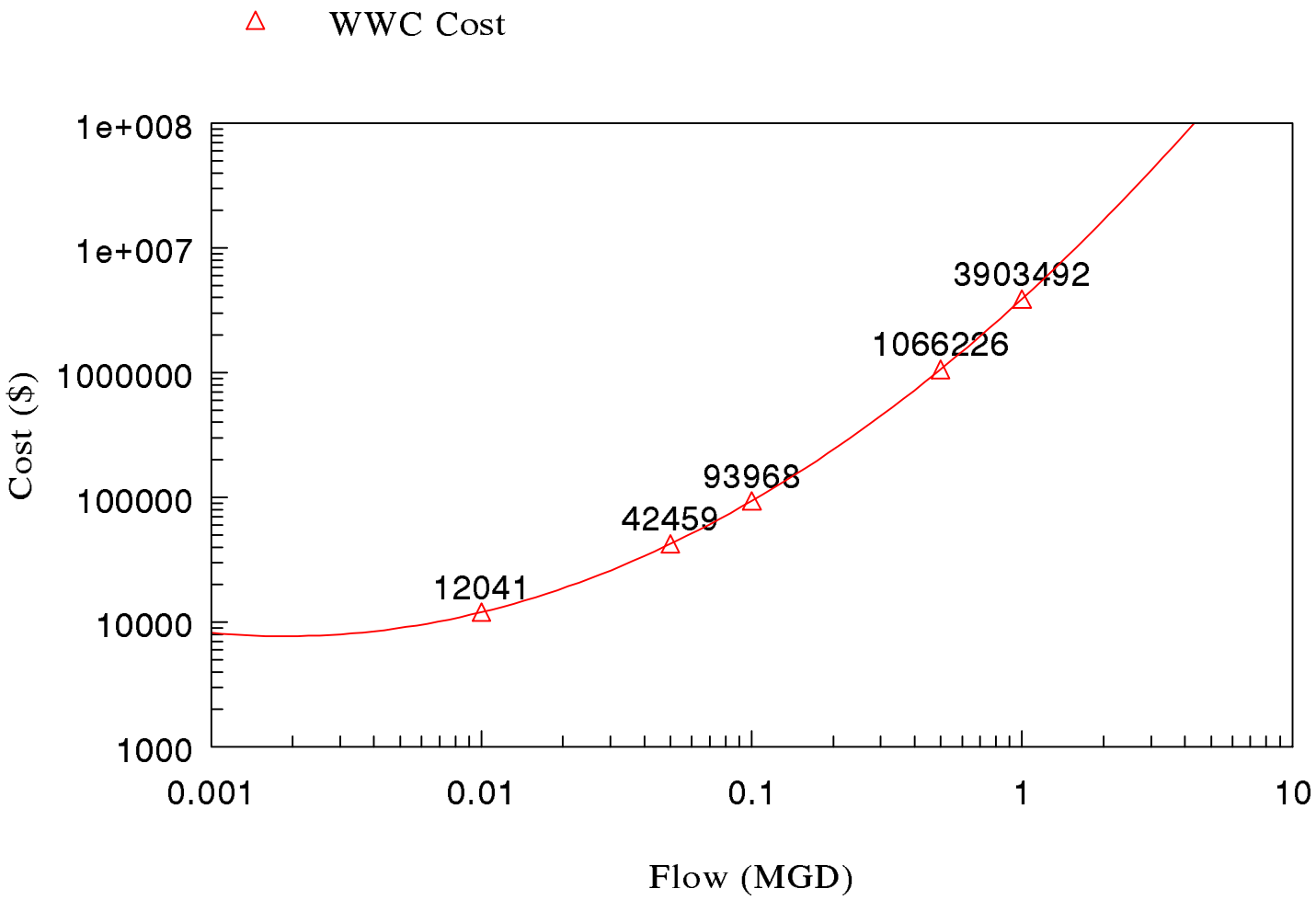


Figure 9-3
Flocculation Capital Cost Curve

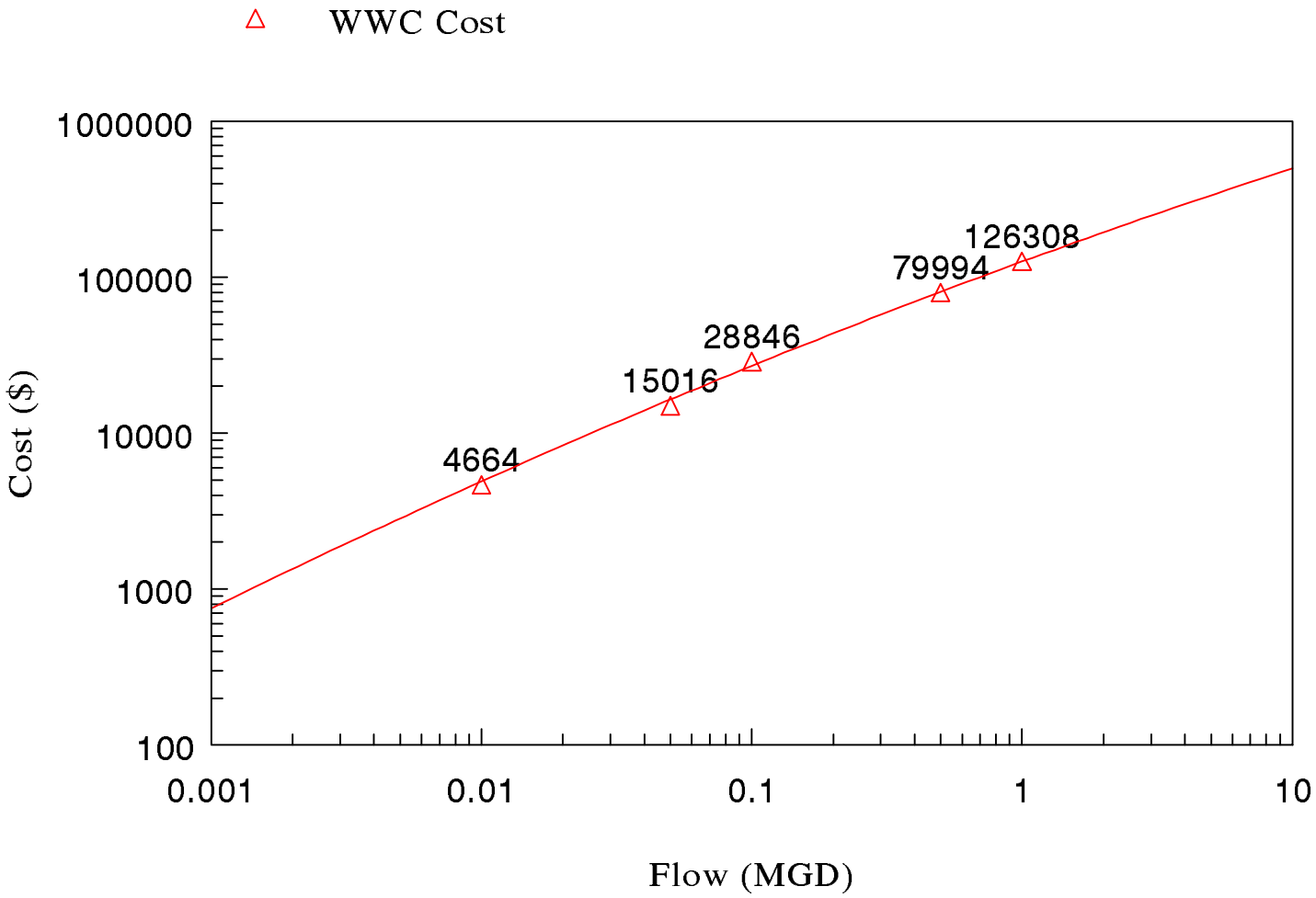


Figure 9-4
Flocculation O&M Cost Curve

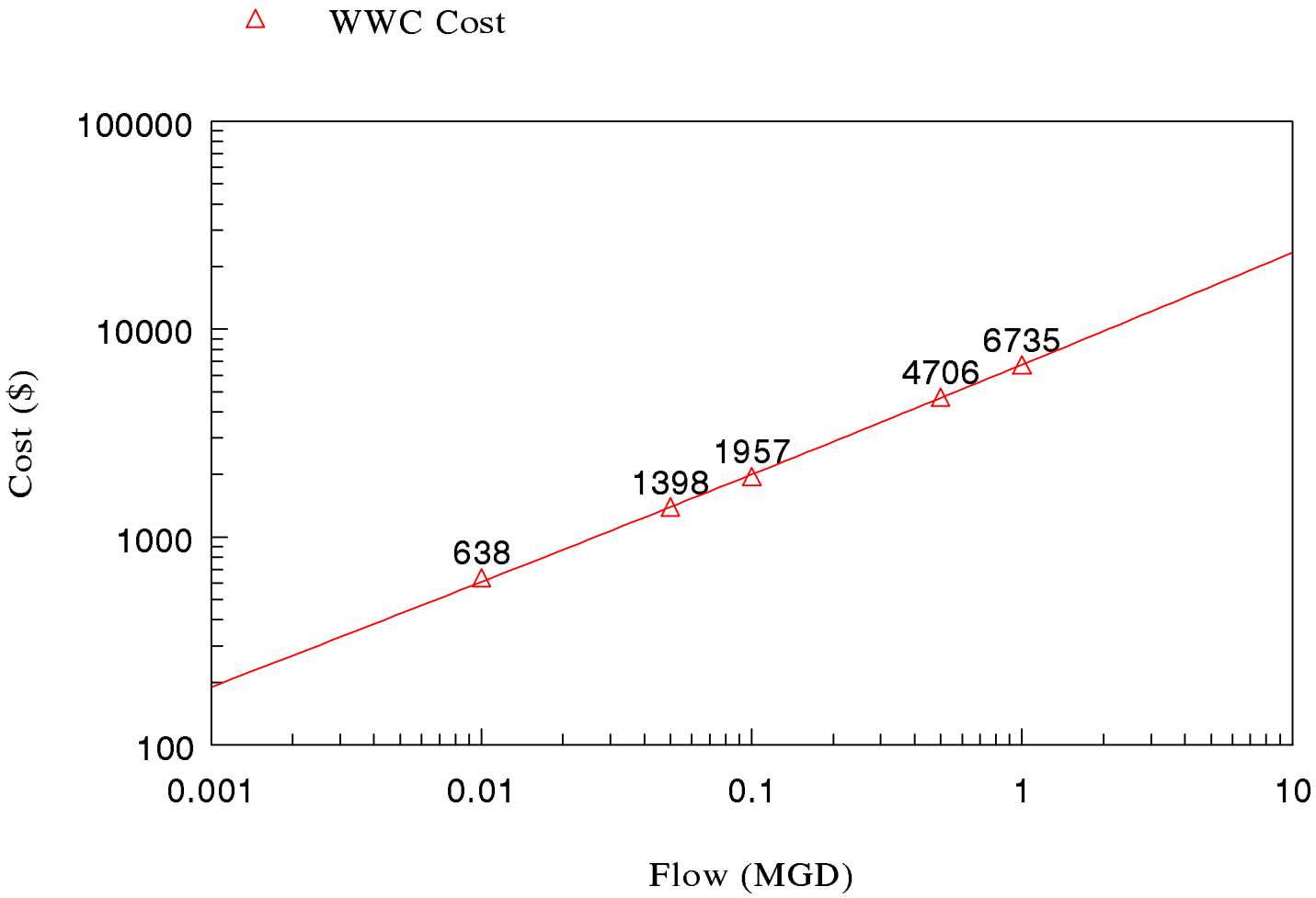


Figure 9-5
Sodium Hydroxide Capital Cost Curve

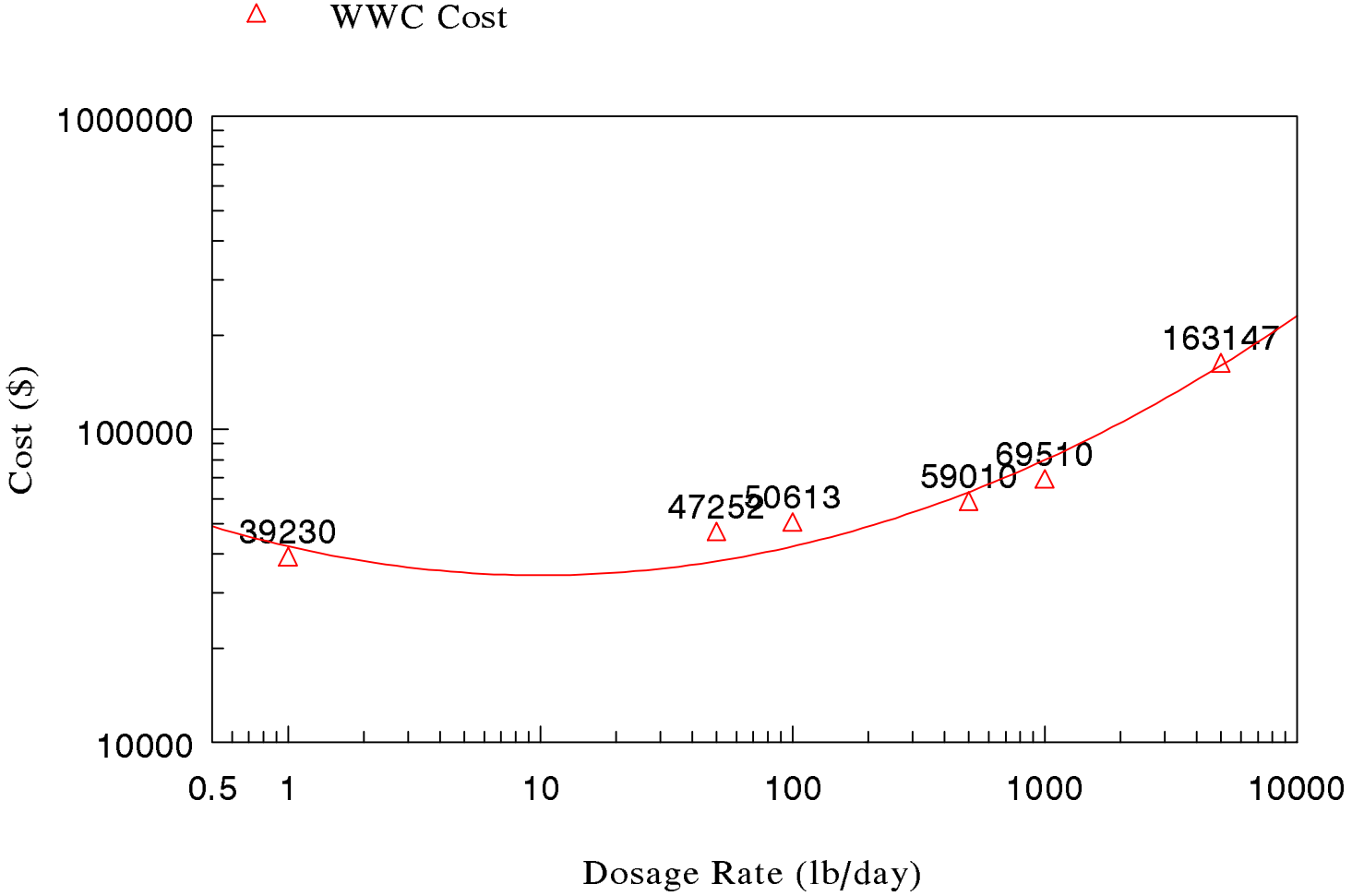


Figure 9-6
Sodium Hydroxide O&M Cost Curve

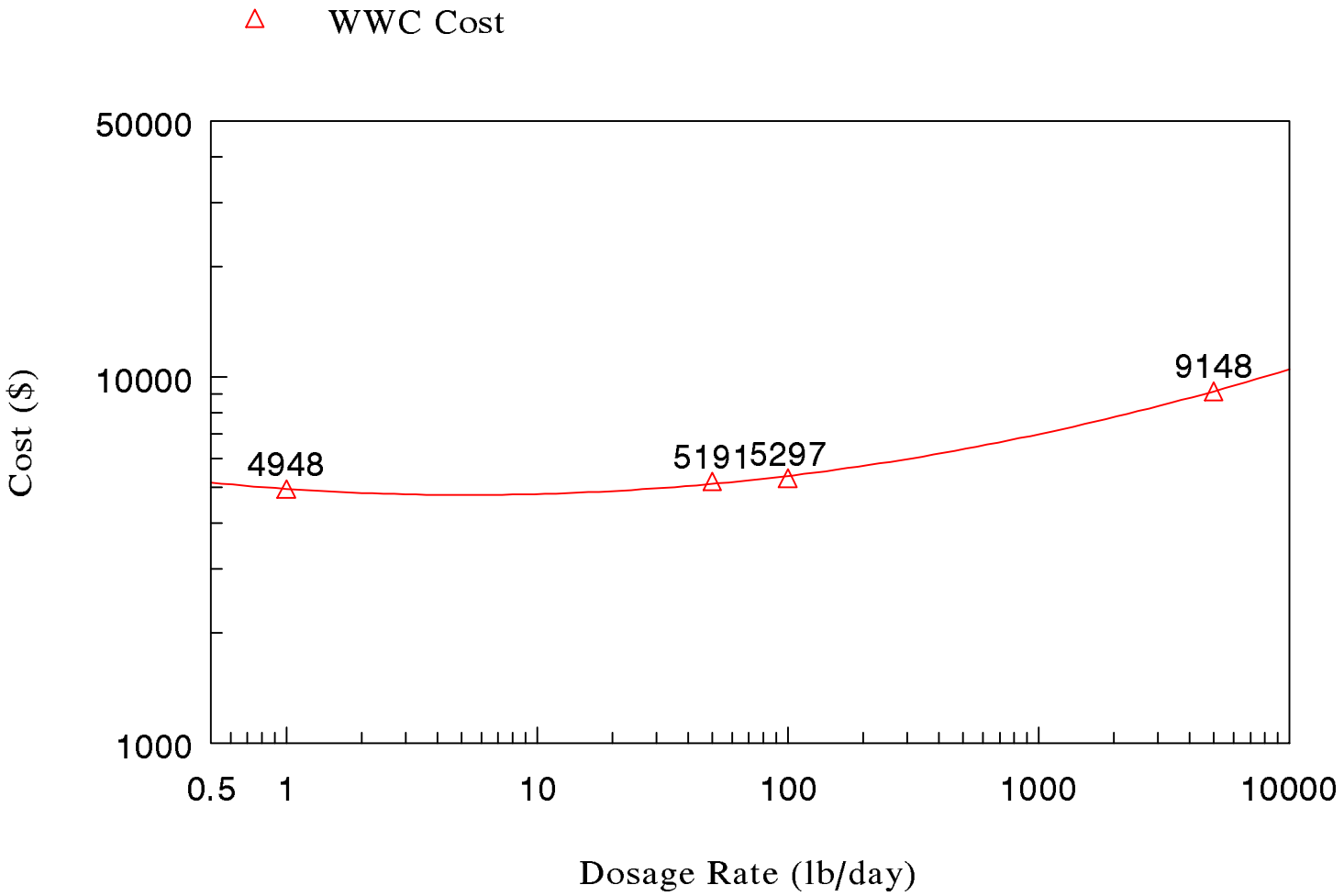


Figure 9-7
Phosphoric Acid Feed Capital Cost Curve

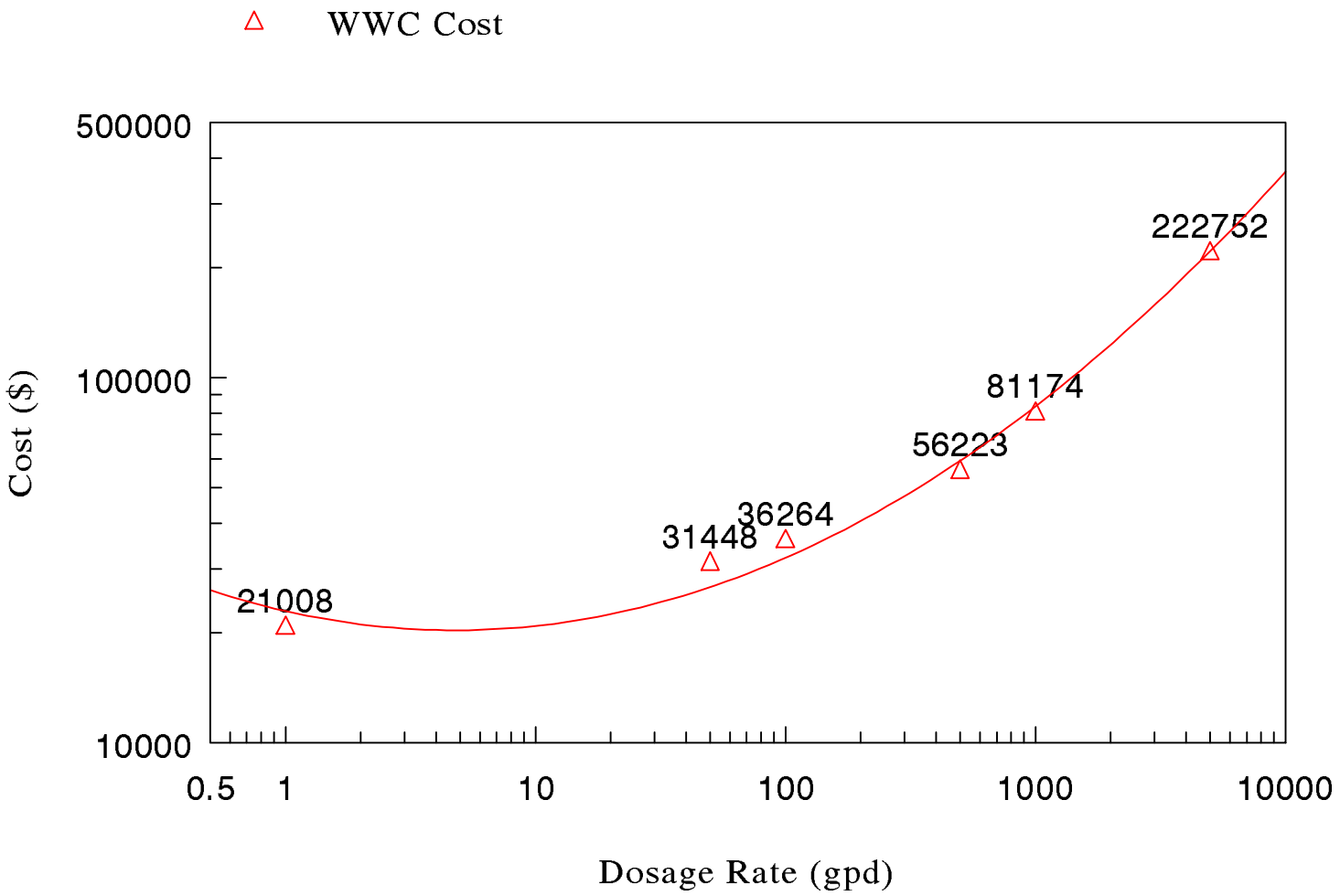


Figure 9-8
Phosphoric Acid Feed O&M Cost Curve

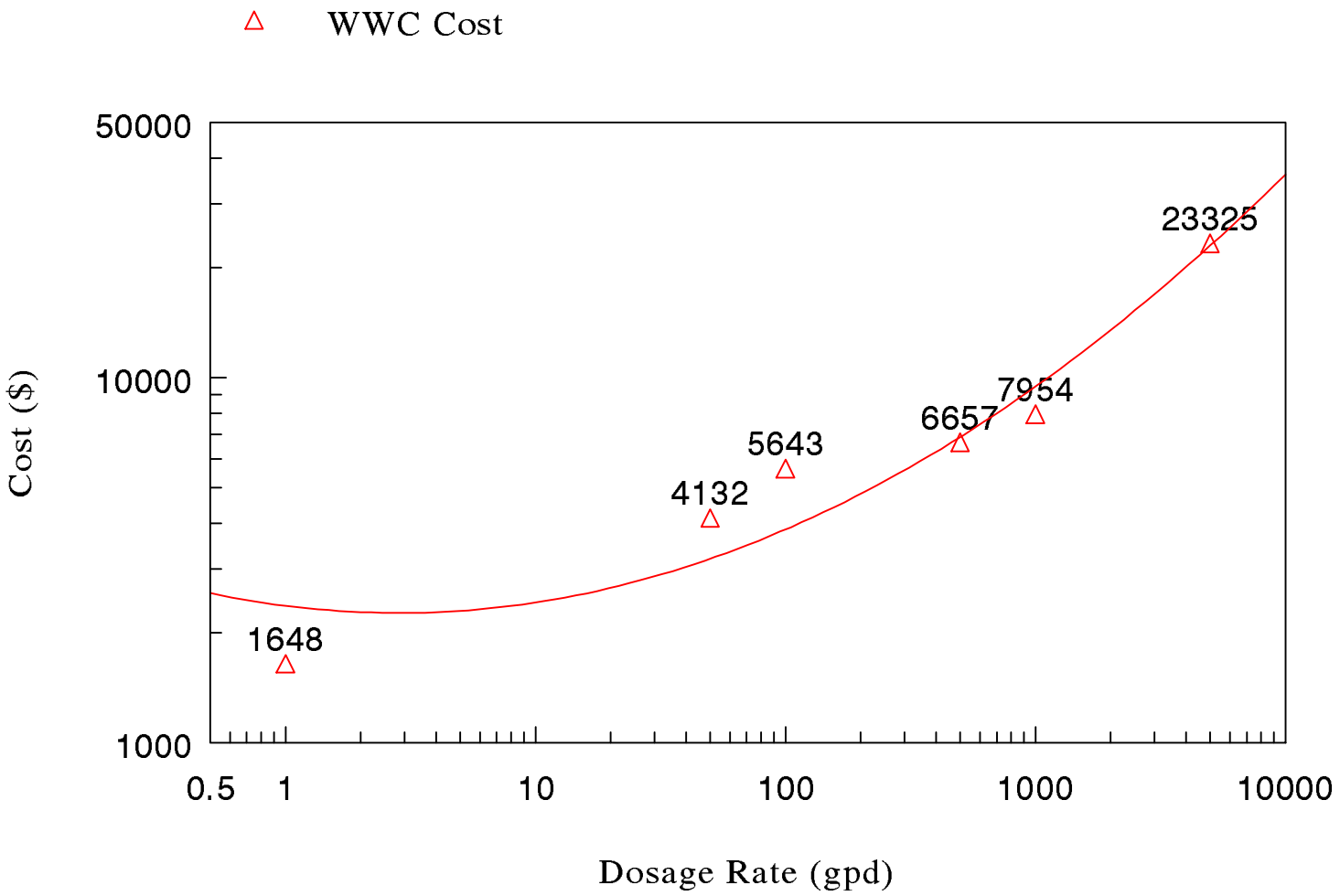


Figure 9-9
Polymer Feed Capital Cost Curve

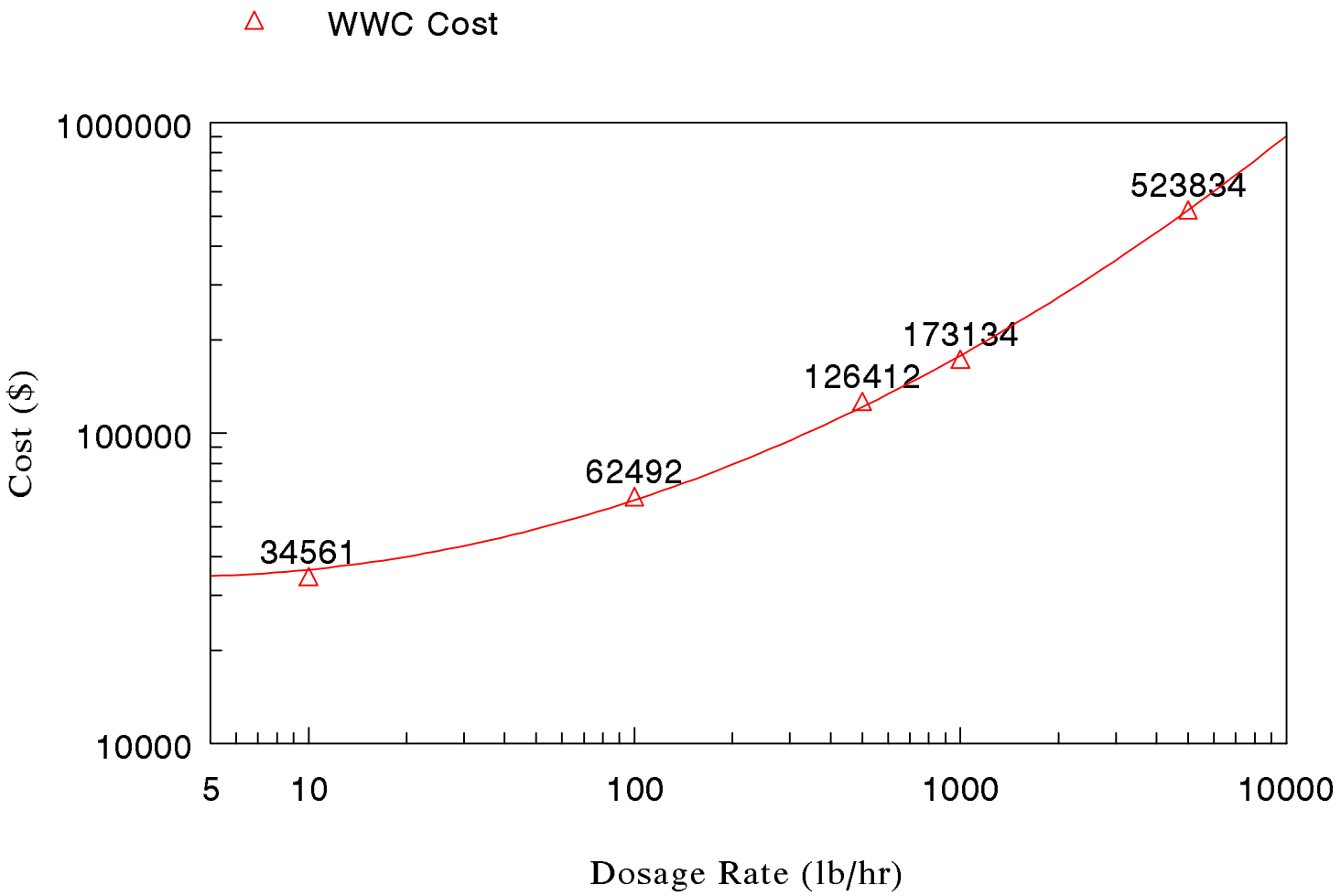


Figure 9-10
Polymer Feed O&M Cost Curve

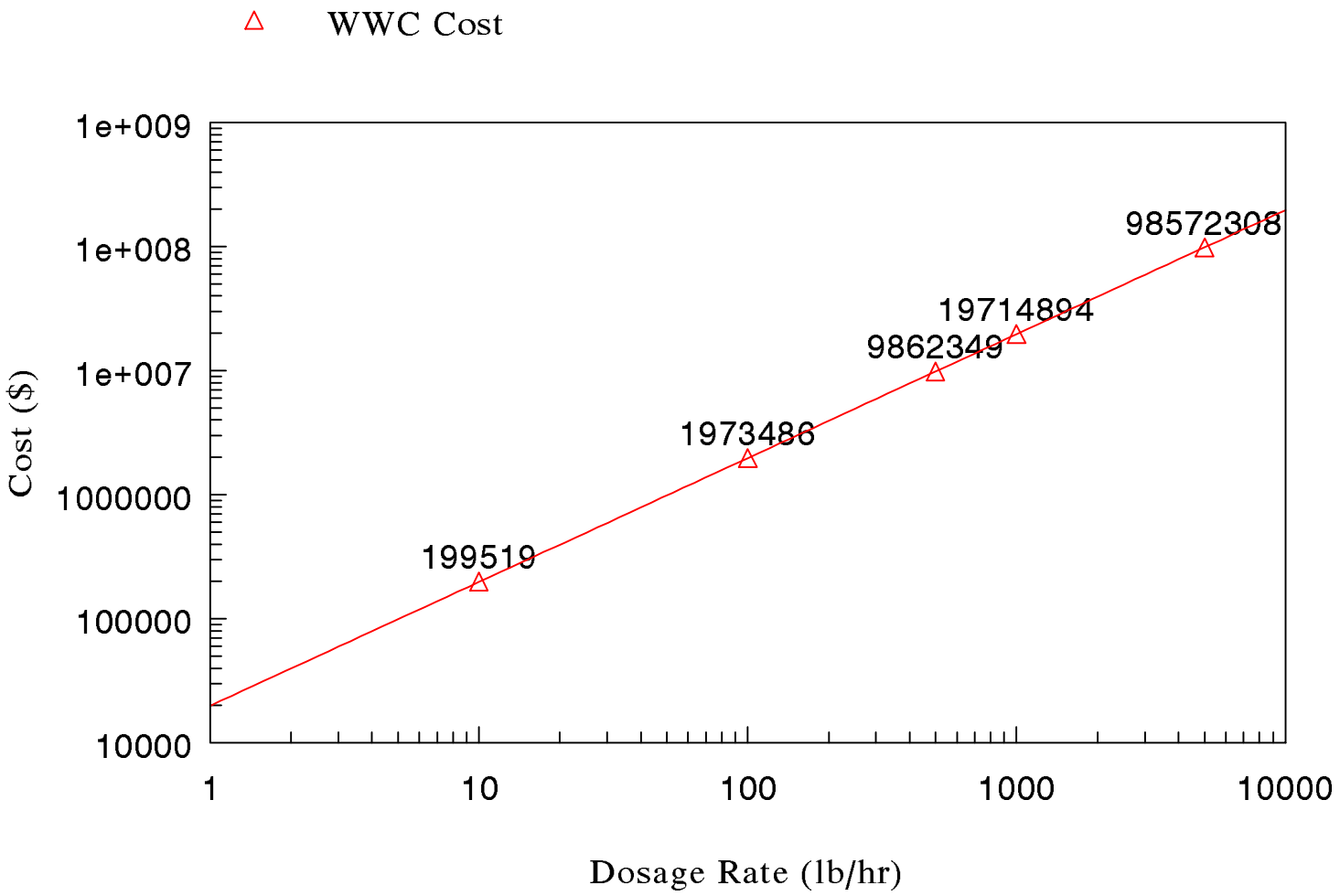


Figure 9-11
Primary Clarifier Capital Cost Curve

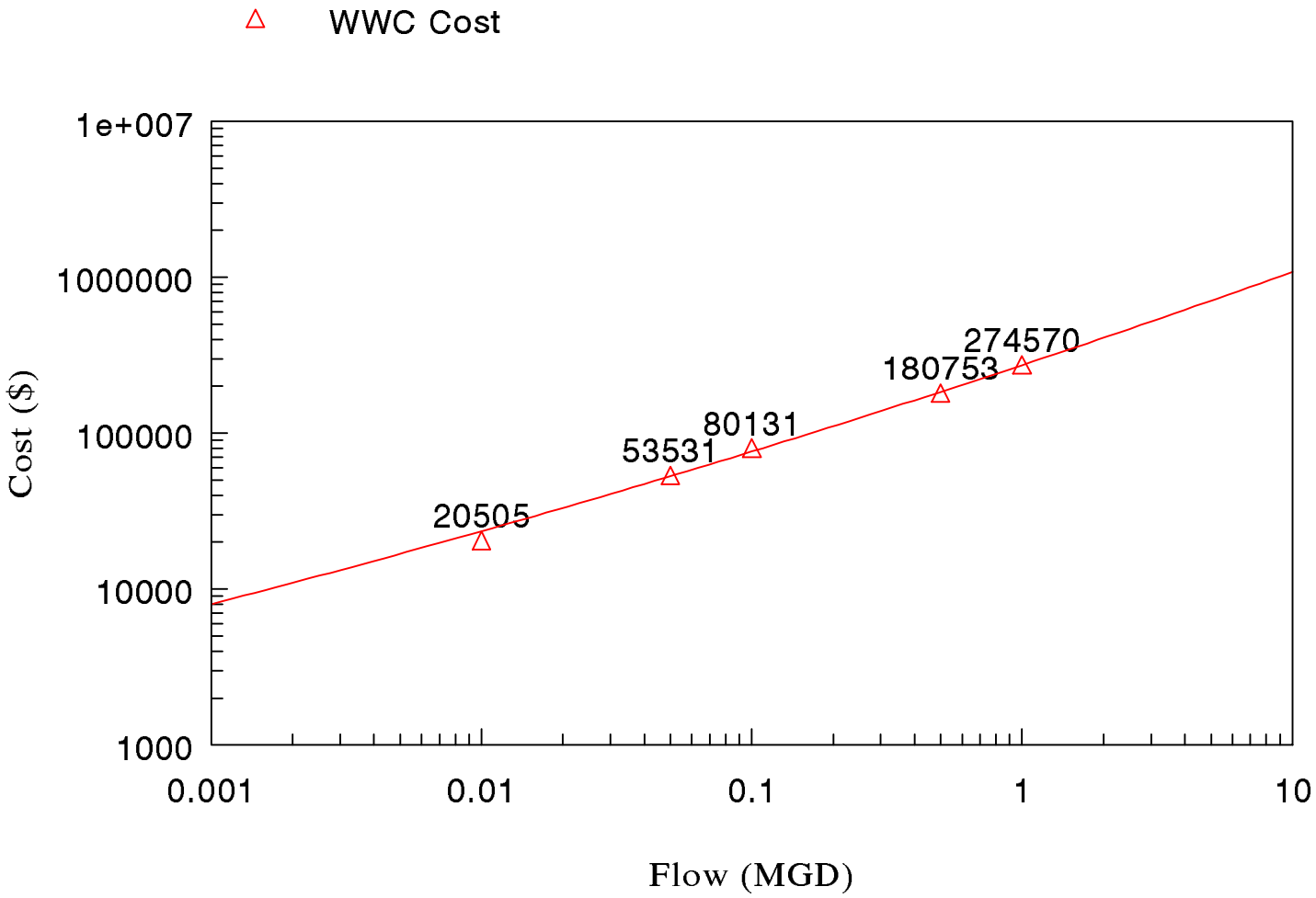


Figure 9-12
Primary Clarifier O&M Cost Curve

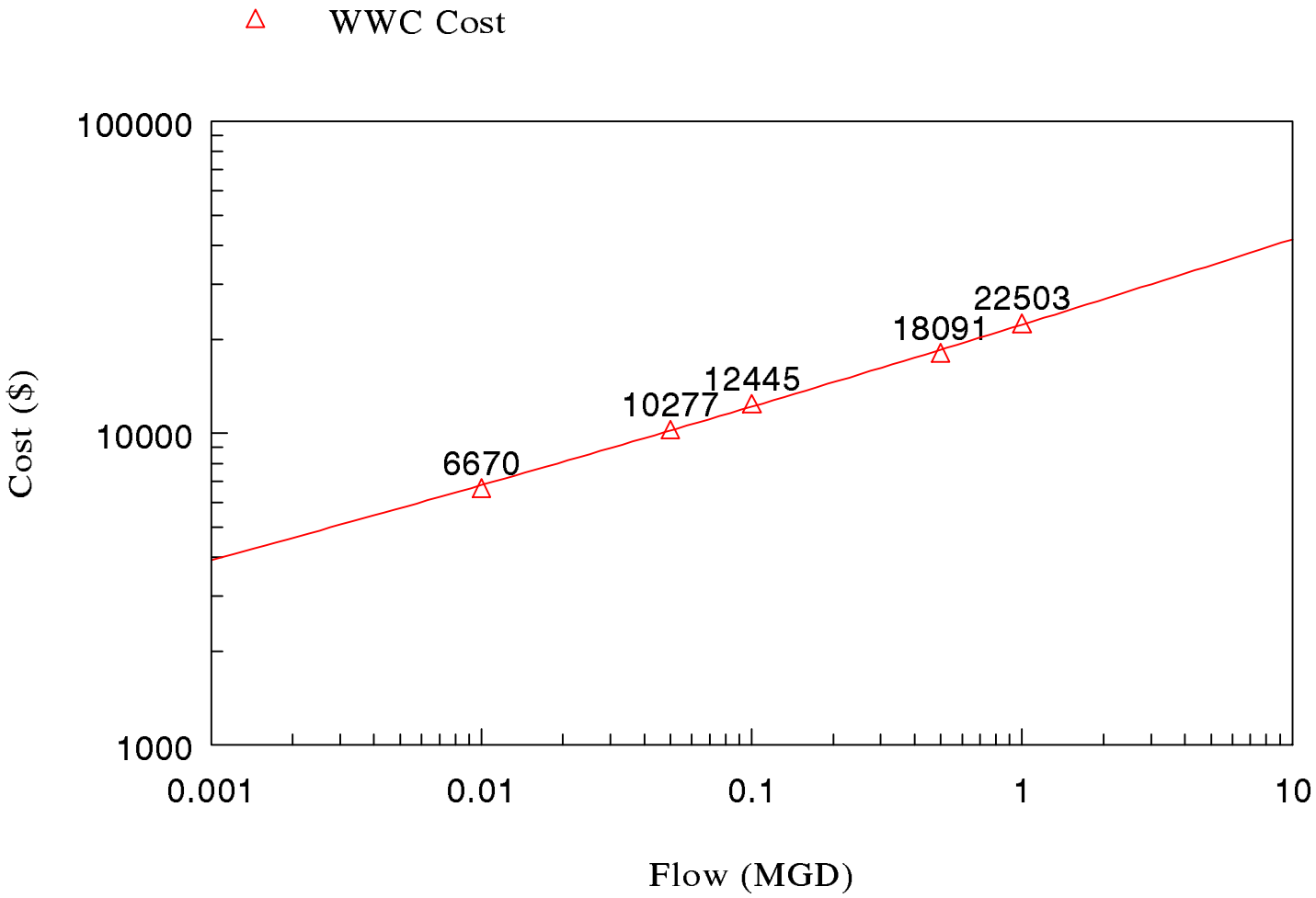


Figure 9-13
Aeration Basin Capital Cost Curve

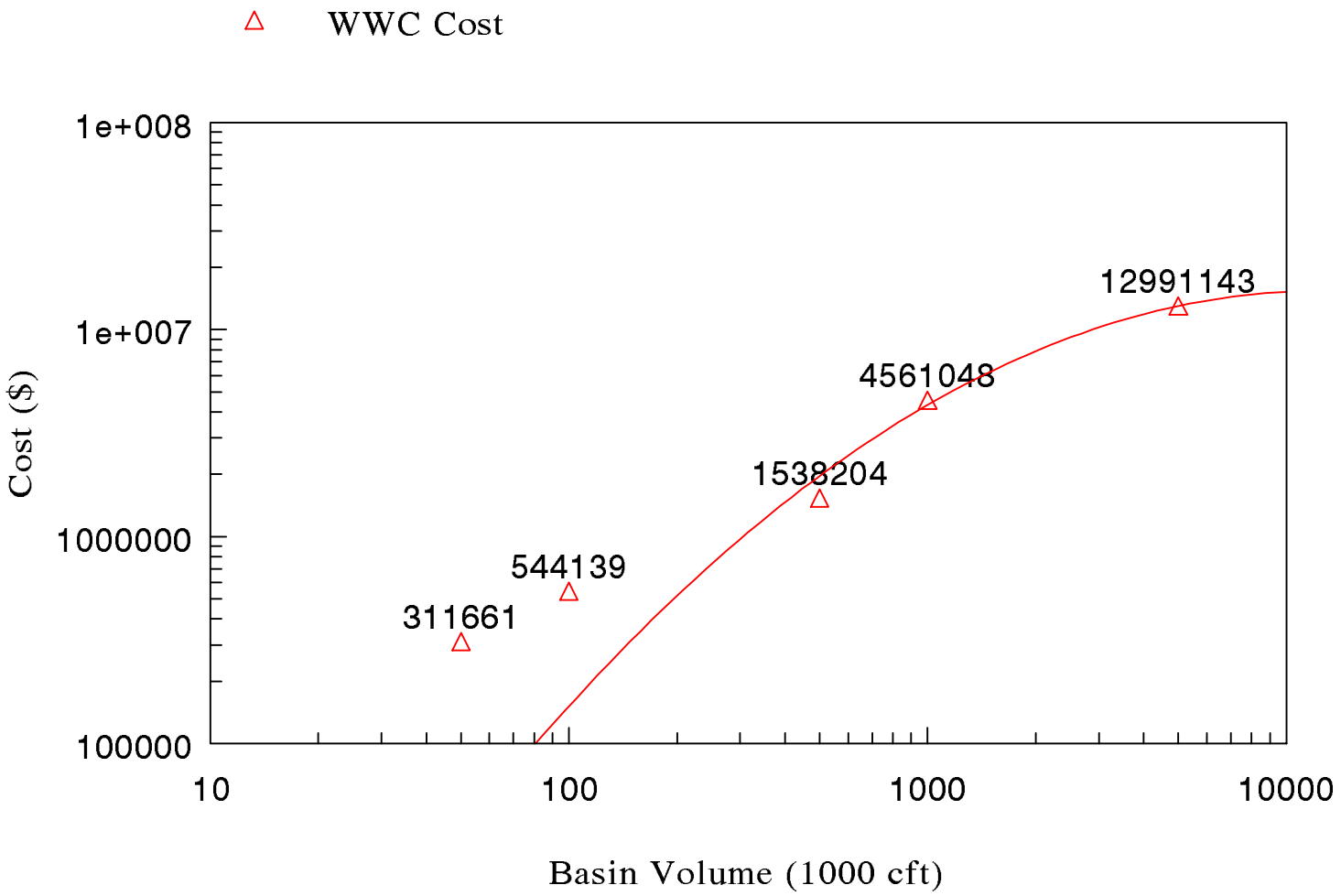


Figure 9-14
Air Diffusion System Capital Cost Curve

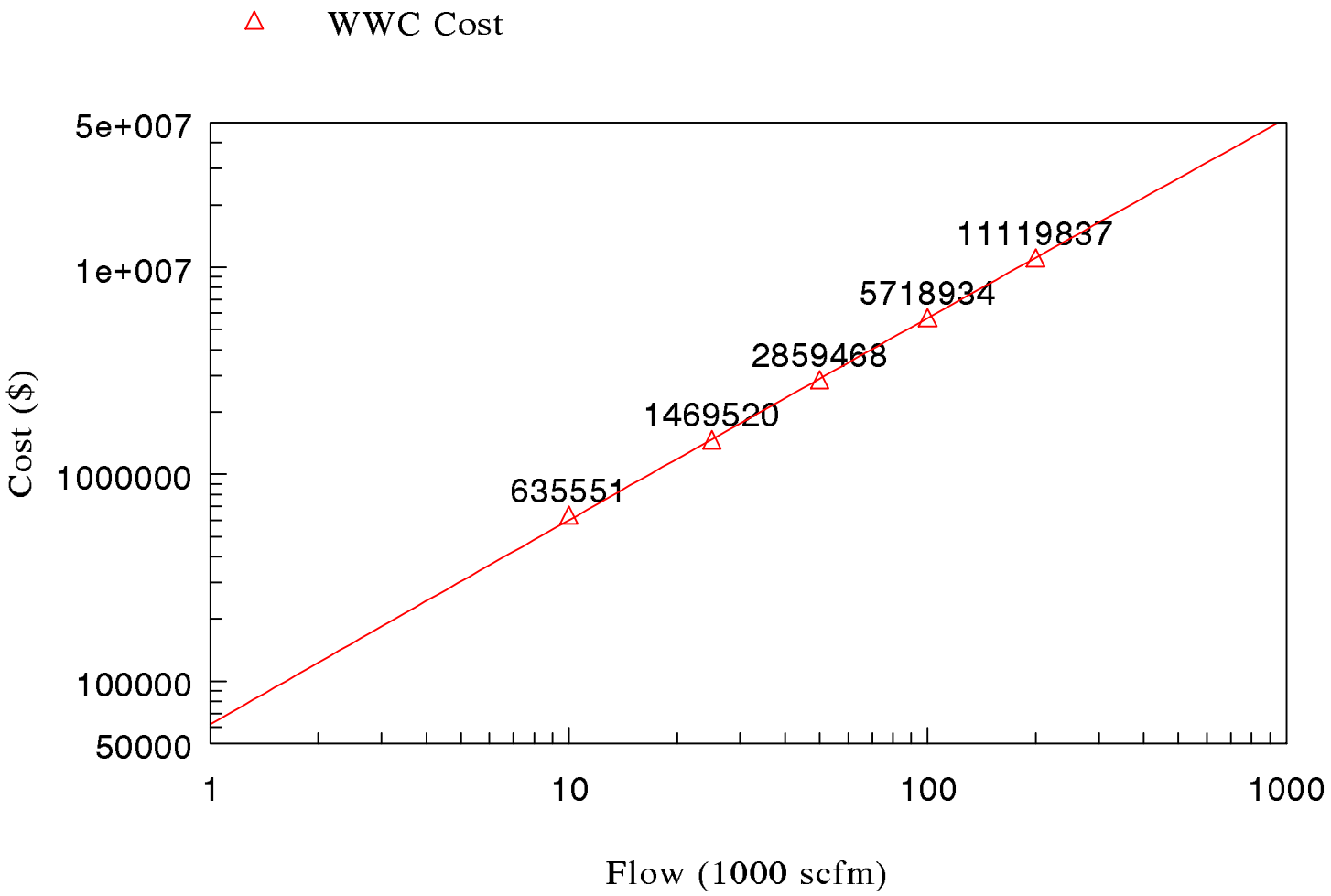


Figure 9-15
Air Diffusion System O&M Cost Curve

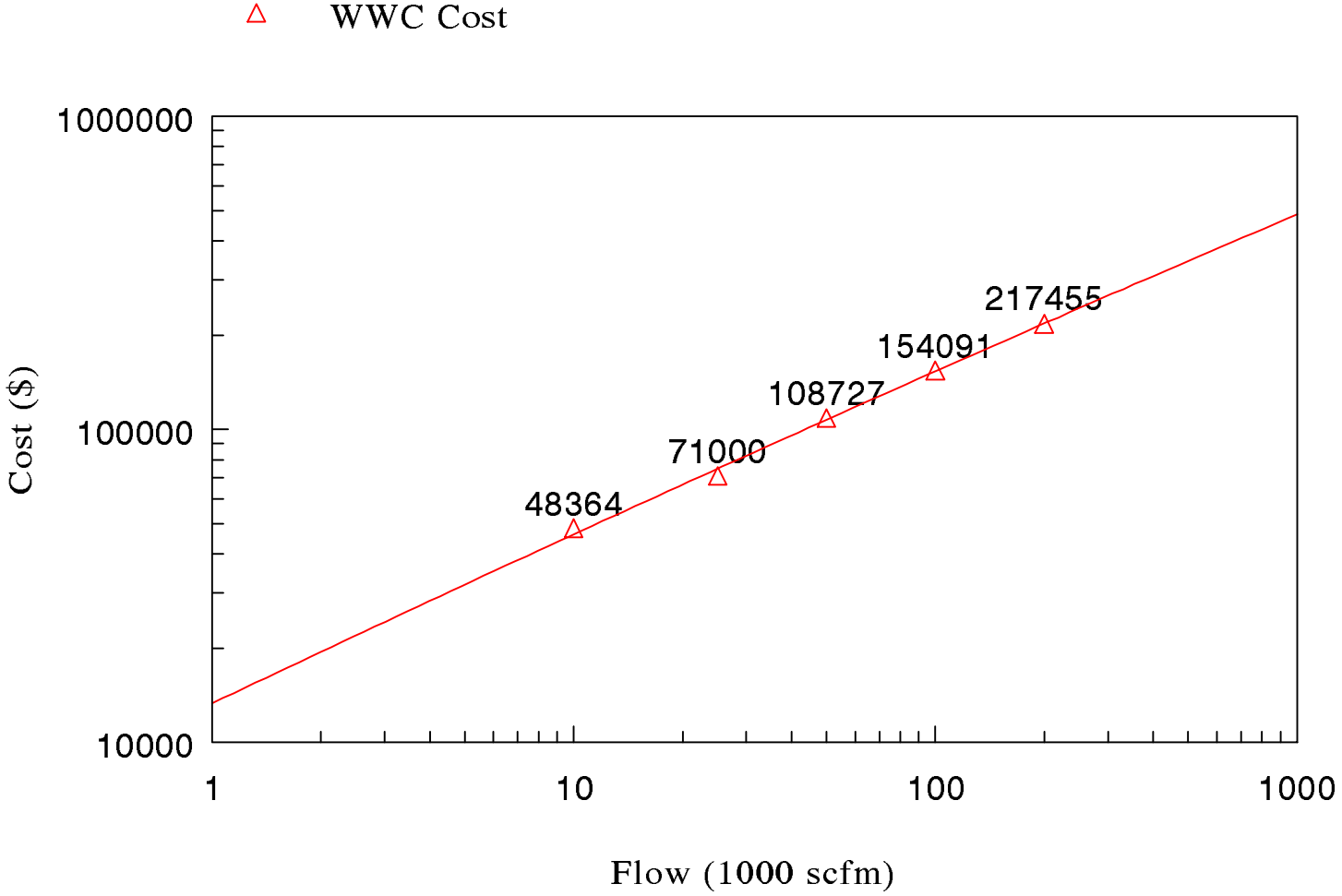


Figure 9-16
Secondary Clarifier Capital Cost Curve

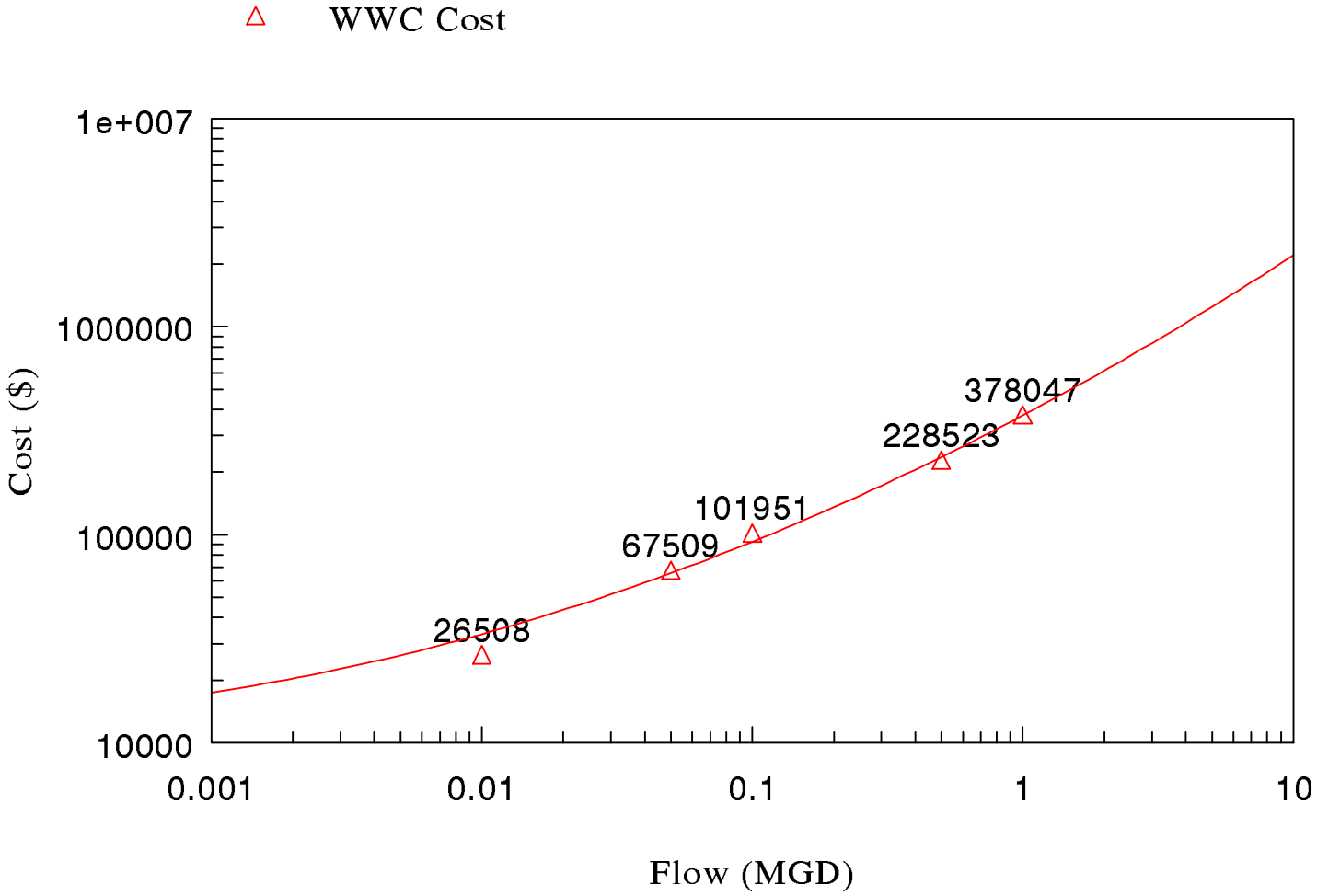


Figure 9-17
Secondary Clarifier O&M Cost Curve

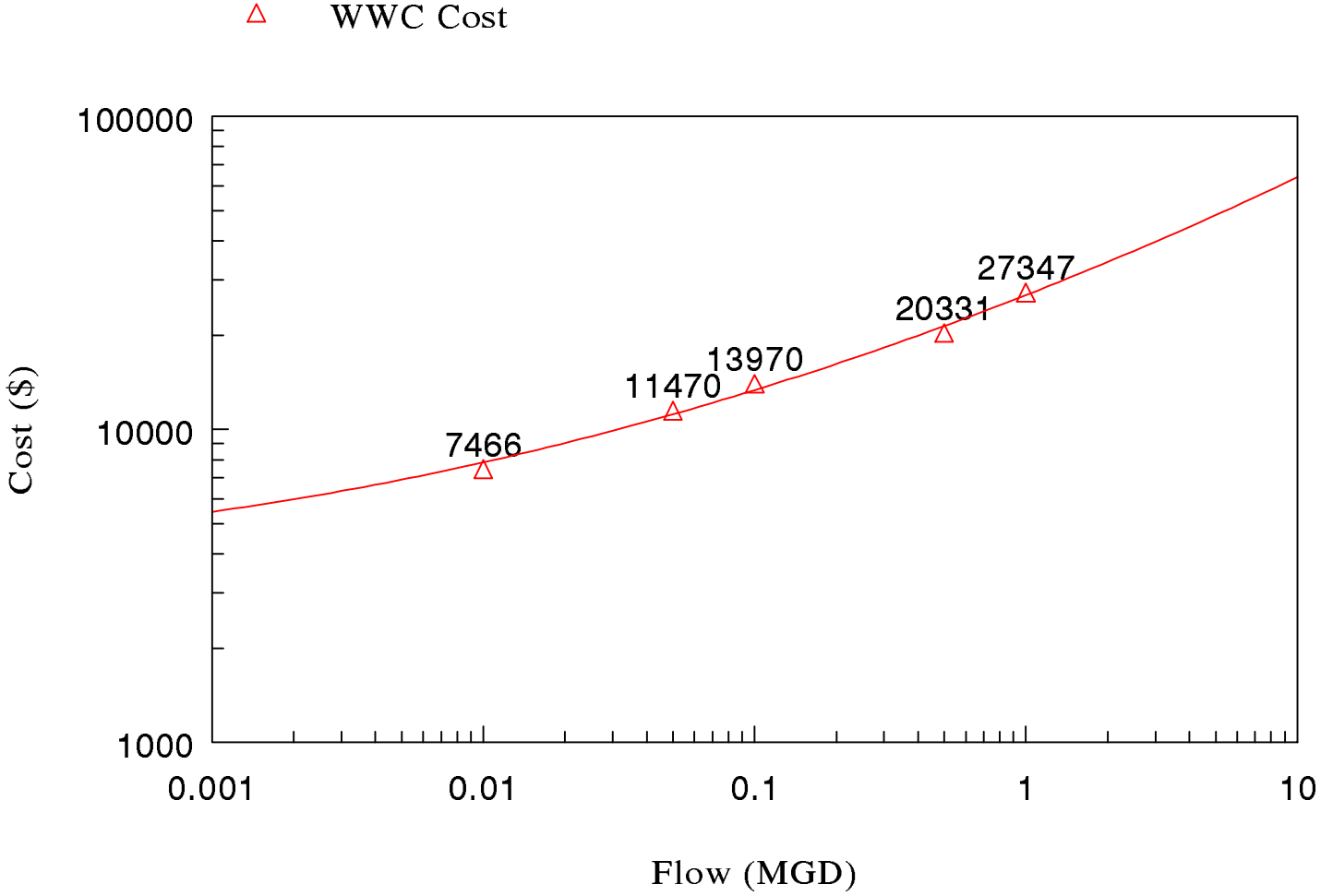


Figure 9-18
Multimedia Filtration Capital Cost Curve

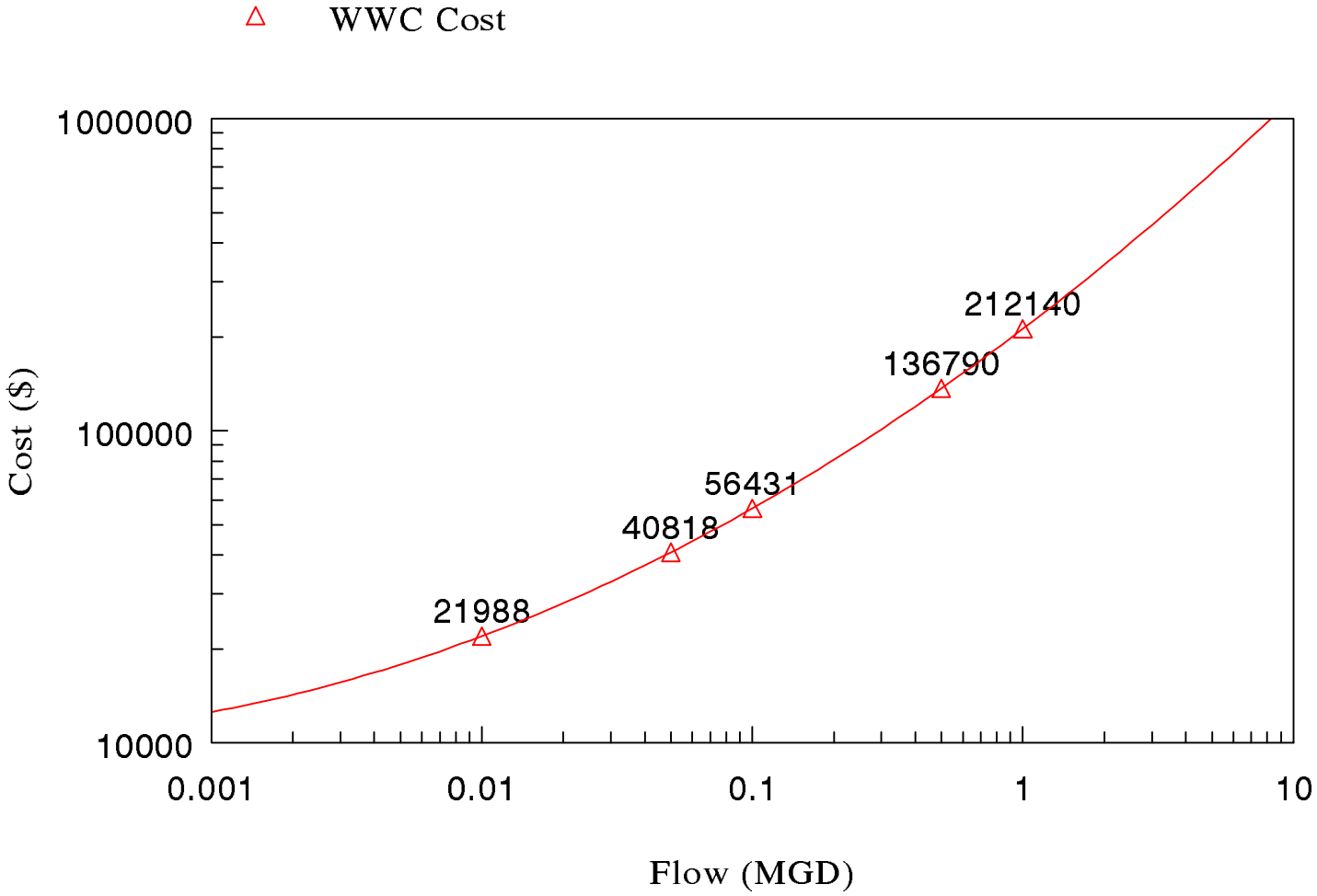


Figure 9-19
Multimedia Filtration O&M Cost Curve

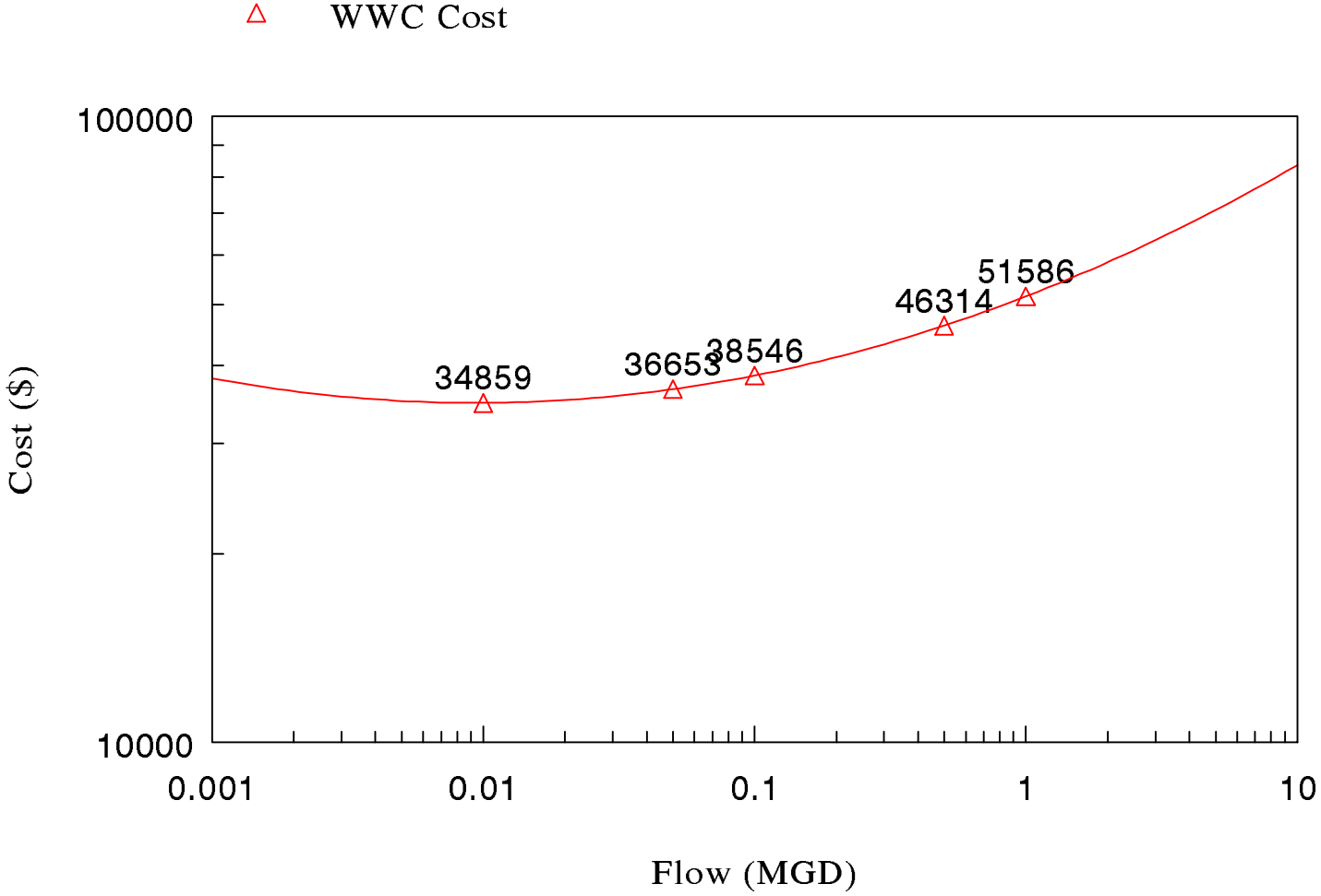


Figure 9-20
Reverse Osmosis Capital Cost Curve

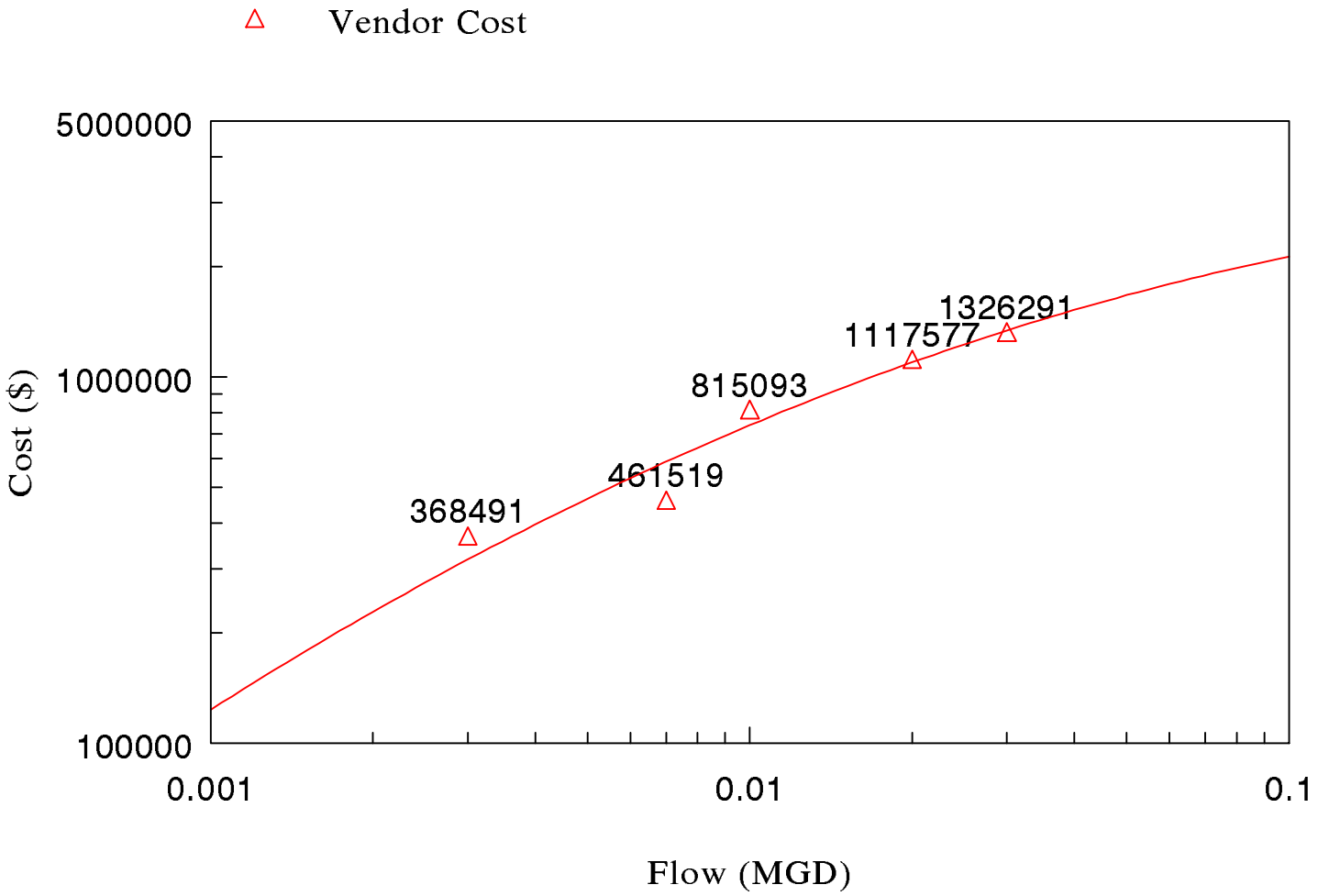


Figure 9-21
Sludge Drying Beds Capital Cost Curve

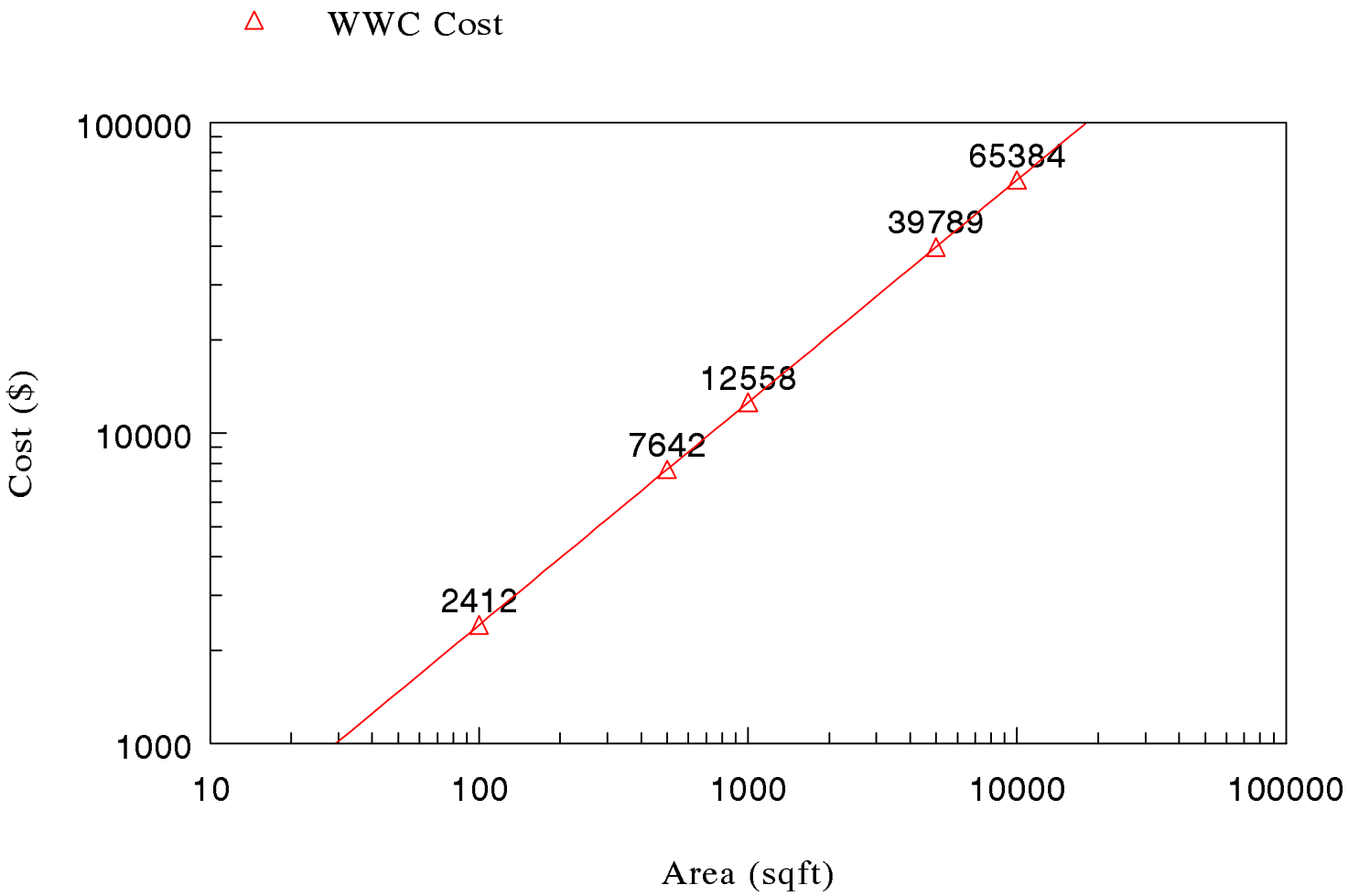


Figure 9-22
Sludge Drying Beds O&M Cost Curve

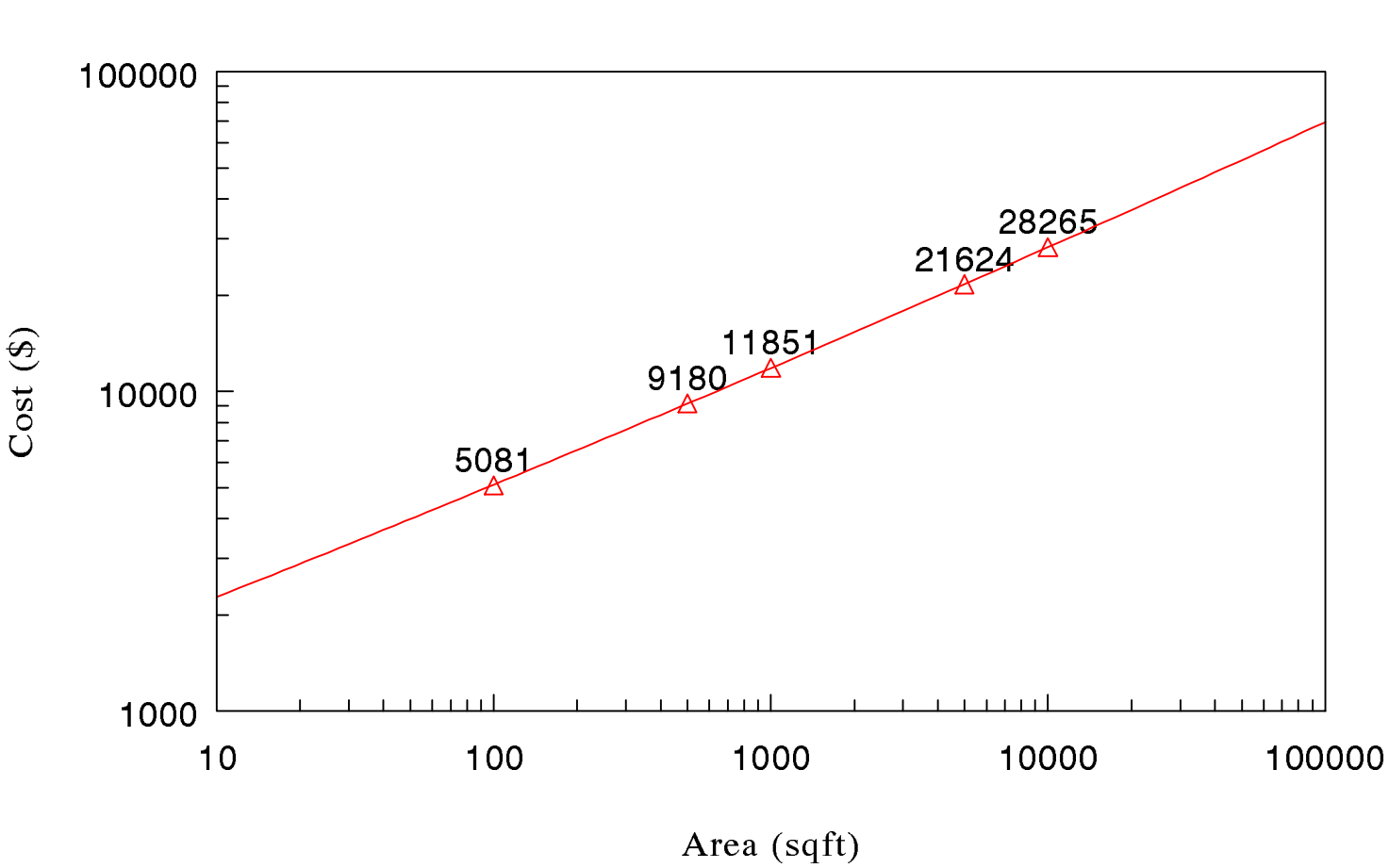


Figure 9-23
GAC Capital Cost Curve

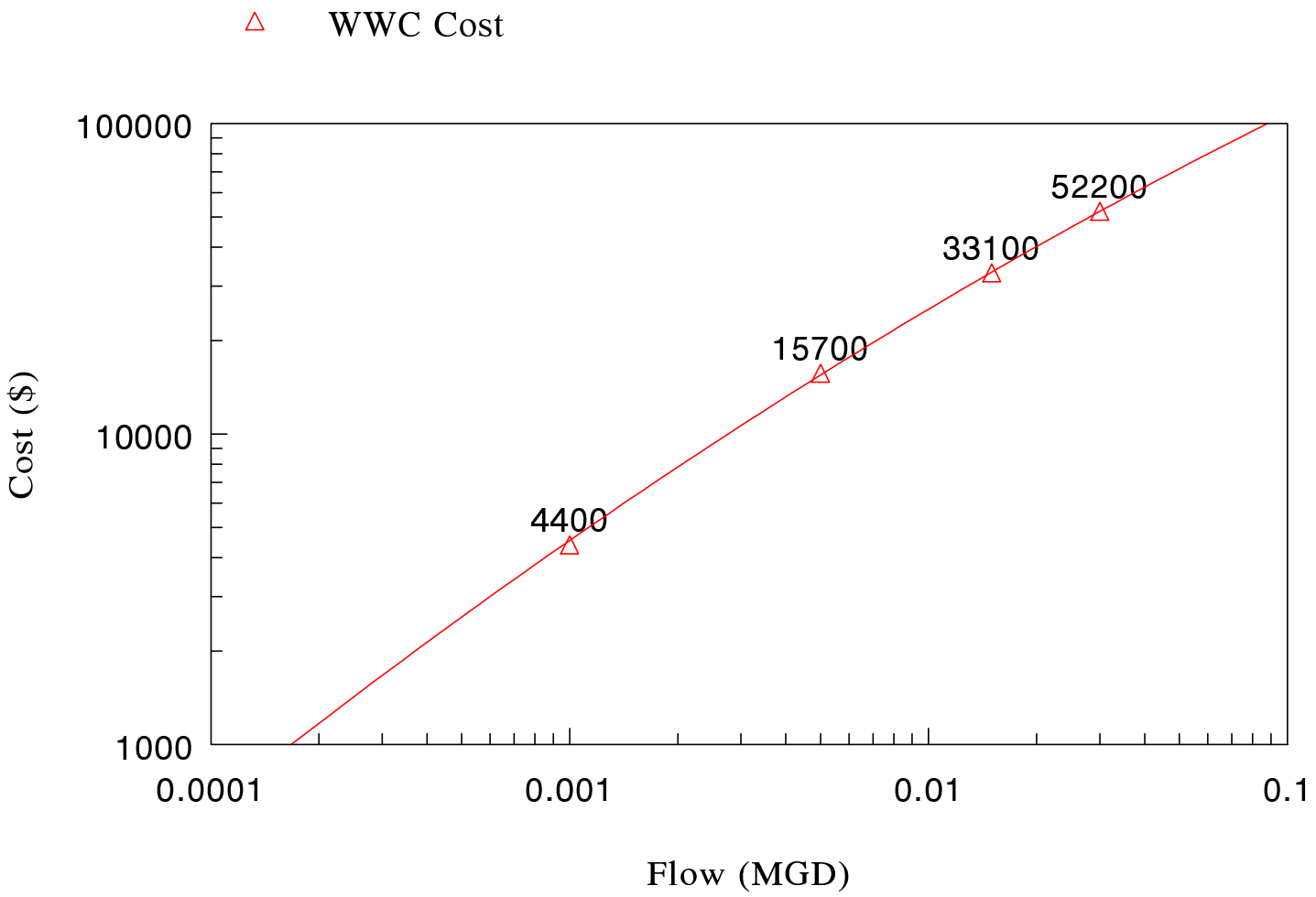


Figure 9-24
GAC O&M Cost Curve

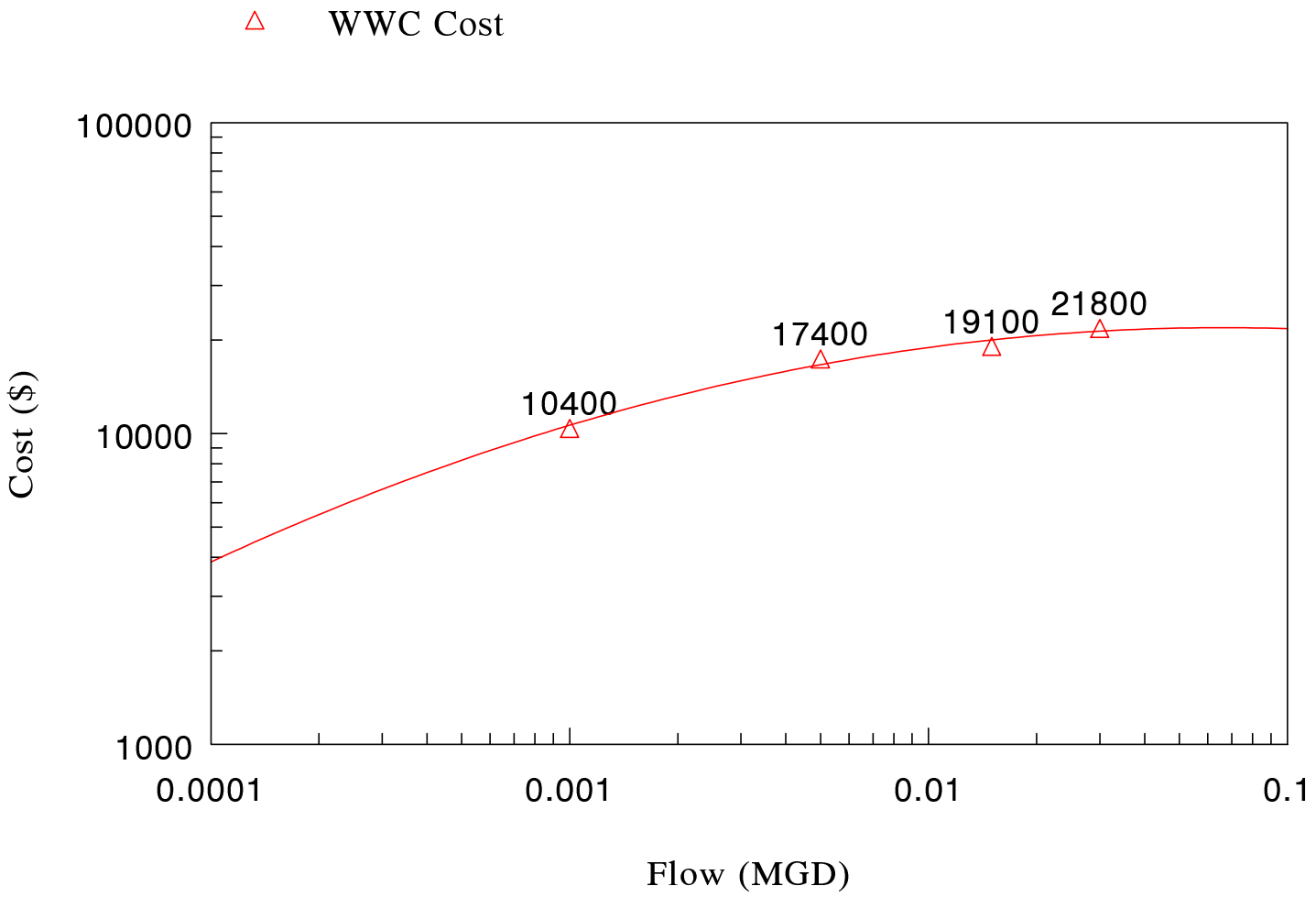


Figure 9-25
Brkpnt Chlorination Capital Cost Curve

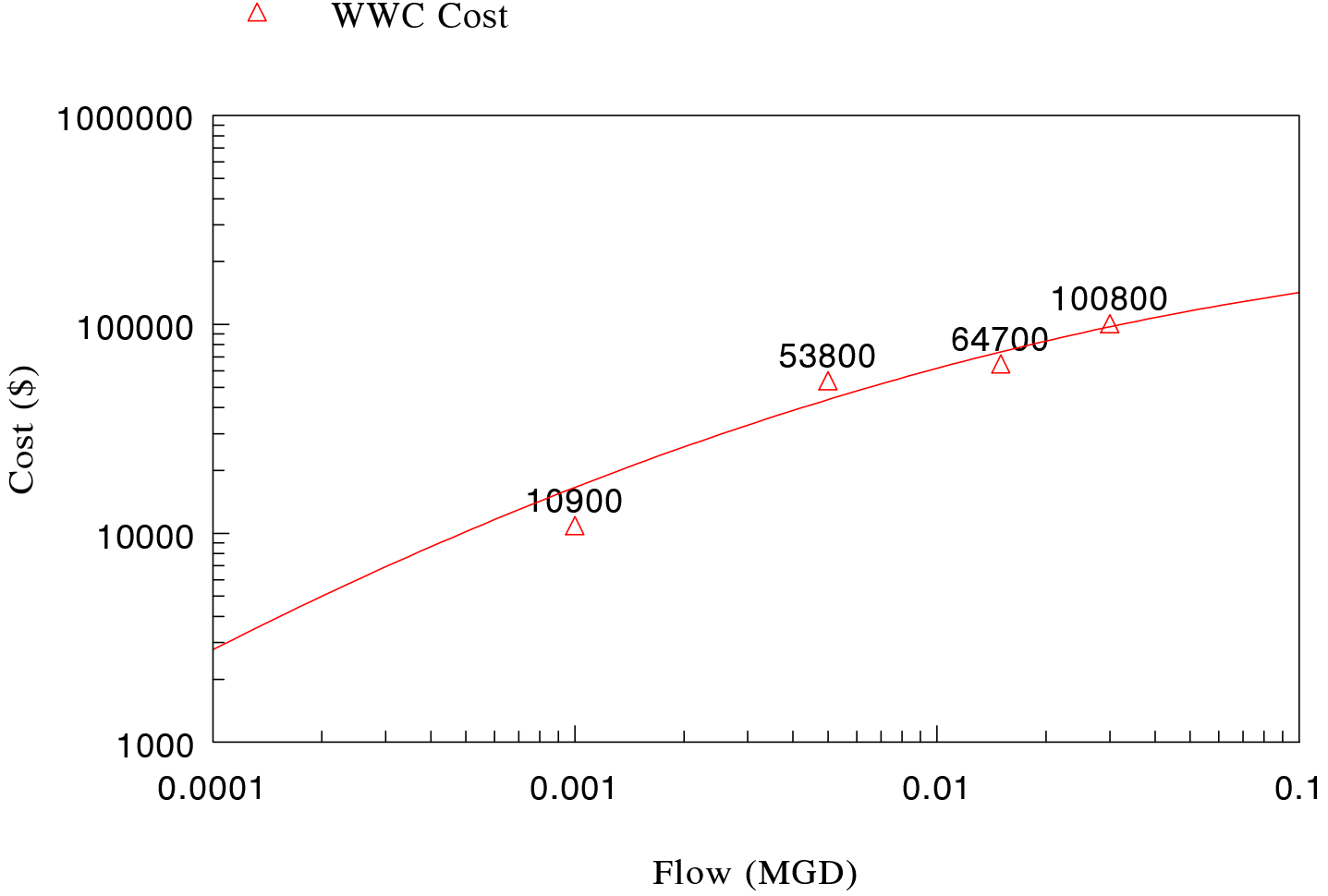
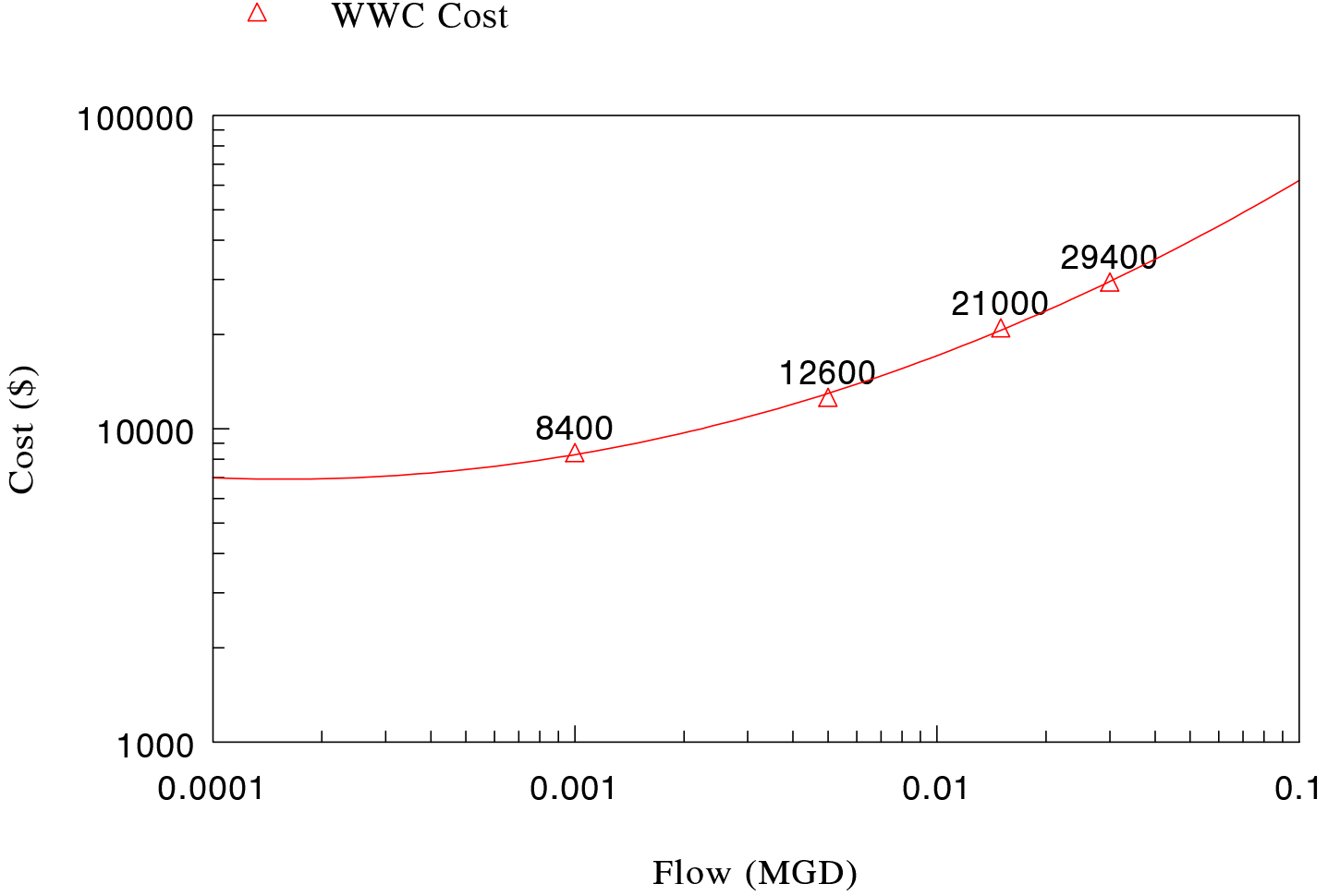


Figure 9-26
Breakpoint Chlorination O&M Cost Curve



10.0 NON-WATER QUALITY IMPACTS

The operation of wastewater treatment systems may have ancillary environmental effects by generating solid and hazardous residuals and air emissions, and by consuming energy in treatment.

The elimination or reduction of one form of pollution may create or aggravate other environmental problems. Therefore, Sections 304(b) and 306 of the Clean Water Act (CWA) require EPA to consider the non-water quality environmental impacts and energy requirements of effluent limitations guidelines and standards. In fulfillment of these requirements, EPA has considered the effect of promulgating the BPT, BCT, BAT, and NSPS regulations for the Landfills industry on the creation of additional air pollution, solid and hazardous waste, and energy consumption.

While it is difficult to balance environmental impacts across all media and energy use, the Agency determined that the impacts identified below do not outweigh the benefits associated with compliance with the limitations and standards.

10.1 Air Pollution

The primary source of air pollution from landfills results from the microbial breakdown of organic wastes from within the landfill. Landfills are known to be major sources of greenhouse gas emissions such as methane and carbon dioxide. These emissions are now regulated under the Clean Air Act (CAA) as a result of the municipal solid waste landfill Standards of Performance for New Stationary Sources and Guidelines for Control of Existing Sources, promulgated by the EPA on March 12, 1996 (Federal Register: Volume 61, Number 49) and codified in 40 CFR 60 Subpart CC-Emission Guidelines and Compliance Times for Municipal Solid Waste Landfills and Subpart WWW-Standards of Performance for Municipal Solid Waste Landfills. Many non-hazardous solid waste landfills are required to collect and combust the gases generated in the landfill. Wastewater collected from within the landfill contains organic compounds which include volatile organic compounds (VOC) and hazardous air pollutants (HAP). This

wastewater must be collected, treated and stored in units which are often open to the atmosphere and may result in the volatilization of certain compounds. Organic pollutants volatilize in reaching an equilibrium with the vapor phase above the wastewater. These volatile organic compounds are emitted to the ambient air surrounding the collection and treatment units. The magnitude of volatile organic compound emissions is dependent on factors such as the physical properties of the pollutants, the temperature of the wastewater, and the design of the individual collection and treatment units.

The landfill effluent guidelines limitations are based on the performance of an aerated biological system. Wastewater aeration may increase the volatilization of certain organic compounds, a potential environmental concern. However, indications are that the potential increase in air emissions due to the final landfill effluent guideline will be minimal. VOCs in hazardous waste landfill leachate are being steadily minimized due to the Resource Conservation and Recovery Act (RCRA) land disposal restriction rules, which typically require aggressive destructive treatment of organics in hazardous wastes before the waste can be landfilled (see 40 CFR 268.40 and 268.48).[1] VOC levels in historic landfill leachate (from both hazardous and non-hazardous waste landfills dating from the 1930s to the mid-1990s) are also at levels which are low enough as not to call into question EPA's determination to base these rules on the performance of aerated biological systems. Tables 6-9, 6-10, and 6-13 in Chapter 6 show the concentrations of VOCs found in landfill wastewater.

Furthermore, EPA's Office of Air and Radiation is currently evaluating the air emissions from wastewater generated at municipal solid waste landfills, and intends to take this rule into account in determining whether further controls under section 112 of the Clean Air Act (which requires technology-based standards for hazardous air pollutants emitted by major sources of emissions of those pollutants) are justified.

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[1] There are certain exceptions to these treatment requirements for hazardous wastewater which is disposed in surface impoundments. RCRA section 3005 (j) (11). However, if this wastewater contains VOCs above a designated concentration level, then the impoundments are subject to rules requiring control of the resulting air emissions. 40 CFR 264.1085 and 263.1086.

(Preliminary indications are that hazardous air pollutant emissions from aeration would be a minor fraction of those from other landfill emission sources such as landfill gas emissions.)

In addition, EPA is addressing emissions of volatile organic compounds from industrial wastewater through a Control Techniques Guideline (CTG) under Section 110 of the CAA. CAA amendments require that State implementation plans for certain ozone nonattainment areas be revised to require the implementation of reasonably available control technology (RACT) for control of volatile organic compound emissions from sources for which EPA has prepared CTGs. In September, 1992, EPA published a draft CTG document entitled "Control of Volatile Organic Compound Emissions from Industrial Wastewater." (EPA-453/0-93-056). This document addresses various industries, including the hazardous waste treatment, storage, and disposal facilities (TSDF) industry, and outlines volatile organic compound emissions expected from their wastewater treatment systems and methods for controlling them. For CTG guideline purposes, EPA has included Subtitle C and D landfills with leachate collection systems in the TSDF industry. EPA estimates that nearly all landfills affected by the Landfills effluent guideline will be subject to this CTG for volatile emissions from their wastewater treatment systems. It was estimated in the CTG draft document that 43 percent of the facilities in the TSDF industry are located in areas of ozone nonattainment. In 1994, the draft CTGs were revised to reflect changes that were made in the wastewater provisions of the Hazardous Organic National Emission Standards for Hazardous Air Pollutants promulgated by the EPA on April 22, 1994 (Federal Register: Volume 59, Number 19). EPA published these changes to the CTGs in a document entitled "Industrial Wastewater Alternative Control Technology".

10.2 Solid and Other Aqueous Waste

Several of the wastewater treatment technologies available to comply with the landfills regulation will generate solid and other aqueous waste. The costs for the disposal of these other waste residuals were included in the compliance cost estimates prepared for the regulatory options. Solid wastes generated by a number of the BPT, BCT, and BAT wastewater treatment technologies include sludge from clarifiers

associated with biological treatment and chemical precipitation systems and backwash waters from filtration systems.

In surveying both subcategories of this industry, EPA determined that it is common practice to dispose of the sludges generated by the on-site wastewater treatment systems directly back into the landfills. This practice eliminates the need for, and the costs associated with, off-site disposal. Analysis of sludge data collected as part of this study also indicates that sludges generated by wastewater treatment systems at landfills in the Subtitle D Non-Hazardous subcategory are non-hazardous, allowing them to be disposed of at the landfill sites from which they are generated.

Waste sludge generated by wastewater treatment facilities at landfills in the Subtitle C Hazardous subcategory may be a hazardous waste, depending upon factors such as the characteristics of the waste deposited in the landfill and the design and operation of the wastewater treatment system. If listed hazardous wastes, as per 40 CFR 261 Subpart D, are disposed of into the landfill, the resultant sludges from the treatment of landfill generated wastewater will be considered a hazardous waste. Based upon the "derived-from" rule found in 40 CFR 261.3(c)(2), the sludge will have the same RCRA waste code as the waste in the landfill for monofills. For hazardous waste landfills which dispose of more than one type of listed hazardous waste and generate a multi-source leachate, the sludge from treatment of the leachate will have the F039 RCRA waste code. Sludges from a treated leachate at a landfill which handles only characteristic wastes, as per 40 CFR 261 Subpart C, will need to be analyzed to determine whether it exhibits any of the characteristics of a hazardous waste as per 40 CFR 261 Subpart C. EPA has developed land disposal restrictions as found in 40 CFR 268. This regulation places restrictions on the land disposal of wastes and specifies treatment standards that must be met before wastes can be land disposed. For purposes of this regulation, EPA has assumed that dried sludges from facilities in the Subtitle C Hazardous subcategory will be returned to the on-site landfill for disposal. Similarly, EPA has assumed dried sludges from Subtitle D non-hazardous facilities will be returned to the on-site landfill for disposal.

Listed or characteristically hazardous waste sludges are to meet applicable treatment standards prior to disposal.

The increased amount of sludge created due to this regulation will be negligible in comparison to the daily volumes of waste processed and disposed in a typical landfill, whether non-hazardous or hazardous. As a result, the practice of on-site disposal has a minimal impact on landfill capacity. For example, based on national estimates, the Subtitle D Non-Hazardous subcategory processed approximately 5,300 million tons of waste in 1992. The BPT/BCT/BAT wastewater treatment options will generate approximately 0.0044 million tons per year of waste solids or only 8.3×10^{-5} percent of the volume of waste disposed into the landfill. For the Subtitle C Hazardous subcategory, the BPT/BCT/BAT option will generate approximately 194 tons per year of solids, as compared to the national estimate of 550 million tons of waste processed, which equates to 3.5×10^{-5} percent.

Filtration backwash waters are generally recycled to the beginning of the wastewater treatment system for reprocessing. This practice eliminates the generation of a waste stream needing disposal.

10.3 Energy Requirements

The operation of wastewater treatment equipment results in the consumption of energy. EPA estimates that the attainment of the BPT, BCT, and BAT standards will increase energy consumption by a very small increment over present industry use. The treatment technologies that are the basis for the limitations and standards are not energy-intensive, and the projected increase in energy consumption is primarily due to the incorporation of components such as power pumps, mixers, blowers, power lighting and controls, and heating devices. The associated energy costs are included in EPA's estimated operating costs for compliance with the guideline presented in Chapter 9. For example, the BPT/BCT/BAT Option 2 for the Subtitle D Non-Hazardous subcategory is estimated to consume 3,300 megawatt-hour per year (Mwhr/year). This is equivalent to approximately 1,800 barrels per year of No.2 fuel oil, as compared to the 1992 rate of consumption in the United States of 40.6 million barrels per year. The additional energy

demand imposed by this regulatory option will represent an insignificant increase in the production or importation of fuel oil. For the Subtitle C Hazardous subcategory, the regulatory option is estimated to consume 37.3 Mwhr/yr or an equivalent 21 barrels per year of No.2 fuel oil.

11.0 DEVELOPMENT OF EFFLUENT LIMITATIONS AND STANDARDS

This chapter presents the final effluent limitations guidelines and standards for the landfills point source category. EPA bases the final effluent limitations upon the performance of selected wastewater treatment systems at landfill facilities and develops limitations expressed as monthly-average and daily-maximum concentrations. The following sections discuss the development of the numerical, technology-based limitations:

- Development of Long-Term Averages, Variability Factors, and Effluent Limitations
- Best Practicable Control Technology Currently Available (BPT)
- Best Conventional Pollutant Control Technology (BCT)
- Best Available Technology Economically Achievable (BAT)
- New Source Performance Standards (NSPS)
- Pretreatment Standards for Existing Sources (PSES)
- Pretreatment Standards for New Sources (PSNS)

11.1 Development of Long-Term Averages, Variability Factors, and Effluent Limitations

The section below presents a summary of the statistical methodology used in the calculation of effluent limitations. (As explained in section 11.6, et.seq., EPA decided not to establish pretreatment standards for landfills). A more detailed explanation can be found in the "Statistical Support Document for the Final Effluent Limitations Guidelines and Standards for the Landfills Point Source Category" (EPA-821-B-99-007).

EPA bases effluent limitations for each subcategory on a combination of long-term average effluent concentrations and variability factors that account for variation in treatment performance within a treatment system over time. The Agency developed variability factors and long-term averages from a database

composed of individual daily measurements of treated effluent at landfills. EPA collected technology performance data from field sampling efforts and from industry-supplied data provided in the Detailed Monitoring Questionnaire. In Chapter 4, EPA presents a detailed description of each data source. While EPA sampling data typically reflects the daily performance of a system over a 5-day period, industry-supplied data for this guideline (collected through the Detailed Monitoring Questionnaire) reflected up to three years of data. The monitoring data obtained through the Detailed Monitoring Questionnaire is unique to each facility in terms of the number of parameters analyzed and the monitoring frequency. Several facilities provided information for dozens of pollutants, while others provided data for only a few parameters. Additionally, monitoring may have been performed weekly, monthly, or quarterly. Wherever possible, when calculating effluent limitations, EPA used a combination of industry-supplied data and EPA sampling data to better account for the variability of the treatment of landfill leachate over time.

EPA used these data to develop long-term average concentrations and variability factors, by pollutant and technology option, for each subcategory. The Agency calculated the final limitations by multiplying long-term average concentrations by the appropriate variability factors. The following paragraphs briefly describe how EPA determined each of these values. As mentioned above, EPA presents the detailed methodology and data in the Statistical Support Document.

11.1.1 Calculation of Long-Term Averages

For each pollutant selected for regulation (see Chapter 7), EPA calculated long-term average effluent concentrations for each regulatory option and subcategory. The first step was to select representative facilities from the EPA database for each option. In Section 11.2, EPA explains the criteria used in facility selection. After selecting the facilities that best represented a technology option, EPA reviewed the influent and effluent data supplied for each of the regulated pollutants. In calculating limitations, the Agency used effluent data from EPA sampling episodes and Detailed Monitoring Questionnaires, but it did not use effluent data from the Detailed Questionnaire. The pollutant data submitted in the Detailed Questionnaire contained the average concentration, the minimum and maximum concentrations, and the number of

samples, whereas EPA sampling data and the Detailed Monitoring Questionnaire consisted of individual daily data. In developing limits, EPA calculated the long-term averages and variability factors using individual daily data. Furthermore, summary data (like the data submitted in the Detailed Questionnaire) may obscure the minimum detection levels used in the sampling data. The use of daily data (like the Detailed Monitoring Questionnaire and EPA sampling data) in developing limitations allows EPA to account for concentration values reported at or below the detection limits. EPA set observations below the sample-specific detection level equal to the detection level for the purposes of calculating a facility-level long-term average. In addition, in many cases, EPA considered reported averages from the Detailed Questionnaires redundant because many facilities also reported the daily data from the Detailed Monitoring Questionnaire for the same time period in 1992 and, therefore, EPA would not have used the data in the calculation of limits. However, in determining whether a pollutant was present at treatable levels, EPA relied on data from any of the three pollutant data sources: Detailed Questionnaire, Detailed Monitoring Questionnaire, and EPA analytical sampling episodes. EPA used effluent data from a facility only if sufficient influent data were available to establish the presence of treatable levels of pollutants. In addition, for each of the regulated pollutants, the Agency analyzed all of the selected facilities to determine if the facility was utilizing treatment technologies, apart from those selected as the technology option, that may provide significant removals of that particular pollutant. For example, the data from a facility that employed carbon adsorption (a treatment technology that was not part of a selected technology option) would not be used in the calculation of the limit for a pollutant that may be treated by carbon adsorption. However, if an intermediate data point that preceded carbon adsorption treatment were available for this facility, then EPA did consider the use of that data point to characterize the performance of the treatment system up to that point. Furthermore, EPA edited EPA sampling data according to the criteria outlined in Chapter 4, Section 4.9.

Once EPA selected the facilities and effluent data points, the Agency calculated the average effluent concentration for each regulated pollutant at each facility. For facilities that EPA had data for both five-day EPA sampling and industry-supplied Detailed Monitoring Questionnaires (representing data collected over

the course of at least a year), EPA calculated long-term averages separately as long as the dates for the two data sets did not overlap. Therefore, by using both data sets, the long-term average accounted for the variability of leachate over a longer period of time.

The Agency estimated the long-term average for each regulated pollutant for each BPT/BAT facility by fitting a modified delta-lognormal distribution to the daily concentration data. The modified delta-lognormal distribution models the data as a mixture of non-detect observations and measured values that follow a lognormal distribution. The Agency selected this distribution because of the following reasons: (1) the data for many analytes consisted of a mixture of non-detects and measured values that were approximately lognormal, and (2) in cases where there are no non-detects, the distribution is equivalent to the usual two-parameter lognormal. This is the same basic distributional model used by EPA in the final rulemakings for the Organic Chemicals, Plastics and Synthetic Fibers (OCPSF; 40 CFR Part 414) and the Pulp and Paper category (40 CFR Part 430) and for the proposed rulemaking for the Centralized Waste Treatment industrial category (proposed 40 CFR Part 437, 64 FR 2280 January 13, 1999). In the Pulp and Paper and the Centralized Waste Treatment studies, the modified delta-lognormal distribution assumes that all non-detects have a value equal to the reported sample-specific detection levels and that the detected values follow a lognormal distribution. EPA again used this model as the basis of estimates of the long-term average at a landfill facility. In the case of the OCPSF rule, EPA used the same basic model but the reported non-detect values were set equal to the pollutant analytical minimum level. A more detailed discussion of the modified delta-lognormal distribution can be found in the "Statistical Support Document for the Final Effluent Limitations Guidelines and Standards for the Landfills Point Source Category" (EPA-821-B-99-007).

After EPA developed the facility-level long-term averages for each regulated pollutant using the criteria outlined above, the Agency determined the median of the facility-level long-term averages for each regulated pollutant in each subcategory. The median of the facility-level long-term averages for each

regulated pollutant was the long-term average used in the calculation of the effluent limitation as described later in this section.

11.1.2 Calculation of Variability Factors

EPA calculated variability factors using the same data sets used to derive the long-term average values. As with the calculation of the long-term averages, EPA fit the daily concentration data to a modified delta-lognormal distribution. The Agency calculated separate variability factors for different averaging periods (either 1-day, 4-day, or 20-day averages). Thus, EPA applied different variability factors to daily data (single measurements without averaging) and to monthly-average data based on four measurements taken once per week ("4-day averages") or 20 measurements taken once each day, five days a week throughout a month ("20-day average").

For those facility data sets that had at least four observations for a given regulated pollutant, including two detected values, EPA used the modified delta-lognormal model to estimate daily and 4-day or 20-day average variability factors. There were several instances where EPA could not calculate variability factors from the landfills database because EPA measured fewer than two samples above the detection limit. In these cases, the Agency transferred variability factors from biological treatment systems used in the final rulemaking of the OCPSF guideline (40 CFR Part 414).

As stated above, in calculating the variability factors, EPA assumed a log-normal distribution of the data. In addition, the Agency used the following:

- The 95th percentile to establish the maximum monthly average.
- The 99th percentile to establish the maximum for any one day.

EPA defines the daily variability factor as the ratio of the estimated 99th percentile of the distribution of daily values to the estimated mean of the distribution. Similarly, the Agency defines the monthly variability

factor as the estimated 95th percentile of the distribution of 4-day or 20-day averages divided by the estimated mean of the monthly averages. EPA derived a monthly-average and daily-maximum variability factor for each pollutant and for each regulatory option. For each subcategory, the Agency defined the daily variability factor for each pollutant as the average of the facility-level daily variability factor. Likewise, EPA defines the 4-day average variability factor for each pollutant as the average of the facility-level 4-day average variability factors and the 20-day average variability factor for each pollutant as the average of the facility-level 20-day average variability factors.

11.1.3 Calculation of Effluent Limitations

The Agency used the median long-term average and the average variability factor for each pollutant in the calculation of the effluent limitations. For each subcategory, EPA calculated the daily-maximum limitations by multiplying the median of the long-term average for a given pollutant by the average daily variability factor for that pollutant. EPA calculated the monthly-maximum limitations by multiplying the median long-term average for a given pollutant by the average 4-day or 20-day variability factors for that pollutant. The Agency used twenty-day average limitations for the conventional pollutants, BOD_5 and TSS, and four-day average limitations for other nonconventional and toxic pollutants.

11.2 Best Practicable Control Technology Currently Available (BPT)

EPA promulgated BPT effluent limitations for the Subtitle D Non-Hazardous and Subtitle C Hazardous subcategories. BPT effluent limitations control identified conventional, toxic, and nonconventional pollutants when discharged from landfill facilities to surface waters of the U.S. Generally, EPA determines BPT effluent levels based on the average of the best existing performance by facilities of various sizes, ages, and unit processes within an industrial category or subcategory. In industrial categories where present practices are uniformly inadequate, however, EPA may determine that BPT requires higher levels of control than any currently in place if the technology to achieve those levels can be practicably applied. BPT may be transferred from a different category or subcategory. BPT normally focuses on end-of-process treatment

rather than process changes or internal controls, except when these technologies are common industry practice.

In addition, the Clean Water Act (CWA) Section 304(b)(1)(B) requires a cost-reasonableness assessment for BPT limitations. In determining the BPT limits, EPA must consider the total cost of treatment technologies in relation to the effluent reduction benefits achieved. This inquiry does not limit EPA's broad discretion to adopt BPT limitations that are achievable with available technology unless the required additional reductions are "wholly out of proportion to the costs of achieving such marginal level of reduction." A Legislative History of the Water Pollution Control Act Amendments of 1972, p. 170. Moreover, the inquiry does not require the Agency to quantify benefits in monetary terms. See e.g. *American Iron and Steel Institute* v. *EPA*, 526 F. 2d 1027 (3rd Cir., 1975).

In assessing the costs relative to the benefits of effluent reduction, EPA considers the volume and nature of expected discharges after application of BPT, the general environmental effects of pollutants, and the cost and economic impacts of the required level of pollution control. In developing guidelines, the Act does not require or permit consideration of water quality problems attributable to particular point sources, or water quality improvements in particular bodies of water. Therefore, EPA has not considered these factors in developing the final limitations. See *Weyerhaeuser Company* v. *Costle*, 590 F. 2d 1011 (D.C. Cir. 1978).

In setting BPT limitations based on a treatment technology, EPA does not require the use of that technology to treat landfill wastewater. Rather, to establish the limits, EPA has demonstrated that the concentration limits are achievable based on a well-operated system using selected technologies. The technologies that may be used to treat wastewater are left entirely to the discretion of the individual landfill operator, as long as the numerical discharge limits are achieved.

11.2.1 BPT Technology Options for the Subtitle D Non-Hazardous Subcategory

In the Agency's engineering assessment of the best practicable control technology currently available for treatment of wastewater from landfills, EPA first considered three technologies commonly in use by landfills and other industries as options for BPT: chemical precipitation, biological treatment, and multimedia filtration.

For its evaluation of chemical precipitation, EPA collected raw wastewater and treated effluent data from several non-hazardous landfills employing this technology. Based on these data, EPA removed chemical precipitation from further consideration as a BPT treatment option. While chemical precipitation is an effective treatment technology for the removal of metals, non-hazardous landfills typically have low concentrations of metals in treatment system influent wastewater. Observed metals concentrations were typically not found at levels that would inhibit biological treatment or that would be effectively removed by a chemical precipitation unit.

EPA sampling data collected at facilities in the Non-Hazardous subcategory showed relatively low levels (less than 1 mg/L) of pollutant of interest metals in untreated landfill generated wastewater. Furthermore, Table 11-1 presents several sources of performance data for metals removals in activated sludge systems along with published biological treatment inhibition ranges and raw wastewater characteristics from the non-hazardous facilities in the EPA database. Performance data for metals from biological treatment systems were obtained from the National Risk Management Research Laboratory (NRMRL) Treatability Database (formerly called the Risk Reduction Engineering Laboratory (RREL) Treatability Database), the 50-POTW Study, and a sampling program conducted at twelve OCPSF facilities that have biological treatment systems. Metal concentrations in the raw wastewater for this subcategory are below, or close to, the published inhibition levels for biological treatment systems. A review of performance data indicates that certain pollutant of interest metals, such as chromium and zinc, are removed by well-operated biological treatment processes at relatively high rates. See Table 11-1.

Based on this analysis, EPA concluded that pollutant of interest metals observed in the Non-Hazardous subcategory generally are present in landfill generated wastewater at levels that should not effect the operation and performance of a biological treatment system. Under these circumstances, biological treatment removes the metals identified as pollutants of interest in the Non-Hazardous subcategory. Therefore, EPA concluded that biological treatment is an adequate BPT control technology for pollutant of interest metals in the Non-Hazardous subcategory.

Based on the above assessment, EPA developed the following BPT regulatory options. Chapter 8 discusses these two technology options in detail and Chapter 9 discusses the cost estimates developed for these options.

Non-Hazardous Subcategory Option I: Biological Treatment

EPA first assessed the pollutant removal performance of equalization and biological treatment. EPA evaluated this as Option I due to its effectiveness in removing the large organic loads commonly associated with leachate. BPT Option I consists of aerated equalization followed by biological treatment. EPA included various types of biological treatment such as activated sludge, aerated lagoons, and anaerobic and aerobic biological towers or fixed film reactors in calculating limits for this option. The Agency based the costs for Option I on the cost of aerated equalization followed by an extended aeration activated sludge system and clarification, including sludge dewatering. Figure 11-1 presents a flow diagram of the treatment system costed for Option I. Approximately thirty percent of the direct-discharging municipal solid waste landfills employed some form of biological treatment, and fifteen percent had a combination of equalization and biological treatment.

Non-Hazardous Subcategory Option II: Biological Treatment and Multimedia Filtration

The second technology option considered for BPT treatment of non-hazardous landfill wastewater was equalization prior to biological treatment, followed by secondary clarification and multimedia filtration. EPA evaluated this as Option II due to its effectiveness in removing the large organic loads and suspended solids

commonly associated with leachate. Approximately nine percent of the direct discharging non-hazardous facilities used the technologies described in Option II. EPA based cost estimates for Option II on the cost of Option I plus a multimedia filtration system. Figure 11-2 presents a flow diagram of the treatment system costed for this option.

Selected BPT Technology Option

EPA selected Option II, equalization prior to biological treatment followed by secondary clarification and multimedia filtration, as the technology basis for BPT limitations for the Non-Hazardous landfills subcategory. EPA selected Option II for the basis of BPT limitations because of the demonstrated ability of biological treatment systems in controlling organics and the effectiveness of multimedia filtration in removing TSS. EPA's decision to base BPT limitations on Option II treatment primarily reflects two factors: the degree of effluent reductions attainable and the total cost of the treatment technologies in relation to the effluent reductions achieved. In assessing BPT, EPA considered the age, size, process, other engineering factors, and non-water quality impacts pertinent to the facilities treating wastes in this subcategory. No basis could be found for identifying different BPT limitations based on age, size, process or other engineering factors. Neither the age nor the size of the landfill facility will directly affect the treatability of the landfill wastewater, as discussed in Chapter 5. For the non-hazardous landfills, the most pertinent factors for establishing the limitations are costs of treatment and the level of effluent reductions obtainable.

EPA has selected Option II based on the comparison of the two options in terms of total costs of achieving the effluent reductions, pounds of pollutant removals, economic impacts, and general environmental effects of the reduced pollutant discharges. BPT Option II removed significantly more pounds of conventional pollutants than Option I with only a moderate associated cost increase. EPA estimated that BPT Option II will cost $340,000 (1998 dollars) more annually than BPT Option I for an additional removal of 142,000 pounds of conventional pollutants (mainly TSS).

Finally, EPA analyzed the costs of both options to determine the economic impact that this rule would have on the Landfills industry. EPA's assessment showed that, under either option, only two facilities would incur significant economic impacts. For this assessment, EPA defined significant economic impacts in two different ways, depending on the ownership of the facility. For privately-owned facilities, significant economic impacts exist when the facility's after-tax cash flow is negative following the addition of compliance costs. For municipally-owned facilities, significant economic impacts occur when the ratio of compliance costs to median household income are greater than one percent. The economic assessment for the final rule is described in the "Economic Analysis for the Final Effluent Limitations Guidelines and Standards for the Landfills Point Source Category." (EPA-821-B-99-005).

11.2.2 BPT Limits for the Subtitle D Non-Hazardous Subcategory

Selection of BPT Facilities

EPA based the final BPT effluent limitations for the Non-Hazardous subcategory on the average of the best existing wastewater treatment systems. The first criterion used in the selection of the average of the best facilities was effective treatment of BOD_5. In selecting BPT facilities, EPA identified facilities that employed either Option I or Option II technologies. Even though EPA selected Option II technologies as the basis for developing the BPT effluent limitations, EPA assumed that very little additional BOD_5 removal would occur because of the multimedia filter employed in Option II and, therefore, facilities employing biological treatment only (Option I) could achieve good removal of BOD_5 and be considered BPT. However, in determining the BPT effluent limitations for TSS, EPA only used the data from the best performers using the entire BPT Option II technology (biological treatment plus a filter) because of the multimedia filtration system's effectiveness in removing suspended solids.

There were 45 municipal solid waste landfill facilities (see Table 11-2) in the EPA database in the Non-Hazardous subcategory that utilized a biological treatment system that was considered for BPT. Even though both Subtitle D municipal solid waste landfills and non-municipal solid waste landfills make up the Non-Hazardous subcategory, EPA only considered municipal solid waste facilities for selection as BPT

for the Non-Hazardous subcategory because the wastewater at these landfills tends to contain a wider array of pollutants than that found at Subtitle D non-municipal facilities. The pollutants found at the non-municipal facilities tended to be a subset of the pollutants found at the municipal facilities. In fact, all nine pollutants of interest for non-municipal facilities were also pollutants of interest for the municipal facilities (see Chapter 7). In addition, EPA's data showed that the pollutants of interest present at non-municipal facilities were present at concentrations similar to, or less than, the concentrations typically found at municipal facilities. Therefore, EPA determined that a treatment system that can adequately control pollutant discharges from a municipal solid waste landfill should also be able to control discharges at Subtitle D non-municipal landfills. EPA discusses its reasons for establishing only one subcategory for non-hazardous landfills in Chapter 5 and discusses alternative technology options and costs of these options in Chapters 8 and 9, respectively.

In addition to the 45 non-hazardous municipal solid waste facilities identified as potential BPT, EPA also evaluated one hazardous facility (16041) in the EPA database. This facility used biological treatment in the form of a sequential batch reactor (SBR) to treat its landfill generated wastewater. The facility commingled leachate from both non-hazardous and hazardous landfills prior to treatment by the SBR. In determining whether it was reasonable to include a facility from the Hazardous subcategory as a potential BPT facility in the Non-Hazardous subcategory, EPA evaluated two different factors. First, because the facility accepted leachate from both hazardous and non-hazardous landfills, EPA sampling data showed that the waste stream contained almost all of the pollutants of interest for the Non-Hazardous subcategory at similar concentrations to those found in the non-hazardous landfill raw wastewater database (see Table 11-3). At this facility, EPA sampling detected all but one of the 32 pollutants of interest for the Non-Hazardous subcategory in the influent concentration (1,4-dioxane) and EPA did not include four others (barium, disulfoton, hexavalent chromium, and n,n-dimethylformamide) in the analytical effort. Therefore, the Agency determined that the raw wastewater concentrations for the non-hazardous pollutants of interest from this hazardous facility were similar to those concentrations found at the non-hazardous facilities. Second, the facility achieved good BOD_5 removal using biological treatment equivalent to BPT Option I.

Therefore, EPA concluded that a treatment system that can adequately control pollutant discharges from a hazardous landfill should also be able to control discharges at non-hazardous landfills.

Based on the assessment above, there were 46 in-scope landfill facilities in the EPA database that employed various forms of biological treatment considered for BPT for the Non-Hazardous subcategory. EPA evaluated these 46 landfill facilities selected as potential BPT candidates to determine the performance across the various types of biological treatment systems. To determine the best performers for biological treatment EPA established a number of criteria. The first criterion used in the selection of the best facilities was effective treatment of BOD_5. Under this criterion, there were several reasons why a facility might be eliminated from the selection of BPT facilities.

Of the 46 landfill facilities treating their wastewater with some form of biological treatment, only 26 facilities provided BOD_5 effluent data in their Detailed Questionnaire or Detailed Monitoring Questionnaire submitted to EPA or in the data that EPA collected during a sampling episode performed at the facility. EPA evaluated these data to assess the performance across the various biological systems. Two facilities, 16119 and 16123, provided carbonaceous BOD (CBOD) data rather than BOD_5 data and, therefore, EPA removed these facilities from further consideration. EPA eliminated the facilities reporting CBOD data because the analytical results of the CBOD tests can differ from the BOD_5 results, especially in cases where ammonia is present in the wastewater. Table 11-4 lists the 20 facilities that EPA eliminated from further consideration as BPT facilities since they did not supply BOD_5 effluent data. Table 11-5 lists the treatment in place at the 26 candidate BPT facilities in the Non-Hazardous subcategory that provided BOD_5 effluent data. Table 11-6 shows, for the 26 candidate BPT facilities, the baseline flow, the facility-average raw wastewater BOD_5 concentration, the facility-average effluent BOD_5 concentration, the influent and effluent BOD_5 concentrations from Section C of the Detailed Questionnaire (DET) data, Detailed Monitoring Questionnaire (DMQ) data, and EPA sampling episodes (ANL) data, and the reason (if any) why EPA eliminated the facility as a BPT facility. EPA determined the average raw wastewater BOD_5 concentration and average effluent BOD_5 concentration at a facility by calculating the flow-weighted average of the facility

data available in Section C of the Detailed Questionnaire, the Detailed Monitoring Questionnaire, and the data collected during the EPA sampling episode.

Because EPA based BPT limitations on the effectiveness of biological treatment, the Agency eliminated facilities that used additional forms of treatment for BOD_5 (other than biological treatment). EPA, therefore, removed two sites (16099, 16125) using carbon treatment in addition to biological treatment from the list of candidate BPT facilities. EPA eliminated another facility from consideration (16117) because it used two separate treatment trains in treating its wastewater, one with biological treatment and the other with chemical precipitation, before commingling the streams at the effluent sample point. After the elimination of these three facilities, 23 potential BPT facilities remained in the EPA non-hazardous landfill database.

To ensure that the facilities were operating effective biological treatment systems, EPA evaluated the influent concentrations of BOD_5 entering the wastewater treatment systems to determine which facilities had influent BOD_5 concentrations that most closely resembled typical non-hazardous landfills. The median concentration of BOD_5 for non-hazardous landfills was 240 mg/L and the average concentration was 1,229 mg/L. EPA determined that facilities with BOD_5 influent concentrations significantly lower than these values would not be representative of typical wastewater concentrations found in the Non-Hazardous subcategory. Therefore, EPA eliminated facilities where the influent BOD_5 was below 100 mg/L. EPA acknowledges that it is possible to operate a biological treatment system with influent BOD_5 concentrations lower than 100 mg/L. In fact, as can be seen in Table 11-6, four of the remaining candidate BPT facilities had influent BOD_5 concentrations much less than 100 mg/L (16077, 16093, 16097, and 16170) and operated biological treatment systems. Three of these four (16077, 16093, 16097) achieved BOD_5 effluent concentrations below the BPT effluent limit despite low influent BOD_5 concentrations. However, EPA did receive a significant number of comments on the proposal stating that the biological treatment option selected as BPT was infeasible for treatment of particular types of landfill leachate (ash monofill wastewater in particular) due to its low organic content. The BOD_5 raw wastewater data submitted by some of these commenters was below 10 mg/L. The Agency acknowledges that in many of these cases

(such as where BOD_5 is less than 10 mg/L), the concentration of organic material in the raw wastewater is too low to support biological treatment. Because the guidelines do not require the installation of any particular technology to meet the limitations, facilities remain free to use whatever technology they choose as long as these technologies can meet the limitations. In response to comments concerning the feasibility of biological treatment for certain types of monofills with very low BOD_5 in their raw leachate, the Agency developed costs for low BOD_5 facilities in the database for alternative, non-biological treatment such as breakpoint chlorination, granular activated carbon, and iron co-precipitation. These alternate forms of non-biological treatment are discussed in Chapter 8 and their associated costs presented in Chapter 9. EPA's decision not to further subcategorize the Non-Hazardous landfill subcategory is discussed in Chapter 5. Therefore, as a result of the influent BOD_5 greater than 100 mg/L edit, EPA did not consider four facilities (16077, 16093, 16097, and 16170) for BPT.

EPA eliminated eight other facilities (16048, 16049, 16052, 16065, 16161, 16164, 16171, and 16176) from BPT consideration because they did not supply BOD_5 influent data (from any data source). EPA did not select two facilities (16127 and 16129) because their raw wastewater streams consisted primarily of non-contaminated storm water or contaminated ground water, which are flows that this regulation does not cover. As discussed in Chapter 6, the Agency did not use monitoring data to characterize landfill generated wastewater from facilities where out-of-scope wastewater contributed greater than 15 percent of the total wastewater flow. Facility 16129 treated a combined raw wastewater influent stream consisting of 92 percent ground water and 7 percent leachate, and facility 16127 treated a combined raw wastewater influent stream consisting of 70 percent storm water and 30 percent leachate. After elimination of these facilities, a total of 9 candidate BPT facilities remained.

The final requirement for BPT selection in the Non-Hazardous landfill subcategory was that the biological treatment system at the facility had to achieve a BOD_5 effluent concentration less than 50 mg/L. EPA determined that facilities not able to maintain an effluent concentration below 50 mg/L were not operating their biological system effectively. Two of the remaining 9 facilities (16088 and 16165) did not achieve

BOD_5 effluent concentrations of less than 50 mg/L, leaving seven facilities in the database. The site-identification numbers for the seven facilities selected as BPT are 16041, 16058, 16118, 16120, 16122, 16132, and 16253.

The seven facilities that met all of the BPT criteria employed various types of biological treatment systems, including activated sludge, a sequential batch reactor, aerobic and anaerobic biological towers or fixed film, and aerated ponds or lagoons. Most of the facilities employed equalization tanks in addition to the biological treatment, while several facilities also employed chemical precipitation and neutralization in their treatment systems. Clarification or sedimentation stages followed the biological treatment systems. Table 11-7 shows the treatment technologies in-place at the facilities selected as BPT for the Non-Hazardous subcategory. EPA used all seven facilities employing well-operated biological treatment systems to calculate the effluent limitations for BOD_5. The average influent BOD_5 concentrations to these seven treatment systems ranged from 150 mg/L to 7,600 mg/L and, as mentioned above, all of the average effluent concentrations for these seven facilities were below 50 mg/L.

While the BOD_5 edits discussed above ensure good biological treatment and a basic level of TSS removal, treatment facilities meeting this level may not necessarily be operated for optimal control of TSS. To ensure that the effluent limitation developed for TSS reflects proper control, EPA established additional editing criteria for TSS.

EPA developed two criteria for editing TSS performance data. In addition to achieving the BOD_5 criteria cited above, EPA required that the facility employ technology sufficient to ensure adequate control of TSS, that is, a sand or multimedia filtration system. Three of the seven well-operated biological systems (16120, 16122, 16253) used sand or multimedia filters as a polishing step for additional control of suspended solids prior to discharge.

The second factor EPA considered was whether the treatment system achieved an effluent TSS concentration less than or equal to 100 mg/L. EPA selected treatment facilities meeting these criteria as the average of the best existing performers for TSS. Table 11-8 lists the baseline flow, the facility-average raw wastewater TSS concentration, the facility-average effluent TSS concentration, the influent and effluent TSS concentrations from Section C of the Detailed Questionnaire (DET) data, the Detailed Monitoring Questionnaire (DMQ) data, and the EPA sampling episode (ANL) data for the seven facilities selected as BPT in the Non-Hazardous subcategory. EPA determined the average raw wastewater TSS concentration and average effluent TSS concentration at a facility by calculating the flow-weighted average of the facility data available in Section C of the Detailed Questionnaire, the Detailed Monitoring Questionnaire, and the data collected during the EPA sampling episode. All three facilities that employed a sand or multimedia filtration system (16120, 16122, and 16253) achieved an effluent TSS concentration far less than 100 mg/L, and therefore EPA included these among the best existing performers for TSS. Although facility 16122 meets the TSS editing criteria, EPA eliminated it from further consideration as BPT for TSS because of potential settling of TSS in aerated tanks immediately prior to the filters that are not part of the selected BPT option. Therefore, EPA selected the remaining two facilities (16120 and 16253) as "average of the best" existing performers for TSS and based the TSS limitations on these two facilities.

EPA determined that the use of a multimedia filter after biological treatment with secondary clarification achieved significantly lower long-term average effluent concentrations of TSS than the other BPT facilities that did not employ multimedia filters after secondary clarification. As shown in Table 11-8, the two facilities (16120 and 16253) that employed multimedia filters after biological treatment with clarification achieved an average effluent TSS concentration of 19.5 mg/L whereas the other BOD_5 BPT facilities without multimedia filters achieved an average effluent concentration of 69.1 mg/L.

Development of BPT Limitations

EPA based the effluent limitations for BOD_5 on all seven non-hazardous BPT facilities; however, the BPT facilities often did not supply data for all of the regulated pollutants. Therefore, EPA used the data available

from the seven non-hazardous BPT facilities to develop the BPT limitations for ammonia, TSS, alpha terpineol, benzoic acid, p-cresol, phenol, and zinc. EPA applied additional editing criteria to the seven BPT facilities to select the "average of the best" existing performers for each of the regulated pollutants. The editing criteria applied to the available data were as follows:

- EPA only used data from the seven facilities which passed the BOD_5 criteria in the calculation of limits (16041, 16058, 16118, 16120, 16122, 16132, and 16253).

- EPA only used data from facilities that passed the TSS criteria in the calculation of TSS limits (16120 and 16253).

C EPA did not use effluent data from the Detailed Questionnaire (16000 series data) in the calculation of effluent limits. The pollutant data submitted in the Detailed Questionnaire contained the average concentration, the minimum and maximum concentrations, and the number of samples, whereas EPA sampling data and the Detailed Monitoring Questionnaire consisted of individual daily data. In developing limits, EPA calculated the long-term averages and variability factors using individual daily data. Furthermore, summary data (like the data submitted in the Detailed Questionnaire) may obscure the minimum detection levels used in the sampling data. The use of daily data (like the Detailed Monitoring Questionnaire and EPA sampling data) in developing limitations allows EPA to account for concentration values reported at or below the detection limits. In addition, in many cases, EPA considered reported averages from Detailed Questionnaires redundant because many facilities also reported the daily data from the Detailed Monitoring Questionnaire for the same time period in 1992 and, therefore, EPA would not have used the data in the calculation of limits. However, EPA did use, in cases where no other influent data were available, influent data from the Detailed Questionnaire to show that a landfill had treatable levels of a pollutant in the wastewater.

- Since chemical precipitation was not part of the selected BPT Option for the Non-Hazardous subcategory, EPA did not use data from BPT facilities employing chemical precipitation when developing limitations for metals. Therefore, since zinc was the only metal regulated, EPA did not include zinc effluent data from four of the seven facilities that employed chemical precipitation in the calculation of zinc limitations (16118, 16120, 16122, and 16253). In the Non-Hazardous subcategory, EPA determined that the levels of zinc found in raw wastewater were at low enough concentrations that chemical precipitation was not a necessary treatment technology. In the Non-Hazardous landfill subcategory, EPA's sampling, for the most part, did not find zinc raw wastewater concentrations that would inhibit biological treatment. In addition, raw wastewater concentrations of zinc were typically less than 1 mg/L, a level that would not be effectively removed by a chemical precipitation system.

C EPA did not use facility data demonstrating zero or negative percent removals in the calculation of limits. No facility data in the Non-Hazardous subcategory met this criterion.

- EPA did not include data from facility 16120 in the calculation of ammonia limitations because the treatment system included air stripping.

- EPA only used effluent data if sufficient influent data were available to establish the presence of treatable levels of pollutants. The Agency only used effluent data in calculating limits if influent data for a given pollutant were available for a facility. In cases where a facility supplied effluent data for a particular pollutant but did not supply influent data in the Detailed Monitoring Questionnaire (or supplied influent data below a treatable level), EPA used the effluent data so long as influent data were available from the EPA sampling episode or the Detailed Questionnaire at a concentration above a treatable level. However, EPA did not use effluent data from EPA sampling episodes to calculate limits unless matching influent data from the EPA sampling episode were at concentrations above treatable levels.

- For the EPA sampling episode at facility 16122, EPA did not use the effluent data collected from sample point 08 in the calculation of the limits because this sample point was located after two aerated holding tanks operated in parallel just prior to the multimedia filter (which was not part of the selected treatment option after biological treatment). Instead, EPA used data from sample point 07 (after biological treatment but before aeration in the holding tanks) in the calculation of limits for the final rule. In addition, EPA did not use effluent data from the Detailed Questionnaire and Detailed Monitoring Questionnaire from facility 16122 in the calculation of limits because the data were from sample point 03, which is located after the aeration tanks.

In Table 11-9, EPA presents the non-hazardous BPT facilities and sample points used to calculate the non-hazardous BPT limitations for conventional, nonconventional, and toxic pollutants. Table 11-10 presents the non-hazardous BPT facilities and sample points that EPA did not use to calculate the BPT limitations and the reason for their exclusion. Table 11-11 presents EPA's final BPT limitations for the Non-Hazardous subcategory.

Tables 11-12 and 11-13 present the national estimates of the pollutant of interest reductions for the BPT/BAT options for the municipal solid waste Subtitle D landfills and non-municipal Subtitle D landfills, respectively. Table 11-14 and Table 11-15 summarize the estimated amount of pollutants discharged

annually from direct discharging municipal landfills and direct-discharging non-municipal landfills, respectively, before and after the implementation of BAT for the Non-Hazardous subcategory.

EPA based all of the estimated costs on a facility installing aerated equalization tanks followed by an activated sludge biological system with clarification and a multimedia filter and included a sludge dewatering system. On a national scale, EPA estimates that the implementation of the BAT effluent limitations will require a capital cost of $18.87 million and annual operating cost of $6.50 million resulting in a total annualized cost of $7.64 million (post-tax) for the Subtitle D Non-Hazardous subcategory (1998 dollars).

11.2.3 BPT Technology Options for the Subtitle C Hazardous Subcategory

EPA's survey of the hazardous landfills industry identified no in-scope respondents that were classified as direct dischargers. All of the hazardous landfills within the scope of the rule are either indirect or zero/alternative dischargers. Consequently, EPA could not evaluate any treatment systems in-place at direct-discharging hazardous landfills for establishing BPT effluent limitations. Therefore, EPA relied on information and data from widely available treatment technologies in use at hazardous landfill facilities discharging indirectly and at non-hazardous landfills discharging directly and indirectly, termed "technology transfer." EPA concluded that the technology in-place at some indirect hazardous landfills is appropriate to use as the basis for regulation of direct dischargers because the wastewater generated at hazardous waste landfills discharging directly would be similar in character to the wastewater from indirect-discharge hazardous waste landfills.

Based on this assessment, EPA developed the following BPT regulatory options for establishing BPT effluent limitations for the Hazardous landfill subcategory: 1) aerated equalization followed by chemical precipitation with clarification and multimedia filtration, 2) aerated equalization followed by chemical precipitation with clarification, biological treatment with secondary clarification, and multimedia filtration, and 3) zero or alternative discharge. Chapter 8 discusses these options in detail and Chapter 9 discusses the cost estimates developed for these options.

Hazardous Subcategory Option I: Chemical Precipitation and Multimedia Filtration

EPA first assessed the pollutant removal performance of equalization, chemical precipitation, and multimedia filtration. EPA evaluated chemical precipitation as a treatment technology because of the metals concentrations typically found in hazardous landfill leachate and the efficient metals removals achieved through chemical precipitation. EPA also evaluated multimedia filtration as an appropriate technology to remove additional levels of metals and TSS following chemical precipitation.

Hazardous Subcategory Option II: Chemical Precipitation, Biological Treatment, and Multimedia Filtration

The second technology option considered for BPT treatment of hazardous landfill wastewater was aerated equalization, chemical precipitation, and biological treatment with secondary clarification, followed by multimedia filtration. EPA evaluated these technologies as Option II because of the effectiveness of chemical precipitation in removing metals and the effectiveness of biological treatment in removing the high organic loads present in the leachate. The Agency considered multimedia filtration to be an appropriate technology for consideration because of its effectiveness in removing TSS and metals remaining after primary or secondary clarification.

Hazardous Subcategory Option III: Zero or Alternative Discharge

Finally, EPA considered a zero or alternative discharge option as BPT Option III because a significant segment of the industry is currently not discharging wastewater to surface waters or to POTWs. The zero or alternative disposal option would require facilities to dispose of their wastewater in a manner that would not result in wastewater discharge to a surface water or a POTW.

Methods of achieving zero or alternative discharge currently in use by hazardous landfills are deep well injection, solidification, and contract hauling of wastewater to a Centralized Waste Treatment (CWT) facility or to a landfill wastewater treatment facility. Thirty seven facilities are estimated to inject landfill

wastewater underground on site, 103 facilities send their wastewater to a CWT or landfill treatment system, and one facility solidifies wastewater.

Selected BPT Technology Option

EPA selected Option II, aerated equalization and chemical precipitation followed by biological treatment with secondary clarification and multimedia filtration, as the technology basis for BPT limitations for the Hazardous landfills subcategory. EPA selected Option II because of the demonstrated ability of biological treatment and multimedia filtration in removing the large organic loads and suspended solids associated with hazardous leachate. Metals in the raw wastewater will be removed prior to the biological treatment system using chemical precipitation. Figure 11-3 presents a flow diagram of the treatment system for this option.

EPA eliminated Option I from consideration because it did not control organic pollutants effectively. In addition, based on consideration of comments submitted on the proposal, EPA decided not to establish BPT limitations based on zero or alternative discharge. EPA concluded that, for the industry as a whole, zero or alternative discharge options are either not viable or the cost is wholly disproportionate to the pollutant reduction benefits and, thus, not "practicable." Furthermore, the commenters' submissions support EPA's decision to reject zero or alternative discharge as the technology basis for BPT (or BAT) limitations for hazardous landfills. While EPA supports the use of zero or alternative discharges particularly where it does not result in media transfer of pollutants, many of the available zero discharge options have identifiable shortcomings, such as transfer of waste residuals to another media (e.g., ground water, soil) or the availability of an alternative disposal option only in certain geographic locations.

For example, one demonstrated alternative disposal option for large wastewater flows is underground injection. However, this is not considered a practically available option on a nationwide basis because it is not allowed in many geographic regions of the country where landfills may be located. These restrictions may preclude underground injection at a given landfill. In such circumstances, landfills would need to resort

to contract hauling to a CWT facility. Unless the CWT itself were a zero discharge facility, the ultimate result would be treatment and discharge to surface waters or a POTW following waste treatment that may be no more effective than that provided on site. This might result in substantial transportation costs for the landfill and associated non-water quality environmental impacts (e.g., truck emissions) resulting in no net reduction in the discharge of pollutants. EPA's survey demonstrated that only landfills with relatively low flows (under 500 gpd) currently contract haul their wastewater to a CWT. The costs of contract hauling are directly proportional to the volume and distance over which the wastewater must be transported, generally making it excessively costly to send large wastewater flows to a CWT, particularly if it is not located nearby.

EPA evaluated the cost of requiring all hazardous landfills to achieve zero or alternative discharge status. For the purposes of costing, EPA assumed that a facility would have to contract haul wastewater off site because it may be impossible to pursue other zero or alternative discharge options. EPA concluded that the cost of contract hauling off site for high flow facilities was unreasonably high and disproportionate to the removals potentially achieved. In addition, EPA concluded that the wastewater shipped to a CWT will typically receive treatment equivalent to that promulgated, and that zero/alternative discharge requirements would result in additional costs to discharge without greater removals for hazardous landfill wastewater. To calculate costs for this option, EPA estimated that all facilities currently discharging to a POTW would have to contract haul wastewater approximately 500 miles to a CWT facility. EPA based cost estimates on a $0.35 per gallon disposal cost at a CWT facility, and $3.00 per loaded mile for transport. EPA estimated the total cost to the industry at approximately $30 million dollars.

11.2.4 BPT Limits for the Subtitle C Hazardous Subcategory

Selection of BPT Facilities

EPA based the BPT effluent limitations for the Hazardous subcategory upon the average of the best existing landfill facilities. Based on the characteristics of hazardous landfill leachate and on an evaluation of appropriate technology options, the Agency selected aerated equalization, chemical precipitation, and

biological treatment followed by secondary clarification and multimedia filtration as BPT technology for the Hazardous subcategory. As previously noted, EPA relied on data from both hazardous and non-hazardous facilities to develop the limitations for this subcategory. Because there are currently no hazardous landfills discharging directly, EPA used data from indirectly discharging facilities to develop the limitations.

Apart from the 139 hazardous, zero, or alternative discharge facilities estimated to be in the U.S. based on the responses to the Detailed Questionnaire, EPA identified only three other hazardous respondents (16017, 16041, and 16087) to the Detailed Questionnaire, all of which discharged indirectly to POTWs. Facility 16017 only collected and treated landfill gas collection condensate which was very dilute, had low flows, and required only minimal treatment (neutralization using ammonia) prior to discharge. Consequently, EPA did not consider this facility as appropriate for establishing BPT limitations. The two remaining facilities (16041 and 16087) both had treatment systems in-place that achieved very good pollutant reductions. The treatment at facility 16087 consisted of equalization and a chemical precipitation unit followed by an activated sludge system with secondary clarification; the other facility (16041) utilized equalization tanks and a sequential batch reactor. The treatment systems in-place at these indirect-discharging hazardous facilities achieved low effluent concentrations with average removals of 88 to 98 percent of organic toxic pollutants, and 55 to 80 percent of metal pollutants. Thus, EPA concluded that both facilities should be used in the development of the Hazardous subcategory BPT limitations for nonconventional and toxic pollutants. Table 11-16 presents the treatment technologies in-place at the facilities selected as BPT for the Hazardous subcategory.

Development of BPT Effluent Limitations

As discussed above, because there were no direct-discharging hazardous facilities in EPA's database, the Agency relied on technology transfer to establish BPT effluent limitations, using performance data from treatment technologies at hazardous landfill facilities discharging indirectly and non-hazardous facilities discharging directly and indirectly. EPA used the data from the two hazardous indirect-discharging facilities (16041 and 16087) to calculate the BPT effluent limitations for the following toxic pollutant parameters:

alpha terpineol, aniline, arsenic (total), benzoic acid, chromium (total), naphthalene, p-cresol, phenol, pyridine, and zinc (total). Chapter 7 discusses the methodology used to select toxic pollutants for regulation.

EPA concluded that establishing BPT effluent limitations for ammonia, BOD_5, and TSS based only on performance data from these two hazardous indirect-discharging facilities was not appropriate. In general, removal of classical pollutant parameters such as ammonia, BOD_5, and TSS in treatment systems at indirect-discharging facilities is incidental to toxic pollutant removals, since these pollutants are a major component of domestic sewage and are adequately treated at POTWs. Since removals of ammonia, BOD_5, and TSS at these two hazardous indirect-discharging facilities ranged from poor to adequate, EPA concluded that the use of performance data from BPT facilities in both the Hazardous and Non-Hazardous subcategories that employed variations of biological treatment would result in more representative hazardous BPT effluent limitations for these pollutants.

EPA supplemented the Hazardous subcategory data for these three pollutants with data from non-hazardous landfill facilities. For calculation of BPT effluent limitations for BOD_5, EPA supplemented the performance data from the two hazardous indirect-discharging facilities (16041 and 16087), with performance data from direct- and indirect-discharging non-hazardous facilities (16058, 16118, 16120, 16122, 16132 and 16253) to obtain a more representative mix of facilities. For calculation of BPT effluent limitations for TSS, because neither of the treatment systems for the two hazardous indirect-discharging facilities included multimedia filtration to control TSS discharges, EPA used technology transfer to establish TSS limitations, using performance data from two non-hazardous facilities (16120 and 16253) that passed the TSS effluent editing criteria for the BPT effluent limitations for Non-Hazardous subcategory.

For calculation of BPT effluent limitations for ammonia, since the treatment system for only one of the two hazardous indirect-discharging facilities was considered a good performer (16041), EPA supplemented

these data with performance data from two non-hazardous BPT facilities (16122 and 16132) that were considered good performers in the Non-Hazardous subcategory.

In addition, EPA applied editing criteria to the data to determine the final list of BPT facilities and sample points used to develop the BPT limits for the Hazardous subcategory. The editing criteria applied to the available data were as follows:

- EPA only used data from the two hazardous facilities selected as BPT (16041 and 16087) in the calculation of limits for toxic pollutants (except ammonia).

C EPA used technology transfer from the Non-Hazardous subcategory in establishing limits for BOD_5, TSS, and ammonia.

- EPA only used data from facilities that passed the TSS criteria in the calculation of TSS limits (16120 and 16253).

C EPA did not use effluent data from the Detailed Questionnaire (16000 series data) in the calculation of effluent limits. The pollutant data submitted in the Detailed Questionnaire contained the average concentration, the minimum and maximum concentrations, and the number of samples, whereas EPA sampling data and the Detailed Monitoring Questionnaire consisted of individual daily data. In developing limits, EPA calculated the long-term averages and variability factors using individual daily data. Furthermore, summary data (like the data submitted in the Detailed Questionnaire) may obscure the minimum detection levels used in the sampling data. The use of daily data (like the Detailed Monitoring Questionnaire and EPA sampling data) in developing limitations allows EPA to account for concentration values reported at or below the detection limits. In addition, in many cases, EPA considered reported averages from Detailed Questionnaires redundant because many facilities also reported the daily data from the Detailed Monitoring Questionnaire for the same time period in 1992 and, therefore, EPA would not have used the data in the calculation of limits. However, EPA did use, in cases where no other influent data were available, influent data from the Detailed Questionnaire to show that a landfill had treatable levels of a pollutant in the wastewater.

C EPA did not use facility data demonstrating zero or negative percent removals in the calculation of limits.

- EPA did not include data from facility 16120 in the calculation of ammonia limitations because the treatment system included air stripping.

- EPA only used effluent data if sufficient influent data were available to establish the presence of treatable levels of pollutants. The Agency only used effluent data in calculating limits if influent data for a given pollutant were available for a facility. In cases where a facility supplied effluent data for a particular pollutant but did not supply influent data in the Detailed Monitoring Questionnaire (or supplied influent data below a treatable level), EPA used the effluent data so long as influent data were available from the EPA sampling episode or the Detailed Questionnaire at a concentration above a treatable level. However, EPA did not use effluent data from EPA sampling episodes to calculate limits unless matching influent data from the EPA sampling episode are at concentrations above treatable levels.

Table 11-17 presents the hazardous BPT facilities and sample points used to calculate the hazardous BPT limitations for conventional, nonconventional, and toxic pollutants. Table 11-18 presents the hazardous BPT facilities and sample points EPA did not use to calculate the BPT limitations and the reason for their exclusion. In Table 11-19, EPA presents the final BPT limitations for the Hazardous subcategory.

Since there are no direct discharging hazardous landfills in the EPA database, EPA could not estimate pollutant reductions as a result of the regulation and the average facility costs for implementation of the regulation.

11.3 Best Conventional Pollutant Control Technology (BCT)

BCT limitations control the discharge of conventional pollutants from direct dischargers. Conventional pollutants include BOD, TSS, oil and grease, and pH. BCT is not an additional limitation, but rather replaces BAT for the control of conventional pollutants. To develop BCT limitations, EPA conducts a cost-reasonableness evaluation, which consists of a two-part cost test: 1) the POTW test and 2) the industry cost-effectiveness test.

In the POTW test, EPA calculates the cost per pound of conventional pollutants removed by industrial dischargers in upgrading from BPT to a BCT candidate technology and then compares this to the cost per pound of conventional pollutants removed in upgrading POTWs from secondary to tertiary treatment. The upgrade cost to industry, which is represented in dollars per pound of conventional pollutants removed,

must be less than the POTW benchmark of $0.25 per pound (in 1976 dollars). In the industry cost-effectiveness test, the ratio of the incremental BPT to BCT cost, divided by the BPT cost for the industry, must be less than 1.29 (i.e. the cost increase must be less than 29 percent).

For the final rule, EPA established effluent limitations guidelines and standards equivalent to the BPT guidelines for the conventional pollutants covered under BPT for both subcategories. In developing BCT limits, EPA considered whether there are technologies that achieve greater removals of conventional pollutants than for BPT and whether those technologies are cost-reasonable according to the BCT cost-reasonableness evaluation. In each subcategory, EPA identified no technologies that can achieve greater removals of conventional pollutants than those promulgated for BPT that are also cost-reasonable under the BCT cost-reasonableness evaluation, and, accordingly, EPA established BCT effluent limitations equal to the BPT effluent limitations guidelines and standards.

11.4 Best Available Technology Economically Achievable (BAT)

The factors considered in establishing a BAT level of control include the following: the age of process equipment and facilities, the processes employed, process changes, the engineering aspects of applying various types of control techniques to the costs of applying the control technology, non-water quality environmental impacts such as energy requirements, air pollution and solid waste generation, and such other factors as the Administrator deems appropriate (Section 304(b)(2)(B) of the Act). In general, the BAT technology level represents the best existing economically achievable performance among facilities with shared characteristics. BAT may include process changes or internal plant controls which are not common in the industry. BAT may also be transferred from a different subcategory or industrial category.

EPA promulgated BAT effluent limitations for both landfill subcategories based upon the same technologies evaluated and selected for BPT. The BAT effluent limitations control identified toxic and nonconventional pollutants discharged from facilities. EPA did not identify any additional technologies beyond BPT that could provide additional toxic pollutant removals and that are economically achievable.

11.4.1 BAT Limits for the Subtitle D Non-Hazardous Subcategory

EPA evaluated reverse osmosis technology as a potential option for establishing BAT effluent limits more stringent than BPT for the control of toxic pollutants for the Non-Hazardous subcategory. EPA considered reverse osmosis for evaluation because of its effective control of a wide variety of toxic pollutants in addition to controlling conventional and nonconventional parameters.

EPA evaluated BAT treatment options as an increment to the baseline treatment technology used to develop BPT limits. Therefore, the BAT Option III consisted of BPT Option II (biological treatment followed by multimedia filtration) followed by a single-stage reverse osmosis unit. Figure 11-4 presents a flow diagram of the treatment system costed for BAT Option III. EPA acknowledges that reverse osmosis treatment of landfill wastewater does not require biological pretreatment. However, in evaluating potential BAT options, EPA considers the removal and costs of BAT in addition to the selected BPT option. Therefore, to analyze the incremental removals and incremental costs, EPA evaluated the reverse osmosis system after the selected BPT option (biological treatment and multimedia filtration).

EPA promulgated limits based on a BAT technology that is equivalent to the BPT technology. After an assessment of costs and pollutant reductions associated with reverse osmosis, EPA concluded that limits should not be established based on more advanced treatment technology than the BPT technology. EPA concluded that a biological system followed by multimedia filtration would remove the majority of toxic pollutants, leaving the single-stage reverse osmosis to treat the very low levels of pollutants that remained. In the Agency's analysis, BPT Option II removed 170,000 pounds of toxic pollutants per year, whereas BAT Option III removed 172,000 pounds of toxic pollutants per year. The small incremental removal of pounds of toxic pollutants achieved by BAT Option III was not justified by the large cost for the reverse osmosis treatment system. According to EPA's costing analysis, the BAT Option III, consisting of BPT Option II plus reverse osmosis, was estimated to cost the Landfills industry $130.3 million in capital costs (1998 dollars) and $45.95 million in annualized costs (pre-tax, 1998 dollars). By contrast, the selected option, BPT Option II, had capital costs of $18.87 million (1998 dollars) and annualized costs of $7.64

million (post-tax, 1998 dollars). It should be noted that reverse osmosis was much more effective than the BPT Option II at removing the often-high quantities of dissolved metals such as iron, manganese, and aluminum. However, these parameters were not included in the calculation of pound-equivalent reductions due to their use as treatment chemicals.

Table 11-20 compares the long-term averages achieved by BPT Option II, consisting of equalization, biological treatment, and multimedia filtration, to the long-term averages achieved by the reverse osmosis treatment system. For the long-term average comparison, the effluent concentrations are from the reverse osmosis treatment system sampled by EPA and described in Section 8.2.1.5, including the flow diagram in Figure 8-30. Table 11-20 shows BPT Option II achieves very low effluent concentrations that are similar to the effluent concentrations achieved by the reverse osmosis system.

Several commenters on the proposal supported EPA's decision to reject reverse osmosis as the selected technology option. While EPA rejected reverse osmosis as the basis for BAT limitations because it was very expensive and achieved very little additional removal of pollutant, other technical factors also supported this decision. EPA agrees with the commenters that there may be additional site-specific costs associated with the operation of reverse osmosis systems at landfills that it could not directly factor into its cost analysis. EPA found that it was difficult to evaluate potential operating and concentrate-disposal problems and the associated potential increase in the cost of operating a reverse osmosis system at a landfill. The fact that reverse osmosis is a technology that concentrates rather than destroys pollutants is an important consideration. These concentrates still need to be treated and disposed and, as noted by one commenter, some states may not allow them to be recycled back into the landfill. Further, recirculation may inhibit rather than stimulate anaerobic decomposition of the landfilled wastes. While the sludges generated by chemical precipitation and biological treatment require minimal treatment prior to disposal, reverse osmosis concentrates may require additional costly treatment steps prior to final disposal.

11.4.2 BAT Limits for the Subtitle C Hazardous Subcategory

As stated in the BPT analysis, EPA's survey of the hazardous landfills industry identified no in-scope respondents which were classified as direct dischargers. All of the hazardous landfills in the EPA survey were indirect or zero or alternative dischargers. Therefore, the Agency based BPT limitations on technology transfer and treatment systems in place for indirect dischargers in the Hazardous subcategory and on treatment systems in-place for BPT facilities in the Non-Hazardous subcategory. In EPA's engineering assessment of the possible BAT technology for direct-discharging hazardous facilities, EPA evaluated the same three potential technology options as those evaluated for BPT for the Hazardous landfill subcategory. These technology options were 1) aerated equalization followed by chemical precipitation with clarification and multimedia filtration, 2) aerated equalization followed by chemical precipitation with clarification, biological treatment with secondary clarification, and multimedia filtration, and 3) zero or alternative discharge, as explained above. EPA has identified no other technologies that would represent BAT level of control for this industry.

EPA determined that BAT limits should be established based on the same technology evaluated for BPT limits. As explained above at Section 11.2.3, zero or alternative discharge is not an available alternative treatment technology for this industry. Therefore, EPA promulgated BAT effluent limitations for the Hazardous landfill subcategory based upon the same treatment technology selected for BPT: equalization prior to chemical precipitation with clarification, followed by biological treatment with secondary clarification, and multimedia filtration.

11.5 New Source Performance Standards (NSPS)

New Source Performance Standards under Section 306 of the Clean Water Act represent the greatest degree of effluent reduction achievable through the application of the best available demonstrated control technology for all pollutants (i.e. conventional, nonconventional, and toxic pollutants). NSPS are applicable to new industrial direct-discharging facilities, for which construction has commenced after the publication of final regulations. Congress envisioned that new treatment systems could meet tighter controls than

existing sources because of the opportunity to incorporate the most efficient processes and treatment systems into plant design. Therefore, Congress directed EPA, in establishing NSPS, to consider the best demonstrated process changes, in-plant controls, operating methods, and end-of-pipe treatment technologies that reduce pollution to the maximum extent feasible.

EPA established New Source Performance Standards (NSPS) that would control the same conventional, toxic, and nonconventional pollutants promulgated for control by the BAT effluent limitations for both subcategories. The treatment technologies used to control pollutants at existing facilities are fully applicable to new facilities. Furthermore, EPA has not identified any other technologies or combinations of technologies that are demonstrated for new sources that are different from those used to establish BPT/BCT/BAT for existing sources. Therefore, EPA established NSPS limitations that are identical to those promulgated in both subcategories for BPT/BCT/BAT.

In the proposed rule, EPA solicited comments and data on other technologies that may be appropriate for the treatment of landfill leachate from new sources. One commenter urged EPA to consider reverse osmosis as an appropriate technology for the treatment of leachate. While EPA acknowledges that reverse osmosis can treat landfill leachate to levels equivalent to, and even lower than, the final BAT limitations, EPA concluded that the reverse osmosis treatment system did not remove significantly more pounds of toxic pollutants than the treatment option selected as BAT. Therefore, EPA concluded that the large costs associated with the installation, operation, and maintenance of a reverse osmosis system would not justify the small incremental removal of pounds of toxic pollutants achieved. Therefore, EPA is promulgating NSPS limitations that are identical to those in each subcategory for BPT/BCT/BAT.

11.6 Pretreatment Standards for Existing Sources (PSES)

Section 307(b) of the Act requires EPA to promulgate pretreatment standards for pollutants that are not susceptible to treatment by POTWs or which would interfere with the operation of POTWs. After a thorough analysis of indirect-discharging landfills in the EPA database, EPA has decided not to establish

PSES for either subcategory in the Landfills Point Source Category. For the proposal, EPA proposed *not* to establish pretreatment standards for indirectly discharging landfills in the Non-Hazardous subcategory. However, for the Hazardous subcategory, EPA proposed effluent limitations and pretreatment standards for six pollutants. In response to its proposal, EPA received a number of comments supporting the decision not to propose pretreatment standards for Subtitle D landfills. In addition, a number of commenters suggested that EPA should also reconsider whether Subtitle C landfills require national categorical pretreatment standards. As a result of these comments, EPA took a second look at its data and determined that pretreatment standards were not necessary for the Landfills Point Source Category.

For both subcategories, EPA looked at a number of factors in deciding whether a pollutant was not susceptible to treatment at a POTW or would interfere with POTW operations – the predicate to establishment of pretreatment standards. First, EPA assessed the pollutant removals achieved at POTWs relative to those achieved by landfills using BAT treatment systems. Second, EPA estimated the quantity of pollutants likely to be discharged to receiving waters after POTW removals. Third, EPA studied whether any of the pollutants introduced to POTWs by landfills interfered with or were otherwise incompatible with POTW operations. EPA, in some cases, also looked at the costs and other economic impacts of pretreatment standards and the effluent reduction benefits in light of treatment systems currently in-place at POTWs. The result of EPA's evaluation showed that POTWs could adequately treat discharges of landfill pollutants. Therefore, EPA is not establishing pretreatment standards for either subcategory in this point source category.

As noted above, among the factors EPA considers before establishing pretreatment standards is whether the pollutants discharged by an industry pass through a POTW or interfere with the POTW operation or sludge disposal practices. One of the tools traditionally used by EPA in evaluating whether pollutants pass through a POTW, is a comparison of the percentage of a pollutant removed by POTWs with the percentage of the pollutant removed by discharging facilities applying BAT. In most cases, EPA has concluded that a pollutant passes through the POTW when the median percentage removed nationwide

by representative POTWs (those meeting secondary treatment requirements) is less than the median percentage removed by facilities complying with BAT effluent limitations guidelines for that pollutant. For a full explanation of how EPA performs its removal analysis, see Chapter 7.

In developing the final guidelines, EPA has made a number of modifications to its calculations of pollutant removal used to compare POTW operations with BAT treatment. For example, the primary source of POTW percent removal data used for removal comparisons is an EPA document, "Fate of Priority Pollutants in Publicly Owned Treatment Works" (EPA 440/1-82/303) commonly referred to as the "50-POTW Study". The 50-POTW Study presents data on 50 well-operated POTWs with secondary treatment in removing toxic pollutants. For its removal comparison for this guideline, EPA eliminated influent values that were close to the detection limit, thereby minimizing the possibility that low POTW removals might simply reflect low influent concentrations instead of being a true measure of treatment effectiveness.

After revising the database, EPA calculated POTW-specific percent removals for each pollutant based on its average influent and average effluent values. The POTW percent removal used for each pollutant for the comparison is the median value of all the POTW-specific percent removals for that pollutant. EPA then compared the median POTW percent removal to the median percent removal for the BAT option treatment technology in order to determine pass through.

11.6.1 EPA's Decision Not to Establish PSES for the Subtitle D Non-Hazardous Subcategory

EPA estimates that there are 756 Subtitle D landfill facilities in the U.S. that discharge landfill wastewater to a POTW. The Agency did not establish pretreatment standards for existing sources (PSES) for the Non-Hazardous landfill subcategory. The Agency decided not to establish PSES for this subcategory after an assessment of the effect of landfill leachate on receiving POTWs and the cost of pretreatment standards.

EPA looked at three measures of effects on POTWs: biological inhibition levels, contamination of POTW biosolids, and a comparison of BAT and POTW removals. For the proposed rule, following procedures outlined above, the removal comparison suggested that one pollutant, ammonia, would pass through in the Non-Hazardous subcategory. However, EPA concluded that ammonia was susceptible to treatment and did not interfere with POTW operations. Therefore, the Agency did not propose to establish national pretreatment standards for ammonia.

Following the proposal, EPA reviewed the data available in the proposed Public Record for both the POTW percent removal calculations and the BAT percent removal calculations and made a number of adjustments. For the proposal, EPA calculated the BAT percent removals using data from well-operated biological treatment facilities in EPA's database. However, some of these facilities did not pass the editing criteria for selection as a BPT/BAT facility. For the revised removal comparison, EPA calculated percent removals using data from only those seven facilities that passed the BPT/BAT editing criteria. In addition, in the proposal, EPA inadvertently neglected to use selected BAT facilities in the calculation of percent removals for several pollutants even though the data for the facility passed the editing criteria.

The result of this revised comparison of removal for the Non-Hazardous subcategory suggested that BAT removal would be greater than POTW removal for four pollutants*:* ammonia, benzoic acid, p-cresol, and phenol. However, as explained below, EPA concluded that these pollutants do not pass through or interfere with POTW operations on a national basis and therefore has not established national categorical pretreatment regulations for these pollutants. Moreover, as discussed later in this chapter, EPA notes that adoption of PSES would result in the removal of only a small quantity of pollutants, approximately 14 toxic pound equivalents per facility per year. Such a reduction is low relative to that seen in other categorical pretreatment standards promulgated by EPA. (See 64 FR 45077).

11.6.1.1 EPA's Rationale for Not Establishing PSES for Ammonia

EPA has decided not to establish ammonia pretreatment standards for several reasons. First, while EPA's

removal comparison suggests that ammonia in landfill leachate is not as amenable to POTW treatment as to pretreatment, in reality, EPA has concluded that ammonia is susceptible to POTW treatment on a national basis. Further, landfill discharges will not result in POTW upsets or interfere with POTW operations. The Public Record indicates that POTWs are not currently experiencing any difficulty in adequately treating ammonia discharges from Subtitle D landfills. No POTWs commenting on the proposal cited any persistent POTW upsets associated with landfill leachate discharges. Finally, EPA has determined that pretreatment standards for ammonia for landfill indirect dischargers would be extremely costly, given the high levels of removal currently observed. In these circumstances, EPA has concluded that ammonia is susceptible to treatment by POTWs and national pretreatment standards are not required.

Ammonia Removals

In the case of ammonia, the median BAT percent removal for the landfills industry is 99 percent compared to the median POTW percent removal which is 39 percent. (For the proposed rule, EPA calculated the POTW percent removal for ammonia to be 60 percent. However, upon applying the revised data editing procedures to the 50-POTW Study, EPA has now determined that ammonia POTW percent removal is 39 percent.) This comparison suggests that ammonia is not susceptible to treatment at a POTW and passes through. However, as discussed below, most subtitle D landfills discharging to POTWs are discharging small quantities of leachate with an ammonia concentration comparable to that observed in raw sewage.

EPA's data show that over 75 percent of indirectly discharging landfills discharge fewer than 10 pounds of ammonia per day at a concentration similar to that observed in raw sewage. Because many POTWs are designed and operated to treat ammonia (and other pollutants) in raw sewage, a POTW will adequately control landfill discharges of ammonia so long as the ammonia loadings to a POTW did not significantly differ from that typically observed. In those circumstances, ammonia will not pass through such POTWs.

Moreover, some POTWs have installed additional treatment to control ammonia. The data on POTW removal used for EPA's comparison does not reflect this fact. POTWs that have installed additional

ammonia treatment (or modified existing treatment) typically achieve removals in excess of 95 percent -- much higher than the 39 percent removal observed for the POTWs in the comparison analysis. Thus, ammonia does not pass through POTWs with nitrification even in cases where significant loadings of ammonia are discharged to a POTW.

In these circumstances, EPA has concluded ammonia at levels discharged by landfills is generally susceptible to POTW treatment. Therefore, EPA concluded that ammonia limits are best established by local POTWs based on site-specific conditions in accordance with the POTW's design treatment capacity and existing mass loadings.

Upset and Interference

EPA also assessed the ammonia concentrations and loads received by POTWs from Subtitle D leachate discharges to evaluate potential upsets or interference with POTW treatment systems. EPA concluded that national pretreatment standards were not required to prevent interference with POTW operations.

In terms of landfill leachate ammonia concentrations discharged to POTWs, only one of the Subtitle D landfill facilities in EPA's database is currently discharging (i.e. after treatment, if treatment is in place) wastewater to a POTW which contains more than 105 mg/L of ammonia. The remainder of the indirect-discharging Subtitle D landfills discharged an average concentration of 37 mg/L of ammonia to POTWs, with one-half of the facilities discharging less than 32 mg/L. Typical ammonia concentrations in raw domestic sewage range from 12 to 50 mg/L ("Operation of Municipal Wastewater Treatment Plants: Manual of Practice, Volume II," Water Pollution Control Federation).

The one facility in EPA's database that was discharging more than 105 mg/L of ammonia to a POTW was discharging 1,018 mg/L of ammonia to a 114 MGD POTW which currently has ammonia control (nitrification) in place. EPA also received influent ammonia data from several POTWs that commented on the proposed rule. The average ammonia influent concentration to POTWs ranged from 14 mg/L to 35

mg/L with an average concentration of 17 mg/L. Therefore, with the exception of the one outlier, the average concentration of ammonia in leachate discharged to POTWs (37 mg/L) noted in EPA's data closely parallels POTW experience (35 mg/L). However, it should be noted that the upper ranges of leachate concentrations were higher than the upper ranges observed in domestic sewage. Nevertheless, in most instances, observed ammonia discharge levels to POTWs fall within a POTW's treatment capabilities. Thus, EPA determined that the vast majority of Subtitle D landfills are discharging ammonia to POTWs at levels comparable to that which POTWs in the ordinary course of operations receive and treat in raw domestic sewage.

No POTWs commenting on the proposal cited any specific incidents where POTW acceptance of landfill leachate containing high levels of ammonia caused persistent upsets at the POTW. The data are consistent with that supplied by commenters and further supported EPA's understanding prior to the proposal of no documented persistent problems at POTWs due to ammonia concentrations in landfill leachate.

EPA also analyzed the effects that ammonia concentrations found in landfill leachate can have on the biological treatment systems at POTWs. In this analysis, EPA compared the concentrations of ammonia found in leachate with the activated sludge biological minimum threshold toxicity value (or inhibition value). With respect to ammonia, the inhibition value for activated sludge systems is 480 mg/L (Guidance Manual on the Development and Implementation of Local Discharge Limitations Under the Pretreatment Program, Volume 1. EPA, November 1987). The average raw wastewater concentration of ammonia found in Subtitle D landfills in EPA's database was 199 mg/L for direct, indirect and zero dischargers. In addition, all of the average and median ammonia concentration values observed in the data submitted to EPA in comments were below the activated sludge inhibition value. EPA has consequently determined that ammonia does not represent a threat to biological treatment systems that would require establishment of pretreatment standards.

Effect on Receiving Streams

Subsequent to the proposal, EPA evaluated total wastewater flows and loads of ammonia to receiving streams associated with non-hazardous landfill indirect dischargers (an estimated 756 facilities). EPA estimated that the non-hazardous landfill industry discharges 2.7 million pounds per year of ammonia to POTWs, which results in 1.6 million pounds per year being discharged to receiving streams, assuming that the POTWs have secondary treatment achieving 39 percent removal but do not have additional treatment for ammonia control. However, as mentioned above, EPA is aware that many POTWs have installed additional treatment specifically for the control of ammonia and typically achieve removals in excess of 95 percent. A review of EPA's 1996 Clean Water Needs Survey and its Permit Compliance System database indicates that approximately 20 percent of the POTWs in the U.S. employ some sort of ammonia control. Over 75 percent of the Subtitle D landfills in EPA's database discharge less than 10 pounds per day to the POTW (3,500 pounds/year), which results in discharging less than six pounds per day (2,100 pounds/year) to receiving streams, again assuming secondary treatment only and no additional POTW ammonia controls. In light of existing ammonia control in place at POTWs, actual discharges to receiving streams are likely to be even smaller.

Cost of Pretreatment Standards

EPA has evaluated the economic costs of ammonia pretreatment standards. EPA's economic assessment of these options demonstrated very high removal costs with low associated pollutant removals. Given the high cost, EPA concluded that it is not appropriate to establish national pretreatment standards to address the limited circumstances in which POTW removal might not match BAT removal performance.

EPA evaluated the costs of pretreatment standards in terms of the toxic pound equivalents. Pound-equivalents is a term used to describe a pound of pollutant weighted by its toxicity relative to copper. These weights are known as toxic weighting factors. The Agency calculates pound-equivalents by multiplying the pounds of a pollutant discharged from a landfill by the toxic weighting factor for that pollutant. The use of pounds-equivalent reflects the fact that some pollutants are more toxic than others.

The first treatment option that EPA evaluated is biological treatment. EPA evaluated PSES Option I equivalent to BPT/BAT Option I, which was equalization plus biological treatment. This option had a total annualized cost of $34.6 million (1998 dollars). Biological treatment removed 10,650 pound-equivalents annually, or an average of 14 pound equivalents per facility per year. This represents a cost of removal of $1,900/lb-equivalents (1981 dollars) and represents the cost of removing all of the pound-equivalents removed, not just ammonia. If EPA took credit only for the pound-equivalents of ammonia removed, the annual removal cost for this option is $7,100/lb-equivalents (1981 dollars). Moreover, these calculations are based on the assumption that POTWs will only remove 39 percent of the ammonia discharged to it. If POTWs remove more ammonia than that assumed, then the cost of each pound of pollutant removed by the industrial user raises. Given the installation of additional ammonia controls at many POTWs, actual ammonia removal by POTWs will be greater than assumed.

The second technology option EPA evaluated for the control of ammonia is ammonia stripping with appropriate air pollution controls. However, according to EPA's survey of the landfills industry, only two percent of survey respondents use this technology for the treatment of landfill leachate. In addition, air or steam stripping is more commonly used for treatment of wastewater containing concentrations of ammonia that are several orders of magnitude greater than those typically found in landfill wastewater. Therefore, EPA concluded that biological treatment systems are more appropriate for the treatment of the ammonia concentrations found in landfill leachate. In addition, air stripping for ammonia removal generally requires warm climates, and therefore this may not be a viable treatment option for all landfills located in the United States. In these circumstances, effluent levels associated with air stripping may not be attainable in all cases and thus not broadly available in the landfill industry. In addition, the air stripping option for the treatment of ammonia has an estimated annualized cost of $15.1 million (1998 dollars, pre-tax costs). The cost-effectiveness for this option is also high, $4,400/lb-equivalents (1981 dollars).

As explained above, EPA concluded that the vast majority of POTWs experience no difficulty in treating the ammonia loads received from landfill indirect dischargers, and that as a result, there is generally no pass through of ammonia from landfill leachate on a national basis. Moreover, the cost of pretreatment is not warranted by the limited circumstances where pretreatment would result in reduced ammonia to surface water. But there are POTWs without additional controls for ammonia that may not be equipped to handle landfill leachate ammonia discharges. Consequently, in the proposal, EPA requested comments on requiring ammonia pretreatment standards for those landfills discharging to POTWs that do not have ammonia controls in place. Several commenters supported no pretreatment standard because of their conclusion that ammonia loads from landfills made up an insignificant amount of the total ammonia loads discharged to POTWs. Others favored pretreatment standards because of smaller POTWs that do not employ nutrient removal systems. EPA, however, is not convinced that national ammonia pretreatment standards are warranted even where landfills are discharging to POTWs without ammonia controls given the high cost of pretreatment and current ammonia concentrations in landfill leachate discharged to POTWs that are generally consistent with values observed in raw sewage. Special ammonia situations are best addressed by the local POTW based on site-specific conditions in accordance with the POTW's design treatment capacity and existing mass loadings.

All of these factors discussed above confirm EPA's decision not to establish national ammonia pretreatment standards. EPA has concluded that landfills typically discharge wastewater to POTWs containing ammonia concentrations that can be adequately treated by POTWs. Further, in cases where ammonia loading rates are at levels which may be of concern or where ammonia discharges are a water quality concern, POTWs retain the ability to establish local limits on ammonia.

11.6.1.2 EPA's Rationale for Not Establishing PSES for Benzoic Acid

Benzoic Acid Pass-Through Analysis

As stated above, for the proposal, benzoic acid was not one of the pollutants EPA determined would pass through. However, after the proposal, EPA reviewed the BAT facilities and the representative POTW

facilities used for the removal comparison and determined that it had not used the appropriate editing rules. As a result of these revisions, the comparison showed that the median percent removal for benzoic acid at the landfills BAT facilities was 99 percent compared to the median POTW percent removal which was determined to be 81 percent. Because the 50-POTW database does not contain information on the percent removal of benzoic acid, EPA used the National Risk Management Research Laboratory Treatability database to estimate the percent removal. (For more information on EPA's use of the NRMRL database, see Chapter 7.)

Despite the difference in the BAT and POTW percent removals, further analysis of the data showed that both systems were achieving the same level of treatment of benzoic acid. That is, both the NRMRL database facilities representing POTWs and the landfills BAT facilities were treating benzoic acid down to non-detect levels (50 ug/L). Therefore, the smaller percent removal achieved by facilities in the NRMRL database (used to represent the POTW percent removal) is a function of lower influent concentrations at those facilities and is not necessarily indicative of inferior treatment at POTWs. EPA concluded that benzoic acid in these circumstances is susceptible to treatment at the POTW and does not pass through.

Benzoic Acid Loads Discharged to POTWs

In addition, EPA also evaluated the total flows and loads of benzoic acid discharged from non-hazardous landfills to POTWs. EPA compared the current discharge loads to the loads that would be anticipated after the implementation of pretreatment standards. As was explained above, EPA selected Option I (biological treatment) as the appropriate treatment technology and has analyzed the costs and benefits of pretreatment standards for the Non-Hazardous subcategory for this option. According to EPA's estimates, non-hazardous landfills currently discharge approximately 4,700 pounds of benzoic acid to POTWs per year resulting in an annual discharge of 900 pounds to receiving streams. PSES Option I (biological treatment) would reduce this annual discharge to receiving streams to 400 pounds per year. The average non-hazardous facility discharges only 6.4 pounds of benzoic acid annually (less than 0.02 pounds per day), and the median discharge is only 1.9 pounds per year. Furthermore, benzoic acid has a toxic weighting factor

of only 0.0003. Therefore, for the entire indirect-discharging non-hazardous landfills population (approximately 756 facilities), Option I would only remove an additional 0.16 pound-equivalents per year.

As a result of the above analysis, EPA determined that national pretreatment standards for benzoic acid are not necessary because benzoic acid is susceptible to treatment by POTWs. POTWs and landfill BAT facilities both treat benzoic acid down to non-detect levels. In addition, EPA determined that the pounds of benzoic acid currently being discharged by landfills are compatible with POTW treatment and that pretreatment standards would result in little further reduction of benzoic acid.

11.6.1.3 EPA's Rationale for Not Establishing PSES for P-Cresol

P-Cresol Pass-Through Analysis

Like benzoic acid, p-cresol also did not pass through POTWs according to EPA's pass-through analysis at proposal. However, the result of its revised removal comparison showed some difference in removal. The landfills median BAT percent removal for p-cresol is 99 percent, while the estimated median POTW percent removal is 68 percent. (Again, because the 50-POTW database does not contain percent removal data for p-cresol, EPA used the NRMRL database to determine POTW removal.)

P-Cresol Concentrations and Loads Discharged to POTWs

EPA also analyzed the flows and loads of p-cresol being discharged from non-hazardous landfills to POTWs. According to EPA's estimates, non-hazardous landfills currently discharge approximately 2,730 pounds of p-cresol to POTWs per year resulting in an annual discharge of 870 pounds to receiving streams. PSES Option I (biological treatment) would reduce this discharge to receiving streams to 130 pounds/year. Furthermore, p-cresol has a toxic weighting factor of only 0.0024. Therefore, the implementation of Option I results in an additional reduction of only 3.0 pound-equivalents per year across the entire Subtitle D indirect discharge population. On average, non-hazardous landfill facilities discharge only 3.4 pounds of p-cresol annually (or 0.01 pounds per day), and the median discharge load is only 0.7 pounds per year.

Based on the data shown above, EPA concluded that the implementation of pretreatment standards for p-cresol would result in only minimal reductions in the pounds of p-cresol discharged to surface waters. In addition, p-cresol is found in non-hazardous landfill leachate at concentrations which will not cause upsets at POTWs nor should POTWs have difficulty effectively treating such concentrations. The median raw wastewater concentration for p-cresol at municipal landfills is 75 ug/L. This concentration is well below the Universal Treatment Standard (UTS) of 770 ug/L established for F039 wastes (multi-source leachate) in 40 CFR 268.48. (EPA bases UTS on the Best Demonstrated Available Treatment Technology (BDAT) for each listed hazardous waste. BDAT represents the treatment technology that EPA concludes is the most effective for treating a particular waste that is also readily available to generators and treaters.)

11.6.1.4 EPA's Rationale for Not Establishing PSES for Phenol

Although phenol appeared to pass through, EPA decided not to establish pretreatment standards for phenol based on the fact that phenol is highly biodegradable and is treated by POTWs to the same degree as the landfill direct dischargers. Furthermore, the Agency concluded that the differences in influent concentrations caused the apparent difference in removal performance between landfill direct dischargers and POTWs. As a result, the performance across the landfills direct dischargers showed higher removals than the performance at the POTWs.

In EPA's landfills database, raw wastewater concentrations of phenol at the BAT facilities in the Non-Hazardous subcategory were much higher than the influent concentrations at the POTWs used in the determination of the POTW percent removal. The average influent concentrations for phenol for the three non-hazardous BAT facilities used in the pass-through analysis ranged from 350 ug/L to 5,120 ug/L. All three of the facilities treated phenol down to the analytical minimum level (10 ug/L), corresponding to a median percent removal of 97.5 percent. For POTW performance, EPA used a total of eight POTWs in the analysis for POTW percent removal of phenol. The average influent concentration for phenol at these eight POTWs was 387 ug/L, and six of the eight effluent values were below the analytical minimum level and therefore assigned values of 10 ug/L. Thus, the average percent removal for the POTWs was 95.3 percent.

In this case, EPA concluded that the differences in removals for POTWs (95.3 percent) and BAT facilities (97.5 percent) is an artifact of the differing influent concentrations and does not necessarily reflect a real difference in treatment performance. Therefore, EPA concluded that phenol is treated to essentially the same level by direct dischargers and POTWs and, therefore, does not pass through.

Based on the pollutant loadings rationale described above for ammonia , benzoic acid, and p-cresol, and based on the highly biodegradable nature of phenol, EPA decided not to set pretreatment standards for landfills in the Non-Hazardous subcategory. In addition, the Agency concluded that in the case of discharges from Subtitle D landfills, problems that may result from elevated ammonia loads in landfill leachate are best addressed at the local level. Furthermore, the Agency has determined that as a result of the ability of POTWs to adequately treat the small quantities of benzoic acid and p-cresol being discharged from landfills, a pretreatment standard for these two pollutants is also unnecessary. EPA also concluded that the cost to implement pretreatment standards for this subcategory is not warranted by the environmental benefits associated with any small additional removals.

11.6.1.5 Public Comments to the Proposed Rule Regarding Non-Hazardous PSES

In support of EPA's proposal not to establish PSES for the Non-Hazardous subcategory, EPA received comments and data following the proposal concerning the treatment of non-hazardous landfill leachate at POTWs. A total of seventeen commenters, representing municipalities, POTWs, privately-owned landfills, trade associations, and engineering consulting firms, stated that in their experience, no POTW upsets or adverse impacts on sludge quality had occurred as a result of a POTW accepting non-hazardous landfill leachate. Several of these commenters supported their claim with data or anecdotal evidence from over 20 landfills discharging leachate to POTWs. Most of these commenters felt that local limits are currently addressing discharges from non-hazardous landfills and that any particular pollutant that may be of concern should be dealt with on a case-by-case basis. Commenters also stated that the implementation of pretreatment standards would be extremely costly for very little improvement in water quality. Commenters stressed that any mandatory pretreatment that did not take into account the ability of receiving POTWs to

handle the wastewater would inevitably result in unnecessary pretreatment of some waste streams. EPA found that this comment is particularly applicable to ammonia because of the varying degrees of treatment that can be achieved by POTWs. Furthermore, several commenters felt that the constituents found in landfill leachate are similar to those found in the influent to POTWs and that the flow contribution from landfills is relatively small.

There were also several commenters who supported the establishment of pretreatment standards for non-hazardous landfill leachate. One municipality was concerned with the effects that landfill leachate can have on small community POTWs with low flows. Specifically, the commenter was concerned with elevated levels of three specific pollutants (zinc, chromium, and cyanide) at three landfills that discharge to the city's POTW. The concentrations cited by the commenter for chromium and zinc were much higher than the median concentration determined for these metals by EPA's data gathering efforts. In addition, EPA did not detect cyanide in its analytical sampling at Subtitle D landfills. As a result, EPA determined that the pollutant concentrations identified by this municipality are not indicative of the concentrations typically present at Subtitle D landfills. Therefore, in cases where elevated levels of pollutants present in landfill leachate may cause problems for a POTW, local, site-specific limits are the best way to implement controls on such discharges. Furthermore, EPA did not receive any comments from POTWs that had experienced persistent upsets as a result of accepting landfill leachate.

One other municipality felt that EPA should set pretreatment standards for non-hazardous landfills since close to 70 percent of the wastewater flow discharged from Subtitle D landfills is discharged to POTWs. EPA establishes pretreatment standards for pollutants that are not susceptible to treatment by POTWs to prevent pass through and interference based on the ability of POTWs to achieve treatment equivalent to that of direct dischargers. The percentage of total flow of an industry being discharged to POTWs is not a basis for establishing pretreatment standards. Furthermore, EPA determined that the total loads of the pollutants that are discharged to POTWs made up only a very small fraction of what the POTW receives, and that the concentrations of these pollutants are at levels that are compatible with POTW treatment.

Other commenters disagreed with EPA's statements that non-hazardous leachate is of the same quality as the headwaters of a POTW. Three of these commenters were particularly concerned with ammonia concentrations in landfill leachate (the data from these commenters was discussed in the ammonia discussion above). EPA reviewed the data submitted by these commenters and, although some pollutants exceeded EPA's median concentrations, the commenters did not cite any specific instances where the reported leachate concentrations created a problem for a receiving POTW. EPA acknowledges that elevated levels of pollutants can exist in landfill wastewater. However, the median concentrations of pollutants determined by EPA's sampling program indicate that, on a national basis, concentration levels of pollutants are not at a level to be of concern to POTWs. In addition, in many cases, the loads of pollutants discharged from landfills to POTWs make up a very small portion of the total pollutant loads received by the POTW.

Another commenter suggested that EPA consider setting pretreatment standards for sulfates and sodium in landfill discharges. The commenter stated that the levels of sodium found in landfill leachate is generally greater than the level of 20 mg/L indicated in EPA's Drinking Water Contaminant Candidate List. EPA did not include limits for sulfates or sodium since these can be found in naturally occurring compounds in landfill soils and are often constituents in treatment chemicals commonly used for wastewater treatment.

One municipality commented in favor of PSES for ammonia since its regional POTWs had to establish a local ammonia pretreatment limit of 100 mg/L to protect water quality in ocean outfalls. However, in this case, the local authority has determined that a 100 mg/L pretreatment standard is adequate for the protection of water quality in the ocean outfalls. EPA acknowledges that there may be circumstances in which POTWs may have to establish local limits in order to prevent upsets or pass through. These situations do not undercut EPA's decision not to establish national pretreatment standards for ammonia. As explained in Section 11.6.1.1, the removal technologies evaluated for PSES would result in very low ammonia discharge levels, much lower than that established by the commenter (100 mg/L). This situation further supports EPA's conclusion that local limits for ammonia provide the most appropriate control and that national pretreatment requirements for ammonia may result in unnecessary pretreatment of some waste streams. In fact, one of

the two landfills discharging leachate to the district's POTW has since installed an SBR. As a result, the leachate ammonia concentration for this landfill has dropped from an average of 393 mg/L to 52 mg/L. The fact that one of the landfills has installed pretreatment to lower ammonia discharges is a good example that existing pretreatment programs are effective at requiring landfills to control their discharges.

One of the commenters in support of PSES already employs biological pretreatment at its landfill. This landfill specifically stated that concentrations of ammonia as nitrogen and total toxic organics should undergo pretreatment prior to discharge to a POTW unless the leachate is a very small constituent of the total flow of the POTW. The raw wastewater ammonia concentrations at this landfill were consistent with the median determined by EPA's sampling efforts and the facility employed biological treatment to achieve an effluent ammonia concentration that was acceptable to the local POTW. In addition, the concentrations of toxic organics found in EPA's sampling of Subtitle D landfill leachate were typically not at levels that would cause inhibition to biological treatment at a POTW. The specific organic pollutants that EPA determined to pass through were found in very low concentrations, resulting in minimal loadings discharged to POTWs. The fact that this landfill already employs pretreatment is a good example that existing pretreatment programs are effective at requiring landfills to control their discharges.

11.6.2 EPA's Decision Not to Establish PSES for the Subtitle C Hazardous Subcategory

In the proposed rule, EPA proposed pretreatment standards for six pollutants that EPA determined to pass through in the Hazardous subcategory. However, after reviewing the comments received and re-evaluating the pollutant loads in the Hazardous subcategory, EPA has decided not to establish national pretreatment standards for Subtitle C landfills.

As previously explained, EPA establishes pretreatment standards for pollutants that are not susceptible to treatment at a POTW or for pollutants that may interfere with POTW operations. As explained at section 11.2.3, for the Hazardous subcategory, EPA identified only three Subtitle C landfills, all of them indirect dischargers. EPA used data from these hazardous landfills to develop the BPT/BAT limitations for toxic

pollutants because these landfills were using the treatment systems for their leachate that EPA determined was the BPT/BAT treatment technology.

EPA also performed an analysis for this subcategory in order to compare POTW removals with BAT treatment systems. As was the case for the Non-Hazardous subcategory, EPA revised the pass-through analysis data editing procedures after the proposal and as a result EPA's removal results have changed. The result of the revised comparison show BAT removals greater than POTW removals for the following eight pollutants: ammonia, alpha terpineol, aniline, benzoic acid, naphthalene, p-cresol, phenol, and pyridine.

For its removal comparison for ammonia, EPA compared the nation-wide median percentage of ammonia removed by well-operated POTWs to the percentage of ammonia removed by BAT treatment systems from both the Hazardous and Non-Hazardous subcategories. (For the reasons explained in section 11.2.4, in the case of ammonia, EPA supplemented the Hazardous subcategory data with data from non-hazardous landfill facilities.) For all other toxic pollutants, in determining whether a pollutant would pass through a POTW, the Agency compared the nation-wide median percentage of a pollutant removed by well-operated POTWs with secondary treatment to the percentage of a pollutant removed by BAT treatment systems from only the Hazardous subcategory. For the proposal, EPA proposed pretreatment standards that were equivalent to the BPT/BAT limitations for the pollutants that passed through. EPA has reconsidered its decision that it should promulgate national pretreatment standards for hazardous landfills. The reasons for this decision are explained in more detail below.

Two of the indirect discharging landfills have treatment technology in place that EPA considers to be BAT, and currently discharge very low concentrations of pollutants to their local POTWs. The third and only other indirectly discharging Subtitle C landfill for which EPA has data discharged less than 1,000 gal/day of landfill gas collection condensate to a POTW. In addition to the low wastewater flow at this landfill, the facility has relatively low raw wastewater pollutant concentrations and employs neutralization with ammonia followed by settling prior to discharge to the POTW.

Several commenters on the proposal questioned EPA's rationale for developing ammonia pretreatment standards for the Hazardous subcategory while not establishing ammonia pretreatment standards for the Non-Hazardous subcategory. EPA's database indicate that the median raw wastewater ammonia concentration for hazardous landfills is 268 mg/L as compared to the raw wastewater ammonia concentration for Subtitle D landfills which is 199 mg/L.[1] EPA has current information on ammonia concentration in wastewater discharges for two of the three Subtitle C landfills in EPA's database. One of the landfills employs biological treatment and discharges an average of 4.9 mg/L of ammonia to the POTW. The other landfill employs chemical precipitation prior to biological treatment and discharges ammonia at an average concentration of 156 mg/L. This discharge level presents no apparent problem to the receiving POTW. According to discussions with this facility and the POTW, the POTW has not set local pretreatment standards for ammonia for this landfill, and the POTW does not perform nitrification nor is there an ammonia limit in the POTW's NPDES permit. Since 1995, the POTW has seen the ammonia concentration at its headworks increase from 13 mg/L to 20 mg/L and has experienced some upsets at the POTW. However, the POTW explained that it was unsure whether the upsets are a result of the increased ammonia concentrations or due to some other constituent in the wastewater. In addition, the POTW is not sure if the landfill leachate discharge is contributing at all to the upsets. As was the case in the Non-Hazardous subcategory, EPA concluded that national pretreatment standards for ammonia are not warranted by the small quantity of ammonia being discharged to POTWs from landfills in this subcategory and due to the site-specific water quality and POTW nitrification issues associated with ammonia.

Although the removal comparison suggests that phenol may pass through, EPA decided not to establish pretreatment standards for it because it is highly biodegradable and is, in fact, treated by POTWs to the same degree as the landfill direct dischargers. The Agency concluded that any apparent difference in

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[1] In the comments received on the proposal, some commenters referred to the Hazardous subcategory median ammonia raw wastewater concentration referred to in Table 6-8 on page 6-44 of the Proposed Landfills Development Document (EPA -821-R-97-022). This table lists the median ammonia raw wastewater concentration of 8.6 mg/L. However, this median concentration included numerous CERCLA facilities with discharges that consisted primarily of ground water. After proposal, EPA recalculated the median ammonia raw wastewater concentration for the Hazardous subcategory using only data from Subtitle C landfills in EPA's database. This results in a median raw wastewater ammonia concentration of 268 mg/L.

removals in the removal comparison is an artifact of differing influent concentrations rather than any difference in performance between landfill direct dischargers and POTWs.

In EPA's landfills database, raw wastewater concentrations of phenol at the two BAT facilities in the Hazardous subcategory were much higher than the influent concentrations at the POTWs used in the determination of the POTW percent removal. The average influent concentrations for phenol for the two hazardous BAT facilities used in the pass-through analysis ranged from 5,120 ug/L to 98,500 ug/L, and the average effluent concentrations ranged from 10 ug/L to 814 ug/L corresponding to an average percent removal of 99.8 percent. For POTW performance, EPA used a total of eight POTWs in the analysis for POTW percent removal of phenol. The average influent concentration for phenol at these eight POTWs was 387 ug/L, and six of the eight effluent values were below the analytical minimum level and therefore assigned values of 10 ug/L. Thus, the average percent removal for the POTWs was 95.3 percent, and therefore EPA determined that the pollutant passed through. In this case, EPA concluded that the pass-through determination is an artifact of the differing influent concentrations and does not necessarily reflect a real difference in removals. Therefore, EPA concluded that phenol is treated to essentially the same level by direct dischargers and POTWs and, therefore, does not pass through.

Further review of the comparison for alpha terpineol, aniline, benzoic acid, naphthalene, and pyridine under the revised analysis showed that all of these pollutants were treated down to non-detect levels in both the landfill's BAT treatment option and in the NRMRL facilities representing POTWs. That is, both BAT facilities and POTWs achieve the same level of treatment for these pollutants, and the differences in removal once again were simply a function of smaller influent concentrations at facilities representing POTWs. (Alpha terpineol and benzoic acid are compounds for which a high removal efficiency would be expected at a POTW due to their relatively high biodegradability.) Therefore, the Agency determined that, not only are the current pollutant loads not a problem for POTWs, but also all of these pollutants are present in concentrations that are treated down to non-detect levels in a well-operated POTW. Thus, given the small

loadings and low concentrations of these pollutants, EPA concluded that these five pollutants are susceptible to treatment at the POTW and do not pass through.

Furthermore, EPA has concluded that while the removal comparison suggests that two pollutants, naphthalene and aniline, may not be susceptible to POTW treatment, in fact, they will receive equivalent treatment. First, the median untreated wastewater concentration observed in EPA's data collection effort for these pollutants is less than the Universal Treatment Standards (UTS) EPA has developed for these pollutants in F039 wastes (multi-source leachate) in 40 CFR 268.48. The UTS for naphthalene is 0.059 mg/L which is slightly greater than the median concentration found in hazardous landfills (0.049 mg/L). The UTS standard for aniline is 0.81 mg/L while the median concentration in hazardous landfills is 0.237 mg/L. Second, aniline and naphthalene (as well as p-cresol and pyridine) will be removed from wastewater through attachment to the biosolids in the POTW's biological treatment system and then undergo subsequent biodegradation while entrained in the biosolids.

In addition, as noted above, the revised comparison shows a lower POTW removal for p-cresol than that achieved by BAT treatment. However, as was the case in the Non-Hazardous subcategory, EPA has concluded that the concentrations of p-cresol and the associated loadings discharged to POTWs from landfills in the Hazardous subcategory would be insignificant compared to the total loads received at the POTW. The median Subtitle C raw wastewater concentration for p-cresol is 144 ug/L (this includes only Subtitle C landfills and not the CERCLA data included in the median on page 6-44 of the Proposed Landfills Development Document) which is less than the UTS developed for p-cresol in F039 wastes which is 770 ug/L (40 CFR 268.48).

Therefore, based on the small quantity of pollutants involved and low pollutant concentrations discharged from landfills in the Hazardous subcategory, EPA concluded that national pretreatment standards for landfills in the Hazardous subcategory are unnecessary. In addition, EPA concluded that local limits are adequately controlling wastewater discharges from Subtitle C landfills.

11.7 Pretreatment Standards for New Sources (PSNS)

Section 307 of the Clean Water Act requires EPA to promulgate both pretreatment standards for new sources and new source performance standards. New indirect-discharging facilities, like new direct-discharging facilities, have the opportunity to incorporate the best available demonstrated technologies, including process changes, in-facility controls, and end-of-pipe treatment technologies.

EPA decided not to establish pretreatment standards for new sources for both subcategories for many of the same reasons that EPA did not establish PSES limits. As stated in the PSES discussions above, EPA concluded that the typical concentrations of pollutants in landfill leachate are not at levels that will cause problems for POTWs. In addition, EPA determined that the relatively small wastewater flows from landfills, coupled with the concentrations of pollutants typically found, result in small pollutant loading rates discharged to POTWs from landfills. Finally, in site-specific cases where a particular pollutant may be found at concentrations that are of concern to the POTW, EPA concluded that local pretreatment standards are the most appropriate means for controlling such discharges.

Table 11-1: Removal of Pollutant of Interest Metals in the Non-Hazardous Subcategory (ug/L)

Non-Hazardous POI Metals	CAS #	Landfills Raw Wastewater Data		NRMRL Treatability Data (1)		Published Inhibition Levels (2)	50-POTW Study (3)			OCPSF 12 Plant Sampling Data (4)	
		Subtitle D Municipal Median Concentration	Subtitle D Non-Municipal Median Concentration	Biological Treatment Systems			Maximum Influent Concentration	Mean Influent Concentration	Median Percent Removal	Biological Treatment Systems	
				Influent Concentration	Percent Removal					Median Influent Concentration	Percent Removal
Barium	7440393	483	822	1,000-10,000	84.0	NA	NA	NA	NA	NA	NA
Chromium	7440473	28	NA	44	45.0	1,000-100,000	2,380	173	82	440	68.5
Strontium	7440246	1,671	4,615	1,000-10,000	14.0	NA	NA	NA	NA	NA	NA
Titanium	7440326	64	11.8	55	34.0	NA	NA	NA	NA	NA	NA
Zinc	7440666	100	93	372	56.0	80-5,000	9,250	723	79	322	58.5

NA - Not applicable or not available.
(1) Source: EPA National Risk Management Research Laboratory (NRMRL) Treatability Database.
(2) Source: EPA Guidance Manual on the Development and Implementation of Local Discharge Limitations Under the Pretreatment Program, Volume 1. EPA Nov 1987.
(3) Source: EPA Fate of Priority Pollutants in Publicly Owned Treatment Works. (EPA 440/1-82/303, September 1982).
(4) Source: EPA Organic Chemicals, Plastics and Synthetic Fibers Public Record.

Table 11-2: List of Subtitle D Municipal Solid Waste Facilities Employing Biological Treatment Considered for BPT in the Non-Hazardous Subcategory

Facility Questionnaire ID Numbers	
16001	16119
16047	16120
16048	16121
16049	16122
16052	16123
16056	16125
16058	16127
16059	16129
16060	16132
16063	16154
16065	16155
16077	16158
16078	16159
16079	16161
16083	16164
16085	16165
16088	16166
16093	16170
16097	16171
16099	16174
16102	16176
16117	16253
16118	

Table 11-3: Comparison of Raw Wastewater Mean Concentrations of Non-Hazardous Pollutants of Interest for Municipal Solid Waste Landfills and Hazardous Facility 16041

Cas No.	Pollutant	Mean concentration of Pollutants of Interest for All Municipal Landfills in EPA Database	Mean Concentration of Pollutants of Interest for Hazardous Facility 16041
C-002	Biochemical Oxygen Demand	1,228,534	877,875
C-004	Chemical Oxygen Demand	2,024,932	2,033,750
C-005	Nitrate/Nitrite	5,844	1,770
C-009	Total Suspended Solids	735,308	191,375
C-010	Total Dissolved Solids	4,195,518	12,275,000
C-012	Total Organic Carbon	661,478	562,250
C-020	Total Phenols	142,838	3,195
106445	P-Cresol	246	218
108101	4-Methyl-2-Pentanone	3,789	2,175
108883	Toluene	166	1,468
108952	Phenol	287	1,553
120365	Dichloroprop	10	2
123911	1,4-Dioxane	118	10
142621	Hexanoic Acid	13,148	1,632
18540299	Chromium (Hexavalent)	77	Not analyzed
20324338	Tripropyleneglycol Methyl Ether	568	1,750
298044	Disulfoton	9	Not analyzed
3268879	OCDD	0.03	6
35822469	1234678-HpCDD	0.002	1
65850	Benzoic Acid	7,220	5,294
67641	2-Propanone	2,407	4,398
68122	N,N-Dimethylformamide	214	Not analyzed
7440213	Silicon	30,913	5,518
7440246	Strontium	1,569	2,846
7440326	Titanium	66	65
7440393	Barium	720	Not analyzed
7440428	Boron	3,005	8,839
7440473	Chromium	46	87
7440666	Zinc	1,476	253
75092	Methylene Chloride	70	49
7664417	Ammonia Nitrogen	238,163	382,250
78933	2-Butanone	5,119	6,398
95487	O-Cresol	298	10
98555	Alpha Terpineol	334	691

Table 11-4: Candidate BPT Facilities for the Non-Hazardous Subcategory
Eliminated from BPT Consideration Because No BOD_5 Effluent Data Was Available

Facility Questionnaire ID Numbers	
16001	16102
16047	16119
16056	16121
16059	16123
16060	16154
16063	16155
16078	16158
16079	16159
16083	16166
16085	16174

Table 11-5: Treatment Systems In Place at Landfill Facilities Considered for BPT Which Supplied BOD_5 Effluent Data

Facility QID	Treatment in Place
16041	Sequencing batch reactor (SBR)
16048	Aerobic (oxidation pond)
16049	Aerobic-anaerobic (facultative pond)
16052	Aerobic-anaerobic (oxidation pond)
16058	Aerated lagoon
16065	Aerobic pond
16077	Aerated lagoon
16088	Equalization, sand filter, carbon adsorption, aerobic
16093	Activated sludge, secondary clarifier, disinfection, multimedia filtration
16097	Activated sludge, secondary clarifier
16099	Equalization, chemical precipitation, flocculation, coalescing, anaerobic, activated sludge with PACT, nitrification, secondary clarifier
16117	Equalization, chemical precipitation, primary clarifier, aerated fixed film, secondary clarifier, denitrification
16118	Equalization, chemical precipitation, primary clarifier, anaerobic, aerobic, secondary clarifier
16120	Settling, aeration, chemical precip, primary clarifier, air stripper, neutralization, activated sludge, secondary clarifier, multimedia filtration, disinfection
16122	Equalization, chemical precipitation, primary clarifier, anaerobic, aerobic, secondary clarifier, aerobic equalization, multimedia filtration
16125	Aeration, chemical precipitation, primary clarifier, SBR, secondary clarifier, carbon adsorption, multimedia filtration
16127	Unstirred tank, aeration
16129	Neutralization (lime), chemical precipitation, primary clarifier, activated sludge, secondary clarifier, sand filter, air stripping
16132	Aerated pond
16161	Aeration, aerobic, settling (aerated pond)
16164	Aeration, chemical precipitation, primary clarifier, neutralization, equalization, aerobic, secondary clarifier
16165	Aerobic, settling (aerated pond)
16170	Equalization, stabilization pond
16171	Equalization, activated sludge, settling
16176	Aeration, activated sludge, settling
16253	Equalization, chemical precipitation, primary clarifier, anaerobic, activated sludge, secondary clarifier, nitrification, multimedia filter

Table 11-6: Landfill Facilities Considered for BPT in the Non-Hazardous Subcategory which Supplied BOD_5 Effluent Data

Facility QID	Bsl Flow (MGD)	BOD5 (mg/L)								Reason Facility Data was not Considered for BOD Limitations
		Facility Avg		DET		DMQ		ANL		
		Inf	Eff	Inf	Eff	Inf	Eff	Inf	Eff	
16041	0.058917	910	47	-	-	-	-	910	45	Data used for calculating BOD limits
16048	0.000005	NA	41	-	-	-	-	-	-	No BOD influent data
16049	0.0017	NA	NA	-	4.8	-	-	-	-	No BOD influent data
16052	0.0546	NA	37	-	37	-	-	-	-	No BOD influent data
16058	0.003	153	24	-	22	-	30	153	-	Data used for calculating BOD limits
16065	0.008	NA	35	-	35	-	-	-	-	No BOD influent data
16077	0.00816	54	10	54	10	-	-	-	-	Average influent BOD concentration below 100 mg/L
16088	0.03621	3799	209	-	200	3799	223	-	-	Effluent BOD concentration greater than 50 mg/L
16093	0.081575	24	8.3	27	6	22	8.3	-	-	Average influent BOD concentration below 100 mg/L
16097	0.019	23	14	-	20	23	14.3	-	-	Average influent BOD concentration below 100 mg/L
16099	0.01533	3600	11.5	-	8	3600	11.5	-	-	Carbon treatment used in addition to biological treatment
16117	0.04	180	4.8	-	4	180	5.5	-	-	Separate treatment trains (BIO/CPR) employed before
16118	0.0288	1990	48	2200	49	1890	46	-	-	Data used for calculating BOD limits
16120	0.042775	790	10	-	16	780	4.6	1290	-	Data used for calculating BOD and TSS limits
16122	0.0255	1007	6.1	-	5.3	-	5.4	1007	30	Data used for calculating BOD and TSS limits
16125	0.014193	1673	57	1141	10	2394	-	379	171	Carbon treatment used in addition to biological treatment
16127	0.003627	NA	40	-	-	-	40	-	-	Wastewater stream consists primarily of storm water
16129	0.00469	214	1.8	-	-	214	1.8	-	-	Wastewater stream consists primarily of ground water
16132	0.03	7609	15.7	5581	7	4741	16	-	-	Data used for calculating BOD limits
16161	0.053	NA	171	-	171	-	-	-	-	No BOD influent data
16164	0.01	NA	487	-	487	-	-	-	-	No BOD influent data
16165	0.030218	1812	974	1812	974	-	-	-	-	Effluent BOD concentration greater than 50 mg/L
16170	0.0048	69	63	-	54	69	72	-	-	Average influent BOD concentration below 100 mg/L
16171	0.024	NA	213	-	213	-	-	-	-	No BOD influent data
16176	0.037272	NA	112	-	112	-	-	-	-	No BOD influent data
16253	0.01776	327	6.4	1000	5.2	159	6.4	-	-	Data used for calculating BOD and TSS limits

Bsl Flow: Baseline flow

Facility Avg: Flow weighted average calculated from all data sources available at the facility (DET, DMQ, ANL)

DET: Detailed Questionnaire data from 1992

DMQ: Detailed Monitoring Questionnaire data from 1992 through 1994

ANL: Analytical data from sampling episodes 1993-1995

NA: Not Available

Table 11-7: Selected BPT Facilities for the Non-Hazardous Subcategory

Detailed Questionnaire ID Number	Discharge Status	Treatment in Place
16041	Indirect	sequential batch reactor
16058	Direct	equalization, aerated lagoon
16118	Indirect	aerated equalization, chemical precipitation, anaerobic fixed film, aerobic fixed film, clarification
16120	Direct	aerated equalization, chemical precipitation, ammonia strip lagoons, neutralization, activated sludge, multimedia filter, chlorination
16122	Direct	aerated equalization, chemical precipitation, flocculation, clarification, neutralization, anaerobic fixed film, aerobic fixed film, neutralization, coagulation, flocculation, clarification, chlorination, aerated equalization, multimedia filter
16132	Indirect	aerated pond
16253	Direct	equalization, chemical precipitation, flocculation, clarification, neutralization, anaerobic filtration, 2-stage activated sludge, multimedia filter

Table 11-8: TSS Data from Landfill Facilities Selected for BPT in the Non-Hazardous Subcategory

Facility QID	Baseline Flow (MGD)	TSS (mg/L)							
		Facility Average		DET		DMQ		ANL	
		Influent	Effluent	Influent	Effluent	Influent	Effluent	Influent	Effluent
16041	0.058917	330	36	364	36	307	35	70	46
16058	0.003	14470	188	-	216	-	188	14470	-
16118	0.0288	NA	NA	-	-	-	-	-	-
16120	0.042775	1221	14	-	14	1241	13.6	200	-
16122	0.0255	267	5.4	-	5.6	-	5.4	267	12.5
16132	0.03	244	47	244	39	-	47	-	-
16253	0.01776	150	25	180	17.5	120	25	-	-

Facility Avg: weighted average calculated from all data sources available at the facility (DET, DMQ, ANL).

DET: Detailed Questionnaire data from 1992

DMQ: Detailed Monitoring Questionnaire data from 1992 through 1994

ANL: Analytical data from sampling episodes 1993-1995

NA: Not Available

Table 11-9: Facilities and Sample Points Used for the Development of BPT/BAT Effluent Limitations for the Non-Hazardous Subcategory

BPT Facility	Data Source	Influent Sample Point	Avg. Influent Concentration	Effluent Sample Point	Avg. Effluent Concentration
Ammonia (mg/L)					
16041	DMQ ANL	02 01, 03, 05, 06	679 475	04 02	5.4 1.4
16122	ANL	01, 02, 03	181	07	1.2
16132	DMQ	01, 02, 03	206	04	5.9
BOD_5 (mg/L)					
16041	ANL	01, 03, 05, 06	910	02	47
16058	DMQ ANL	- 01, 02	- 153	01 -	29.7 Only influent conc. used
16118	DMQ	01	1,890	02	45.5
16120	DMQ	01	780	02	4.6
16122	ANL	01, 02, 03	1,007	07	35.2
16132	DMQ	01, 02, 03	4,740	04	15.8
16253	DMQ	01	159	02	6.4
TSS (mg/L)					
16120	DMQ	01	1,240	02	13.6
16253	DMQ	01	120	02	24.9
Alpha Terpineol (ug/L)					
16041	ANL	01, 03, 05, 06	653	02	10
16122	ANL	01, 02, 03	123	07	10
Benzoic Acid (ug/L)					
16041	ANL	01, 03, 05, 06	15,400	02	50
16122	ANL	01, 02, 03	9,300	07	50
P-Cresol (ug/L)					
16041	ANL	01, 03, 05, 06	1,360	02	10

Table 11-9: Facilities and Sample Points Used for the Development of BPT/BAT Effluent Limitations for the Non-Hazardous Subcategory (continued)

BPT Facility	Data Source	Influent Sample Point	Avg. Influent Concentration	Effluent Sample Point	Avg. Effluent Concentration
Phenol (ug/L)					
16041	ANL	01, 03, 05, 06	5,120	02	10
16118	DET DMQ	01 01	350 -	02 02	Only influent conc. used 11
16120	DMQ ANL	01 01	16 712	02 -	27.7 Only influent conc. used
16122	ANL	01, 02, 03	395	07	10
Zinc (ug/L)					
16041	DMQ ANL	02 01, 03, 05, 06	505 310	04 02	214 87
16058	DMQ ANL	- 01, 02	- 995	01 -	59 Only influent conc. used
16132	DMQ	01, 02, 03	490	04	50

ANL: Analytical data
DET: Detailed Questionnaire data
DMQ: Detailed Monitoring Questionnaire data

Table 11-10: BPT Facility Data Excluded from the Calculation of Non-Hazardous BPT/BAT Limitations

BPT Facility	Influent Sample Point	Avg. Influent Conc.	Effluent Sample Point	Avg. Effluent Conc.	Reason for Exclusion
BOD_5 (mg/L)					
16041 DMQ	02 02	- -	04 04	- -	No data No data
16058	-	-	01	22	Detailed questionnaire data was not used
16118	01	2,200	02	49	Detailed questionnaire data was not used
16120 ANL	- 01	- 1,290	02 -	15.9 -	Detailed questionnaire data was not used No effluent data
16122 DMQ	01 01	- -	03 03	5.3 5.4	Effluent sample point 03 located after aerated equalization
16132	01,02,03	5,581	04	7	Detailed questionnaire data was not used
16253	01	1,000	02	5.2	Detailed questionnaire data was not used
TSS (mg/L)					
16041 DMQ ANL	02 02 1, 3, 5, 6	364 307 70	04 04 02	36 35 46	Facility wastewater treatment system does not employ filtration
16058 DMQ ANL	- - 01, 02	- - 14,470	01 01 -	216 188 -	Facility wastewater treatment system does not employ filtration
16118 DMQ	01 01	- -	02 02	- -	Facility wastewater treatment system does not employ filtration
16120 ANL	- 01	- 200	02 -	14 -	Detailed questionnaire data was not used No effluent data
16122 DMQ ANL	01 01 01,02,03	- - 267	03 03 07	5.6 5.4 12.5	Facility eliminated due to settling that can occur in equalization tanks prior to filtration
16132 DMQ	01,02,03 01,02,03	244 -	04 04	39 47	Facility wastewater treatment system does not employ filtration
16253	01	180	02	17.5	Detailed questionnaire data was not used

Table 11-10: BPT Facility Data Excluded from the Calculation of Non-Hazardous BPT/BAT Limitations (continued)

BPT Facility	Influent Sample Point	Avg. Influent Conc.	Effluent Sample Point	Avg. Effluent Conc.	Reason for Exclusion
Ammonia (mg/L)					
16041	02	554	04	5.0	Detailed questionnaire data was not used
16058 DMQ ANL	- - 01, 02	- - 2,900	01 01 -	- - -	No data No data No effluent data
16118 DMQ	01 01	- -	02 02	- -	No data No data
16120 DMQ ANL	- 01 01	- 362 245	02 02 -	1.35 5.98 -	Facility wastewater treatment system employed an air stripper
16122 DMQ	01 01	136 135	03 03	0.87 0.48	Effluent sample point 03 located after aerated equalization
16132	01,02,03	-	04	-	No data
16253 DMQ	01 01	- -	02 02	- 0.01	No data No influent data
Alpha Terpineol (ug/L)					
16041 DMQ	02 02	- -	04 04	- -	No data No data
16058 DMQ ANL	- - 01, 02	- - -	01 01 -	- - -	No data No data No data
16118 DMQ	01 01	- -	02 02	- -	No data No data
16120 DMQ ANL	- 01 01	- - -	02 02 -	- - -	No data No data No data
16122 DMQ	01 01	- -	03 03	- -	No data No data
16132 DMQ	01,02,03 01,02,03	- -	04 04	- -	No data No data
16253 DMQ	01 01	- -	02 02	- -	No data No data

Table 11-10: BPT Facility Data Excluded from the Calculation of Non-Hazardous BPT/BAT Limitations (continued)

BPT Facility	Influent Sample Point	Avg. Influent Conc.	Effluent Sample Point	Avg. Effluent Conc.	Reason for Exclusion
Benzoic Acid (ug/L)					
16041	02	-	04	-	No data
DMQ	02	-	04	-	No data
16058	-	-	01	-	No data
DMQ	-	-	01	50	No influent data
ANL	01, 02	50	-	-	Influent concentration < 10xMDL
16118	01	-	02	-	No data
DMQ	01	-	02	-	No data
16120	-	-	02	-	No data
DMQ	01	-	02	-	No data
ANL	01	-	-	-	No data
16122	01	-	03	-	No data
DMQ	01	-	03	-	No data
16132	01,02,03	-	04	-	No data
DMQ	01,02,03	-	04	-	No data
16253	01	-	02	-	No data
DMQ	01	-	02	20	No influent data
P-Cresol (ug/L)					
16041	02	-	04	-	No data
DMQ	02	-	04	-	No data
16058	-	-	01	-	No data
DMQ	-	-	01	10	No influent data
ANL	01, 02	48	-	-	Influent concentration < 10xMDL
16118	01	-	02	-	No data
DMQ	01	-	02	-	No data
16120	-	-	02	-	No data
DMQ	01	30	02	10	Influent concentration < 10xMDL
ANL	01	10	-	-	Influent concentration < 10xMDL
16122	01	-	03	-	No data
DMQ	01	425	03	-	No effluent data
ANL	01,02,03	10	07	10	Influent concentration < 10xMDL
16132	01,02,03	-	04	-	No data
DMQ	01,02,03	-	04	-	No data

Table 11-10: BPT Facility Data Excluded from the Calculation of Non-Hazardous BPT/BAT Limitations (continued)

BPT Facility	Influent Sample Point	Avg. Influent Conc.	Effluent Sample Point	Avg. Effluent Conc.	Reason for Exclusion
16253	01	-	02	-	No data
DMQ	01	-	02	-	No data
Phenol (ug/L)					
16041	02	-	04	-	No data
DMQ	02	-	04	-	No data
16058	-	-	01	10	Detailed questionnaire data was not used
DMQ	-	-	01	10	No influent data
ANL	01, 02	10	-	-	Influent concentration < 10xMDL
16120	-	-	02	-	No data
16122	01	3,050	03	10	Effluent sample point 03 located after aerated equalization
DMQ	01	-	03	-	
16132	01,02,03	-	04	-	No data
DMQ	01,02,03	-	04	-	No data
16253	01	-	02	-	No data
DMQ	01	-	02	-	No data
Zinc (ug/L)					
16041	02	1,130	04	200	Detailed questionnaire data was not used
16058	-	-	01	10	Detailed questionnaire data was not used
16118	01	380	02	50	Facility wastewater treatment system includes chemical precipitation
DMQ	01	295	02	45	
16120	-	-	02	40	Facility wastewater treatment system includes chemical precipitation
DMQ	01	230	02	37	
ANL	01	85	-	-	
16122	01	212,000	03	16	Facility wastewater treatment system includes chemical precipitation
DMQ	01	805	03	22	
ANL	01,02,03	120	07	12	
16132	01,02,03	575	04	10	Detailed questionnaire data was not used
16253	01	20	02	38	Facility wastewater treatment system includes chemical precipitation
DMQ	01	90	02	50	

ANL: Analytical data
DET: Detailed Questionnaire data
DMQ: Detailed Monitoring Questionnaire data

Table 11-11: BPT/BAT Limitations for the Non-Hazardous Subcategory

Pollutant or Pollutant Property	Maximum for 1 day (mg/L)	Monthly Average Shall Not Exceed (mg/L)
BOD_5	140	37
TSS	88	27
Ammonia	10	4.9
Alpha Terpineol	0.033	0.016
Benzoic Acid	0.12	0.071
P-Cresol	0.025	0.014
Phenol	0.026	0.015
Zinc	0.20	0.11
pH	([1])	([1])

([1]) pH shall be in the range 6.0 - 9.0 pH units.

Table 11-12: National Estimates of Pollutant of Interest Reductions for BPT/BAT Options for Municipal Solid Waste Landfills - Direct Dischargers

		National Estimates			
Pollutant of Interest CAS Number	Pollutant of Interest	Current Discharge Loads (pounds/yr)	BPT/BAT Option I Loads (pounds/yr)	BPT/BAT Option II Loads (pounds/yr)	BAT Option III -RO Loads (pounds/yr)
C-020	TOTAL PHENOLS (CHLOROFORM EXTRACTION)	1,005	166	125	125
C-012	TOTAL ORGANIC CARBON	692,275	352,957	231,875	127,805
C-010	TOTAL DISSOLVED SOLIDS	13,158,362	13,109,304	12,086,905	621,714
C-009	TOTAL SUSPENDED SOLIDS	319,754	195,173	92,491	21,328
C-005	NITRATE/NITRITE	109,494	109,494	109,494	3,527
C-004	CHEMICAL OXYGEN DEMAND	2,364,028	1,597,988	1,497,581	373,389
C-002	BIOCHEMICAL OXYGEN DEMAND	478,004	144,915	105,561	105,561
98555	ALPHA-TERPINEOL	247	53	53	53
95487	O-CRESOL	62	53	53	53
78933	2-BUTANONE	2,846	2,846	2,846	2,846
7664417	AMMONIA NITROGEN	174,382	26,279	16,978	16,978
75092	METHYLENE CHLORIDE	385	385	385	385
7440666	ZINC	857	249	249	249
7440473	CHROMIUM	110	103	103	103
7440393	BARIUM	1,449	926	926	639
7440326	TITANIUM	123	20	20	20
7440246	STRONTIUM	3,404	1,812	1,812	533
68122	N,N-DIMETHYLFORMAMIDE	69	53	53	53
67641	2-PROPANONE	1,642	1,642	1,642	1,642
65850	BENZOIC ACID	350	265	265	265
298044	DISULFOTON	22	22	22	11
20324338	TRIPROPYLENEGLYCOL METHYL ETHER	840	528	528	528
18540299	CHROMIUM (HEXAVALENT)	117	117	117	50
142621	HEXANOIC ACID	9,183	53	53	53
123911	1,4-DIOXANE	55	55	55	55
120365	DICHLOROPROP	16	6	6	5
108952	PHENOL	298	56	56	56
108883	TOLUENE	191	191	191	191
108101	4-METHYL-2-PENTANONE	228	228	228	228
106445	P-CRESOL	151	48	48	48
35822469	1234678-HPCDD	6E-04	3E-04	3E-04	3E-04
3268879	OCDD	7E-03	2E-03	1E-03	1E-03

Table 11-13: National Estimates of Pollutant of Interest Reductions for BPT/BAT Options for Non-Municipal Solid Waste Landfills - Direct Dischargers

		National Estimates			
Pollutant of Interest CAS Number	Pollutant of Interest	Current Discharge Loads (pounds/yr)	BPT/BAT Option I Loads (pounds/yr)	BPT/BAT Option II Loads (pounds/yr)	BAT Option III -RO Loads (pounds/yr)
C-002	Biochemical Oxygen Demand	24,492	24,492	24,492	24,492
C-004	Chemical Oxygen Demand	5,633,111	1,033,662	907,417	147,359
C-009	Total Suspended Solids	22,451	22,451	22,451	8,164
C-005	Nitrate/Nitrite	73,475	1,939	1,939	1,359
C-020	Total Phenols	241	78	53	53
C-012	Total Organic Carbon	55,107	55,107	55,107	51,025
C-010	Total Dissolved Solids	69,189,296	13,878,575	6,385,329	339,723
7664417	Ammonia as Nitrogen	153,074	11,062	6,994	6,994
7440246	Strontium	61,229	54,494	54,494	204

Table 11-14: Annual Pollutant Discharge Before and After the Implementation of BPT for Subtitle D Municipal Solid Waste Landfill Facilities in the Non-Hazardous Subcategory

Pollutant Group	Current Annual Pollutant Discharge (pounds)	Annual Pollutant Discharge After Implementation of BPT (pounds)	Annual Amount of Pollutants Removed by BPT (pounds)
Conventional Pollutants[1]	800,000	200,000	600,000
Nonconventional Pollutants[2]	16,500,000	13,950,000	2,550,000
Metal Pollutants[3]	6,000	3,200	2,800
Organic Pollutants[4]	16,500	6,500	10,000 [7]
Pesticides[5]	40	29	11
Dioxins/ Furans[6]	0.0075	0.0013	0.0062

(1) Includes BOD_5 and TSS
(2) Includes ammonia, COD, TDS, TOC, total phenols, and nitrate/nitrite
(3) Includes barium, chromium, hexavalent chromium, strontium, titanium, and zinc
(4) Includes alpha terpineol, benzoic acid, hexanoic acid, N,N-Dimethylformamide, o-cresol, p-cresol, phenol, tripropyleneglycol methyl ether, methylene chloride, 1,4 dioxane, 2-butanone, 2-propanone, 4-methyl-2-pentanone, and toluene
(5) Includes dichloroprop and disulfoton
(6) Includes OCDD and 1,2,3,4,6,7,8-HpCDD
(7) EPA did not include the removal of the following volatile organic compounds: methylene chloride, 1,4 dioxane, 2-butanone, 2-propanone, 4-methyl 2-pentanone, and toluene

Table 11-15: Annual Pollutant Discharge Before and After The Implementation of BPT for Subtitle D Non-Municipal Solid Waste Landfill Facilities in the Non-Hazardous Subcategory

Pollutant Group	Current Annual Pollutant Discharge (pounds)	Annual Pollutant Discharge After Implementation of BPT (pounds)	Annual Amount of Pollutants Removed by BPT (pounds)
Conventional Pollutants (1)	47,000	47,000	0
Nonconventional Pollutants (2)	75,100,000	7,350,000	67,750,000
Metal Pollutants (3)	61,200	54,500	6,700

(1) Includes BOD_5 and TSS. Both facilities in the database were already in compliance with the BOD_5 and TSS limits.
(2) Includes ammonia, nitrate/nitrite, TDS, TOC, total phenol, and COD.
(3) Includes strontium - the only metal pollutant of interest for non-municipal solid waste landfills.

Table 11-16: Selected BPT Facilities for the Hazardous Subcategory

Detailed Questionnaire ID Number	Discharge Status	Treatment in Place
16041	Indirect	sequential batch reactor
16087	Indirect	stirred equalization, chemical precipitation, flocculation, neutralization, clarification, activated sludge, chemical oxidation

Table 11-17: Facilities and Sample Points Used for the Development of BPT/BAT Effluent Limitations for the Hazardous Subcategory

BPT Facility	Data Source	Influent Sample Point	Avg. Influent Concentration	Effluent Sample Point	Avg. Effluent Concentration
Ammonia (mg/L)					
16041	DMQ ANL	02 01, 03, 05, 06	679 475	04 02	5.4 1.4
16122	ANL	01, 02, 03	181	07	1.2
16132	DMQ	01, 02, 03	206	04	5.9
BOD_5 (mg/L)					
16041	ANL	01, 03, 05, 06	910	02	47
16058	DMQ ANL	- 01, 02	- 153	01 -	29.7 Only influent conc. used
16087	DMQ	01	2,929	05	29
16118	DMQ	01	1,890	02	45.5
16120	DMQ	01	780	02	4.6
16122	ANL	01, 02, 03	1,007	07	35.2
16132	DMQ	01, 02, 03	4,740	04	15.8
16253	DMQ	01	159	02	6.4
TSS (mg/L)					
16120	DMQ	01	1,240	02	13.6
16253	DMQ	01	120	02	24.9
Alpha Terpineol (ug/L)					
16041	ANL	01, 03, 05, 06	653	02	10
Aniline (ug/L)					
16041	ANL	01, 03, 05, 06	1,060	02	10
16087	ANL	01	533	03	10
Benzoic Acid (ug/L)					
16041	ANL	01, 03, 05, 06	15,400	02	50
16087	ANL	01	64,957	03	50

Table 11-17: Facilities and Sample Points Used for the Development of BPT/BAT Effluent Limitations for the Hazardous Subcategory (continued)

Pollutant	BPT Facility	Influent Sample Point	Avg. Influent Concentration	Effluent Sample Point	Avg. Effluent Concentration
Naphthalene (ug/L)					
16041	ANL	01, 03, 05, 06	645	02	10
P-Cresol (ug/L)					
16041	ANL	01, 03, 05, 06	1,360	02	10
16087	ANL	01	5,022	03	10
Phenol (ug/L)					
16041	ANL	01, 03, 05, 06	5,120	02	10
16087	ANL	01	65,417	03	29.7
Pyridine (ug/L)					
16087	ANL	01	301	03	10
Arsenic (ug/L)					
16087	DMQ ANL	01 01	1,400 584	05 03	325 312
Chromium (ug/L)					
16087	DMQ ANL	01 01	730 415	05 03	312 82
Zinc (ug/L)					
16041	DMQ ANL	02 01, 03, 05, 06	505 310	04 02	214 85
16087	DMQ	01	550	05	380

ANL: Analytical data
DET: Detailed Questionnaire data
DMQ: Detailed Monitoring Questionnaire data

Table 11-18: BPT Facility Data Excluded from the Calculation of Hazardous BPT/BAT Limitations

BPT Facility	Influent Sample Point	Avg. Influent Conc.	Effluent Sample Point	Avg. Effluent Conc.	Reason for Exclusion
BOD_5 (Transferred from the Non-Hazardous subcategory) (mg/L)					
16041 DMQ	02 02	- -	04 04	- -	No data No data
16058	-	-	01	22	Detailed questionnaire data was not used
16087 ANL	01 01	2,980 3,721	03 03	258 66	Effluent concentration above 50 mg/L Effluent concentration above 50 mg/L
16118	01	2,200	02	49	Detailed questionnaire data was not used
16120 ANL	- 01	- 1,290	02 -	15.9 -	Detailed questionnaire data was not used No effluent data.
16122 DMQ	01 01	- -	03 03	5.3 5.4	Effluent sample point 03 located after aerated equalization
16132	01,02,03	5,581	04	7	Detailed questionnaire data was not used
16253	01	1,000	02	5.2	Detailed questionnaire data was not used
TSS (Transferred from the Non-Hazardous subcategory) (mg/L)					
16041 DMQ ANL	02 02 1, 3, 5, 6	364 307 70	04 04 02	36 35 46	Facility wastewater treatment system does not employ filtration
16058 DMQ ANL	- - 01, 02	- - 14,470	01 01 -	216 188 -	Facility wastewater treatment system does not employ filtration
16087 DMQ ANL	01 01 01	586 579 172	03 05 03	51 114 78	Facility wastewater treatment system does not employ filtration
16118 DMQ	01 01	- -	02 02	- -	Facility wastewater treatment system does not employ filtration
16120 ANL	- 01	- 200	02 -	14 -	Detailed questionnaire data was not used No effluent data
16122 DMQ ANL	01 01 01,02,03	- - 267	03 03 07	5.6 5.4 12.5	Facility eliminated due to settling that can occur in equalization tanks prior to filtration
16132 DMQ	01,02,03 01,02,03	244 -	04 04	39 47	Facility wastewater treatment system does not employ filtration
16253	01	180	02	17.5	Detailed questionnaire data was not used

Table 11-18: BPT Facility Data Excluded from the Calculation of Hazardous BPT/BAT Limitations (continued)

BPT Facility	Influent Sample Point	Avg. Influent Conc.	Effluent Sample Point	Avg. Effluent Conc.	Reason for Exclusion
Ammonia (Transferred from the Non-Hazardous subcategory) (mg/L)					
16041	02	554	04	5.0	Detailed questionnaire data was not used
16058 DMQ ANL	- - 01, 02	- - 2,900	01 01 -	- - -	No data No data No effluent data
16087 DMQ ANL	01 01 01	- - 209	03 05 03	- - 153	No data No data Minimal ammonia removal
16118 DMQ	01 01	- -	02 02	- -	No data No data
16120 DMQ ANL	- 01 01	- 362 245	02 02 -	1.35 5.98 -	Facility wastewater treatment system employed an air stripper
16122 DMQ	01 01	136 135	03 03	0.87 0.48	Effluent sample point 03 located after aerated equalization
16132	01,02,03	-	04	-	No data
16253 DMQ	01 01	- -	02 02	- 0.01	No data No influent data
Alpha Terpineol (ug/L)					
16041 DMQ	02 02	- -	04 04	- -	No data No data
16087 DMQ ANL	01 01 01	- - 10	03 05 03	- - 10	No data No data Influent concentration < 10xMDL
Aniline (ug/L)					
16041 DMQ	02 02	- -	04 04	- -	No data No data
16087 DMQ	01 01	- -	03 05	- -	No data No data
Benzoic Acid (ug/L)					
16041 DMQ	02 02	- -	04 04	- -	No data No data
16087 DMQ	01 01	- -	03 05	- -	No data No data

Table 11-18: BPT Facility Data Excluded from the Calculation of Hazardous BPT/BAT Limitations (continued)

BPT Facility	Influent Sample Point	Avg. Influent Conc.	Effluent Sample Point	Avg. Effluent Conc.	Reason for Exclusion
Naphthalene (ug/L)					
16041	02	-	04	-	No data
DMQ	02	-	04	-	No data
16087	01	-	03	-	No data
DMQ	01	-	05	-	No data
ANL	01	25	03	10	Influent concentration < 10xMDL
P-Cresol (ug/L)					
16041	02	-	04	-	No data
DMQ	02	-	04	-	No data
16087	01	-	03	-	No data
DMQ	01	-	05	-	No data
Phenol (ug/L)					
16041	02	-	04	-	No data
DMQ	02	-	04	-	No data
16087	01	98,500	03	814	Detailed questionnaire data was not used
DMQ		-	05	-	No data
Pyridine (ug/L)					
16041	02	-	04	-	No data
DMQ	02	-	04	-	No data
ANL	1, 3, 5, 6	23	02	10	Influent concentration < 10xMDL
16087	01	-	03	-	No data
DMQ	01	-	05	-	No data

Table 11-18: BPT Facility Data Excluded from the Calculation of Hazardous BPT/BAT Limitations (continued)

BPT Facility	Influent Sample Point	Avg. Influent Conc.	Effluent Sample Point	Avg. Effluent Conc.	Reason for Exclusion
Arsenic (ug/L)					
16041	02	-	04	-	No data
DMQ	02	-	04	-	No data
ANL	1, 3, 5, 6	535	02	569	Negative percent removal
16087	01	1,420	03	193	Detailed questionnaire data was not used
Chromium (ug/L)					
16041	02	210	04	120	Detailed Questionnaire data was not used
DMQ	02	419	04	417	No removal
ANL	1, 3, 5, 6	82	02	46	Influent concentration < 10xMDL
16087	01	731	03	501	Detailed questionnaire data was not used
Zinc (ug/L)					
16041	02	1,130	04	200	Detailed questionnaire data was not used
16087	01	560	03	279	Detailed questionnaire data was not used
ANL	01	126	03	52	Influent concentration < 10xMDL

ANL: Analytical data
DET: Detailed Questionnaire data
DMQ: Detailed Monitoring Questionnaire data

Table 11-19**:** BPT/BAT Limitations for the Hazardous Subcategory

Pollutant or Pollutant Property	Maximum for 1 day (mg/L)	Monthly Average Shall Not Exceed (mg/L)
BOD_5	220	56
TSS	88	27
Ammonia	10	4.9
Alpha Terpineol	0.042	0.019
Aniline	0.024	0.015
Benzoic Acid	0.119	0.073
Naphthalene	0.059	0.022
P-Cresol	0.024	0.015
Phenol	0.048	0.029
Pyridine	0.072	0.025
Arsenic	1.1	0.54
Chromium	1.1	0.46
Zinc	0.535	0.296
pH	([1])	([1])

([1]) pH shall be in the range 6.0 - 9.0 pH units.

Table 11-20: Comparison of Long-Term Averages for Nonconventional and Toxic Pollutants Regulated Under BAT for the Non-Hazardous Subcategory

Pollutant	BPT Option II: Equalization + Biological + Multimedia Filter (mg/L)	Reverse Osmosis single stage effluent (mg/L)	Reverse Osmosis second stage effluent (mg/L)
Ammonia	5.4	13	0.59
Alpha Terpineol	0.010 ND	0.010 ND	0.010 ND
Benzoic Acid	0.050 ND	0.093	0.050 ND
P-Cresol	0.010 ND	0.253	0.022
Phenol	0.010 ND	0.185	0.029

ND: Non-detect

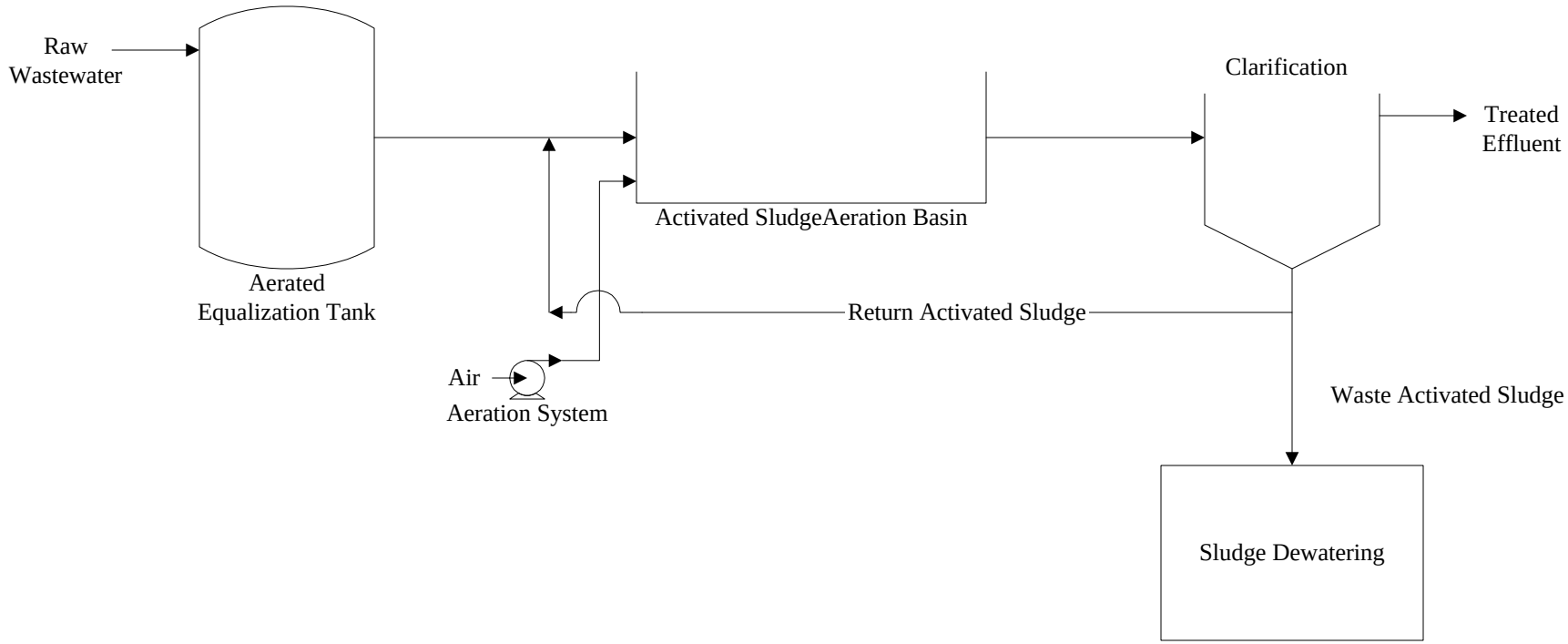


Figure 11-1: BPT/BCT/BAT/PSES/PSNS Non-Hazardous Subcategory Option I Flow Diagram

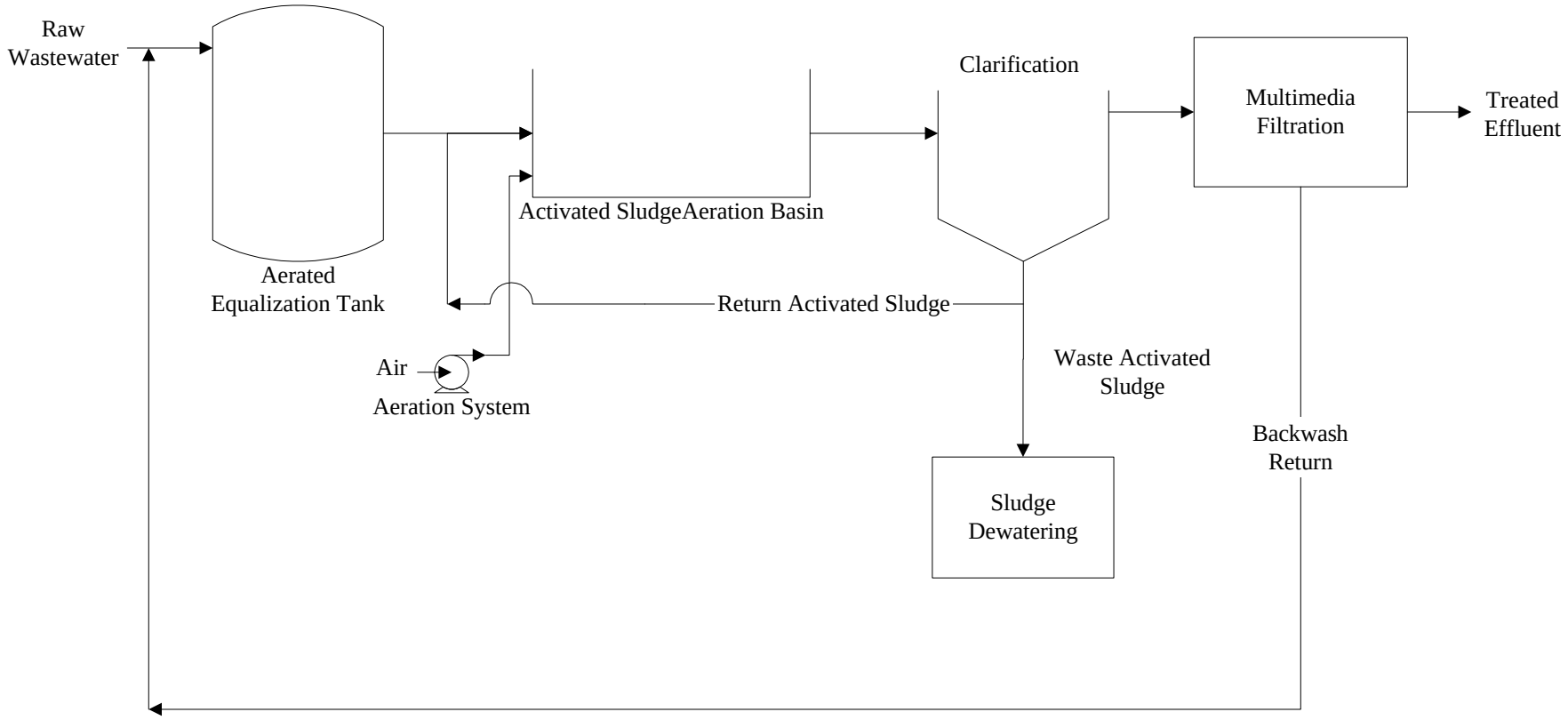


Figure 11-2: BPT/BCT/BAT Non-Hazardous Subcategory Option II & NSPS Flow Diagram

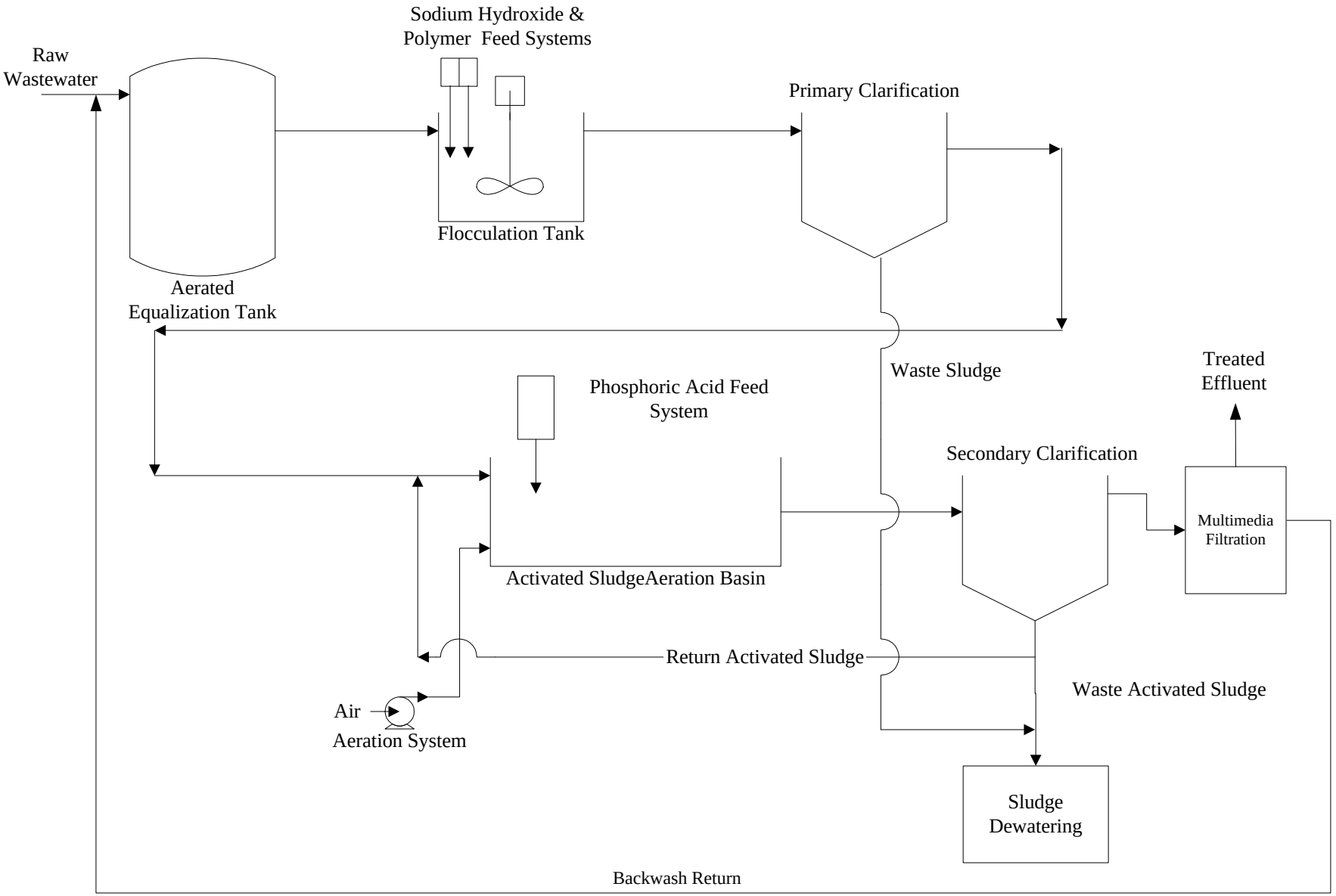


Figure 11-3: BPT/BCT/BAT Hazardous Subcategory Option II & NSPS Flow Diagram

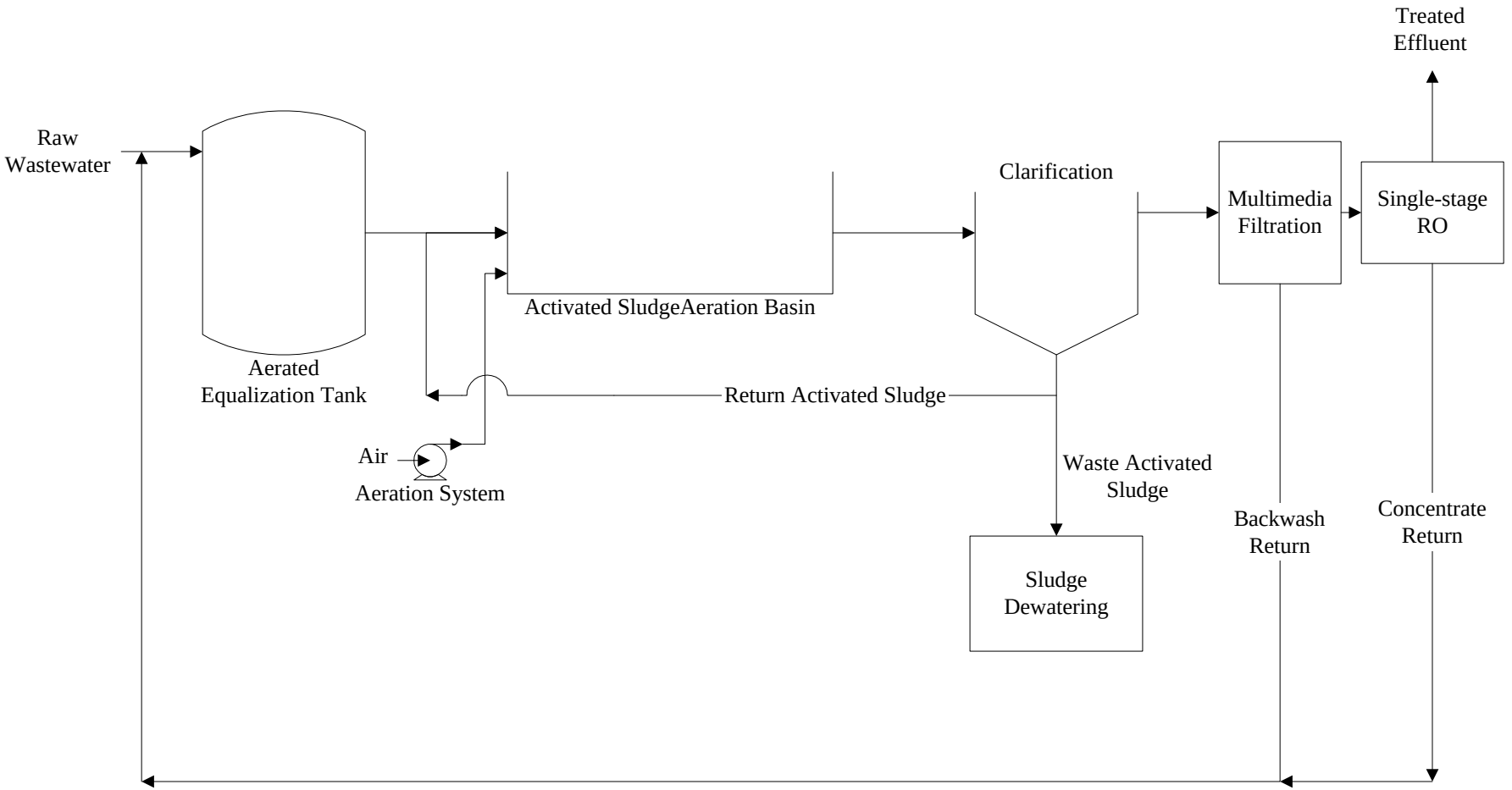


Figure 11-4: BAT Non-Hazardous Subcategory Option III Flow Diagram

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APPENDIX A:

SECTION 308 SURVEY FOR LANDFILLS - INDUSTRY POPULATION ANALYSIS

Appendix A: Section 308 Survey for Landfills-Industry Population Analysis

The list of landfills needed to define the landfill population in the United States was developed from various sources: state environmental and solid waste departments, and other state contacts; the National Survey of Hazardous Waste Treatment Storage, Disposal, and Recycling Facilities respondent list; Environmental Ltd.'s 1991 Directory of Industrial and Hazardous Waste Management Firms; the Resource Conservation and Recovery Act (RCRA) 1992 list of Municipal Solid Waste Landfills; and the Resource Conservation and Recovery Information System (RCRIS) National Oversight Database.

The information provided by state environmental departments was requested during early stages of the rulemaking process for Centralized Waste Treatment and represented 1987-88 data for both active and inactive landfills. This information was incomplete to some extent. For 18 of the 50 states only limited or no information was available. Hence, these states were contacted during the data gathering effort for the development of effluent guidelines and standards for Landfills and Incinerators to obtain updated lists of landfills and wastewater collection information.

The duplication of landfill entries among various sources was eliminated as far as possible by cross checking using computer programs. However, some duplication in Subtitle D landfills is inevitable as some of the various identifiers were unclear.

Landfill population was divided into two categories: Subtitle C (hazardous waste) and Subtitle D (non-hazardous waste). In total, mailing addresses were compiled for 595 Subtitle C landfills and 9,882 Subtitle D landfills. In addition, 448 Subtitle D landfills were identified for which addresses were inadequate for delivery. Thus the population of Subtitle D amounted to 10,330. Table 1 provides a list of the number of landfills with deliverable mailing addresses in each state by category.

Selection of the landfills to survey

From the identified landfill population of 10,925 Subtitle C and D facilities, screener surveys were mailed to 4996. Facilities receiving the screener survey included all of the 595 Subtitle C landfills and a sample of the 9,882 Subtitle D facilities with mailable addresses.

TABLE 1. COUNT OF LANDFILLS WITH MAILABLE ENTRIES IN EACH STATE

State	Subtitle-D	Subtitle-C	Total
Alabama	238	38	276
Alaska	201	1	202
Arizona	90	2	92
Arkansas	134	3	137
California	630	16	646
Colorado	216	12	228
Connecti-cut	125	22	147
Delaware	8	14	22
Florida	91	9	100
Georgia	277	17	294
Hawaii	15	1	16
Idaho	112	6	118
Illinois	182	14	196
Indiana	101	29	130
Iowa	118	13	131
Kansas	118	8	126
Kentucky	121	33	154
Louisiana	73	17	90

State	Subtitle-D	Subtitle-C	Total
Maine	291	2	293
Maryland	50	5	55
Massachu-setts	722	1	723
Michigan	762	9	771
Minnesota	257	4	261
Mississippi	97	3	100
Missouri	128	7	135
Montana	257	1	258
Nebraska	41	8	49
Nevada	127	3	130
New Hapmshire	58	0	58
New Jersey	467	8	475
New Mexico	121	7	128
New York	565	10	575
North Carolina	244	39	283
North Dakota	85	1	86
Ohio	119	24	143
Oklahoma	189	7	196
Oregon	231	10	241
Pennsyl-vania	41	22	63
Rhode Island	12	0	12
South Carolina	127	9	136

State	Subtitle-D	Subtitle-C	Total
South Dakota	193	0	193
Tennessee	112	9	121
Texas	601	70	671
Utah	92	7	99
Vermont	73	0	73
Virginia	440	8	448
Washing-ton	72	9	81
West Virginia	57	5	62
Wisconsin	183	3	186
Wyoming	218	45	263
Puerto Rico	0	3	3
Guam	0	1	1
Total	9882	595	10477

The remaining 4401 screener surveys were sent to Subtitle D landfills. A statistical approach was taken to sample the 9882 deliverable Subtitle D facilities. For sampling purposes, the 9882 Subtitle D landfills were stratified into three categories:

1) landfills with known wastewater collection
2) landfills from states with fewer than 100 landfills and
3) landfills from states with more than 100 landfills.

All landfills with known wastewater collection were included in the landfill survey sample. The population included 134 landfills with known wastewater collection (1.35%).

Landfills in states with fewer than 100 landfills were stratified from the landfills in states with more than 100 landfills. This was simply a sampling technique for random sampling and was done to ensure the inclusion of a representative number of facilities from each stratum.

There were 16 states with under 100 landfills each (after exclusion of known wastewater collectors), which accounted for 892 landfills. A screener survey was mailed to each of these 892 landfills. The remaining 24 states, with over 100 landfills each, accounted for 8856 landfills. A random sample of 3375 was taken from this strata, and a screener survey was mailed to each of these randomly selected landfills. Table 2 summarizes the stratification.

Screener surveys were distributed by both Federal Express and U.S. certified mail: 1916 surveys were sent via Federal Express, which resulted in 94% receipt confirmation; 3080 surveys were sent via U.S. certified mail, which resulted in 92% receipt confirmation. Twenty three additional screener surveys were mailed because of change of ownership, or different mailing address, even though the physical location of the landfill remained same. A summary of analysis on these additional surveys is presented in Table 3. Thus, a total of 5020 landfill screener surveys were distributed.

TABLE 2. SUMMARY OF STRATIFICATION

Strata #	Population	# in frame	# in sample
1	Subtitle C	595	595
2	Subtitle D -known wastewater generators	134	134
3	Subtitle D - states with ≤ 100 landfills	892	892
4	Subtitle D - states with >100 landfills	8856	3375
	Total	10477	4996

A completed screener survey was received from 3628 landfills excluding the late arrivals. This includes response from a pre-test screener survey. The status of remaining screener surveys is:

- 353 surveys were deemed non-deliverables due to incorrect/non-traceable addresses and were returned to the sender
- 1008 landfills were presumed to be non-respondents
- 4 landfills were found to be out-of-business
- 26 landfills were declared ineligible to participate in the survey for reasons discovered during the mid-point remainder calls
- 1 respondent refused to respond to the survey.

For statistical analysis purposes, screener surveys in each of the above categories were traced back to the respective strata. Table 4 presents a breakdown of these remaining screener surveys by strata.

TABLE 3. SUMMARY OF ADDITIONAL SCREENER SURVEY ANALYSIS

Screener ID	Original ID	Stratum	Reason for re-assignment
15100	13235	4	screener sent to former owner or incorrect address
15101	14044	4	screener sent to former owner or incorrect address
15102	13876	4	screener sent to former owner or incorrect address
15103	11594	4	screener sent to former owner or incorrect address
15104	14117	4	screener sent to former owner or incorrect address
15105	13953	4	screener sent to former owner or incorrect address
15106	13264	4	screener sent to former owner or incorrect address
15107	10985	4	additional screener resp. was obtained for a new landfill
15108	14449	4	additional screener resp. was obtained for a new landfill
15109	12167	1	additional screener resp. was obtained for a new landfill
15110	12883	4	additional screener resp. was obtained for a new landfill
15111			response transferred from pre-test screener survey
15112	14112	4	screener sent to former owner or incorrect address
15113	11319	3	screener sent to former owner or incorrect address
15114	12327	4	screener sent to former owner or incorrect address
15116	11528	4	screener sent to former owner or incorrect address
15117	13389	3	screener sent to former owner or incorrect address
15118	13995	4	screener sent to former owner or incorrect address
15119	14779	4	screener sent to former owner or incorrect address
15120	11422	4	screener sent to former owner or incorrect address
15121	13976	4	screener sent to former owner or incorrect address
15122	12422	1	screener sent to former owner or incorrect address
15123	11299	4	screener sent to former owner or incorrect address
15124	10851	4	screener sent to former owner or incorrect address

Among the 3628 survey responses received, a total of 3581 surveys were sent to data entry; 44 were declared ineligible upon reviewing their response, and were not processed any further; 3 remained incomplete because of unsuccessful attempts to contact the respondents to complete the review process. A total of 859 respondents were found collecting some type of wastewater (landfills collecting only storm water were not included) generated from their landfill operations, and were considered as in scope population from which a sample of facilities will be selected to receive the detailed Section 308 landfill questionnaire. The rest of the surveys sent to data entry were considered out of scope. For statistical analysis purposes, screener surveys not sent to data entry, the out of scope surveys, and the in scope surveys were traced back to the respective strata, and a count of these in each strata is presented in Table 4.

A response bias query was conducted on about 5.65% (57 landfills) of the 1008 presumed non-respondents. Each of these 57 randomly-selected landfills was called to discern the reasons that the screener survey was not received. The result of this effort is as follows:

- 25 facility contacts said that they over looked/misplaced/forgotten the survey (1 in stratum 2; 1 in stratum 3; and 23 in stratum 4)

- 19 facility contacts said that they did not recall receiving any survey (2 in stratum 1; 3 in stratum 3; and 14 in stratum 4)

- 7 facility contacts said that they did not feel it was applicable to them (1 in stratum 1; 2 in stratum 3; and 4 in stratum 4)

- 3 facility contacts said that they forgot and would complete the survey and return (2 in stratum 3; and 1 in stratum 4)

- 2 facility contacts said that they received duplicate surveys, and this was checked and found correct (these 2 are in stratum 4)

- 1 facility contact said that they are under bankruptcy proceedings (this is in stratum 1).

A total of 39 landfill screener survey responses were received past the deadline. since these were received after the close of the screener survey database, they were not considered for any further analyses. Among these 39 late arrivals, only four landfills collected wastewater generated from landfill operations (landfill leachate and contaminated groundwater), and none of these four landfills have any on-site treatment. Additional information on these four landfills is: two were municipal, non-commercial, and discharged untreated wastewater to a Publicly Owned Treatment Works (POTW); one was government, commercial, and discharged untreated wastewater to a POTW; one was private and sent their wastewaters for off-site disposal.

Questionnaire distribution

A total of 859 landfill operators reported that they collect one or more type of wastewater generated from the landfill operations (landfills collecting only storm water were not included). These landfills were considered as the sample frame to receive the Section 308 questionnaire for landfills. Facilities with treatment were targeted most heavily, while some facilities without treatment but collect wastewater were randomly selected to receive only Section A of the questionnaire. The facilities selected fall into any of the following eight categories:

1. Commercial private, municipal, or government facilities which have wastewater treatment and are direct or indirect dischargers. A census was conducted of this part of the industry.

2. Commercial private, municipal, or government facilities which have wastewater treatment and are zero dischargers (do not discharge to surface water or to a POTW). Approximately 25% of these were randomly chosen to receive the questionnaire.

3. Non-commercial private facilities with wastewater treatment. Approximately 40% of these were randomly chosen to receive the questionnaire.

4. Facilities with no wastewater treatment. Approximately 10% of these were randomly chosen to receive only Section A of the questionnaire.

5. Commercial facilities who accept PCB wastes. Only one facility was in this category, and was chosen.

6. Municipal hazardous waste landfills. There were two facilities in this category, and a census was conducted of this part of the industry.

7. Small business with no wastewater treatment. A census was conducted of this part of the industry.

8. Pre-test facility which was not in the screener population. Only one facility was in this category, and was chosen based on knowledge of the industry and professional judgement.

For statistical analysis purposes, the facilities in each of the aforementioned categories were traced back through their screener surveys to the respective strata, and a count of these in each strata is presented in Table 5.

Section 308 Questionnaires were sent to a total of 252 mailing addresses that were considered in scope from their screener responses. The questionnaire response was received from 248 landfills. The remaining four landfills were presumed to be non-respondents. The questionnaire responses received included four responses from pre-test questionnaires. Thus a total of 248 responses were available for further review.

Among the survey responses obtained, 22 were declared out of scope upon reviewing their response and were not processed any further; 226 were reviewed for completeness and technical accuracy and

were entered into the landfill questionnaire database. For statistical analysis purposes, the 252 questionnaires that were sent, including the 226 questionnaires reviewed and placed in the database, were traced back to the original screener population strata, and a count of these in each strata is presented in Table 4.

TABLE 4. COUNT OF SCREENER SURVEYS IN EACH CATEGORY BY STRATA[1]

Category	Stratum 1	Stratum 2	Stratum 3	Stratum 4	Total
Non-respondents	69	15	170	755	1009
Ineligible[2]	79	9	45	294	427
Incomplete	2	0	1	0	3
In scope	141	91	222	405	859
Out of scope	305	20	456	1941	2722
Quest. recipients	51	35	77	88	252[3]
Quest. in database	46	32	71	76	226[3]
Quest. out of scope	4	3	4	11	22
Quest. non-response	1	0	2	1	4

[1]For each of the category presented below, a list of Survey ID numbers and their respective strata # is presented in Appendix A.

[2]This includes all non-deliverables, out-of-business, and duplicate addresses.

[3]An additional one is the pre-test questionnaire, which is not part of any stratum.

TABLE 5. QUESTIONNAIRE SELECTION BY CATEGORY

Category	Stratum 1	Stratum 2	Stratum 3	Stratum 4	Total
Pri/com/muni/govt./with treat/D-I discharge	12	27	51	38	128
Pri/non-com/with treatment	30	2	3	7	42
Pri/com/muni/govt./with treat/Zero discharge	1	0	7	0	8
No treatment	5	6	14	38	63
PCB facilities with treatment	0	0	1	0	1
Municipal/hazardous	2	0	0	0	2
Small business/no treatment	1	0	1	5	7
Pre-test not in Screener population[4]	-	-	-	-	1
Totals	51	35	77	88	252

[4]This is a pre-test questionnaire and is not in any stratum because, it was not in the screener database.

TABLE 6. IN SCOPE SCREENERS NOT SELECTED FOR QUESTIONNAIRE BY CATEGORY

Category	Stratum 1	Stratum 2	Stratum 3	Stratum 4	Total
Pri/com/muni/govt./with treat/D-I discharge	0	0	0	0	0
Pri/non-com/with treatment	31	0	6	27	64
Pri/com/muni/govt./with treat/Zero discharge	7	2	9	7	25
No treatment	52	54	130	283	519
PCB facilities with treatment	0	0	0	0	0
Municipal/hazardous	0	0	0	0	0
Small business/no treatment	0	0	0	0	0
Totals	90	56	145	317	608

APPENDIX B:

DEFINITIONS,
ACRONYMS, AND ABBREVIATIONS

APPENDIX B: DEFINITIONS, ACRONYMS, AND ABBREVIATIONS

ADMINISTRATOR: The Administrator of the U.S. Environmental Protection Agency.

AGENCY: The U.S. Environmental Protection Agency.

AVERAGE MASTER FILE: A method of calculating the average raw wastewater concentration for each pollutant of interest in a subcategory. The Average Master File was calculated using all available data collected in the Landfills industry study.

BASELINE FLOW: Estimated wastewater discharge flow rate for a selected facility in 1992 based on their Detailed Questionnaire response.

BAT: The best available technology economically achievable, applicable to effluent limitations to be achieved by July 1, 1984, for industrial discharges to surface waters, as defined by Sec. 304(b)(2)(B) of the CWA.

BCT: The best conventional pollutant control technology, applicable to discharges of conventional pollutants from existing industrial point sources, as defined by Sec. 304(b)(4) of the CWA.

BOD_5: Biochemical oxygen demand - Five Day. A measure of the biochemical decomposition of organic matter in a water sample. It is determined by measuring the dissolved oxygen consumed by microorganisms to oxidize the organic contaminants in a water sample under standard laboratory conditions of five days and 70 degrees Celsius. BOD_5 is not related to the oxygen requirements in chemical combustion.

BPT: The best practicable control technology currently available, applicable to effluent limitations to be achieved by July 1, 1977, for industrial discharges to surface waters, as defined by Sec. 304(b)(1) of the CWA.

CAPDET: Computer-Assisted Procedure for the Design and Evaluation of Wastewater Treatment Systems. Developed by the U.S. Army Corp. of Engineers, CAPDET is intended to provide planning level cost estimates to analyze alternate design technologies for wastewater treatment systems.

CAPTIVE: Used to describe a landfill that is directly associated with an industrial or commercial operation. See Chapter 2 for the conditions that a captive landfill must meet in order to be excluded from the landfill effluent guideline.

CELL: An area of a landfill that is separated from other areas by an impervious structure. Each cell has a separate leachate collection system or would require a separate leachate collection system if one were installed. Individual leachate collection systems that are combined at the surface are considered separate systems by this definition.

CLEAN WATER ACT (CWA): The Federal Water Pollution Control Act Amendments of 1972 (33 U.S.C. Section 1251 et seq.), as amended by the Clean Water Act of 1977 (Pub. L. 95-217), and the Water Quality Act of 1987 (Pub. L. 100-4).

CLEAN WATER ACT (CWA) SECTION 308 QUESTIONNAIRE:
A questionnaire sent to facilities under the authority of Section 308 of the CWA, which requests information to be used in the development of national effluent guidelines and standards.

CLOSED: A facility or portion thereof that is currently not receiving or accepting wastes and has undergone final closure.

COMMERCIAL FACILITY: A facility that treats, disposes, or recycles/recovers the wastes of other facilities not under the same ownership as this facility. Commercial operations are usually made available for a fee or other remuneration. Commercial waste treatment, disposal, or recycling/recovery does not have to be the primary activity at a facility for an operation or unit to be considered "commercial".

CONTAMINATED GROUND WATER: Water below the land surface in the zone of saturation which has been contaminated by landfill leachate. Contaminated ground water occurs at landfills without liners or at facilities that have released contaminants from a liner system. Ground water may also become contaminated if the water table rises to a point where it infiltrates the landfill or the leachate collection system.

CONTAMINATED STORM WATER: Storm water which comes in direct contact with landfill wastes, the waste handling and treatment areas, or wastewater that is subject to the limitations and standards. Some specific areas of a landfill that may produce contaminated storm water include (but are not limited to): the open face of an active landfill with exposed waste (no cover added); the areas around wastewater treatment operations; trucks, equipment or machinery that has been in direct contact with the waste; and waste dumping areas.

CONVENTIONAL POLLUTANTS: Constituents of wastewater as determined by Sec. 304(a)(4) of the CWA, including pollutants classified as biochemical oxygen demand, total suspended solids, oil and grease, fecal coliform, and pH.

DEEP WELL INJECTION: Disposal of wastewater into a deep well such that a porous, permeable formation of a larger area and thickness is available at sufficient depth to ensure continued, permanent storage.

DETAILED MONITORING QUESTIONNAIRE (DMQ): Questionnaires sent to collect monitoring data from 27 selected landfill facilities based on responses to the Section 308 Questionnaire.

DIRECT DISCHARGER: A facility that discharges or may discharge treated or untreated wastewater into waters of the United States.

DRAINED FREE LIQUIDS: Aqueous wastes drained from waste containers (e.g., drums, etc.) prior to landfilling. Landfills which accept containerized waste may generate this type of wastewater.

EFFLUENT LIMITATION: Any restriction, including schedules of compliance, established by a State or the Administrator on quantities, rates, and concentrations of chemical, physical, biological, and other constituents

which are discharged from point sources into navigable waters, the waters of the contiguous zone, or the ocean. (CWA Sections 301(b) and 304(b)).

EPA: The U.S. Environmental Protection Agency.

EXISTING SOURCE: Any facility from which there is or may be a discharge of pollutants, the construction of which is commenced before the publication of the proposed regulations prescribing a standard of performance under Sec. 306 of the CWA.

FACILITY: All contiguous property owned, operated, leased or under the control of the same person or entity.

GAS CONDENSATE: A liquid which has condensed in the landfill gas collection system during the extraction of gas from within the landfill. Gases such as methane and carbon dioxide are generated due to microbial activity within the landfill, and must be removed to avoid hazardous conditions.

GROUND WATER: The body of water that is retained in the saturated zone which tends to move by hydraulic gradient to lower levels.

HAZARDOUS SUBCATEGORY: For the purposes of this guideline, Hazardous subcategory refers to all landfills regulated under Subtitle C of RCRA.

HAZARDOUS WASTE: Any waste, including wastewater, defined as hazardous under RCRA (40 CFR 261.3).

INACTIVE: A facility or portion thereof that is currently not treating, disposing, or recycling/recovering wastes.

INDIRECT DISCHARGER: A facility that discharges or may discharge wastewater into a publicly-owned treatment works (POTW).

INTRA-COMPANY: A facility that treats, disposes, or recycles/recovers wastes generated by off-site facilities under the same corporate ownership. The facility may also treat on-site generated wastes.

LANDFILL: An area of land or an excavation in which wastes are placed for permanent disposal, that is not a land application or land treatment unit, surface impoundment, underground injection well, waste pile, salt dome formation, a salt bed formation, an underground mine or a cave.

LANDFILL GENERATED WASTEWATER: Wastewater generated by landfill activities and collected for treatment, discharge or reuse, include: leachate, contaminated ground water, storm water runoff, landfill gas condensate, truck/equipment washwater, drained free liquids, floor washings, and wastewater from recovering pumping wells.

LEACHATE: Leachate is a liquid that has passed through or emerged from solid waste and contains soluble, suspended, or miscible materials removed from such waste. Leachate is typically collected from a liner system

above which waste is placed for disposal. Leachate may also be collected through the use of slurry walls, trenches or other containment systems.

LEACHATE COLLECTION SYSTEM: The purpose of a leachate collection system is to collect leachate for treatment or alternative disposal and to reduce the depths of leachate buildup or level of saturation over the low permeability liner.

LINER: The liner is a low permeability material or combination of materials placed at the base of a landfill to reduce the discharge to the underlying or surrounding hydrogeologic environment. The liner is designed as a barrier to intercept leachate and to direct it to a leachate collection system.

LONG-TERM AVERAGE (LTA): For purposes of the effluent guidelines, average pollutant levels achieved over a period of time by a facility, subcategory, or technology option. LTAs are used in developing the limitations and standards in the landfill regulation.

NATIONAL POLLUTANT DISCHARGE ELIMINATION SYSTEM (NPDES) PERMIT:
A permit to discharge wastewater into waters of the United States issued under the National Pollutant Discharge Elimination System, authorized by Section 402 of the CWA.

NEW SOURCE: As defined in 40 CFR 122.2, 122.29, and 403.3 (k), a new source is any building, structure, facility, or installation from which there is or may be a discharge of pollutants, the construction of which commenced (1) for purposes of compliance with New Source Performance Standards (NSPS) established under CWA section 306, after the promulgation of these standards; or (2) for the purposes of compliance with Pretreatment Standards for New Sources (PSNS), after the publication of proposed standards under CWA section 307 (c), if such standards are thereafter promulgated in accordance with that section.

NONCONVENTIONAL POLLUTANTS: Pollutants that are neither conventional pollutants listed at 40 CFR Part 401.16 nor priority pollutants listed in Appendix A of 40 CFR Part 423.

NON-CONTAMINATED STORM WATER: Storm water which does not come in direct contact with landfill wastes, the waste handling and treatment areas, or wastewater that is subject to the limitations and standards. Non-contaminated storm water includes storm water which flows off the cap, cover, intermediate cover, daily cover, and/or final cover of the landfill.

NON-HAZARDOUS SUBCATEGORY: For the purposes of this report, Non-Hazardous subcategory refers to all landfills regulated under Subtitle D of RCRA.

NON-WATER QUALITY ENVIRONMENTAL IMPACT: Deleterious aspects of control and treatment technologies applicable to point source category wastes, including, but not limited to air pollution, noise, radiation, sludge and solid waste generation, and energy usage.

NSPS: New Source Performance Standards, applicable to new sources of direct dischargers whose construction is begun after the publication of the proposed effluent regulations under CWA section 306.

OCPSF: Organic chemicals, plastics, and synthetic fibers manufacturing point source category. (40 CFR Part 414).

OFF-SITE: Outside the boundaries of a facility.

ON-SITE: The same or geographically contiguous property, which may be divided by a public or private right-of-way, provided the entrance and exit between the properties is at a crossroads intersection, and access is by crossing as opposed to going along the right-of-way. Non-contiguous properties owned by the same company or locality but connected by a right-of-way, which it controls, and to which the public does not have access, is also considered on-site property.

PASS THROUGH: A pollutant is determined to "pass through" POTWs when the nationwide median percentage removed by well-operated POTWs achieving secondary treatment is less than the percentage removed by the industry's direct dischargers that are using the BAT technology.

POINT SOURCE: Any discernable, confined, and discrete conveyance from which pollutants are or may be discharged.

POLLUTANTS OF INTEREST (POIs): Pollutants commonly found in landfill generated wastewater. For the purposes of this report, a pollutant of interest is a pollutant that is detected three or more times above a treatable level at a landfill, and must be present at more than one facility.

PRIORITY POLLUTANT: One hundred twenty-six compounds that are a subset of the 65 toxic pollutants and classes of pollutants outlined in Section 307 of the CWA. The priority pollutants are specified in the NRDC settlement agreement (Natural Resources Defense Council et al v. Train, 8 E.R.C. 2120 [D.D.C. 1976], modified 12 E.R.C. 1833 [D.D.C. 1979]).

PRODUCT STEWARDSHIP: These activities mean the acceptance for treatment and disposal of only the following materials: spent, or unused products; shipping and storage containers with product residue; off-specification products.

PSES: Pretreatment standards for existing sources of indirect discharges, under Sec. 307(b) of the CWA.

PSNS: Pretreatment standards for new sources of indirect discharges, applicable to new sources whose construction has begun after the publication of proposed standards under CWA section 307 (c), if such standards are thereafter promulgated in accordance with that section.

PUBLIC SERVICE: The provision of landfill waste disposal services to individual members of the general public, publicly-owned organizations (schools, universities, government agencies, municipalities) and not-for-profit organizations for which the landfill does not receive a fee or other remuneration.

PUBLICLY OWNED TREATMENT WORKS (POTW): Any device or system, owned by a state or municipality, used in the treatment (including recycling and reclamation) of municipal sewage or industrial wastes of a liquid nature that is owned by a state or municipality. This includes sewers, pipes, or other conveyances only if they convey wastewater to a POTW providing treatment (40 CFR 122.2).

RCRA: The Resource Conservation and Recovery Act of 1976 (RCRA) (42 U.S.C. Section 6901 et seq.), which regulates the generation, treatment, storage, disposal, or recycling of solid and hazardous wastes.

SUBTITLE C LANDFILL: A landfill permitted to accept hazardous wastes under Sections 3001 and 3019 of RCRA and the regulations promulgated pursuant to these sections, including 40 CFR Parts 260 through 272.

SUBTITLE D LANDFILL: A landfill permitted to accept only non-hazardous wastes under Sections 4001 through 4010 of RCRA and the regulations promulgated pursuant to these sections, including 40 CFR Parts 257 and 258.

SURFACE IMPOUNDMENT: A natural topographic depression, man-made excavation, or diked area formed primarily of earthen materials (although it may be lined with man-made materials), used to temporarily or permanently treat, store, or dispose of waste, usually in the liquid form. Surface impoundments do not include areas constructed to hold containers of wastes. Other common names for surface impoundments include ponds, pits, lagoons, finishing ponds, settling ponds, surge ponds, seepage ponds, and clarification ponds.

TOXIC POLLUTANTS: Pollutants declared “toxic” under Section 307(a)(1) of the Clean Water Act.

TRUCK/EQUIPMENT WASHWATER: Wastewater generated during either truck or equipment washes at the landfill. During routine maintenance or repair operations, trucks and/or equipment used within the landfill (e.g., loaders, compactors, or dump trucks) are washed and the resultant washwaters are collected for treatment.

VARIABILITY FACTOR: The daily variability factor is the ratio of the estimated 99th percentile of the distribution of daily values divided by the expected value, median or mean, of the distribution of the daily data. The monthly variability factor is the estimated 95th percentile of the distribution of the monthly averages of the data divided by the expected value of the monthly averages.

ZERO DISCHARGE: No discharge of pollutants to waters of the United States or to a POTW. Also included in this definition are alternative discharge or disposal of pollutants by way of evaporation, deep-well injection, off-site transfer, and land application.

INDEX

Activated Sludge
- Capital Cost Curves (9-71, 9-72)
- Costing (9-20, 9-36)
- Evaluated as BAT - Hazardous Subcategory (11-31, 11-84)
- Evaluated as BAT - Non-Hazardous Subcategory (11-29, 11-82, 11-83, 11-85)
- Evaluated as BPT - Hazardous Subcategory (11-20, 11-84)
- Evaluated as BPT - Non-Hazardous Subcategory (11-8, 11-82, 11-83)
- O&M Cost Curve (9-73)
- Technology Description (8-34, 8-79)
- Treatment Performance (8-50, 8-58, 8-88)
- Types (8-36)

Age (see Landfills Industry)

Air Pollution Reduction Impacts (10-1)

Air Stripping
- Number of Landfills currently using (8-53)
- Technology Description (8-14, 8-67)

Ammonia
- Limitations - Hazardous Subcategory (2-20, 11-80)
- Limitations - Non-Hazardous Subcategory (2-21, 11-68)
- National Estimates of Pollutant of Interest Reductions (11-69, 11-70)
- Pretreatment Standards - Hazardous Subcategory (2-5, 11-48)
- Pretreatment Standards - Non-Hazardous Subcategory (2-5, 11-35)
- Raw Wastewater Concentrations - Hazardous Subcategory (5-20, 5-25, 6-50, 6-51, 8-56, 8-58, 11-56, 11-74, 11-77)
- Raw Wastewater Concentrations - Non-Hazardous Subcategory (5-20, 5-23, 5-24, 5-27, 5-30, 6-49, 6-51, 8-54, 8-55, 8-60, 11-56, 11-62, 11-65)
- Treated Effluent Concentrations - Hazardous Subcategory (8-56, 8-58, 11-56, 11-74, 11-77)
- Treated Effluent Concentrations - Non-Hazardous Subcategory (8-54, 8-55, 8-60, 11-56, 11-62, 11-65)

Anaerobic Biological Systems
- Technology Description (8-30, 8-75)
- Technology Performance (8-46, 8-54, 8-85)

Applicable Waste Streams (2-2, 6-1, 6-8, 6-30)

INDEX

Aqueous Waste Disposal Impacts (10-3)

Attached-Growth Biological Systems
- Technology Description (8-31, 8-76, 8-77, 8-78)
- Types (8-31)

BAT (1-2)
- Costs (9-31, 9-32, 9-40, 9-44, 9-49, 9-54)
- Limitations - Hazardous Subcategory (2-4, 2-20, 11-31, 11-80)
- Limitations - Non-Hazardous Subcategory (2-4, 2-21, 11-29, 11-68)
- Technology Description (8-1)
- Technology Options - Hazardous Subcategory (11-84)
- Technology Options - Non-Hazardous Subcategory (11-81, 11-82, 11-83, 11-85)

BCT (1-2)
- Costs (9-30, 9-31, 9-40, 9-44, 9-49)
- Limitations (2-4, 11-27)
- Technology Options (11-82, 11-83, 11-84)

Best Management Practices (8-1)

Benzoic Acid
- Limitations - Hazardous Subcategory (2-20, 11-80)
- Limitations - Non-Hazardous Subcategory (2-21, 11-68)
- National Estimates of Pollutant of Interest Reductions (11-69)
- Pretreatment Standards - Hazardous Subcategory (11-48)
- Pretreatment Standards - Non-Hazardous Subcategory (11-41)
- Raw Wastewater Concentrations - Hazardous Subcategory (5-20, 5-25, 6-50, 6-53, 8-56, 8-58, 11-56, 11-74, 11-77)
- Raw Wastewater Concentrations - Non-Hazardous Subcategory (5-20, 5-22, 5-24, 5-27, 5-30, 6-49, 6-53, 8-54, 8-55, 8-60, 11-56, 11-62, 11-66)
- Treated Effluent Concentrations - Hazardous Subcategory (8-56, 8-58, 11-56, 11-74, 11-77)
- Treated Effluent Concentrations - Non-Hazardous Subcategory (8-54, 8-55, 8-60, 11-56, 11-62, 11-66)

Biological Treatment
- As a selection criteria for BPT facilities (11-8, 11-9, 11-10, 11-20, 11-21, 11-22, 11-55, 11-58, 11-59, 11-60, 11-73, 11-82, 11-83, 11-84)
- Number of Landfills currently using (3-37, 8-26, 8-53)

INDEX

Technology Description (8-24)
Types (8-26)

BOD_5 (4-2, 5-13, 6-10, 6-18)
As a selection criteria for BPT facilities (11-11, 11-55, 11-57, 11-58, 11-59)
Concentration with Age of Landfill (5-11, 5-30, 6-10)
Limitations - Hazardous Subcategory (2-20, 11-80)
Limitations - Non-Hazardous Subcategory (2-21, 11-68)
National Estimates of Pollutant of Interest Reductions (11-69, 11-70)
Raw Wastewater Concentrations - Hazardous Subcategory (5-20, 5-25, 6-50, 6-51, 8-56, 8-58, 11-56, 11-74, 11-76)
Raw Wastewater Concentrations - Non-Hazardous Subcategory (5-20, 5-23, 5-24, 5-27, 5-30, 6-31, 6-49, 6-51, 8-54, 8-55, 8-60, 11-56, 11-59, 11-62, 11-64)
Regulated Pollutant (7-14, 7-21)
Treated Effluent Concentrations - Hazardous Subcategory (8-56, 8-58, 11-74, 11-76)
Treated Effluent Concentrations - Non-Hazardous Subcategory (8-54, 8-55, 8-60, 11-62, 11-64)

BPT (1-1, 11-6)
Costs (9-29, 9-30, 9-40, 9-44, 9-49)
Limitations - Hazardous Subcategory (2-4, 2-20, 11-23, 11-80)
Limitations - Non-Hazardous Subcategory (2-4, 2-21, 11-11, 11-68)
Selected Facilities (11-60, 11-73)
Technology Options - Hazardous Subcategory (11-20, 11-84)
Technology Options - Non-Hazardous Subcategory (11-8, 11-82, 11-83)

Breakpoint Chlorination
Capital Cost Curve (9-83)
Costs (9-27, 9-36)
O&M Cost Curve (9-84)
Technology Description (8-12)

Captive/Intra-Company Facilities
Definition (3-2)
Exemption from Guideline (2-10, 3-13)
Number in Landfills Population (3-12, 3-25)

Carbon Adsorption
Capital Cost Curve (9-81)
Costs (9-26, 9-36)

INDEX

Number of Landfills currently using (8-53)
O&M Cost Curve (9-82)
Technology Description (8-21, 8-71)
Treatment Performance (8-48, 8-55)

Chemical Oxidation/Reduction
Number of Landfills currently using (8-53)
Technology Description (8-11, 8-12, 8-65, 8-66)

Chemical Precipitation
Capital Cost Curves (9-63, 9-65, 9-67)
Costs (9-14, 9-36, 9-41, 9-42, 9-43)
Evaluated as BAT (11-28, 11-31, 11-84)
Evaluated as BPT - Hazardous Subcategory (11-20, 11-21, 11-22 11-84)
Evaluated as BPT - Non-Hazardous Subcategory (11-8)
Number of Landfills currently using (3-37, 8-53)
O&M Cost Curves (9-64, 9-66, 9-68)
Technology Description (8-8)
Treatment Performance (8-46, 8-48, 8-50, 8-54, 8-55, 8-58, 8-85, 8-86, 8-88)

Clarification
Capital Cost Curves (9-69, 9-74)
Costs (9-19, 9-22, 9-36)
Evaluated as BAT - Hazardous Subcategory (11-31, 11-84)
Evaluated as BAT - Non-Hazardous Subcategory (11-29, 11-82, 11-83, 11-85)
Evaluated as BPT - Hazardous Subcategory (11-20, 11-84)
Evaluated as BPT - Non-Hazardous Subcategory (11-8, 11-82, 11-83, 11-85)
O&M Cost Curves (9-70, 9-75)
Technology Description (8-6, 8-62)

Contaminated Ground Water
CERCLA Ground Water Data (4-13, 6-7)
Concentration of Pollutants (5-8, 5-25, 5-27)
Definition (3-19, 6-4)
Exclusion from Guideline (2-5, 3-12, 3-19, 5-8, 6-1, 6-4)
Monitoring (3-6, 3-8, 3-9)
Quantity of Flow Generated (3-19, 3-34, 6-26, 6-27, 6-29)

INDEX

Costs (9-1)
- Additional Cost Factors (9-8, 9-37)
- BAT (9-31, 9-40, 9-44, 9-49, 9-54)
- BCT (9-30, 9-40, 9-44, 9-49)
- BPT (9-29, 9-40, 9-44, 9-49)
- Land Costs (9-9)
- Methodology (9-6, 9-7, 9-11)
- Models (9-1, 9-3, 9-34, 9-35, 9-40)
- Monitoring Costs (9-10, 9-38)
- NSPS (9-32, 9-40)
- Off-Site Disposal Costs (9-11, 9-39, 11-23)
- Option Specific Costing Logic Flow Diagram (9-59)
- Residual Disposal Costs (9-9)
- Retrofit Costs (9-9)
- Treatment Chemicals (9-14, 9-41, 9-42, 9-43)
- Treatment Technologies (5-14, 9-11, 9-36, 9-40)

Cost Models (9-1)

Current Discharge Concentrations (7-3)
- Alternate Methodology - Hazardous Subcategory (7-5)
- Alternate Methodology - Non-Hazardous Subcategory (7-5)

Denitrification Systems
- In Removal of Nitrate/Nitrite (7-10, 7-17)
- Technology Description (8-41)

Discharge Information
- Discharge Types (3-22, 3-42)
- Quantity of Flow Discharged (3-34, 6-4, 6-5, 6-6, 6-26, 6-27, 6-29, 6-30)
- Raw Wastewater Concentrations - Hazardous Subcategory (5-20, 5-25, 6-17, 6-24, 6-25, 6-50, 6-51, 6-52, 6-53, 6-55, 8-56, 8-58, 11-56, 11-74, 11-76)
- Raw Wastewater Concentrations - Non-Hazardous Subcategory (5-20, 5-22, 5-24, 5-27, 5-30, 6-17, 6-21, 6-23, 6-49, 6-51, 6-52, 6-53, 6-54, 8-54, 8-55, 8-60, 11-54, 11-56, 11-59, 11-61, 11-62, 11-64)
- Sources of Wastewater (3-16, 6-1)

Drained Free Liquids (2-2, 3-1, 3-12, 3-17, 6-1, 6-3)
- Quantity of Flow Generated (3-17, 3-34, 6-14, 6-26, 6-27, 6-29, 6-30)

INDEX

Energy Requirements (5-15, 10-5)

Equalization
- Capital Cost Curve (9-60)
- Costs (9-12, 9-36)
- Evaluated as BAT - Hazardous Subcategory (11-31, 11-84)
- Evaluated as BAT - Non-Hazardous Subcategory (11-29, 11-82, 11-83, 11-85)
- Evaluated as BPT - Hazardous Subcategory (11-20, 11-84)
- Evaluated as BPT - Non-Hazardous Subcategory (11-8, 11-82, 11-83)
- Number of Landfills currently using (3-37, 8-53)
- Technology Description (8-3, 8-61)

Equipment/Truck Washwater (see Truck/Equipment Washwater)

Filtration (8-14)
- Diatomaceous Earth (8-17)
- Fabric Filters (8-21)
- Membrane Filtration (8-18)
  - Reverse Osmosis (8-19, 8-70, 9-78)
  - Ultrafiltration (8-18, 8-69)
- Multimedia Filtration (8-17, 8-68, 9-76, 9-77)
- Number of Landfills currently using (3-37, 8-53)
- Sand Filtration (8-15)

Flocculation
- Capital Cost Curve (9-61)
- Costs (9-13, 9-36)
- O&M Cost Curve (9-62)
- Technology Description (8-5, 8-62)

Floor Washings (2-2, 3-12, 6-1)
- Quantity of Flow Generated (3-18, 3-34, 6-26, 6-27, 6-29, 6-30)

Fluidized Bed Biological Reactor
- Technology Description (8-33, 8-78)

Granular Activated Carbon (see Carbon Adsorption)

INDEX

Gravity Assisted Separation (also see Clarification)
Number of Landfills currently using (8-53)
Technology Description (8-6, 8-62)

Ground Water (see Contaminated Ground Water)

Hazardous Subcategory (2-1, 5-2)
BAT Options (11-31, 11-84)
BPT Options (11-20, 11-84)
Limitations (2-20, 11-23, 11-31, 11-80)
Pollutants Considered for Regulation - Indirect Dischargers (7-27)
Pollutants of Interest (7-30)
Pollutants Selected for Regulation - Direct Dischargers (7-21)
Pretreatment Standards (2-5, 11-32, 11-48)
Raw Wastewater Concentrations (5-20, 5-25, 6-24, 6-25, 6-50, 6-51, 6-52, 6-53, 6-55, 8-56, 8-58, 11-56, 11-74, 11-76)
Subcategorization Approach (5-1, 5-2)
Wastewater Flow and Discharge (6-6, 6-26, 6-30)

Impacts
Air Pollution (7-11, 7-12, 7-13, 7-19, 10-1)
Energy Requirements (10-5)
Solid and Other Aqueous Waste Disposal (10-3)
Non-Water Quality (5-15, 10-1)

Intra-Company/Captive Facilities
Definition (3-2)
Exemption from Guideline (2-10, 3-13)
Number in Landfills Population (3-12, 3-25)

Ion Exchange
Technology Description (8-23, 8-72)

Iron Coprecipitation
Technology Description (8-11)

Laboratory-Derived Wastewater (2-2, 3-1, 3-12, 3-18, 6-1, 6-3, 6-8)

INDEX

Lagoon Systems
- Aerated (8-28, 8-73)
- Aerobic (8-27)
- Anaerobic (8-28)
- Facultative (8-29, 8-74)
- Number of Landfills currently using (8-26)
- Technology Description (8-26)

Land Costs (9-9)

Landfill Gas Condensate (2-2, 3-1, 3-12, 3-17, 6-1, 6-2, 6-13)
- Definition (3-17, 6-2)
- Monitoring Data (6-32)
- Quantity of Flow Generated (3-17, 3-34, 6-13, 6-26, 6-27, 6-29, 6-30)
- Treatment (6-13)

Landfill Leachate (2-2, 3-1, 3-12, 3-16, 6-1, 6-2, 6-8)
- Definition (3-16, 6-2)
- Literature Data (6-12, 6-31)
- Quantity of Flow Generated (3-17, 3-34, 6-11, 6-26, 6-27, 6-29, 6-30, 6-31)

Landfills Industry
- Age (5-11, 5-28, 5-30)
- Area (3-26, 3-27)
- Cells (3-28, 5-6, 5-13)
- General Information (3-14)
- Industry Description (3-1)
- Location of Landfills (3-11, 3-12, 3-24, 5-9)
- Number of Facilities (3-11, 3-12, 3-24)
- Ownership Types (3-1, 3-25, 5-9)
- Population (3-11, 3-24, 3-29, 3-30)
- Regulatory History of the Landfills Industry (3-3)
- Regulatory Types (3-2)
- Waste Received (3-15, 3-31, 3-32, 3-33, 5-3)

Leachate (see Landfill Leachate)

Leachate Collection Systems (3-20, 3-35)

INDEX

Limitations
- BPT/BAT/NSPS Limitations - Hazardous Subcategory (2-4, 2-20, 11-23, 11-80)
  - Facilities and Sample Points Selected (11-74)
  - Facilities and Sample Points Excluded (11-76)
- BPT/BAT/NSPS Limitations - Non-Hazardous Subcategory (2-4, 2-21, 11-11, 11-68)
  - Facilities and Sample Points Selected (11-62)
  - Facilities and Sample Points Excluded (11-64)
- Calculations of Effluent Limitations (11-6)
- Long-Term Averages (11-2, 11-81)
- PSES (11-32, 11-34, 11-48)
- Variability Factors (11-5)

Long-Term Averages (11-1, 11-2, 11-81)

Monitoring Costs (9-10)

Monofills (5-5, 5-22, 5-24, 11-14)

Multimedia Filtration
- As a selection criteria for BPT facilities (11-9, 11-10, 11-16, 11-20, 11-21, 11-22, 11-60, 11-61, 11-83, 11-84)
- Capital Cost Curve (9-76)
- Costs (9-23, 9-36)
- Evaluation as BAT - Hazardous Subcategory (11-31, 11-84)
- Evaluation as BAT - Non-Hazardous Subcategory (11-29, 11-83, 11-85)
- Evaluation as BPT - Hazardous Subcategory (11-20, 11-21, 11-22, 11-84)
- Evaluation as BPT - Non-Hazardous Subcategory (11-9, 11-10, 11-16, 11-83)
- O&M Cost Curve (9-77)
- Technology Description (8-17, 8-68)
- Treatment Performance (8-46, 8-48, 8-54, 8-55, 8-85, 8-86)

National Estimates (3-11, 3-43)

National Risk Management Research Laboratory (NRMRL) Treatability Database (see Pass-Through Analysis)

Neutralization
- Number of Landfills currently using (8-53)
- Technology Description (8-4, 8-61)

INDEX

Nitrification Systems
- In Removal of Nitrate/Nitrite (7-10, 7-17)
- Technology Description (8-40)

Non-Hazardous Subcategory (2-1, 5-3)
- BAT Options (11-29, 11-82, 11-83, 11-85)
- BPT Options (11-8, 11-82, 11-83)
- Limitations (2-21, 11-11, 11-29, 11-68)
- Pollutants Considered for Regulation - Indirect Dischargers (7-25)
- Pollutants of Interest (7-29)
- Pollutants Selected for Regulation - Direct Dischargers (7-14)
- Pretreatment Standards (2-5, 11-32, 11-34)
- Raw Wastewater Concentrations (5-20, 5-22, 5-24, 5-27, 5-30, 6-21, 6-23, 6-31, 6-49, 6-51, 6-52, 6-53, 6-54, 8-54, 8-55, 8-60, 11-54, 11-56, 11-59, 11-61, 11-62, 11-64)
- Subcategorization Approach (5-1, 5-3)
- Wastewater Flow and Discharge (6-5, 6-27, 6-29, 6-30)

Non-Water Quality Impacts (10-1)

NSPS (1-3, 2-5,)
- Costs (9-32, 9-40)
- Limitations (2-20, 2-21, 11-31, 11-68, 11-80, 11-83, 11-84)

Off-Site Disposal Costs (9-11, 9-39, 11-23)

Ownership Status (see Landfills Industry)

P-Cresol
- Limitations - Hazardous Subcategory (2-20, 11-80)
- Limitations - Non-Hazardous Subcategory (2-21, 11-68)
- National Estimates of Pollutant of Interest Reductions (11-69)
- Pretreatment Standards - Hazardous Subcategory (11-48)
- Pretreatment Standards - Non-Hazardous Subcategory (11-43)
- Raw Wastewater Concentrations - Hazardous Subcategory (5-20, 6-50, 6-53, 8-56, 8-58,11-56, 11-75, 11-78)
- Raw Wastewater Concentrations - Non-Hazardous Subcategory (5-20, 5-22, 5-24, 5-30, 6-49, 6-53, 8-54, 8-55, 8-60, 11-56, 11-62, 11-66)
- Treated Effluent Concentrations - Hazardous Subcategory (8-56, 8-58, 11-75, 11-78)
- Treated Effluent Concentrations - Non-Hazardous Subcategory (8-54, 8-55, 8-60, 11-62, 11-66)

INDEX

Pass-Through Analysis
- Approach (7-22)
- BAT Performance Data (7-22, 7-32, 7-34)
- Data Editing (7-23)
- NRMRL Treatability Database (4-15, 7-22, 7-31)
- 50-POTW Study Database (4-14, 7-22, 7-31)
- Results - Hazardous Subcategory (7-27, 7-35, 11-48)
- Results - Non-Hazardous Subcategory (7-25, 7-33, 11-34)

pH (6-18)
- Limitations (2-20, 2-21, 11-68, 11-80)
- Raw Wastewater Concentrations (6-31, 6-51)

Phenol
- Limitations - Hazardous Subcategory (2-20, 11-80)
- Limitations - Non-Hazardous Subcategory (2-21, 11-68)
- National Estimates of Pollutant of Interest Reductions (11-69)
- Pretreatment Standards - Hazardous Subcategory (11-48)
- Pretreatment Standards - Non-Hazardous Subcategory (11-44)
- Raw Wastewater Concentrations - Hazardous Subcategory (5-20, 5-26, 6-50, 6-53, 11-56, 11-75, 11-78)
- Raw Wastewater Concentrations - Non-Hazardous Subcategory (5-20, 5-22, 5-24, 5-27, 5-30, 6-49, 6-53, 11-56, 11-63, 11-67)
- Treated Effluent Concentrations - Hazardous Subcategory (8-56, 8-58, 11-75, 11-78)
- Treated Effluent Concentrations - Non-Hazardous Subcategory (8-54, 8-55, 8-60, 11-63, 11-67)

Pollutants Considered for Regulation - Indirect Dischargers
- Hazardous Subcategory (7-22, 7-27, 7-35)
- Non-Hazardous Subcategory (7-22, 7-25, 7-33)

Pollutants Not Selected for Regulation - Direct Dischargers
- Hazardous Subcategory (7-15, 7-16, 7-18)
- Non-Hazardous Subcategory (7-9, 7-10, 7-13)

Pollutants of Interest
- Determination (7-2, 7-7, 7-36)
- Hazardous Subcategory List (6-50, 7-30)
- Metals (11-54)
- Non-Hazardous Subcategory List (6-49, 7-29)

INDEX

Pollutant Reductions (7-3, 11-35, 11-41, 11-43, 11-44, 11-48, 11-69, 11-70, 11-71, 11-72)

Pollutants Selected for Regulation - Direct Dischargers
- Determination of (7-7, 7-37)
- Hazardous Subcategory (2-20, 7-14, 7-21, 11-80)
- Non-Hazardous Subcategory (2-21, 7-8, 7-10, 11-68)

Population (see Landfills Industry)

50-POTW Study (see Pass-Through Analysis)

Powdered Activated Carbon
- Technology Description (8-38, 8-80)

Preliminary Data Summary (4-1)

Pressure Filtration
- Technology Description (8-42, 8-83)

Pretreatment Methods for Wastes Received at Landfills (3-21, 3-36)

Primary Clarification (see Clarification)

PSES (1-3, 2-5, 11-32)
- Hazardous Subcategory (11-48)
- Non-Hazardous Subcategory (11-34, 11-82)
  - Ammonia (11-35)
  - Benzoic Acid (11-41)
  - P-Cresol (11-43)
  - Phenol (11-44)
  - Public Comments (11-45)

PSNS (1-4, 2-5, 11-53)
- Hazardous Subcategory (11-53)
- Non-Hazardous Subcategory (11-53, 11-82)

Public Comments
- PSES (11-45)

INDEX

QA/QC and Other Data Editing Procedures (4-15)

Quantity of Wastes Received (3-15, 3-31, 3-32, 3-33)

Questionnaires (4-3, Appendix A)
- Detailed Monitoring Questionnaire (4-9)
- Detailed Technical Questionnaire (4-6)
- Screener Surveys (4-4)

RCRA
- Subtitle C (3-3)
- Subtitle D (3-6)

Recovering Pumping Wells (2-3, 3-1, 3-12, 3-20, 6-4)
- Definition (3-20, 6-4)
- Quantity of Flow Generated (3-20, 3-34)

Residual Disposal Costs (9-9)

Retrofit Costs (9-9)

Reverse Osmosis
- Capital Cost Curve (9-78)
- Costs (9-24, 9-36, 11-29)
- Evaluated as BAT - Non-Hazardous Subcategory (11-29, 11-41, 11-69, 11-70, 11-81, 11-85)
- Technology Description (8-19, 8-70)
- Treatment Performance (8-51, 8-60, 8-89, 11-69, 11-70, 11-81)

Rotating Biological Contactors
- Technology Description (8-31, 8-76)

Sampling Program
- Analytical Data (5-24, 5-25, 5-27, 6-37, 6-54, 6-55, 8-54, 8-55, 8-56, 8-58, 8-60, 11-56, 11-59, 11-61, 11-62, 11-64, 11-74, 11-76)
- Pollutants Analyzed in EPA Sampling (6-15, 6-33, 6-37)
- Sampling Episodes (4-11, 4-12, 4-23, 4-24, 4-25)
- Sampling Results for Performance of Treatment Processes (8-45, 8-54, 8-55, 8-56, 8-58, 8-60)
- Site Visits (4-10, 4-11, 4-12, 4-23, 4-25)

INDEX

Scope of the Regulation (2-2)

Screener Surveys (see Questionnaires)

Secondary Clarification (see Clarification)

Section 304 (m) (1-4)

Sequencing Batch Reactors
- BPT Facility for Hazardous Subcategory (11-24)
- BPT Facility for Non-Hazardous Subcategory (11-12)
- Number of Landfills currently using (8-26)
- Technology Description (8-39, 8-81)
- Treatment Performance (8-48, 8-49, 8-55, 8-56, 8-86, 8-87)

Sludge Handling (8-41, 8-53)
- Gravity Thickening (8-42, 8-82)
- Pressure Filtration (8-42, 8-83)
- Sludge Dewatering (8-42, 8-53, 9-25, 11-9)
- Sludge Drying Beds (8-43, 8-84, 9-25, 9-36, 9-79, 9-80)
- Sludge Slurrying (8-42)

Solid Waste Disposal Impacts (10-3)

Storm Water
- Contaminated (2-7, 3-1, 3-12, 3-19, 6-1, 6-3)
- Non-Contaminated (non-contact) (2-7, 3-12, 3-19, 6-1, 6-4)
- Quantity of Flow Generated (3-19, 3-34, 6-26, 6-27, 6-29)

Stripping (see Air Stripping)

Subcategories (see Hazardous and Non-Hazardous Subcategories)
- Landfill Subcategories (2-1, 5-2, 5-3)

Subcategorization (2-1, 5-1)
- Age (5-11, 5-28, 5-30)
- Approach (5-1)
- Factors Considered (5-3)
- Landfill Subcategories (2-1, 5-2, 5-16)

INDEX

Waste Types (5-3)
Wastewater Characteristics (5-7)

Treatment Performance (8-45)
Episode 4626 (8-46, 8-54, 8-85)
Episode 4667 (8-48, 8-55, 8-86)
Episode 4721 (8-49, 8-56, 8-87)
Episode 4759 (8-50, 8-58, 8-88)
Episode 4687 (8-51, 8-60, 8-89)

Trickling Filters
Technology Description (8-32, 8-77)

Truck/Equipment Washwater (2-2, 3-1, 3-12, 3-18, 6-1, 6-3, 6-14)
Definition (3-18, 6-3, 6-14)
Quantity of Flow Generated (3-18, 3-34, 6-15, 6-26, 6-27, 6-29, 6-30)

TSS (6-18)
As a selection criteria for BPT facilities (11-16, 11-25, 11-61)
Limitations - Hazardous Subcategory (2-20, 11-80)
Limitations - Non-Hazardous Subcategory (2-21, 11-68)
National Estimates of Pollutant of Interest Reductions (11-69, 11-70)
Raw Wastewater Concentrations - Hazardous Subcategory (5-20, 5-26, 6-50, 6-51, 8-56, 8-58, 11-56, 11-74, 11-76)
Raw Wastewater Concentrations - Non-Hazardous Subcategory (5-20, 5-23, 5-24, 5-27, 5-30, 6-31, 6-49, 6-51, 8-54, 8-55, 8-60, 11-56, 11-61, 11-62, 11-64)
Regulated Pollutant (7-14, 7-21)
Treated Effluent Concentrations - Hazardous Subcategory (8-56, 8-58, 11-74, 11-76)
Treated Effluent Concentrations - Non-Hazardous Subcategory (8-54, 8-55, 8-60, 11-62, 11-64)

Ultrafiltration (also see Filtration)
Technology Description (8-18, 8-69)

Variability Factors (11-1, 11-5)

Waste Receipts (3-15, 3-31, 3-32, 3-33, 5-3)

Wastewater (6-1)
CERCLA Ground Water Data (4-13, 6-18)

INDEX

Characterization (5-7, 6-7, 6-17, 6-31, 6-32, 6-37, 6-49, 6-50, 6-51, 6-52, 6-53, 6-54, 6-55)
Industry Supplied Data (4-13)
Quantity of Flow Generated (3-16, 3-34, 6-4, 6-5, 6-6, 6-26, 6-27, 6-29, 6-30)
Sources (2-2, 3-16, 6-1)
Technology for Treatment of (3-22, 3-37, 5-14, 8-53)
Treatment Performance (8-45)

Wetlands Treatment
Technology Description (8-41)

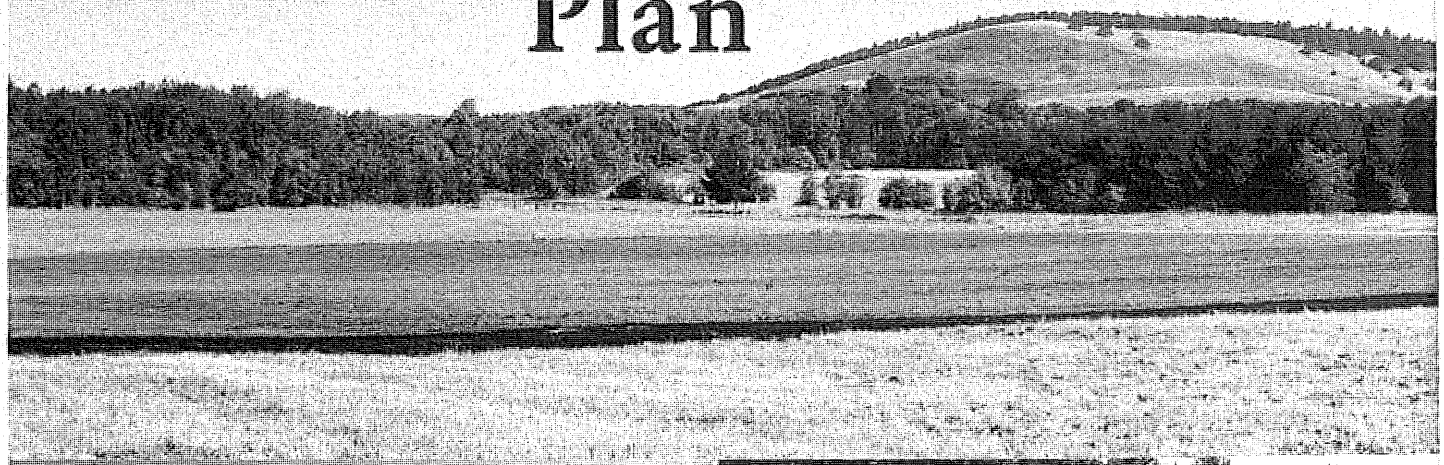
Zero/Alternative Discharge Treatment
Cost (9-7, 9-11, 9-39, 11-20, 11-23)
Discharge Types (3-22, 3-42)
Evaluated as BAT - Hazardous subcategory (11-31)
Evaluated as BPT - Hazardous subcategory (11-21)
Number of Landfills currently using (6-7, 11-24)
Technology Description (8-44)

CORVALLIS
ENHANCING COMMUNITY LIVABILITY

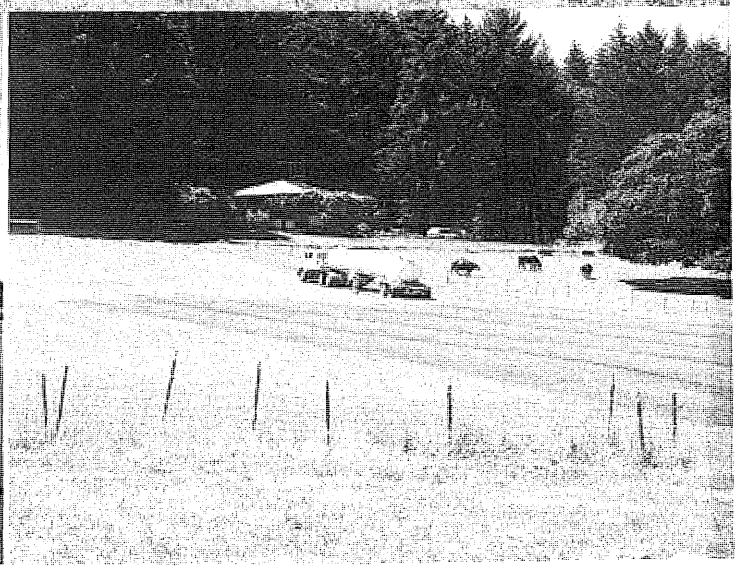
City of Corvallis

Jan 2007

Biosolids Management Plan







CORVALLIS
ENHANCING COMMUNITY LIVABILITY

TABLE OF CONTENTS

LIST OF FIGURES ii
LIST OF TABLES ii

EXECUTIVE SUMMARY 1

INTRODUCTION 2

FACILITY DESCRIPTION 3
- Liquid Flowstream 3
- Solids Processing 3
- Septage 4
- Leachate 4
- Pre-Treatment Program 6

SOLIDS TREATMENT PROCESSES 7
- Vector Attraction Reduction 7
- Pathogen Reduction 8

FACULTATIVE TREATMENT AND STORAGE 8

CONTINGENCY PLANNING 9
- Process Failure 9
- Spill Control 9

BIOSOLIDS CHARACTERISTICS 9

REPORTING 11

BIOSOLIDS UTILIZATION PROGRAM 11
- Transportation and Application 11
- Site Selection and Inventory 11
- Application Rates 22
- Soil Sampling and Testing 23

FUTURE BIOSOLIDS PROGRAM ENHANCEMENTS 23

REFERENCES 24

APPENDIX A 2006 Annual Report
APPENDIX B Site Authorization Letters
APPENDIX C DEQ Biosolids Management Rules And Guidance
APPENDIX D OSU Extension Publication: "Fertilizing With Biosolids"
APPENDIX E Glossary

Updated: January 2011

LIST OF FIGURES

No.	Title	Page no.
1	Process Flow Schematic	5
2	Site Layout	5a
3	Vicinity Map	12
4A	Detailed Site Map – Airport	13
4B	Detailed Site Map – Anderson	14
4C	Detailed Site Map – Bressler 100	15
4D	Detailed Site Map – Gray	16
4E	Detailed Site Map – OSU	17
4F	Detailed Site Map – Sanders 30	18
4G	Detailed Site Map – Corvallis Dedicated Site	19
4H	Detailed Site Map – Pimm 101	20

LIST OF TABLES

No.	Title	Page no.
1	Corvallis WWRP Biosolids Characteristics	10
2	Land Application Sites	21
3	Maximum Biosolids Application Rates (Typical)	22

EXECUTIVE SUMMARY

The City of Corvallis Wastewater Reclamation Plant (WWRP) operates a biosolids land application program in the local agricultural community. Biosolids are treated at the WWRP through anaerobic digestion then stored for several months in a facultative lagoon. During storage, solids undergo further pathogen reduction and stabilization. In a typical year, lagoon solids are harvested beginning in May when approved land application sites are accessible. Land application generally runs through September. For the balance of the year solids are stored in the lagoon at the WWRP.

Approximately 1,500 acres have been approved by the Oregon Department of Environmental Quality (DEQ) for biosolids application by the City of Corvallis and additional approvals are pending. Many of these farms are within 5 miles of the treatment facility; all are within 10 miles. Crops grown include forage and grass seed.

The Corvallis biosolids program has operated successfully for nearly 20 years, providing a valued source of fertilizer to agricultural cooperators. The product is applied at an agronomic rate of approximately 3 to 4 dry tons (DT)/acre on 100 to 400 acres/year. Since over 1,500 acres have been approved for application, demand for the product exceeds supply.

Lagoon storage at the WWRP lends flexibility to the biosolids program so that delivery can be made at times when growers need it the most and local field concentrations are favorable. The lagoon also provides additional stabilization (odor) and pathogen reduction. Monitoring for the past year indicated that Corvallis biosolids came close to qualifying as Class A (low fecal coliform) product. Trace metals concentrations are also low due to an effective industrial pretreatment and source control program that has been implemented by city staff.

Wastewater Reclamation Plant staff have worked diligently to develop and maintain the biosolids program consistent with State and Federal standards, and look forward to continued cooperation with the agricultural community. This Management Plan provides basic information about how the program is operated as well as monitoring and testing data to document quality of the product.

INTRODUCTION

Biosolids consist of a semi-solid or liquid residue generated as the end product of biological treatment processes at the WWRP facility. They contain organic matter and nutrients that are beneficial for soil, crop, and livestock productivity. As such they are commonly used as a fertilizer and soil amendment.

Biosolids are regulated by both the U.S. Environmental Protection Agency (EPA) and Oregon Department of Environmental Quality (DEQ) to ensure that quality standards are met. Standards cover trace metal concentrations, pathogen content, odor potential, and basic operational practices. Biosolids produced at the Corvallis Wastewater Reclamation Plant (WWRP), for example, can only be applied to approved farm sites and at controlled rates consistent with crop needs. When managed in this way, biosolids land application provides beneficial reuse or recycling of the nutrients and organic matter. This is one of the ways that treatment plant staff and the city contribute to a better environment for Oregon.

Beneficial reuse of biosolids has long been preferred over historical disposal practices such as incineration or landfilling. Land application practices are encouraged as an alternative to disposal by state and federal regulatory authorities. In Oregon, 99 percent of biosolids produced are recycled on farmland in this way. Recent EPA and DEQ rule making and risk assessments have added credibility and certainty to the concept. Numerous publications from EPA and academic institutions like Oregon State University (OSU) provide valuable information regarding biosolids management practices. A reference list is provided at the end of this report, and a recent OSU bulletin titled "Fertilizing with Biosolids" is included as Appendix D.

This Management Plan outlines treatment practices, monitoring, and methods of operation for the biosolids program. The plan was originally developed in 1980 when many solids handling improvements were designed for the Wastewater Reclamation Plant. The most recent update to the plan was in 2006. The plan is being updated at this time in conjunction with the city's National Pollutant Discharge Elimination System (NPDES) wastewater discharge permit renewal and to address recent modifications in regulatory policy.

FACILITY DESCRIPTION

Liquid Flowstream

The liquid treatment process at the City of Corvallis WWRP is divided into two trains, the WWRP and the CSO plants. The WWRP is designed for an average dry weather flow of 9.7 million gallons per day (mgd). The peak design capacity of the WWRP is 28 mgd. Influent plant flow is pumped by four raw sewage pumps and split into two process trains that each consist of a mechanically cleaned bar screen, a parshall flume for flow measurement, an aerated grit chamber and a rectangular primary sedimentation tank. Primary effluent is conveyed to the two trickling filters and the two aeration basins for secondary treatment. Both of the aeration basins are equipped with fine bubble diffusers and mechanical mixers. Mixed liquor from the aeration basins is conveyed to two secondary clarifiers, and the secondary effluent is disinfected in a single chlorine contact tank. Final effluent gravity flows to the Willamette River under normal conditions and is pumped during higher river levels.

The combined sewer overflow (CSO) has an 85 mgd influent pumpstation with a wet weather treatment capacity of 47 mgd. Influent cso plant flow is pumped by seven submersible sewage pumps and split into two trains that each consist of a mechanically cleaned barscreen, magmeter for flow measurement, aerated grit chamber, chlorine injection, primary sedimentation and dechlorination injection. In addition, a 5 million gallon (MG) lagoon is available for storage of wet weather flow rates exceeds the peak wet weather treatment capacity of 47 mgd. A process flow schematic is provided in Figure 1. The site layout is shown in Figure 2.

Solids Processing

Solids produced during the liquid treatment process are stabilized through anaerobic digestion. A portion of the waste solids generated from the trickling filters and activated sludge process are returned to the gravity thickners where they are combined with raw sludge and thickened. The combined thickened sludge is conveyed to the primary digester, which is mechanically mixed using pumped recirculation. The sludge is heated to 35.6 degrees C for approximately 30 days, and the volatile solids reduction during digestion is approximately 60 percent (see Appendix A).

Septage

The City of Corvallis WWRP receives minor amounts of septage and other sanitary waste generated in the region that are trucked or hauled to the plant. Septic haulers are required by DEQ to secure an approved disposal site. All dischargers of trucked and hauled waste must apply for approval prior to discharging at the plant, and the applicable procedures for a new service discharger must be followed. Domestic septage dischargers are issued a general notification letter and are required to provide a manifest with each load and a certification statement confirming that no industrial waste is being disposed of. Mobile carpet cleaning operations must follow the same requirements as the domestic septage dischargers and certify that they are only discharging carpet-cleaning residuals. Disposal of any other types of trucked or hauled industrial waste will be considered on a case by case basis by the city. The city will not accept any trucked or hauled hazardous waste under any circumstance.

Leachate

The City of Corvallis WWRP receives leachate from Coffin Butte landfill. During wet weather and prior to the summer irrigation season the landfill leachate storage lagoon may reach capacity. The City recognizes this situation and has agreed to treat leachate during such times to avoid any environmental impacts. The discharge of leachate to the WWRP is permitted and regulated by the City's industrial waste management staff. The landfill must apply for approval prior to discharging at the plant.

City of Corvallis
Wastewater Reclamation Plant
1304 N.E. Second Street

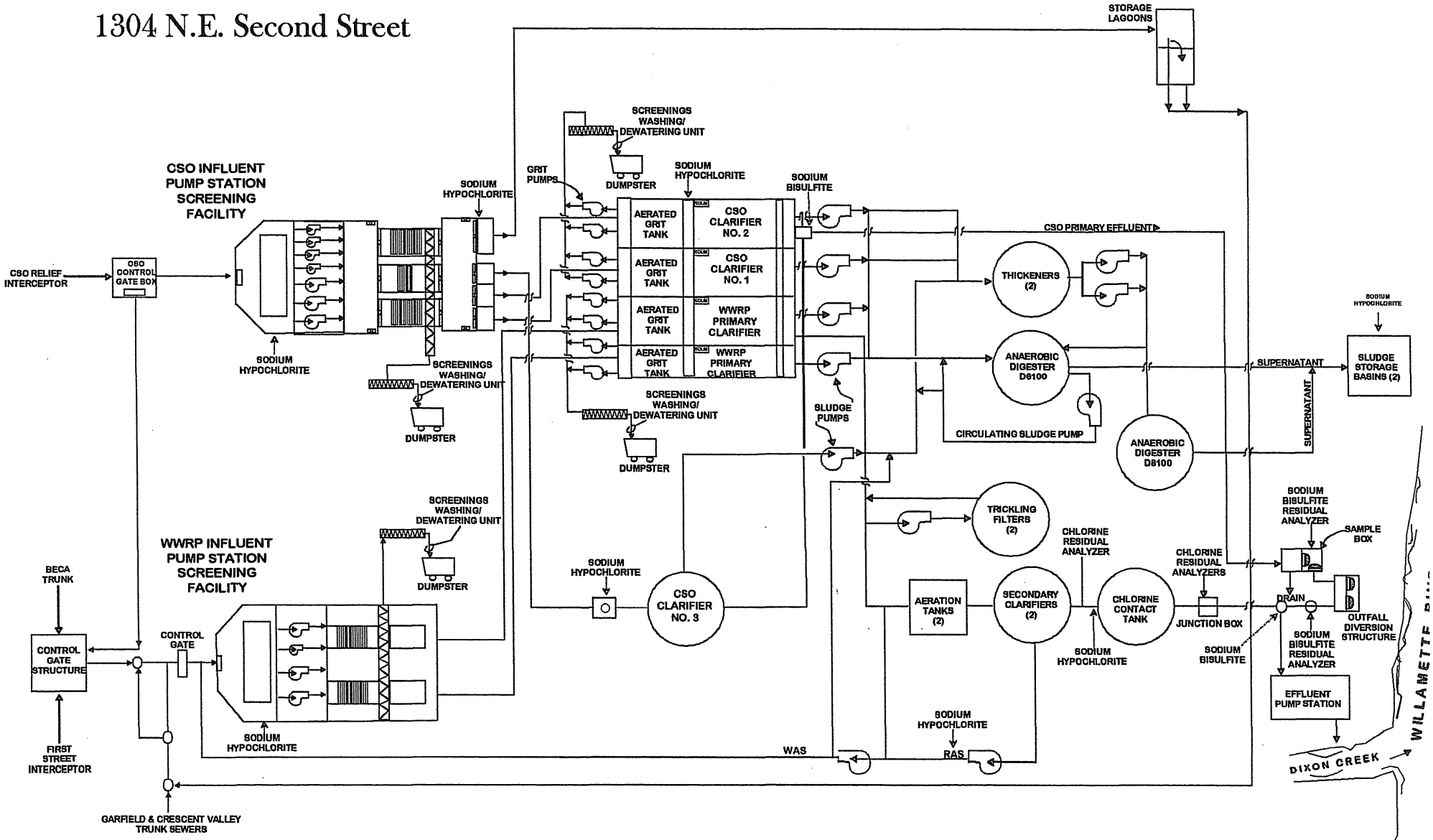


FIGURE 1

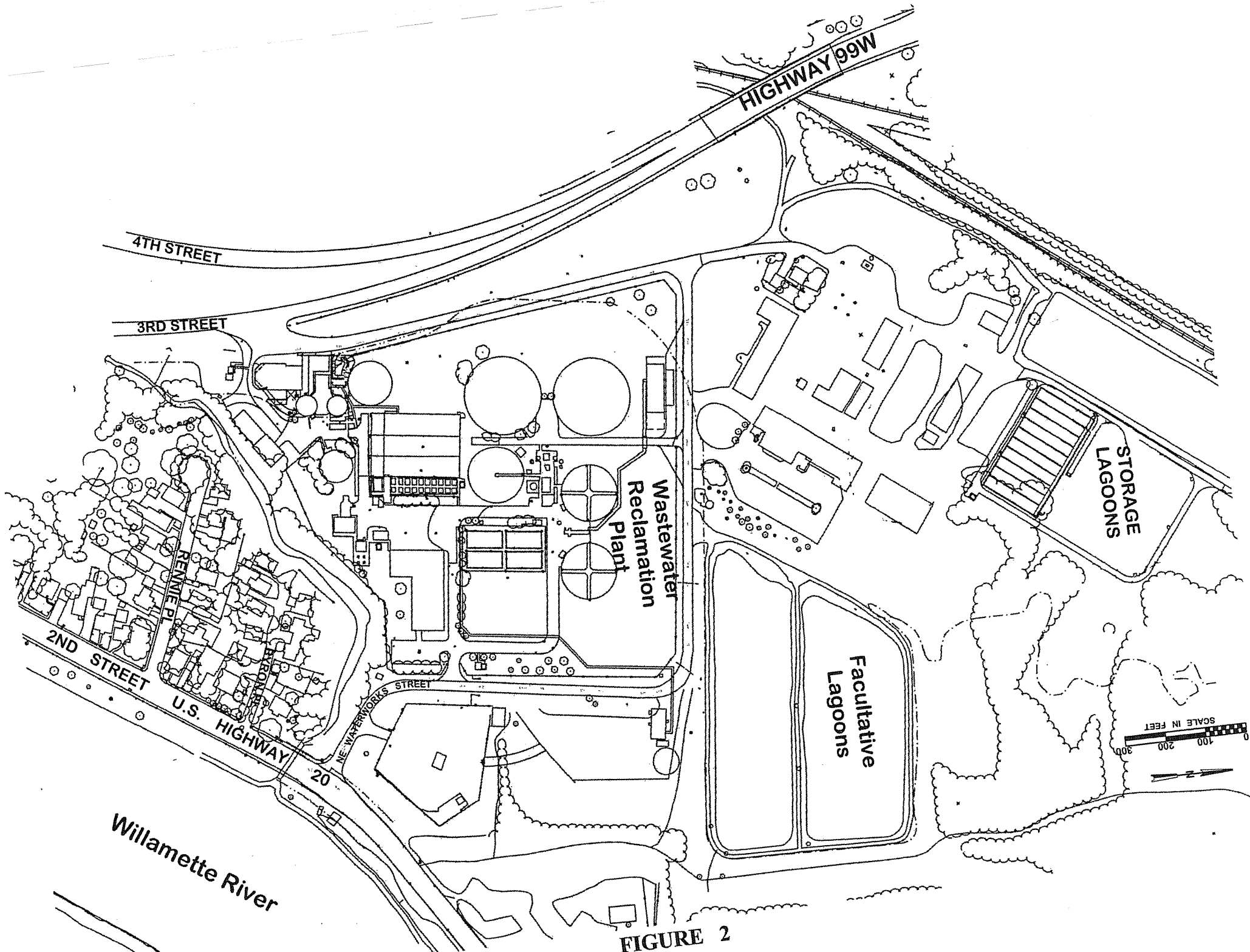

HIGHWAY 99W
4TH STREET
3RD STREET
Wastewater Reclamation Plant
STORAGE LAGOONS
Facultative Lagoons
RENNIE PL
2ND STREET
U.S. HIGHWAY 20
NE WATERWORKS STREET
Willamette River
SCALE IN FEET
0
100
200
300

FIGURE 2

Pre-Treatment Program

The 1977 amendments to the Clean Water Act required municipalities to develop and implement pretreatment programs to control pollutants from industries discharging to the public sewer. The City of Corvallis implemented its pretreatment program in 1983 after approval by DEQ.

The purpose of the pretreatment program is to prevent the introduction of pollutants into the sewer system in quantities that might:

1. interfere with the operation of the wastewater treatment plant and collection system;

2. pass through the wastewater treatment plant to pollute the Willamette River;

3. contaminate the biosolids produced at the wastewater treatment plant, preventing biosolids use as an agricultural fertilizer; or

4. cause health and safety risks to the public and wastewater treatment system personnel.

In order to fulfill the purpose of the pretreatment program, pollutants from industrial sources are regulated. The regulated pollutants include substances that cause fire or explosion in the sewer system, corrode the sewer system, cause sewer flow obstruction, or adversely affect the operation of the city treatment system.

Wastewater from industries is required to be treated to remove excessive amounts of pollutants before discharge to the city sewer. This pretreated industrial wastewater is regulated by the city's industrial wastewater pretreatment program.

In order to ensure that the industry knows to what levels it must reduce its pollutants, an industrial wastewater discharge permit is issued by the city. The discharge permit specifies the maximum amounts of pollutants that the industry can discharge into the sewer system.

Industries issued discharge permits are required to submit periodic monitoring reports to demonstrate compliance with discharge

limits. The city also collects samples and conducts inspections to verify compliance.

When violations occur, the city responds with an enforcement action. The level of enforcement corresponds to the nature and degree of the violation. Response actions range from a telephone notification to penalties of up to $20,000 per violation per day.

Through the implementation of a comprehensive pretreatment program, the City of Corvallis is committed to working with industry in a cooperative spirit to enhance community livability by protecting the environment and the community's investment in the wastewater treatment system.

SOLIDS TREATMENT PROCESSES

Under the federal Part 503 rules, pathogen and vector attraction reduction for biosolids is required to control product quality and protect public health. Biosolids can be categorized as Class A or Class B, depending on pathogen reduction methods. Class A biosolids require pathogen reduction processes that reduce pathogen levels below specified limits. Class B requirements are less stringent and reduce pathogens to levels that are unlikely to pose a threat to public health as long as management practices are controlled. Pathogen and vector attraction reduction (or stabilization) occur simultaneously at the Corvallis WWRP during anaerobic digestion. Additional pathogen reduction and stabilization occur during the period when biosolids are held in the facultative lagoons.

Vector Attraction Reduction

The purpose of vector attraction reduction processes is to eliminate the transport of pathogens to humans or animals by vector transmission. Vectors can include insects, rodents, and birds. Biosolids that are stabilized to reduce or eliminate vector attraction are also much less likely to produce offensive odor.

The vector attraction reduction requirement of the EPA 503 rule is met by reducing the volatile suspended solids by at least 38 percent [Option 1, 503.33(b)(1)]. The volatile solids reduction during digestion at Corvallis is approximately 60 percent. Additional (but unmeasured) volatile solids loss occurs during lagoon storage. Reducing the volatile solids by at least 38 percent produces a stable product that will not generate offensive odors and attract vectors.

The Corvallis WWRP greatly exceeds requirements for this quality parameter.

Pathogen Reduction

Pathogen reduction requirements of the 503 rule are met by monitoring the digested solids for fecal coliform [Alternative 1, 503.32(b)(2)]. Seven samples are collected each quarter. The geometric mean of the samples must be less than 2 million MPN per gram of dry solids to meet Class B requirements. Corvallis biosolids greatly exceed these requirements (Table 1). These results are a reflection of long digester detention times and additional treatment provided by the facultative storage lagoons. Pathogen reduction requirements are also met through use of a "Process to Significantly Reduce Pathogens," or PSRP [503.32(b)(3)]; anaerobic digestion for >15 days @ 35 degrees C. Site restrictions are practiced when land applying Class B biosolids. These include restrictions for harvesting crops, grazing animals and public contact.

FACULTATIVE TREATMENT AND STORAGE

The City of Corvallis has two facultative solids storage basins (SSB9100 and SSB9200) with an area of 2.2 acres each. The depth of each lagoon is maintained at 15 feet using a supernatant structure that returns supernatant above 15 feet to the headworks. Digested solids are discharged to the lagoons by gravity through a floating line, which results in an even distribution of solids across the bottom of the lagoon. Profiles of the facultative lagoons are performed as needed to determine when to move the floating line. The design loading rate for the lagoons is 20 pounds VS/1,000 sf/day.

Digested solids are discharged to the basins on alternate years. While solids are being fed to one lagoon, solids are being removed from the other. Isolating solids in this way provides maximum pathogen and vector attraction reduction. Solids are removed by a dredge that pumps through a floating line to the bank of the lagoon and then through a fixed line to an above ground storage tank. Circulation is provided in the storage tank to prevent solids from settling.

CONTINGENCY PLANNING

Process Failure

In the event that a spill or illegal discharge of some hazardous substance were to occur, the primary digester would need to be shutdown and cleaned. The accumulated solids would not meet pathogen and vector attraction reduction criteria and would need to be dewatered and disposed of in a landfill. If the hazardous spill killed all the biological organisms in the activated sludge and trickling filter media, those processes would need to be re-seeded with healthy organisms. These are extremely unlikely occurrences and have not occurred at the Corvallis WWRP.

Spill Control

The city is responsible for cleaning up any spills of biosolids that occur while transporting the biosolids to land application sites. Maintenance staff and equipment are available to respond to such an incident. The driver will immediately clean up any biosolids that have spilled. Any spillage that cannot be cleaned up will be covered with dry lime, and the area will be posted where appropriate.

The DEQ Western Region-Salem office (503/378-8240) will be notified within one hour in the event of any spill over 50 gallons or any other threats to the environment that occur. Spills occurring after normal DEQ working hours shall be reported to the Emergency Management Division (800/452-0311) within one hour.

BIOSOLIDS CHARACTERISTICS

Biosolids are drawn from the solids storage basins (SSBs) after nine to ten months of storage for disposal by land application. The solids are sampled for nutrients, metals, and pathogens during the hauling season, which is May through October. The required monitoring frequency depends upon the size of the facility. The City of Corvallis is a facility that produces greater than 290 but less than 1,500 metric tons of solids per year (561) dry metric tons in (2006), which corresponds to a required monitoring frequency of once per quarter (4 times per year).

Table 1 provides the biosolids characteristics that were measured for 2006. The table shows that the city's biosolids meet all pollutant concentration (PC) limits for all of the metals. Metals limits are specified in both the EPA 503 and Oregon DEQ Division 50 biosolids management rules. As long as PC limits are met, tracking long term metals addition to land application sites is not required.

Table 1. Corvallis WWRP Biosolids Characteristics

	2006					
Parameter	June	July	August	August	Mean[1]	503 PC Limits[2] (metals, mg/kg)
TS, %	6.6	8.4	7.3	7.8	7.5	
VS, %	51.0	48.3	54.8	51.2	51.3	
TKN, %	4.26	1.95	4.31	4.36	3.72	
NH_4-N, %	0.910	0.954	1.360	1.530	1.188	
NO_3-N, %	0.0001	0.0001	0.0007	0.0012	0.0005	
P, %	1.430	0.419	0.117	0.161	0.532	
K, %	0.271	0.295	0.245	0.263	0.268	
Ag, mg/kg	39.8	34.8	36.4	41.3	38.1	
As, mg/kg	4.2	6.3	6.3	6.4	5.8	41
Cd, mg/kg	3.1	3.4	3.1	2.7	3.1	39
Cr, mg/kg	53	59	50	52	53.5	
Cu, mg/kg	595	550	571	514	557.5	1500
Pb, mg/kg	51	57	46	46	50	300
Hg, mg/kg	2.38	0.61	0.68	2.5	1.54	17
Mo, mg/kg	8.9	6.9	7.8	10.1	8.4	75
Ni, mg/kg	28	32	26	27	28.2	420
Se, mg/kg	11.3	10.0	8.0	7.6	9.2	36
Zn, mg/kg	1060	1030	1070	1070	1058	2800
Fecal coliform, MPN/g[3]	21	7	6	7	10	

[1] All values are arithmetic means except fecal coliform, which is a geometric mean.
[2] PC Limits refers to Pollutant Concentration Limits (503.13, Table 3)
[3] Part 503.32 Bulk Class B Pathogen Reduction requirement are based on a geometric mean fecal coliform density of <2,000,000 fecal coliform per gram.

REPORTING

The city is required to submit an annual solids handling report to DEQ and the EPA by February 19 of the year following the calendar reporting year. The report includes an overall description of the solids handling process from generation to application, the total annual solids production, a description of the pollutant monitoring requirements and results, and descriptions of the vector attraction and pathogen reduction processes. An information and application report and pollutant loading summary for each site is also included. In addition, EPA requires that the agency that produces the biosolids provide certification that vector attraction/pathogen reduction requirements are being met. They also require certification by the applier at each site that management practices and site restrictions for Class B biosolids are being met. The annual report for 2006 is included for reference (Appendix A).

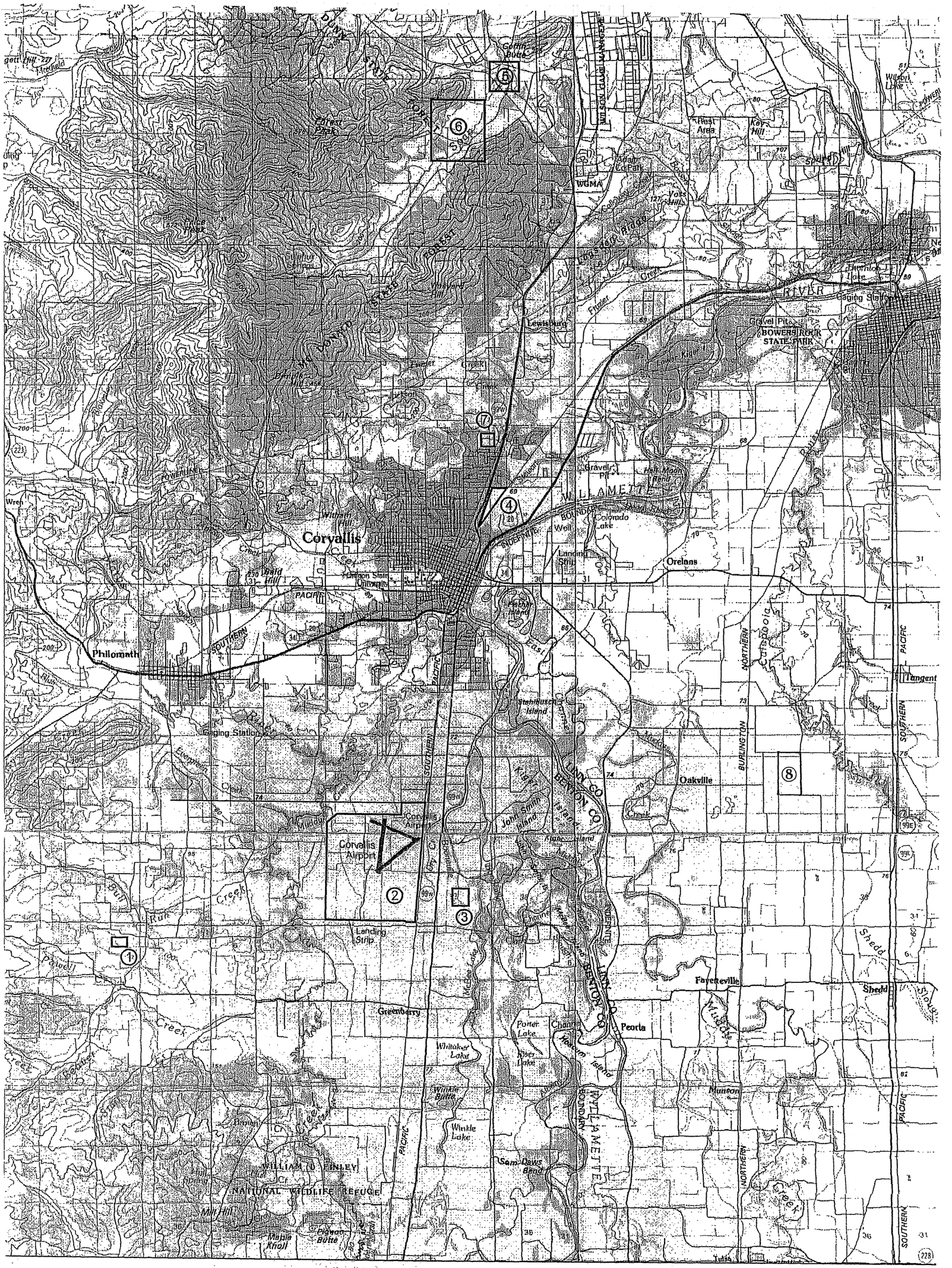
BIOSOLIDS UTILIZATION PROGRAM

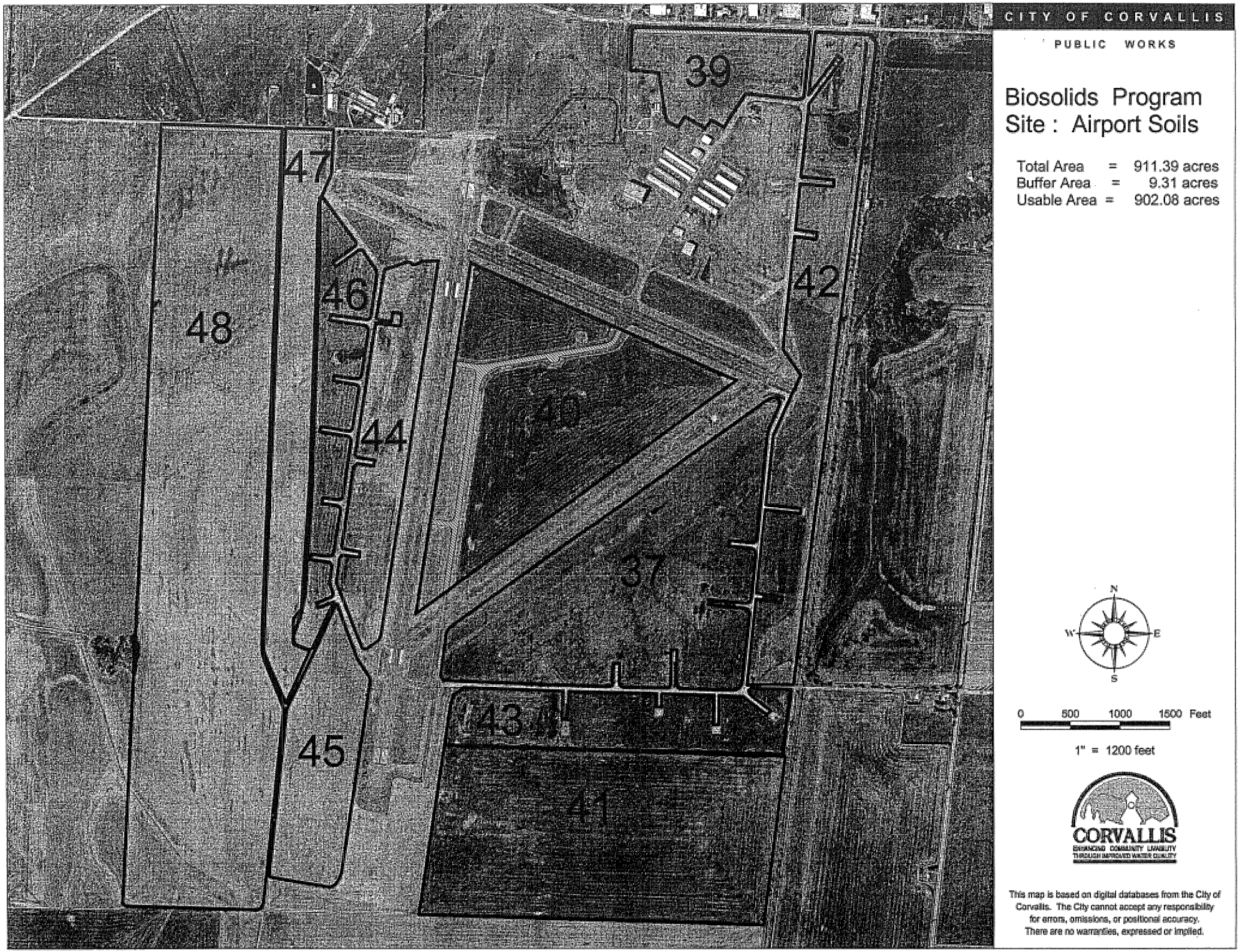
Transportation and Application

Biosolids are pumped from the above ground storage tank to tank trucks using a recirculating pump. The city owns two 6,000 gallon tank trucks to transport biosolids and to apply the biosolids directly to the site. The trucks are equipped with a splash plate attached to the back of the tank. An irrigation system can be used to surface apply biosolids, but it is only used if fields cannot support truck traffic. It has only been used once in the last ten years. If drivers employed by the city are unable to handle the entire biosolids volume at a given time, hauling contractors are hired and will remain an option if the need arises. Daily records are kept of the quantity of biosolids hauled to each site and the type of application (surface or injection).

Site Selection and Inventory

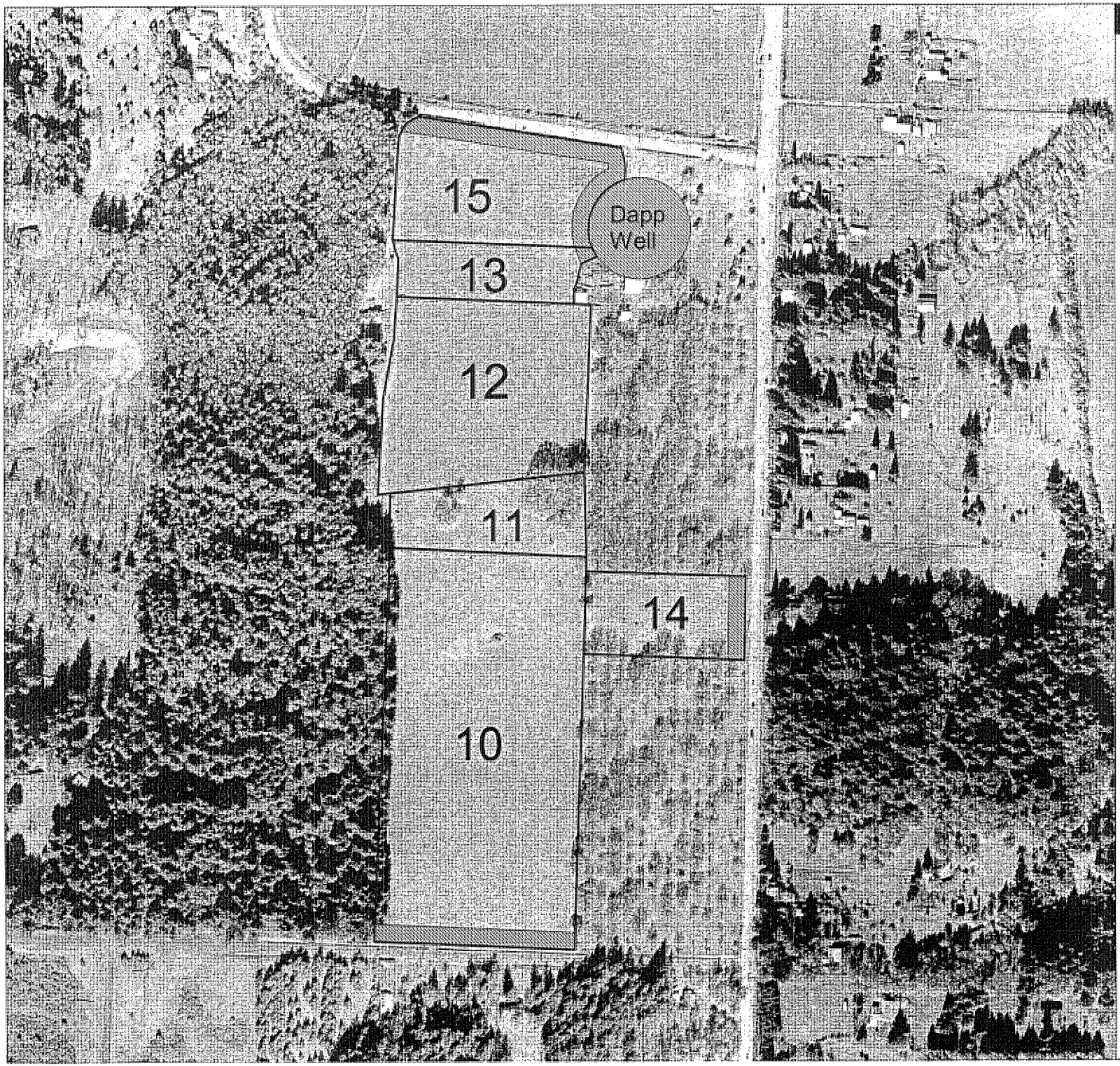
The City of Corvallis land applies Class B biosolids on DEQ approved sites. Figure 3 is a general location map of the application sites. Table 2 provides a list of the sites and the DEQ authorization date, acreage, and field number for each. Figures 4A, B, C, D, E, F and G are detailed site maps that are kept on file. DEQ site authorization letters are included in Appendix B.


Corvallis
Philomath
Lewisburg
Orleans
Oakville
Fayetteville
Peoria
Greenberry
Tangent
Gravel Pit
BOWERS ROCK STATE PARK
Corvallis Airport
Landing Strip
WILLIAM L. FINLEY NATIONAL WILDLIFE REFUGE
Oregon State University
Coffin Butte
Colorado Lake
Half Moon Bend
Porter Lake
Riser Lake
Whitaker Lake
Winkle Lake
Sam Daws Bend
Munson
Shedd
Albany
Peoria Road
Pacific
Southern Pacific
Burlington Northern
Willamette River
Marys River
Benton Co.
Linn Co.
Polk Co.
Yamhill Co.
Willamette Forest
Powerline
Landing Strip
Well
Gaging Station
Mill Hill
Wildlife Area
WMA
Campground
Boat Ramp
Kiger Island
Stahlbusch Island
John Smith Island
Pigeon Butte
Marys Peak
Kiger Hill
Vake Hill
Wilson Lake
Harrison Lake
Fitzer
Lewis and Clark
Soap Creek
Dry Creek
Muddy Creek
Mill Creek
Shedd
Sloughs
Tidbits
Wren
Bald Hill
Witham Hill
Mahlie Knoll
Pigeon Butte
Sam Daws Bend
Tidbits
Gaging Station
Gravel Pit
Well
Landing Strip
Boat Ramp
Campground
Wildlife Area
WMA



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FIGURE 4A



CITY OF CORVALLIS

PUBLIC WORKS

Biosolids Program
Site : Anderson Soils

Total Area = 35.46 acres
Buffer Area = 2.10 acres
Usable Area = 33.36 acres

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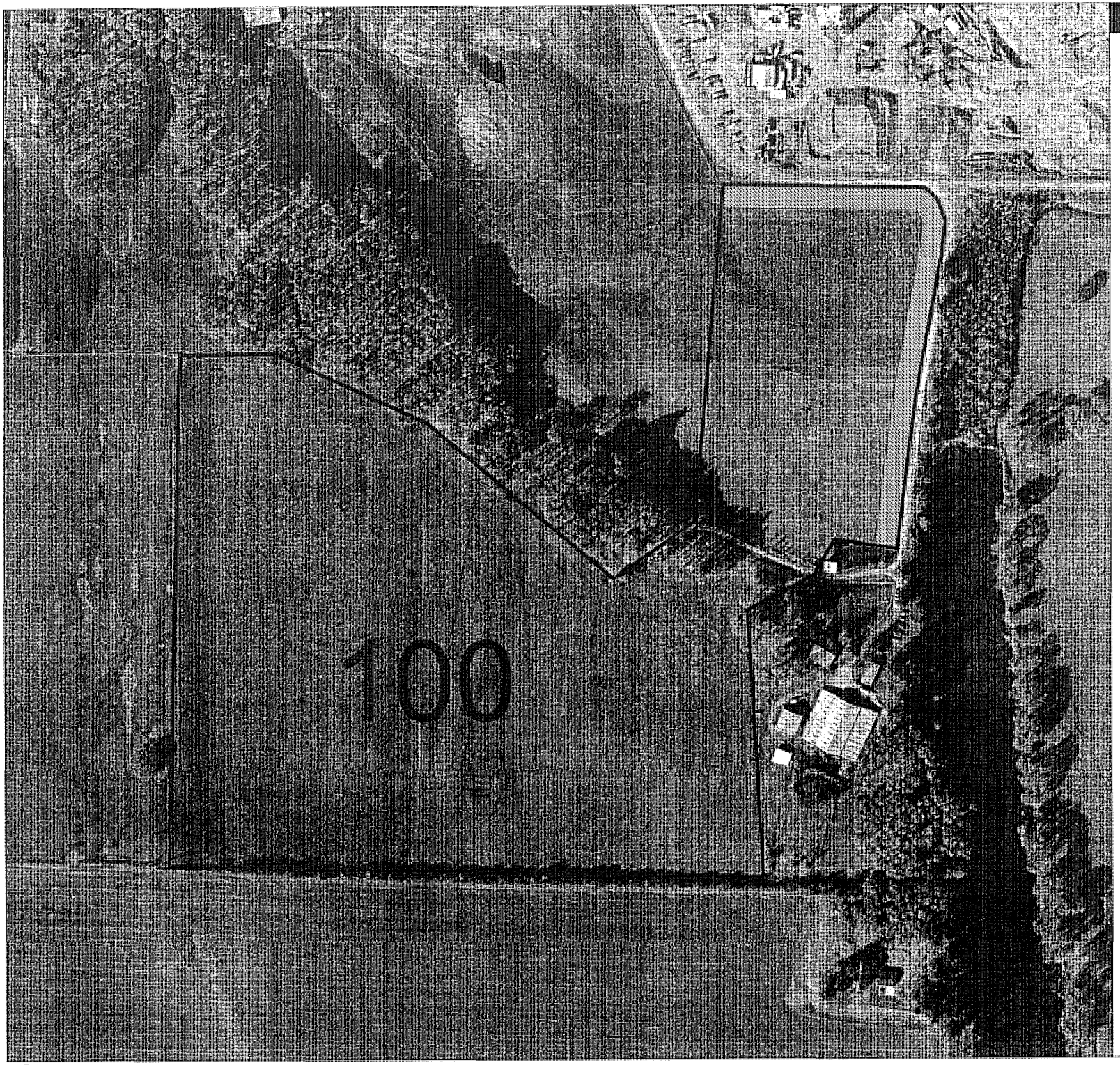
1" = 400 feet

CORVALLIS
ENHANCING COMMUNITY LIVABILITY THROUGH IMPROVED WATER QUALITY

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FIGURE 4B



CITY OF CORVALLIS

PUBLIC WORKS

Biosolids Program
Site : Bressler 100

Total Area	=	36.54 acres
Buffer Area	=	1.43 acres
Usable Area	=	35.11 acres

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1" = 300 feet

CORVALLIS
ENHANCING COMMUNITY LIVABILITY THROUGH IMPROVED WATER QUALITY

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FIGURE 4C



CITY OF CORVALLIS

PUBLIC WORKS

Biosolids Program Site : Gray Soils

Total Area	=	168.41 acres
Buffer Area	=	10.95 acres
Usable Area	=	157.46 acres

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1" = 1000 feet

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FIGURE 4D



CITY OF CORVALLIS

PUBLIC WORKS

Biosolids Program Site : OSU Soils

Total Area	=	253.29 acres
Buffer Area	=	15.73 acres
Usable Area	=	237.56 acres

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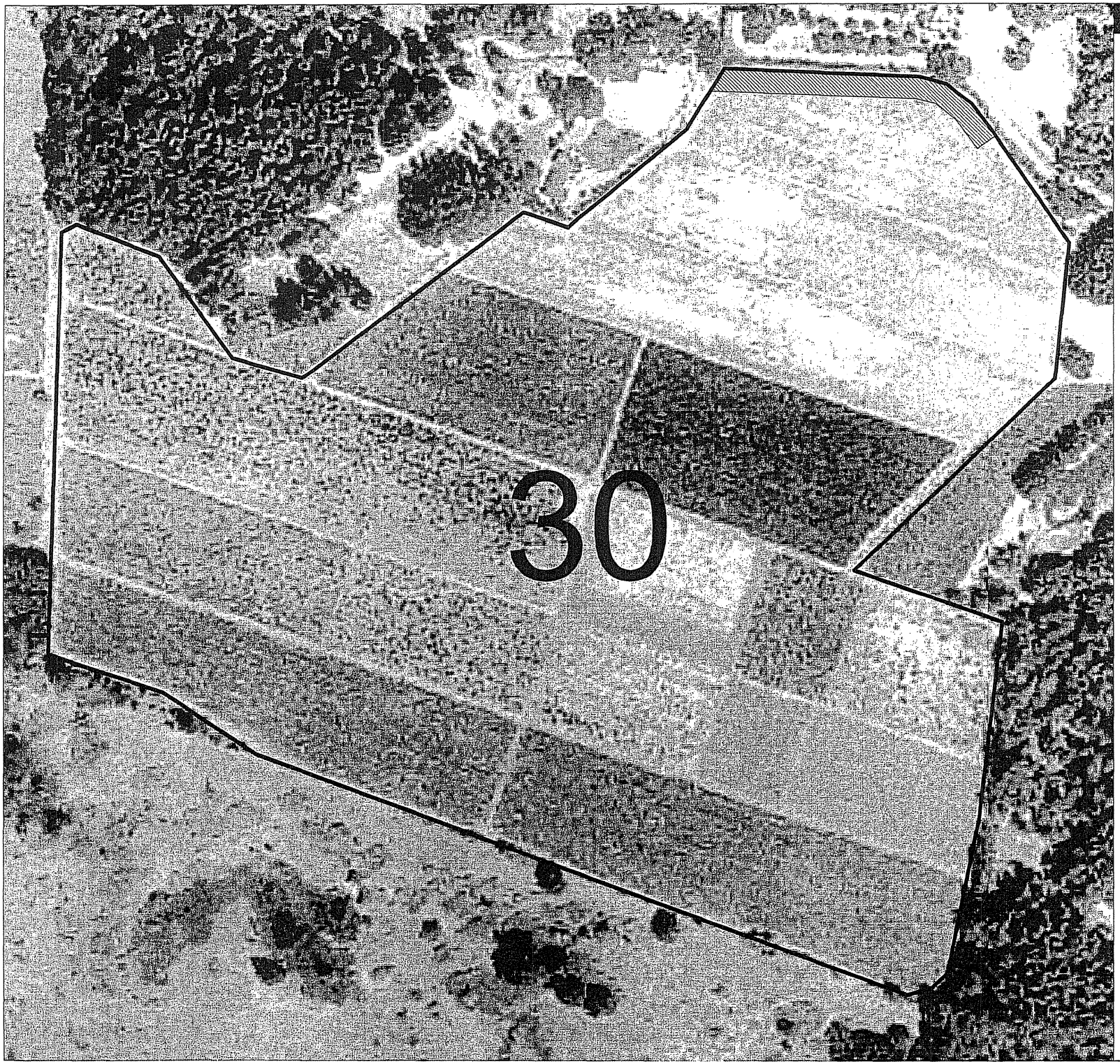
1" = 1000 feet

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FIGURE 4E



CITY OF CORVALLIS

PUBLIC WORKS

Biosolids Program
Site : Sanders 30

Total Area	=	67.86 acres
Buffer Area	=	0.77 acres
Usable Area	=	67.09 acres

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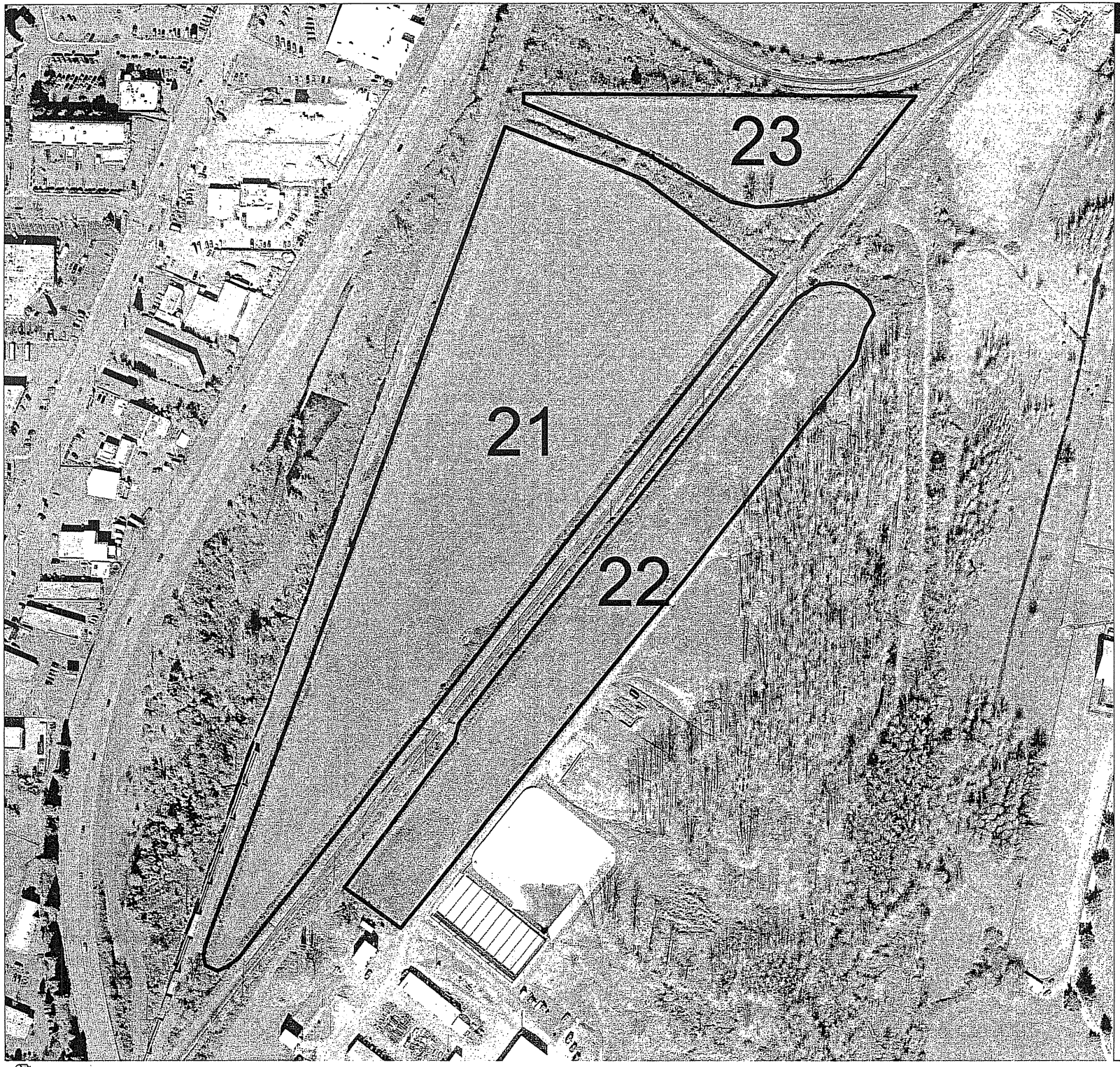
1" = 300 feet

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FIGURE 4E



CITY OF CORVALLIS

PUBLIC WORKS

Biosolids Program Site : DLD Soils

Total Area = 27.37 acres
Buffer Area = 0.00 acres
Usable Area = 27.37 acres

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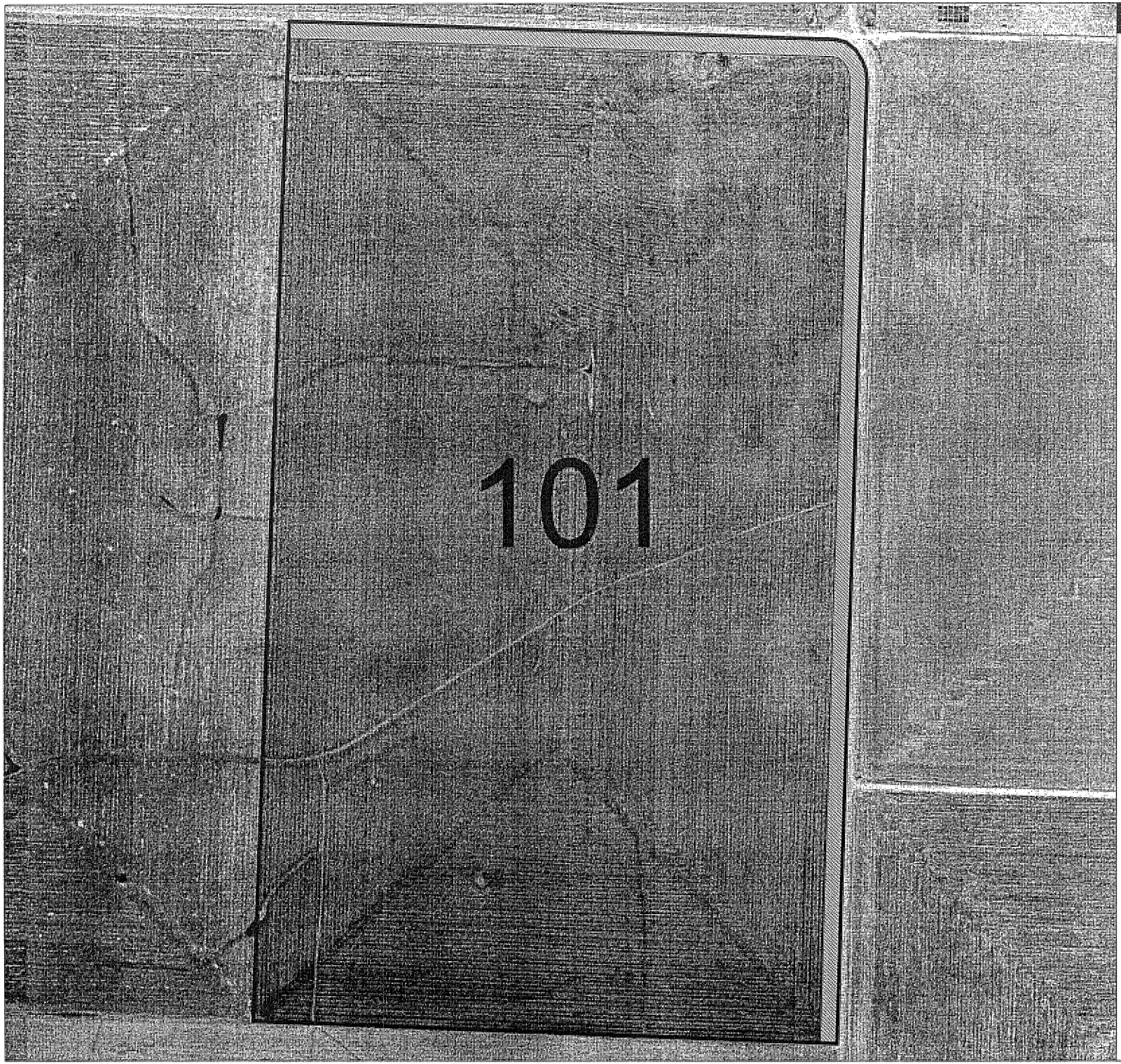
1" = 300 feet

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FIGURE 4C



CITY OF CORVALLIS

PUBLIC WORKS

Biosolids Program
Site : Pimm 101

Total Area = 117.13 acres
Buffer Area = 5.28 acres
Usable Area = 111.85 acres

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1" = 400 feet

CORVALLIS
ENHANCING COMMUNITY LIVABILITY THROUGH IMPROVED WATER QUALITY

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FIGURE 4H

Table 2. Land Application Sites

Site name	DEQ authorization date	Acres	Field Number(s)
Airport	May 1999	902.08	37, 39-48
Anderson	May 1981	33.36	10-15
	Modified January 2011		16
OSU	August 1999	237.92	70,75-78,80
Corvallis Dedicated Site	May 1999	27.37	21-23
Pimm Farm	August 2007	111.85	101
Gray	June 1995	157.46	60, 62-69
Sanders 30	August 1992	67.09	30
Bressler 100	June 1996	35.11	100
Total acres		**1572.24**	

Site approval must be consistent with local land use plans. The following site conditions must be considered when determining the suitability of a site for land application:

- Stability of the geologic formation of the site and the possibility of flooding.
- Minimum depth to permanent groundwater should be 4 feet.
- Topography of the site–liquid sludge should not be applied on slopes greater than 12 percent.

I. Minimum soil rooting depth of 24 inches.

- Proximity to residential areas–a buffer strip shall be provided with the size depending upon the type of application.
- Proximity to well-traveled highways and waterways–a buffer strip shall be provided.

These and other site selection considerations are outlined in DEQ's biosolids management rules (OAR 340 Div. 50), which are provided in Appendix C. An adequate land base is available for current and future operations. Approximately 400 acres are used in a given year. More than three times this amount of land has been authorized by DEQ. Authorization to additional sites owned by OSU is pending. New sites would be in the general vicinity shown on the area map

Application Rates

Based on 2006 nutrient levels, Corvallis biosolids provide 27.2 pounds of nitrogen per dry ton of solids. This assumes a mineralization rate of 30 percent and a 50 percent loss of ammonia due to volatilization. The maximum application rate differs for each site, but typical application rates for different crops are provided in Table 3.

Table 3. Maximum Biosolids Application Rates (Typical)

Crop	N required lb/ac	Biosolids DT/ac[1]	Biosolids gal/ac[1]	Biosolids inches/ac
Grass pasture/grass hay[2]	100	3.7	14,532	0.54
Grass seed	130	4.8	18,892	0.70

[1] At average nutrient and solids concentrations listed for 1998.
[2] Assumes a healthy stand and 5 ton/ac harvest.

Application rates may vary depending on solids concentration and percent TKN and NH_4-N. An example calculation is provided below.

Average biosolids N concentrations (Table 1)

TKN, % 3.72
NH_4-N, % 1.19
NO_3-N, % 0

Available N calculation

% effective inorganic N =
(%NH_4-N)(0.5[1]) + (%NO_3-N) = (1.3%)(0.5) + (0%) = **0.60**

% organic N = (%TKN) – (%NH_4-N) = 3.67 – 1.3 = **2.53**

lbs mineralized organic N/dry ton (DT) biosolids =
(%organic N/100)(mineralization rate/100)(2000 lb/DT) =
(2.37/100)(30[2]/100)(2000) = **15.2 lbs N/DT**

lbs effective inorganic N/DT = (% eff. Inorganic N/100)(2000 lb/DT) =
(0.65/100)(2000) = **12 lbs N/DT**

Total lbs available N/DT = mineralized organic N + effective inorganic N =14.2 + 13 = **27.2 lbs N/DT**

[1] 50 percent loss of NH_4-N due to volatilization
[2] Mineralization rate for lagoon-stored biosolids is 30 percent

For this example, the biosolids application rate for a site requiring 100 lbs N/ac would be (100 lbs N/ac)/(27.2 lbs N/DT), or 3.7 DT/ac, as shown in Table 3.

Additional information on biosolids application rates and nutrient management is provided in the OSU Extension publication included as Appendix D.

Soil Sampling and Testing

Sampling and testing of application site soils is conducted in the event of routine annual application to the same field exceeding 2 out of 3 years. Available N in the soil profile is factored into the agronomic rate calculations such that biosolids application may be reduced to account for carry-over from previous applications.

FUTURE BIOSOLIDS PROGRAM ENHANCEMENTS

The Corvallis WWRP has maintained a successful biosolids land application program for nearly 20 years. Facultative lagoon treatment and storage for biosolids has resulted in the ability to deliver a quality product at times when it is most needed at cooperating farms. Additional farm sites will be added in the near future to provide even greater flexibility in scheduling operations and product delivery. One of the goals of operations staff is to make the product available to a wider user group, within reasonable limits for hauling distance and cost.

Historically, WWRP staff have relied on a computerized management system for calculating application rates and maintaining records. A shift is underway to a more "hands-on" approach to provide greater operator involvement and knowledge of variables such as biosolids nutrient value and crop requirements. Soil samples will be collected at sites that receive frequent biosolids application to help fine tune application rates.

Staff are also bringing the county-wide GIS (graphic information system) into the program as a tool for tracking application sites and record keeping. Aerial photo-based site maps (as represented in previous sections of this report) provide an excellent method for assisting with application site management and monitoring, and the use of the GIS system will continue to be refined.

REFERENCES

National Research Council. 1996. Use of Reclaimed Water and Sludge in Food Crop Production. National Academy Press, Washington, D.C.

USEPA. 1995. A Guide to the Biosolids Risk Assessment for the EPA Part 503 Rule. EPA/832/B-93/005.

USEPA. 1994. A Plain English Guide to the EPA Part 503 Biosolids Rule. EPA/832-R-93/003.

Sullivan, Dan. 1998. Fertilizing with Biosolids. PNW508, Oregon State University, Corvallis.

APPENDIX A

2006 ANNUAL REPORT

CITY OF CORVALLIS
BIOSOLIDS MANAGEMENT PROGRAM

DEQ/EPA ANNUAL REPORT - 2006

I. Background/General

A. Facility: City of Corvallis Wastewater Reclamation Plant
P.O. Box 1083 Corvallis OR 97339
Longitude: 123 deg. 15' Latitude: 44 deg. 34'

NPDES Permit # : 100984, DEQ File 20151
Facility contact : Dan Hanthorn, Wastewater Operations Manager

B. Reporting Period: Calendar year 2006

C. Narrative Description:

1. Sludge generation process: The City of Corvallis operates a secondary treatment plant using both trickling filter and activated sludge processes. In this process a portion of the biological solids generated in the secondary system is wasted to the gravity thickeners where it is combined with raw sludge and thickened. It is then pumped to one of the primary digesters. An average of 30,310 gallons of sludge is pumped daily to the anaerobic digester. The sludge is then heated to 96 deg. F for approximately 26 days prior to discharge to one of two facultative lagoons where the sludge is stored for eight to nine months prior to land application.

- Two headworks structures with mechanically raked bar screens.
- Five primary clarifiers, four rectangular with aerated grit removal and one circular.
- Two standard rate trickling filters.
- Two aeration basins with fine bubble diffusers and mechanical mixers.
- Two circular secondary clarifiers.
- One circular chlorine contact basin.
- Two gravity thickeners.
- Two 800,000 gallon primary digesters, mechanically mixed , hot water heated by dual fuel boilers. Operation is in the mesophilic temperature range.
- Two facultative solids storage basins (2.2 acres each).

Background/General (cont.)

2. Biosolids application process:

Biosolids are pumped from the facultative storage lagoon by a manned dredge to an above ground storage tank. Biosolids are recirculated continuously to prevent solids separation. Trucks load from the storage tank with the recirculating pump. Biosolids are then applied at agronomic rates to approved sites with two 6,000 gallon tank trucks once crops have been harvested and the ground is stable enough to support the trucks. Biosolids application is restricted to dryer months and typically runs between May, 15 and October, 15.

- two 6,000 gallon tank trucks
- one off-road 3,000 gallon injection truck

3. Process Flow Diagram (See attachment A1)

D. Total Annual Production:

1. From Corvallis to sites:

		Metric	U.S
a.	Tons generated:	510	561
b.	Area covered:	45 ha	112 ac

E. Pollutant Monitoring: 40 CFR 503.16

1. Required frequency is four per year based on >290 but < 1500 metric tons applied per 365 day period. Samples are taken during the active season, from May through October. Application of biosolids is regulated by the weather and harvest, and can start as late as August. (See attachment A2)

F. Management Practices: 40 CFR 503.14

1. Biosolids shall not be applied to the land if it is likely to adversely affect a threatened or endangered species listed under section 4 of the Endangered Species Act or its designated critical habitat.

2. Biosolids shall not be applied to agricultural land, forest, a public contact site, or a reclamation site that is flooded, frozen, or snow-covered so that the biosolids enters a wetland or other waters of the United States, as defined in 40 CFR 122.2, except as provided in a permit issued pursuant to section 402 or 404 of the CWA.

Management Practices(cont.)

3. Biosolids shall be applied to agricultural land, forest, a public contact site or a reclamation site at a whole sludge application rate that is equal to or less than the agronomic rate for the biosolids, unless, in the case of a reclamation site, otherwise specified by the permitting authority.

4. A 50-foot setback shall be maintained between biosolids application areas and all highways, public roadways, and property lines.

5. A minimum setback of 200 feet shall be maintained from biosolids application areas and all wells and other water sources.

6. A minimum setback of 50 feet shall be maintained from seasonal drainage ways and 100 feet from Soap Creek.

7. Biosolids shall be applied evenly and thinly in a manner that will prevent ponding and runoff.

8. No biosolids shall be applied, when within the prior 24 hours, 0.25 inches of rain or greater has fallen in the Corvallis area.

9. No food crops for direct human consumption, whose harvested parts are grown below ground, shall be planted on any biosolids amended area for at least 38 months following the last biosolids application.

10. No food crops for direct human consumption, whose harvested parts are grown above ground (e.g., tomatoes) shall be grown on any biosolids amended area for at least 14 months following the last biosolids application.

11. A 30-day fallow period shall follow the application of biosolids prior to grazing livestock or harvesting animal feed crops from the amended area.

12. Public access to field sites shall be restricted for at least 12 months after biosolids application has ceased.

13. In the event an odor problem is reported at any land application site, immediate steps, such as, but not limited to, the addition of liming materials, must be taken to counteract that condition.

14. The City shall maintain daily records for the biosolids application sites. The records shall show where, when, and how many gallons were applied.

CITY OF CORVALLIS
BIOSOLIDS MANAGEMENT PROGRAM

DEQ/EPA ANNUAL REPORT - 2006

I. Vector Attraction Reduction: 40 CFR 503.33

A. Facility: City of Corvallis Wastewater Reclamation plant
P.O. box 1083 Corvallis OR 97339
Longitude: 123 deg. 15' Latitude: 44 deg. 34'

NPDES Permit # : 100984, DEQ File 20151
Facility contact : Dan Hanthorn, Wastewater Operations Manager

B. Reporting Period: Calendar year 2006

2. Process Description:

A. Primary digester (2)

- 800,000 gallon capacity each (only one on-line at a time)
- Average volume raw sludge pumped to primary digester; 30,310 gals. @ 78%vs.
- Sludge detention time averaged 26 days
- Digester temperature averaged 96.0 degrees F.
- Digester average pH was 7.5
- The average concentration leaving the digester was 1.9% @ 57% vs.
- The average volatile solids reduction for the digester was 62%.

B. Facultative solids storage basins (2)

- 2.2 acres each
- total depth 15 ft.
- 51% volatile solids (4 per yr)(season average)
- 4.7 % solids removed (season average)
- Nutrient analysis (4 per yr)
- Metals analysis (4 per yr)
- Operation: Solids enter the SSB from the anaerobic digester by gravity through a floating line. This allows the sludge to be discharged below the surface of the water, with the normal wave action on the SSB's, solids are evenly distributed across the bottom. Profiles of the SSB's are preformed on a quarterly basis to determine if the floating line needs to be moved.

Lagoons are alternated every year, feeding one and removing solids from the other. Removal of the solids is done with a floating dredge that pumps through a floating line to the bank and into fixed piping to an above ground storage tank. The storage tank is recirculated to keep the solids in suspension.

3. Alternative 1, 40 CFR 503.33 (b)(1)

Has the mass of volatile solids in the sewage sludge been reduced by at least 38 percent? Yes

Frequency: Volatile solids reduction is verified __4__ times per season.

CITY OF CORVALLIS
BIOSOLIDS MANAGEMENT PROGRAM

DEQ/EPA ANNUAL REPORT - 2006

I. Pathogen Reduction: Class B - Alternative 1, 40 CFR 503.32(b)(2)

A. Facility: City of Corvallis Wastewater Reclamation plant
P.O. box 1083 Corvallis OR 97339
Longitude: 123 deg. 15' Latitude: 44 deg. 34'

NPDES Permit # : 100984, DEQ File 20151
Facility contact : Dan Hanthorn, Wastewater Operations Manager

B. Reporting Period: Calendar year 2006

2. Sampling and Analysis:

A. During sampling, were seven samples of biosolids collected for analysis? Yes

B. Was the geometric mean of the density of fecal coliform sample collected less than either (I) 2,000,000 Most Probable Number (MPN) per gram of total solids (dry weight basis) OR (ii) 2,000,000 Colony Forming Units (CFU) per gram of total solids (dry weight basis)? Yes

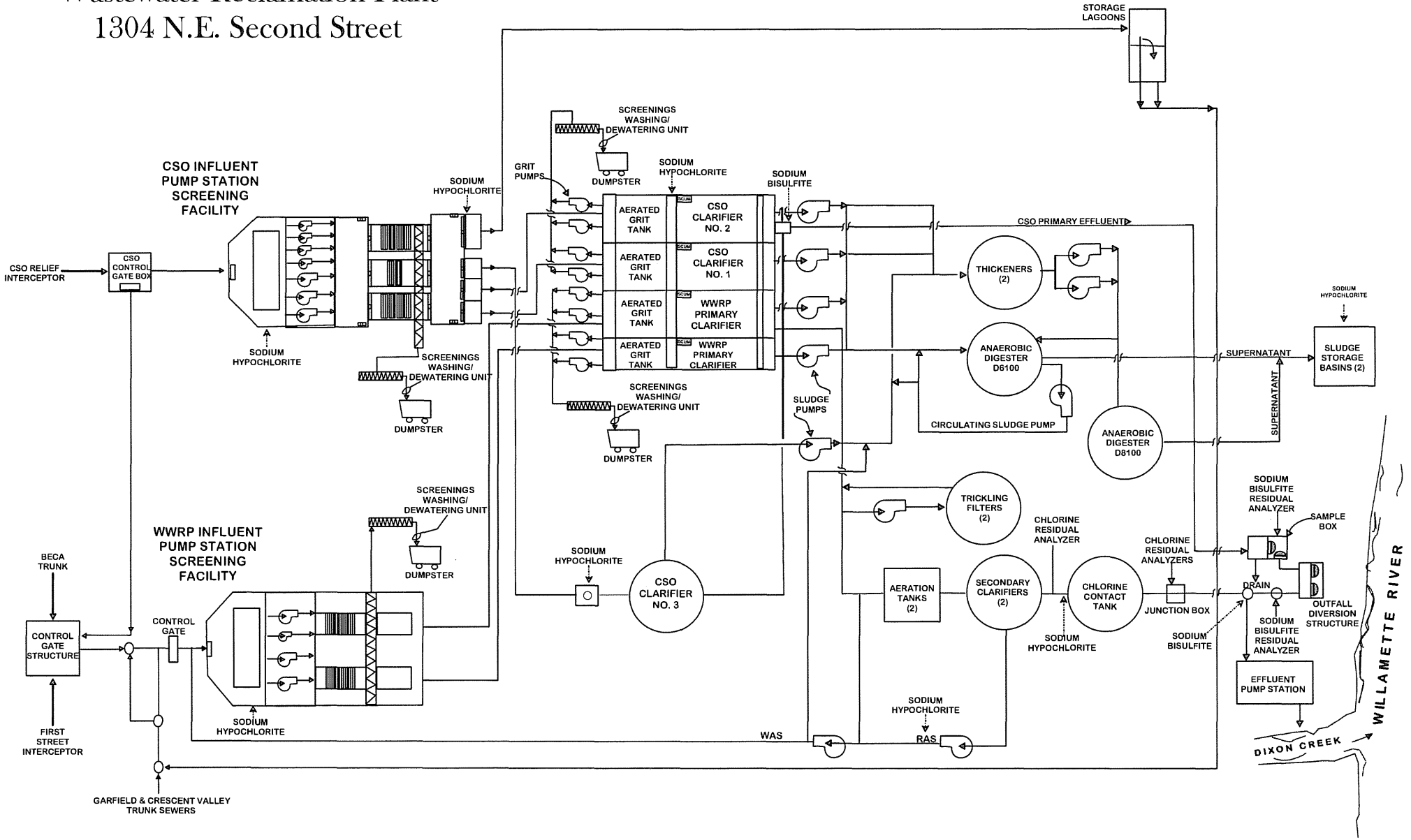
3.Monitoring Data

Parameter	Allowable level		Pathogen density		Frequency of	Sample	Analytical
Fecal	In sludge /	Date /	Value /	Units /	Analysis /	Type /	Technique
Coliform	$2*10^6$ MPN	6/06/06	21	MPN	4 per season	Grab	SM9221C
		7/25/06	7	MPN			
		8/01/06	6	MPN			
		8/22/06	7	MPN			

4. Site restrictions:

Public access is restricted for one year, crop harvest and grazing are restricted for 30 days, after the final application of biosolids.

City of Corvallis
Wastewater Reclamation Plant
1304 N.E. Second Street



ATTACHMENT A1

CITY OF CORVALLIS WASTEWATER RECLAMATION PLANT
BIOSOLIDS ANALYSIS FOR AGRICULTURAL USE
(SSB 9200 SLUDGE STABILIZATION BASIN)

All values in mg/kg dry weight basis unless otherwise noted

DATE	%TS	%VS	%P	%NH_3-N	%NO_3-N	%TKN-N	Ag	As	Cd	Cr	Cu	Pb	Hg	Mo	Ni	Se	Zn	K
06/06/06	6.60	51.0	1.430	0.910	0.00009	4.26	39.8	4.2	3.10	53	595	51	2.38	8.90	28	11.3	1060	2710
07/25/06	8.40	48.3	0.419	0.954	0.00012	1.95	34.8	6.3	3.40	59	550	57	0.61	6.90	32	10.0	1030	2950
08/01/06	7.30	54.8	0.117	1.360	0.00068	4.31	36.4	6.3	3.10	50	571	46	0.68	7.80	26	8.0	1070	2450
08/22/06	7.80	51.2	0.161	1.530	0.00123	4.36	37.7	6.4	2.70	52	514	46	2.50	10.10	27	7.6	1070	2630

ATTACHMENT A2

MANAGEMENT PRACTICES CERTIFICATION 2006

POTW: City of Corvallis WWRP

Source Name: SSB 9200 - Class B

Source Period: Jan-1-2006 - Dec-31-2006

I certify, under penalty of law, that the management practices in 503.14 and the site restrictions in 503.32(b)(5)(v) have been met. This determination has been made under my direction and supervision in accordance with the system designed to ensure that qualified personnel properly gather and evaluate the information used to determine that the management practices and site restriction requirements have been met. I am aware that there are significant penalties for false certification including the possibility of fine and imprisonment.

Signature : _____ Date : _____

PATHOGEN CERTIFICATION
2006

POTW: City of Corvallis WWRP

Source Name: SSB 9200 - Class B

Source Period: Jan-1-2006 - Dec-31-2006

I certify, under penalty of law, that the Class B pathogen reduction requirement in 503.32(b) Alt (2)(3) has been met. This determination has been made under my direction and supervision in accordance with the system designed to ensure that qualified personnel properly gather and evaluate the information used to determine that the Class B pathogen reduction requirement has been met. I am aware that there are significant penalties for false certification including the possibility of fine and imprisonment.

Signature : _____ Date : _____

VECTOR ATTRACTION REDUCTION CERTIFICATION 2006

POTW: City of Corvallis WWRP

Source Name: SSB 9200 - Class B

Source Period: Jan-1-2006 - Dec-31-2006

I certify, under penalty of law, that the Class B vector attraction reduction requirement in 503.33(b)(1) has been met. This determination has been made under my direction and supervision in accordance with the system designed to ensure that qualified personnel properly gather and evaluate the information used to determine that the Class B vector attraction reduction requirement has been met. I am aware that there are significant penalties for false certification including the possibility of fine and imprisonment.

Signature : _____ Date : _____

BDMS FACILITY REPORT

Date Printed: 1/3/2007

Facility Name **CORVALLIS WWRP** Permit Number **OR002636-1**

GENERAL FACILITY INFORMATION

Annual Report Submitte 1/3/2007
Alternate Permit# 101714
Region X
Hydrologic Unit 22
Field Office

EPA ID Code
Biosolids Facility Indicator (S/O/E
Authority Name CORVALLIS, CITY OF
Use for Analysis
Latitude 44.574 Longitude 123.253
Plant Type TF/AS

Facility Physical Address:
Street 1304 NE 2ND
Zip Code 97330
Phone 541-766-6934
County BENTON
City/St CORVALLIS, OR

Facility Mailing Address:
Street PO BOX 1083
Zip Code 97339
Phone 541-766-6934
City/St CORVALLIS, OR
County BENTON

Operator-in-charge DAN HANTHORN
Title PLANT SUPERVISOR Phone 541-754-1757 Fax 541-766-6753
License No. T-4571 Certification Grade IV

Responsible Official DAN HANTHORN
Title PLANT SUPERVISOR

Contact Person GENE FREEL
Title BIOSOLIDS COORDINATOR E-mail GENE.FREEL@CI.CORVALLIS.OR.US
Phone 541-766-6729 EXT527 Fax 541-766-6753

Permit Issued 08/01/1999 Permit Expires 06/01/2004 Last Inspection Date 06/30/2005
Influent Flow (MGD) 0.00 Influent TSS 0.00 Influent BOD (mg/L) 0.00
Activity Status? (A/I) A MaJor/MiNor? (J/N) J Pretreatment? (Y/N) Y
Indian Lands? (Y/N) N Class 1? (Y/N) Y Report Required? Y
Facility send biosolids out of state? (Y/N) N

BDMS FACILITY REPORT

Date Printed: 1/3/2007

Facility Name **CORVALLIS WWRP** Permit Number **OR002636-1**

CONTRACT APPLIER(S)/HAULER(S)

Name

Street

City/State Zip

County

Contact Phone

Amount of Use: (Majority/Occassional)

BDMS FACILITY REPORT

Date Printed: 1/3/2007

Facility Name **CORVALLIS WWRP** Permit Number **OR002636-1**

BIOSOLIDS TREATMENT PROVIDED (No. of Units or 999 as default)

THICKENING:

1.	Gravity	0
2.	DAF	0
3.	Centrifuge	0
4.		0

STABILIZATION:

5.	Aerobic Dig.	0
6.	Anaerobic Dig.	2
7.	Heat Treat	0
8.	Wet Oxidation	0
9.	Chemical (Lime	0
10.	Composti	0
11.	Biosolids Lagoons	2
12		0

CONDITIONING:

13.	Chemical Cond.	0
14		0

DEWATERING:

15.	Vacuum Filter	0
16.	Pressure Filter	0
17.	Belt Filter	0
18.	Drying Bed	0
19.	Drying Lagoon	0
20.	Heat Drying	0
21.	Centrifuge	0
22		0

OTHER:

23.	Wastewater Lagoon	0
24.	Mixing of Biosolid	0
25.	Oxidation Ditch	0
26.	Incineration	0
27.	Septage	0
28		0

BDMS FACILITY REPORT

Date Printed: 1/3/2007

Facility Name **CORVALLIS WWRP** Permit Number **OR002636-1**

FINAL USE/DISPOSAL PRACTICES (dmt/yr) FOR MONITORING YEAR 2006

TOTAL ANNUAL PRODUCTION (Predigestion)		0.00	
TOTAL ANNUAL PRODUCTION (Postdigestion)		510.00	
LAND APPLICATION (Total)	510.00		
Bulk Biosolids:	510.00	Derived Material:	0.0
Agricultural Land	510.00	Agricultural Land	0.0
Range Land	0.00	Range Land	0.0
Forest	0.00	Forest	0.0
Public Contact Site	0.00	Public Contact Site	0.0
Reclamation Site	0.00	Reclamation Site	0.0
Sold or Given Away	0.00	Sold or Given Away	0.0
Lawn or Garden	0.00	Lawn or Garden	0.0
SURFACE DISPOSAL (Total)	0.00		
With Liner & LCS	0.00		
Without Liner & LCS	0.00		
LANDFILL (Total)	0.00		
Landfill Disposal	0.00	Landfill Cover	0.0
Landfill Name			
Does landfill meet 40CFR258? (Y/N)			
INCINERATION	0.00		
TRANSPORTED TO ANOTHER FACILITY(S)	0.00	**RECEIVED FROM ANOTHER FACILITY(S)**	0.00
Name		Name	
Address		Address	
State(s)		State(s)	
Phone		Phone	
OTHER	0.00		
STORED	0.00		
CALCULATED PRODUCED	510.00		
LONG TERM TREATMENT	0.00		

Was land application used for any of the sites? (Y/N)

Pathogen Certification? (Y/N/NA)	**Y**
VAR Certification? (Y/N/NA)	**Y**
Management Practice Certification? (Y/N/NA)	**Y**
Landfill Certification? (Y/N/NA)	**NA**

BDMS FACILITY REPORT

Date Printed:

Facility Name **CORVALLIS WWRP** Permit Number **OR002636-1**

Monitoring Year **2006** Use/disposal Practice **LA BULK AGRICULTURAL LAND**
No. of Analysis required per monitoring year? **4**
Approved methods? **Y** All in dry weight? (Y/N) **Y** Required Metals? (Y/N) **Y**

Pollutant	Average	Maximum	Analysis/yr	Comments
Arsenic (mg/kg)	5.80	6.40	4	
Cadmium (mg/kg)	3.08	3.40	4	
Chromium (mg/kg)	53.50	59.00	4	
Copper (mg/kg)	557.50	595.00	4	
Lead (mg/kg)	50.00	57.00	4	
Mercury (mg/kg)	1.54	2.50	4	
Molybdenum (mg/kg)	8.42	10.10	4	
Nickel (mg/kg)	28.25	32.00	4	
Selenium (mg/kg)	9.22	11.30	4	
Zinc (mg/kg)	1,057.50	1,070.00	4	
Salmonella (#/g)	0.00	0.00	0	
Helminth OVA (#/g)	0.00	0.00	0	
Enteric Virus (PFU/g)	0.00	0.00	0	
Nitrite & Nitrate (%)	0.00	0.00	0	
TKN (%)	3.72	4.36	4	
Ammonia (%	1.19	1.53	4	
Solids (%)	7.52	8.40	4	
Phosphorus (%)	0.53	1.43	4	

	Geometric Mean	Maximum	Analysis/yr	Comments
Fecal Coliform (#/g)	10.25	21.00	4	

Paint Filter Test? (P/F) TCLP? (P/F)

LAND APPLICATION SITE INFORMATION FOR CORVALLIS WWRP OR002636-1

Monitoring Year: **2006**
Site Name: **ANDERSON** Site No: **10**
Hydrologic Unit:
Owner: **LLOYD ANDERSON**
Operator: **LLOYDANDERSON**
Applier: **CITY OF CORVALLIS**
Latitude: Longitude:
Location: **4210 NW CRESCENT VALLEY DRIVE**
Section: **14**
Township: **T11S** Range: **R5W**
Size (acres): **15.03** Size (hectares): **6.08**
Crop: **GRASS HAY**
Application Rate (tons/acre): **3.44** Application Rate (MT/HA): **7.70**

	English Units		Metric Units	
Recommended N for Crop	**100.00**	lbs/Acre	**112.00**	Kg/Ha
Nitrogen Available:	**22.50**	lbs/Dry Ton	**54.68**	Kg/Metic Ton
Total Biosolids Applied:	**51.63**	Dry Tons	**46.84**	Dry Metric Tons
Actual Application Rate:	**3.44**	Dry Tons/Acre	**7.70**	Dry Metric Tons / Ha
Total Recommended N:	**1,503.00**	lbs/field	**681.75**	Kg/Field
Total Applied N:	**1,161.70**	lbs/field	**526.94**	Kg/Field
% of Recommended N:	**77.29**	%	**77.29**	%
Total P Applied:	**1,476.65**	lbs/field	**669.80**	Kg/Field
Total K Applied:	**279.84**	lbs/field	**126.93**	Kg/Field

LAND APPLICATION SITE INFORMATION FOR CORVALLIS WWRP OR002636-1

Individual Load Tracking

Haul # / Desc	Date Hauled	Date Applied	Gallons	% Total Solids	Wet Tons	Dry Tons	lbs N / load	lbs N / DT	lbs P / load	lbs P / DT	lbs K / load	lbs K / DT
6565/1	7/1/02	7/1/02	16,800.00	4.00	70.06	2.80	73.45	26.21	100.88	36.00	9.53	3.40
6565/2	7/2/02	7/2/02	50,400.00	3.30	210.17	6.94	181.78	26.21	249.68	36.00	23.58	3.40
6565/3	7/3/02	7/3/02	5,600.00	4.50	23.35	1.05	27.54	26.21	37.83	36.00	3.57	3.40
6575/1	6/24/02	6/24/02	22,400.00	3.10	93.41	2.90	75.89	26.21	104.24	36.00	9.85	3.40
6575/2	6/25/02	6/25/02	28,000.00	3.10	116.76	3.62	94.87	26.21	130.30	36.00	12.31	3.40
6575/3	6/26/02	6/26/02	56,000.00	3.00	233.52	7.01	183.62	26.21	252.20	36.00	23.82	3.40
6575/4	6/27/02	6/27/02	56,000.00	3.40	233.52	7.94	208.10	26.21	285.83	36.00	26.99	3.40
6575/5	6/28/02	6/28/02	5,600.00	3.40	23.35	0.79	20.81	26.21	28.58	36.00	2.70	3.40
6575/6	7/1/02	7/1/02	11,200.00	4.00	46.70	1.87	48.96	26.21	67.25	36.00	6.35	3.40
6575/7	7/2/02	7/2/02	56,000.00	3.30	233.52	7.71	201.98	26.21	277.42	36.00	26.20	3.40
6575/1	7/9/03	7/9/03	11,200.00	5.90	46.70	2.76	75.56	27.42	121.24	44.00	11.57	4.20
6575/2	7/10/03	7/10/03	44,800.00	5.90	186.82	11.02	302.23	27.42	484.97	44.00	46.29	4.20
6575/3	7/11/03	7/11/03	44,800.00	6.10	186.82	11.40	312.47	27.42	501.41	44.00	47.86	4.20
6575/4	7/14/03	7/14/03	39,200.00	5.20	163.46	8.50	233.07	27.42	374.01	44.00	35.70	4.20
6575/5	7/15/03	7/15/03	39,200.00	5.60	163.46	9.15	251.00	27.42	402.78	44.00	38.45	4.20
6575/6	7/16/03	7/16/03	11,200.00	5.70	46.70	2.66	73.00	27.42	117.13	44.00	11.18	4.20
6565/1	7/10/03	7/10/03	5,600.00	5.90	23.35	1.38	37.78	27.42	60.62	44.00	5.79	4.20
6565/2	7/11/03	7/11/03	39,200.00	6.10	163.46	9.97	273.41	27.42	438.74	44.00	41.88	4.20
6565/3	7/14/03	7/14/03	44,800.00	5.20	186.82	9.71	266.37	27.42	427.43	44.00	40.80	4.20
6565/4	7/15/03	7/15/03	39,200.00	5.60	163.46	9.15	251.00	27.42	402.78	44.00	38.45	4.20
6565/5	7/16/03	7/16/03	11,200.00	5.70	46.70	2.66	73.00	27.42	117.13	44.00	11.18	4.20
6575-1	7/15/04	7/15/04	22,400.00	5.90	93.41	5.51	117.50	21.32	24.03	4.36	28.88	5.24
6575-2	7/16/04	7/16/04	44,800.00	4.80	186.82	8.97	191.18	21.32	39.46	4.40	46.99	5.24
6575-3	7/19/04	7/19/04	44,800.00	4.60	186.82	8.59	183.21	21.32	37.81	4.40	45.03	5.24
6575-4	7/23/04	7/23/04	44,800.00	4.20	186.82	7.85	167.28	21.32	34.52	4.40	41.11	5.24
6575-5	7/26/04	7/26/04	50,400.00	4.40	210.17	9.25	197.15	21.32	40.69	4.40	48.46	5.24
6575-6	7/27/04	7/27/04	28,000.00	5.00	116.76	5.84	114.66	19.64	8.52	1.46	29.07	4.98
6565-1	7/20/04	7/20/04	39,200.00	5.20	163.46	8.50	181.22	21.32	37.40	4.40	44.54	5.24
6565-2	7/21/04	7/21/04	39,200.00	4.80	163.46	7.85	167.28	21.32	34.52	4.40	41.11	5.24
6565-3	7/22/04	7/22/04	44,800.00	5.00	186.82	9.34	199.15	21.32	41.10	4.40	48.95	5.24
6565/4	7/10/06	7/10/06	44,800.00	4.50	186.82	8.41	189.15	22.50	240.43	28.60	45.56	5.42
6565/5	7/11/06	7/11/06	44,800.00	4.90	186.82	9.15	205.96	22.50	261.80	28.60	49.61	5.42
6565/6	7/12/06	7/12/06	5,600.00	3.40	23.35	0.79	17.86	22.50	22.71	28.60	4.30	5.42
6565/7	7/17/06	7/17/06	5,600.00	4.50	23.35	1.05	23.64	22.50	30.05	28.60	5.70	5.42
6565/8	7/18/06	7/18/06	5,600.00	5.00	23.35	1.17	26.27	22.50	33.39	28.60	6.33	5.42
6575/1	7/10/06	7/10/06	11,200.00	4.50	46.70	2.10	47.29	22.50	60.11	28.60	11.39	5.42
6575/2	7/11/06	7/11/06	33,600.00	4.90	140.11	6.87	154.47	22.50	196.35	28.60	37.21	5.42
6575/3	7/18/06	7/18/06	5,600.00	5.00	23.35	1.17	26.27	22.50	33.39	28.60	6.33	5.42
6565/1	7/5/06	7/5/06	11,200.00	5.90	46.70	2.76	62.00	22.50	78.81	28.60	14.94	5.42
6565/2	7/6/06	7/6/06	44,800.00	5.70	186.82	10.65	239.59	22.50	304.55	28.60	57.71	5.42
6565/3	7/7/06	7/7/06	39,200.00	4.60	163.46	7.52	169.19	22.50	215.05	28.60	40.75	5.42

Avg % Total Solids:	**4.75**
Total Gallons:	**1,248,800.00**
Total Dry Tons:	**244.31**
Gallons / Acre:	**83,087.16**

LAND APPLICATION SITE INFORMATION FOR
CORVALLIS WWRP
OR002636-1

Total Cumulative Metal Loading (Kg/Hectare)

As Average:	**0.48**
As Maximum:	**0.53**
Cd Average:	**0.26**
Cd Maximum	**0.28**
Cu Average:	**46.43**
Cu Maximum	**49.56**
Pb Average:	**4.16**
Pb Maximum	**4.75**
Hg Average:	**0.13**
Hg Maximum	**0.21**
Mo Average:	**0.70**
Mo Maximum	**0.84**
Ni Average:	**2.35**
Ni Maximum:	**2.67**
Se Average:	**0.77**
Se Maximum	**0.94**
Zn Average:	**88.08**
Zn Maximum:	**89.12**

Comments:

LAND APPLICATION SITE INFORMATION FOR CORVALLIS WWRP OR002636-1

Monitoring Year: **2006**
Site Name: **ANDERSON** Site No: **12**
Hydrologic Unit:
Owner: **LLOYD ANDERSON**
Operator: **LLOYD ANDERSON**
Applier: **CITY OF CORVALLIS**
Latitude: Longitude:
Location: **4210 NW CRESCENT VALLEY DRIVE**
Section: **14**
Township: **T11S** Range: **R5W**
Size (acres): **7.43** Size (hectares): **3.01**
Crop: **GRASS HAY**
Application Rate (tons/acre): **3.89** Application Rate (MT/HA): **8.71**

	English Units		Metric Units	
Recommended N for Crop	**100.00**	lbs/Acre	**112.00**	Kg/Ha
Nitrogen Available:	**22.50**	lbs/Dry Ton	**54.68**	Kg/Metic Ton
Total Biosolids Applied:	**28.91**	Dry Tons	**26.23**	Dry Metric Tons
Actual Application Rate:	**3.89**	Dry Tons/Acre	**8.71**	Dry Metric Tons / Ha
Total Recommended N:	**743.00**	lbs/field	**337.02**	Kg/Field
Total Applied N:	**650.47**	lbs/field	**295.05**	Kg/Field
% of Recommended N:	**87.55**	%	**87.55**	%
Total P Applied:	**826.82**	lbs/field	**375.04**	Kg/Field
Total K Applied:	**156.69**	lbs/field	**71.07**	Kg/Field

LAND APPLICATION SITE INFORMATION FOR CORVALLIS WWRP OR002636-1

Individual Load Tracking

Haul # / Desc	Date Hauled	Date Applied	Gallons	% Total Solids	Wet Tons	Dry Tons	lbs N / load	lbs N / DT	lbs P / load	lbs P / DT	lbs K / load	lbs K / DT
6565/1	7/5/02	7/5/02	11,200.00	4.50	46.70	2.10	55.09	26.21	75.66	36.00	7.15	3.40
6575/1	7/5/02	7/5/02	11,200.00	4.50	46.70	2.10	55.09	26.21	75.66	36.00	7.15	3.40
6565/2	7/8/02	7/8/02	28,000.00	2.20	116.76	2.57	67.33	26.21	92.47	36.00	8.73	3.40
6575/2	7/8/02	7/8/02	33,600.00	2.20	140.11	3.08	80.79	26.21	110.97	36.00	10.48	3.40
6565/3	7/8/02	7/8/02	16,800.00	2.90	70.06	2.03	53.25	26.21	73.14	36.00	6.91	3.40
6575/3	7/8/02	7/8/02	16,800.00	2.90	70.06	2.03	53.25	26.21	73.14	36.00	6.91	3.40
6565/4	7/9/02	7/9/02	5,600.00	3.50	23.35	0.82	21.42	26.21	29.42	36.00	2.78	3.40
6575/4	7/9/02	7/9/02	44,800.00	3.50	186.82	6.54	171.38	26.21	235.39	36.00	22.23	3.40
6565/1	7/17/03	7/17/03	33,600.00	5.50	140.11	7.71	211.30	27.42	339.07	44.00	32.37	4.20
6565/2	7/18/03	7/18/03	33,600.00	5.30	140.11	7.43	203.62	27.42	326.74	44.00	31.19	4.20
6575/1	7/17/03	7/17/03	28,000.00	5.50	116.76	6.42	176.09	27.42	282.56	44.00	26.97	4.20
6575/2	7/18/03	7/18/03	28,000.00	5.30	116.76	6.19	169.68	27.42	272.28	44.00	25.99	4.20
6575-1	7/29/04	7/29/04	28,000.00	5.10	116.76	5.95	116.95	19.64	8.69	1.46	29.65	4.98
6575-2	7/30/04	7/30/04	22,400.00	5.00	93.41	4.67	91.73	19.64	6.82	1.46	23.26	4.98
6565-1	8/30/04	8/30/04	5,600.00	2.60	23.35	0.61	11.92	19.64	0.89	1.46	3.02	4.98
6565-2	8/31/04	8/31/04	33,600.00	2.60	140.11	3.64	71.55	19.64	5.32	1.46	18.14	4.98
6575/3	8/31/04	8/31/04	28,000.00	2.60	116.76	3.04	59.62	19.64	4.43	1.46	15.12	4.98
6565-3	9/1/04	9/1/04	44,800.00	4.50	186.82	8.41	165.11	19.64	12.27	1.46	41.87	4.98
6575-4	9/1/04	9/1/04	28,000.00	4.50	116.76	5.25	103.19	19.64	7.67	1.46	26.17	4.98
6565-4	9/2/04	9/2/04	11,200.00	5.20	46.70	2.43	47.70	19.64	3.55	1.46	12.09	4.98
6565/1	7/12/06	7/12/06	11,200.00	3.40	46.70	1.59	35.73	22.50	45.41	28.60	8.61	5.42
6565/2	7/13/06	7/13/06	44,800.00	4.80	186.82	8.97	201.76	22.50	256.46	28.60	48.60	5.42
6565/3	7/14/06	7/14/06	22,400.00	4.60	93.41	4.30	96.68	22.50	122.89	28.60	23.29	5.42
6575/1	7/12/06	7/12/06	5,600.00	3.40	23.35	0.79	17.86	22.50	22.71	28.60	4.30	5.42
6575/2	7/13/06	7/13/06	44,800.00	4.80	186.82	8.97	201.76	22.50	256.46	28.60	48.60	5.42
6575/3	7/14/06	7/14/06	22,400.00	4.60	93.41	4.30	96.68	22.50	122.89	28.60	23.29	5.42

Avg % Total Solids:	**4.06**
Total Gallons:	**644,000.00**
Total Dry Tons:	**111.93**
Gallons / Acre:	**86,675.64**

Total Cumulative Metal Loading (Kg/Hectare)

As Average:	**0.48**
As Maximum:	**0.53**
Cd Average:	**0.26**
Cd Maximum	**0.28**
Cu Average:	**46.43**
Cu Maximum	**49.56**
Pb Average:	**4.16**
Pb Maximum	**4.75**
Hg Average:	**0.13**
Hg Maximum	**0.21**
Mo Average:	**0.70**
Mo Maximum	**0.84**
Ni Average:	**2.35**
Ni Maximum:	**2.67**
Se Average:	**0.77**
Se Maximum	**0.94**
Zn Average:	**88.08**
Zn Maximum:	**89.12**

LAND APPLICATION SITE INFORMATION FOR CORVALLIS WWRP OR002636-1

Monitoring Year: **2006**
Site Name: **ANDERSON** Site No: **13**
Hydrologic Unit:
Owner: **LLOYD ANDERSON**
Operator: **LLOYD ANDERSON**
Applier: **CITY OF CORVALLIS**
Latitude: Longitude:
Location: **4210 NW CRESCENT VALLEY DRIVE**
Section: **14**
Township: **T11S** Range: **R5W**
Size (acres): **2.05** Size (hectares): **0.83**
Crop: **GRASS HAY**
Application Rate (tons/acre): **4.33** Application Rate (MT/HA): **9.70**

	English Units		Metric Units	
Recommended N for Crop	**100.00**	lbs/Acre	**112.00**	Kg/Ha
Nitrogen Available:	**22.50**	lbs/Dry Ton	**54.68**	Kg/Metic Ton
Total Biosolids Applied:	**8.87**	Dry Tons	**8.05**	Dry Metric Tons
Actual Application Rate:	**4.33**	Dry Tons/Acre	**9.70**	Dry Metric Tons / Ha
Total Recommended N:	**205.00**	lbs/field	**92.99**	Kg/Field
Total Applied N:	**199.66**	lbs/field	**90.56**	Kg/Field
% of Recommended N:	**97.40**	%	**97.39**	%
Total P Applied:	**253.79**	lbs/field	**115.12**	Kg/Field
Total K Applied:	**48.10**	lbs/field	**21.82**	Kg/Field

Individual Load Tracking

Haul # / Desc	Date Hauled	Date Applied	Gallons	% Total Solids	Wet Tons	Dry Tons	lbs N / load	lbs N / DT	lbs P / load	lbs P / DT	lbs K / load	lbs K / DT
6565/1	7/3/02	7/3/02	16,800.00	4.50	70.06	3.15	82.63	26.21	113.49	36.00	10.72	3.40
6565/2	7/5/02	7/5/02	22,400.00	4.50	93.41	4.20	110.17	26.21	151.32	36.00	14.29	3.40
6575/1	7/5/02	7/5/02	11,200.00	4.50	46.70	2.10	55.09	26.21	75.66	36.00	7.15	3.40
6565/1	7/18/03	7/18/03	5,600.00	5.30	23.35	1.24	33.94	27.42	54.46	44.00	5.20	4.20
6565/2	7/21/03	7/21/03	11,200.00	5.10	46.70	2.38	65.31	27.42	104.80	44.00	10.00	4.20
6575/1	7/18/03	7/18/03	11,200.00	5.30	46.70	2.48	67.87	27.42	108.91	44.00	10.40	4.20
6575/2	7/21/03	7/21/03	11,200.00	5.10	46.70	2.38	65.31	27.42	104.80	44.00	10.00	4.20
6575-1	9/1/04	9/1/04	5,600.00	4.50	23.35	1.05	20.64	19.64	1.53	1.46	5.23	4.98
6565-1	9/2/04	9/2/04	39,200.00	5.20	163.46	8.50	166.94	19.64	12.41	1.46	42.33	4.98
6565-2	9/3/04	9/3/04	11,200.00	5.50	46.70	2.57	50.45	19.64	3.75	1.46	12.79	4.98
6565/1	7/20/06	7/20/06	28,000.00	4.90	116.76	5.72	128.73	22.50	163.63	28.60	31.01	5.42
6575/1	7/17/06	7/17/06	16,800.00	4.50	70.06	3.15	70.93	22.50	90.16	28.60	17.09	5.42

Avg % Total Solids: **4.91**
Total Gallons: **190,400.00**
Total Dry Tons: **38.93**
Gallons / Acre: **92,878.05**

LAND APPLICATION SITE INFORMATION FOR
CORVALLIS WWRP
OR002636-1

Total Cumulative Metal Loading (Kg/Hectare)

As Average:	**0.48**
As Maximum:	**0.53**
Cd Average:	**0.26**
Cd Maximum	**0.28**
Cu Average:	**46.43**
Cu Maximum	**49.56**
Pb Average:	**4.16**
Pb Maximum	**4.75**
Hg Average:	**0.13**
Hg Maximum	**0.21**
Mo Average:	**0.70**
Mo Maximum	**0.84**
Ni Average:	**2.35**
Ni Maximum:	**2.67**
Se Average:	**0.77**
Se Maximum	**0.94**
Zn Average:	**88.08**
Zn Maximum:	**89.12**

Comments:

LAND APPLICATION SITE INFORMATION FOR CORVALLIS WWRP OR002636-1

Monitoring Year: **2006**
Site Name: **ANDERSON** Site No: **14**
Hydrologic Unit:
Owner: **LLOYD ANDERSON**
Operator: **LLOYD ANDERSON**
Applier: **CITY OF CORVALLIS**
Latitude: Longitude:
Location: **4210 NW CRESCENT VALLEY DRIVE**
Section: **14**
Township: **T11S** Range: **R5W**
Size (acres): **2.42** Size (hectares): **0.98**
Crop: **GRASS HAY**
Application Rate (tons/acre): **4.15** Application Rate (MT/HA): **9.30**

	English Units		Metric Units	
Recommended N for Crop	**100.00**	lbs/Acre	**112.00**	Kg/Ha
Nitrogen Available:	**22.50**	lbs/Dry Ton	**54.68**	Kg/Metic Ton
Total Biosolids Applied:	**10.04**	Dry Tons	**9.11**	Dry Metric Tons
Actual Application Rate:	**4.15**	Dry Tons/Acre	**9.30**	Dry Metric Tons / Ha
Total Recommended N:	**242.00**	lbs/field	**109.77**	Kg/Field
Total Applied N:	**225.93**	lbs/field	**102.48**	Kg/Field
% of Recommended N:	**93.36**	%	**93.36**	%
Total P Applied:	**287.18**	lbs/field	**130.26**	Kg/Field
Total K Applied:	**54.42**	lbs/field	**24.68**	Kg/Field

Individual Load Tracking

Haul # / Desc	Date Hauled	Date Applied	Gallons	% Total Solids	Wet Tons	Dry Tons	lbs N / load	lbs N / DT	lbs P / load	lbs P / DT	lbs K / load	lbs K / DT
6565/1	7/3/02	7/3/02	11,200.00	4.50	46.70	2.10	55.09	26.21	75.66	36.00	7.15	3.40
6565/2	7/5/02	7/5/02	28,000.00	4.50	116.76	5.25	137.71	26.21	189.15	36.00	17.86	3.40
6575/1	6/28/02	6/28/02	22,400.00	3.40	93.41	3.18	83.24	26.21	114.33	36.00	10.80	3.40
6565/1	7/16/03	7/16/03	28,000.00	5.70	116.76	6.66	182.49	27.42	292.83	44.00	27.95	4.20
6575/1	7/16/03	7/16/03	33,600.00	5.70	140.11	7.99	218.99	27.42	351.40	44.00	33.54	4.20
6575/2	7/17/03	7/17/03	5,600.00	5.50	23.35	1.28	35.22	27.42	56.51	44.00	5.39	4.20
6575-1	7/27/04	7/27/04	16,800.00	5.00	70.06	3.50	68.79	19.64	5.11	1.46	17.44	4.98
6575-2	7/28/04	7/28/04	44,800.00	5.10	186.82	9.53	187.12	19.64	13.91	1.46	47.45	4.98
6575-3	7/29/04	7/29/04	16,800.00	5.10	70.06	3.57	70.17	19.64	5.22	1.46	17.79	4.98
6565/1	7/12/06	7/12/06	28,000.00	3.40	116.76	3.97	89.32	22.50	113.54	28.60	21.52	5.42
6575/1	7/12/06	7/12/06	28,000.00	3.40	116.76	3.97	89.32	22.50	113.54	28.60	21.52	5.42
6575/2	7/17/06	7/17/06	11,200.00	4.50	46.70	2.10	47.29	22.50	60.11	28.60	11.39	5.42

Avg % Total Solids: **4.65**
Total Gallons: **274,400.00**
Total Dry Tons: **53.10**
Gallons / Acre: **113,388.43**

LAND APPLICATION SITE INFORMATION FOR
CORVALLIS WWRP
OR002636-1

Total Cumulative Metal Loading (Kg/Hectare)

As Average:	**0.48**
As Maximum:	**0.53**
Cd Average:	**0.26**
Cd Maximum	**0.28**
Cu Average:	**46.43**
Cu Maximum	**49.56**
Pb Average:	**4.16**
Pb Maximum	**4.75**
Hg Average:	**0.13**
Hg Maximum	**0.21**
Mo Average:	**0.70**
Mo Maximum	**0.84**
Ni Average:	**2.35**
Ni Maximum:	**2.67**
Se Average:	**0.77**
Se Maximum	**0.94**
Zn Average:	**88.08**
Zn Maximum:	**89.12**

Comments:

LAND APPLICATION SITE INFORMATION FOR CORVALLIS WWRP OR002636-1

Monitoring Year: **2006**
Site Name: **ANDERSON** Site No: **15**
Hydrologic Unit:
Owner: **LLOYD ANDERSON**
Operator: **LLOYD ANDERSON**
Applier: **CITY OF CORVALLIS**
Latitude: Longitude:
Location: **4210 NW CRESCENT VALLEY DRIVE**
Section: **14**
Township: **T11S** Range: **R5W**
Size (acres): **3.70** Size (hectares): **1.50**
Crop: **GRASS HAY**
Application Rate (tons/acre): **4.59** Application Rate (MT/HA): **10.27**

	English Units		Metric Units	
Recommended N for Crop	**100.00**	lbs/Acre	**112.00**	Kg/Ha
Nitrogen Available:	**22.50**	lbs/Dry Ton	**54.68**	Kg/Metic Ton
Total Biosolids Applied:	**16.98**	Dry Tons	**15.40**	Dry Metric Tons
Actual Application Rate:	**4.59**	Dry Tons/Acre	**10.27**	Dry Metric Tons / Ha
Total Recommended N:	**370.00**	lbs/field	**167.83**	Kg/Field
Total Applied N:	**381.98**	lbs/field	**173.26**	Kg/Field
% of Recommended N:	**103.24**	%	**103.24**	%
Total P Applied:	**485.54**	lbs/field	**220.24**	Kg/Field
Total K Applied:	**92.01**	lbs/field	**41.74**	Kg/Field

Individual Load Tracking

Haul # / Desc	Date Hauled	Date Applied	Gallons	% Total Solids	Wet Tons	Dry Tons	lbs N / load	lbs N / DT	lbs P / load	lbs P / DT	lbs K / load	lbs K / DT
6565/1	7/10/02	7/10/02	39,200.00	2.70	163.46	4.41	115.68	26.21	158.89	36.00	15.01	3.40
6575/1	7/10/02	7/10/02	39,200.00	2.70	163.46	4.41	115.68	26.21	158.89	36.00	15.01	3.40
6565/1	7/21/03	7/21/03	28,000.00	5.10	116.76	5.95	163.28	27.42	262.01	44.00	25.01	4.20
6575/1	7/21/03	7/21/03	33,600.00	5.10	140.11	7.15	195.94	27.42	314.41	44.00	30.01	4.20
6565-1	9/3/04	9/3/04	39,200.00	5.50	163.46	8.99	176.57	19.64	13.13	1.46	44.77	4.98
6565-2	9/7/04	9/7/04	11,200.00	4.10	46.70	1.91	37.61	19.64	2.80	1.46	9.54	4.98
6575-1	9/7/04	9/7/04	11,200.00	4.10	46.70	1.91	37.61	19.64	2.80	1.46	9.54	4.98
6565/1	7/14/06	7/14/06	22,400.00	4.60	93.41	4.30	96.68	22.50	122.89	28.60	23.29	5.42
6565/2	7/17/06	7/17/06	39,200.00	4.50	163.46	7.36	165.51	22.50	210.38	28.60	39.87	5.42
6575/1	7/14/06	7/14/06	16,800.00	4.60	70.06	3.22	72.51	22.50	92.17	28.60	17.47	5.42
6575/2	7/17/06	7/17/06	11,200.00	4.50	46.70	2.10	47.29	22.50	60.11	28.60	11.39	5.42

Avg % Total Solids: **4.32**
Total Gallons: **291,200.00**
Total Dry Tons: **51.72**
Gallons / Acre: **78,702.70**

LAND APPLICATION SITE INFORMATION FOR
CORVALLIS WWRP
OR002636-1

Total Cumulative Metal Loading (Kg/Hectare)

As Average:	**0.48**
As Maximum:	**0.53**
Cd Average:	**0.26**
Cd Maximum	**0.28**
Cu Average:	**46.43**
Cu Maximum	**49.56**
Pb Average:	**4.16**
Pb Maximum	**4.75**
Hg Average:	**0.13**
Hg Maximum	**0.21**
Mo Average:	**0.70**
Mo Maximum	**0.84**
Ni Average:	**2.35**
Ni Maximum:	**2.67**
Se Average:	**0.77**
Se Maximum	**0.94**
Zn Average:	**88.08**
Zn Maximum:	**89.12**

Comments:

LAND APPLICATION SITE INFORMATION FOR CORVALLIS WWRP OR002636-1

Monitoring Year: **2006**
Site Name: **DLD** Site No: **21**
Hydrologic Unit:
Owner: **CITY OF CORVALLIS**
Operator: **CITY OF CORVALLIS**
Applier: **CITY OF CORVALLIS**
Latitude: Longitude:
Location: **1304 NE 2ND**
Section: **10**
Township: **713S** Range: **R6W**
Size (acres): **17.14** Size (hectares): **6.94**
Crop: **GRASS HAY**
Application Rate (tons/acre): **3.96** Application Rate (MT/HA): **8.88**

	English Units		Metric Units	
Recommended N for Crop	**100.00**	lbs/Acre	**112.00**	Kg/Ha
Nitrogen Available:	**22.50**	lbs/Dry Ton	**54.68**	Kg/Metic Ton
Total Biosolids Applied:	**67.91**	Dry Tons	**61.61**	Dry Metric Tons
Actual Application Rate:	**3.96**	Dry Tons/Acre	**8.88**	Dry Metric Tons / Ha
Total Recommended N:	**1,714.00**	lbs/field	**777.46**	Kg/Field
Total Applied N:	**1,527.92**	lbs/field	**693.05**	Kg/Field
% of Recommended N:	**89.14**	%	**89.14**	%
Total P Applied:	**1,942.16**	lbs/field	**880.95**	Kg/Field
Total K Applied:	**368.06**	lbs/field	**166.95**	Kg/Field

LAND APPLICATION SITE INFORMATION FOR CORVALLIS WWRP OR002636-1

Individual Load Tracking

Haul # / Desc	Date Hauled	Date Applied	Gallons	% Total Solids	Wet Tons	Dry Tons	lbs N / load	lbs N / DT	lbs P / load	lbs P / DT	lbs K / load	lbs K / DT
6565/1	7/11/02	7/11/02	44,800.00	4.40	186.82	8.22	215.44	26.21	295.92	36.00	27.95	3.40
6575/1	7/11/02	7/11/02	67,200.00	4.40	280.22	12.33	323.17	26.21	443.87	36.00	41.92	3.40
6565/2	7/15/02	7/15/02	128,800.00	4.40	537.10	23.63	619.40	26.21	850.76	36.00	80.35	3.40
6575/2	7/15/02	7/15/02	128,800.00	4.40	537.10	23.63	619.40	26.21	850.76	36.00	80.35	3.40
6565/3	7/16/02	7/16/02	33,600.00	4.40	140.11	6.16	161.58	26.21	221.94	36.00	20.96	3.40
6575/3	7/16/02	7/16/02	16,800.00	4.40	70.06	3.08	80.79	26.21	110.97	36.00	10.48	3.40
6565/1	7/15/03	7/15/03	11,200.00	5.60	46.70	2.62	71.71	27.42	115.08	44.00	10.98	4.20
6565/2	9/4/03	9/4/03	11,200.00	5.50	46.70	2.57	20.70	8.06	14.28	5.56	15.31	5.96
6565/3	9/5/03	9/5/03	11,200.00	5.30	46.70	2.48	19.95	8.06	13.76	5.56	14.75	5.96
6565/4	9/8/03	9/8/03	39,200.00	5.00	163.46	8.17	65.88	8.06	45.44	5.56	48.71	5.96
6575/1	7/15/03	7/15/03	11,200.00	5.60	46.70	2.62	71.71	27.42	115.08	44.00	10.98	4.20
6575/2	9/5/03	9/5/03	33,600.00	5.30	140.11	7.43	59.85	8.06	41.29	5.56	44.26	5.96
6575/3	9/8/03	9/8/03	44,800.00	5.00	186.82	9.34	75.29	8.06	51.93	5.56	55.67	5.96
6575/4	9/17/03	9/17/03	33,600.00	5.00	140.11	7.01	56.47	8.06	38.95	5.56	41.75	5.96
6575/5	9/18/03	9/18/03	22,400.00	5.00	93.41	4.67	37.64	8.06	25.97	5.56	27.84	5.96
6575-1	7/15/04	7/15/04	5,600.00	5.90	23.35	1.38	27.06	19.64	2.01	1.46	6.86	4.98
6565-1	8/3/04	8/3/04	16,800.00	5.00	70.06	3.50	68.79	19.64	5.11	1.46	17.44	4.98
6575-2	8/4/04	8/4/04	67,200.00	4.70	280.22	13.17	258.67	19.64	19.23	1.46	65.59	4.98
6575-3	8/5/04	8/5/04	67,200.00	4.70	280.22	13.17	258.67	19.64	19.23	1.46	65.59	4.98
6565-2	8/9/04	8/9/04	67,200.00	5.20	280.22	14.57	286.19	19.64	21.27	1.46	72.57	4.98
6565-3	8/10/04	8/10/04	67,200.00	4.40	280.22	12.33	242.16	19.64	18.00	1.46	61.40	4.98
6565-4	8/11/04	8/11/04	61,600.00	4.80	256.87	12.33	242.16	19.64	18.00	1.46	61.40	4.98
6565-5	8/12/04	8/12/04	28,000.00	5.10	116.76	5.95	116.95	19.64	8.69	1.46	29.65	4.98
6565/1	7/18/06	7/18/06	28,000.00	5.00	116.76	5.84	131.35	22.50	166.97	28.60	31.64	5.42
6565/2	7/19/06	7/19/06	50,400.00	4.60	210.17	9.67	217.52	22.50	276.50	28.60	52.40	5.42
6565/3	7/20/06	7/20/06	22,400.00	4.90	93.41	4.58	102.98	22.50	130.90	28.60	24.81	5.42
6565/4	7/21/06	7/21/06	39,200.00	4.30	163.46	7.03	158.15	22.50	201.03	28.60	38.10	5.42
6565/5	7/24/06	7/24/06	39,200.00	2.70	163.46	4.41	99.30	22.50	126.23	28.60	23.92	5.42
6575/1	7/18/06	7/18/06	22,400.00	5.00	93.41	4.67	105.08	22.50	133.57	28.60	25.31	5.42
6575/2	7/19/06	7/19/06	78,400.00	4.60	326.93	15.04	338.37	22.50	430.11	28.60	81.51	5.42
6575/3	7/20/06	7/20/06	33,600.00	4.90	140.11	6.87	154.47	22.50	196.35	28.60	37.21	5.42
6575/4	7/21/06	7/21/06	33,600.00	4.30	140.11	6.02	135.56	22.50	172.31	28.60	32.65	5.42
6575/5	7/24/06	7/24/06	33,600.00	2.70	140.11	3.78	85.12	22.50	108.19	28.60	20.50	5.42

Avg % Total Solids:	**4.74**
Total Gallons:	**1,400,000.00**
Total Dry Tons:	**268.27**
Gallons / Acre:	**81,680.28**

LAND APPLICATION SITE INFORMATION FOR
CORVALLIS WWRP
OR002636-1

Total Cumulative Metal Loading (Kg/Hectare)

As Average:	**0.48**
As Maximum:	**0.53**
Cd Average:	**0.26**
Cd Maximum	**0.28**
Cu Average:	**46.43**
Cu Maximum	**49.56**
Pb Average:	**4.16**
Pb Maximum	**4.75**
Hg Average:	**0.13**
Hg Maximum	**0.21**
Mo Average:	**0.70**
Mo Maximum	**0.84**
Ni Average:	**2.35**
Ni Maximum:	**2.67**
Se Average:	**0.77**
Se Maximum	**0.94**
Zn Average:	**88.08**
Zn Maximum:	**89.12**

Comments:

LAND APPLICATION SITE INFORMATION FOR CORVALLIS WWRP OR002636-1

Monitoring Year: **2006**
Site Name: **DLD** Site No: **23**
Hydrologic Unit:
Owner: **CITY OF CORVALLIS**
Operator: **CITY OF CORVALLIS**
Applier: **CITY OF CORVALLIS**
Latitude: Longitude:
Location: **1304 NE 2ND**
Section: **10**
Township: **T13S** Range: **R6W**
Size (acres): **3.00** Size (hectares): **1.21**
Crop: **GRASS HAY**
Application Rate (tons/acre): **3.89** Application Rate (MT/HA): **8.76**

	English Units		Metric Units	
Recommended N for Crop	**100.00**	lbs/Acre	**112.00**	Kg/Ha
Nitrogen Available:	**22.50**	lbs/Dry Ton	**54.68**	Kg/Metic Ton
Total Biosolids Applied:	**11.68**	Dry Tons	**10.60**	Dry Metric Tons
Actual Application Rate:	**3.89**	Dry Tons/Acre	**8.76**	Dry Metric Tons / Ha
Total Recommended N:	**300.00**	lbs/field	**136.08**	Kg/Field
Total Applied N:	**262.71**	lbs/field	**119.16**	Kg/Field
% of Recommended N:	**87.57**	%	**87.57**	%
Total P Applied:	**333.93**	lbs/field	**151.47**	Kg/Field
Total K Applied:	**63.28**	lbs/field	**28.70**	Kg/Field

Individual Load Tracking

Haul # / Desc	Date Hauled	Date Applied	Gallons	% Total Solids	Wet Tons	Dry Tons	lbs N / load	lbs N / DT	lbs P / load	lbs P / DT	lbs K / load	lbs K / DT
6565/1	7/10/02	7/10/02	16,800.00	2.70	70.06	1.89	49.58	26.21	68.09	36.00	6.43	3.40
6575/1	7/10/02	7/10/02	16,800.00	2.70	70.06	1.89	49.58	26.21	68.09	36.00	6.43	3.40
6565/2	7/11/02	7/11/02	5,600.00	4.40	23.35	1.03	26.93	26.21	36.99	36.00	3.49	3.40
6575/2	7/11/02	7/11/02	5,600.00	4.40	23.35	1.03	26.93	26.21	36.99	36.00	3.49	3.40
6565/1	7/16/03	7/16/03	16,800.00	5.70	70.06	3.99	109.49	27.42	175.70	44.00	16.77	4.20
6575/1	7/16/03	7/16/03	16,800.00	5.70	70.06	3.99	109.49	27.42	175.70	44.00	16.77	4.20
6565-1	8/2/04	8/2/04	39,200.00	4.30	163.46	7.03	138.05	19.64	10.26	1.46	35.00	4.98
6565-2	8/3/04	8/3/04	16,800.00	5.00	70.06	3.50	68.79	19.64	5.11	1.46	17.44	4.98
6565/1	7/18/06	7/18/06	28,000.00	5.00	116.76	5.84	131.35	22.50	166.97	28.60	31.64	5.42
6575/1	7/18/06	7/17/06	28,000.00	5.00	116.76	5.84	131.35	22.50	166.97	28.60	31.64	5.42

Avg % Total Solids: **4.49**
Total Gallons: **190,400.00**
Total Dry Tons: **36.03**
Gallons / Acre: **63,466.67**

LAND APPLICATION SITE INFORMATION FOR
CORVALLIS WWRP
OR002636-1

Total Cumulative Metal Loading (Kg/Hectare)

As Average:	**0.48**
As Maximum:	**0.53**
Cd Average:	**0.26**
Cd Maximum	**0.28**
Cu Average:	**46.43**
Cu Maximum	**49.56**
Pb Average:	**4.16**
Pb Maximum	**4.75**
Hg Average:	**0.13**
Hg Maximum	**0.21**
Mo Average:	**0.70**
Mo Maximum	**0.84**
Ni Average:	**2.35**
Ni Maximum:	**2.67**
Se Average:	**0.77**
Se Maximum	**0.94**
Zn Average:	**88.08**
Zn Maximum:	**89.12**

Comments:

LAND APPLICATION SITE INFORMATION FOR CORVALLIS WWRP OR002636-1

Monitoring Year: **2006**
Site Name: **OSU, (SOAP CREEK RANCH)** Site No: **75**
Hydrologic Unit:
Owner: **OREGON STATE UNIVERSITY**
Operator: **OSU DEPT. OF ANIMAL SCIEN(**
Applier: **CITY OF CORVALLIS**
Latitude: Longitude:
Location:
Section: **24**
Township: **T10S** Range: **R5W**
Size (acres): **34.07** Size (hectares): **13.79**
Crop: **GRASS HAY**
Application Rate (tons/acre): **4.77** Application Rate (MT/HA): **10.70**

	English Units		Metric Units	
Recommended N for Crop	**100.00**	lbs/Acre	**112.00**	Kg/Ha
Nitrogen Available:	**26.27**	lbs/Dry Ton	**63.84**	Kg/Metic Ton
Total Biosolids Applied:	**162.62**	Dry Tons	**147.53**	Dry Metric Tons
Actual Application Rate:	**4.77**	Dry Tons/Acre	**10.70**	Dry Metric Tons / Ha
Total Recommended N:	**3,407.00**	lbs/field	**1,545.39**	Kg/Field
Total Applied N:	**4,281.96**	lbs/field	**1,942.26**	Kg/Field
% of Recommended N:	**125.68**	%	**125.68**	%
Total P Applied:	**487.64**	lbs/field	**221.19**	Kg/Field
Total K Applied:	**840.67**	lbs/field	**381.32**	Kg/Field

LAND APPLICATION SITE INFORMATION FOR CORVALLIS WWRP OR002636-1

Individual Load Tracking

Haul # / Desc	Date Hauled	Date Applied	Gallons	% Total Solids	Wet Tons	Dry Tons	lbs N / load	lbs N / DT	lbs P / load	lbs P / DT	lbs K / load	lbs K / DT
6565/1	8/2/02	8/2/02	39,200.00	4.70	163.46	7.68	189.38	24.65	322.68	42.00	29.19	3.80
6565/2	8/7/02	8/7/02	44,800.00	5.30	186.82	9.90	244.07	24.65	415.85	42.00	37.62	3.80
6565/3	8/8/02	8/8/02	44,800.00	5.10	186.82	9.53	234.86	24.65	400.16	42.00	36.20	3.80
6565/4	8/9/02	8/9/02	16,800.00	6.10	70.06	4.27	105.34	24.65	179.48	42.00	16.24	3.80
6565/5	8/12/02	8/12/02	44,800.00	5.30	186.82	9.90	244.07	24.65	415.85	42.00	37.62	3.80
6565/6	8/13/02	8/13/02	44,800.00	5.30	186.82	9.90	244.07	24.65	415.85	42.00	37.62	3.80
6565/7	8/14/02	8/14/02	44,800.00	5.20	186.82	9.71	254.81	26.23	272.00	28.00	27.20	2.80
6565/8	8/15/02	8/15/02	44,800.00	5.10	186.82	9.53	249.91	26.23	266.77	28.00	26.68	2.80
6565/9	8/16/02	8/16/02	33,600.00	5.70	140.11	7.99	209.48	26.23	223.62	28.00	22.36	2.80
6565/10	8/19/02	8/19/02	44,800.00	5.90	186.82	11.02	289.11	26.23	308.62	28.00	30.86	2.80
6565/11	8/20/02	8/20/02	5,600.00	5.80	23.35	1.35	35.53	26.23	37.92	28.00	3.79	2.80
6575/2	8/7/02	8/7/02	44,800.00	5.30	186.82	9.90	244.07	24.65	415.85	42.00	37.62	3.80
6575/3	8/8/02	8/8/02	39,200.00	5.10	163.46	8.34	205.50	24.65	350.14	42.00	31.68	3.80
6575/4	8/9/02	8/9/02	22,400.00	6.10	93.41	5.70	140.45	24.65	239.31	42.00	21.65	3.80
6575/5	8/12/02	8/12/02	44,800.00	5.30	186.82	9.90	244.07	24.65	415.85	42.00	37.62	3.80
6575/6	8/13/02	8/13/02	39,200.00	5.30	163.46	8.66	213.56	24.65	363.87	42.00	32.92	3.80
6575/7	8/14/02	8/14/02	44,800.00	5.20	186.82	9.71	254.81	26.23	272.00	28.00	27.20	2.80
6575/8	8/15/02	8/15/02	39,200.00	5.10	163.46	8.34	218.67	26.23	233.43	28.00	23.34	2.80
6575/9	8/20/02	8/20/02	5,600.00	5.80	23.35	1.35	35.53	26.23	37.92	28.00	3.79	2.80
6575/1	8/2/02	8/2/02	33,600.00	4.70	140.11	6.59	162.33	24.65	276.58	42.00	25.02	3.80
6565/4	8/29/05	8/29/05	33,600.00	4.50	140.11	6.31	153.97	24.42	20.18	3.20	35.31	5.60
6565/5	8/30/05	8/30/05	39,200.00	3.40	163.46	5.56	135.72	24.42	17.78	3.20	31.12	5.60
6565/6	8/31/05	8/31/05	16,800.00	4.10	70.06	2.87	70.14	24.42	9.19	3.20	16.08	5.60
6575/1	9/2/05	9/2/05	39,200.00	4.00	163.46	6.54	159.67	24.42	20.92	3.20	36.62	5.60
6575/2	9/6/05	9/6/05	39,200.00	4.50	163.46	7.36	179.63	24.42	23.54	3.20	41.19	5.60
6575/3	9/7/05	9/7/05	39,200.00	5.40	163.46	8.83	215.56	24.42	28.25	3.20	49.43	5.60
6575/4	9/8/05	9/8/05	39,200.00	5.60	163.46	9.15	223.54	24.42	29.29	3.20	51.26	5.60
6575/5	9/12/05	9/12/05	39,200.00	4.60	163.46	7.52	183.62	24.42	24.06	3.20	42.11	5.60
6575/6	9/13/05	9/13/05	39,200.00	4.10	163.46	6.70	163.66	24.42	21.45	3.20	37.53	5.60
6575/7	9/14/05	9/14/05	33,600.00	5.50	140.11	7.71	188.18	24.42	24.66	3.20	43.15	5.60
6565/1	8/24/05	8/24/05	22,400.00	6.20	93.41	5.79	141.42	24.42	18.53	3.20	31.85	5.50
6565/2	8/25/05	8/25/05	39,200.00	5.90	163.46	9.64	235.52	24.42	30.86	3.20	54.01	5.60
6565/3	8/26/05	8/26/05	39,200.00	5.30	163.46	8.66	211.56	24.42	27.72	3.20	48.52	5.60
6565/1	8/15/06	8/15/06	11,200.00	4.40	46.70	2.05	52.22	25.41	4.81	2.34	10.07	4.90
6565/2	8/18/06	8/18/06	33,600.00	6.40	140.11	8.97	227.86	25.41	20.98	2.34	43.94	4.90
6565/3	8/21/06	8/21/06	44,800.00	5.00	186.82	9.34	237.35	25.41	21.86	2.34	45.77	4.90
6565/4	8/22/06	8/22/06	44,800.00	4.40	186.82	8.22	218.98	26.64	26.47	3.22	43.24	5.26
6565/5	8/23/06	8/23/06	44,800.00	4.90	186.82	9.15	243.86	26.64	29.48	3.22	48.15	5.26
6565/6	8/24/06	8/24/06	39,200.00	4.40	163.46	7.19	191.61	26.64	23.16	3.22	37.83	5.26
6565/7	8/25/06	8/25/06	39,200.00	4.50	163.46	7.36	195.96	26.64	23.69	3.22	38.69	5.26
6565/8	8/28/06	8/28/06	39,200.00	4.30	163.46	7.03	187.25	26.64	22.63	3.22	36.97	5.26
6565/9	8/29/06	8/29/06	39,200.00	4.80	163.46	7.85	209.02	26.64	25.26	3.22	41.27	5.26
6565/10	8/30/06	8/30/06	44,800.00	4.80	186.82	8.97	238.89	26.64	28.87	3.22	47.17	5.26
6565/11	8/31/06	8/31/06	39,200.00	4.10	163.46	6.70	178.54	26.64	21.58	3.22	35.25	5.26
6575/1	8/14/06	8/14/06	5,600.00	4.40	23.35	1.03	26.11	25.41	2.40	2.34	5.03	4.90
6575/2	8/15/06	8/15/06	11,200.00	4.40	46.70	2.05	52.22	25.41	4.81	2.34	10.07	4.90
6575/3	8/18/06	8/18/06	39,200.00	6.40	163.46	10.46	265.83	25.41	24.48	2.34	51.26	4.90
6575/4	8/21/06	8/21/06	33,600.00	5.00	140.11	7.01	178.01	25.41	16.39	2.34	34.33	4.90
6575/5	8/22/06	8/22/06	39,200.00	4.40	163.46	7.19	191.61	26.64	23.16	3.22	37.83	5.26
6575/6	8/23/06	8/23/06	39,200.00	4.90	163.46	8.01	213.38	26.64	25.79	3.22	42.13	5.26
6575/7	8/24/06	8/24/06	44,800.00	4.40	186.82	8.22	218.98	26.64	26.47	3.22	43.24	5.26
6575/8	8/25/06	8/25/06	39,200.00	4.50	163.46	7.36	195.96	26.64	23.69	3.22	38.69	5.26
6575/9	8/28/06	8/28/06	39,200.00	4.30	163.46	7.03	187.25	26.64	22.63	3.22	36.97	5.26
6575/10	8/29/06	8/29/06	39,200.00	4.80	163.46	7.85	209.02	26.64	25.26	3.22	41.27	5.26
6575/11	8/30/06	8/30/06	39,200.00	4.80	163.46	7.85	209.02	26.64	25.26	3.22	41.27	5.26
6575/12	8/31/06	8/31/06	33,600.00	4.10	140.11	5.74	153.04	26.64	18.50	3.22	30.22	5.26

LAND APPLICATION SITE INFORMATION FOR
CORVALLIS WWRP
OR002636-1

Avg % Total Solids:	4.98
Total Gallons:	2,004,800.00
Total Dry Tons:	414.54
Gallons / Acre:	58,843.56

Total Cumulative Metal Loading (Kg/Hectare)

As Average:	0.48
As Maximum:	0.53
Cd Average:	0.26
Cd Maximum	0.28
Cu Average:	46.43
Cu Maximum	49.56
Pb Average:	4.16
Pb Maximum	4.75
Hg Average:	0.13
Hg Maximum	0.21
Mo Average:	0.70
Mo Maximum	0.84
Ni Average:	2.35
Ni Maximum:	2.67
Se Average:	0.77
Se Maximum	0.94
Zn Average:	88.08
Zn Maximum:	89.12

Comments:

LAND APPLICATION SITE INFORMATION FOR CORVALLIS WWRP OR002636-1

Monitoring Year: **2006**
Site Name: **OSU, (SOAP CREEK RANCH)** Site No: **76**
Hydrologic Unit:
Owner: **OREGON STATE UNIVERSITY**
Operator: **OSU DEPT. OF ANIMAL SCIEN(**
Applier: **CITY OF CORVALLIS**
Latitude: Longitude:
Location:
Section: **23/24**
Township: **T10S** Range: **R5W**
Size (acres): **48.43** Size (hectares): **19.60**
Crop: **GRASS HAY**
Application Rate (tons/acre): **4.14** Application Rate (MT/HA): **9.27**

	English Units		Metric Units	
Recommended N for Crop	**100.00**	lbs/Acre	**112.00**	Kg/Ha
Nitrogen Available:	**21.77**	lbs/Dry Ton	**52.90**	Kg/Metic Ton
Total Biosolids Applied:	**200.36**	Dry Tons	**181.77**	Dry Metric Tons
Actual Application Rate:	**4.14**	Dry Tons/Acre	**9.27**	Dry Metric Tons / Ha
Total Recommended N:	**4,843.00**	lbs/field	**2,196.75**	Kg/Field
Total Applied N:	**4,430.76**	lbs/field	**2,009.76**	Kg/Field
% of Recommended N:	**91.49**	%	**91.49**	%
Total P Applied:	**867.09**	lbs/field	**393.31**	Kg/Field
Total K Applied:	**1,038.05**	lbs/field	**470.85**	Kg/Field

LAND APPLICATION SITE INFORMATION FOR CORVALLIS WWRP OR002636-1

Individual Load Tracking

Haul # / Desc	Date Hauled	Date Applied	Gallons	% Total Solids	Wet Tons	Dry Tons	lbs N / load	lbs N / DT	lbs P / load	lbs P / DT	lbs K / load	lbs K / DT
6565/1	7/17/02	7/17/02	16,800.00	5.10	70.06	3.57	93.64	26.21	128.62	36.00	12.15	3.40
6565/2	7/18/02	7/18/02	39,200.00	5.20	163.46	8.50	222.79	26.21	306.00	36.00	28.90	3.40
6565/3	7/19/02	7/19/02	28,000.00	5.00	116.76	5.84	153.01	26.21	210.17	36.00	19.85	3.40
6565/4	7/22/02	7/22/02	44,800.00	4.70	186.82	8.78	216.44	24.65	368.77	42.00	33.37	3.80
6565/5	7/23/02	7/23/02	44,800.00	4.70	186.82	8.78	216.44	24.65	368.77	42.00	33.37	3.80
6565/6	7/24/02	7/24/02	44,800.00	4.90	186.82	9.15	225.65	24.65	384.47	42.00	34.79	3.80
6565/7	7/25/02	7/25/02	44,800.00	5.00	186.82	9.34	230.25	24.65	392.31	42.00	35.50	3.80
6565/8	7/26/02	7/26/02	33,600.00	5.20	140.11	7.29	179.60	24.65	306.00	42.00	27.69	3.80
6565/9	7/29/02	7/29/02	44,800.00	5.00	186.82	9.34	230.25	24.65	392.31	42.00	35.50	3.80
6565/10	7/30/02	7/30/02	44,800.00	5.10	186.82	9.53	234.86	24.65	400.16	42.00	36.20	3.80
6565/11	7/31/02	7/31/02	44,800.00	5.10	186.82	9.53	234.86	24.65	400.16	42.00	36.20	3.80
6565/12	8/1/02	8/1/02	39,200.00	4.70	163.46	7.68	189.38	24.65	322.68	42.00	29.19	3.80
6575/1	7/17/02	7/17/02	16,800.00	5.10	70.06	3.57	93.64	26.21	128.62	36.00	12.15	3.40
6575/2	7/18/02	7/18/02	39,200.00	5.20	163.46	8.50	222.79	26.21	306.00	36.00	28.90	3.40
6575/3	7/19/02	7/19/02	44,800.00	5.00	186.82	9.34	244.82	26.21	336.27	36.00	31.76	3.40
6575/4	7/22/02	7/22/02	28,000.00	4.70	116.76	5.49	135.27	24.65	230.48	42.00	20.85	3.80
6575/5	7/23/02	7/23/02	44,800.00	4.70	186.82	8.78	216.44	24.65	368.77	42.00	33.37	3.80
6575/6	7/24/02	7/24/02	44,800.00	4.90	186.82	9.15	225.65	24.65	384.47	42.00	34.79	3.80
6575/7	7/25/02	7/25/02	33,600.00	5.00	140.11	7.01	172.69	24.65	294.24	42.00	26.62	3.80
6575/8	7/26/02	7/26/02	39,200.00	5.20	163.46	8.50	209.53	24.65	357.01	42.00	32.30	3.80
6575/9	7/29/02	7/29/02	44,800.00	5.00	186.82	9.34	230.25	24.65	392.31	42.00	35.50	3.80
6575/10	7/30/02	7/30/02	44,800.00	5.10	186.82	9.53	234.86	24.65	400.16	42.00	36.20	3.80
6575/11	7/31/02	7/31/02	44,800.00	5.10	186.82	9.53	234.86	24.65	400.16	42.00	36.20	3.80
6575/12	11/19/02	11/19/02	44,800.00	4.70	186.82	8.78	216.44	24.65	368.77	42.00	33.37	3.80
6565/1	7/5/05	7/5/05	5,600.00	7.00	23.35	1.63	39.92	24.42	5.23	3.20	8.99	5.50
6565/2	7/6/05	7/6/05	39,200.00	7.00	163.46	11.44	279.43	24.42	36.62	3.20	64.08	5.60
6565/3	7/7/05	7/7/05	44,800.00	3.30	186.82	6.16	150.55	24.42	19.73	3.20	34.52	5.60
6565/4	7/12/05	7/12/05	39,200.00	4.80	163.46	7.85	191.61	24.42	25.11	3.20	43.94	5.60
6565/5	7/13/05	7/13/05	39,200.00	5.10	163.46	8.34	203.58	24.42	26.68	3.20	46.69	5.60
6565/6	7/14/05	7/14/05	39,200.00	4.00	163.46	6.54	159.67	24.42	20.92	3.20	36.62	5.60
6565/7	7/15/05	7/15/05	39,200.00	5.40	163.46	8.83	215.56	24.42	28.25	3.20	49.43	5.60
6565/8	7/19/05	7/19/05	39,200.00	5.70	163.46	9.32	227.53	24.42	29.82	3.20	52.18	5.60
6565/9	7/20/05	7/20/05	44,800.00	5.20	186.82	9.71	237.23	24.42	31.09	3.20	54.40	5.60
6565/10	7/21/05	7/21/05	33,600.00	5.70	140.11	7.99	195.03	24.42	25.56	3.20	44.72	5.60
6565/11	7/22/05	7/22/05	39,200.00	5.10	163.46	8.34	203.58	24.42	26.68	3.20	46.69	5.60
6565/12	7/25/05	7/25/05	39,200.00	4.80	163.46	7.85	191.61	24.42	25.11	3.20	43.94	5.60
6565/13	7/26/05	7/26/05	39,200.00	4.00	163.46	6.54	159.67	24.42	20.92	3.20	36.62	5.60
6565/14	7/27/05	7/27/05	11,200.00	3.90	46.70	1.82	44.48	24.42	5.83	3.20	10.20	5.60
6575/1	8/8/05	8/8/05	11,200.00	4.40	46.70	2.05	50.18	24.42	6.58	3.20	11.51	5.60
6575/2	8/11/05	8/11/05	33,600.00	6.30	140.11	8.83	215.56	24.42	28.25	3.20	49.43	5.60
6575/3	8/12/05	8/12/05	39,200.00	4.40	163.46	7.19	175.64	24.42	23.02	3.20	40.28	5.60
6565/15	8/15/05	8/15/05	39,200.00	5.10	163.46	8.34	203.58	24.42	26.68	3.20	46.69	5.60
6575/4	8/16/05	8/16/05	39,200.00	4.60	163.46	7.52	183.62	24.42	24.06	3.20	42.11	5.60
6565/16	8/17/05	8/17/05	5,600.00	4.60	23.35	1.07	26.23	24.42	3.44	3.20	6.02	5.60
6575/5	8/17/05	8/17/05	33,600.00	4.60	140.11	6.45	157.39	24.42	20.62	3.20	36.09	5.60
6565/17	8/18/05	8/18/05	28,000.00	6.30	116.76	7.36	179.63	24.42	23.54	3.20	41.19	5.60
6565/18	8/19/05	8/19/05	39,200.00	5.30	163.46	8.66	211.56	24.42	27.72	3.20	48.52	5.60
6565/19	8/22/05	8/22/05	39,200.00	4.50	163.46	7.36	179.63	24.42	23.54	3.20	41.19	5.60
6565/20	8/23/05	8/23/05	39,200.00	4.50	163.46	7.36	179.63	24.42	23.54	3.20	41.19	5.60
6565/21	8/24/05	8/24/05	16,800.00	6.20	70.06	4.34	106.07	24.42	13.90	3.20	24.32	5.60
6565/1	7/24/06	7/14/06	11,200.00	2.70	46.70	1.26	28.37	22.50	36.06	28.60	6.83	5.42
6565/2	7/25/06	7/25/06	28,000.00	4.10	116.76	4.79	64.77	13.53	40.12	8.38	28.24	5.90
6565/3	7/26/06	7/26/06	39,200.00	4.50	163.46	7.36	99.53	13.53	61.64	8.38	43.40	5.90
6565/4	7/27/06	7/27/06	33,600.00	5.70	140.11	7.99	108.06	13.53	66.93	8.38	47.12	5.90
6565/5	7/28/06	7/28/06	22,400.00	5.60	93.41	5.23	70.77	13.53	43.83	8.38	30.86	5.90
6565/6	7/31/06	7/31/06	22,400.00	4.00	93.41	3.74	50.55	13.53	31.31	8.38	22.04	5.90

LAND APPLICATION SITE INFORMATION FOR
CORVALLIS WWRP
OR002636-1

6565/7	8/1/06	8/1/06	33,600.00	4.00	140.11	5.60	142.41	25.41	13.11	2.34	27.46	4.90
6565/8	8/2/06	8/2/06	33,600.00	4.40	140.11	6.16	156.65	25.41	14.43	2.34	30.21	4.90
6565/9	8/3/06	8/3/06	39,200.00	4.60	163.46	7.52	191.07	25.41	17.60	2.34	36.84	4.90
6565/10	8/4/06	8/4/06	39,200.00	4.60	163.46	7.52	191.07	25.41	17.60	2.34	36.84	4.90
6565/11	8/7/06	8/7/06	39,200.00	5.10	163.46	8.34	211.83	25.41	19.51	2.34	40.85	4.90
6565/12	8/8/06	8/8/06	39,200.00	4.40	163.46	7.19	182.76	25.41	16.83	2.34	35.24	4.90
6565/13	8/9/06	8/9/06	28,000.00	5.40	116.76	6.31	160.21	25.41	14.75	2.34	30.89	4.90
6565/14	8/10/06	8/10/06	39,200.00	5.70	163.46	9.32	236.76	25.41	21.80	2.34	45.66	4.90
6565/15	8/11/06	8/11/06	39,200.00	5.50	163.46	8.99	228.45	25.41	21.04	2.34	44.05	4.90
6565/16	8/14/06	8/14/06	33,600.00	4.40	140.11	6.16	156.65	25.41	14.43	2.34	30.21	4.90
6575/1	7/24/06	7/24/06	11,200.00	2.70	46.70	1.26	28.37	22.50	36.06	28.60	6.83	5.42
6575/2	7/25/06	7/25/06	33,600.00	4.10	140.11	5.74	77.72	13.53	48.14	8.38	33.89	5.90
6575/3	7/27/06	7/27/06	33,600.00	5.70	140.11	7.99	108.06	13.53	66.93	8.38	47.12	5.90
6575/4	7/28/06	7/28/06	28,000.00	5.60	116.76	6.54	88.47	13.53	54.79	8.38	38.58	5.90
6575/5	7/31/06	7/31/06	33,600.00	4.00	140.11	5.60	75.83	13.53	46.97	8.38	33.07	5.90
6575/6	8/1/06	8/1/06	33,600.00	4.00	140.11	5.60	142.41	25.41	13.11	2.34	27.46	4.90
6575/7	8/2/06	8/2/06	33,600.00	4.40	140.11	6.16	156.65	25.41	14.43	2.34	30.21	4.90
6575/8	8/3/06	8/3/06	33,600.00	4.60	140.11	6.45	163.77	25.41	15.08	2.34	31.58	4.90
6575/9	8/4/06	8/4/06	33,600.00	4.60	140.11	6.45	163.77	25.41	15.08	2.34	31.58	4.90
6575/10	8/7/06	8/7/06	33,600.00	5.10	140.11	7.15	181.57	25.41	16.72	2.34	35.01	4.90
6575/11	8/8/06	8/8/06	39,200.00	4.40	163.46	7.19	182.76	25.41	16.83	2.34	35.24	4.90
6575/12	8/9/06	8/9/06	33,600.00	5.40	140.11	7.57	192.25	25.41	17.70	2.34	37.07	4.90
6575/13	8/10/06	8/10/06	39,200.00	5.70	163.46	9.32	236.76	25.41	21.80	2.34	45.66	4.90
6575/14	8/11/06	8/11/06	33,600.00	5.50	140.11	7.71	195.81	25.41	18.03	2.34	37.76	4.90
6575/15	8/14/06	8/14/06	33,600.00	4.40	140.11	6.16	156.65	25.41	14.43	2.34	30.21	4.90

Avg % Total Solids:	**4.89**
Total Gallons:	**2,805,600.00**
Total Dry Tons:	**574.09**
Gallons / Acre:	**57,931.03**

Total Cumulative Metal Loading (Kg/Hectare)

As Average:	**0.48**
As Maximum:	**0.53**
Cd Average:	**0.26**
Cd Maximum	**0.28**
Cu Average:	**46.43**
Cu Maximum	**49.56**
Pb Average:	**4.16**
Pb Maximum	**4.75**
Hg Average:	**0.13**
Hg Maximum	**0.21**
Mo Average:	**0.70**
Mo Maximum	**0.84**
Ni Average:	**2.35**
Ni Maximum:	**2.67**
Se Average:	**0.77**
Se Maximum	**0.94**
Zn Average:	**88.08**
Zn Maximum:	**89.12**

Comments:

APPENDIX B

SITE AUTHORIZATION LETTERS

Oregon
John A. Kitzhaber, M.D., Governor

Department of Environmental Quality
Western Region
Salem Office
750 Front St. NE
Suite 120
Salem, OR 97310
(503) 378-8240
(503) 378-3684 TTY

May 5, 1999

Dan Hanthorn
Wastewater Operations Supervisor
City of Corvallis
Post Office Box 1083
Corvallis, OR 97339

RE: WQ/SDS-City of Corvallis
NPDES No. 100984
File No. 20151
Biosolids Land Application Authorization
Corvallis Airport Site
T12S; R5W; Sec 22,27,28, 33,34 -900 acres
Benton County

Dear Mr. Hanthorn:

The Department of Environmental Quality (Department) reevaluated your request for authorization to land apply anaerobically digested biosolids from the Corvallis Wastewater Treatment Plant (WWTP) onto the Corvallis City property (referenced above). The site was initially approved by the Department on March 8, 1990, and biosolids have been beneficially used on the site since then to supply nutrients for grass seed. The re-evaluation was based upon updated maps of individual fields and current Biosolids analysis.

Based upon the Department's evaluation of the information submitted; your request to use the property is approved for the application of anaerobically digested biosolids from the City of Corvallis from April 15th to October 15th of each year provided:

1. Solids processing and handling will comply with Oregon sludge rules and guidelines (OAR 340-50-005 to OAR 340-50-080) and all other applicable statutes, rules, and federal regulations;

2. Sludge volatile solids shall be reduced by 38% or more via anaerobic digestion prior to land spreading;

3. Prior to land application, the biosolids must meet one of the three pathogen reduction standards required under 40 CFR 503.32(b);

4. Based on the City's most recent biosolids analysis, existing crop, topographic, and soil conditions of the field, 100 pounds of available nitrogen per acre, per year 4.4 dry tons per acre, can be applied. Changes in biosolids characteristics or cropping practices will necessitate appropriate adjustments in the annual application rate to maintain nitrogen loadings consistent with crop demands;

5. If other sources of nitrogen will be applied to the fields, the biosolids application rate must be reduced so that commercial fertilizer nitrogen, animal manure, or green manure nitrogen, plus biosolids nitrogen, do not exceed the agronomic loading rate of any field within the site;

6. The City's truck operator shall immediately clean up any spillage of biosolids. Spillage which cannot be completely cleaned up must be covered with dry lime and posted where appropriate;

7. The DEQ Salem Office ((503) 378-8240) shall be notified within one hour of any biosolids spills greater than 50 gallons, or other threats to the environment that may occur. Spills that occur after normal working hours shall be reported to the Emergency Management Division (EMD) within one hour. The telephone number of EMD is 1-800-452-0311;

8. A 50-foot minimum (300 feet minimum if biosolids gun application is used) setback shall be maintained between biosolids application areas and all highways, public roadways, and property lines;

9. A minimum setback of 200 feet shall be maintained between biosolids application areas and all wells and other water sources;

10. A minimum setback of 50 feet shall be maintained from seasonal drainage ways;

11. Biosolids shall be applied evenly and thinly in a manner that will prevent ponding and runoff. No Biosolids shall be applied, when within the prior 24 hours, 0.25 inches of rain or greater has fallen in the Corvallis area;

12. No food crops for direct human consumption, whose harvested parts are grown below ground shall be planted on any biosolids amended area for at least 38 months following biosolids land spreading;

13. No food chain crops for direct human consumption, whose harvested parts are grown above ground (e.g., tomatoes) shall be grown on any biosolids amended area for at least 14 months following biosolids land spreading;

14. A 30-day fallow period shall follow the application of biosolids prior to grazing livestock on any biosolids amended field or harvesting animal feed crops from biosolids amended areas;

15. Public access to field sites shall be restricted for at least 12 months after biosolids land spreading has ceased;

16. In the event an odor problem is reported at any land application site, immediate steps, such as, but not limited to, the addition of liming materials, must be taken to counteract that condition;

17. The City shall maintain daily records for the biosolids land application site which indicate (on a field map grid system) where, when, and how many (gallons) biosolids were land applied to the specific fields on the property to verify that annual loading limitations are not being exceeded;

18. Site records shall indicate the date, location and amount of biosolids applied;

19. The Department shall have the right to enter the Corvallis' WWTP's place of record keeping to review biosolids management operations and records, obtain copies of any records required under the terms of this authorization and the WWTP's biosolids (sludge) management plan, inspect any monitoring equipment required under this authorization, inspect any collection, transport, or land spreading vehicles, and obtain any photographic documentation or evidence deemed appropriate;

20. The City shall provide DEQ an annual summary of biosolids processing and land application activities;

21. The Department may impose, at a future date, additional restrictions or conditions deemed necessary to assure adequate biosolids management. Any variations from the City's (DEQ approved) biosolids management plan and this authorization letter must receive prior written approval from the Western Region - Salem Office; and,

22. This authorization is subject to revocation should health hazards, environmental degradation, or nuisance conditions develop as a result of inadequate solids treatment or site management. If operations are not conducted in accordance with terms specified under this authorization, the Department will initiate necessary remedial action.

If you have any questions regarding this authorization, please contact me at (503) 378-8240, extension 236.

Sincerely,

Tom Fisher
Environmental Specialist
Western Region-Salem Office

TRF:clp

x:\tfisher\CorvallissaBA
cc: Doug Peters, Municipal Wastewater Section
Dept of Environmental Quality

Oregon
Theodore Kulongoski, Governor

Department of Environmental Quality
Western Region Eugene Office
165 East 7th Avenue, suite 100
Eugene, OR 97401
(541) 687-7439
FAX (541) 686-7551

PUBLIC WORKS
DEPARTMENT
JAN 07 2011

Janaury 5, 2011

Gene Freel, Wastewater Operator
Wastewater Regional Plant
City of Corvallis
1245 NE 3rd Street
P.O. Box 1083
Corvallis OR 97339

Phone ((541) 754-1757

Re: File Number: 20151
Permit Number: 101714
City of Corvallis Biosolid Land Application Site Reauthorization
WR-2010-0017-BS
Lloyd Anderson property owner, Anderson Farms
4210 NW Crescent Valley Dr., Corvallis OR 97330
Twp. 11S, R. 5 W. Sec. 14 Tax Lot
Parcel total approx. 36 acres
Benton County

Dear Mr. Gene Freel:

Based on the current property owner's approval, this letter represents approval of your request to land apply Class B Biosolid on the above referenced property. This biosolid authorization is good until the property ownership changes, the farming management/practices change such that there is significant non-compliance with the biosolid rules (threat to public health or the environment), or the current property owner requests the city to discontinue the land application of biosolid on this site. This authorization is subject to revocation through a contested hearing process by the Department if significant noncompliance issues arise.

This authorization to apply biosolids the above referenced property is subject to criteria detailed in the Oregon Administrative Rules, Chapter 340, Division 50 and the following conditions:

Responsibility:

It is the responsibility of the City of Corvallis to insure the proper handling and application of all biosolids generated. Transportation of the biosolids to the application site shall be done in such a manner as to prevent leaking or spilling the biosolids onto the highways, streets, roads, waterways or other land surfaces not approved for biosolids application.

Site Use Limitations:

1. Controlled access to the biosolids site must be maintained for a period of 12 months following biosolids application.

2. Grazing of domestic animals should not be allowed on this site for 30 days following the last day biosolids were land applied on this site. There is a 90 day grazing restriction for lactating animals (dairy cattle).

Accidental Spillage:

The permittee shall immediately clean up any spillage of biosolids and notify the DEQ Roseburg office at 440-3338 of any such occurrences. Call Oregon Emergency Response System (ORES) at 1-800- 452-0311 for any spills that make it to or could make it to waters of the state. Spillage which cannot be completely cleaned up shall be covered with hydrated lime (Calcium Hydroxide) or lime (calcium oxide). A 50-lb. bag of liming material shall remain available during transportation of the biosolids.

Monitoring:

1. Written daily land application records shall be kept on a field grid map or other easily readable system. Corvallis is responsible for tracking the land application of biosolids on daily basis (number of dry pounds nitrogen land applied per acre). Corvallis shall provide the DEQ with monthly summaries of biosolids land application activities along with a current biosolids analysis in Corvallis' annual report.

2. A copy of the current year's biosolid analysis should be carried with all biosolids that are to be land applied at this site.

3. A copy of this site authorization letter and a signed biosolid pathogen and vector attraction reduction certification statement should accompany all biosolids land applied at this site. The responsible parties who apply biosolids should review the site authorization letter prior to land applying biosolids to this site

4. If this site is used 2 out 3 consecutive years (loaded at agronomic rates) then a soil carry over nitrogen test is required for those fields in question prior to the end of the third year.

If you have any questions regarding this approval, please call me at (541) 687-7439.

Sincerely,

Paul Kennedy

Paul Kennedy
Natural Resource Specialist 3

Cc: Portland-DEQ Biosolid Program
Marion County Health Department

6/11/96

Oregon

June 14, 1996

DEPARTMENT C
ENVIRONMENTA
QUALITY

Western Region -
Salem Office

Michael K. Clark
City of Corvallis
Wastewater Treatment Plant
Post Office Box 1083
Corvallis, OR 97339

Re: WQ/SDS-City of Corvallis
Bressler Farm
Section 35; TL 12S; R5W
Benton County
Biosolids Land Application
Authorization

Dear Mr. Clark:

The Department of Environmental Quality (Department) has evaluated your request for authorization to land apply anaerobically digested biosolids from the Corvallis Wastewater Treatment Plant (WWTP) onto the Bressler Farm referenced above. The biosolids will supply nutrients for 35 acres of hay. Material you submitted with the biosolids land application site authorization request included information characterizing soils, topography, biosolids, crop, crop management, annual available nitrogen loading (agronomic loading rate), and a copy of a signed agreement regarding the use of biosolids.

Based upon our evaluation of the information submitted, your request to use the property is approved for dry weather (May through October) applications of anaerobically digested biosolids from the City provided:

1. Solids processing and handling will comply with Oregon Sludge Rules and Guidelines (OAR 340-50-005 to OAR 340-50-080) and all other applicable statutes, rules, and Federal regulations;

2. Sludge volatile solids shall be reduced by 38% or more via anaerobic digestion prior to land spreading;

3. Prior to land application, the biosolids must meet one of the three pathogen reduction standards required under 40 CFR 503.32(b);

750 Front St. NE
Suite 120
Salem, OR 97310
(503) 378-8240
(503) 378-3684 TDD

4. Based on the City's most recent biosolids analysis, existing crop, topographic, and soil conditions of the field, 100 pounds of available nitrogen per acre, per year (7.85 dry tons per acre) can be applied. Changes in biosolids characteristics or cropping practices will necessitate appropriate adjustments in the annual application rate to maintain nitrogen loadings consistent with crop demands;

5. If other sources of nitrogen will be applied to the fields, the biosolids application rate must be reduced so that commercial fertilizer nitrogen, animal manure, or green chop nitrogen, plus biosolids nitrogen, do not exceed the agronomic loading rate of any field within the site;

6. The City's truck operator shall immediately clean up any spillage of biosolids. Spillage which cannot be completely cleaned up must be covered with dry lime and posted where appropriate;

7. The Western Region-Salem Office (503-378-8240) shall be notified within one hour of any biosolids spills greater than 50 gallons, or other threats to the environment that may occur. Spills that occur after normal working hours shall be reported to the Emergency Management Division (EMD) within one hour. The telephone number of EMD is 1-800-452-0311;

8. A 50 foot minimum (75 feet minimum if biosolids gun application is used) setback shall be maintained between biosolids application areas and all highways, public roadways, and property lines;

9. A minimum setback of 200 feet shall be maintained between biosolids application areas and all wells and other water sources;

10. Biosolids shall be applied evenly and thinly in a manner that will prevent ponding and runoff;

11. No food crops for direct human consumption, whose harvested parts are grown below ground, shall be planted on any biosolids amended area for at least 38 months following biosolids land spreading;

12. No food chain crops for direct human consumption, whose harvested parts are grown above ground (e.g., tomatoes), shall be grown on any biosolids amended area for at least 14 months following biosolids land spreading;

13. A 30-day fallow period shall follow the application of biosolids prior to grazing livestock on any biosolids amended field or harvesting animal feed crops from biosolids amended areas;

14. Public access to fields site shall be restricted for at least 12 months after biosolids land spreading has ceased;

15. In the event an odor problem is reported at any land application site, immediate steps, such as, but not limited to, the addition of liming materials, must be taken to counteract that condition;

16. The City shall maintain daily records for the biosolids land application site which indicate (on a field map grid system) where, when, and how much (gallons) biosolids were land applied to the specific fields on the property to verify that annual loading limitations are not being exceeded;

17. Site records shall indicate the date, location, and amount of biosolids applied; segments of each field that received biosolids, sludge, pounds of nutrients, arsenic, cadmium, chromium, copper, lead, mercury, molybdenum, nickel, selenium and zinc applied to each area receiving biosolids; and the type of crop grown. This data shall be maintained at the WWTP for a minimum of five (5) years and be made available to the Department for review upon request;

18. The Department shall have the right to enter the Corvallis WWTP's place of recordkeeping to review biosolids management operations and records; obtain copies of any records required under the terms of this authorization and the WWTP's biosolids (sludge) management plan; inspect any monitoring equipment required under this authorization; inspect any collection, transport, or land spreading vehicles; and obtain any photographic documentation or evidence deemed appropriate;

19. The City shall provide DEQ an annual summary of biosolids processing and land application activities;

20. The Department may impose, at a future date, additional restrictions or conditions deemed necessary to assure adequate biosolids management. Any variations from the City's DEQ approved biosolids management plan, and this authorization letter must receive prior written approval from the Western Region-Salem Office; and,

21. This authorization is subject to revocation should health hazards, environmental degradation, or nuisance conditions develop as a result of inadequate solids treatment or site management. If operations are not conducted in accordance with terms specified under this authorization, the Department will initiate necessary remedial action.

If you have any questions regarding this authorization, please contact me at (503) 378-8240, extension 236.

Sincerely,

Tom Fisher
Environmental Specialist
Western Region-Salem Office

TRF:lmd
x:\tfisher\corvalli

cc: Doug Peters, Municipal Wastewater Section, DEQ
Scott Bressler, 28078 Payne Road, Corvallis, OR 97333

RECEIVED
JUN 21 1995
PUBLIC WORKS

Oregon

DEPARTMENT OF ENVIRONMENTAL QUALITY

Western Region - Salem Office

June 20, 1995

Dan R. Hanthorn
Wastewater Operations Manager
P.O.Box 1083
Corvallis, OR 97339

Re: WQ/SDS-City of Corvallis
Grey Farm
Section 24/13; TL 10S; R 4/5 W
Benton County
Sludge Land Application Authorization

Dear Mr. Hanthorn:

The Department has evaluated your request for authorization to land apply anaerobically digested biosolids from the Corvallis Wastewater Treatment Plant (WWTP) onto the Grey Farm referenced above. The biosolids will supply nutrients for 180 acres of grass pasture. Material you submitted with the biosolids land application site authorization request included information characterizing soils, topography, biosolids, crop, crop management, annual available nitrogen loading (agronomic loading rate), site life, and a copy of a signed agreement regarding the use of biosolids.

Based upon our evaluation of the information submitted, your request to use the property is approved for dry weather (May through October) applications of anaerobically digested biosolids from the City, provided:

1. Solids processing and handling will comply with Oregon Sludge Rules and Guidelines (OAR 340-50-005 to OAR 340-50-080) and all other applicable statutes, rules, and Federal regulations;

2. Sludge volatile solids shall be reduced by 38% or more via anaerobic digestion prior to land spreading;

3. Prior to land application, the biosolids must meet one of the three pathogen reduction standards required under 40 CFR 503.32(b);

4. Based on the City's most recent biosolids analysis, existing crop, topographic, and soil conditions of the field, 100 pounds of available nitrogen per acre, per year (3.6 dry tons per acre) can be applied. Changes in biosolids characteristics or cropping practices will necessitate appropriate adjustments in the annual application rate to maintain nitrogen loadings consistent with crop demands;

750 Front St. NE
Suite 120
Salem, OR 97310
(503) 378-8240
(503) 378-3684 TDD

5. If other sources of nitrogen will be applied to the fields, the biosolids application rate must be reduced so that commercial fertilizer nitrogen, animal manure, or green chop nitrogen, plus biosolids nitrogen, do not exceed the agronomic loading rate of any field within the site;

6. The City's truck operator shall immediately clean up any spillage of biosolids. Spillage which cannot be completely cleaned up must be covered with dry lime and posted where appropriate;

7. The Western Region-Salem Office (503-378-8240) shall be notified within one hour of any biosolids spills greater than 50 gallons, or other threats to the environment that may occur. Spills that occur after normal working hours shall be reported to the Emergency Management Division (EMD) within one hour. The telephone number of EMD is 1-800-452-0311;

8. [redacted] (75 feet minimum if biosolids gun application is used) setback shall be maintained between biosolids application areas and [redacted] public roadways and property lines;

9. A minimum setback of [redacted] shall be maintained between biosolids application areas and all [redacted] wells and other water sources;

10. Biosolids shall be applied evenly and thinly in a manner that will prevent ponding and runoff;

11. Because the runoff potential of fields is high, [redacted] or the ground is frozen. An interval of at least three days shall occur between heavy periods of precipitation and biosolids land application;

12. No food crops for direct human consumption, whose harvested parts are grown below ground, shall be planted on any biosolids amended area for at least 38 months following biosolids land spreading;

13. No food chain crops for direct human consumption, whose harvested parts are grown above ground (e.g., tomatoes), shall be grown on any biosolids amended area for at least 14 months following biosolids land spreading;

14. A 30-day fallow period shall follow the application of biosolids prior to grazing livestock on any biosolids amended field or harvesting animal feed crops from biosolids amended areas;

15. Public access to fields site shall be restricted for at least 12 months after biosolids land spreading has ceased;

16. In the event an odor problem is reported at any land application site, immediate steps, such as, but not limited to, the addition of liming materials, must be taken to counteract that condition;

17. The City shall maintain daily records for the biosolids land application site which indicate (on a field map grid system) where, when, and how much (gallons) biosolids were land applied to the specific fields on the property to verify that annual loading limitations are not being exceeded;

18. Site records shall indicate the date, location, and amount of biosolids applied; segments of each field that received biosolids, sludge, pounds of nutrients, arsenic, cadmium, chromium, copper, lead, mercury, molybdenum, nickel, selenium and zinc applied to each area receiving biosolids; and the type of crop grown. These data shall be maintained at the WWTP for a minimum of five (5) years and be made available to the Department for review upon request;

19. The Department shall have the right to enter the Corvallis' WWTP's place of record keeping to review biosolids management operations and records; obtain copies of any records required under the terms of this authorization and the WWTP's biosolids (sludge) management plan; inspect any monitoring equipment required under this authorization; inspect any collection, transport, or land spreading vehicles; and obtain any photographic documentation or evidence deemed appropriate;

20. The City shall provide DEQ an annual summary of biosolids processing and land application activities;

21. The Department may impose, at a future date, additional restrictions or conditions deemed necessary to assure adequate biosolids management. Any variations from the City's DEQ approved biosolids management plan, and this authorization letter must receive prior written approval from the Western Region-Salem Office; and,

22. This authorization is subject to revocation should health hazards, environmental degradation, or nuisance conditions develop as a result of inadequate solids treatment or site management. If operations are not conducted in accordance with terms specified under this authorization, the Department will initiate necessary remedial action.

If you have any questions regarding this authorization, please contact me at (503) 378-8240, extension 236.

Sincerely,

Tom Fisher
Environmental Specialist
Western Region-Salem Office

TRF:klw
x:\trf\corvalli

cc: Bijan Pour, Municipal Wastewater Section
Dennis Grey, 39247 Rifle Range Road, Corvallis, OR 97339

Oregon
John A. Kitzhaber, M.D., Governor

Department of Environmental Quality
Western Region – Salem Office
750 Front St. NE, Ste. 120
Salem, OR 97301-1039
(503) 378-8240
(503) 378-3684 TTY

August 6, 1999

Dan Hanthorn
Wastewater Operations Supervisor
City of Corvallis
Post Office Box 1083
Corvallis, OR 97339

RECEIVED
AUG 16 1999
PUBLIC WORKS

Re: WQ/SDS-City of Corvallis
NPDES # 100984
Biosolids Land Application
OSU, OSU Dept. of Animal Science Authorization
Sec, 23,24,26;26; TL10S;R5W
Field # 70-22.81 acres; Field # 75- 33.41 acres; Field # 76-43.94 acres
Field # 77- 32.21 acres; Field # 78 46.36 acres; Field # 80-42.25 acres
Benton County

Dear Mr. Hanthon:

The Department of Environmental Quality (Department) evaluated your request for authorization to land apply anaerobically digested biosolids from the Corvallis Wastewater Treatment Plant (WWTP) onto the OSU farm (referenced above). The biosolids will supply nutrients for grass hay/pasture. Material you submitted with the biosolids land application site authorization request included: information characterizing soils, topography, biosolids, crop, crop management, annual available nitrogen loading (agronomic loading rate), site life, and a copy of a signed agreement regarding the use of the Bioslids.

During my December 10, 1998 review I found inclusions of poorly drained soils on all six fields listed above, therefore application of Biosolids is permitted only from April 15 to October 15 of each year.

Based upon the Department's evaluation of the information submitted your request to use the property is approved for the application of anaerobically digested biosolids from the City of Corvallis (City) provided:

1. Solids processing and handling will comply with Oregon sludge rules and guidelines (OAR 340-50-005 to OAR 340-50-080) and all other applicable statutes, rules, and federal regulations;

2. Sludge volatile solids shall be reduced by 38% or more via anaerobic digestion prior to land spreading;

3. Prior to land application, the biosolids must meet one of the three pathogen reduction standards required under 40 CFR 503.32(b);

4. Based on the City's most recent biosolids analysis, existing crop, topographic, and soil conditions of the field, 100 pounds of available nitrogen per acre, per year (3.7 dry tons per acre, can be applied. Changes in biosolids characteristics or cropping practices will necessitate appropriate adjustments in the annual application rate to maintain nitrogen loadings consistent with crop demands.

5. If other sources of nitrogen will be applied to the fields, the biosolids application rate must be reduced so that commercial fertilizer nitrogen, animal manure, or green manure nitrogen, plus biosolids nitrogen, do not exceed the agronomic loading rate of any field within the site;

6. The City's truck operator shall immediately clean up any spillage of biosolids. Spillage which cannot be completely cleaned up must be covered with dry lime and posted where appropriate;

7. The DEQ Salem Office ((503) 378-8240) shall be notified within one hour of any biosolids spills greater than 50 gallons, or other threats to the environment that may occur. Spills that occur after normal working hours shall be reported to the Emergency Management Division (EMD) within one hour. The telephone number of EMD is 1-800-452-0311;

8. A 50-foot minimum (300 feet minimum if biosolids gun application is used) setback shall be maintained between biosolids application areas and all highways, public roadways, and property lines.

9. A minimum setback of 200 feet shall be maintained between biosolids application areas and all wells and other water sources.

10. A minimum setback of 50 feet shall be maintained from seasonal drainage ways and 100 feet from Soap Creek.

11. Biosolids shall be applied evenly and thinly in a manner that will prevent ponding and runoff. No Biosolids shall be applied, when within the prior 24 hours, 0.25 inches of rain or greater has fallen in the Corvallis area;

12. No food crops for direct human consumption, whose harvested parts are grown below ground shall be planted on any biosolids amended area for at least 38 months following biosolids land spreading;

13. No food chain crops for direct human consumption, whose harvested parts are grown above ground (e.g., tomatoes) shall be grown on any biosolids amended area for at least 14 months following biosolids land spreading;

14. A 30-day fallow period shall follow the application of biosolids prior to grazing livestock on any biosolids amended field or harvesting animal feed crops from biosolids amended areas;

15. Public access to field sites shall be restricted for at least 12 months after biosolids land spreading has ceased;

16. In the event an odor problem is reported at any land application site, immediate steps, such as, but not limited to, the addition of liming materials, must be taken to counteract that condition;

17. The City shall maintain daily records for the biosolids land application site which indicate (on a field map grid system) where, when, and how many (gallons) biosolids were land applied to the specific fields on the property to verify that annual loading limitations are not being exceeded;

18. Site records shall indicate the date, location and amount of biosolids applied;

19. The Department shall have the right to enter the Corvallis' WWTP's place of record keeping to review biosolids management operations and records; obtain copies of any records required under the terms of this authorization and the WWTP's biosolids (sludge) management plan; inspect any monitoring equipment required under this authorization; inspect any collection, transport, or land spreading vehicles; and obtain any photographic documentation or evidence deemed appropriate;

20. The City shall provide DEQ an annual summary of biosolids processing and land application activities;

21. The Department may impose, at a future date, additional restrictions or conditions deemed necessary to assure adequate biosolids management. Any variations from the City's (DEQ approved) biosolids management plan and this authorization letter must receive prior written approval from the Western Region - Salem Office;

22. All neighbors residing around the site must be notified prior to the proposed Biosolids application. Records of this notification must be kept on file with the land application site log. and,

23. This authorization is subject to revocation should health hazards, environmental degradation, or nuisance conditions develop as a result of inadequate solids treatment or site management. If operations are not conducted in accordance with terms specified under this authorization, the Department will initiate necessary remedial action.

If you have any questions regarding this authorization, please contact me at 378-8240, extension 236.

Sincerely,

Tom Fisher
Environmental Specialist
Western Region-Salem Office

TRF:sms
x:\tfisher\Corvallissa
cc: Doug Peters, Municipal Wastewater Section
Scott Faxton, OSU Dept of Animal Science, Withycombe Hall Corvallis, OR 97339

August 25, 1992

Oregon

RECEIVED

AUG 2 6 1992

PUBLIC WORKS

Dan Hanthorn
City of Corvallis
Wastewater Treatment Plant
P.O. Box 1083
Corvallis, OR 97339

DEPARTMENT OF ENVIRONMENTAL QUALITY

Willamette Valley Region

Re: WQ-SDS; City of Corvallis; Sludge Land Application Authorization; Gary Sander Site; T.L. 00500, Sec. 2, T13S., R,6W, W.M., Benton County; File No. 20151

Dear Mr. Hanthorn:

Regional staff evaluated the Sander site on August 24, 1992, to determine if the site would be suitable for the land application of anaerobically digested sludge from the Corvallis WWTP. At that time you provided us with a request for authorization to land apply solids to a 75 acre christmas tree site on the property (referenced above), which included information characterizing soils, topography, sludge, crop, crop management, annual available nitrogen loading (agronomic loading rate) and site life, and a signed agreement between the City and Gary and Audrey Sanders.

It is our understanding that the frequency of application to this site will be limited to one application every five (5) years.

Based upon our evaluation of the site, and the information submitted, I am pleased to advise you that the property described above is approved for summer use only (June 1-September 30) application of digested sewage sludge from the City provided:

1. Sludge processing and handling will comply with Oregon sludge rules and guidelines (OAR 340-50-005 to OAR 340-50-080) and all other applicable statutes, rules and federal regulations.

2. Sludge volatile solids shall be reduced by 38% or more via aerobic digestion prior to land spreading.

3. Based on the City's most recent sludge analysis, existing crop, topographic and soil conditions of the field, up to 100 pounds of available nitrogen [19,845 gallons or 8.5 dry tons of biosolids per acre per year] can be applied to areas of the field acceptable for receiving solids. Changes in sludge characteristics or cropping practices will necessitate appropriate adjustments in the application rate to maintain nitrogen loadings consistent with crop demands.

BARBARA ROBERTS
Governor

750 Front St. NE
Suite 120
Salem, OR 97310
(503) 378-8240

4. If other sources of nitrogen will be applied to the field, the sludge application rate must be reduced so that commercial, animal manure or green chop nitrogen, plus sludge nitrogen, does not exceed the agronomic loading rate of any field within the site.

5. Based upon sludge analytical data reported under the City's proposal, the solids land application site has a cumulative loading capacity of 235 dry tons per acre. Cadmium is the sludge constituent that limits the number of dry tons solids that may be land applied.

6. Sludge land spreading shall be via tanker with a spreader gate.

7. Immediately following land spreading, the sludge tanker operator shall wash down (at the application site) areas of equipment coated with sludge to prevent sludge from leaking onto public highways.

8. The City's truck operator shall clean up any spillage of sludge. Spillages which cannot be completely cleaned up must be covered with dry lime and posted where appropriate.

9. The Willamette Valley Region Office shall be notified within one hour of any spills of more than 50 gallons, or other threats to the environment that may occur. Failure to provide notification within one hour may be considered cause for taking enforcement action against the City. Spills that occur after normal working hours shall be reported to the Emergency Management Division (EMD) within one hour. The telephone number of EMD is 1-800-452-0311;

10. A 50 foot (minimum), (75 feet minimum where sludge cannon is used), setback shall be maintained from any county road or state highway and points of sludge application.

11. A minimum setback of 200 feet shall be maintained from all wells and other water sources.

12. Sludge shall be applied evenly and thinly in a manner that will prevent ponding and runoff.

13. Areas where sludge has been applied shall be clearly marked by flag pins or stakes which note the date when sludge was last applied or the permittee shall keep accurate daily records of the exact location of daily loads.

14. No food crops (for direct human consumption) whose harvested parts are grown below ground shall be planted on any sludge amended area for at least 60 months following sludge land spreading.

15. No food chain crops (for direct human consumption) whose harvested parts are grown above ground (e.g., tomatoes) shall be grown on any sludge amended area for at least 18 months following sludge land spreading.

16. A 30-day fallow period shall follow the application of sludge prior to grazing livestock on any field or feeding harvested crops from sludge areas to animals.

17. Public access to the site shall be restricted for at least 12 months after sludge land spreading has ceased.

18. In the event an odor problem is reported to the Department after sludge has been land spread at the Sander Site, immediate steps, such as, but not limited to, the addition of liming materials, must be taken to counteract that condition.

19. The City shall maintain daily records for the sludge land application site which indicate (on a field map grid system) where, when, and how much (gallons) sludge was land applied to a particular site.

20. Site records shall indicate the date, location and amount of sludge applied, segments of each field that received sludge, pounds of nutrients, cadmium, copper, lead, nickel, and zinc applied to each area receiving sludge, and the type of crop grown. These data shall be maintained at WWTP and be made available to the Department for review upon request.

21. The Department shall have the right to, (at reasonable times); enter City of Corvallis WWTP's place of record keeping to review sludge management operations and records; obtain copies of any records required under the terms of this authorization and the WWTP's sludge management plan; inspect any monitoring equipment required under this authorization; inspect any collection, transport, or land spreading vehicles and obtain any photographic documentation or evidence deemed appropriate.

22. The City shall provide DEQ a monthly summary of sludge processing and land application activities on DMR's required under the sewage treatment facility's NPDES permit.

23. The Department may impose any additional restrictions or conditions deemed necessary to assure adequate sludge management. Any variations from the City's DEQ approved sludge management plan, and this authorization letter, must receive prior written approval from the Willamette Valley Region Office.

24. This authorization is subject to revocation should health hazards, environmental degradation, or nuisance conditions develop as a result of inadequate sludge treatment or site management. This authorization is considered to be part of your approved sludge management plan. Therefore, if operations are not conducted in accordance with terms specified under this authorization, the Department will initiate an enforcement action which may lead to the assessment of a civil penalty.

If you have any questions regarding this authorization, please contact me at 378-8240.

Sincerely,

Joe Petrovich
Environmental Specialist
Willamette Valley Region

JAP:bkb

X:\JAP\Corvall.sds

Enc: (1) Beneficial Use Application
(2) Annual Solids Production Summary

cc: Gary & Audrey Sander, 24274 Neuman Road, Corvallis, OR 97339
Mark Ronayne, WQ w/att

Oregon
John A. Kitzhaber, M.D., Governor

Department of Environmental Quality
Western Region
Salem Office
750 Front St. NE
Suite 120
Salem, OR 97310
(503) 378-8240
(503) 378-3684 TTY

May 5, 1999

Dan Hanthorn
Wastewater Operations Supervisor
City of Corvallis
Post Office Box 1083
Corvallis, OR 97339

RE: WQ/SDS-City of Corvallis
NPDES No. 100984
File No. 20151
Biosolids Land Application Authorization
Corvallis Dedicated Disposal Site
T13S; R6W; Sec 10 -27.38 acres
Benton County

Dear Mr. Hanthon:

The Department of Environmental Quality (Department) reevaluated your request for authorization to land apply anaerobically digested biosolids from the Corvallis Wastewater Treatment Plant (WWTP) onto the Corvallis City property (referenced above). The site was initially approved by the Department on August 29, 1990, and biosolids have been beneficially used on the site since then to supply nutrients for grass hay. The reevaluation was based upon updated maps of individual fields and current Biosolids analysis.

Based upon the Department's evaluation of the information submitted; your request to use the property is approved for the application of anaerobically digested biosolids from the City of Corvallis from April 15th to October 15th of each year provided:

1. Solids processing and handling will comply with Oregon sludge rules and guidelines (OAR 340-50-005 to OAR 340-50-080) and all other applicable statutes, rules, and federal regulations;

2. Sludge volatile solids shall be reduced by 38% or more via anaerobic digestion prior to land spreading;

3. Prior to land application, the biosolids must meet one of the three pathogen reduction standards required under 40 CFR 503.32(b);

4. Based on the City's most recent biosolids analysis, existing crop, topographic, and soil conditions of the field, 100 pounds of available nitrogen per acre, per year 4.4 dry tons per acre, can be applied. Changes in Biosolids characteristics or cropping practices will necessitate appropriate adjustments in the annual application rate to maintain nitrogen loadings consistent with crop demands;

5. If other sources of nitrogen will be applied to the fields, the biosolids application rate must be reduced so that commercial fertilizer nitrogen, animal manure, or green manure nitrogen, plus biosolids nitrogen, do not exceed the agronomic loading rate of any field within the site;

6. The City's truck operator shall immediately clean up any spillage of biosolids. Spillage which cannot be completely cleaned up must be covered with dry lime and posted where appropriate;

7. The DEQ Salem Office ((503) 378-8240) shall be notified within one hour of any biosolids spills greater than 50 gallons, or other threats to the environment that may occur. Spills that occur after normal working hours shall be reported to the Emergency Management Division (EMD) within one hour. The telephone number of EMD is 1-800-452-0311;

8. A 50-foot minimum (300 feet minimum if biosolids gun application is used) setback shall be maintained between biosolids application areas and all highways, public roadways, and property lines;

9. A minimum setback of 200 feet shall be maintained between biosolids application areas and all wells and other water sources;

10. A minimum setback of 50 feet shall be maintained from seasonal drainage ways;

11. Biosolids shall be applied evenly and thinly in a manner that will prevent ponding and runoff. No Biosolids shall be applied, when within the prior 24 hours, 0.25 inches of rain or greater has fallen in the Corvallis area;

12. No food crops for direct human consumption, whose harvested parts are grown below ground shall be planted on any biosolids amended area for at least 38 months following biosolids land spreading;

13. No food chain crops for direct human consumption, whose harvested parts are grown above ground (e.g., tomatoes) shall be grown on any biosolids amended area for at least 14 months following biosolids land spreading;

14. A 30-day fallow period shall follow the application of biosolids prior to grazing livestock on any biosolids amended field or harvesting animal feed crops from biosolids amended areas;

15. Public access to field sites shall be restricted for at least 12 months after biosolids land spreading has ceased;

16. In the event an odor problem is reported at any land application site, immediate steps, such as, but not limited to, the addition of liming materials, must be taken to counteract that condition;

17. The City shall maintain daily records for the biosolids land application site which indicate (on a field map grid system) where, when, and how many (gallons) biosolids were land applied to the specific fields on the property to verify that annual loading limitations are not being exceeded;

18. Site records shall indicate the date, location and amount of biosolids applied;

19. The Department shall have the right to enter the Corvallis' WWTP's place of record keeping to review biosolids management operations and records, obtain copies of any records required under the terms of this authorization and the WWTP's biosolids (sludge) management plan, inspect any monitoring equipment required under this authorization; inspect any collection, transport, or land spreading vehicles, and obtain any photographic documentation or evidence deemed appropriate;

20. The City shall provide DEQ an annual summary of biosolids processing and land application activities;

21. The Department may impose, at a future date, additional restrictions or conditions deemed necessary to assure adequate biosolids management. Any variations from the City's (DEQ approved) biosolids management plan and this authorization letter must receive prior written approval from the Western Region - Salem Office; and,

22. This authorization is subject to revocation should health hazards, environmental degradation, or nuisance conditions develop as a result of inadequate solids treatment or site management. If operations are not conducted in accordance with terms specified under this authorization, the Department will initiate necessary remedial action.

If you have any questions regarding this authorization, please contact me at (503) 378-8240, extension 236.

Sincerely,

Tom Fisher
Environmental Specialist
Western Region-Salem Office

TRF:clp

x:\tfisher\CorvallissaBDS
cc: Doug Peters, Municipal Wastewater Section
Dept of Environmental Quality

Oregon
Theodore Kulongoski, Governor

Department of Environmental Quality
Western Region CORVALLIS Office
725 SE Main
CORVALLIS, OR 97470
(541) 440-3338
FAX (541) 440-3396

Received
JAN 10 2008
PUBLIC WORKS

August 31, 2007

Daniel Hanthorn, Wastewater Operation s Supervisor
Wastewater Regional Plant
City of Corvallis
1245 NE 3rd Street
P.O. Box 1083
Corvallis OR 97339

File Number: 20151
Phone ((541) 754-1757

RE: WQ-S-Benton County
Biosolid Land Application (approx. 100 acres)
Twp. 12S, R.4W, Sec. 23 W.M. Tax Lot
Property Owner: Jack and Penny Pimm
30854 Abraham Dr.
Halsey OR 97348

Dear Mr. Daniel Hanthorn:

Based on the current property owner's approval, this letter represents approval of your request to land apply biosolid on the above referenced property. This biosolid authorization is good until the property ownership changes, the farming management/practices change such that there is significant non-compliance with the biosolid rules (threat to public health or the environment), or the current property owner requests the city to discontinue the land application of biosolid on this site. This authorization is subject to revocation through a contested hearing process by the Department if significant noncompliance issues arise.

This authorization to apply biosolids the above referenced property is subject to criteria detailed in the Oregon Administrative Rules, Chapter 340, Division 50 and the following conditions:

Responsibility:
It is the responsibility of the City of Corvallis to insure the proper handling and application of all biosolids generated. Transportation of the biosolids to the application site shall be done in such a manner as to prevent leaking or spilling the biosolids onto the highways, streets, roads, waterways or other land surfaces not approved for biosolids application.

Site Description:
The site is located near Halsey, Oregon. The Pimm property is approximately 117 acres of grass fields, of which about 100 acres can be used for biosolid land application at this time. The biosolid application site is comprised mostly of Amity silty clay loam, Concord silty clay loam, and Woodburn silt loam soils. This site is only those portions of this parcel that is shaded on the enclosed map.

Based upon an evaluation of this property the Department is pleased to authorize the City to land apply stabilized biosolids subject to the conditions under your National Pollutant Discharge Elimination (NPDES) permit and the following stipulations:

1. This site is approved for summer application (June 1 through Oct. 31) of biosolids. During biosolid land application, care should be taken to avoid wet soil conditions, which may have occurred as a result of precipitation, especially in low and concave areas.

2. No liquid biosolids shall be applied on slopes greater than 12 percent, No cake biosolid shall be applied on slopes greater than 30 percent.

3. Biosolids shall be applied evenly at approved agronomic rate and in a manner to prevent ponding or runoff.

4. Biosolids shall not be applied closer than 10'for cake and 50' feet for liquid to any low access public property or road way, 50' to any drainage ditch, channel, pond, or waterway, or within 200 feet of a domestic water source or well.

5. For biosolid land application there is a minimum 48" setback from the ground surface and the permanent soil water table at these fields. Prior to land application of biosolids each year the permanent soil water table depths must be checked and a daily written recorded kept. Once the soil water table depths at these fields respectively have dropped below 48" from the ground surface the water table measurements can cease; should the groundwater table rise above 48" then daily water table depth shall be monitored for setback land application compliance.

6. Biosolid application rate shall not exceed approximately 140 lb. Plant Available Nitrogen (PAN) /acre/year. Changes in the biosolids characteristics or crops management may necessitate appropriate adjustments in the biosolid application rate to maintain proper plant available nitrogen for desired crop growth.

7. If other sources of nitrogen are used, the biosolids application rate must be reduced so that irrigation of reclaimed water or commercial nitrogen in combination with biosolids nitrogen does not exceed base agronomic loading rate of this site (140 lb. PAN/acre/years). Note Nitrogen loading rate may be adjusted to account for multiple grass crop harvests (see condition 6 above).

Site Use Limitations:

1. Controlled access to the biosolids site must be maintained for a period of 12 months following biosolids application.

2. Grazing of domestic animals should not be allowed on this site for 30 days following the last day biosolids were land applied on this site. There is a 90 day grazing restriction for lactating animals (dairy cattle).

DEQ-DC1

Accidental Spillage:
The permittee shall immediately clean up any spillage of biosolids and notify the DEQ Roseburg office at 440-3338 of any such occurrences. Call Oregon Emergency Response System (ORES) at 1-800- 452-0311 for any spills that make it to or could make it to waters of the state. Spillage which cannot be completely cleaned up shall be covered with hydrated lime (Calcium Hydroxide) or lime (calcium oxide). A 50-lb. bag of liming material shall remain available during transportation of the biosolids.

Monitoring:
1. Written daily land application records shall be kept on a field grid map or other easily readable system. Corvallis is responsible for tracking the land application of biosolids on daily basis (number of dry pounds nitrogen land applied per acre). Corvallis shall provide the DEQ with monthly summaries of biosolids land application activities along with a current biosolids analysis in Corvallis' annual report.

2. A copy of the current year's biosolid analysis should be carried with all biosolids that are to be land applied at this site.

3. A copy of this site authorization letter and a signed biosolid pathogen and vector attraction reduction certification statement should accompany all biosolids land applied at this site. The responsible parties who apply biosolids should review the site authorization letter prior to land applying biosolids to this site

4. If this site is used 2 out 3 consecutive years then a soil carry over nitrogen test is required for those fields in question prior to the end of the third year.

If you have any questions regarding this approval, please call me at (541) 440-3338, ext. 228.

Sincerely,

Paul Kennedy

Paul Kennedy
Natural Resource Specialist 3

Cc: Judy Johndohl, Portland-DEQ Biosolid Program
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DEQ-DC1

APPENDIX C

DEQ BIOSOLIDS MANAGEMENT RULES AND GUIDANCE

RECEIVED
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AMENDMENTS TO OREGON ADMINISTRATIVE RULES

OAR 340
DIVISION 50

LAND APPLICATION OF DOMESTIC WASTEWATER TREATMENT FACILITY BIOSOLIDS, BIOSOLIDS DERIVED PRODUCTS, AND DOMESTIC SEPTAGE

CONTENTS

OAR Number	Title	Page
	Oregon Administrative Rules	
340-50-005	Purpose	2
340-50-006	Policy	3
340-50-010	Definitions	3
340-50-015	Permit or License Required	5
340-50-020	Responsibility	7
340-50-025	Limitations and Restricted Uses	7
340-50-026	General Standards	8
340-50-030	Biosolids and Domestic Septage Land Application Site Selection and Approval	9
340-50-031	Biosolids and Domestic Septage Management Plans	10
340-50-032	Biosolids Facilities Plans and Specifications	12
340-50-035	Monitoring, Recordkeeping, and Reporting	13
	Best Management Practices for Site Selection and the Use and Application of Bulk Biosolids and Domestic Septage	
340-50-060	Purpose	15
340-50-065	Use Limitations	15
340-50-070	Criteria for Site Selection and Approval	16
340-50-080	Application of Biosolids and Domestic Septage	17

PURPOSE

340-50-005

(1) It is the purpose of these rules and best management practices to protect the environment and public health in Oregon by prescribing the methods, procedures and restrictions required for the safe handling and use of domestic wastewater treatment facility solids, biosolids, biosolids derived products, and domestic septage. These rules implement a program for biosolids and domestic septage management which satisfies or exceeds minimum federal regulations pertaining to land application.

(2) Industrial process water solids , agricultural wastes and sewerage wastewater are not included in these rules.

POLICY

340-50-006

The Environmental Quality Commission (EQC) encourages the land application of treated domestic wastewater biosolids, biosolids derived products, and domestic septage which are managed in a manner which protects the public health and maintains or improves environmental quality. These beneficial recyclable materials improve soil tilth, fertility, and stability and their use enhances the growth of agricultural, silvicultural, and horticultural crops.

DEFINITIONS

340-50-010

Unless otherwise indicated in this Division, definitions appearing under federal regulations 40 CFR §503.9, §503.11, and §503.31 shall apply. In addition, as used in these rules, unless differently required by context, the following definitions apply:

(1) "Agronomic Application Rate" means a rate of biosolids or domestic septage application which matches nutrient requirements for a specific crop on an annual basis.

(2) "Beneficial Use Site" means any Department approved site for application of a regulated amount of biosolids or domestic septage used for crop or livestock production, soil reclamation and stabilization, or soil improvement. Application rates and site management practices shall assure continued

(10) "Dewatered Solids" means domestic wastewater treatment facility solids or biosolids with a solids concentration between ten (10) and fifty (50) percent.

(11) "Domestic Septage" means liquid or solid material removed from a septic tank, cesspool, portable toilet, Type III marine sanitation device, holding tanks, or similar treatment works that receive only domestic wastewater. Domestic septage does not include liquid or solid material removed from a septic tank, cesspool, or similar treatment works that receive either commercial wastewater or industrial wastewater and does not include grease removed from a grease trap at a restaurant.

(12) "Domestic Wastewater" means the water-carried human wastes from residences, buildings, industrial establishments or other places, together with such groundwater infiltration and surface water as may be present that flow to wastewater treatment facilities.

(13) "Dried Solids" means domestic wastewater treatment facility solids or biosolids with a solids concentration of greater than fifty (50) percent accomplished by mechanical means or air drying .

(14) "Exceptional Quality Biosolids" means domestic wastewater treatment facility solids containing trace pollutant concentrations which are below federal alternative pollutant limits recognized under 40 CFR §503.13(b)(3) that have been treated by a Class A pathogen reduction process recognized under 40 CFR §503.32(a) and one of the vector attraction reduction procedures established under 40 CFR §503.33(b)(1) through (8). These solids are recognized as soil amendments which are acceptable for distribution and marketing to the public.

(15) "Federal Regulations" means Part 503 Standards for the Use or Disposal of Sewage Sludge, Subchapter O in Chapter I of Title 40 of the Code of Federal Regulations (§503).

(16) "Liquid Biosolids" means domestic wastewater treatment facility biosolids with a solids concentration of less than ten (10) percent.

(17) "Non-Digested Solids" means raw domestic primary, secondary, or advanced wastewater treatment facility solids, or solids that have accumulated in a digester which is either not operating efficiently or has not provided adequate time or temperature for digestion to occur; a lagoon where settled solids have not decomposed to the extent that they comply with 40 CFR §503.32 pathogen or 40 CFR §503.33 vector attraction reduction requirements; or a septic tank process whose function is confinement and/or separation of liquids and solids.

biosolids, or domestic septage, shall have in their possession a valid NPDES or WPCF permit issued pursuant to ORS 468B.050.

(3) Persons who land apply either Bulk Class B biosolids derived from sources outside Oregon or alkaline stabilized domestic septage shall first obtain a valid WPCF permit obtained pursuant to ORS 468B.050.

(4) Permit or license issuance or renewal applicants shall submit a biosolids or domestic septage management plan to the Department developed pursuant to OAR 340-50-031. No plan shall be approved by the Department unless, at a minimum, the plan demonstrates compliance with all requirements specified under OAR 340-50-026.

(5) Conditions in Department biosolids and septage management plan approvals and site authorization letters may be appealed to the Commission pursuant to ORS Chapter 183 and OAR Chapter 340, Division 11. Appeals shall be limited to topics pertaining to Department approved biosolids or septage management plan or site authorization letters.

(6) Any person operating a sewage disposal service business shall comply with all license conditions required under Department approved septage management plans.

(7) Where biosolids will be land applied during the term of a permit, permit applications submitted to the Department shall include a land application plan developed pursuant to 40 CFR §501.15 and OAR 340-50-031(7).

(8) Biosolids management and land application plans shall be subject to review and comment during the public participation process required prior to permit issuance or renewal under OAR 340-14-025, OAR 340-45-035, or OAR 340-71-162.

(9) All conditions contained in Department approved biosolids or septage management plan approval and site authorization letters are considered permit requirements.

(10) Except as otherwise required by this Division, the permitting provisions in 40 CFR Part 122; 40 CFR Part 501; OAR Chapter 340, Division 14; OAR Chapter 340, Division 45; and OAR Chapter 340, Division 71 are applicable.

GENERAL STANDARDS

340-50-026

(1) Unless the Department determines that the context requires otherwise, all general provisions of 40 CFR §503.1 through §503.9 and land application requirements in 40 CFR §503.10 through §503.18 are applicable for the purpose of this rule.

(2) To be considered acceptable for land application, biosolids or domestic septage shall meet:

(a) Pollutant concentration and cumulative pollutant loading limits required under 40 CFR §503.13. *(Note: not required for domestic septage.);*

(b) One of the pathogen reduction standards established under 40 CFR §503.32;

(c) One of the vector attraction reduction standards required under 40 CFR §503.33; and

(d) Management practices required under 40 CFR §503.14 and 40 CFR §503.32(b)(5).

(3) In addition to meeting pollutant concentration and loading limits required under 40 CFR §503.13, biosolids derived products must meet federal Class A pathogen reduction standards pursuant to 40 CFR §503.32(a) and one of the vector attraction reduction standards required under 40 CFR §503.33(b)(1) through (8).

(4) Biosolids imports must meet the following criteria:

(a) 40 CFR §503.13(b)(1), Table 1 pollutant ceiling concentration limits;

(b) One of the 40 CFR §503.32(b) Class B pathogen reduction standards;

(c) One of the 40 CFR §503.33(b)(1) through (8) vector attraction reduction standards;

(d) Minimum biosolids quality requirements of the County, State, or Regional government where they are produced; and

(e) They must be applied within beneficial use (agronomic) rates.

(2) Prior to the approval of any proposed site that may be sensitive with respect to residential housing, runoff potential or threat to groundwater, the Department shall ensure that an opportunity is provided for public comment and, if required as noted in (a) below, public hearing.

(a) If, during the public comment period, at least 10 people, or an organization representing at least 10 people, indicate concerns about the proposed action, then opportunity shall be provided for public hearing.

(b) The Department shall take final action on site authorization within 30 days of the closure of the public comment period, or 30 days of the closure of the hearing's record.

(3) Land application site authorization letters are considered an integral part of the biosolids or domestic septage management plan. Provisions specified by the Department in site authorization letters, in accordance with a Department approved biosolids or domestic septage management plan, shall be considered enforceable conditions under the permitted source's NPDES, WPCF, or Solid Waste Disposal permit.

BIOSOLIDS AND DOMESTIC SEPTAGE MANAGEMENT PLANS

340-50-031

(1) Any person who intends to land apply biosolids or domestic septage shall submit a solids management plan to the Department for review and approval. Unless otherwise authorized in writing by the Department, solids management plans shall be submitted a minimum of sixty (60) days before biosolids land application commences.

(2) Provisions established in Department approved biosolids or domestic septage management plans shall be considered NPDES, WPCF, or Solid Waste Disposal permit conditions.

(3) The biosolids or domestic septage management plan shall be kept current and remain on file with the permit or license. Plan modification requires written Department approval.

(4) Septage management plans must address the quantities of septages handled annually, types of septage processed, solid storage facilities, and solids land application or disposal facilities.

(l) A description of biosolids or domestic septage land application site monitoring, recordkeeping, and reporting procedures; and

(m) A description of remedial procedures that would be implemented in the event of a solids treatment process failure (e.g., digester breakdown or upset), solids spill at the wastewater treatment facility or solids generating source, or biosolids or domestic septage spill between the generating source and land application site.

(6) New Department authorized sites shall be made a part of the biosolids or domestic septage management plan.

(7) Biosolids Land Application plans shall, at a minimum, list:

(a) All known sites that will receive biosolids during the life of the permit;

(b) The geographic location of new sites which are not specifically listed at the time of application;

(c) Criteria which will be used in the selection of new sites; and

(d) Management practices which will be implemented where new sites are authorized by the Department.

BIOSOLIDS FACILITIES PLANS AND SPECIFICATIONS

340-50-032

(1) Plans and specifications for biosolids or domestic septage impoundments, reservoirs, tanks, or other containment structures proposed for storage may be required by the Department as part of an upgrade or as a permit or solids management plan approval condition.

(2) Detailed plans and specifications for biosolids or septage composting facilities shall, at a minimum, meet requirements included under OAR 340-96-020.

(3) Plans and preliminary engineering reports must be submitted to the Department pursuant to OAR 340-52-015. Written Department approval shall be obtained prior to the construction or use of biosolids or septage storage or composting facilities.

§503.13(b)(2), Table 2 and maintain records adequate to demonstrate that pollutant additions do not exceed Table 2 limits.

(5) Unless otherwise required by the Department, the quality and quantity of biosolids and land application actions shall be reported to the Department at least once annually. The Department may require more frequent reporting on biosolids production, treatment, characteristics, and land application activities. Monthly reporting is not required where exceptional quality biosolids are land applied.

(6) Annually, by February 19, each domestic NPDES, WPCF, and Solid Waste permitted source that has generated and land applied bulk biosolids or domestic septage, or prepared biosolids or biosolids derived products for distribution and marketing during the prior year, shall provide the Department with a comprehensive report that describes solids handling activities for the previous year. At a minimum, the report shall include, but is not limited to:

(a) Data, on each site that received solids, which is adequate to characterize solids quality and to demonstrate that solids were applied within agronomic loading rates and other required site management practices;

(b) Sources generating and applying biosolids that are required to track cumulative pollutant additions pursuant to 40 CFR §503.13(b)(2), Table 2, shall also be required to submit information on annual and cumulative pollutant additions;

(c) Information sufficient to demonstrate that solids met pathogen reduction requirements required under 40 CFR §503.32 and vector attraction reduction standards required under 40 CFR §503.33;

(d) Information describing any substantive modifications to solids handling or land application site management practices;

(e) A detailed description of any violation of 40 CFR §503 or OAR Chapter 340, Division 50 and remedial actions taken to prevent the recurrence of similar violations in the future.

(7) Annually, as a requirement of license renewal, sewage disposal service businesses shall submit information adequate to characterize solids handling activities which occurred during the previous licensing period. Information must include data describing the quantities and varieties of septages handled and locations where solids are used and disposed.

required where Class A biosolids derived products are land applied to sites used for the cultivation of fresh market fruits or vegetables.

(6) Grazing animals should not be allowed on pasture to forage nor should livestock feed crops be harvested where bulk Class B biosolids or domestic septage have been applied for a minimum of thirty (30) days after application.

(7) Exceptional Quality biosolids and biosolids derived products may be used on indoor and outdoor ornamental plants, shrubs, trees, home gardens and lawns, and high public contact areas without restricting public access.

CRITERIA FOR SITE SELECTION AND APPROVAL

340-50-070

(1) Normally, tillable agricultural land is suitable for the land application of biosolids and domestic septage.

(2) To be considered for biosolids or domestic septage land application, sites should meet all the following conditions:

(a) Sites should be on a stable geologic formation not subject to flooding or excessive runoff from adjacent land. If periodic flooding cannot be avoided, the period of application should be restricted and soil incorporation is recommended.

(b) At the time when liquid biosolids or domestic septage are applied, the minimum depth to permanent groundwater should be four (4) feet and the minimum depth to temporary groundwater should be one (1) foot. Sites approved for year-round application should be evaluated carefully to ensure that groundwater separation distances conform with these requirements.

(c) Topography of the site should be suitable to allow normal agricultural operations. Where needed, runoff and erosion control measures should be constructed. In general, liquid biosolids or domestic septage should not be surface applied on bare soils where the ground slope exceeds twelve (12) percent. Well vegetated sites with slopes up to thirty (30) percent may be used for dewatered or dried biosolids, or for liquid biosolids or domestic septage application with appropriate management to prevent runoff.

(2) The application of biosolids or domestic septage on agricultural land should be managed to utilize the fertilizer and organic matter value to the maximum extent possible. The recommended rate of biosolids or domestic septage application is normally based on the nitrogen requirement of the crop grown and will vary depending on the nitrogen content of the solids.

(3) Crop nitrogen requirements are used routinely to determine application rates for commercial fertilizer and these figures are readily available from state or county Extension Service offices. Applying biosolids and domestic septage within these limits helps ensure that solids nitrogen will be utilized for plant growth and that excess nitrogen which could leach into groundwater will not be of concern.

(4) Exceeding crop nitrogen requirements may occasionally be justified on a temporary basis in order to achieve rapid soil improvement or to prolong beneficial effects. Biosolids applications exceeding normal crop nitrogen requirements may be approved on a case-by-case basis for reclamation or soil improvement, if justification for a single high rate application is provided to the Department.

(a) Where a site has previously been amended with biosolids at soil improvement or reclamation rates, documentation of background soil nitrate-nitrogen (NO_3-N) shall be submitted for Department approval prior to the application of additional biosolids;

(b) Soil samples shall be collected and tested according to protocols published by Oregon State University and the American Society of Agronomy.

(5) Sites proposed for routine annual application at agronomic rates must periodically be assessed to determine the impact of nitrogen from biosolids and other sources.

(a) The criteria for requiring evaluation or performance monitoring [e.g., soil testing for carry-over nitrate-nitrogen (NO_3-N)] will be biosolids applications exceeding two (2) out of three (3) successive years at agronomic rates;

(b) Soil samples shall be collected and tested according to protocols published by Oregon State University and the American Society of Agronomy.

Oregon State Department of Environmental Quality
Water Quality Division

Policies for Biosolids Land Application

Biosolids Program Guidance DRAFT

INTRODUCTION

These biosolids policies help implement the Oregon state rule on land application of biosolids, Chapter 340 Division 50 Oregon Administrative Rule. Ten position papers covering these subjects were previously developed with the assistance of the DEQ Domestic Biosolids Technical Advisory Committee. The papers were presented in June 1994 as part of the state biosolids rule revision process, which included adoption by reference of the Environmental Protection Agency (EPA) technical standards in 40 CFR Part 503. The position papers were developed to clarify how to interpret both the new state and federal rules. They served as interim guidance, undergoing field testing for practicality while the new state regulation was implemented during the past two years.

These position papers were reviewed and are the basis for these Dept. of Environmental Quality (DEQ) Biosolids Policies. These guidance documents are being issued in draft form, to supplement the proposed draft biosolids land application rule amendments, and to solicit comments from all interested parties. Notice of these draft documents is being made to the public, the Domestic Biosolids Technical Advisory Committee, the Northwest Biosolids Management Association, and any interested Biosolids Committee or other members of the Oregon Association of Clean Water Agencies. Comments will be addressed and final drafts are then planned to be released together with the amended biosolids rule in 1998.

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6. Cumulative Pollutant Loading 15

Biosolids metals higher than Table 3 values must be tracked at sites 15

Sampling and testing protocols required for testing metals loading rates 15

7. Biosolids Trace Pollutant Forecasting 17

Statistical analyses of data can be used to forecast biosolids quality 17

EPA recommends 95% confidence interval in statistical analyses of biosolids 17

Example of biosolids statistical analysis 18

8. Public Notification 22

All permits are subject to public notification 22

Public notice process by publishing in a newspaper and mailing to list of interested people 22

Biosolids management plans may require separate public notification 22

Land application sites usually not subject to separate public notice 23

Sensitive sites may require public hearing 23

Public notice to adjacent land owners and occupants is recommended 23

9. Winter Biosolids Storage Policy 24

Adequate storage is needed for winter rainy periods and for plant upsets 24

DEQ determines what is adequate storage on a case-by-case basis 24

Guidelines assist in establishing adequate storage 24

Biosolids Storage Guidelines 25

Climatic Factors and Application Indices 29

10. Vector Attraction Reduction - Volatile Solids Reduction 33

VAR must be documented and certified 33

Minimum of seven samples required to determine VAR 33

Monitoring frequency based on quantities produced 33

Samples must be representative 33

Van Kleeck equation 34

Example Calculations for Agronomic Rate Determination

1. Biosolids N concentrations

TKN	Total Kjeldahl Nitrogen	55,000 mg/kg (5.5%)
NH4-N	Ammonium Nitrogen	15,000 mg/kg (1.5%)
NO3-N	Nitrate N	1,000 mg/kg (0.1%)

2. Available N calculations

% inorganic N = (%NH4-N) (.5) + (%NO3-N) = (5.5%)(.5) + (0.1%) = (2.75%) + (0.1%) = 0.85% (using 50% volatilization rate for surface application)

% organic N = (%TKN) - (%NH4-N) = 4.0%

lb mineralized N/dry ton biosolids = (%organic N/100) (mineralization rate/100) (2000) = (4/100)(30/100)(2000) = 24 lb mineralized N/dry ton biosolids (using 30% mineralization rate for anaerobic digestion)

lb inorganic N/dry ton biosolids = (% inorganic N/100)(2000) = (.85)(2000) = 17 lb inorganic N/dry ton biosolids

Total lb available N/dry ton biosolids = lb mineralized N+ lb inorganic N =24+17 = 41 lb available N/dry ton biosolids

3. Biosolids Application Rate

crop N requirement (lb N/acre) / (lb available N/dry ton biosolids) = dry ton biosolids/acre

Example crop: pasture grass/hay requiring 150 lb N/acre

150 lb N/acre / 41 lb available N/dry ton biosolids = 3.66 dry tons biosolids/acre

References are provided on pg. 4 of other publications which discuss methods to calculate agronomic application rates.

References

EPA Sept. 1995, Process Design Manual for Land Application of Sewage Sludge and Domestic Septage, EPA 625/R-95/001.

EPA 1993, Standards for the Use or Disposal of Sewage Sludge , 40 CFR Part 503.

EPA 1992, Technical Support Document for the Land Application of Sewage Sludge, EPA 822/R-93-001a.

D. Sullivan, D. Granatstein, C. Cogger, C. Henry, K. Dorsey, 1993, Biosolids Management Guidelines for Washington State, Washington State Dept. of Ecology Publication 93-80.

A.L. Page et al., 1982, Methods of Soil Analysis, Agronomy No. 9, Part 2.

H.L. Huddleston and M.P. Ronayne, 1995, Guide to Soil Suitability and Site Selection for Beneficial Use of Domestic Wastewater Biosolids, OSU, Manual 8.

D. Sullivan, Jan.1998 projected publishing date, Fertilizing with Biosolids, OSU, information brochure.

C. Henry, D. Sullivan, R. Rynk, K. Dorsey, C. Cogger, M. Van Ham, 1998 projected publishing date, Managing Nitrogen from Biosolids, unpublished manuscript now in process funded by the Washington State Dept. of Ecology and the Northwest Biosolids Management Association.

Oregon Administrative Rules, Chapter 340 Division 50, 1995.

Sampling to follow EPA protocol

Sample preparation and testing for fecal coliform and *Salmonella* sp. should follow protocols listed in EPA's ***Control of Pathogens and Vector Attraction in Sewage Sludge Manual*** (EPA/-92/013, December 1992), in **Appendices F** and **G**, on pages 103-106 and 107-115 respectively.

Composted biosolids must be monitored for either fecal coliform or *Salmonella* sp.. Where fecal coliform testing is used as a surrogate pathogen indicator, densities based on the geometric mean density must be less than 1,000 fecal coliform Most Probable Number (MPN) per gram of total solids (dry weight basis). If *Salmonella* sp. are used to determine pathogen regrowth, the arithmetic mean density of *Salmonella* sp. must be less than three (MPN) per four grams total solids (dry weight basis).

To assure fecal coliform densities adequately reflect actual microbial populations present at the time of sampling, composite samples must be collected over a period not to exceed one hour. Bacterial samples should be gathered during a period within two weeks immediately prior to the time when the compost would be marketed or distributed. Testing frequency would be quarterly thereafter, as long as product remains available for marketing and distribution.

The heterogeneous nature of some composted biosolids causes problems in sampling. For example, a composted product may be in a large pile comprised of wood chips and fine composted sludge. Most augers and sampling tools will be ineffective in getting a representative sample of the interior of such a pile. There may be no substitute for digging with a shovel to get to the desired location. When there is a large temperature gradient in the pile, it is important to include an analysis of the composted biosolids from the cooler section of the pile where the chance of regrowth of bacterial populations is greatest. It is also important that wood chips or other materials used as bulking agents are included in sampling the compost. In any case, samples from a large pile should be taken at various depths along its length.

Samples must be taken at least quarterly

To assess the impact of seasonal variation, testing of finished (cured) compost bacterial population shall be required at least once quarterly unless the compost will be processed, cured and distributed or marketed to the public in less than 90 days. Where composts would be made available to the public less than 90 days after the composting process was initiated, the Department will require more frequent testing. Testing frequency will be decided on a case-by-case basis.

3. Importing Biosolids to Oregon

Imported biosolids must meet quality standards in the state of origin

The land application of imported bulk biosolids is restricted to biosolids which meet:

1) § 503.13, Table 1 limits;

2) minimum EPA Class B pathogen quality in § 503.32(b);

3) minimum EPA vector attraction reduction requirements in § 503.33(b)(1)-(8), representing in-plant treatment options;

4) minimum biosolids quality requirements in the County, State, or Regional government where solids are produced; and

5) application rates which do not exceed agronomic rates for beneficial use.

Permit required for biosolids land application

Persons applying imported biosolids must be issued a Water Pollution Control Facility (WPCF) permit. Costs for permitting imported biosolids land application operations will be consistent with permit fees assessed for biosolids produced by Oregon sources. Until a new, separate permit fee category is established for this activity, permit fees will be based either on the design flow of the source (like they are for Oregon sources) or they will be the same as fees assessed minor industrial permits.

Landfill disposal of biosolids may be restricted in Oregon

Oregon Revised Statute (ORS) 459.055 (9) requires Oregon municipal solid waste landfills to manage solid waste from another state as it would be managed if disposed in that state.

> "Not withstanding any other provision of law relating to solid waste disposal, if the laws of any state prohibit or restrict the disposal of any kind of solid waste within the state of origin, such prohibition or restriction also shall apply to the disposal of such solid waste in Oregon."

Biosolids Annual Report Outline

I. Background/General

A. Facility
   1. Name, (District, Municipality, Other)
   2. Address
   3. Facility permit numbers
      a. NPDES or WPCF Permit #
   4. Facility contact and contact phone #

B. Reporting Period - previous calendar year

C. Narrative Descriptions of Processes
   1. Biosolids generation
   2. Biosolids preparation
   3. Biosolids application
   4. Annual goals, accomplishments, problems, (e.g., non-compliance with rules and regulations) changes

D. Total Annual Production (specify either dry tons or dry metric tons)
   1. Amount generated
   2. Amount prepared
   3. Amount applied

E. Total Application Acres Utilized

II. Pollutant Monitoring

A. Required Frequency

B. Parameter
   1. Sample measurement
   2. Annual average

Reporting Example:

Sample	#1	#2	#3	#4	Avg.	Exceedences
Hg (mg/kg)	5.0	6.4	7.2	5.6	6.0	0

III. Pathogen Reduction Monitoring and Recordkeeping

A. Description of Pathogen Reduction Process and § 503.32 Alternative

B. Operational Parameters Met During Sampling Interval (e.g., time/temperature)

C. Pathogen Monitoring Data
   1. Parameter measured
   2. Mean during required monitoring interval
   3. Annual mean
   4. Certification Statement

5. Biosolids Management Plans

Plans are required of biosolids producers and land appliers

A biosolids management plan provides a basic description of all processing and management aspects of biosolids production and land application. It needs to cover solids sources, treatment, handling, transport and land application. It also needs to include information on all land application sites and site selection criteria. All biosolids management plans should follow the format suggested in this policy to ensure all needed aspects are addressed and to ease DEQ staff review and approval.

Plans are the main administrative tool of the Oregon Biosolids Program.

Biosolids management plans serve to guide the specific program operations at facilities. Together with the facility permits and site authorizations, the plans provide assurance that biosolids activities are addressed in a comprehensive way, and problems with compliance are minimized. Plan approval conditions are enforceable extensions of the facility permit.

Separate Biosolids Policies cover certain elements of biosolids management plans and related policy and regulatory issues.

Plans must be kept current

Plans need to be kept current to address new circumstances in both the production of biosolids and the addition of new land application sites. Reviewing the plan during preparation of the annual report is a good idea to ensure it is accurate. Any change to the plan requires DEQ approval. As long as new application sites meet the site selection criteria in the approved biosolids management plan, no permit modification is needed.

On the following pages is a ***Biosolids Management Plan Outline.***

VII. Biosolids Utilization Program

A. Land Application
   1. Means of Transport
   2. Storage at Application Sites
   3. Application Methods
      a. Liquid Biosolids
         i. Truck application [splash plate/spray gun]
         ii. Sprinkler [spray gun]
      b. Dewatered Biosolids
         i. Truck application
         ii. Spreaders
      c. Dried Biosolids
         i. Spreaders

   4. Site Selection (Reference OAR 340-50-030)
      a. Landscape position
      b. Soils
      c. Sampling
         i. Soil Sampling
         ii. Well Sampling
      d. Notification of Property Owner
      e. Authorization

   5. Application Rates
   6. Testing/Sampling
   7. Recordkeeping/Monitoring

B. Composting
   1. Process Description
   2. Marketing/Distribution Plan
   3. Testing/Sampling
   4. Recordkeeping/Monitoring

C. Alkaline Stabilization
   1. Process Description
   2. Testing/Sampling
   3. Recordkeeping/Monitoring

D. Thermal Drying
   1. Process Description
   2. Marketing/Distribution Plan
   3. Testing/Sampling
   4. Recordkeeping/Monitoring

Once baseline levels of soil pollutants have been established, continuous tracking of biosolids pollutant additions is required. Pollutant additions would be computed by multiplying the number of dry tons of biosolids land applied after baseline soil concentrations have been determined, by the concentration of each pollutant, following the protocol indicated on page 9333, Volume 58, No. 32 of the Federal Register.

This approach measures the actual quantity of each pollutant present at a particular site. It takes into account not only the pollutants introduced via the land application of biosolids, but considers any naturally occurring pollutant levels as well as any pollutants already introduced to the site as a result of any of human activities. This is the most protective approach to determining pollutant loading since it accounts for actual inorganic element levels, including pollutants introduced via the land application of biosolids.

The Department may assist small agencies with limited staff in tracking biosolids dry matter and pollutant accumulations. Sources must report sufficient information to establish biosolids quantity and quality where solids have been applied to a particular site. This data will enable metal accumulations to be calculated.

To decrease expense associated with soils monitoring, a certain percentage (e.g., ≥20%) of an agency's actively used land application sites might be tested each year and baseline metal values at all sites might be evaluated in installments over a five year period. This would enable an agency to budget for soils monitoring, lessening the immediate economic burden. Under this approach, those sites which have received solids over the greatest period of time, sites with the highest annual loading rate, and sites receiving the largest quantity of solids should be assessed first. Prioritization of testing in subsequent years should be approached on a similar basis.

Further Considerations:

Agencies with biosolids where one or more pollutants exceed 67% of the National Sewage Sludge Survey value should determine the probable cause of elevated pollutant levels. This could involve a review of pretreatment program activities.

Under this evaluation method, a § 503.13 (b)(3) trace pollutant concentration confidence level assessment would be conducted at the beginning of each year. An agency's data sets for each of the 9 tested trace pollutants would be subjected to a statistical analysis to determine if any of the pollutants exhibited a reasonable potential to exceed the pollutant concentrations listed in § 503.13 (b)(1) & (3).The previous year's sampling data for an agency's land applied biosolids would be used to classify the current year's biosolids as either "clean" [§ 503.13 (b)(1)] or "very clean" [§ 503.13 (b)(3)]. Information resulting from the statistical analyses of data from the previous year would be used as a basis for an agency's planning (budgets, operations, maintenance, sampling, and analyses) for each upcoming new year.

The recommended confidence level interval for biosolids trace pollutant forecasting is 95%. Based on the analysis, if the confidence level is less than 95% that an agency can meet § 503.13(b)(1) or § 503.13(b)(3) (i.e., indicating that a reasonable potential exists for exceeding Table 1 or Table 3 concentrations), then the agency's biosolids will be reclassified to the appropriate biosolids category ("very clean", "clean" or "not acceptable" for land application under § 503, Subpart B) and managed in accordance with that quality rating the following year. The agency's biosolids will remain reclassified until it can be demonstrated by the agency that a confidence level for a higher or lower classification is warranted.

The recommended statistical approach is the method presented in EPA Manual, Test Methods for Evaluating Solid Waste, utilizing the **Student "*t*" statistic.**

Example of biosolids statistical analysis

An example calculation follows of biosolids copper concentration data from the Unified Sewerage Agency's Rock Creek Wastewater Treatment Facility.

The following Confidence Level evaluation was computed using the approved EPA method. The Student "t" is a continuous random variable whose probability distribution is completely specified by a single parameter, referred to as the number of degrees of freedom (df). For the Student t distribution, this parameter is determined by the sample size, n, and always df = n -1. The smallest value for df is 1 and therefore the Student "t" statistical method can be used for a sample size of 2 values. The determination of Confidence Level, using the Student "t" statistic, is calculated using the following equation:

OBJECTIVE: Define the 95 % confidence limit for 42 biosolids copper data points, using the **Student "*t*" statistic** approach.

Data is from the Unified Sewerage Agency's Rock Creek advanced wastewater treatment facility biosolids samples gathered between Jan. 3, 1990 and July 14, 1993.

Statistical Evaluation of Analytical Test Data (Biosolids Cake, Rock Creek WTP)		
Sample No.	Sample Value	$(x - \bar{x})^2$
1	875	19,600
2	745	100
3	816.2	6,593
4	858.6	15,277
5	830	9,025
6	679	3,136
7	1,350	378,225
8	916	32,761
9	702	1,089
10	763	784
11	654	6,561
12	747	12,544
13	832.9	9,584
14	975.8	57,985
15	906.9	29,550
16	831.2	9,254
17	687.3	2,275
18	553	33,124
19	598.4	18,660
20	729.4	31
21	788.4	2,852

8. Public Notification

All permits are subject to public notification

All wastewater treatment facilities seeking new or renewed permits are subject to public notice requirements. The permit application should include a biosolids management plan, which includes all approved land application sites and site selection criteria for any new sites.

Public notice process by publishing in a newspaper and mailing to list of interested people

Each DEQ main regional office has a permit coordinator who handles the process of public notice, including maintaining a list of interested people that receive notices. Any person can ask to be placed on any relevant mailing list. The permit coordinator arranges for publishing a statement describing the proposed activity in a newspaper of general circulation in the same area proposed for biosolids land spreading activities, including the following minimum information:

- name, address, and telephone number of a contact person from whom additional information can be obtained. For a proposed permit action this would include the regional DEQ office permit coordinator or permit writer responsible for the permit action for which public notice is being given.

- name and address of the permit applicant and, if different, the facility or activity regulated by the permit.

- a brief description of the activity described in the permit application.

- a brief description of the comment procedures and the time and place of any meeting or hearing that will be held (if any). If no meeting or hearing is scheduled, the statement includes the procedures for requesting a meeting or hearing, and any other procedures by which the public can participate in the final permit decision.

Biosolids management plans may require separate public notification

All permit applicants need to develop a biosolids management plan. If the plan is submitted with the permit application, it is available for public review with the permit. If the plan is developed later under a compliance schedule, it will be subject to a separate public notice at that time.

9. Winter Biosolids Storage Policy

Adequate storage is needed for winter rainy periods and for plant upsets

DEQ discourages the application of biosolids on wet or frozen ground when conditions would increase the potential for contaminants to runoff the site or enter groundwater. Adequate biosolids storage capacity is needed to avoid having to apply biosolids during unfavorable conditions. In the Willamette Valley, a minimum of three (3) to six (6) months storage capacity is recommended.

Besides biosolids storage, sewage treatment agencies need to have adequate capacity to store solids or liquids in the event of a unit process upset at the treatment works. Storage can also enable operators to optimize wastewater or solids treatment, while protecting water quality.

DEQ determines what is adequate storage on a case-by-case basis

Actual storage volume needs are based on a case-by-case evaluation of each facility. Factors used to evaluate the adequacy of storage capacity include:

- average climatic conditions
- biosolids total solids content (moisture content)
- availability of sufficient land application sites that are level to gently sloping, deep, and well drained to allow land application during the fall, winter, and spring periods when low precipitation and non-saturated soils allow some level of land application.

Agency solids storage needs are considered during the evaluation of facility plans for proposed treatment works upgrades. In a few cases, storage has been ordered as a condition of biosolids management plan approval. Agencies that treat domestic wastewater are required to either build or lease sufficient storage capacity based on the expected quantity of solids that will be generated during a 20 year design life.

Guidelines assist in establishing adequate storage

Guidelines are provided to define when it is appropriate to land apply biosolids and help define how much storage is necessary at a particular treatment works. Solids storage needs are, in part, influenced by the number and quality of sites available to agencies for land application.

4. Amount of Storage.

Storage needs will be evaluated according to the following priority:

a. POTWs that produce biosolids and are upgrading their treatment facilities and are proposing to or are currently land spreading solids will be required to provide adequate storage capacity for the 20 year design life as required under facilities planning. Long term biosolids storage facilities for liquid or cake sludge shall be designed to provide sufficient flexibility to store biosolids for a minimum of three to six months annually. The actual volume of storage needed will vary depending upon average climatic conditions, biosolids total solids concentration, and the availability of well drained land application sites. Storage capacity shall be designed to recognize the total solids density of the sludge product to be stored. Lesser storage capacity may be approved if the source can demonstrate to the Department's satisfaction, that it has sufficient inventory of well drained sites which can be managed to allow the application of solids during normal wet weather months. This assessment will be based on the evaluation of the local soil and climatic conditions, the availability of suitable winter biosolids land application sites, or other pertinent considerations. Storage capacity may not include storage in clarifiers, oxidation ditches or aeration basins, unless these are extra or unused for the digestive requirements.

b. For POTWs not upgrading their facilities, and who have 5 or more years of remaining design life, but need additional biosolids storage, an engineering report shall be submitted regarding the upgrading of the biosolids storage capacity. POTWs that are not upgrading their facilities, but are having biosolids management problems due to lack of storage, will be required to either build or lease sufficient storage capacity to accommodate minimum storage needs for the remainder of their design life. The volume of storage needed will vary depending upon average climatic conditions, biosolids total solids concentration, and the availability of well drained land application sites. The storage requirements will be based on the projected remaining design life left in the POTW and would be as follows:

Remaining Design Life of POTW	Design Year Storage Requirements
0 - 5 years	Take action to address immediate needs and start facility planning
6 - 10 years	10 years
10 - 15 years	15 years
15 - 20 years	20 years

c. Western Oregon facilities that are not in the process of upgrading and are not encountering solids storage problems are not required to expand their biosolids storage capacity.

Storage Plan elements include:

- **Empty Lagoons**

Begin each rainy season with empty storage lagoons (four lagoons with a combined capacity of 6,338,755 gallons).

- **Utilize Thickening**

Institute low feed rate biosolids thickening at the onset of rainy season to a dedicated lagoon. This maintains storage capacity in the digesters. It is used in conjunction with hauling operations to get ahead of production volumes. Centrate is recycled and lagoon volume is reduced.

Another option for storage is running all biosolids through dewatering systems prior to lagoon storage. This increases solids from 2% to 10%, increasing lagoon storage capacity from approximately 70 days to 350 days.

- **Limit Biosolids Hauling and Land Application During Inclement Weather**

When inclement weather is prolonged, limiting hauling operations, and when production is greater than thickening plus hauling operations, biosolids are transferred directly to an empty lagoon. Decanting begins upon biosolids separation and decant is recycled back to the main wastewater treatment process at a non-interfering rate. Approximately 30 % of the volume is decanted.

- **Apply Biosolids When Conditions Allow**

Storage period generally runs from November through June. Storage capacity is 6,338,755 gallons. During the 1992-1993 fall, winter and spring period, storage was required periodically during a 242 day interval. No application of biosolids occurred for 105 days during this interval (see Table). Storage capacity available at the end of the rainy season in 1993 was 2,845,503 gallons.

Through a combined use of storage lagoons, thickening and land application on well-drained sites, Salem has been able to successfully manage its biosolids in an area of the Willamette Valley which receives an annual average of 41.3 inches of precipitation. Solids generated by the Willow Lake treatment facility have been effectively handled using two to three months storage in combination with thickening, decanting, hauling, and land application when conditions allow. Six months storage capacity has not been necessary.

Critical climatic factors consider evapotranspiration (reflected by the season of the year) and occurrence of rainfall prior to the desired biosolids application date. These determinants are used to calculate the index since they are readily available from maps and soil surveys (slope and available water holding capacity) and on-site measurements (daily rainfall).

The past rainfall index (PRI) is calculated as follows:

For slopes ≤**25%**:

PRI = (1/S) + (1/WHC) + 0.3P4 + 0.5P3 + 0.8P2 + P1 + 1.2PO

For slopes ≥**25%**:

PRI = S + (1/WHC) + 0.3P4 + 0.5P3 + 0.8P2 + P1 + 1.2PO

Where:

S = Percent slope;

WHC = Water holding capacity of the A horizon in inches of water per inch soil (obtainable from SCS Soil Survey data for principal soil series at a particular slope phase);

Pn = Precipitation, in inches, for day 2, 3, etc., prior to application, and

PO = the predicted amount for the current day (If PO is not available, it is dropped from the equation).

The PRI is calculated each morning. If the index exceeds the specified critical value, a walk over inspection is made of the site, and in conjunction with the long-term weather forecasts (5-day), a determination is made to see if biosolids application may proceed as planned or whether solids should be stored or applied at an alternative location.

The source site manager is to make daily observations and keep a log of the conditions with the calculated PRI index so an experience base can be built to a set of critical PRI index values for future use.

Other Climatic Conditions

Other climatic conditions that must be considered in determining the suitability of a site include snow, frozen soil, wind speed and visibility.

Snow

In western Oregon, biosolids may be land applied following a light dusting of snow , but if there is greater than one-half inch of snow and the application of solids will not cause the snow to melt within a couple days from heat conducted from the solids and the underlying soil, biosolids should not be applied. If subfreezing temperatures are forecast for a few days following biosolids application and rainfall is not likely, the allowable period for melting can be extended by a few days.

Wind

If wind speed is such that biosolids would be blown into buffer strips, waterways, roads, trails or ditches or onto the application vehicle itself, application should stop immediately until the wind has stopped or slowed so that no significant drifting occurs.

Fog

If fog causes such poor visibility that the applicator vehicle operator is not able to clearly see buffer boundaries or areas where biosolids have been previously applied, application must cease immediately. The presence of fog also creates unsafe operating conditions. When fog limits clear visibility to less than 1.5 times the distance of the spread stream, application should be discontinued until visibility improves.

Lighting

Uniform biosolids application relies on the operator's ability to see that the surface of the application site receives a complete coat of biosolids and that over-application and/or spotty application does not occur. Loss of visibility from poor lighting during dawn and dusk will restrict operations and application must cease.

Samples must be representative

The primary purpose of Vector Attraction Reduction (VAR) criteria is to ensure that solids are stable enough to be land applied without resulting in odor or nuisance complaints. Samples of VS stabilized should be representative of the biosolids to be land applied. The sample point for VS raw is normally the point where solids exit a primary clarifier, gravity thickener, gravity belt thickener, Dissolved Air Flotation Thickener or other thickening process. Performance of individual digesters is not a primary concern and is monitored separately. If compliance can be met at the outlet of a digester, this is acceptable, but not specifically required for compliance purposes. Both inflow and discharge from individual digesters should be pooled on a representative basis, to show VAR compliance for the product going out to beneficial use. Digesters failing to meet normal operational criteria should not be included in the pooled sample and should not be blended with other digester solids for land application. Also, caution is advised for interpreting results from any samples mixed with polymer.

Arithmetic means can then be used to evaluate § 503 compliance. These can be computed on a "running average" basis, so that the facility operator always knows his status regarding VAR levels. To go a step further, a 95 percent confidence interval can be computed as outlined in the biosolids pollutant trace forecasting policy on page 15.

Van Kleeck equation

Ten vector attraction reduction options are available to wastewater treatment facilities who produce biosolids for bulk land application under § 503.33(b). Option 1 [§ 503.33(b)(1)] requires a demonstration that volatile solid concentrations are reduced by 38 percent prior to land application. There are a variety of equations that may be used in performing this calculation, but the most widely used equation is the Van Kleeck equation, which is usually shown as follows:

$$FVSR = \frac{VS_{(raw)} - VS_{(stabilized)}}{VS_{(raw)} - [VS_{(raw)} \times VS_{(stabilized)}]}$$

where:
FVSR = Fractional reduction of volatile solids (multiply by 100 for percent)
$VS_{(raw)}$ = Volatile fraction in raw sewage sludge
$VS_{(stabilized)}$ = Volatile fraction in stabilized sewage sludge

APPENDIX D

OSU EXTENSION PUBLICATION "FERTILIZING WITH BIOSOLIDS"

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Fertilizing with Biosolids

D. Sullivan

Biosolids are a by-product of municipal wastewater treatment. They contain organic matter and nutrients that are beneficial for soil, crop, and livestock productivity. Raw sewage solids must be processed to meet U.S. Environmental Protection Agency (USEPA) standards before they can be called biosolids.

This publication focuses on how biosolids can be used to supply nutrients to crops. Other aspects of biosolids utilization also are addressed briefly, including:

- Biosolids quality factors (trace element concentrations and pathogen levels)
- Applicable regulations
- Questions about biosolids safety and application practices

Biosolids as a Nutrient Source

Nutrient content

Biosolids usually are applied at rates designed to supply crops with adequate nitrogen. They also contain other nutrients that reduce fertilizer requirements. Table 1 shows the typical macronutrient content of biosolids produced by Pacific Northwest wastewater treatment facilities. The total nutrient content shown in Table 1 includes rapidly and slowly available nutrient forms.

- Inorganic nitrogen (ammonium) and sulfur (sulfide) compounds in biosolids are rapidly available. Nitrogen and sulfur-containing organic compounds are slowly converted via microbial activity to plant-available forms.
- Field research in the Northwest has shown that about 40 percent of the total sulfur (S) supplied by anaerobically digested biosolids becomes available during the first year of application. See "Biosolids Nitrogen" for details on nitrogen (N) availability for biosolids.
- Approximately half of the total phosphorus in biosolids is in inorganic forms that are rapidly available for plant uptake. This inorganic P is present as Al, Ca, and Fe phosphates and as Fe, Al, and Mn hydrous oxides. These forms of P are similar to those formed shortly after fertilizer P (Ca phosphate) application to soils.
- Calcium and magnesium concentrations in biosolids vary widely, depending on the hardness of the local water supply and whether alkaline materials are used in processing (see "Soil pH").
- All biosolids are low in potassium, which is water-soluble and is removed with treatment plant effluent (reclaimed water). For crop production, the amount of K needed from other sources (e.g., fertilizer, manure) is *not* affected by a biosolids application.
- Biosolids also provide essential micronutrients including copper, boron, molybdenum, zinc, and iron.

Table 1.—Biosolids organic matter and macronutrients (dry wt. basis).[a]

Nutrient	Usual range[b]	
	Low	High
	%	%
Organic matter[c]	45	70
Nitrogen	3	8
Sulfur	0.6	1.3
Phosphorus	1.5	3.0
Calcium	1.0	4.0
Magnesium	0.4	0.8
Potassium	0.1	0.6

[a]Nutrient concentrations are expressed on a total elemental basis; not all elemental content is in a plant-available form. Concentrations for organic matter, nitrogen, phosphorus, and potassium are based on a 1995 Oregon Association of Clean Water Agencies survey. Sulfur, calcium, and magnesium concentrations are based on biosolids analyses from seven Pacific Northwest wastewater treatment facilities.

[b]The usual nutrient concentration range includes approximately 80 percent of the biosolids analyses reported. Biosolids composts and alkaline-stabilized biosolids were not included in the calculation of the usual nutrient concentration range.

[c]Organic matter as determined by loss on ignition (volatile solids).

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Balance between biosolids nutrient inputs and crop nutrient removal

The nutrients supplied by biosolids are not present in the same ratio as they are found in plant tissue. Therefore, biosolids supplied at an agronomic rate for nitrogen supply an excess of some nutrients (e.g., phosphorus) and not enough of other nutrients (e.g., potassium).

Table 2 gives a general example of the relationship between biosolids nutrient inputs and crop removal of nutrients. The general nutrient balance illustrated in Table 2 (positive or negative for a particular nutrient) is valid for most crops. Nutrients accumulate in the soil to a greater extent if nutrients are recycled onsite via crop residues (e.g., grass managed for seed production) or if harvest is by grazing.

Table 2.—Example: Annual change in soil nutrient status with a 4-dry-ton-per-acre biosolids application. Tall fescue grown for grass hay or silage.

Total nutrient applied	Biosolids nutrient input[a] lb/acre	Grass nutrient removal[b] lb/acre	Net nutrient input to soil[c] lb/acre
Nitrogen	400	220	130
Phosphorus	160	32	128
Potassium	24	240	-216
Sulfur	80	40	40
Zinc	6.8	0.4	6.4
Copper	4.7	0.1	4.6

[a]Estimated total nutrient input with application of 4 dry tons biosolids per acre per year. Loss of 50 lb N/acre as ammonia is estimated at application.

[b]Based on field research in western Oregon and Washington (tall fescue harvested four to five times each year, yielding about 1 ton of dry forage per harvest).

[c]Calculated as total nutrient inputs minus grass harvest removal. Leaching losses of all nutrients are assumed to be insignificant.

Fertilizer replacement value of biosolids

The table below estimates the fertilizer replacement value of biosolids nitrogen, phosphorus, potassium, and sulfur for the first year after application. This estimate is based on typical biosolids analyses and estimates of nutrient availability from university field trials. The first-year nitrogen value of a 4-dry-ton/acre biosolids application supplying 120 lb available N is approximately $36 per acre.

The fertilizer replacement value for nutrients besides N depends on existing soil test values and the cropping system. Typically, a biosolids application to meet crop N requirements eliminates the need for annual P and S applications. Biosolids have the greatest fertilizer replacement value when they are applied to soils testing low in available P.

The fertilizer replacement value calculated below does not include the potential benefits to soil quality from biosolids (Table 9). Soil quality benefits are difficult to express in simple economic terms.

Fertilizer replacement value per dry ton of anaerobically digested biosolids. First year after application.

Nutrient	Expressed as[a]	Total nutrient % dry wt.	Available nutrient[b] % dry wt.	Nutrient cost[c] $ per lb	Value per dry ton
Nitrogen	N	4.0	1.5	0.30	$9.00
Phosphorus	P	2.0	1.0	0.55	$11.00
Potassium	K	0.3	0.3	0.35	$2.10
Sulfur	S	1.0	0.4	0.06	$0.48
Total					$22.58

[a] To convert P and K to units used in fertilizer marketing: multiply P by 2.29 to get P_2O_5; multiply K by 1.20 to get K_2O.

[b]Estimated plant-available nutrient released in the first year after biosolids application.

[c]Approximate nutrient cost is based on Willamette Valley bulk fertilizer prices, March 1996. The actual cost of a pound of fertilizer N, P, or K varies depending on nutrient form and analysis (e.g., costs differ for anhydrous ammonia and calcium nitrate as a source of N), transportation and application charges, market conditions, and the quantity purchased.

Soil testing at biosolids application sites

Soil testing can be used to monitor the change in soil fertility resulting from biosolids application. For elements such as P and Zn that are relatively immobile, take soil samples to a depth of 12 inches, since test interpretations usually are based on this depth (e.g., Marx et al., 1996). For mobile nutrients such as N and S, a sampling depth greater than 12 inches often is recommended.

When biosolids are applied to perennial grasses or other cropping systems without tillage, smaller sampling increments near the soil surface (e.g., 0-3 inches and 4-12 inches) give better information on changes in nutrient availability, soil pH, and salinity (conductivity).

To get the best information from soil testing, we recommend that growers work with a qualified agronomist to develop a sampling plan to fit their cropping system and biosolids application techniques.

Table 3 shows an example of how soil testing can help growers realize the greatest benefit from biosolids application. In the example, soil test phosphorus and zinc increased as the amount of biosolids applied increased. After a 3-dry-ton application, soil test P and Zn were well above the minimum soil test values needed for dryland wheat (10 mg/kg bicarbonate-extractable P, and 0.5 mg/kg DTPA-extractable Zn). Instead of repeat applications at the same site, more benefit would be obtained by correcting P and Zn deficiencies at other sites with low soil test values.

Table 3.—Example: biosolids effects on soil test phosphorus and zinc (dryland winter wheat).[a]

Biosolids applied	Soil test (0–12 inch)[b]		Soil test interpretation[c]	
	P	Zn		
dry ton/acre	mg/kg[d]	mg/kg	P	Zn
0	13	0.7	medium	possible deficiency
3	18	1.2	medium	sufficient
6	24	1.9	high	sufficient
9	32	2.6	high	sufficient

Adapted from: Sullivan et al., 1995.

[a]Example data represent the average of four locations in central Washington cropped to dryland winter wheat. Soil was sampled approximately 18 months after biosolids application. Average biosolids P concentration was 2.65 percent, and average biosolids Zn concentration was 870 mg/kg (dry weight basis).

[b]Soil test phosphorus via bicarbonate extraction, zinc via DTPA extraction. Soil test P and Zn are indexes of soil nutrient status; they cannot be used to calculate lb of available nutrient per acre.

[c]Soil test interpretation from OSU Extension publication, EC 1478, *Soil Test Interpretation Guide* (Marx et al., 1996).

[d]The units mg/kg are equivalent to parts per million (ppm) on a dry weight basis.

Applying more P to soils that already have high soil test P values is a potential environmental risk. Runoff and eroded sediment can move P to lakes or streams, where it can reduce water quality. At repeat application sites with a high potential for offsite P movement, soil testing can help prevent excessive accumulations of P.

Biosolids Nitrogen

Nitrogen forms

Wastewater treatment facilities routinely monitor biosolids ammonium, nitrate, and organic N concentrations. The organic N concentration is determined by subtracting the amount of ammonium-N from total N (TKN) concentration. Nitrate-N is present at insignificant concentrations in most biosolids, so it is ignored in the remainder of this discussion.

The proportions of ammonium and organic N in biosolids vary with the stabilization process (e.g., digestion, composting) used at the wastewater treatment facility (Table 4). Anaerobically digested liquid biosolids often contain more ammonium than organic N. Other biosolids contain most of the N in organic form.

Ammonium-N is available to plants immediately after application. Organic N provides slow-release N for crops after application.

Table 4.—Effect of biosolids processing on N forms in biosolids.[a]

Processing	Moisture content	Ammonium N	Organic N
		Percent of total N content[b]	
Anaerobic digestion	liquid	60	40
Aerobic digestion	liquid	25	75
Aerobic/anaerobic digestion and storage in lagoon > 6 months	liquid	25	75
Anaerobic digestion and dewatering	semi-solid	25	75
Drying bed	solid	25	75
Heat-drying	solid	10	90
Composting	solid	10	90

[a]Sources: 1989–94 Oregon wastewater facility data compiled by Oregon DEQ, and 1995 Oregon facility data compiled by the Oregon Association of Clean Water Agencies.

[b]Example using table data: For biosolids containing 4 percent total N (dry weight basis), anaerobically digested liquid biosolids would contain approximately 2.4 percent ammonium N and 1.6 percent organic N, while drying bed biosolids would contain approximately 3 percent organic N and 1 percent ammonium N.

Plant-available Nitrogen

Two processes affect the amount of plant-available N (PAN) provided by a biosolids application: ammonium-N retention and organic N mineralization. These processes are described briefly below.

Ammonium N retained after application

When biosolids are surface applied (not incorporated into soil at application), a portion of the biosolids ammonium-N is lost as ammonia gas. Ammonia loss is very rapid during the first hours after application. Estimates of ammonium retention after surface application range from 0 to 70 percent of the ammonium-N applied. An estimate of 50 percent ammonium-N retention commonly is used for planning purposes. Immediate tillage after biosolids application increases ammonium retained, but is not practical or desirable for many cropping situations.

Organic N mineralization

Organic N in biosolids is converted to plant-available forms (ammonium and nitrate) by soil microorganisms after field application (a process known as mineralization). The biosolids treatment process affects the organic N mineralization rate after land application (Table 5). Freshly digested biosolids usually contain more mineralizable N than do biosolids produced with more intensive stabilization processes (composting, long-term lagoon storage).

Table 5.—Effect of biosolids processing technologies on first-year organic N mineralization rate.

Processing	First-year organic N mineralization rate
Anaerobic digestion	20–40
Aerobic digestion	30–45
Aerobic/anaerobic digestion and storage in lagoon > 6 months	15–30
Anaerobic digestion and dewatering	20–40
Drying bed	15–30
Heat-drying	20–40
Composting	0–20

The organic N mineralization rate also is affected by soil temperature and moisture conditions. Mineralization is most rapid in warm, moist soils.

Mineralization proceeds most rapidly immediately after biosolids application, provided soil temperature and moisture are favorable. Usually, more than half of first-year N mineralization occurs within the first 3 to 6 weeks after biosolids application.

The quantity estimated for the first year (Table 5) represents cumulative mineralization in 2 to 5 months in spring, summer, or fall. In winter, when soil temperatures are less than 40°F, mineralization is very slow.

Soil and plant tissue testing can be used to determine actual N availability after a biosolids application.

Repeated Application to the Same Site

The organic N not mineralized during the first year after application is mineralized slowly in succeeding years. Thus, when biosolids are applied repeatedly to the same site, the annual application rate needed to meet crop needs is reduced.

University field research with perennial grasses demonstrates this slow-release N contribution of biosolids (Table 6). To compare fertilizer N to biosolids total N, we use the term "relative effectiveness." At 50 percent relative effectiveness, an annual application of 2 lb total N as biosolids is equivalent to application of 1 lb of fertilizer N.

In the first year of application, biosolids total N was 27–40 percent as effective as fertilizer N. In the second and third years of continuous application, biosolids total N was 41–59 percent as effective as fertilizer N.

Thus, for the tall fescue example (Table 6), biosolids application rates needed to equal an annual fertilizer N application rate of 200 lb N/acre were 540–740 lb biosolids total N/acre in year 1, and 340-490 lb biosolids total N/acre in years 2 and 3.

Table 6.—Relative effectiveness of biosolids total N for tall fescue production compared to fertilizer N[a]

Biosolids Type	Location	Years of continuous biosolids application: 1	2	3
		Relative effectiveness, percent of fertilizer N[b]		
Dewatered	Puyallup, WA	37	59	49
Heat-dried	Puyallup, WA	40	49	41
Liquid	Aurora, OR	27	41	44
Average		35	50	45

[a] Biosolids were applied at the same total N application rate in years 1, 2, and 3 of each study. Data sources: Puyallup, WA, 1993-95 (Cogger et al., unpublished data); Aurora, OR, 1976-78 (Kiemnec et al., 1987).

[b] Relative effectiveness in increasing harvested grass N uptake. Relative effectiveness of 50 percent means that a current-season application of 2 lb biosolids total N was equivalent to 1 lb fertilizer N (ammonium nitrate, 34-0-0).

Converting nitrogen availability estimates to application rates

The estimated plant-available N (PAN) provided by a biosolids application usually is reported on a dry weight basis (e.g., 1.5 percent PAN or 30 lb PAN per dry ton). For field application, solids never are dry. Factors that can be used to convert an estimate of biosolids PAN to a field N application rate are shown at the right.

To use these conversion factors, you must know the percent total solids content of the biosolids. For example, biosolids containing 98 percent liquid contain 2 percent total solids.

Converting from dry weight PAN estimate to application units.

To convert from % PAN, dry weight basis to:	**Multiply % PAN, dry weight basis by:**
lb PAN/1,000 gal	percent total solids x 0.834
lb PAN/acre-inch	percent total solids x 22.65
lb PAN/wet ton	percent total solids x 0.2

Note: This table converts from biosolids PAN as a percentage of dry weight. If, on the other hand, the municipality lists PAN in terms of lb/dry ton, divide by 20 to get % PAN on a dry weight basis. For example, biosolids with estimated PAN of 30 lb N/dry ton contain 1.5 percent PAN on a dry weight basis.

Examples: Conversion from %PAN, dry weight basis to application rates

A city produces biosolids with an estimated PAN of 1.7 percent (dry weight basis). The biosolids are applied as a slurry containing 4 percent total solids. You estimate that 100 lb PAN/acre should be applied to meet the crop yield goal. You can calculate the amount of liquid biosolids to apply as follows:

Example A: Conversion to gallons per acre

Step 1. Convert % PAN on a dry weight basis to application units using the table above:

Biosolids PAN (lb/1,000 gal) = % PAN, dry weight basis x % total solids x 0.834
= 1.7 x 4 x 0.834
= 5.67 lb PAN/1,000 gal

Step 2. Calculate the biosolids application rate:

Biosolids application rate (thousand gal/a) = lb N/acre ÷ lb PAN/1,000 gal
= 100 ÷ 5.67
= 17.6 (17,600 gal/acre)

Example B: Conversion to acre-inches

Step 1. Convert % PAN on a dry weight basis to application units using the table above:

Biosolids PAN (lb/acre-inch) = % PAN, dry weight basis x $ total solids x 22.65
= 1.7 x 4 x 22.65
= 154 lb PAN/acre-inch

Step 2. Calculate the biosolids application rate:

Biosolids application rate (acre-inches) = lb N/acre ÷ lb PAN/acre-inch
= 100 ÷ 154
= 0.65 acre-inches

Soil pH

Maintaining an appropriate soil pH is essential in maintaining the productivity of cropping systems. Biosolids application can increase or decrease soil pH (Table 7), depending primarily on whether alkaline materials are used in biosolids processing.

Table 7.—Biosolids effects on soil pH.

Type of biosolids	Biosolids input	Effect on soil pH	Soil process
Alkaline-stabilized	Ca oxides and hydroxides	increase	Converts soil acidity to water and carbon dioxide.
All	soluble salts	temporary decrease	Cations displace acidity from soil clays and organic matter, bringing it into soil solution.
All	organic N and S	decrease	Oxidation in soil to nitrate and sulfate produces acidity.
All	exchangeable Ca and Mg	increase	Supplies non-acidifying cations.

Alkaline-stabilized biosolids

Some biosolids are stabilized with alkaline materials (calcium oxide or calcium hydroxide) to reduce odors and meet USEPA pathogen reduction requirements. In a typical alkaline stabilization process, the biosolids/alkaline material mixture reaches a pH of 12 or greater for at least 2 hours before application. At this high pH, ammonia loss is rapid. Ammonia loss produces acidity, which neutralizes some of the alkaline material before application.

After application, the residual alkaline material rapidly neutralizes soil acidity, thereby increasing soil pH (first process in Table 7). Thus, alkaline-stabilized biosolids act as a replacement for agricultural limestone. Post-application increases in soil pH, exchangeable soil calcium, and soil SMP buffer pH can be used to determine the liming value of a biosolids/alkaline materials application.

Biosolids stabilized without alkaline materials

When biosolids that do not contain added alkaline materials are applied, the change in soil pH is the net result of the last three factors listed in Table 7. Shortly after application, a drop in soil pH (acidification) may be observed, usually 0.2 to 0.5 pH units. Acidity is generated by the addition of soluble salts and by the oxidation of organic N and S compounds.

This pH drop usually is short lived and is balanced by the calcium and magnesium added by biosolids. Because several factors are involved, soil testing is recommended to monitor soil pH at biosolids application sites.

The effect of biosolids vs. fertilizer N application on soil pH is shown in Table 8 for one Willamette Valley application site. The soil pH values listed were measured after 7 years of biosolids or fertilizer N (ammonium sulfate) application. The liquid anaerobically digested biosolids applied in the study did not contain added alkaline materials. The soil pH after 7 years was the same for biosolids application and for an annual application of 100 lb N per acre as ammonium sulfate.

Table 8.—Surface soil pH (0–12 in) after 7 years of biosolids or ammonium sulfate application to a Willamette Valley silt loam soil.[a]

N Source	Total N applied lb/acre/yr	Soil pH
None	0	5.4
Biosolids[b]	140–210	5.2
	290–420	5.1
Ammonium sulfate	100	5.1
	200	4.9
	300	4.7

Source: Kiemnec et al., 1990.

[a] Location: Aurora, OR. Soil pH (0–8 inches) at the start of the study was 5.7. The soil had a cation exchange capacity of 15 meq/100g soil.

[b] Average for anaerobically digested liquid biosolids from three Oregon wastewater treatment facilities: Portland, Salem, and Rock Creek (Unified Sewerage Agency, Washington County). All biosolids sources in the study performed similarly.

Soil Quality

Biosolids application also affects soil chemical, physical, and biological properties. Together, these properties determine soil quality. Biosolids supply both nutrients and organic matter for soil quality improvement.

The organic matter provided by biosolids does not by itself increase soil organic matter a great deal. However, when combined with other soil management practices that favor accumulation of organic matter (e.g., reduced tillage, use of cover crops), biosolids application helps improve soil quality. The changes in soil quality accompanying biosolids application generally improve the capacity of crops to utilize nutrients. Potential benefits of biosolids application on soil quality are listed in Table 9.

Table 9.—Ways that soil quality can improve with biosolids application.

Physical	Greater water-holding capacity
	Improved tilth
	Reduced soil erosion
Chemical and biological	Increased cation exchange capacity
	Plant-available nitrogen and sulfur slowly released from organic forms
	Micronutrient deficiencies corrected
	Increased earthworm and microbial activity

To utilize the slow-release N provided by biosolids, cover crops may be needed to capture the organic N mineralized from biosolids in late summer and early fall. Otherwise, this slow-release N is likely to be lost via nitrate leaching.

What about heavy metals?

Heavy metals were a concern when land application of biosolids began 25 years ago. However, since then, source control activities have been very effective in decreasing metal inputs to wastewater treatment facilities:

- Wastewater treatment facilities now issue permits to major industrial wastewater sources and routinely monitor metal concentrations in the industrial wastewater. To comply with permit requirements, industries have adopted manufacturing processes that keep metals out of the sewer system.
- Adjustment of city water supplies to a higher pH has reduced pipe corrosion and the concentrations of zinc, copper, and lead in household wastewater.
- The use of PVC pipe instead of metal pipe also has reduced metal concentrations.

The table below shows the impact of source control activities on biosolids metal concentrations. Metals in Portland biosolids have been reduced by a factor of 2 to 10 times over the past 15 years. Today, biosolids metal concentrations are similar for cities with industrial inputs (Portland biosolids concentrations, below) and smaller treatment facilities with few industrial inputs (Average Oregon biosolids concentrations, page 9).

Decrease in metal content of City of Portland biosolids between 1981 and 1996.

	City of Portland biosolids[a]	
	1981	1996
Metal	mg/kg	mg/kg
Cadmium	40	5
Copper	1,000	481
Lead	900	181
Nickel	190	39
Zinc	2,200	834

Crop Yield and Quality

Biosolids fertilization often increases crop yield potential over that attainable with other nutrient sources that do not supply organic matter or micronutrients. The most dramatic improvements in site productivity are observed when biosolids are applied to severely eroded cropland.

As with any nitrogen source, over-application of biosolids can reduce crop yield and quality. Research with dryland wheat grown in the 10- to 14-inch annual precipitation zone (east of the Cascades) has demonstrated that biosolids applications that supply too much N can reduce grain yield and quality. Supplying too much N is less of a crop quality problem for pasture and grass forage production, because the N is utilized for increased protein content. Biosolids rates for woody vines and tree crops should be monitored carefully, as N supplied in excess late in the growing season can result in increased winter injury.

Forage Quality

Macronutrients

When similar amounts of available N are supplied with biosolids or fertilizer N, the biosolids-fertilized forage usually has higher phosphorus and sulfur content (Table 10). Unlike most dairy and poultry manures, biosolids are low in potassium and do not increase forage potassium, except when the typical forage is nitrogen-deficient.

Table 10.—Tall fescue macronutrient concentrations after 4 years of biosolids or fertilizer N application.[a]

	Nutrient concentration	
	Grass grown with biosolids	Grass grown with fertilizer N
Nutrient	mg/kg	mg/kg
Calcium	0.62	0.48
Magnesium	0.26	0.24
Potassium	3.2	3.4
Nitrogen	2.3	2.1
Phosphorus	0.39	0.30
Sulfur	0.49	0.19

[a] Tall fescue grown for grass silage, Puyallup, WA on a Puyallup sandy loam soil. Biosolids were applied at approximately 500 lb total N/acre/yr; fertilizer N (34-0-0) was applied at 300 lb N/acre/yr.

Site soil test data (0–12 in): Soil pH: 5.5. Exchangeable Ca: 4 meq/100g; Mg: 1 meq/100g; K: 0.25 meq/100g. Phosphorus (Bray P1): 200+ mg/kg. For grass grown with fertilizer N, applications of K, Mg, and S fertilizers were applied to prevent nutrient deficiencies.

Micronutrients

Biosolids provide plant-available copper, zinc, and manganese. Biosolids application can help correct deficiencies of these elements in animal diets. Research with forage grasses (Table 11) has shown that typical micronutrient concentrations in biosolids-fertilized grasses are far below levels considered detrimental to animal performance.

Table 11.—Tall fescue micronutrient concentrations after 4 years of biosolids or fertilizer N application.[a]

Nutrient	Grass grown with biosolids mg/kg	Grass grown with fertilizer N mg/kg	Required concentration in total diet[b] mg/kg	Recommended maximum concentration in total diet[b] mg/kg
Copper	9	7	4–11	25-115
Manganese	155	168	20–50	1,000
Zinc	51	26	20–40	500–750

[a] Tall fescue was grown at the same site and with the same management described in Table 10.

[b] Required concentration in total diet and recommended maximum concentration in total diet for beef cattle, sheep, and horses. Source: Cuhna, 1987.

Site Approvals for Biosolids Application

Is site approval needed?

Whether you need a permit or site approval for biosolids application to cropland depends on:

- Trace element concentrations in the biosolids relative to USEPA standards
- Whether the biosolids meet USEPA Class A or Class B pathogen reduction requirements

Site approvals always are required for Class B biosolids application. More information on trace element and pathogen reduction requirements is given on page 9, "Biosolids Quality."

Who obtains site approval?

When needed, site approvals for biosolids application generally are obtained by wastewater treatment plant representatives. Some private companies, working under contract with the wastewater treatment facility, also may assist in obtaining site approval.

Biosolids quality

Biosolids quality is based on:

- Trace element concentrations in biosolids
- The completeness of the pathogen reduction process used by the wastewater treatment facility

These biosolids quality factors are described briefly here. Additional details can be found in USEPA, 1993, and Sullivan et al., 1993.

Together, these two quality factors determine whether site approvals by a regulatory agency are required for biosolids application.

Trace elements

Current trace element concentrations usually are well below concentration standards set by USEPA for biosolids (see table at right). The current concentrations of trace metals in biosolids allow most wastewater treatment facilities to produce a beneficial product suitable for land application.

The USEPA standards, adopted in 1993, are based on more than 20 years of field research and a comprehensive risk assessment. The lower concentration standard for trace elements (Exceptional Quality Limit in the table) is for biosolids products distributed to the public. The higher concentration standard (Ceiling Limit in the table) applies to all land-applied biosolids. If one element in the product exceeds the Ceiling Limit, then the product cannot be called biosolids and cannot be land applied.

Pathogen reduction

Pathogens are organisms, such as bacteria, that can cause disease in humans. Before application to cropland, biosolids must be processed to meet either USEPA Class A or Class B pathogen reduction standards.

Class A biosolids are essentially pathogen-free and usually are sold or distributed in urban areas for landscaping or turf fertilization. Because of the greater flexibility in management afforded by Class A treatment, more facilities are moving toward Class A biosolids production.

Class B biosolids have been processed to significantly reduce, but not eliminate, pathogens. Biosolids of Class B quality usually are delivered to farmers for land application. The most common processing methods for Class B biosolids are aerobic or anaerobic digestion. Some treatment facilities add alkaline materials such as calcium hydroxide to kill pathogens and meet Class B standards.

After land application, pathogens in Class B biosolids are killed by exposure to sunlight, drying conditions, unfavorable pH, and other environmental factors. Site management practices required at Class B biosolids application sites protect public health.

Trace elements in Oregon biosolids in comparison with USEPA concentration limits for land application.

Trace element	Oregon biosolids average[a] mg/kg	USEPA Exceptional Quality limit[b] mg/kg	USEPA Ceiling Limit[b] mg/kg
Arsenic	6	41	75
Cadmium	5	39	85
Copper	589	1,500	4,300
Lead	94	300	840
Mercury	2	17	57
Molybdenum	8	75[c]	75
Nickel	32	420	420
Selenium	9	100	100
Zinc	852	2,800	7,500

[a]Average concentration for 37 Oregon biosolids products as reported by the Oregon Association of Clean Water Agencies (1995 data).

[b]Source: USEPA guidelines for pollutant concentrations in biosolids (40 CFR Part 503). The Exceptional Quality Limit applies to biosolids suitable for distribution without site approvals; the Ceiling Limit is the maximum for land application of biosolids. To be called "Exceptional Quality Biosolids" by USEPA, biosolids also must be Class A for pathogen reduction.

[c]The USEPA Exceptional Quality Limit for molybdenum currently is under review.

What agency is responsible for site approvals?

Oregon:	Department of Environmental Quality
Washington:	County health departments and Washington State Department of Ecology
Idaho:	USEPA

Site management practices

Table 12 shows the site management practices required for biosolids meeting USEPA's Class A or Class B pathogen reduction requirements. The additional site management practices for Class B biosolids application sites protect public health.

Class A and B refer only to the completeness of pathogen reduction processes. Biosolids also must meet standards for trace element concentration (see "Biosolids Quality," page 9).

For most crops, the waiting period is 30 days after Class B biosolids application before grazing or crop harvest. For crops whose edible portion may directly contact the biosolids (e.g., root crops such as potatoes or carrots, and low-growing vegetable and fruit crops such as lettuce and strawberries), the waiting period between biosolids application and harvest is 14 to 38 months. Thus, Class B biosolids application is impractical for these crops.

Table 12.—Site management requirements for Class A and B biosolids.[a]

Biosolids class	Limit public access?[b]	Waiting period before grazing or crop harvest?	Regulatory agency authorization required?	Buffer areas required?
A	no	no	no	no
B	365 days	30 days	yes	yes

[a] This table gives general explanatory information on site management required by USEPA. For Class B biosolids, the required site management practices are included in site approvals issued by the regulatory agency.

[b] Privately owned farmland meets the USEPA's definition of "limited public access."

Questions about biosolids

What limits the rate of biosolids that can be applied?

Application rates are based on supplying N for the crop, without excessive nitrate leaching.

Is there a limit to the amount of biosolids that can be applied to a site?

If biosolids meet Exceptional Quality (EQ) limits (see page 9), there is no limit to the cumulative amount of biosolids that can be applied. (Annual application rates are based on crop N needs.) If one element exceeds the Exceptional Quality Limit, but is below the Ceiling Limit, then site life is based on the cumulative amount of that trace element applied to the site.

For most biosolids that don't meet EQ standards, the calculated site life is more than 50 years. The maximum cumulative amount of a trace element that can be applied during a site's life includes a large margin of safety. After the cumulative limit is reached, the site still will be productive for growing crops.

Are permitted levels for heavy metals and other trace elements likely to change in the future?

The federal biosolids rule adopted by USEPA in 1993 was based on more than 20 years of university research. This research included numerous worst-case studies, where sewage sludges containing high trace element concentrations (2 to 20 times greater than current concentrations) were applied at very high rates (10 to 50 times the agronomic rate). The current biosolids regulations are based on this extensive research base, and are unlikely to change in the near future.

The National Academy of Sciences reviewed USEPA's trace element standards in 1995 and published an extensive report concluding: "While no disposal or reuse option can guarantee complete safety, the use of these materials (biosolids) in the production of crops for human consumption, when practiced in accordance with existing federal guidelines and regulations, presents negligible risk to the consumer, to crop production, and to the environment."

How big of a problem are odors generated by biosolids application?

Odors produced by biosolids application generally are less than for animal by-products such as dairy or chicken manure. The same processes that kill pathogens in biosolids (e.g., digestion) also reduce odor after application. Tillage after biosolids application reduces odor.

What concerns might my neighbors have about a biosolids application?

Neighbors may object to potential nuisances associated with biosolids application such as truck traffic, dust, and odor. They also have concerns about protection of local water resources such as wells, streams, and lakes. They also may have health concerns. Biosolids transportation and application practices can be modified to address neighbor concerns. Ask the agency that will provide the biosolids to visit with neighbors before the biosolids application.

How much time is required for biosolids application relative to conventional inorganic fertilizer application?

Biosolids application is much slower than conventional fertilizer application, because of the large quantities of material involved. Several weeks may be required to complete application to large fields. Because of the slower pace of biosolids application, perennial grass pastures are the most common application sites. Annual crops can be fertilized with biosolids, but careful scheduling is required.

If biosolids are a valuable resource, why are they provided at no charge to the landowner?

Not all biosolids are free. Some municipalities charge a transportation fee, an application fee, or a fee equal to the nitrogen value of the biosolids. In other cases, biosolids application is free, which compensates for landowners' time and effort. Often, field operations must be scheduled to accommodate biosolids application, and management practices specified by the regulatory agency (see Table 12) must be followed after application.

For More Information

OSU Extension publications

Huddleston, J.H. and M.P. Ronayne. Revised 1995. *Guide to soil suitability and site selection for beneficial use of domestic wastewater biosolids*, Manual 8 (Oregon State University, Corvallis, OR). $3.25

Marx, E.S., J. Hart and R.G. Stevens. 1996, reprinted 1997. *Soil test interpretation guide*, EC 1478 (Oregon State University, Corvallis, OR). $1.50

Sullivan, D.M. 1996. *Biosolids recycling in Oregon*, EC 1471 (Oregon State University, Corvallis, OR). No charge.

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WSU Cooperative Extension publication

Cogger, C.G. and D.M. Sullivan. 1998. *Worksheet for calculating biosolids application rates in agriculture* (Washington State University, Pullman, WA, in review).

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Government publications

Sullivan, D.M., D.M. Granatstein, C.G. Cogger, C.L. Henry, and K.P. Dorsey. 1993. *Biosolids management guidelines for Washington State.* Washington State Department of Ecology Publication 93-80 (Olympia, WA).

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USEPA. 1993. *A plain English guide to the EPA biosolids rule*, EPA Publ. 832-R-93-003 (Office of Wastewater Enforcement and Compliance).

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Cogger, C.G., D.M. Sullivan, A.I. Bary, and J.A. Kropf. 1998. Matching plant-available nitrogen from biosolids with dryland wheat needs. J. Prod. Ag. 11:41–47.

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Research article available from Dan Sullivan

Contact the Department of Crop and Soil Science, Oregon State University.

Sullivan, D.M., A.I. Bary, J.A. Kropf, and D.M. Granatstein. 1995. Biosolids fertilization for dryland Pacific Northwest wheat production. p. 2.1-2.16. In: *Proc. USEPA Biosolids Symposium, Effects of land application of biosolids in arid and semi-arid environments* (Colorado State University, Fort Collins, CO, 16–19 May 1995).

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APPENDIX E

GLOSSARY

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Aerobic Digestion: The degradation of organic matter brought about through the action of microorganisms in the presence of elemental oxygen for purposes of stabilization, volume reduction, and pathogen reduction. The process is carried out in a tank or other vessel called a digester.

Agricultural Land: Land on which food, feed, or fiber crops are grown. This includes range land and/or land used as pasture.

Agronomic Rate: The whole biosolids application rate designed to provide the amount of nitrogen needed for the crop or vegetation grown on the land and designed to minimize the amount of nitrogen in the biosolids that passes below the root zone of the crop or vegetation grown on the land, to the ground water.

Ammonia Nitrogen: The quantity of elemental nitrogen present in the form of ammonia (NH_3) normally present at fairly high levels in biosolids. This form of nitrogen is readily converted to nitrate nitrogen and, therefore is considered to be immediately available to plants, or may be lost to the atmosphere (volatized if surface applied.)

Anaerobic Digestion: The degradation of organic matter brought about through the action of microorganisms in the absence of elemental oxygen for purposes of stabilization, volume reduction and pathogen reduction. The process is carried out in a tank or other vessel called a digester.

Annual Pollutant Loading Rate (APLR): The maximum amount of a pollutant that can be applied to a unit area of land during a 365-day period. This term describes pollutant limits for biosolids that is given away or sold in a bag or other container for application to the land.

Annual Whole Biosolids Application Rate: The maximum amount of biosolids on a dry weight basis that can be applied to a land application site during a 365-day (1-year) period.

Beneficial Use: Taking advantage of the nutrient content and soil conditioning properties of an organic waste product to supply some or all of the fertilizer needs of an agronomic crop or stabilizing vegetative cover.

Biosolids: A primarily organic solid product produced by wastewater treatment processes that can be beneficially used. They are the treated solid, semi-solid, or liquid residues generated during the treatment of domestic sewage in a wastewater treatment facility (such facilities may also receive an industrial component). Biosolids meet all of the Part 503 pollutant concentration, pathogen reduction, and vector attraction reduction criteria. The residues include, but are not limited to, scum or solids removed in primary, secondary, or advanced wastewater treatment processes, but do not include grit and screenings generated during the preliminary treatment of domestic sewage.

Buffer: An area of land that designates a zone of separation between possible conflicting land uses.

Bulk Biosolids: Biosolids not sold or given away in a bag or other container (with a load capacity of one metric ton or less) for application to the land.

California Environmental Quality Act (CEQA): The California legislative enactment which requires public agencies to assess the environmental impacts of public and private projects before governmental authorization of the project is granted.

CFR: Code of Federal Regulations. A document of the United States Government presenting federal agency rules, regulations, and guidelines.

Class A Biosolids: Biosolids meeting Part 503 Class A pathogen reduction requirements.

Class B Biosolids: Biosolids meeting Part 503 Class B pathogen reduction requirements.

Cumulative Pollutant Loading Rate (CPLR): The maximum amount of an inorganic pollutant that can be applied to an area of land. This term applies to bulk biosolids that are land applied.

CWEA: California Water Environment Association. A professional organization of administrators, engineers, ant treatment plan operators engaged in water quality enhancement and water pollution abatement in California.

Dewatered Biosolids: The biosolids remaining after removal of water by draining, centrifugation, filtering or pressing.

Domestic Sewage: Waste and wastewater from humans or household operations that are discharged to or otherwise enter a treatment works.

Exceptional Quality Biosolids: Biosolids that meet the most stringent limits for pollutant concentrations, pathogen reduction and vector attraction reduction.

Feces: Excrement from the gastrointestinal tract, consisting of residue from food digestion and bacterial action.

Hazardous Waste: Any waste that is potentially damaging to environmental health because of toxicity, ignitability, corrosivity, chemical reactivity, or other action.

Humus: The dark-colored carboniferous residue in the soil resulting from the decomposition of vegetable tissues of plants. The relatively resistant fraction of soil organic matter that forms during biological decomposition of organic residues. Humus usually constitutes the major fraction of soil organic matter.

Inorganic Nitrogen: Nitrogen that is in the ammonium (NH_4^+) or nitrate (NO_3^-) form, either in biosolids or in the soil.

Land Application: The placement of biosolids on land at a predetermined rate to support vegetative growth.

Land Application Field: A discrete area of land within a land application site that is the smallest unit of land for which monitoring, record keeping, and reporting requirements apply.

Land Application Site: An area of land covered by a single permit on which biosolids are land applied to condition the soil or to fertilize crops or vegetation grown in the soil.

Leachate: Liquid that has percolated through permeable material and has extracted soluble dissolved or suspended materials from it.

Leaching: Removal of soluble minerals, nutrients, organic chemicals, and pesticides from the soil by water passing through the soil.

Legume: A crop that forms a specific association with soil bacteria that are capable of fixing nitrogen, that is, transforming nitrogen gas to organically combined nitrogen. Common legumes include alfalfa, clovers, peas, and soybeans. Nitrogen fixation can provide most of the nitrogen of the crop, and it can provide large amounts of residual nitrogen for succeeding crops.

Macronutrients: An element required in large amounts for the growth and development of plants such as nitrogen, phosphorus, and potassium.

Micronutrients: Essential plant foods usually required in minute quantities such as zinc, iron, copper.

Microorganisms: Microscopic organisms, either plant or animal, invisible or barely visible to the naked eye. Examples are algae, bacteria, fungi, protozoa, and viruses.

Mineralization: Biochemical conversion of nitrogen in the organic matter of soils and biosolids to inorganic nitrogen. Mineralization produces nitrogen in the ammonium (NH_4^+) form, which is then converted to the nitrate (NO_3^-) form by the nitrification process.

Moisture Content: The quantity of water present in soil, biosolids, or residual solids, usually expressed in percentage of wet weight.

Monitoring: Routine observation, sampling, and testing of designated locations and/or parameters to determine efficiency of treatment or compliance with standards or requirements.

Nitrate Nitrogen: A form of nitrogen which is oxidized from ammonia nitrogen. It is a soluble form which is immediately available to plants, does not volatilize, and may move downward through the soil to contaminate groundwater. Nitrate is usually present in small amounts in biosolids.

Nitrogen: An essential nutrient often present in biosolids as ammonia, nitrate, nitrite, and organic nitrogen.

Nutrient: Any substance that is assimilated by organisms and promotes growth. The term generally refers to nitrogen, phosphorus, and potassium in agriculture, but can also apply to other essential and trace elements.

Organic Nitrogen: Nitrogen that is combined in the molecular structure of organic compounds. Most of the organic nitrogen in soils occurs as proteins and amino acids or amine groups.

PAN: Plant Available Nitrogen. Nitrogen primarily in the form of nitrate or ammonia ions readily available to plants.

Pathogens: Disease-causing organisms such as bacteria, protozoa, viruses, and parasites.

Percolation: The movement of flow of water through the interstices or the pores of a coil or other porous medium.

Permeability: The rate that water moves through the soil. Permeability depends on the amount, size, and interconnectedness of soil pores. These in turn are related to soil texture, soil structure, and soil density.

PFRP: Process to further reduce pathogens as defined in Appendix B to Part 503-Pathogen Treatment Processes.

pH: A number that indicates the relative acidity or alkalinity of a material. A pH of 7.0 indicates a neutral material. Lower numbers indicate acidic materials. Higher numbers indicate alkaline materials.

Ponding: The process of creating a still, shallow body of water, smaller than a lake, usually very small accumulations of water on the soil surface.

Water Table: The top of a zone of saturated soil. Water tables in soils are classified as perched, apparent, or artesian. A perched water table refers to a zone of saturation that is underlaid by unsaturated soil. Perched water tables are associated with restrictive layers. An apparent water table refers to a thick zone of saturated soil in which there is no evidence of restrictive layers. An artesian water table refers to water under pressure that is trapped beneath an impermeable layer. The water table rises when the impermeable layer is breached.