**PFAS in Municipal Solid Waste Landfills: Sources, Leachate Composition, Chemical Transformations, and Future Challenges**

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**Abstract**

Observed trends in municipal solid waste landfills reveal a distinct disparity between composition of PFAS entering in waste, mostly as diPAP and FTOH, and leaching as FTCA and PFCA. These patterns are elucidated by compiling PFAS compositions in paper, textiles, and carpet, and known precursor transformations responsible for FTCA and PFCA predominance in leachate. Future research must assess the role of precursor PFAS like diPAP and FTOH in landfill mass balances, particularly in leachate, along with the potential for release of semi-volatile PFAS to the atmosphere. Closing these knowledge gaps is critical as landfills will increasingly serve as PFAS sources.

**Key Words**: emerging contaminants, leachate transformation, PFAS, pollution, repository, waste disposal

**Introduction**

Municipal solid waste landfills (MSWLs) are a critical component of the engineered environment, serving as a terminal repository for commercial, consumer, and industrial solid wastes. Modern MSWL safely isolate these wastes from the biosphere with clay and geotextile liners to prevent groundwater contamination and through collection of leachate and gas [1](Figure 1). Leachate is generated within MSWLs from the infiltration of precipitation and waste degradation, and typically requires treatment at wastewater treatment plants (WWTPs). Solid residuals from WWTPs are often disposed of in MSWLs, resulting in continuous cycling and interaction between aqueous and solid waste streams.

Per- and polyfluorinated alkyl substances (PFAS), are a recalcitrant family of 10,000+ compounds containing at least one fluorinated carbon, are known to cause adverse health impacts at very low concentrations [2,3], and are widely used in manufacturing, commercial, and consumer products. MSWLs have received PFAS-laden waste for decades, and subsequently, leachate is contaminated with PFAS [4–7]. WWTPs cannot effectively treat or remove PFAS, and with the recent promulgation of environmental regulations restricting PFAS in effluent, reducing PFAS in MSWL leachate has become a priority.

This review focuses on the current state of knowledge of PFAS in MSWLs by emphasizing the disparity between the composition of PFAS in waste disposed of and leachate generated. This difference is highlighted by compiling measured PFAS concentrations in municipal solid waste, current knowledge of the chemical transformations of PFAS in landfills, and observed trends in leachate composition. For brevity, the broad class of PFAS are categorized into five groups containing multiple compounds: fluorotelemer phosphate diester (diPAP), fluorotelemer alcohol (FOTH), fluorotelemer carboxylic acid (FTCA), perfluorinated carboxylic acid (PFCA), and perfluorinated sulfonic acid (PFSA). Lastly, future challenges for PFAS characterization and management in leachate are identified.

**Sources of PFAS in Municipal Solid Waste**

Characterizing PFAS composition of municipal solid waste (MSW) and identifying compounds of concern is essential for determining the role of landfill function within the PFAS cycle [4,7]. Detected PFAS in MSW leachate originates from a variety of consumer and industrial products. Some MSW contain PFAS already in the aqueous, mobile phase (e.g., liquid wastes), whereas PFAS bound to solid wastes are released over time by decomposition and leaching processes (Figure 1). MSW characterized into subgroups by the USEPA 2018 consists of 24.14% food, 18.46% plastics, 11.78% paper and paperboard, 9.53% metals, 8.32% wood, 7.73% textiles, 7.21% yard trimmings, 5.17% glass, 3.42% rubber and leather, 2.24% miscellaneous inorganic waste, and 2.01% other.

Several studies have characterized PFAS in consumer products, which are direct exposure pathways [8–11]. An organic solvent, usually methanol, is used to extract leachable PFAS, which is then quantified using liquid chromatography or gas chromatography in tandem with mass spectrometry [7]. While limited by target analytes, these studies provide a basis for discerning PFAS in consumer products in MSW. We reviewed studies characterizing PFAS in food/compostable MSW fraction [12–14], plastics [15–17], paper and paperboard [11,17–25], textiles and carpets [9–11,16,19,23,24,26–31], rubber and leather [23,30], electronics [16,23], metal non-stick wear [23,24], and waterproofing/polishing/cleaning liquids [11,23,24,30,32,33]. Data reported as a mass per unit area (μg/m2) were excluded from our compilation to facilitate direct comparisons of PFAS concentration.

Table1 lists the concentrations of five PFAS groups (diPAP, FTOH, FTCA, PFCA, PFSA) compiled from multiple studies in paper (e.g., food contact material, packaging, paper fibers), textiles (e.g., clothing, furniture), and carpets. Due to the extreme variability in PFAS concentrations measured for individual products, average values were computed from median PFAS concentrations reported in each study. For example, 9 different food contact papers ranged from non-detect to 12700 ng/g 6:2 FTOH in a single study[11]. MSW subcategories (paper, textiles, carpet) summarized in this review have the most available data on PFAS composition, together comprise 19.5% of total incoming landfilled mass, and are readily biodegradable, which is expected to be an important mechanism of PFAS release [34,35]. Figure 2 reports the combined average PFAS concentrations in paper, textiles, and carpet by weight percent of MSW in ng/g.

Wide variation between extraction methods, analytes, number of samples, and products made it difficult to characterize the PFAS composition and care should be taken when making interpretations; however, this compilation yields several clear trends. Most notably, polyfluorinated compounds, especially FTOH, comprise the majority of PFAS extracted. Given their grease resistant design, popcorn bags and paper plates appear to have the highest concentrations of FTOH for the paper products [18]. Studies that analyzed for diPAP (FTOH precursor) revealed its prevalence in paper products [17,20–22] with concentrations as high as 600-9000 µg/g measured in Danish food packing [25].

FTOH is also abundant in impregnation, waterproofing, and cleaning agents [11,23,33], floor waxes and sealants [11,33], and leather [23,30]. A broad analysis of 115 individual products indicates widespread usage of FTOH in municipal products[30], likely an important contributor to PFAS mass in MSWLs. Other products such as ski wax [30], non-stick cookware [23,24], and cosmetics [8] have high concentrations of PFAS, but little is known about their overall mass in landfills. Studies analyzing electronics [16,23,36] and plastics [15–17,36] found mainly PFCA and PFSA, but small sample size, limited analyte suites, and selective removal from waste streams make it difficult to extrapolate these findings to the landfill scale.

Recently, leaching tests on MSW screenings in water collected from a Florida landfill produced 15 to 88 ng/g PFCA, 0.22 to 0.79 ng/g PFSA, non-detect to 0.12 ng/g FTS, and more interestingly, non-detect FTCA over a range of liquid to solid ratios [37]. While PFCA appear to account for most of the PFAS mass in leachate, many key polyfluorinated compounds compiled in Table 1, such as diPAP and FTOH, were excluded. Given the abundance and relatively high concentrations of FTOH measured across consumer products, total leachable PFAS concentrations in the incoming MSW are likely much greater. Studies comparing PFAS leached from consumer products to total oxidizable precursors [19,31] reveal concentration increases by factors of 2-3, while total organic fluorine methods [19,31,38] yielded orders of magnitude increases. Nonetheless, assuming an average of 50 ng/g PFAS weight combined with EPA estimates of 141 million tons of MSW landfilled per year, a conservative estimate of 6,400 kilograms of PFAS enters MSWLs each year in the U.S. Actual PFAS loading is likely orders of magnitude higher considering the limited analyte suite. Moreover, this estimate is an order of magnitude higher than estimated total mass of PFAS leaving landfills in leachate, indicating that MSWLs will become increasingly important sources [4,5,7].

**Trends in MSWL Leachate**

Leachate collected from 18 landfills across the United States indicated of the 70 PFAS analyzed most of the detected mass leaving landfills was 5:3 fluorotelomer carboxylic acid (FTCA), followed by short chain PFCAs with minimal contributions from other PFAS [5]. FTCA, and other polyfluorinated compounds, were also prevalent in the limited studies analyzing for them [6,39–41]. Other studies of U.S. landfills [4,42] show high concentrations of perfluorinated alkyl acids, mainly shorter chained PFCA (≤C8), along with Chinese [43], Australian [44], Norwegian [45], and Spanish [46] landfills; however, these studies did not account for polyfluorinated compounds (especially FTCA) and likely grossly underestimate total PFAS. Compositions and concentrations are observed to range widely between landfills within the same country and with different countries. This variability can probably be mostly attributed to differences in past and current waste [7], although climate is another important factor, with MSWL leachate in wet climates having concentrations over twice that of arid regions [5]. While active landfills reported the highest concentrations, elevated PFAS concentrations were still observed from non-active landfills [47]. Leachate impacted groundwater surrounding historic unlined landfills also show elevated levels of PFAS [48], although few studies have focused on them. There are large fluctuations in leached PFAS concentrations over short time periods of weeks to months, correlated with increased precipitation and leachate flow rates [6,49]. This temporal variation makes it difficult to compare different MSW landfills and should be considered when determining annual PFAS mass contributions.

**Chemical Transformations in Landfill Leachate**

A distinct compositional discrepancy occurs between PFAS in incoming waste and in leachate, particularly for FTCA. For example, Liu et al. (2022) reported leachate leaving the landfill at 2500 ng/g 5:3 FTCA [37], typical of MSW landfill leachate [5,50], yet none was shown to leach from the incoming screenings. Another study showed a similar trend, reporting increased FTCA concentrations between the incoming waste sampled from the collection truck to the leachate leaving the landfill [51]. These trends strongly support the prevalence of polyfluorinated precursor transformation within landfills. Hamid et al. (2018) provides a thorough review of PFAS transformations, and Hamid et al. (2020) detail transformation pathways from diPAP to FTOH, FTOH to FTCA, and then ultimately FTCA to PFCA. Variable transport processes, preferential leaching of different PFAS within landfills, heterogeneity of MSW composition, and variability in water content may also play a role in PFAS transformations [50,53].

Laboratory experiments simulating landfill conditions also demonstrate accumulation of FTCA in biologically active reactors from carpet [35] and MSW [34], with no accumulation under abiotic conditions. These studies concluded that abiotic temporal changes in PFAS concentrations are associated with adsorption and leaching, while biotic degradation of the substrate increases leaching and biotransformations, evidenced by simultaneous precursor disappearance and accumulation of transformation products [34,35]. Substantially greater PFAS leaching in biologically-active vs. abiotic reactors suggest biotransformation of polyfluorinated precursors is a controlling factor on PFAS composition of MSWL leachate. Leaching of stain-guard treated carpets in distilled water resulted in 1 ng/L more PFAS than simulated leachate suggesting that the copresence of multivalent cations inhibits leaching [29]. These combined mechanisms may explain why leachate from landfills in wetter climates [5] with higher leachate flow volumes [6,49] are correlated with higher PFAS concentrations. Higher water content within the landfill may stimulate biologic activity[54], dilute competing cations [29], and increase leaching and transport [53].

Importantly, these reactor studies [29,34,35] did not monitor for FTOH or any other potential volatile PFAS in the gas produced, likely underestimating total release. FTOH are semi-volatile and have been found in the headspace of experimental reactors at 10 times the leachate concentration [55]. While no studies have yet characterized PFAS in gas vented from landfills, the prevalence of semi-volatile PFAS in MSW makes it highly likely that they are present in landfill gas. FTOH have been measured in air samples from wastewater treatment plants [56,57] and downwind of landfills [50] . These observations point to a potential pathway for release of semi-volatile precursors, such as FTOH, to the environment.

**Future Challenges**

This review highlights the importance of thoroughly characterizing PFAS in incoming MSW to better understand transformation processes inside MSWLs affecting leachate composition. However, the limited number of studies to date have large variability in waste composition, climate, and many other factors that leaves much uncertainty. To better quantify and manage the total mass PFAS in MSWLs, future work should focus on better characterizing PFAS in incoming MSW, with an emphasis on precursor compounds (e.g., FTOH and diPAP). Given their prevalence in consumer products, semi-volatile PFAS should be investigated in MSWL gas for more precise mass balances of PFAS, and as a pathway for atmospheric release. Temporal variability of PFAS in leachate remains poorly understood, along with mechanisms for PFAS release from MSW. These knowledge gaps must be addressed to more accurately quantify PFAS release from MSWLs and aid in the design of efficient on-site treatment technologies.

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**Data Archive:** https://github.com/dmreeves1/COEHS\_MSW\_PFAS

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**Tables**

Table 1. Average PFAS composition of MSW products by subcategory for paper, textiles, and carpet.

|  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- |
| **Material** | **PFAS Group** | **Concentration (ng/g)** | | **n** | **References** |
| **Average** | **Range** |
| Paper | diPAP | 629 | 84-7800 | 5-59 | D'eon et al., 2009; Guo et al., 2009; Liu et al., 2015; Kotthoff 2015; Yuan et al., 2016; Timshina et al., 2021; Zabaleta et al., 2020 |
| FTOH | 2572 | ND-41130 | 7-85 |
| FTS | ND | ND | 29 |
| FTCA | ND | ND-333 | 25 |
| PFCA | 28 | ND-15400 | 98-165 |
| PFSA | 0.2 | ND-14 | 29-87 |
| Textiles | diPAP | ~ | ~ | ~ | Guo et al., 2009; Liu et al., 2015; Becanova et al., 2016; Van der Veen et al., 2022 |
| FTOH | 257 | ND-74200 | 22-48 |
| FTS | ~ | ~ | ~ |
| FTCA | ~ | ~ | ~ |
| PFCA | 33 | ND-3300 | 35-102 |
| PFSA | 0.3 | ND-97 | 26-61 |
| Carpet | diPAP | ~ | ~ | ~ | Guo et al., 2009; Liu et al., 2015; Becanova et al., 2016; Wu et al., 2020 |
| FTOH | 284 | ND-8070 | 29-34 |
| FTS | 44 | ND-2850 | 29 |
| FTCA | ~ | ~ | ~ |
| PFCA | 21 | ND-2010 | 29-44 |
| PFSA | 3 | ND-629 | 29-36 |

FTS = fluorotelemer sulfonate

n = number of samples

ND = non-detect

**Figures**

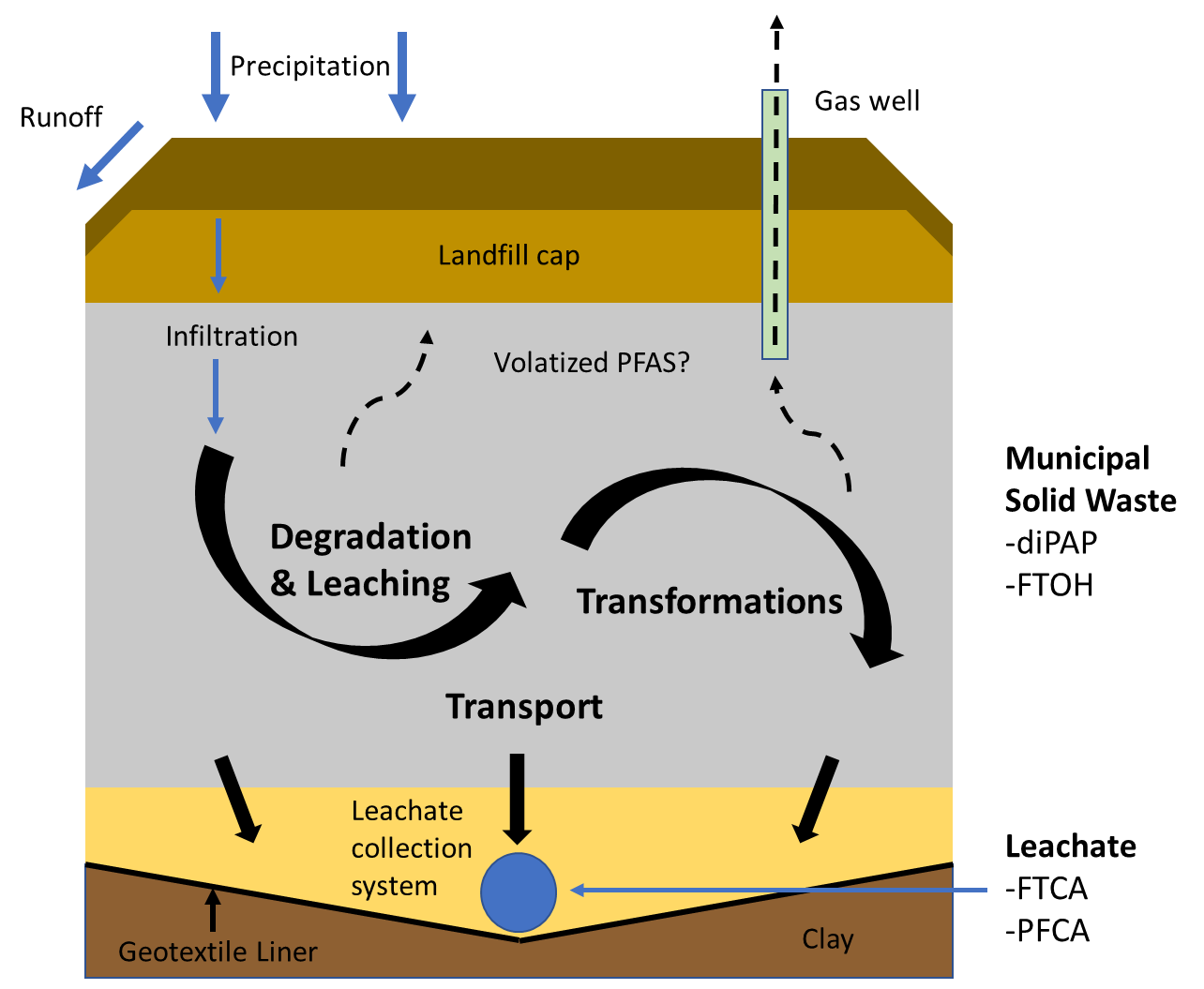


Figure 1. Modern municipal solid waste landfill design elements depicting major processes affecting PFAS composition and leachate generation. Note the evolution in PFAS from diPAP and FTOH in municipal solid waste to leachate dominant in FTCA and PFCA.

Figure 2. Combined mass of PFAS for paper (11.8%), textiles (6.0%), and carpet (1.7%) by weight in ng/g based on compilation in Table 1.