

Transport of per-polyfluoroalkyl substances (PFAS) in high-density polyethylene (HDPE) geomembrane liners

Aamir Ahmad¹, and Dr. Kuo Tian^{1*}

¹Department of Civil, Environmental, and Infrastructure Engineering, George Mason University, Fairfax, VA, 22030, USA

Abstract. The objective of this study is to investigate the fate and transport of per-polyfluoroalkyl substances (PFAS) through a high-density polyethylene (HDPE) geomembrane (GM) that is commonly used in landfill composite liner systems. Tests were conducted to measure the sorption and diffusion of four per-polyfluoroalkyl substances (PFAS) with varying numbers of carbons in chain and functional groups on HDPE GM. Perfluoroalkyl carboxyl acids (PFCAs), and perfluoroalkyl sulphonic acids (PFSAs) were investigated in this study. The partition coefficients (K_d) on HDPE GM ranged from 5 to 12 L/Kg. K_d showed an increasing trend with chain length and were found to be sensitive to functional groups of PFAS. Molecular weight directly affected the K_d . The diffusion coefficients (D_g) of PFCAs and PFSAs through 0.1-mm HDPE GM were found to be in the orders of 10^{-18} to 10^{-17} m²/s. The D_g decreased with increasing molar mass and were also observed to be dependent on the functional group. D_g of PFSAs was lower than that of PFCAs for a similar number of carbons in the chain.

1 INTRODUCTION

High-density polyethylene (HDPE) geomembranes (GM) are an integral component of a composite landfill liner (Nefso and Burns, 2007; Rowe, 2005) to reduce the transport of contaminants from municipal solid waste (MSW) landfill leachate into the subsoil/groundwater (Foose et al., 2002). HDPE GMs are primarily composed of polyethylene resin, which is a thermoplastic polymer (Tian et al. 2017). In addition to the resin antioxidants, stabilizers, and UV absorbers are also incorporated into the GM to enhance mechanical properties (e.g., tensile strength, tear, and puncture resistance) and make it resistant to chemicals, UV degradation, and weathering (Tian et al., 2018).

HDPE GMs consist of non-polar molecules and provide an excellent impedance to the advective migration of contaminants by the reduction of water flow. HDPE GMs also act as an ideal barrier to the diffusive transport of inorganic (polar) contaminants (Rowe et al., 1995). However, non-polar organic contaminants can migrate through intact HDPE GMs by diffusion (Rowe et al., 1996). The HDPE GM migration process involves three stages: (I)

* Corresponding author : ktian@gmu.edu

sorption of contaminants from landfill leachate to the upper GM layer, (II) diffusion through GM layers, and (III) desorption or partitioning between the GM's lower surface and the underlying landfill liner components like geosynthetic clay liners and compacted clay liners (Foose et al., 2002).

Perfluoroalkyl substances (PFAS) pose a growing concern for landfill facilities. These fluoro organic contaminants, known for their high stability, resistance to heat and chemicals, and hydrophobic nature, are classified based on functional groups, carbon chain lengths, and structural forms. Major groups within PFAS include perfluorocarboxylic acids (PFCAs), and perfluorosulfonic acids (PFSAs) (Buck et al., 2011). PFAS are extensively used in the production of various domestic and industrial products, such as fabrics, food packaging, water-repellent paints, carpets, upholstery, non-stick cookware, fluoroplastics, composite resins, aqueous fire-fighting filming foams (AFFF), and semiconductor industries (Buck et al., 2011; Lohmann et al., 2020). Municipal solid waste (MSW) landfills serve as a significant repository for PFAS, with leachates containing a range of PFAS compounds, as reported by Lang et al. (2017). These compounds span 14 classes of PFAS, with concentrations in the leachates ranging from 250 ng/L to 3500 ng/L. Yan et al. (2015) found that PFOA accounted for 36.8% of the total concentration of PFAS, which ranged from 7280 to 292,000 ng/L in landfill leachate collected from 5 MSW leachates in China.

Limited information exists in the literature concerning the migration of PFAS through HDPE GM. Di Battista et al. (2020) examined PFOA and PFOS diffusion and partitioning in 0.1- and 0.75-mm linear low-density polyethylene (LLDPE) GMs using double-compartment column and batch sorption tests. For LLDPE, K_d values ranged between 0.9 to 1.4 for PFOA and 2.8 to 5.3 for PFOS. The diffusion coefficient of PFOA in LLDPE was $< 3.1 \times 10^{-16} \text{ m}^2/\text{s}$, $< 13 \times 10^{-16} \text{ m}^2/\text{s}$, and $< 19 \times 10^{-16} \text{ m}^2/\text{s}$ at 23, 35, and 50°C, respectively. For PFOS, the measured diffusion coefficients were $< 3.2 \times 10^{-16} \text{ m}^2/\text{s}$, $< 40 \times 10^{-16} \text{ m}^2/\text{s}$, and $< 52 \times 10^{-16} \text{ m}^2/\text{s}$ at the same temperatures (Di Battista et al., 2020). The diffusion coefficients of PFOA and PFOS may be even lower because of the greater tortuosity through the amorphous crystalline nature of HDPE GMs compared to LLDPE (Di Battista et al., 2020).

The objective of our study was to quantify the partition and diffusion coefficients of four PFAS compounds i.e., PFBA, PFBS, PFOA, and PFOS in HDPE GM, and to understand how these parameters are affected by changing molar mass and functional group. These parameters will help researchers to predict the PFAS diffusive flux through HDPE GM and give insight into the effectiveness of HDPE GMs as part of the composite liner systems for the containment of PFAS.

2 Materials and methods

2.1 Materials

A smooth 0.1-mm HDPE GM custom manufactured to be used in this study. The 0.1mm HDPE GM was procured to be used as part of broader research including long-term double-compartment diffusion tests for PFAS. The engineering properties including thickness, puncture resistance, tear resistance, and tensile strength were measured according to ASTM D5944, ASTM D4833, ASTM D1004, and ASTM D6693 respectively. All HDPE GM specimens were washed with DI water, before testing, to clean any impurities/dust present on the surface. The puncture and tear resistance were found to be 36.5 and 13.3 N respectively whereas 1.31 kN/m tensile strength at the yield point was observed with a 54.4% elongation. A tensile strength of 1.56 kN/m corresponding to a 264% elongation was observed at the breakpoint.

2.2 Batch Testing

Static batch tests were conducted to measure the partition coefficients of PFAS on the 0.1-mm HDPE GM, according to procedures outlined in Park et al. (2012), and Eun et al. (2018). Four PFAS were investigated in this study i.e., PFBA, PFBS, PFOA, and PFOS. The selected PFAS can be classified based on carbon chain length and functional group. The number of carbons in the chain of these PFAS were C-4 (PFBA, PFBS) and C-8 (PFOA, PFOS). Functional groups include carboxylic acid (PFCAs), and sulphonic acid (PFSAs). The CAS No., acronyms, formulae, number of carbons in the chain, molar weight, and functional groups of these selected PFAS are shown in Table 1.

Table 1. Physical properties of selected PFAS

Name	CAS No.	Acronym	Formula	Carbon Chain Length	Mol. Wt. (g/mol)	Functional Group Classification
Perfluorobutanoic acid	375-22-4	PFBA	C ₄ HF ₇ O ₂	4	214.04	Perfluoroalkyl Carboxylic Acids
Perfluorooctanoic acid	335-67-1	PFOA	C ₈ HF ₁₅ O ₂	8	414.07	
Perfluorobutane sulfonic acid	375-73-5	PFBS	C ₄ HF ₉ O ₃ S	4	300.10	Perfluoroalkane Sulfonic Acids
Perfluorooctane sulfonic acid	1763-23-1	PFOS	C ₈ HF ₁₇ O ₃ S	8	500.13	

PFAS solutions ranging from 200 to 800 µg/L were prepared using Milli-Q Type I water with a near-neutral pH (6.7 to 6.9). 0.4 g of HDPE GM specimens were added to 10 mL of these solutions in a 15 mL Falcon polypropylene tube. The sealed vials were rotated at 60 rpm for 8 days at room temperature (20-25 °C) to reach equilibrium, a duration determined from initial trials. Duplicate tests were performed.

After the equilibration period, HDPE GM strips were removed, and the polypropylene vials were centrifuged at 2000 rpm for 20 minutes to ensure PFAS solution homogeneity. The supernatant from each vial was sequentially filtered using 0.2-micron polypropylene syringe filters from Fisher Scientific (Waltham, Massachusetts) and then 0.05-micron Polyether sulfone (PES) syringe filters from Tisch Scientific (Clevs, Ohio). To prevent potential contamination, all reusable glassware was pre-rinsed with methanol. A 0.4-mL aliquot of the filtrate was diluted 1:1 in methanol, spiked with calibration standards, and analyzed using liquid chromatography-tandem mass spectrometry (LC-MS/MS, as detailed in a later section). Each experiment was performed in triplicate for accuracy and consistency. To check for any contamination or interaction with container walls, control tests were conducted using 15-mL Falcon tubes with 10 mL of multi-PFAS solutions at each concentration. The K_d (L/Kg or mL/g) was calculated using equation (1) below, as the ratio between the concentration of a sorbed contaminant in the GM (C_s in nmol/g) and its concentration in the aqueous phase at equilibrium (C_{eq} in nmol/mL), with C_{eq} values determined directly via LC-MS/MS.

$$K_d = \frac{C_s}{C_{eq}} \tag{1}$$

The corresponding concentrations of PFAS sorbed by HDPE GM were calculated as the difference from the initial test concentration. Linear fitting was used to fit the sorption isotherms and a least squares fitting of the sorption isotherms was performed with the Solver function in Microsoft Excel™ to find the set of adjustable parameters that minimized the root mean square error.

Kinetic batch tests were conducted on 0.1-mm HDPE GM to measure the diffusion coefficient of four PFAS (PFBA, PFBS, PFOA, and PFOS). This method was adapted from Park et al., (2012) and Eun et al., (2018). Like the static batch tests, 0.4 g HDPE GM strips were placed in 15-mL Falcon tubes with 10 mL of multi-solute PFAS solutions, yielding a concentration of 300 µg/L for each PFAS compound. Duplicate vials were decommissioned, and samples were collected daily from day 1 to day 8. The PFAS concentration data was analyzed assuming a planar HDPE GM sheet initially free of solutes, suspended with both sides exposed in a well-stirred solution of fixed volume and mass. The diffusion coefficients were obtained using Crank's analytical solution (1975).

$$\frac{C_t}{C_0} = \exp\left(\frac{D_g t (K_d)^2}{A^2}\right) \operatorname{erfc}\left(\frac{D_g t (K_d)^2}{A^2}\right)^{0.5} \quad (2)$$

where C_t = concentration of the PFAS in the solution [ML^{-3}] at time t [T]; C_0 = initial concentration of solute in the solution [ML^{-3}]; D_g = diffusion coefficient [L^2T^{-1}] in the geomembrane; K_d = partition coefficient; and A = half thickness of the solution in contact with both sides of the geomembrane [L], which is calculated by dividing the volume of the solution by the area of the geomembrane.

2.3 LC-MS/MS Analysis

Shimadzu 8050 LC-MS/MS (Shimadzu, Columbia, MD) was used for analyzing all PFAS samples. The instrument included dual Nexera 20ADXR UHPLC pumps, SIL-20ACXR autosampler, and a C18 Restek Raptor Column (50 mm x 2.1 mm, 1.8 µm particle diameter). Analyses were performed at a constant flow rate of 0.4 mL/min. The mobile phases consisted of 5 mM ammonium acetate in water (A) and 5 mM ammonium acetate in methanol (B). The gradient elution method followed this sequence: 0.10 min at 90% A, 1.00 min at 70% A, 9.00 min at 30% A, 9.50 min at 5% A, 10.00 min at 5% A, 10.10 min at 90% A, and 13.10 min at 90% A. The C18 column was maintained at 40 °C. The MS detector operated with a capillary voltage of 2.8 kV, cone gas flow rate of 50 L/h, desolvation gas flow rate of 100 L/h, source temperature at 120 °C, and desolvation temperature at 400 °C. Nitrogen (99.9%) served as the cone gas, and argon (99.9%) functioned as the collision gas. All PFAS compounds were analyzed in negative electrospray ionization and MRM detection modes. The identification process utilized three characteristic MRM ions, with the first (MRM-1) used for quantification.

The calibration method employed internal injection standards (IS) and an eight-point calibration curve, following USEPA method 537 with forced origin calibration curves. Target suppression for IS and sorption test percentage recovery was evaluated using surrogate standards (SS). High-purity analytical standards were procured as single compounds from Wellington Laboratories (Guelph, Ontario, Canada) in 1.2 mL glass ampules, including six isotopically labeled standards (3 x IS and 3 x SS). IS comprised 13C2 perfluorooctanoic acid (13C2-PFOA), 13C4 perfluorooctanesulfonate (13C4-PFOS), and d3-N-methyl perfluorooctane-sulfonamido acetic acid (d3-N-MeFOSAA). SS included 13C2 perfluorohexanoic acid (13C2-PFHxA), 13C2 perfluorodecanoic acid (13C2MPFDA), and d5-N-ethyl perfluorooctane-sulfonamido acetic acid (d5-N-EtFOSAA).

Polypropylene labware was methanol-rinsed to prevent contamination. HDPE GM sample blanks were assessed for their contribution to compound concentrations. Control tests using the same materials didn't show notable compound losses or degradation due to PFAS sorption. Reproducibility, calculated as the relative standard deviation of triplicate analyses, fell within the acceptable 5-15% range.

3 Results and Discussion

3.1 Partition coefficients

The K_d of PFCAs and PFSAs for the HDPE GM evaluated in this study show an increasing trend with increasing molar mass on the HDPE GM. The K_d for PFBA (C-4) was found to be 5.8 L/kg which increased to 9.8 L/kg for PFOA (C-8). The K_d of PFBS (C-4) was found to be 8.3 L/kg which increased to 12.1 L/kg for PFOS (C-8). The sorption of PFAS depends on their molecular mass, primarily the number of carbon atoms. This is because as the molecular mass increases, the surface area to volume ratio decreases, resulting in stronger Van der Waals forces among the PFAS molecules (Atkins et al., 2017). These temporary dipoles in PFAS lead to increased hydrophobicity, causing them to repel the aqueous phase and enhancing their likelihood of physically interacting with the HDPE GM structure. A similar pattern has been previously noted with chlorinated hydrocarbons, where the sorption of 1,2-dichloroethane (1,2-DCA) (C-2) was observed to be higher than that of dichloroethane (DCM) (C-1) on HDPE GM (Sangam and Rowe, 2001).

Table 2. Partition and diffusion coefficients of four PFAS with HDPE GM.

Name	Acronym	K_d (L/Kg)	D_g (m ² /s)
Perfluorobutanoic acid	PFBA	5.8	4.67E-17
Perfluorooctanoic acid	PFOA	9.8	1.30E-17
Perfluorobutane sulfonic acid	PFBS	8.3	2.81E-17
Perfluorooctane sulfonic acid	PFOS	12.1	6.42E-18

The functional group of PFAS affects the K_d on HDPE GM, i.e., carboxyl vs. sulfonic. For example, PFBA and PFBS have a similar number of carbons in the chain but consist of different terminal functional groups. The K_d of C-4 PFAS increased from 5.8 L/kg to 8.3 L/kg with a change in the functional group (PFBA vs PFBS). Similarly, the K_d of PFOS is 12.1 L/kg, whereas the K_d of PFOA is 9.8 L/kg, as shown in Table. 2. PFSAs exhibit greater sorption, primarily attributed to the higher hydrophobicity of the sulfonic acid functional group in comparison to the carboxylic functional group (Xing et al., 2020). This elevated hydrophobicity in PFSAs, even with an equal number of carbon atoms, results from an extra carbon atom in the chain/tail of PFSA compared to PFCA. This difference leads to a stronger affinity for sorption on HDPE GM.

3.2 Diffusion coefficients

The D_g of four PFAS (PFCAs and PFSAs) were determined from kinetic batch tests. As an example, the measured relative concentration (C_t/C_o) profile of PFOA (C-8) as a function of time is shown in Fig. 1. The D_g were obtained by fitting the relative concentration curve data from the kinetic batch tests with equation (2) using non-linear squares regression with the Solver tool in Microsoft Excel. The fittings for different PFAS were made by solving only for the value of D_g while the measured values of K_d were used from the results of the static batch sorption tests.

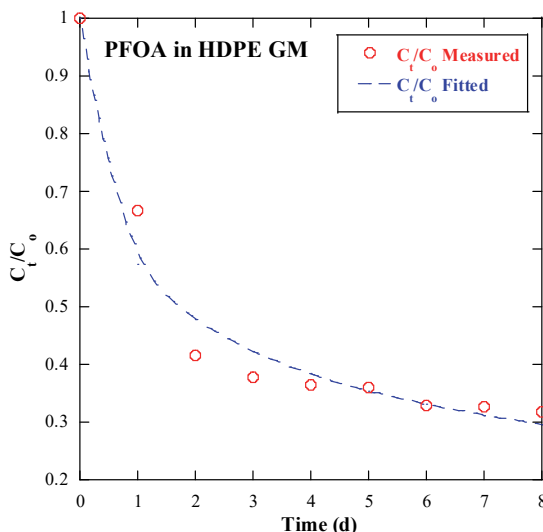


Fig. 1. Example fitting of relative concentration versus time for determination of diffusion coefficient (D_g) for PFOA(C-8).

The D_g of PFCAs and PFSA in 0.1-mm HDPE GM are shown in Table. 2. Generally, a decreasing trend of D_g was observed for all PFAS with increasing molar mass (number of carbons in the chain). The D_g of PFBA (C-4) was observed to be $4.7 \times 10^{-17} \text{ m}^2/\text{s}$ which decreased to a value of $1.3 \times 10^{-17} \text{ m}^2/\text{s}$ for PFOA (C-8). A similar decreasing trend was also seen for PFSA where the values of D_g observed for PFBS (C-4), and PFOS (C-8) were $2.8 \times 10^{-17} \text{ m}^2/\text{s}$, and $6.4 \times 10^{-18} \text{ m}^2/\text{s}$ respectively.

The effect of the functional group on the D_g can also be observed. At a similar number of carbons in the chain, the D_g for PFSA was found to be lower than that of PFCAs, e.g., PFOS has a D_g of $6.4 \times 10^{-18} \text{ m}^2/\text{s}$ whereas PFOA has a D_g of $1.3 \times 10^{-17} \text{ m}^2/\text{s}$. Similar findings were observed for C-4 (PFBS and PFBA), as shown in Table 2. The greater molecular mass of PFSA molecules, along with their higher sorption tendencies, likely leads to reduced permeation through the HDPE GM structure, resulting in lower diffusion coefficients compared to PFCAs. Generally, diffusion coefficients (D_g) tend to decrease as the molar mass or the number of carbons in the chain increases. Additionally, it was observed that, for PFAS with similar chain lengths, PFSA exhibit lower D_g values compared to PFCAs.

4 Conclusions

PFAS migration through landfill liners is currently a particular area of concern for landfill owners, regulators, and geo-environmental engineers. HDPE GMs are a critical component of MSW landfill liners to prevent contaminants from leaching out to the environment. Sorption and diffusion are two important mechanisms that potentially dictate the contaminants to pass through HDPE GMs. In this study, static and kinetic batch tests were conducted to measure the partition and diffusion coefficients of PFAS migrating through HDPE GM.

Four PFAS with varying numbers of carbons in the chain and functional groups have been investigated. The results demonstrate that all PFAS investigated tend to sorb onto HDPE GM and can diffuse through HDPE GM. The partition coefficient ranged from 5 to 12 L/kg. The partition coefficient was also observed to be dependent on the no of carbons in the chain and the terminal functional group. The diffusion coefficients of PFAS in HDPE GMs varied in

order from 10^{-18} m²/s to 10^{-17} m²/s. The diffusion coefficients showed an inverse relationship with an increase in molar mass/number of carbons in the chain. The diffusion coefficients of PFASs were smaller than PFCAs for a similar number of carbons.

References

1. ASTM. (2013a). "Standard test method for index puncture resistance of geomembranes and related products." ASTM D4833 M-07, West Conshohocken, PA.
2. ASTM. (2013b). "Standard test method for tear resistance (graves tear) of plastic film and sheeting." ASTM D1004-13, West Conshohocken, PA.
3. ASTM. (2015a). "Standard test method for determining tensile properties of nonreinforced polyethylene and nonreinforced flexible polypropylene geomembranes." ASTM D6693 M-04, West Conshohocken, PA.
4. ASTM. (2015b). "Standard test method for measuring core thickness of textured geomembranes." ASTM D5944 M-10, West Conshohocken, PA.
5. Atkins, P. J., De Paula, J., & Keeler, J. (2017). *Atkins' physical chemistry* (11th ed.). Oxford University Press.
6. Buck, R. C., Franklin, J., Berger, U., Conder, J. M., Cousins, I. T., Voogt, P. De, Jensen, A. A., Kannan, K., Mabury, S. A., & van Leeuwen, S. P. J. (2011). Perfluoroalkyl and polyfluoroalkyl substances in the environment: Terminology, classification, and origins. *Integrated Environmental Assessment and Management*, 7(4), 513–541. <https://doi.org/10.1002/ieam.258>
7. Crank, J. (1975). "The mathematics of diffusion." 2nd Ed., Clarendon, Oxford, U.K., 56–60.
8. Di Battista, V., Rowe, R. K., Patch, D., & Weber, K. (2020). PFOA and PFOS diffusion through LLDPE and LLDPE coextruded with EVOH at 22 °C, 35 °C, and 50 °C. *Waste Management*, 117, 93–103. <https://doi.org/10.1016/j.wasman.2020.07.036>
9. Eun, J., Tinjum, J. M., Benson, C. H., & Edil, T. B. (2018). Equivalent Transport Parameters for Volatile Organic Compounds in Coextruded Geomembrane-Containing Ethylene-Vinyl Alcohol. *Journal of Geotechnical and Geoenvironmental Engineering*, 144(7), 1–14. [https://doi.org/10.1061/\(asce\)gt.1943-5606.0001888](https://doi.org/10.1061/(asce)gt.1943-5606.0001888)
10. Foose, G. J., Benson, C. H., & Edil, T. B. (2002). Comparison of Solute Transport in Three Composite Liners. *Journal of Geotechnical and Geoenvironmental Engineering*, 128(5), 391–403. [https://doi.org/10.1061/\(asce\)1090-0241\(2002\)128:5\(391\)](https://doi.org/10.1061/(asce)1090-0241(2002)128:5(391))
11. Lang, J. R., Allred, B. M. K., Field, J. A., Levis, J. W., & Barlaz, M. A. (2017). National Estimate of Per- and Polyfluoroalkyl Substance (PFAS) Release to U.S. Municipal Landfill Leachate. *Environmental Science and Technology*, 51(4), 2197–2205. <https://doi.org/10.1021/acs.est.6b05005>
12. Lohmann, R., Cousins, I. T., DeWitt, J. C., Glüge, J., Goldenman, G., Herzke, D., Lindstrom, A. B., Miller, M. F., Ng, C. A., Patton, S., Scheringer, M., Trier, X., & Wang, Z. (2020). Are Fluoropolymers Really of Low Concern for Human and Environmental Health and Separate from Other PFAS? *Environmental Science & Technology*, 54(20), 12820–12828. <https://doi.org/10.1021/acs.est.0c03244>
13. Nefso, E. K., & Burns, S. E. (2007). Comparison of the equilibrium sorption of five organic compounds to HDPE, PP, and PVC geomembranes. *Geotextiles and Geomembranes*, 25(6), 360–365. <https://doi.org/10.1016/j.geotextmem.2006.12.002>

14. Park, M. G., Benson, C. H., & Edil, T. B. (2012). Comparison of batch and double compartment tests for measuring voc transport parameters in geomembranes. *Geotextiles and Geomembranes*, 31, 15–30. <https://doi.org/10.1016/j.geotextmem.2011.09.001>
15. Rowe, R. K. (2005). Long-term performance of contaminant barrier systems. *Geotechnique*, 55(9), 631–678. <https://doi.org/10.1680/geot.2005.55.9.631>
16. Rowe, R., Hrapovic, L., & Armstrong, M. D. (1996). Diffusion of organic pollutants through HDPE geomembrane and composite liners and its influence on groundwater quality. 737–742.
17. Rowe, R., Hrapovic, L., & Kosaric, N. (1995). Diffusion of Chloride and Dichloromethane Through an HDPE Geomembrane. *Geosynthetics International*, 2. <https://doi.org/10.1680/gein.2.0021>
18. Sangam, H. P., & Rowe, R. K. (2001). through HDPE geomembranes \$. *Geotextiles and Geomembranes*, 19, 329–357.
19. Tian, K., Benson, C. H., Tinjum, J. M., & Edil, T. B. (2017). Antioxidant Depletion and Service Life Prediction for HDPE Geomembranes Exposed to Low-Level Radioactive Waste Leachate. *Journal of Geotechnical and Geoenvironmental Engineering*, 143(6), 1–11. [https://doi.org/10.1061/\(asce\)gt.1943-5606.0001643](https://doi.org/10.1061/(asce)gt.1943-5606.0001643)
20. Tian, K., Benson, C. H., Yang, Y., & Tinjum, J. M. (2018). Radiation dose and antioxidant depletion in a HDPE geomembrane. *Geotextiles and Geomembranes*, 46(4), 426–435. <https://doi.org/10.1016/j.geotextmem.2018.03.003>
21. USEPA. (2020). 537.1 - 1. 1(2009), 1–50.
22. Xing, D. Y., Chen, Y., Zhu, J., & Liu, T. (2020). Fabrication of hydrolytically stable magnetic core-shell aminosilane nanocomposite for the adsorption of PFOS and PFOA. *Chemosphere*, 251, 126384. <https://doi.org/10.1016/j.chemosphere.2020.126384>
23. Yan, H., Cousins, I. T., Zhang, C., & Zhou, Q. (2015). Perfluoroalkyl acids in municipal landfill leachates from China: Occurrence, fate during leachate treatment and potential impact on groundwater. *Science of The Total Environment*, 524–525, 23–31. <https://doi.org/https://doi.org/10.1016/j.scitotenv.2015.03.111>