Science of the Total Environment 912 (2024) 169026

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Review

Environmental pitfalls and associated human health risks and ecological impacts from landfill leachate contaminants: Current evidence, recommended interventions and future directions

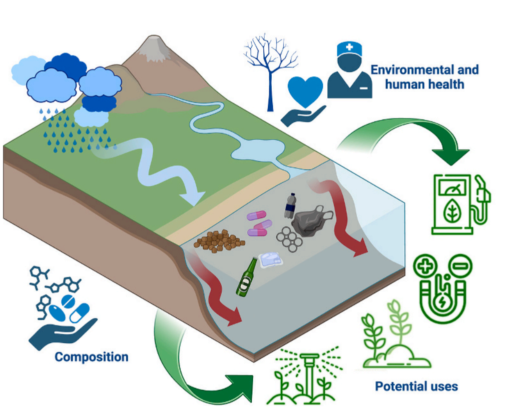
Viraj Gunarathne [a,b,1], Ankur J. Phillips [c,1], Alessandra Zanoletti [d], Anushka Upamali Rajapaksha [a,e], Meththika Vithanage [a], Francesco Di Maria [f], Alberto Pivato [g], Ewa Korzeniewska [h], Elza Bontempi [d,*]

[a] Ecosphere Resilience Research Centre, Faculty of Applied Sciences, University of Sri Jayewardenepura, Nugegoda, CO 10250, Sri Lanka
[b] University of Wuppertal, School of Architecture and Civil Engineering, Institute of Foundation Engineering, Water- and Waste-Management, Laboratory of Soil- and Groundwater-Management, Pauluskirchstraβe 7, 42285 Wuppertal, Germany
[c] Department of Microbiology, College of Basic Sciences and Humanities, Govind Ballabh Pant University of Agriculture and Technology, Pantnagar, 263145, Uttarakhand, India
[d] INSTM and Chemistry for Technologies Laboratory, Department of Mechanical and Industrial Engineering, University of Brescia, Via Branze, 38, 25123 Brescia, Italy
[e] Instrument Center, Faculty of Applied Sciences, University of Sri Jayewardenepura, Nugegoda 10250, Sri Lanka
[f] LAR5 Laboratory, Dipartimento di Ingegneria, University of Perugia, via G. Duranti 93, 06125 Perugia, Italy
[g] DICEA - Department of Civil, Environmental and Architectural Engineering, University of Padova, Via Marzolo 9, 35131 Padova, Italy
[h] Department of Water Protection Engineering and Environmental Microbiology, The Faculty of Geoengineering, University of Warmia and Mazury in Olsztyn, Prawocheńskiego 1 Str., 10-719 Olsztyn, Poland

HIGHLIGHTS

- Landfill and uncontrolled dumps increase the risk of hazardous leachate release.
- The environmental contamination by leachate and the consequent human health risks are investigated.
- The overall perspective of leachate contamination through living communities is discussed.
- Leachate pollution index, potential ecological risks of leachate and emerging contaminants are evaluated.

GRAPHICAL ABSTRACT



* Corresponding author.
E-mail address: elza.bontempi@unibs.it (E. Bontempi).
[1] These authors contributed equally to this work.

https://doi.org/10.1016/j.scitotenv.2023.169026
Received 3 August 2023; Received in revised form 17 October 2023; Accepted 29 November 2023
Available online 4 December 2023

A R T I C L E I N F O

Editor: Damià Barceló

Keywords:
Municipal solid waste
Antibiotics and antimicrobial resistance
Groundwater and surface water bodies
Emerging contaminants
Microplastics
Toxicological and ecological risks

A B S T R A C T

The improper management of solid waste, particularly the dumping of untreated municipal solid waste, poses a growing global challenge in both developed and developing nations. The generation of leachate is one of the significant issues that arise from this practice, and it can have harmful impacts on both the environment and public health. This paper presents an overview of the primary waste types that generate landfill leachate and their characteristics. This includes examining the distribution of waste types in landfills globally and how they have changed over time, which can provide valuable insights into potential pollutants in a given area and their trends. With a lack of specific regulations and growing concerns regarding environmental and health impacts, the paper also focuses on emerging contaminants. Furthermore, the environmental and ecological impacts of leachate, along with associated health risks, are analyzed. The potential applications of landfill leachate, suggested interventions and future directions are also discussed in the manuscript. Finally, this work addresses future research directions in landfill leachate studies, with attention, for the first time to the potentialities that artificial intelligence can offer for landfill leachate management, studies, and applications.

1. Introduction

Municipal solid waste generation is mainly attributable to human activity advancements. The rapid explosion of the human population led to a significant expansion of industrial waste and municipal solid waste generation (Abdel-Shafy and Mansour, 2018; Adamović et al., 2018; Alobaid et al., 2018; Jouhara et al., 2017). In the last 50 years, the world population has risen from about 3 billion to >7 billion. This figure is expected to reach about 8.6 billion by 2030 and 9.8 billion by 2050 (United Nations, 2023). Due to the population increase, the amount of waste generated is expected to increase from about 1.2 billion Mg in 2010 to about 2.2 billion Mg in 2025 (Di Maria et al., 2018a). Norbu et al. (2005) claimed that the municipal solid waste generated by Asians living in cities would be around 5.2 million m^3 or 1.8 million tonnes every day by 2025.

Municipal solid waste generation along with mismanagement is a serious and challenging issue concerning environmental pollution, social harmony, and sustainable economic advancement (de Souza Melaré et al., 2017; Kawai and Tasaki, 2016). Of particular interest are also effects related to environmental pollution, public health risks, social concern, and economic integrality worldwide (de Souza Melaré et al., 2017; Dolar et al., 2016).

Fig. 1 reports the projection of waste generation worldwide in 2016, 2030, and 2050, by region (www.worldbank.org). The global production of waste is anticipated to persist in its upward trajectory across the world in the coming decades. In 2016, the collective volume of waste generated in East Asia and the Pacific reached 468 million metric tons. Forecasts indicate that by 2050, the waste generation within this region is poised to surge to 714 million metric tons.

Almost half of the world's population, mainly in low-incoming countries, cannot have access to basic services concerning waste collection and disposal, with the consequence that a large amount of the world's generated waste is improperly disposed of. These landfills are unregulated, then they are often uncontrolled and located close to urban areas. In a recent report (STATISTA, 2023) it was estimated that about 60 million people live <10 km away from the 50 largest dumpsites in the world, with severe health and environmental consequences.

Once municipal solid waste has been arranged in a landfill, it continues to decompose. The main decomposition products are gases and leachate. In particular, the excessive moisture content in the wastes and their exposition to rainwater can overflow through the deposited wastes resulting in leachate generation (Anand et al., 2021). Then, leachate is a

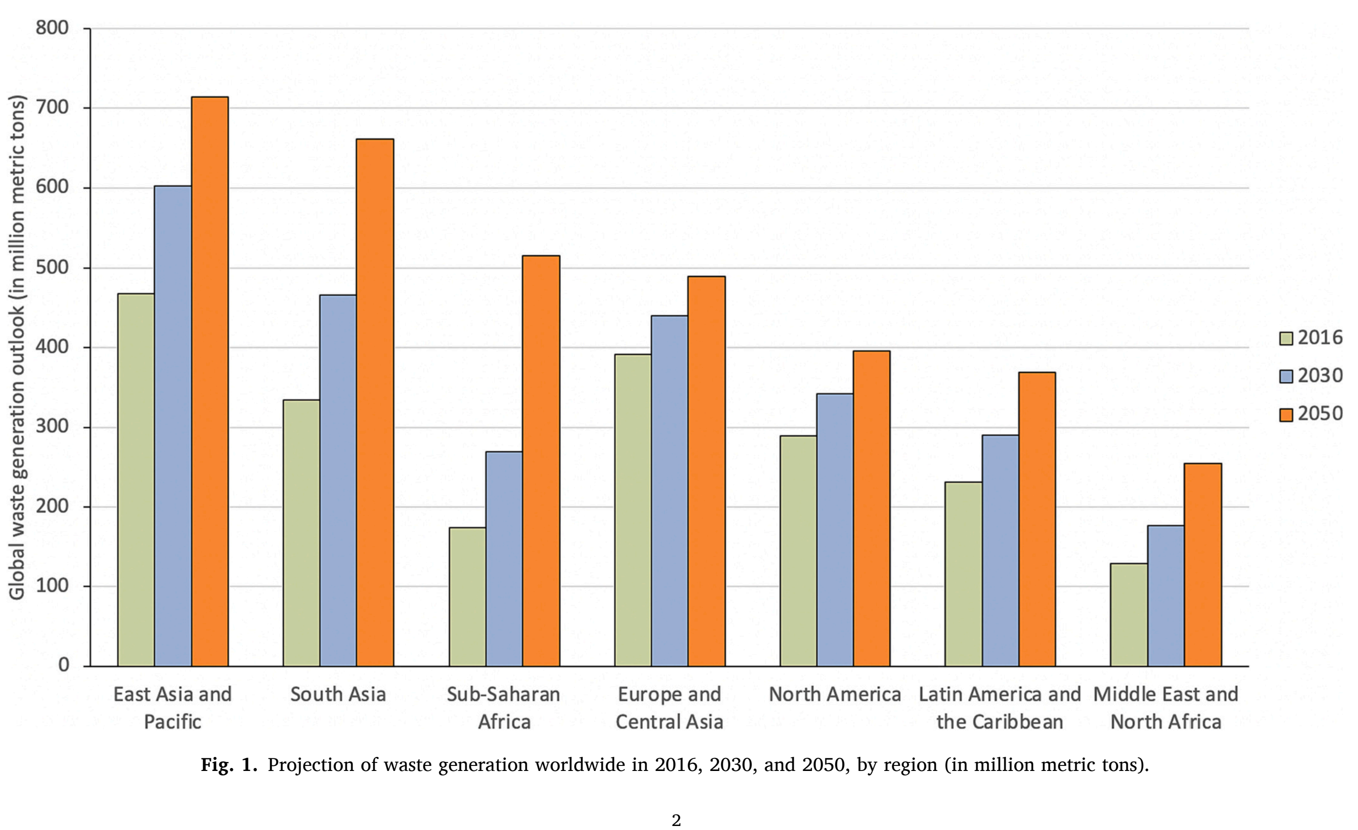


Fig. 1. Projection of waste generation worldwide in 2016, 2030, and 2050, by region (in million metric tons).

dark brown liquid mixture, with a foul smell, which can be formed of biodegradable and non-biodegradable compounds. Leachate generally accumulates at the bottom of a landfill.

It is considered hazardous wastewater, due to the presence of common and toxic chemicals, heavy metals, and inorganic compounds such as Ca^{2+}, Mg^{2+}, Na^{+}, NH_4^{+}, Fe^{3+}, Mn^{2+}, Cl^{-}, SO_4^{2-}, HCO_3^{-}, in large quantities (Anand et al., 2021). Moreover, landfill leachate may contain pathogenic microorganisms, and then promote virus migration in the environment, which may be critical during a pandemic (Anand et al., 2022). This was recently verified for COVID-19, with SARS-CoV-2 RNA fragments detection in landfill leachate (Mondelli et al., 2022).

In the landfill leachate, both aerobic and anaerobic degradation processes occur, and its composition mainly depends on the characteristics of the waste that is being dumped. Depending on the management practices, the thickness of the layer of waste placed and the age of the landfill, the degradation processes can be grouped into the following main four phases (Luo et al., 2020): aerobic phase occurring for shallow waste during the first days from their disposal; acid phase occurring for waste placed at higher depth in anaerobic/anoxic conditions; methanogenic phase lasting for several years, occurring after the acid phase in anaerobic conditions; stabilization phase, occurring after quite a long period once the larger part of biodegradable compounds was degraded. As reported by Teng et al. (2021), leachate pollutants such as chemical oxygen demand and biological oxygen demand can be decreased up to >90 % passing from a less than five-year-old to a more than ten-year-old landfill.

Uncontrolled dumpsites are open dumps where wastes are left uncovered and untreated, leaving the refuse open to the full effects of the atmospheric elements for example rain and water, without any proper management of gaseous and liquid emissions. In low-income countries, about 93 % of waste is burned or dumped on roads, open land, or waterways, whereas in high-income countries only 2 % of waste is dumped (see Table 1). Furthermore, no controls are performed on the amount and characteristics of the waste disposed of. Due to the absence of proper systems, the leachate generated by uncontrolled dumping represents a serious threat to the contamination of the water resource by percolation through the soil and surface runoff (Anand et al., 2021).

As a consequence, leachate generated from landfills can have significant environmental and health impacts (Essien et al., 2022; Gupta and Arora, 2016; Kooch et al., 2023). The major potential environmental impacts related to landfill leachate are pollution of groundwater and surface waters. The released leachate from landfills greatly affects the soil physicochemical, biological, and groundwater properties associated with agricultural activity and human health (Anand et al., 2021). The infiltration of leachate can negatively impact soil quality and fertility. The accumulation of contaminants in soil can hinder plant growth and disrupt terrestrial ecosystems. Moreover, landfills can emit volatile compounds and odorous gases into the atmosphere. These emissions can contribute to air pollution and potentially affect the health of nearby communities (Jayawardhana et al., 2019). The impact on human health is related to exposure to pollutants in drinking water, which can lead to various health problems, including neurological, respiratory, and gastrointestinal disorders. The inhalation of odorous gases and volatile compounds can irritate the respiratory system and cause discomfort for people, for example living in proximity to landfills. Contaminants from landfills can enter the food chain when crops and livestock are exposed to polluted soil or water. This can lead to the consumption of contaminated food products, posing health risks to consumers (Parvin and Tareq, 2021).

Engineering and sanitary landfills are designed and managed to minimize emissions. In these facilities, the prevention of leachate formation and material dragging effect due to wind are pursued by several management practices such as the daily cover of the disposed waste, optimization of the waste placement and waste compaction. Furthermore, sanitary/engineered landfills are also equipped with proper systems for both gas and leachate collection and treatment (Di Maria et al., 2018b) and tight controls are performed on the amount and characteristic of the waste placed. However, some minor risks concerning water contamination can also occur for engineering and sanitary landfills due to the cracking of the natural and artificial barrier systems causing some leakages of both leachate and landfill gas. Such events can happen during the whole life of the landfill with a probability higher than expected and can be caused by many factors such as waste settlement, bad design and/or choice of materials, installation damage and ageing (Pivato, 2011).

Over the years, advanced concepts of sanitary landfills management have been also proposed based on past experiences and according to the new environmental challenges as the “sustainable landfills” (Cossu and Stegmann, 2018). The sustainable landfill is functionally designed to accelerate the degradation of waste to an inert state in the shortest possible period, generally <30 years, for achieving a lower and possibly sustainable impact on the environment. Enhancing the degradation processes also affects the amount of pollutants in the leachate by decreasing, among others, the chemical oxygen demand and biological oxygen demand content. Currently, most of the literature available focuses on the contaminants typically encountered in landfill leachate and are included in national regulations. However, there is little information on emerging contaminants eventually present in this waste.

This paper offers a comprehensive examination of landfill leachate, its origins, and distinctive characteristics. Furthermore, it presents, for the first time, a comprehensive analysis of the global distribution of various waste types in landfills and their evolving patterns over time. This analysis offers insights into the potential pollutants that could be encountered in respective areas and their likely trajectories in the future. The work's primary objective is to underscore the critical need for a more thorough evaluation of the environmental and health ramifications linked to leachate originating from diverse disposal methods. In light of the absence of specific regulations and in light of the escalating concerns about environmental and health implications, particular attention is dedicated to emerging contaminants. While considering the existing body of literature on the risk assessment of landfill leachate contaminants, this paper seeks to provide a broader perspective. The work also aims to give an overview of the possibilities to fit the modern needs of the circular economy, by proposing some reuses of landfill leachate. Finally, it presents the challenges and perspectives for landfill leachate management, offering for the first time a vision of the future perspectives offered by artificial intelligence (AI).

Table 1
Per cent distribution of municipal solid waste treatment and disposal worldwide in 2016, by region and by method.

Region	Open/uncontrolled dump	Sanitary/engineered landfill	Composting	Recycling	Incineration
North America	–	54	1	33	12
South Asia	75	4	16	5	–
Middle East and North Africa	53	34	4	9	–
Europe and Central Asia	26	26	11	20	18
Sub-Saharan Africa	69	24	–	7	–
Latin America and Caribbean	27	69	–	4	–
East Asia and Pacific	18	46	2	9	24

Source: Kaza et al. (2018).

2. Geographical distribution of landfill waste

Many activities such as municipal solid waste recycling performed by the informal sector, for example, waste scavengers and keepers, domestic and open burning, littering, and open dumping cause serious and challenging problems with potentially adverse environmental and health consequences. Globally, a large percentage of municipal solid waste is disposed of in open dumpsites or sanitary/engineered landfills, as shown in Table 1. In particular, for developing economies, >61 % of waste is still not collected (Kaza et al., 2018), up to 93 % of waste is dumped or openly burned and <3 % is disposed of in engineered landfills (Maalouf et al., 2020). It is evident that waste disposal methods exhibit notable disparities based on both income levels and geographical regions. In lower-income nations, open dumping is widespread due to the absence of established landfill facilities.

At the world level, the presence of waste on open land, roads and waterways due to improper and uncontrolled waste management and disposal, for example, open burning, dumping, and littering, has been estimated on about 35–40 % of the whole waste generated affecting a population of about 3.5–4 billion peoples (Atlas, 2014). The 50 largest dumpsites in Africa, Asia and Latin America affected the life of about 64,000,000 people in the area, and manifest waste and leachate go virtually into the rivers and the sea as 38 of these dumpsites are built in coastal areas (ISWA, 2016). In India, Indonesia, and the Philippines, about 9,000,000 people were at high risk of exposure to hazardous chemical pollutants released from about 370 dumpsites (Chatham-Stephens et al., 2013). All of this represents a serious environmental risk and a possible source of disease outbreaks.

Fig. 2 reports a ranking of some of the world's largest dump sites as of 2019, showing their size in acres (Worldatlas, 2022). Data are available in the Supporting information (S1). Xinfeng landfill site in China, is one of the largest landfills in Asia, receiving daily 7000 tons of generated waste. This landfill has a leachate collection system. Deonar landfill site, located in Mumbai, India, receives daily approximately 5500 tons of waste and the Puente Hills landfill site, located in Los Angeles, California, USA, could daily take up to 13,200 tons of waste. The leachate head exceeded 8 m in the Laogang landfill in Shanghai (Touze-Foltz et al., 2021).

Among the largest landfills in the world, about one-half receive both municipal solid waste and hazardous waste. This can have deleterious consequences on human health, considering that larger sites involve a high population, as shown in S1. For example, Williams et al. (2019) estimated that 0.4–1 million people die each year in developing nations because of diseases caused by improper waste management. Similarly, Kodros et al. (2016) found that uncontrolled waste burning is responsible for about 270,000 premature deaths of adult individuals whereas Vaccari et al. (2019) quantified that the death of about 9,000,000 people is directly related to several diseases caused by emerging pollutants that are released from municipal solid waste every year. Uncontrolled dumping and engineered/sanitary landfilling are still likely to remain the most common waste disposal option and remain so for the foreseeable future (Table 1).

3. Characteristics of leachate in municipal solid waste landfill

3.1. Leachate generation and characterization

Leachate is heavily polluted wastewater, mainly generated by the penetration of precipitated water through the municipal solid waste body. Also, the water originally contained in the waste contributes to leachate composition (Teng et al., 2021). The amount produced leachate can be greatly influenced by several factors including waste composition; climatic conditions; landfill management system; landfill liners and structure; and age (Renou et al., 2008).

As shown in Table 1, landfill and uncontrolled dump remain the first disposal option for waste. However, separate waste collection is realized in many countries, even if at different levels. This is expected to promote a better waste recovery strategy in the next future. At present, data about the specific volume of different wastes destined for landfilling (see Fig. 3) can help to have an idea of the potential pollutants that may be found and their possible trend. For example, the data evolution concerning landfill, available in the USA, shows that food is the most landfilled waste, with 35.3 million tons discarded in 2018, with a continuously increasing trend. Plastic volumes about doubled in 2018, corresponding to 26.97 million tons, if compared to data from 1990, corresponding to 13.78 million tons, but their trend is almost constant, and it is expected to decrease due to an increase in separate collection of packaging waste and the restriction about the use of petroleum-derived

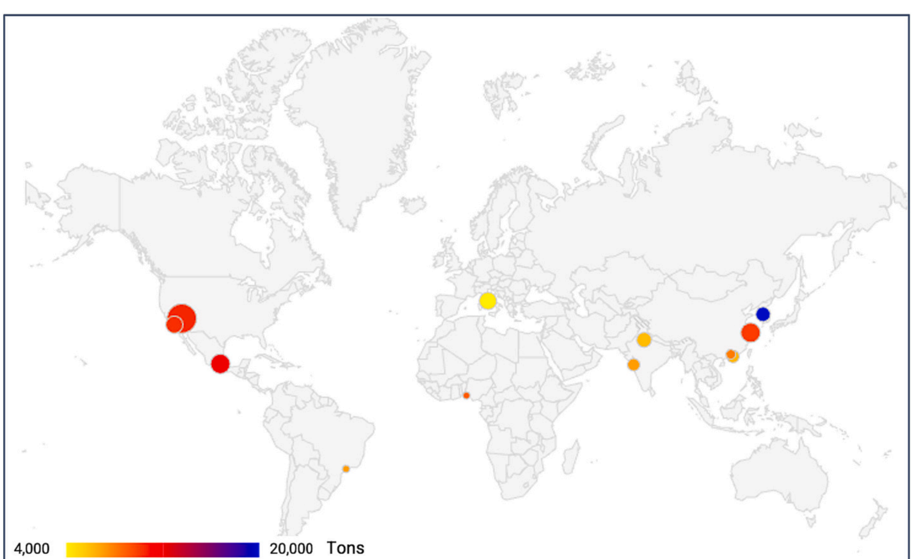


Fig. 2. World map showing the size of the largest dump sites globally (Worldatlas, 2022), which are proportional to the corresponding ball dimensions, with the daily volume of waste dumped in 2021, represented by ball colour. The Apex Regional site in Las Vegas is the largest existing landfill, covering a land of about 2200 acres. Unfortunately several of the existing and unregulated landfill sites are often located in the proximity of urban areas. Then it was estimated that about 60 million people in the world live <10 km away from the largest dumpsites. All the data, concerning the largest landfilling sites and the population in their proximity, are available as Supporting information (S1).

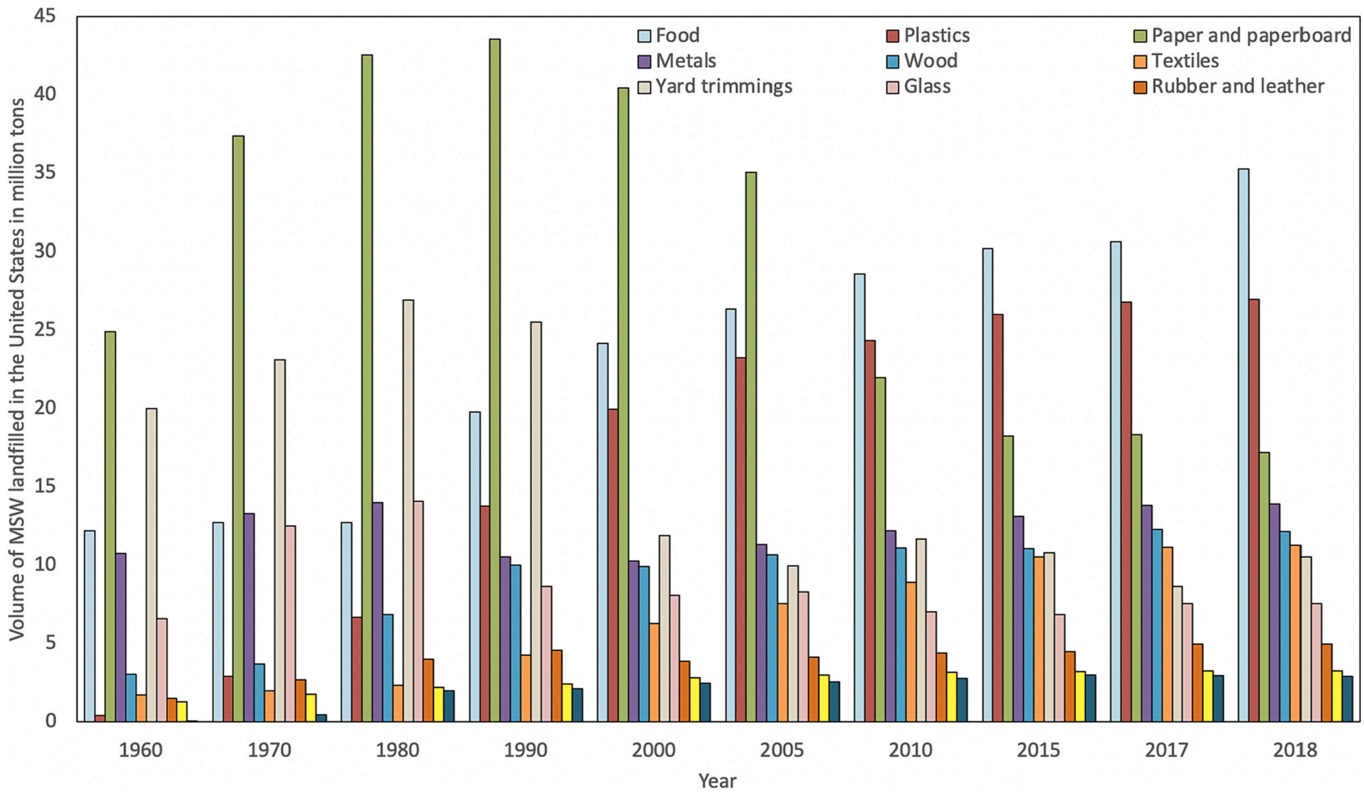


Fig. 3. Volume municipal solid waste landfilled in the USA from 1960 to 2018, by material (Source: Environmental Protection Agency (Environmental Protection Agency, 2021)). It is possible to highlight the decrease in paper and paperboard waste generation, with an increase in food and plastic waste. In 2018 the municipal solid waste generated in the USA was about 292 million tons. Of the landfilled wastes, most are plastics, inorganic waste, yard trimmings, and food waste. Last year's loss of landfill capacity with the consequent decrease in landfilling sites represents a problem for the USA, with detrimental effects on the environment.

materials. However, this has several implications in terms of released contaminants, such as microplastics.

Along with landfill gas, leachate represents the main pollutant stream generated from waste disposal (Brennan et al., 2016; Koda et al., 2017; Shen et al., 2018). Indeed, a series of potentially toxic compounds based on the original waste composition can be introduced into the landfill: for example, organic nitrogen, aromatic and sulfurous compounds, and hydrocarbons, including chlorinated, bromated, and fluorinated hydrocarbons. Due to the increasing use of a wide variety of new drugs and chemicals, nanoparticles and other microcontaminants can also be found in the waste. Some of these pollutants are partly or completely degradable and may end up in landfill leachate. It is extremely important to highlight that in some cases the degradation compounds are more toxic than the original products (Erythropel et al., 2014).

Common chemical pollutants found in leachate can be grouped into the following classes:

- Organic compounds detectable by the chemical oxygen demand or the total organic compounds, the volatile fatty acids and the fulvic-like and humic-like compounds resulting in more resistance to biological degradation;
- Macro inorganic ions as Ca^{2+}, Mg^{2+}, Na^{+}, NH_4^{+}, Cl^{-}, SO_4^{2-}, HCO_3^{-};
- Heavy metal as Cd, Cr, Cu, Pd, Ni, Zn, Hg, Fe, Mn, Co;
- Organic compounds, such as polyfluoroalkyl substances, polycyclic aromatic hydrocarbons like persistent organic pollutants and volatile organic compounds;
- Emerging contaminants include xenobiotic organic compounds such as aromatic hydrocarbons, phenols, chlorinated aliphatics, and pesticides, plasticizers, antibiotics, microbial contaminants;
- Microplastics.

The composition of the landfill leachate depends on the country/region, waste type, and age of the landfill. Table 2 reports the contaminants and their concentrations in landfill leachates concerning the country and waste type. Landfill age becomes a determinant factor in leachate composition due to the degradation phase that leachate goes through.

Furthermore, a great amount of unsorted wastes arising from different activities such as wastewater treatment sludge, personal and absorbing hygiene products, health care waste, drugs, faeces, electronic waste, oils, batteries and chemical products are often disposed of in dumpsites (Yukalang et al., 2018; Zhao et al., 2018). Such waste represents a major anthropogenic load of different classes of antibiotics and antibiotic-resistant bacteria (Liu and Wong, 2013; Threedeach et al., 2012). Notably, leachate can be an important source of antibiotic resistance genes and contains a plethora of emerging contaminants such as lincomycin, bisphenol A, caffeine, gemfibrozil, crotamiton, sulfamethazine, acetaminophen, diclofenac, salicylic acid, N, N-diethyl-m-toluamide and clofibric acid as well as perfluorinated chemicals into the environment (Eggen et al., 2010; Masoner et al., 2016; Yi et al., 2017).

Due to the degradation processes occurring in the waste mass, the age of the landfill is a major factor in characterizing the quality of leachate and is broadly categorized into three parts (Foo and Hameed, 2009): <5 years old, i.e. young, 5–10 years old, i.e. medium, and >10 years old, i.e. highly mature. Additionally, landfill leachate characteristics are also highly dependent on physiochemical parameters such as alkalinity, total Kjeldahl nitrogen, chloride, biological oxygen demand, suspended solids, conductivity, chemical oxygen demand, total dissolved solids, phosphorus, temperature, pH, temperature, as shown in Table 3.

There are several case studies available discussing the contaminants present in landfill leachate. One recent study conducted in China investigated the occurrence and distribution of emerging organic

Table 2
Contaminants in landfill leachates concerning the country and waste typology.

Country	Waste type	Contaminants and their concentrations	Reference
China	Municipal solid waste	Perfluoroalkyl carboxylic acids - 70 to 214,000 ng/L Polyfluoroalkyl substances - 30 to 416,000 ng/L	(Yan et al., 2015)
	Municipal solid waste	Microplastic - 4 to 13 items/L	(Su et al., 2019)
	Six different landfills in China	Total nitrogen - 950.0 to 3810.0 mg/L Cl^- - 219.0 to 18,145.4 mg/L SO_4^{2-} - 99.3 to 1320.7 mg/L COD - 1680.6 to 18,181.3 mg/L BOD - 25.5 to 7121.6 mg/L Ammoniacal nitrogen - 44.7 to 566.2 mg/L Total phosphorus - 1.0 to 19.0 mg/L	(Song et al., 2015)
Canada	Municipal solid waste incineration ash Unburned waste	Polyfluoroalkyl substances - 290 ng/L Polyfluoroalkyl substances - 11,000 ng/L	(Liu et al., 2022b)
United States	Municipal wastewater treatment plant waste	Polyfluoroalkyl substances - 19,800 to 48,700 ng/L Bisphenol A - 628 to 516,000 ng/L 3-Beta-coprostanol - 32,700 to 176,200 ng/L 4,4′-Bisphenol F - 1100 to 1280 ng/L Hormone - 86 to 341 ng/L Nonprescription pharmaceuticals - 1400 to 123,000 ng/L Prescription pharmaceuticals - 1620 to 42,700 ng/L	(Masoner et al., 2020)
	Heterogeneous mixture of municipal waste, construction debris, wastewater sludge (biosolids), and nonhazardous commercial and industrial waste	Plant/animal sterols - 1000 to 100,000 ng/L Nonprescription pharmaceuticals - 100 to 10,000 ng/L Prescription pharmaceuticals - 10 to 10,000 ng/L Steroid hormones - 10 to 100 ng/L	(Masoner et al., 2016)
	Municipal solid waste	Carbamazepine - 23 to 282 ng/L N, N-diethyl-m-toluamide - 31 to 143,500 ng/L Gemfibrozil - 8 to 2110 ng/L Perfluorooctanoic acid - 177 to 2500 ng/L Perfluorooctane sulfonate - 26 to 92 ng/L Primidone - 65 to 1000 ng/L Sucralose - 296 to 620,500 ng/L Sulfamethoxazole - 254 ng/L Trimethoprim - 64 ng/L	(Clarke et al., 2015)
Portugal	Municipal solid waste	COD – 5700 mg/L BOD - 400 mg/L Dissolved organic carbon - 2400 mg/L Total polyphenols - 750 mg gallic acid L^{-1} Total suspended solids - 130 mg/L Fe - 4.1 mg/L Zn - 0.7 mg/L As - 37.0 μg/L Pb - 28.5 μg/L Cd - 1.1 μg/L Cu - 46.9 μg/L Cr - 2.2 mg/L Ni - 3.0 μg/L	(Amor et al., 2015)
Turkey	Municipal solid waste	BOD - 211 mg/L COD - 800 mg/L PO_4^{3-} - 5.42 mg/L Ammoniacal nitrogen - 150 mg/L Ammonium nitrogen - 1100 mg/L Total dissolved solids - 11.84 g/L Salinity - 14.37 % Alkalinity - 1700 mg/L Cl - 10,000 mg/L Na - 1500 mg/L K - 2710 mg/L Hg - 5.9 mg/L Pb - 5.15 mg/L Cd - 4.66 mg/L Zn - 4.88 mg/L Ni - 4.42 mg/L Cu - 4.44 mg/L Cr - 3.78 mg/L Fe - 11.09 mg/L S - 17.54 mg/L Ca - 1.5 mg/L Mg - 3.26 mg/L	(Khanzada, 2020)
India	Municipal solid waste	Total dissolved solids - 2027 mg/L COD - 10,400 mg/L BOD - 1500 mg/L Cl^- - 660 mg/L Ca - 400 mg/L Na - 3710 mg/L K - 1675 mg/L SO_4^{2-} - 40 mg/L Fe - 11.16 mg/L Cu - 0.151 mg/L Ag - 0.035 mg/L Cd - 0.035 mg/L Cr - 0.021 mg/L Pb - 0.3 mg/L Zn - 3 mg/L Ni - 1.339 mg/L Nitrate - 22.36 mg/L Ammoniacal nitrogen - 1803 mg/L	(Naveen et al., 2017)
	Municipal solid waste	Total dissolved solids - 2322 mg/L Dissolved oxygen - 7.70 mg/L Cl^- - 1221 mg/L F^- - 1.65 mg/L NO_3^- - 66 mg/L BOD - 1335 mg/L COD - 8332 mg/L Ca - 350 mg/L Mg - 201 mg/L Na - 745 mg/L K - 1246 mg/L Cr - 1.77 mg/L Cu - 0.33 mg/L Fe - 5.40 mg/L PO_4^{3-} - 30.4 mg/L	(Mishra et al., 2019)

(continued on next page)

Table 2 (*continued*)

Country	Waste type	Contaminants and their concentrations	Reference
	Municipal solid waste	Total nitrogen - 1150 ± 20 mg/L COD - 8900 ± 120 mg/L BOD - 1800 ± 15 mg/L Total dissolved solids - 17,320 ± 30 mg/L	(Mahtab et al., 2021)
Pakistan	Municipal solid waste	Total dissolved solids - 1513 ± 8.03 mg/L BOD - 925 ± 8.09 mg/L COD - 1560 ± 8.11 mg/L Zn - 1.67 ± 0.08 mg/L Pb - 0.86 ± 0.05 mg/L Cu - 0.73 ± 0.03 mg/L Fe - 1.25 ± 0.07 mg/L Ni - 1.83 ± 0.09 mg/L	(Abbas et al., 2019)
Sri Lanka	Municipal solid waste	Dissolved oxygen - 0.98 to 1.12 mg/L BOD - 950 to 3230 mg/L PO_4^{3-} - 2.67 to 38 mg/L Nitrate–N - 67.2 to 2445 mg/L Nitrite–N - 0.276 to 2.14 mg/L Benzene - 1.3 to 12.12 μg/L	(Jayawardhana et al., 2019)

COD = chemical oxygen demand, BOD = biological oxygen demand.

contaminants in landfill leachates and their ecological risks in the surrounding environment (Wang et al., 2021). This was the first paper reporting the role of landfill leachates in the emerging organic contaminants presence in both aquatic and soil environments in East China. The results suggested that the individual emerging organic contaminants posed medium to high risks to aquatic organisms in groundwater while negligible effects were found for human health through consumption of vegetables. Another study conducted in Iran analyzed the impact of landfill leachate on groundwater quality and identified heavy metal concentrations, COD, BOD5, TOC, EC, NO3-, Cl-, TDS, and pH as sources of contamination, by using a special statistical approach including a main factor (age of leachate) and a subfactor (distance from the source of pollutant) (Vahabian et al., 2019), showing that a high variation in the contaminants (i.e., organic compounds, salts, and heavy metals) can be related to leachate age. A case study conducted in Kuwait examined the chemical characteristics of leachate and the mechanism of leachate formation in two unlined municipal solid waste landfills (Al-Yaqout and Hamoda, 2003). The authors proposed a water balance at the landfill site with a dedicated model designed to account for leachate generation due to the rising water table, capillary water and moisture content of the waste.

Hence, landfill leachate composition is tremendously heterogeneous and remarkably variable. Consequently, numerous strategies such as waterproofing layers, cover layers and lining are being used to control water entry into the landfill and thus the amount of leachate produced (Dajić et al., 2016). Since leachate is a complex mixture of diverse contaminants, the characterization of the quality of landfill leachates is a challenging task (Boonnorat et al., 2018), particularly for emerging pollutants such as the ones discussed in the following paragraphs.

3.2. *Composition of leachate/fate and transport of contaminants in landfill leachate*

3.2.1. *Dissolved organic matter*

Landfills can be considered bioreactors in which chemical, biochemical, and physical processes take place. These mainly depend on the incoming waste composition, moisture, and climatic conditions. The municipal household solid waste is constituted by some substances, which are very similar worldwide, although they vary in amount. Approximately 75–85 % of the refuse destinated for landfill is biodegradable, as shown in Fig. 3. Biodegradable waste can be divided into two fractions: the readily biodegradable part, corresponding for example to food and garden wastes, and a moderately biodegradable portion, for example, wood, paper, and textiles. In a landfill, organic waste is largely degraded under anaerobic conditions. Fig. 4 shows the chemistry of major organic material anaerobic degradation pathways (Renou et al., 2008) occurring in a sanitary landfill, which can be divided into four phases: hydrolysis, acidogenic fermentation, acetogenic fermentation, and methanogenesis.

The first phase of biochemical degradation is fundamental for the reduction of complex organic matter, such as lipids, carbohydrates, and proteins into fatty acids, glycerin, saccharides, and amino acids. The hydrolysis phase is the process with the lower rection kinetics, then it regulates all subsequent reactions, slowing down the entire process.

In the second phase, which is defined as the acidogenic phase, biodegradable organic matter supports anaerobic fermentation, which produces volatile fatty acids as the main fermentation products. The process is enhanced by water presence, due for example to waste moisture or rainfall events, leading to the production of high quantities of volatile fatty acids, as much as 95 % of the organic content. In the acetogenic phase, methanogenic microorganisms develop, and the volatile fatty acids are converted to biogas. i.e. CH_4, CO_2. Aromatic hydrocarbons can be converted into acetic acid if they contain oxygen. The leachate organic part becomes dominated by refractory non-biodegradable compounds such as humic substances. The last phase of anaerobic degradation is methanogenesis: methane and carbon dioxide are produced, starting from hydrogen and acetic acid as a substrate. The presence of water vapor depends on the process efficiency.

Several microbial groups take part in the rate-determining steps of fermentation and methanogenesis. In these chemical reactions, organic parts are the main electron donors (predominant in the landfill environment). Carbon sulfate and dioxide are the major electron receptors.

Degradation of organics needs more long time if the process occurs in the solid phase in comparison to the liquid one, as the organic compounds need to migrate from the solid phase into the liquid phase by means of diffusion and dissolution. The dissolved organic matter in the

Table 3
Range of chemical and physical composition of landfill leachate (mg/L), adopted from Kjeldsen et al. (2002).

Parameter	Range	Parameter	Range	Parameter	Range
pH	4.5–9	Organic nitrogen	14–2500	Potassium	50–3700
Specific electrical conductivity (μScm^{-1})	2500-35,000	Total phosphorous	0.1–23	Ammonium nitrogen	50–2200
Total organic carbon	30–29,000	Chloride	150–4500	Calcium	10–7200
BOD	25–57,000	Sulphate	8–7750	Magnesium	30–15,000
COD	140–152,000	Hydrogen bicarbonate	610–7320	Iron	3–5500
BOD/COD	0.02–0.80	Sodium	70–7700	Manganese	0.03–1400
Silica	4–7000	Arsenic	0.01–1	Cadmium	0.0001–0.4
Chromium	0.02–1.5	Cobalt	0.005–1.5	Copper	0.005–10
Lead	0.001–5	Mercury	0.00005–0.16	Nickel	0.015–13
Zinc	0.03–1000				

BOD = biological oxygen demand, COD = chemical oxygen demand.

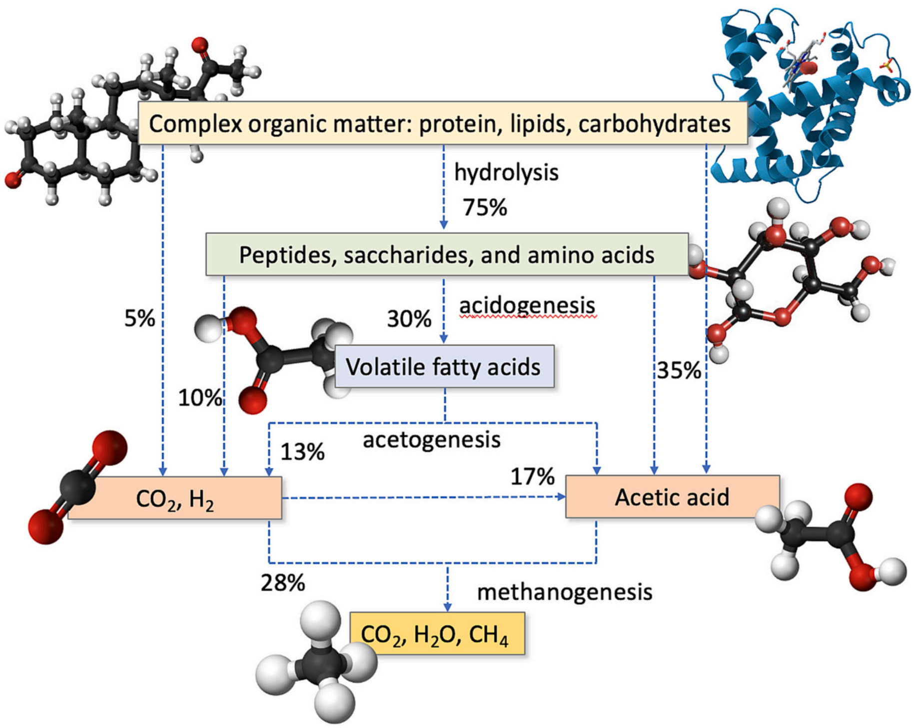


Fig. 4. Predominant decomposition pathways for common organic waste (adapted from Pazoki and Ghasemzadeh (2020)) with their per cent efficiency. In the first phase, large molecules are broken by hydrolysis into oligomer or monomeric units; for example, proteins are broken into peptides and lipids are converted into glycerol and fatty acid. In the second stage, hydrolytic products are fermented to volatile fatty acids such as propionate, valerate, and acetate, by the acidogenesis reactions. The following step, which is named acetogenesis, consists of the acid phase products conversion into hydrogen and acetates. In the methanogenesis phase, methane can be produced either by carbon dioxide reduction or acetic acid fermentation. Therefore, the precursors for methane formation are carbon dioxide, hydrogen and acetic acid, which are the products of the previous phase.

leachate represents the organic matter fraction that can be passed through a 0.45 μm filtration membrane. It includes organic compounds with a wide range of molecular weights and sizes. As explained, this organic matter generally includes fats and compounds with a nature similar to humic acids and fulvic acids serving as the abundant source of microbiological processes, thereby considerably influencing the fate of organic contaminants in their vicinity (Lyngkilde and Christensen, 1992).

Therefore, the composition of leachate reflects a typical condition of the dominant biological processes in a given period. Then, the experimental determination of individual organic compounds in leachate is hard to realize and hence properties such as biological oxygen demand, chemical oxygen demand and total organic carbon and volatile fatty acids are typically used to characterize leachate samples (Luo et al., 2020; Wijekoon et al., 2022). Generally, the elevated levels of biological oxygen demand and chemical oxygen demand suggest the presence of high concentrations of dissolved organic matter in leachate. Further, a low biological oxygen demand/chemical oxygen demand ratio indicates a low amount of volatile fatty acids and higher amounts of humic and fulvic-like compounds (Adhikari and Khanal, 2015; Wijekoon et al., 2022).

Leachate analysis of the Mavallipura landfill in India with an intermediate leachate age (5–10 years) showed high chemical oxygen demand concentrations corresponding to ∼2000–3000 mg/L (Naveen et al., 2017). Also, a study conducted by Vithanage et al. (2017), for the analysis of dissolved organic carbon fraction in the leachate of Gohagoda dumpsite, Sri Lanka, showed recorded maximum values of 56,955 and 28,493 mg/L respectively for total organic carbon and dissolved organic carbon. They have extracted and purified the dissolved organic carbon fraction including humic acid, fulvic acid and hydrophilic fractions with the use of resin techniques. Recorded maximum values for total organic carbon and dissolved organic carbon were 56,955 and 28,493, respectively. It has been shown that the age of the landfill and the precipitation as the main factors which control the concentrations of the chemical oxygen demand and biological oxygen demand (Ma et al., 2022). Humic and fulvic-like substances can form complexes with metal ions. Therefore, recent attention has been paid to investigating the transport and complexation of toxic heavy metals with dissolved organic matter fractions (Wijekoon et al., 2022; Wijesekara et al., 2014).

3.2.2. *Antibiotics and antimicrobial resistance*

Antibiotics are extensively used as medicine to treat infections, livestock food additives in the management of animal husbandry (promoting growth in animal farming), and to prevent and treat plant and animal infections (Cabello, 2006; Singer et al., 2003; Zhou et al., 2013). As a result, the demand for antibiotics is exponentially increasing to confer all these applications, consequently leading to the release of huge amounts in the natural environment (Kokoszka et al., 2022). In recent years, the occurrence of antibiotics in the environment has gained the highest attention due to the leverage on the blooming of antibiotic resistance potentiating a global threat to public health (World Health Organization, 2015). An in-depth assessment of municipal solid wastes suggests the potential impact on the environmental settings at different amplitudes, as shown in Fig. 4.

Globally, the emergence of antibiotic-resistant bacteria and antibiotic resistance genes among microbial communities imposes a major threat to humans and the environment (Bengtsson-Palme and Larsson, 2015; Berendonk et al., 2015; Larsson et al., 2018; Martinez et al., 2015; Pal et al., 2015; Topp et al., 2018). Rather than the accumulation of point mutations, horizontal gene transfer plays a pivotal role in the propagation, multiplication, and progression of antibiotic resistance genes in the environment (Amaro and Martín-González, 2021; Anthony et al., 2020; Baquero et al., 2019; Baquero et al., 2021; Buta-Hubeny et al., 2022; Vrancianu et al., 2020). Improper, overused or poorly controlled usage of antibiotics without proper medication resulted in the development of antibiotic-resistant bacteria (Frieri et al., 2017) and the emergence of antibiotic resistance genes both in the human gut upon initial intake and into the environment after release (Bound and Voulvoulis, 2004).

Several studies over the past few years showed that antibiotics are hardly metabolized and then directly released with excreta into the environment which ultimately goes into the landfill (Dolliver and Gupta, 2008). Surprisingly antibiotics are regarded as both emerging contaminants (Das et al., 2019) as well as food pollutants (Cabello, 2006; Martinez, 2009). To date, the widespread occurrence of antibiotic resistance genes has been observed in plenitude directions including natural ecosystems and engineered ecosystems, like air, surface water, soil, wastewater treatment plants, resulting in the potential spread of antibiotic resistance genes in the environment and human microbes (Buta-Hubeny et al., 2022; Guo et al., 2017; Hubeny et al., 2021; Levin-Reisman et al., 2017). Although landfills play the central character in the management and treatment of solid wastes (Buta et al., 2021a; Chakravarty and Kumar, 2019), the leachate generated during the waste decomposition process is regarded as a major hotspot for transmission of antibiotic-resistant bacteria and antibiotic resistance genes and necessary efforts are to be taken to control it (Wu et al., 2015) for the betterment of the environment and society.

Although remarkable/noteworthy relationships have been found among municipal solid waste leachate, antibiotics, and the levels of antibiotic resistance genes associated, very limited research has been carried out in the light of their potential relation at the metagenomics level. Moreover, the composition, toxicity level, diversity, and identification of mobile genetic elements and antibiotic resistance genes in municipal solid waste landfill is still largely unexplored. Hence, research in this domain must be elucidated since municipal solid waste harbour a great number of different classes of anthropogenic compounds and antibiotics in large quantities (Anand et al., 2021). As a result of this, discarded antibiotics and the development of new antibiotic resistance genes significantly pass into the leachate which ultimately greatly pollutes nearby environments, as reported in Table 4. For instance, the study by Threedeach et al. (2012) showed 80.8–87.5 % of *Escherichia coli* isolates from leachate of anaerobic and semi-aerobic landfills in Thailand were highly resistant to four common antibiotics, tetracycline, doxycycline, cephalothin, and minocycline.

Moreover, discharge of treated landfill leachate is also chiefly correlated with antibiotic resistance genes introduction to the downstream ecosystem/surroundings as effluent receiving water, which ultimately leads to accelerating the increase and spread of antibiotic resistance genes concentration (Wang et al., 2020b). Surprisingly, the increase in the humic acid content greatly impacts the accumulation of the antibiotic resistance genes vitally (Yu et al., 2016). Besides, mobile genetic elements also harbour a greater degree of correlation/connectivity to antibiotic resistance genes in leachates (Wu et al., 2017). It has been documented that the numbers of bacterial 16S ribosomal RNA gene copies including *sulII*, *tetO*, *sulI*, *intl1*, *tetW*, and *dfrA* in landfill leachate have a strong positive correlation to developing antibiotic resistance (Wang et al., 2015; Yi et al., 2017).

According to the findings of Song et al. (2016) numerous physicochemical factors, including nitrate concentrations and moisture content in landfill waste, exhibited strong correlations with antibiotics and antibiotic resistance genes, claiming that chemical conditions significantly influence antibiotic and antibiotic resistance genes delivery in landfill settings. Recently, several studies have discovered that depositing waste containing fluoroquinolones and β-lactams into landfills could have a remarkable impact on the dissemination of antibiotic resistance genes and antibiotic-resistant pathogens (Su et al., 2017a; Wu et al., 2017; Wu et al., 2015; You et al., 2018).

Biocide utilization also contributed to the rise of multidrug resistance (Pal et al., 2015). As many studies indicated that antibiotic introduction has been found to influence the increased frequency of mutation and

Table 4
Range of the concentration of selected antibiotics in raw landfill leachate.

Group of antibiotics	Antibiotics	Concentration (ng/L) (Min. – Max.)	References
β-Lactams	Amoxicillin	nd[a]-1.5	(Rodríguez-Navas et al., 2013; Wu et al., 2017; You et al., 2018)
	Ampicillin	9–396	(Lu et al., 2016; Zhao et al., 2018)
	Cefotaxime	3–72	(Zhao et al., 2018)
	Cephalexin	3–1.5	(Chung et al., 2018; Wu et al., 2017; You et al., 2018; Zhao et al., 2018)
	Cephalosporin	12–537	(You et al., 2018)
	Penicillin G	22–160	(Behr et al., 2010; Rodríguez-Navas et al., 2013)
Tetracyclines	Chlorotetracycline	nd-497	(Su et al., 2017b; Wu et al., 2015)
	Deoxytetracycline	23–472	(Wu et al., 2017)
	Doxycycline	nd-228	(You et al., 2018)
	Doxycycline hyclate	nd-542	(Su et al., 2017b; Wu et al., 2015)
	Doxycycline hydrochloride	nd-2.37	(Zhao et al., 2018)
	Oxytetracycline	nd-3.25	(Su et al., 2017b; Wu et al., 2017; Wu et al., 2015; You et al., 2018)
	Tetracycline	1–19.0	(Su et al., 2017b; Topal and Arslan Topal, 2016; Wu et al., 2017; Wu et al., 2015; You et al., 2018; Zhang et al., 2016; Zhao et al., 2018)
Quinolone	Ciprofloxacin	5–4.48	(Wu et al., 2015; Zhao et al., 2018)
	Enrofloxacin	3–4.03	(Su et al., 2017b; Wu et al., 2015; You et al., 2018; Zhao et al., 2018)
	Norfloxacin	26–21.03	(Behr et al., 2010; Dai et al., 2015; Su et al., 2017b; Wu et al., 2015; You et al., 2018; Zhao et al., 2018)
	Ofloxacin	9–190	(Dai et al., 2015; Su et al., 2017b; Topal and Arslan Topal, 2016; Velpandian et al., 2018; You et al., 2018; Zhao et al., 2018)
	Sarafloxacin	nd-1.26	(Behr et al., 2010)
Sulfonamide	Sulfadiazine	15–29.2	(Behr et al., 2010; Dai et al., 2015; Müller et al., 2011; Peng et al., 2014; Su et al., 2017b; Sui et al., 2017; Wu et al., 2017; Wu et al., 2015; You et al., 2018; Yu et al., 2016; Zhao et al., 2018)
	Sulfadimethoxine	52–51.400	(Behr et al., 2010; Masoner et al., 2016; Masoner et al., 2014)
	Sulfamethazine	25–16.2	(Behr et al., 2010; Dai et al., 2015;

(*continued on next page*)

Table 4 (*continued*)

Group of antibiotics	Antibiotics	Concentration (ng/L) (Min. – Max.)	References
			Peng et al., 2014; Su et al., 2017b; Sui et al., 2017; Wu et al., 2017; Wu et al., 2015; Yi et al., 2017; You et al., 2018; Yu et al., 2016; Zhao et al., 2018)
	Sulfamethoxazole	1–8.5	(Behr et al., 2010; Chung et al., 2018; Clarke et al., 2015; Inam et al., 2015; Masoner et al., 2014; Su et al., 2017b; Sui et al., 2017; Wu et al., 2015; You et al., 2018; Yu et al., 2016; Zhao et al., 2018)
	Sulfathiazole	3–2.2	(Behr et al., 2010; Su et al., 2017b; Wu et al., 2015; Yu et al., 2016)
Macrolides	Erythromycin	12–40	(Chung et al., 2018; Lu et al., 2016; Masoner et al., 2016; Masoner et al., 2014; Wu et al., 2015)
Lincosamides	Lincomycin	1–6.8	(Behr et al., 2010; Inam et al., 2015; Yi et al., 2017; Zhao et al., 2018)
Chemotherapeutic	Trimethoprim	1–8.0	(Chung et al., 2018; Clarke et al., 2015; Masoner et al., 2014; Sui et al., 2017; Wu et al., 2017; You et al., 2018; Zhao et al., 2018)
Chloramphenicol	Chloramphenicol	10–879	(Behr et al., 2010; Chung et al., 2018; Su et al., 2017b; Wu et al., 2015; Zhao et al., 2018)

[a] nd: not detected.

recombination in bacteria through SOS response (Blázquez et al., 2012; López et al., 2007). Therefore, the release of environmental bacteria to various concentrations of antibiotics is likely to create modifications with a high rate of genetic reorganization and this higher rate enables bacteria to quickly acquire favourable mutation and associated genetic modification, and upon contact with antibiotics may lead to the high frequency of genetic reorganizing bacteria establishment in the environment (Gillings and Stokes, 2012).

3.2.3. Microplastics

Undoubtedly, plastics are one of the most widely used materials globally. It has become an inextricable component of the material world, permeating everything from plastic packaging (bags and bottles), clothing, and equipment parts to construction materials. The world's plastic generation exceeded 348 million tons in 2017 from 2 million tons in the 1950s (Plastics Europe, 2018) and 359 million tons in 2018 (Plastics Europe, 2019). The generation of plastic waste is really blooming throughout the world and is a leading environmental problem as well as an emerging issue. Worldwide, microplastics are arising as a potential pollutant and received the highest attention from the municipality, public audience, and major research societies.

Microplastics and large microplastics are classified as particles of plastic waste material that have the highest size from 1 μm to 1 mm (Eerkes-Medrano et al., 2015; Koelmans et al., 2015; Marchesi et al., 2023) and from 1 mm to 5 mm, respectively (ISO, 2023). Recently, microplastics originating from municipal solid waste landfill leachate have been recognized as an emerging threat to the natural ecosystem (He et al., 2019). Solid waste landfills responsible for the release of a total of 17 types of microplastics from different landfill sites were recently documented (He et al., 2019). Moreover, several studies claim that microplastics have been enormously found in marine waters, freshwater bodies, and globally (Auta et al., 2017; Cole et al., 2011; Novotna et al., 2019). However, the concentration of microplastics in the leachate ranged between 0.42 and 24.58 items/L (He et al., 2019).

When compared to other emerging contaminants, such as pharmaceuticals and personal care products (PPCPs), per- and polyfluoroalkyl substances (PFAS), and nanomaterials, microplastics have several unique characteristics and importance: 1) Ubiquity and persistence, 2) Wide range of sources, 3) Regulatory action (Some regions and countries have introduced regulations and bans on microplastic-containing products, reflecting the growing recognition of their environmental and health risks), 4) Challenges in management (Unlike some emerging contaminants that can be effectively removed or treated through conventional water and wastewater treatment processes, the removal of microplastics from the environment poses significant challenges due to their small size and ubiquity (Anand et al., 2023)).

The source of microplastics in the leachate was the resultant of polythene $(C_2H_4)_n$ degradation after being buried in the landfill. Several studies in the recent past have mainly focused on the size of microplastics in different habitats which ranged from >1 μm (81–92 %) (Pivokonsky et al., 2018), <20 μm (96 %) (Triebskorn et al., 2019), <300 μm (61 %) (Leslie et al., 2017), <2000 μm (80 %) (Wang et al., 2017), and <500 μm (Yuan et al., 2019). The formation, accumulation, and transport of microplastics at the landfill site are long-term phenomena. Their study concluded landfill sites as the potential source of release of microplastics rather than serving as a sink for polyethene.

It has been reported that huge quantities of plastics are dumped into landfills every year. Landfilling, globally recognized as the most common waste management technique, was projected to store 21–42 % of the global plastic waste generation (Nizzetto et al., 2016). Currently, increasing awareness is also oriented to emerging contaminants, for example, antibiotics and flame retardants, in municipal solid waste landfills that are still not regulated and for which specific thresholds are not defined yet.

3.2.4. Macro inorganic contaminants

Macro inorganic components represent one of the four major groups of constituents in landfill leachates (Kjeldsen et al., 2002). Macro inorganic compounds in leachates mainly comprise different ions including Na^+, K^+, Ca^{2+}, Mg^{2+}, Fe^{2+}, Zn^{2+}, Al^{3+}, NH_4^+, Cl^-, HCO_3^- NO_2^-, NO_3^-, SO_4^{2-}, and PO_4^{3-} (Chakrabarty et al., 2016; Robinson, 2007; Shehzad et al., 2016). However, Li^+, Ba^{2+}, Co^{2+}, Hg^{2+}, BO_3^{3-}, S^{2-}, SeO^{2-}, and $AsO_4{}^{3-}$ could also be found in leachate as macro inorganic contaminants in minute concentrations and thus have less importance (Robinson, 2007). Macro inorganic contaminants typically appear in higher concentrations in leachates. However, the current stage of landfill stabilization, the composition of wastes, and other leachate characteristics affect the concentration of some macro inorganic compounds (Adhikari and Khanal, 2015; Wijekoon et al., 2022). For example, methanogenic leachate associated with elevated pH values favours the sorption of cations such as Fe^{2+}, Ca^{2+}, and Mg^{2+} into organic components that reduce the concentrations of cations together with the formation of precipitation with anions. Similarly, the concentration of SO_4^{2-} is decreased in the methanogenic phase since the reduction of SO_4^{2-} to S^{2-} via microbial activities (Adhikari and Khanal, 2015).

Some macro inorganic contaminants such as Cl^-, Na^+, and K^+ are considered conservative pollutants due to their negligible reaction abilities for removal through complexation, precipitation, or sorption (De et al., 2016). Chloride shows insignificant reactions for complexation, precipitation, and sorption (Kjeldsen et al., 2002) while Na^+ is not removed by complexation and precipitation (Erses et al., 2008). Other than being a conservative pollutant, K^+ is identified as a dilution indicator (Demirbilek et al., 2013).

Moreover, Fernández et al. (2014) indicated the usefulness of the concentration of macro inorganic compounds as indicators to access groundwater contamination with leachate plumes. In this study, strong distribution patterns of leachates in groundwater were identified with macro inorganic compounds such as sulfate, chloride, and iron.

3.2.5. Heavy metals

The municipal solid waste landfill leachate containing an unacceptable level of hazardous inorganic contaminants including metals and metalloids may eventually affect the nearby soil profiles, groundwater (O'Shea et al., 2018; Regadío et al., 2012), and the surface water environment. The mobilization of metals was influenced by the characteristics of leachate, in addition to the redox status and pH of minerals present in aquifers. Øygard et al. (2007) have shown the immobilization of unbound heavy metals present in landfill leachate to some extent after aerobic storage of 48 h. The rapid sorption of cations occurred because of increased pH nevertheless, there was very little adsorption of arsenic and antimony at such elevated pH indicating the enhanced mobilization of these metals from pore water to the nearby aquifer system.

However, arsenic as the inorganic contaminant of landfill leachate was recently presented by (Hu et al., 2019). The fate of leachate containing arsenic was assessed using a suitable extraction process and the overall content of arsenic in the landfill ranged from 15.26 to 38.41 mg/kg. There was an increase in the arsenic content downstream of the landfill up to 19 m however, subsequent zones displayed a decrement in the amount of studied metalloid. The content of arsenic varied significantly in different fractions like F1, defined as the exchangeable part, F2 corresponding to the oxidizable one, F3 which is the reducible fraction, and F4 the residual part. The allocation of arsenic in different fractions was affected by the presence of nitrate, carbonate, oxygen availability, the content of ferrous and ferric ions as well as crystalline forms of different mineral species.

3.2.6. Emerging organic contaminants

Leachates originating from different domestic and industrial landfills comprise a diverse array of potentially toxic recalcitrant organic contaminants (Hu et al., 2019) with ample chances of transport into nearby aquifers and subsequent contamination, if not managed properly. Organic pollutants in landfill leachate are found to be diverse in types. In general, the municipally derived landfill leachate consists of increased content of xenobiotics of organic origin apart from the organic materials resulting from the breakdown of landfilled organic components. Persistent organic pollutants such as albendazole, lincomycin, acetaminophen, diclofenac, naproxen bisphenol A, endosulfan and 2,4-Di-tert-butylphenol have been detected in landfill leachate (Aziz et al., 2018; Wu et al., 2021).

The utilization of organic components via microbial activities and transport of leachate from one region to another region of the aquifer may create different conditions varying from anaerobic like methane produced in the proximity of landfilled location, S, Fe, Mn, and NO_3^- reducing followed by oxidizing environment prevailing at the plume. The occurrence of different anaerobic and aerobic environments beneath the landfilled site is regarded as an important characteristic for evaluating the fate of organic contaminants present in the plume of leachates. The breakdown of little content of organic contaminants derived from landfill leachates is attributed to the microbiological processes and availability of electron acceptors in aerobic and anaerobic zones of plumes. Based on the analysis of leachate plume from a total of 75 points, Lyngkilde and Christensen (1992) concluded the contribution of microbe mediated biodegradation, in addition to adsorption in the different reduction-oxidation environments as the key factor regulating the fate of organic contaminants. Most of the contaminants could not be identified up to 100 m of the plume, however, one of the agrochemicals (herbicide) mecoprop ($C_{10}H_{11}ClO_3$) was observed to transport for long distances. Overall, the study signified the role of the ferrogenic environment in regulating the fate of contaminants.

Disposal of waste containing polyfluoroalkyl substances into landfills has been started a few decades ago however, increasing in quantities every year (Wei et al., 2019). Per- and polyfluoroalkyl substances are considered a major threat due to their environmental and human health-related concerns. The polyfluoroalkyl compounds in landfills potentially leach out and contaminate adjacent soil and groundwater resources (Wei et al., 2019). Among >5000 man-made polyfluoroalkyl compounds, perfluoroalkyl carboxylic acids and perfluoroalkyl sulfonic acids belong to the perfluoroalkyl acids group and are frequently reported in landfill leachates (Rahman et al., 2014). Abiotic leaching such as desorption and biotic leaching processes involves for release of polyfluoroalkyl substances from solid wastes (Allred et al., 2015). Short chained perfluoroalkyl acids consist of 4–7 carbon atoms that are highly soluble and persistent in aquatic medium thus, predominant in leachates compared to the longer chain perfluoroalkyl acids (Benskin et al., 2012; Hamid et al., 2018). The complex nature of the matrix, low biodegradability, high thermal stability, and high water solubility cause the removal of polyfluoroalkyl substances through conventional techniques ineffective (Yu et al., 2009). Therefore, additional, and expensive treatment technologies are required to remove polyfluoroalkyl substances from leachates.

Besides, local differences in the transformation of organic contaminants of aliphatic nature were also observed. Although the transformation of organic contaminants was observed for both in situ and laboratory-based experiments, in some instances, the in-situ transformations were more prevalent. The environmental conditions prevailing in leachate plume were identified as methanogenic, ferrogenic and nitrate-reducing (Lyngkilde and Christensen, 1992). The conversion of nitrophenol with a reduced lag phase was rapid under a reducing environment in the proximity of the landfill site, however, the transformation was remarkably slower in the oxidizing zone of the investigated plume.

In contrast, the conversion of phenol, C_6H_5OH, with very little, which was much prevalent for up to two months in the distal part of the plume characterized by nitrate-reducing, ferrogenic and manganogenic environments. The conversion of dichlorophenol was noticeably prominent under a highly reducing environment represented by reducing conditions, in the vicinity of landfill sites, having a lag phase reaching up to after 9 days. The differences in the transformation of the study's organic contaminants under in situ and laboratory experiments may be due to changes in the redox environment (Borch et al., 2010; von der Heyden and Roychoudhury, 2015).

The contaminant transformation may also be regulated by the presence of other ions in plumes, the chemistry of contaminants, the nature, and diversity of microorganisms involved as well as the reducing and oxidizing environments prevailing in the plume, aquifer, and sediments. Apart from the biological factors, abiotic factors may also participate actively in the transformation of contaminants (Kotthoff et al., 2019). The natural remediation of recalcitrant organic contaminants stemming from landfill leachate is presented by Baun et al. (2003). Reinvestigation of a previously studied landfill site, for a total of 49 groundwater samples with references to redox responsive substances and organic contaminants, surrounded by the leachate plume was carried out (Baun et al., 2003).

The characteristics of the studied plume were unchanged even after 10 years except for minor alterations in the content of chloride, organic contaminants, and non-volatile organic carbon. Most of the contaminants present in the leachate plume were susceptible to degradation except the mecoprop, benzene, and non-volatile organic carbon. In the leachate plume, toluene was degraded to benzyl succinic acid. The assessment of toxic organic contaminants in the plume could provide insight into the mechanistic details of the natural attenuation process. Reductions in the concentrations of perfluoroalkyl acids in landfill leachate as compared to raw samples during biological treatment have been reported by Yan et al. (2015).

Interestingly, the loss in perfluoroalkyl acids content was not the result of the biodegradation process, as the contaminant was recalcitrant to microbiological activities (Higgins et al., 2007). Since the vapor pressure of perfluoroalkyl acids degradation products was very little, the reduction in its concentration through volatilization was not possible. The decrease in perfluoroalkyl acids content was attributed to the sorption by activated sludge materials.

4. Environmental impacts, risks, and their assessment

Landfill leachate is a very complex high-strength wastewater, which contains suspended and dissolved materials such as heavy metals, inorganic salts, nutrients, microbial contaminants and various organic compounds removed from the decomposing waste in the landfill body (Arunbabu et al., 2017; Wdowczyk et al., 2022). The mixing of landfill leachate into groundwater, surface water bodies and soil undoubtedly generates environmental risks (Ashraf et al., 2019). The main categories that can be directly or indirectly affected are several: land, soil, water, air, climate, biodiversity, material assets, cultural heritage, landscape, and population and human health. Therefore, this section deeply discusses the adverse effects of landfill leachates on ground and surface water, soil physicochemical and biological properties, development of antibiotic resistance as well as risks of microplastics in leachates and negative impacts on ecology. Moreover, this section points out the important aspects of the Leachate Pollution Index, which has been recognized as a valuable tool to estimate the pollution threats from landfill leachates generated from different landfills/open dumps thus, an important implementation for policymaking.

4.1. Effect of municipal solid waste landfill leachates on groundwater and surface water bodies

Municipal solid waste landfills are considered to be important sources of groundwater contamination due to the leakage of leachate, a complex mixture of pollutants having high chemical oxygen demand, high ammonium nitrogen content, high heavy metal content and lasting toxicological characteristics (Li et al., 2014; Teta and Hikwa, 2017). Many studies have indicated that the main pollutants from landfills found in groundwater include chloride, sodium, and ammonium ions, total hardness, total dissolved solids, organic matter such as chemical oxygen demand, heavy metals (Azizi et al., 2015; Cheng et al., 2013), and phosphate (Milosevic et al., 2012; Smahi et al., 2013).

Hepburn et al. (2019) determined a maximum concentration of 5200 ng/L of per-and polyfluoroalkyl substances, which was the sum of 14 compounds, in a landfill site in Australia. The same study found that PFHxS has the highest concentration (2.6–280 ng/L) among 14 per-and polyfluoroalkyl compounds followed by perfluorooctane sulfonate, perfluorohexanoic acid, and perfluorooctanoic acid, resulting 1.3–4800, $0 \leq -46$, and 1.7–74, respectively. A positive correlation was reported for per-and polyfluoroalkyl substances and leachate contaminants like ammoniacal nitrogen and bicarbonate. Especially in unlined landfills, leachate can contaminate groundwater with potentially hazardous chemicals at concentrations that violate drinking water standards (Reyes-López et al., 2008).

Municipal solid waste landfill leachates can contaminate water resources through groundwater underflow, runoff, infiltration, and precipitation (Mor et al., 2006). Leachate disperses in both horizontal and vertical pathways due to dilution and advection (Abiriga et al., 2021a). After contaminants reach the groundwater, the pollutants in leachate can mix with the aquifers. For example, many simply constructed or non-standard landfills have resulted in groundwater contamination to different extents in developing countries; these landfills include the Henchir El Yahoudia landfill in Tunis (Marzougui and Mammou, 2006), Matuail landfill in Bangladesh (Azim et al., 2011), Ondo landfill in Nigeria (Akinbile, 2012), Mediouna site in Morocco (Smahi et al., 2013), Matang landfill in Malaysia (Zawawi et al., 2012), and the Suchi landfill in India (Bhalla et al., 2012).

About 75 % of the 55,000 landfills in the USA have polluted the water resources close to them (Lee and Jones-Lee, 1993). Moreover, in Spain (Regadío et al., 2012), Denmark (Milosevic et al., 2012) and Greece (Fatta et al., 2002), the groundwater near some landfills has been contaminated by nitrate, nitrite, and ammonium. The studies investigation revealed the availability of different organic wastewater contaminants in groundwater samples. The cholesterol was identified at a site upstream of the landfill sites and documented for its presence at all locations. The leachate plume generated from landfill sites was also noticed for the existence of insecticides and flame retardants. The sampling locations nearby were observed to have higher concentrations of contaminants as compared to those located at distant positions. The number of organic contaminants at sampled locations varied from a minimum of 4 to a maximum of 17. The presence of hazardous organic contaminants even after long-duration examination of landfill sites revealed the persistent nature and long-range transport of targeted contaminants in groundwater.

The toxic byproducts of the leachate are manifold, of which the heavy metals play a significant role as a contaminant of water pollution (Ebadi Torkayesh et al., 2019). Studies show that it is challenging to evaluate the behavior of heavy metals in leachate-polluted groundwater as they are strongly bound to microscopic colloidal matter and organic molecules (Matura et al., 2012). The effect of landfill leachate on groundwater characteristics concerning size and heavy metal content in colloids has also been investigated by (Zhai et al., 2019). The size of metal-bearing colloidal particles in groundwater downstream of the landfill site was $>$200 nm, however, at upstream locations, the size was $<$10 nm.

Furthermore, the concentration of iron was higher in downgradient sampling locations as compared to upgradient sites, suggesting the transport and storage of colloidal materials in an aqueous environment. The study revealed the contribution of colloids in controlling the fate and transport of particulate as well as inorganic contaminants. The analysis of collected leachate samples had higher values of electrical conductivity, nitrate, chloride, biological oxygen demand and chemical oxygen demand during dry environmental conditions. The biological oxygen demand and chemical oxygen demand values of water samples were higher than the prescribed limits indicating the employment necessity of appropriate remedial procedures to make the contaminated water suitable for intended purposes. The increased electrical conductivity and enhanced concentrations of important cations and anions in groundwater collected near the landfill site are also demonstrated by Ahamad et al. (2019). Overall, previous studies have claimed that leachates have four chief components which include heavy metals, nutrients, toxic organic compounds and volatile organic compounds (Arunbabu et al., 2017; Budi et al., 2016; Kumarathilaka et al., 2016; Moody and Townsend, 2017).

Moreover, the water resources are observed to be contaminated with high levels of leachate during the wet seasons of the year. The leachate contamination is often propelled by rainy weather through surface runoff and infiltration (Alemayehu et al., 2019). Furthermore, the leachate flow increases linearly with increasing precipitation (Yu et al., 2021). Therefore, the changes in the rainfall highly affect the amount and characteristics of the leachate (Abunama et al., 2021b). Usually, the contamination levels in the groundwater reduce with increasing downstream distances, from the landfill (Sharma et al., 2020). Compared to the proximity of the landfills, there is less influence of other attributes such as topography, type, and state of the landfill to the degree of leachate contamination of water bodies (Akinbile and Yusoff, 2011).

4.2. Effect of landfill leachates on soil physico-chemical and biological properties

The concentrated leachate that reaches the bottom of the landfill infiltrates through different soil layers before reaching the groundwater system. The infiltration of leachate not only contaminates the soil but also alters the physico-chemical and biological properties of the soil layers through which it passes (Ingle, 2022). Studies show that the surrounding soil environments closer to landfills are directly contaminated by leachate when the leachate is not properly treated (Gu et al., 2022). The passage of landfill leachate through soil layers can negatively affect the soils' engineering properties including factors like shear strength, and volume changes as well as chemical properties such as adsorption and retention of heavy metals (Emami et al., 2019; Ingle, 2022).

Industrial and household metal garbage such as light bulbs and electrical equipment are the major sources of heavy metal pollution in landfills (Adelopo et al., 2018; Jaishankar et al., 2014) which are major anthropogenic sources of soil contamination through landfill leachates (Ojuri et al., 2016; Pohrebennyk et al., 2017; Pohrebennyk et al., 2016). Hussein et al. (2021) indicated the impacted soils in landfills contained high concentrations of heavy metals compared to the natural soils and the soil samples taken from non-sanitary landfills were moderate to strongly polluted. A case study in Pune, India highlights that the soil samples near the MSW dumpsite are highly contaminated with heavy metals and organic pollutants when compared to the farthermost soil samples (Ingle, 2022). As landfill leachate undergoes both horizontal and vertical migration through the soil and it consequently contaminates multiple layers of soil (Gu et al., 2022).

Landfills are significant contributors to microplastic contamination in the environment, and the presence of microplastics in landfills is independent of the age of the landfill (Puthcharoen and Leungprasert, 2019). Microplastics are produced in landfills mainly due to the accumulation of large quantities of plastic waste from municipal and industrial sources in landfills (Kabir et al., 2023). The impact of microplastics transported via leachate in soil systems remains a largely unexplored (Bharath et al., 2022). The primary origins of microplastics in landfill leachate stem from two key sources: solid waste and by-products of wastewater treatment plants (Golwala et al., 2021). The distinct characteristics of microplastics can bring about changes in soil texures and structures, ultimately affecting the physico-chemical properties of the soil (Guo and Fei, 2023; Wan et al., 2019).

A recent study conducted by Wan et al. (2022) observed drastically high microplastics contamination in the soil underneath the landfill than the microplastics in leachates and groundwater. This study found 570 to 14,200 items/L of microplastics underneath soil while 3 to 25 items/L of microplastics in leachates and 11 to 17 items/L in groundwater in the landfill site. The majority of microplastics documented in this study belonged to polyethylene, polyethylene terephthalate, and polypropylene. Microplastics presence in the soil can reduce the soil adsorption characteristics, leading to increased bioavailability of hazardous inorganic and organic contaminants in soil (Hüffer et al., 2019). Studies further show that microplastic contamination reduces the residual moisture content of compacted soils, independent of microplastic size and concentration and the impact of microplastics on air-entry pressures in soil varies depending on the size of the microplastics (Xie et al., 2023). The modification of physicochemical properties in microplastics as they age in landfill leachate is a critical reservoir for both microplastics and antibiotic-resistant genes (Anand et al., 2021). The microbial community residing on microplastics includes antibiotic-resistant genes (Liu et al., 2023). Compared to the leachate, the bacterial community on microplastics displays a higher capacity for biofilm formation and pathogenic potential. Moreover, quantitative data on antibiotic-resistant genes indicates that microplastics exhibit a selective enrichment of antibiotic-resistant genes at ratios ranging from 5.7 to 103 times higher than leachate, and the ageing process of the leachate (and microplastics) further enhance the enrichment potential of antibiotic-resistant genes (Su et al., 2021).

Research findings revealed that the introduction of antibiotics has altered the microbial denitrification process, a dominant pathway in reactive nitrogen removal (Hinshaw and Dahlgren, 2013). The study by Wu et al. (2017) strongly supported the hypothesis that the acquisition of antibiotic resistance genes in the form of the mobile genetic element by antibiotic-resistant bacteria influences the denitrification process affecting nitrogen intermediates, causing nitrogen imbalance in the soil.

4.3. Antibiotic resistance and antimicrobial resistance

Most antibiotics are poorly absorbed in the guts of humans and animals and remain unchanged when excreted, resulting in as much as 30 %–90 % of compounds discharged via manure or urine and ultimately released into the soil environment. The prolonged usage of agriculturally important antibiotics likeazithromycin, tetracycline, aminoglycoside, and streptomycin, inducing the development of resistance has attracted national and international concerns (Topp et al., 2018). Illegal supplementation of antibiotics with animal feed also contributes to the excessive release of antibiotics into the environment through animal manure (ur Rahman and Mohsin, 2019).

The majority of antibiotics are released into the environment from the defecation of livestock and humans as their waste contains either non-metabolized antibiotics or active metabolites (Marx et al., 2015). The persistence of released antibiotics, antibiotic-resistant bacteria, and antibiotic resistance genes are noted to be responsible for their environmental dissemination. For instance, Buta et al. (2021b) emphasize the risk of antibiotic resistance gene accumulation in plants. Therefore, it is necessary to use the required doses of antimicrobials for livestock and human being to reduce the onset and spread of antibiotic resistance genes and antibiotic-resistant bacteria in the environment (Pruden et al., 2013; Williams-Nguyen et al., 2016).

According to previous studies, the overall concentration of antibiotics in municipal solid waste landfill leachate is comparatively higher than the total content of antibiotics at wastewater treatment plants (Wu et al., 2015). Thus, landfill leachate can be considered as a "sink and source" for antibiotics and antibiotic resistance genes posing a potential hazard to humans and surrounding ecosystem health. These antibiotics and antibiotic resistance genes diffuse to the surrounding soils and water bodies through a range of physical, biological, as well as microbial processes (Liu et al., 2022a).

Sanitary landfilling is the most frequent waste disposal method, and it involves disposing of antibiotics from various sources like pharmaceuticals, personal care products, and toilet papers from hospitals and households (Eggen et al., 2010; Threedeach et al., 2012). Municipal refuse and landfills released into the soil via leaching caused by environmental events such as rainfall cause their massive dissemination (Salleh and Hamid, 2013). The landfill leachate flow bearing antibiotic resistance genes and metals ultimately disseminate into the environment posing high risks to the ecosystem and human health (Xie et al., 2014). Young leachates could exhibit higher antibiotic concentrations, whereas old leachates may contain a greater amount of antibiotic resistance genes due to the high stability of physicochemical characteristics of older leachates (Wu et al., 2017). A case study in Shanghai, China reveals that sulfonamides, quinolones, and macrolide were found in higher levels in MSW leachate and all leachate samples examined consisted of antibiotic resistance genes (Wu et al., 2015).

Recent studies by experts provide evidence of the propagation of antibiotic-resistant bacteria and antibiotic resistance genes in a landfill area through horizontal gene transfer (Wu et al., 2017; Yu et al., 2016). Studies have concluded that the production and transmission of soil antibiotic resistance genes are affected mainly by the half-lives of antibiotics and their potential sorption to soil particles, which can induce selective pressures (Lau et al., 2017; Xie et al., 2018), the co-selection of antibiotic resistance genes with metal- and biocide-resistance genes in metal or biocide contaminated environments (Imran et al., 2019), and mobile genetic elements (Von Wintersdorff et al., 2016). Wu et al. (2015) showed that the presence of antibiotic resistance genes significantly correlates with heavy metal levels such as Cd and Cr in MSW leachate contaminated soils.

The release of antibiotics in soil and the formation of degradation products rely on the physicochemical properties and microbial activity of the soil, hence antibiotics may exert constant, longer periods of selective pressure among soil microbes (Pan and Chu, 2016). As an example, in the Great Lakes Basin of humid- temperate regions, the applied antibiotics were enriched, leading to their availability for several months and possibly for more than a single crop growing season (Marti et al., 2014). The environmental risk further increases when the treated wastewater or landfill leachate is mixed with river/canal water and reused for agricultural irrigation at places where freshwater availability is under scarcity (Pina et al., 2020). Some studies have indicated that the waste treatment plant may help to partially eliminate the antibiotics, antibiotic-resistant bacteria, and antibiotic resistance genes (Le-Minh et al., 2010; Michael et al., 2013).

Antimicrobial resistance is an emerging threat to human health as it results in significant challenge to a plethora of antimicrobial treatment regimens in use (McGowan Jr, 2001). Studies show that expired medicines and pharmaceuticals mostly are disposed municipal solid wastes (Okeke et al., 2022; Rogowska et al., 2019). Experiments conducted in Ghana have revealed that, when subjected to screening with certain antibiotics, Enterobacteriaceae, along with specific *Bacillus* and *Listeria* species isolated from soil and leachate samples from MSW landfills, exhibited resistance to antibiotics (Borquaye et al., 2019).

Besides, the emergence, survival, and proliferation of antibiotic-resistant bacteria carrying mobile genetic elements can present the risk of dissemination of antibiotic resistance genes within the diverse pool of soil bacteria and ultimately to human pathogens is now one of the prime concerns worldwide (Xu et al., 2021). Therefore, all management practices including manure-derived amendments should be taken into consideration during the development of policy and practice for mitigating the spread of antibiotic resistance in the natural environment.

4.4. Risk of microplastics

Landfills and open dump sites receive a substantial volume of plastic waste from industrial and household sectors, accounting for approximately 42 % of global plastic waste (generated in 2018) (Hahladakis et al., 2018). Plastic waste generation is accelerated throughout the world contributing more and more plastic waste to landfills. Microplastics originating from municipal solid waste landfill leachate are recognized as an emerging threat to the natural ecosystem (He et al., 2019). Microplastics released from municipal solid waste sites could potentially pose risks to both human and environmental health due to the capacity of microplastics to adsorb toxic and long-lasting hazardous chemicals (Silva et al., 2021). The concentrations of microplastics in leachates can vary, and the diversity of microplastic types in leachates is often linked to the kinds of plastic waste present in the respective landfills (Shen et al., 2022). The study conducted by Sun et al. (2021) detected respectively 11.4 μg/L and 235.4 item/L of microplastics mass and concentration in landfill leachates. A similar study from South East Europe revealed microplastic concentrations ranging from 0.64 mg/L to 2.16 mg/L in landfill leachate (Narevski et al., 2021). The concentration of microplastics in young landfills is higher than in old landfills, and specific polymers such as polypropylene, polystyrene, nylon, and polycarbonate contribute significantly to microplastic contamination (Singh et al., 2023). Because of the hydrophobic properties, microplastics have the potential to act as carriers of persistent hazardous chemicals. If not treated well (in the landfills or leachate), microplastics can easily contaminate the surrounding environment (Hartmann et al., 2017). Despite being an emerging contaminant, research on the levels of microplastics in landfill leachates remains limited (Silva et al., 2021).

The buoyancy effect resulting from low density coupled with uneven shapes of microplastics causes challenging situations when removing them through the sedimentation process. Therefore, contamination of groundwater and surface water sources with microplastics resulting from leachate leakages and environmental disposal of treated leachates is inevitable. Natesan et al. (2021) detected 2–80 items/L of microplastics in groundwater resources around Kodungaiyur and Perungudi municipal solid waste landfill sites in India. Microplastics in groundwater possibly create human health-related consequences over long-term use.

Microplastics occur in landfill leachates together with high concentrations of other contaminants including organic pollutants and heavy metals. Therefore, other pollutants could bind with microplastics while making it a vector and carried away into the natural environment with leachate discharge creating harmful environmental impacts (Su et al., 2019). Weathering of microplastics further increases their surface area and hydrophilic activity permitting the binding of wide varieties of organic and inorganic pollutants (Duan et al., 2021). Weathered microplastics can adsorb heavy metals and metalloids by creating hydrogen bonds, electrostatic interactions, and ion complexation (Dong et al., 2020; Wang et al., 2020c). Similarly, electrostatic interactions and hydrogen bonding permit the binding of hydrophilic organic pollutants into weathered microplastics (Liu et al., 2019; Wu et al., 2020). This binding process of organic and inorganic contaminants into microplastics induces environmental threats via increasing bioavailability and distribution of contaminants.

Studies showed the positive involvement of microplastics for antibiotic resistance genes in soil and aquatic environments. Su et al. (2021) found out bacterial communities associated with microplastics exhibit greater pathogenic potential and high biofilm formation compared with the microbial communities in leachates. The same study discovered a positive correlation between microplastics ageing with antibiotic resistance gene enrichment. A similar study conducted by Shi et al. (2020) observed an extended enrichment of antibiotic resistance genes in microplastics particle sizes between 200 and 500 nm. Furthermore, prolonged exposure to microplastics driven bacterial communities to be closely associated with antibiotic resistance genes. Exposure to microplastics further induces the generation of reactive oxygen species which alters the membrane permeability of bacteria making them susceptible to receiving antibiotic resistance genes through intra-bacterial community transferring of genetic materials. Therefore, thorough investigations of the presence of microplastics in leachate and factors affecting their mobilization would help mitigate the associated risks.

The removal of microplastics from leachate is a challenging process due to the diversity of leachates as well as microplastics (Chamanee et al., 2023). In addition to the existing microplastic removal methods, there is a pressing need to develop innovative technologies for microplastic removal. These approaches should be backed by rigorous data collection and supporting evidence and then integrated into waste management systems to effectively reduce the presence of microplastics (Rafiq and Xu, 2023). To address the environmental impact of microplastics in leachates, it is imperative to minimize the disposal of plastics in landfills. This can be accomplished by prioritizing strategies such as reduction, recycling, and waste-to-energy conversion. Furthermore, public awareness campaigns should be launched to encourage changes in consumer behavior, specifically emphasizing the avoidance of short-lived items, like single-use plastics (Silva et al., 2021).

4.5. Ecological risk

Leachate can pose ecological risks if not properly managed. As shown, leachate can contaminate surface water, groundwater, soil, and ecosystems due to the presence of various pollutants. For example, landfill leachate when accumulated at high levels can have consequent negative effects on the ecology and food chains such as genotoxicity in living organisms (Mukherjee et al., 2015). The genotoxic effects of leachate on DNA molecule alterations have been proven by many studies (Gajski et al., 2012; Phoonaploy et al., 2016; Promsid et al., 2015). Genotoxicity arises as a consequence of the presence of various contaminants in leachates and the continuous interaction among these contaminants (Kwasniewska et al., 2012). A recent study related to the genotoxic and oxidative stress potential of landfill leachate in rats has revealed alterations in the antioxidant status within the liver, kidney, and testes of rats subjected to landfill leachate exposure. Furthermore, the study detected the presence of specific toxic chemicals, elevated levels of heavy metals, and an increased concentration of microbes in the rats exposed to landfill leachate, thus showing the potential of genotoxicity to living organisms (Arojojoye et al., 2022).

Some prohibited chemicals like the organophosphate insecticide methamidophos were found in concentrated leachate (Wang et al., 2020a). Methamidophos affects the nervous system of living organisms and is therefore banned in many countries in the world (Tosun et al., 2001). Through long-term bioaccumulation, the emerging organic contaminants such as per- and polyfluoroalkyl substances found in the leachate may pose threats to aquatic organisms, plants, and subsequently to humans (Gunarathne et al., 2023; Wang et al., 2021).

On the other hand, the heavy metals present in the leachates could cause detrimental effects on soil and aquatic organisms. Heavy metals including, Cr (VI), Cd, As, Hg and Pb are considered non-threshold contaminants due to their highly toxic nature toward organisms and can produce lethal impact even at small concentrations (Jayanthi et al., 2016; Rahman and Singh, 2019). Their non-biodegradable nature and bioaccumulation through food chains provoke long-lasting ecological risks. Disruption of natural biological equilibrium and retardation of self-purification processes in nature were reported in response to heavy metal contamination through landfill leachates (Gworek et al., 2016; Öman and Junestedt, 2008; Talalaj, 2015). Findings from several studies concerning the ecological risk assessment of exposure to leachate highlight the significance of ongoing research into landfill leachates and the cumulative environmental risks they pose to neighboring ecosystems, as well as the health of humans and other organisms (Gholampour Arbastan and Gitipour, 2022; Qi et al., 2018; Rouhani et al., 2022).

The study conducted by Gu et al. (2022) in an informal landfill site in southwest China found altered microbial composition and co-occurrence patterns in vertical and horizontal surface soils impacted by landfill leachates compared to the uncontaminated soil. The micro-organism communities involved in carbon, nitrogen and sulfur cycles in contaminated soils showed a significant shift compared to the uncontaminated soil. The anammox and denitrification microbial communities dominated the contaminated soil while retarding the growth of aerobic chemoheterotrophy, and cellulolysis communities resulting hindered nitrogen fixation process. This kind of microbial community shift highly affects the typical ecological function in soil. Furthermore, the microbial communities of aquifers altered drastically due to the contamination of landfill leachate (Abiriga et al., 2021b).

In summary, the ecological risks include:

a) Water pollution: if leachate is not properly collected, treated, and managed, it can potentially contaminate surface water bodies, such as rivers, lakes, and streams, as well as groundwater resources. This can have detrimental effects on aquatic ecosystems, including fish and other aquatic organisms, by disrupting their habitats, impairing water quality, and affecting their survival and reproduction.
b) Soil contamination: leachate can also seep into the soil, potentially contaminating nearby soils and affecting the health of plants and other organisms in the soil ecosystem. Contaminated soil may lose its fertility, and pollutants in the soil can be taken up by plants, potentially leading to bioaccumulation and biomagnification in the food chain.
c) Ecological habitat disruption: leachate can impact the ecological habitats around landfill sites. Surface runoff from leachate-contaminated areas can potentially disrupt the natural habitats of nearby ecosystems, leading to changes in plant and animal populations, as well as alterations in nutrient cycling, soil structure, and other ecosystem processes.
d) Biodiversity loss: leachate pollution can negatively impact biodiversity by contaminating habitats, reducing the availability of suitable food and shelter, and causing direct harm to plants and animals. This can lead to changes in species composition and abundance, loss of biodiversity, and disruption of ecosystem functions and services.
e) Accumulation of persistent pollutants: some pollutants in leachate can persist in the environment for a long time and accumulate in the biota, leading to long-term ecological risks. This can affect the health and survival of organisms in affected ecosystems and potentially impact the overall ecological integrity of the area.

Due to the diversity of changing environments for example soil edaphic factors and groundwater, and the types of municipal solid waste leachate, it is challenging to predict the immediate ecological risks, magnitude, and likelihood of undesired effects of leachate in different regions of the world. Thus, long-term continuous surveys are needed to propose conclusions.

4.6. *Leachate pollution index*

The Leachate Pollution Index was established as a tool to estimate the pollution potential of leachates generated by open dumps and landfills at a particular time (Rajoo et al., 2020). It was formulated by the expert knowledge of 80 scientists considering 18 selected parameters out of 50 parameters identified by a questionnaire survey. In the formulation process of the Leachate Pollution Index, the selected parameters were weighted according to their significance. The significance of the parameters was rated using a scale of 1–5 (Kumar and Alappat, 2005). The Leachate Pollution Index parameters and their significance and assigned weights are listed in Table 5.

The Leachate Pollution Index calculation procedure requires three major steps: testing of pollutants in leachates, calculation of sub-index values, and sub-index values aggregation. Testing of leachates includes performing laboratory testing for the aforementioned 18 pollutants or alternative use of available data. The second step involves the computation of sub-index values/‘*p*’ values for each parameter considered using sub-index curves. Finally, the Leachate Pollution Index can be obtained by multiplying sub-index values with respective weightage denoted in Table 5 and calculating the weighted sum. There are two equations to calculate the Leachate Pollution Index based on the data availability of leachate parameters. Eq. (1) can be used if data for all 18 parameters are available and Eq. (2) is applicable when the values for certain parameters are absent (Kumar and Alappat, 2005).

$$LPI = \sum_{i=1}^{n} w_i p_i \tag{1}$$

$$LPI = \frac{\sum_{i=1}^{m} w_i p_i}{\sum_{i=1}^{m} w_i} \tag{2}$$

where,

w_i = weight of *i*th pollutant variable
p_i = sub-index score for *i*th pollutant variable
n = number of pollutants variables used to calculate the Leachate Pollution Index
m = number of pollutant parameters with available data

$\sum_{i=1}^{n} w_i = 1$ and $\sum_{i=1}^{m} w_i < 1$.

Table 5
The leachate pollution index is a measurement used to assess the potential pollution risk associated with leachate, which is the liquid that is generated when water passes through or comes into contact with solid waste in a landfill or other waste disposal site. The parameters used in the evaluation of the Leachate Pollution Index, their significance, and assigned weights are reported.

No.	Pollutant	Significance	Weightage
01	pH	3.509	0.055
02	Total dissolved solids	3.196	0.050
03	BOD	3.902	0.061
04	COD	3.963	0.062
05	Kjeldahl nitrogen	3.367	0.053
06	Ammonia-N	3.250	0.051
07	Total Fe	2.830	0.045
08	Cu	3.170	0.050
09	Ni	3.321	0.052
10	Zn	3.585	0.056
11	Pb	4.019	0.063
12	Total Cr	4.057	0.064
13	Hg	3.923	0.062
14	As	3.885	0.061
15	Phenolic compounds	3.627	0.057
16	Chlorides	3.078	0.048
17	Cyanide	3.694	0.058
18	Total coliform bacteria	3.289	0.052
	Sum	63.165	1.000

BOD = biological oxygen demand, COD = chemical oxygen demand.
Source: Kumar and Alappat (2005).

The lack of waste disposal regulations or the presence of weak regulations in many countries is resulted mainly due to inadequate information regarding leachate characteristics (Abu-Daabes et al., 2013; Ansari et al., 2018). Therefore, the Leachate Pollution Index can provide basic information for the understanding of the leachate pollution potential of landfills and open dumps, to establish rules and regulations for effective and appropriate waste disposal and leachate management. For example, phytotoxicity experiments showed that leachate can inhibit root and shoot growth, showing a statistically significant correlation can be established between toxicity and Leachate Pollution Index (Wdowczyk and Szymańska-Pulikowska, 2021).

Abunama et al. (2021a) conducted a meta-analysis to identify the variation of leachate characteristics of waste disposal facilities distributed throughout the world with the aid of the Leachate Pollution Index. This study found a significant variation of this index among leachates originating from waste disposal facilities located on different continents and highlighted its importance for policymaking decisions about waste disposal sites, throughout the world.

However, despite that Leachate Pollution Index can be a useful instrument to provide valuable information for the policy-making authorities and the public about the pollution threats of leachates from landfills, its' applicability to compare dissimilar landfill leachates that originated in different parts of the world is in doubt (Rajoo et al., 2020). The characteristics of leachates mainly alter with the composition of waste, which is drastically influenced by the development status of the country or the municipality or the industry if wastes disposed of were by-products of an industrial process (Ismail and Manaf, 2013). Therefore, the use of the Leachate Pollution Index could imply inaccurate representations when using it to determine the impacts of leachates generated from landfills located in different areas of the globe.

Therefore, some researchers are involved in developing more accurate indices based on the Leachate Pollution Index. For instance, Lothe and Sinha (2017) developed a mathematical model that is useful to calculate the real Leachate Pollution Index values more accurately, when the situation with the absence of some pollutant parameters. Moreover, Rajoo et al. (2020) introduced a modification called "Leachate Pollution Index for Developing Countries" by introducing more important parameters such as leachate volume and landfill liners to the original Leachate Pollution Index and it has proved its' high precision to evaluate the leachate pollution in developing countries compared to the original index. Furthermore, Abunama et al. (2021a) suggested the use of advanced laboratory analysis and modeling such as machine learning and AI to enhance the accuracy of the Leachate Pollution Index. Bisht et al. (2023) introduced the revised leachate pollution index (r-LPI). r-LPI is based on three distinct categories related to basic pollutants, heavy metals, and toxicants. The weighted additive function introduced in the r-LPI ensures higher sensitivity to the changes in subindex values and minimal ambiguity in calculating Leachate Pollution Index. Integrating the Leachate Pollution Index with various remediation techniques, such as phytoremediation, can aid in predicting the optimal leachate concentration for effective leachate treatment processes (Arunbabu et al., 2017). Fig. 5 summarizes the overall perspective of leachate contaminations through living communities.

5. Potential uses of landfill leachate

Landfill leachate can potentially be used for various purposes (see Fig. 6), depending on its quality and treatment. Leachate has been long studied to be used as a potential substrate for bio-electrochemical systems. Exceptional electrical conductivity generally associated with landfill leachates provides favourable conditions for use in bio-electrochemical systems (Iskander et al., 2016). These systems can be used to produce electricity and they can be decontaminated using electrochemical reactions and biological processes (Damiano et al., 2014; Iskander et al., 2016). Ongoing studies show that leachate can be used to extract a wide range of metals such as Fe, Mn, Cu, Zn, and Pb, as well through advanced bio-electrochemical systems (Nancharaiah et al., 2015; Wu et al., 2015). These works represent extremely interesting case studies because the extraction of metals from landfill leachate has the possibility to address the primary economic challenges associated with landfill mining. Some more specific examples of potential successful applications of landfill leachate as a sustainable resource show that some leachate derived from electronic scrap can be considered. For example, selective recovery of strategic metals like Co, Ni, and Li may be obtained starting from the leachate of waste generated during the manufacture of Li-ion batteries (Nguyen et al., 2014). Moreover, these studies have shown that metal concentrations within landfill waste exhibit significant variability, with rare earth elements (REEs) typically present at very low levels, often around 1 or 2 micrograms per gram of waste. This low concentration poses a challenge to the economic feasibility of recovery. Generally, metal concentrations of over 1 % are required to ensure a cost-effective recovery process (Umeda et al., 2011).

A recent work of Lee et al. (2022) reported that several factors may contribute to operational costs, including the expense of leachate collection, which amounts to $9.56 per cubic meter for treatment. Labour costs for plant operation are approximately $30, while electricity costs per hour total $0.1042. Additionally, the cost of recovery through chemical leaching is estimated at $1060 per cubic meter. Then, these studies also show that methods have certain limitations: physicochemical approaches can be energy and capital-intensive due to the expenses associated with chemicals, oxidants, and membranes (Lee et al., 2022).

Landfill leachate can also contain high levels of organic matter, which can be converted into biogas through anaerobic digestion. Biogas is a renewable and sustainable energy source that can be used for electricity generation or as a source of heat, providing an opportunity for energy recovery from landfill leachate. However, the anaerobic conversion processes used in biogas production contain limitations as resulting volatile fatty acids byproducts hinder the activity of methanogens which, limits the production of biomethane, the main energy source of biogas. In search of a solution, Pinpatthanapong et al. (2022) researched propionate-cultured sludge bioaugmentation with landfill leachates and achieved increased biomethane production rates and accelerated rates of micropollutant degradation. Another study carried out by Srivastava and Chakma (2021) utilized a dry tomb-bioreactor landfilling technique with leachate recirculation to maximize

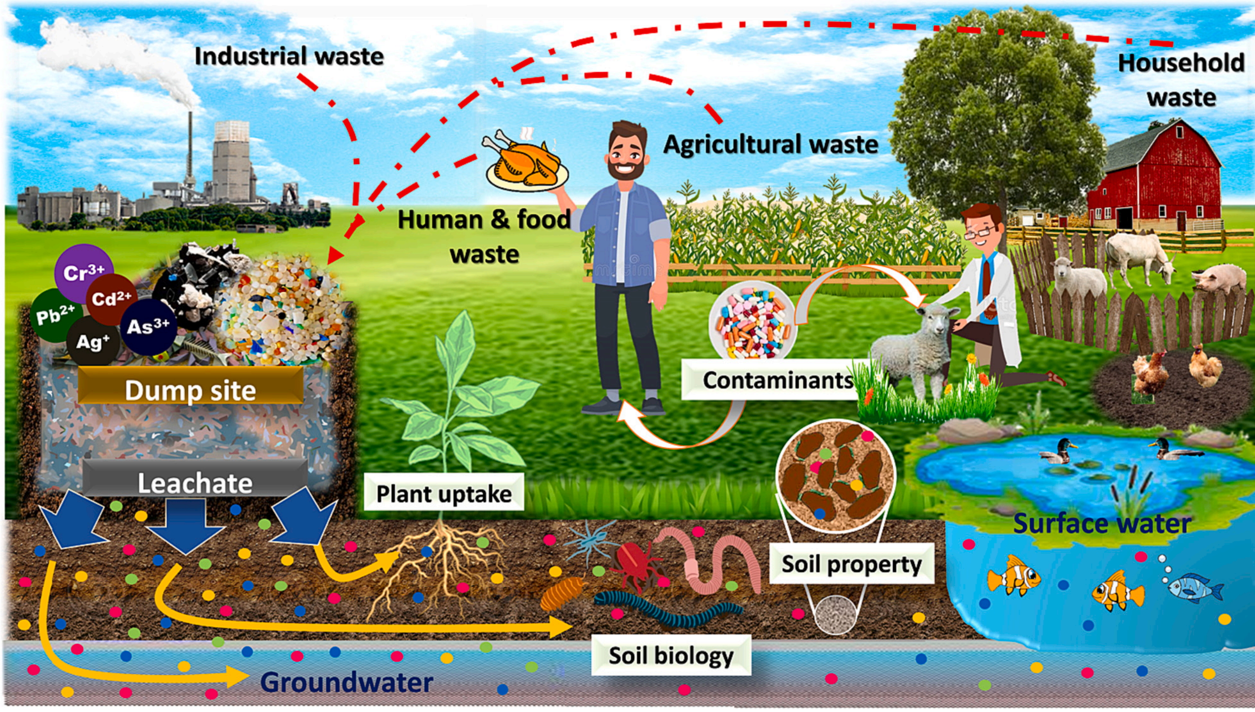


Fig. 5. The overall perspective of leachate contaminations through living communities, discussed in this work. Waste loads from industries, agriculture, and households ultimately end up in municipal landfill sites. Leachate generation occurs by the separation of aqueous components of wastes and mixing with soluble compounds including organic molecules and heavy metals. Once generated, drainage of leachates takes place downwards to the bottom of the landfill. Improper management practices often result in leachate contaminations in nearby soil and water resources threatening the stability of aquatic and terrestrial ecosystems as well as human health.

biomethane production while achieving decontamination of landfill leachates.

Due to the high nutrient content associated, landfill leachates provide an adequate medium for the cultivation of microalgae in the third-generation biofuel production process. Tang et al. (2023) highlighted the successful utilization of microalgae, *Chlorella vulgaris* and *Scenedesmus dimorphu* for lipid production while achieving simultaneous removal of nutrients from landfill leachates. The co-cultivation of both microalgae in landfill leachates diluted with recycled harvesting water yield 27.6 % of lipid content which is a primary source for biofuel production. A similar study conducted by Hu et al. (2021) cultivated microalgae *Chlorella vulgaris* and *Scenedesmus dimorphus* in landfill leachates pre-treated with NaClO. The study found that 10 % dilution of landfill leachate provides optimal growth of microalgae and the highest nutrient removal from leachate. Furthermore, Chang et al. (2019) used an advanced membrane photobioreactor technique to increase the efficiency of lipid production through microalgae and obtained the maximum lipid production of 404.98 mg/d. The harvested algal biomass can be converted to energy using different technologies such as biochemical conversion, thermochemical conversion, direct combustion, and chemical reactions (Alam et al., 2015).

Landfill leachates contain valuable nutrients such as nitrogen and phosphorus, which can be extracted and used as fertilizer in agricultural or other applications. Nutrient recovery from landfill leachate can reduce the need for synthetic fertilizers and contribute to circular economy practices. For example, the supply of ammoniacal nitrogen as an alternative source of nitrogen is the most attractive application of landfill leachates which may give a positive contribution to plant growth and development under nutrient-deprived conditions (Cheng and Chu, 2011). Bio-electrochemical systems can also be utilized for ammonia recovery from landfill leachates. Simultaneous ammonium migration along with electricity generation could be utilized to recover ammonium through stripping and transforming ammonium to ammonia with elevated pH conditions (Liu et al., 2021). The precipitation process could be employed to recover phosphorous which is typically present in landfill leachates with an elevated concentration (Wijekoon et al., 2022).

Depending on its quality and treatment, landfill leachate may be treated to a level that makes it suitable for certain non-potable uses, such as irrigation, dust control, or other industrial processes. This can reduce the demand for freshwater resources and provide a sustainable source of water for certain applications (Aronsson et al., 2010). A recent study shows a sustainable and economical application of garbage enzymes derived from fruit waste, for landfill leaching pretreatment (Nalladiyil et al., 2023). Microalgae were also applied as a pretreatment to remove nutrients and suspended solids from landfill leachate (Tang et al., 2023; Zhang et al., 2021a). Moreover, studies have utilized membrane technologies including reverse osmosis, ultrafiltration, microfiltration, and nanofiltration to recover purified water from leachates (Renou et al., 2008). Leachate concentrate or slurry generated after recovery water provides easy access for leachate management as well as further resource recovery such as heavy metals recovery through technologies such as hydrometallurgy (Gunarathne et al., 2022).

Finally, landfill leachate can also be used for research and monitoring purposes to better understand the characteristics and impacts of leachate pollution, as well as to assess the effectiveness of leachate management and treatment measures. This can help improve waste management practices and develop better strategies for minimizing environmental impacts.

It's important to note that the potential uses of landfill leachate depend on its quality, regulatory requirements, and local conditions. Proper treatment and management of landfill leachate are essential to protect the environment and human health and ensure that any potential uses are conducted in a safe and sustainable manner.

Treating emerging pollutants in landfill leachate requires specialized methods due to the presence of complex and diverse contaminants.

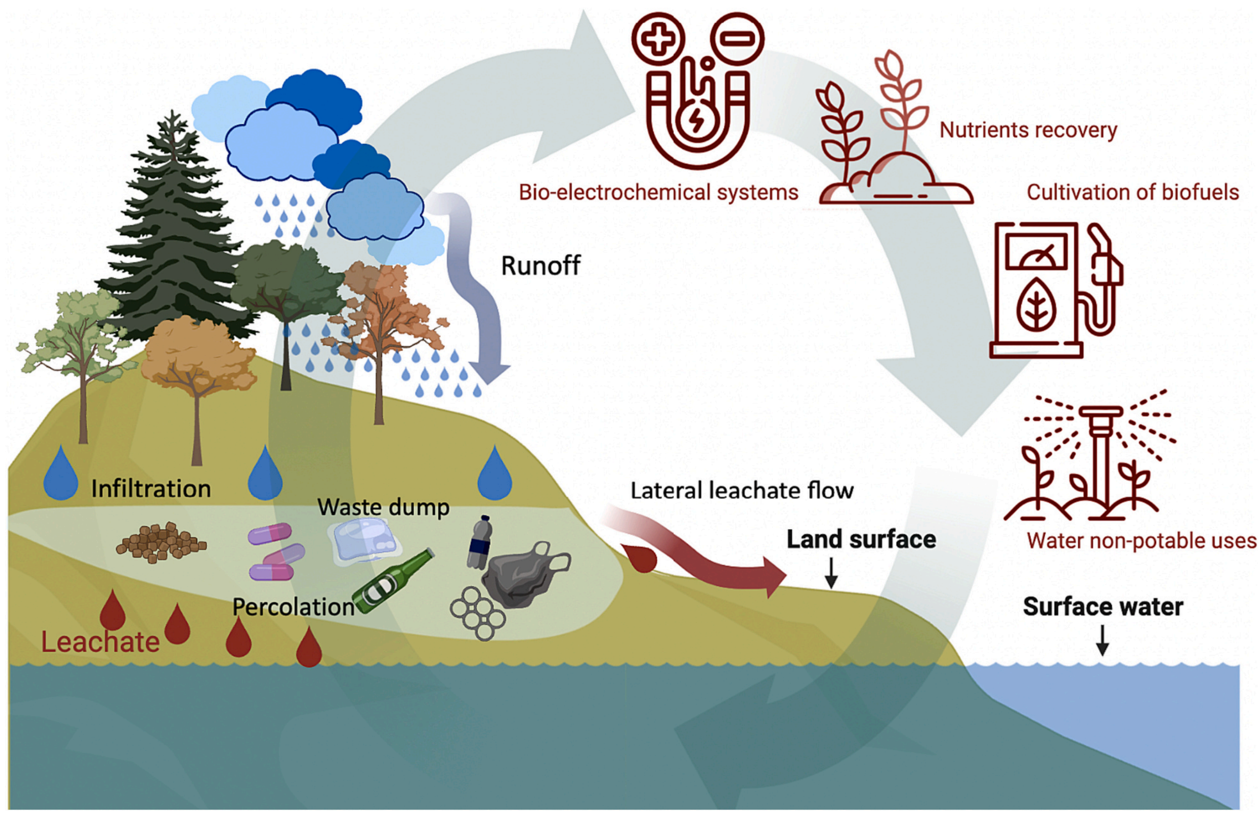


Fig. 6. The potential use of landfill leachate. Depending on its quality and treatment, landfill leachate can be reused for certain applications, like as a potential substrate for bio-electrochemical systems, and as a fertilizer, due to valuable nutrients such as nitrogen and phosphorus, which can be extracted. It can also result in an adequate medium for the cultivation of microalgae in the third-generation biofuel production process. Finally, it can be applied for certain non-potable uses, such as irrigation.

Some common methods to treat emerging pollutants in landfill leachate include: Advanced Oxidation Processes by UV/H_2O_2 and ozone (Gautam et al., 2019), Activated Carbon Adsorption (Yang et al., 2022), Photo-electrochemical methods (Divyapriya et al., 2021), Biological Processes (Villamizar et al., 2022), Membrane Filtration (Chen et al., 2022), and Chemical Degradation (Leung et al., 2022).

6. Research needs and future directions

Future research directions in landfill leachate studies include the need to better quantify the amount of chemicals sent to wastewater treatment facilities, potential impacts on treatment processes, and the significance of landfill leachate as a source of surface water contamination. In this frame, support may derive from the numerical simulation studies concerning the migration and transformation of pollutants in various soil types, that have not kept pace with the recent advancements in engineering technology. Therefore, it will be crucial for researchers to engage in discussions regarding models for understanding pollutant migration and transformation.

It will be also fundamental to design more suitable systems for landfill protection, experiment with combined treatment methods for efficient pollutant removal, and investigate the capacity for material reuse.

The long-term and generally uncontrolled emissions of biogas and leachate have been identified as the most critical problems associated with landfills. As such, it is essential to have protective barriers in place to efficiently safeguard the environment in landfill areas. To address the issue of landfill leachates, various lining systems made of materials such as polymers can be used. This is particularly crucial when the natural geological barrier of the landfill is inadequate in limiting leachate infiltration. The use of a bottom barrier can significantly decrease the migration of leachate. Additionally, a drainage system must be integrated with the barrier to reduce the hydraulic head of the leachate (Touze-Foltz et al., 2021). In this frame, it is fundamental to highlight that regular and continuous monitoring of leachate must be conducted, which can be achieved through dedicated collection systems such as a lysimeter. Moreover, the potentialities of implementing automated monitoring systems must be evaluated, with the aim to provide real-time data on leachate parameters such as pH, temperature, dissolved oxygen, conductivity, and specific ions. The aim is to generate alerts when certain thresholds or regulatory limits are exceeded, allowing for prompt action to be taken. The research must also investigate the synergic contribution of innovative approaches, such as the use of remote sensing technologies like drones or satellites equipped with sensors, which can aid in monitoring landfill sites from a broader perspective. These technologies may capture images, thermal data, or multispectral information to identify potential areas of concern or changes in leachate patterns. This monitoring process should be implemented even for older landfills, as studies have shown that the release of leachate can persist for several decades after their abandonment. By conducting regular monitoring of landfill leachate under various environmental conditions, both in situ and in laboratory settings, eco-friendly techniques to manage the global contamination problem caused by leachate from different waste deposited in landfill sites may be developed. To achieve proper and complete monitoring, the establishment of standard tests and procedures for emerging contaminants is crucial. For example, standard analysis protocols for novel per- and polyfluoroalkyl compounds in landfills are not well established even with the fast advancement of analytical techniques suitable for their detection (Abunada et al., 2020; Kucharzyk et al., 2017). Therefore, more research should be focused on the identification of emerging contaminants in landfill leachates and the development of standard test procedures to monitor them.

The sustainable utilization of landfill leachate is a crucial aspect that can mitigate environmental risks while generating economically valuable products and facilitating energy recovery from waste. However, further research is necessary to improve the efficiency of energy recovery processes, such as bio-electrochemical systems and anaerobic digestion. These methods have some drawbacks, such as low power generation, high costs, and microbial inhibition resulting from unfavourable conditions that arise during prolonged operation (Wijekoon et al., 2022). In addition, to establish effective biofuel production using leachates as a growth media, it will be mandatory to identify and study new microalgae species having a high tolerance toward high pollution levels including heavy metals of leachates.

Future research directions in landfill leachate treatment include the feasibility of extracting secondary metal resources, that are scarce in numerous countries, but are fundamental for the ecological transition needs. While metal recovery from leachate remains a relatively unexplored area, previous research has shown that metals can be successfully reclaimed from wastewater and aqueous solutions (Gunarathne et al., 2022). Challenges persist in the recovery of valuable materials within landfills due to uncertainties surrounding the concentrations and distributions of metals, which may not meet economically viable thresholds.

Indeed, great attention must also be devoted to the economic implications of landfill leachate management. Pre-treatment and recirculation may minimize the aftercare period length and achieve both sustainability and economic benefits (Wang et al., 2012). However, the cost of treating leachate varies depending on the treatment technology, but it may be reduced by implementing a circular economy and enhancing sustainable development (Ololade et al., 2019). The economic implications of landfill leachate management are also influenced by the production of renewable energy from landfill waste, which is emphasized in the recent global event COP 27 (Ghosh et al., 2023). Overall, the economic implications of landfill leachate management depend on the specific context and require a comprehensive evaluation of the life cycle cost and benefits.

If the utilization of leachate is still not an economically feasible solution, remediation is the next possibility. For that, both in situ and ex-situ approaches can be used according to the available space and the resources in the landfill site. However, in both cases, the utilization of sustainable and environmentally friendly techniques should be prioritized while deciding on the most applicable and viable remediation methods for certain landfills. Next studies should be devoted to the use of low-cost materials such as biochar for the adsorption of various pollutants in landfill leachates. In this regard, the use of biochar as permeable reactive barriers to treat landfill leachates before releasing them into the natural environment should be better considered (Gunarathne et al., 2018). Furthermore, some green technologies, such as phytoremediation with constructed wetlands be one of the most effective, environmentally friendly, and sustainable techniques to treat landfill leachates. However, it is necessary to extend research aiming to find out the most suitable plant species and filling materials to enhance the removal efficiencies for pollutants in landfill leachates (Wdowczyk et al., 2022).

Currently, the main available technologies for leachate treatment are devoted to treating organic substances in this waste, for example, volatile acids, but also ammonia is considered. However, leachate organic pollutants are highly concentrated and different in their typology, making their treatment extremely difficult. Accordingly, the main remediation available methods for landfill leachate are better suited for use for organic contaminants with an extremely high molecular weight, such as humic acids, or substances with an average molecular weight such as fulvic compounds. Thus, physic-chemical treatments are mainly addressed to the removal of organic substances that are largely refractory to bio-stabilization, except for heavy metals, salts, and ammonia. However, for substances with low molecular weights, no method can guarantee a specific removal efficiency, mainly due to the issue of their polarity and low dimensions. Then, to increase the removal efficiency of degradable organic substances, physical-chemical treatment, coupled with biological treatment, is a research field which must be better investigated (Cerminara and Cossu, 2018).

In this frame, phytoremediation has also proven its' potential to remove various types of contaminants from wastes including landfill leachates. Moreover, also in this case, the selection of plant type and construction technology should be decided after a careful study of the leachate characteristics as well as the geographical characteristics of the landfill area. However, further scientific studies should be conducted to increase the phytoremediation efficiency as well as the environmental adaptation of plant types. Transgenic plant technology is such a technique to improve phytoremediation by introducing genetic modification to plants (Gunarathne et al., 2019). Indeed, further research in this area is strategic to produce environmentally safe plant types that are capable of efficient remediation of landfill leachates.

The use of phytoremediation together with biochar has shown increased effectiveness in pollutant remediation (Paz-Ferreiro et al., 2014; Zhang et al., 2019). Moreover, despite advanced leachate treatment techniques such as ultraviolet catalytic persulfate use and advanced oxidation process coupled with biochar adsorption have been introduced in recent years for effective treatment of landfill leachate to minimalize the pollution capacity (Kwarciak-Kozłowska and Fijałkowski, 2021; Wang et al., 2021), further studies should be devoted to reducing operational costs before large-scale implementation.

It is also important to highlight that the removal of microplastics by the current leachate treatment facilities is still generally ineffective and poorly explored. In particular, only 50 % of fiber MPs are generally removed in biological treatment and advanced treatment (Zhang et al., 2021b). Thus, it is crucial to develop novel technologies and strategies to treat the microplastics in landfill leachate, as for example the separation of microplastics from the sludge dewatering liquor before its recirculation.

Landfill failure cases have been reported due to the breakdown of lining material and seeping of leachate through damages into the groundwater table (Pivato, 2011). This is especially a case found in abundant landfill sites because current materials used for lining have durability issues (Pivato, 2011; Suter et al., 1993). Therefore, it is a necessary area to improve through research by introducing novel materials that are durable and hazard-free. Simultaneously, the new research can be directed to introduce more environmentally safe and efficient landfill designs in terms of leachate collection and treatment.

Moreover, the hazards from landfill leachate can be caused by the gaps in current laws and regulations. Still, some hazardous chemical compounds such as per- and polyfluoroalkyl substances are disposed into landfills in many countries (Gunarathne et al., 2023; Stoiber et al., 2020). Therefore, more research should be directed to identify emerging contaminants, their environmental and human health-related risks, and their capacity to solubilize with leachates and escape to the natural environment while introducing new laws and regulations that should be in line with research findings. It is also important to note that landfill waste distribution is influenced by a wide range of factors and can vary greatly depending on local conditions, waste management policies, and infrastructure availability. Proper planning, site selection, and management practices are critical to minimize environmental impacts and protecting public health in landfill waste disposal.

Finally, while there may not be many specific examples of AI applications in landfill leachate management, it may be integrated into various aspects of waste management and environmental monitoring, with potential applications in managing landfill leachate, like as:

- **Predictive Modeling**: AI algorithms may analyze historical data on leachate quality and quantity, weather patterns, and landfill conditions to create predictive models. These models may forecast leachate production and composition, helping landfill operators plan for treatment and disposal.

- **Real-time Monitoring**: AI-powered sensors and monitoring systems can continuously collect data on leachate quality, flow rates, and environmental conditions within and around landfills. AI may analyze this real-time data to detect anomalies or trends that may indicate potential issues, such as leachate leakage or contamination.
- **Optimized Treatment**: AI may optimize the operation of leachate treatment facilities by adjusting treatment processes in real time based on the characteristics of incoming leachate. This can improve treatment efficiency and reduce operational costs.
- **Resource Recovery**: AI may assist in identifying valuable materials or resources within leachate that can be recovered, such as metals or nutrients, contributing to sustainability efforts.
- **Decision Support Systems**: AI-based decision support systems may help landfill operators make informed decisions about leachate management, including treatment options, disposal methods, and environmental impact assessments.
- **Environmental Risk Assessment**: AI may be used to assess the environmental risks associated with landfill leachate, including potential impacts on groundwater quality and nearby ecosystems. It may provide early warnings and recommendations for mitigating risks.
- **Data Integration**: AI may integrate data from various sources, such as monitoring sensors, satellite imagery, weather forecasts, and laboratory analysis, to provide a comprehensive view of landfill leachate dynamics and potential environmental effects.
- **Regulatory Compliance**: AI may assist in automating compliance reporting and ensuring that landfill operations adhere to environmental regulations and standards related to leachate management.

While specific examples of successful AI applications in landfill leachate management may still be limited, the potential benefits of using AI in this field are significant. As AI technology continues to advance, it is expected that more innovative solutions and success stories will emerge, leading to more efficient and environmentally responsible landfill leachate management practices.

7. Conclusions

This paper presents a comprehensive overview of the composition, environmental impact and ecological risks associated with landfill leachate, which includes a global overview of the main landfilling sites and the characteristics of the waste that generates leachate. The goal of these techniques is to minimize environmental risks while simultaneously generating valuable products and facilitating energy recovery from waste. Apart from the conventional remediation methods, other treatments should be developed to reduce operational costs before large-scale implementation, such as phytoremediation and physical-chemical treatment. Some of these technologies have shown potential in removing contaminants from landfill leachates, with the possibility of increasing the removal efficiency through a combination of different methods. Biochar is also a promising material for treating landfill leachates before releasing into the environment. Transgenic plant technology is another technique to improve phytoremediation by introducing genetic modifications to plants. Overall, these techniques offer a range of solutions to address the challenges of managing landfill leachates, while minimizing environmental risks and promoting sustainable waste management practices. In many cases, advanced treatment processes are necessary to purify leachate to meet the standards required for reuse applications. However, more scientific studies are necessary to increase the efficiency of phytoremediation and the environmental adaptation of plant types.

In the frame of the current need for the reduction of carbon footprint and increase the values of wastes, this paper also gives an overview of the possibilities for reusing landfill leachate, although it requires careful treatment and consideration due to its potential contaminants. These concerns the possibility of extracting and reusing strategic metals and organic matter. The use of irrigation is also proposed. It's important to note that the decision to reuse landfill leachate should be made cautiously, with a thorough understanding of its composition, treatment requirements, and potential risks to the environment and public health. Regulatory compliance and monitoring are essential to ensure that the reuse is carried out safely and sustainably. Finally, this work addresses future research directions in landfill leachate studies, with great attention to the more recent scientific updates concerning AI and the possibilities that it offers for landfill leachate management, studies, and applications.

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2023.169026.

Funding

This study was carried out by the University of Brescia within the MICS (Made in Italy – Circular and Sustainable) Extended Partnership and received funding from the European Union Next-GenerationEU (PIANO NAZIONALE DI RIPRESA E RESILIENZA (PNRR) – MISSIONE 4 COMPONENTE 2, INVESTIMENTO 1.3 – D.D. 1551.11-10-2022, PE00000004). This manuscript reflects only the authors' views and opinions, neither the European Union nor the European Commission can be considered responsible for them.

The authors used biorender.com for some figures' design.

CRediT authorship contribution statement

All the authors of this manuscript have substantially contributed to the concept, literature mining, writing and methodology of the review, provided critical feedback and revised the manuscript critically. All authors contributed to the writing or revision of the final manuscript. **VG** and **AJP** contributed to the study idea, manuscript structure, and literature survey, wrote, and prepared the original draft, created tables, arranged references and the final draft. **AZ** literature survey, wrote and prepared the original draft, created tables, drawings, data compilation and arrangement. **AUR**, **MV**, **FDM**, **AP**, and **EK** reviewed the write-up, manuscript editing, tables preparation, suggestions, critical assessment, and finalized the draft. **EB** conceptualized the study idea, manuscript structure, investigation, methodology, created figures, writing-review, and editing, supervision, final draft. All authors have read and approved the final version of the manuscript for submission to this journal.

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.

Acknowledgements

The authors are thankful to their respective departments/institutes for providing space and other necessary facilities which helped to draft this manuscript. The authors thank Vecteezy.com for the images.

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