From:	Richard Honour
To:	Benton Public Comment
Cc:	Debbie P; mailto:writerguy@writerguy.com; mailto:clearwater@peak.org; mailto:mayeager@gmail.com; Philipp Schmidt-Pathmann; Richard Honour
Subject:	I oppose LU-24-027
Date:	Thursday, April 10, 2025 6:35:31 PM
Attachments:	LandfillLeachates.04.10.25.pdf

To: The Planning Commissioners of Benton County, Oregon
My opposition to LU-24-027 - Expansion Application for the Coffin Butte Landfill – 04.10.25

Landfills in general are far out-of-date by plan, design, and intent, and are dysfunctional facilities by default that were established historically to receive municipal, industrial, medical, construction and other forms of solid waste, aka, garbage, as generated by society, but without any plan or intent for matters of Public Health, Environmental Health, or Beneficial Reuse.

No matter the name or title of the waste sources or materials, or the intent of the disposer, land-disposal or land-filling of any form of solid waste into any landfill or water environment, damages any or all of air, soil, water and life forms, including Humans, for land-disposed solid waste is nothing other than land-disposal of toxic waste material.

Landfilling or any other form of land disposal of solid or other wastes represents an out-of-date practice based on expedience and convenience, rather than on the critical factors of Human and Environmental Health. Alternatives must be explored.

Benton County's "2040 Thriving Communities Initiative's Core Values, Based on the Principles of Equity & Health," strives to ensure a solid basis for the establishment and sustainability of vibrant, safe and livable Communities that foster a high quality of life, assuring careful attention to the core values of Human and Environment Health, biodiversity and sustainability, free from the risks, threat and harm that may be incited by natural or Human-incited practices or disasters that may threaten or compromise the well-being of communities, as a direct function of policies, practices and operations of communities, including landfills.

In this light, I strongly oppose LU-24-027, the application to expand Coffin Butte

Landfill, as it will surely compromise and otherwise seriously interfere with the character and quality of the landfill area, as well as of adjacent properties and the natural environment.

A landfill that may be extant or planned imposes an undue burden on opportunities for public improvements, facilities, utilities, or services that are available or that may be needed for future growth in the core or adjacent regions. Growth and expansion of communities and their essential infrastructure must be anticipated or forevermore precluded from realization.

Any landfill by any design or engineering principle interferes with immediate and adjacent lands by the very nature of its toxic, odorous and otherwise noxious gasses, disturbances of local wildlife patterns, highly toxic landfill leachates, and importantly, the endless disturbances incited by intense truck traffic that disrupts the local environment with its inherent pollution, noise, and noxious fumes. Such interference would be an added burden, compromising a fully earned and anticipated safe and comfortable lifestyle.

It is for certain that the proposed expansion of the Coffin Butte Landfill would be contradictory to, and in conflict with, the stated and approved criteria in the County's Comprehensive Plan and Development Code. It is clear, based on the toxic content of the waste materials in the current landfill alone, much less when considering the porous nature of the underlying aggregate geology, plus proximity to the critical waters of the adjacent Willamette River and connecting surface and ground waters, that an expansion of the Coffin Butte Landfill will impose an undue burden on life in Benton and surrounding counties.

Attached for your information is a document I prepared that includes two new and highly attenuated peer-reviewed science articles (Pages 1 and 24) with intent to enlighten the Planning Commision members on some of the toxic consequences associated with landfills anywhere on Earth. I edited the content of the attached file with intent to make it easier to explore and digest, hopefully to reveal just some of the toxic consequences associated with any landfill, much less with an aged active landfill that seeks expansion and continuance of the release of its toxic loads to the near and adjacent environments. The included references are attainable by the embedded URL links within the text.

If you may ever seek a full presentation on the core issues of landfill toxics, I will be pleased to make such presentation as may be desired.

Sincerely, Richard C. Honour, PhD
Environmental Scientist: Specialist in Wastewater Treatment Plants and toxic wastes, as well as Landfills, Landfill Leachates, Landfill Gasses and compromised adjacent environments.

Landfill Leachates:

The leachates from nearly all landfills are known to contain toxic chemicals, metals and infectious disease agents, both bacterial and fungal.

Recent studies (Presented as Numbers 1 and 2 below and on Pages 1 and 24) analyzed the hazardous components of landfill leachates, highlighting the toxic chemicals, heavy metals, and pathogenic microorganisms. Below are the brief summaries of thw two scientific articles that discuss the toxic content of nearly any landfill leachate from nearly any municipal landfill:

1. A 2023 comprehensive review published in *Environmental Technology & Innovation* details that landfill leachate contains toxic organic and inorganic pollutants, heavy metals, and ammonia. The study discusses the environmental and health problems caused by these contaminants and evaluates current biological and physiochemical treatment methods. ScienceDirect
2. A 2024 review in *Science of The Total Environment* explored the environmental and health risks associated with landfill leachate contaminants. It emphasized the presence of emerging contaminants, including microplastics and antimicrobial-resistant bacteria, and discussed their ecological impacts and potential interventions. ScienceDirect

These studies provide detailed analyses of the dangerous contents of landfill leachates and their implications for environmental and human health.

1. Hazardous wastes and management strategies of landfill leachates: A comprehensive review

https://www.sciencedirect.com/science/article/pii/S2352186423001463?utm_source=chatgpt.com

Mohamed T. El-Saadony [a] [1], et al. https://doi.org/10.1016/j.eti.2023.103150

Highlights:

- Leachate contains toxic organic/inorganic pollutants, heavy metals and ammonia.
- Landfill leachate (LFL) causes serious environmental, ecological, and health problems.
- Classification, composition and factors affecting of leachate production are described.
- Current treatment of biological and physiochemical treatments for LFLs are discussed.
- Future prospects of the methods/techniques for treating LFL are proposed.

Abstract

Leachate usually includes toxic organic and inorganic pollutants, heavy metals, ammonia nitrogen compounds, and other dissolved and suspended contaminants. Careful management of landfill leachate (LFL) is essential to reduce leachate quantity and prevent the unfortunate fate of leachate contamination. In contrast to the management of solid waste incinerators, LFL disposal has serious ecological and health impacts in most countries, mainly associated with groundwater, soil, and air pollution. This could be attributed to the absence of cost-effective treatment technology or ideal disposal guidelines by the cities/municipalities. Therefore, ecological evaluation and sustainable managing of LFL collection and disposal treatment are highly recommended. This review describes the classification and composition of leachates, the factors affecting leachate production, and the conventional (current) treatment options for LFLs. Procedures of aerobic and anaerobic biological treatment employing microbial operated separately or in combination along with physiochemical management processes are also

discussed. This review examines the prospects of LFL treatment methods/techniques, how to economize LFL treatment methods (e.g., waste pretreatment) using readiness level technology, and recycling sorting technology.

1. Introduction

Generation and management of waste have existed throughout human existence (Nanda and Berruti, 2021, Sun et al., 2022). The pollution of leachate results from the yearly rise in municipal solid waste (MSW) production, reaching 2.2 billion tons (Chen et al., 2020). Since the emergence of humans, primitive trash has been discarded in metropolitan places, contributing to urban pollution and development of fatal illnesses (Martin, 2008). To improve living conditions, municipal and town authorities have begun rubbish gathering and disposal in landfills or marine habitats. Landfill leachate (LFL) discharge is both an environmental problem and a threat to human health (Fig. 1).

At the turn of the 20th century, waste treatment procedures evolved, and garbage collection and disposal became commonplace in metropolitan areas (Herbert, 2007). The first landfills were intentionally created pits or trenches where trash was deposited and then covered with dirt (Rajoo et al., 2020). This method degraded the earth and neighboring soil through LFL. In the 21st century, the scientific and technological community began recognizing the hazards and negative environmental effects of landfilling (Quan et al., 2020, Sun et al., 2022). The trash output has shifted significantly from biodegradable resources to nonbiodegradable, artificial plastic, and hazardous garbage, each of which adds to environmental damage (Ward, 2011, Kumar et al., 2023). Multiple industrialized nations have enacted legislation to control landfills to avoid and reduce environmental degradation (Herbert, 2007, Abunama et al., 2021). When possible, reduce, reuse, and recycle waste to avoid disposing it in landfills (McCarthy et al., 2010, Can-Güven et al., 2021, Ishii et al., 2022).

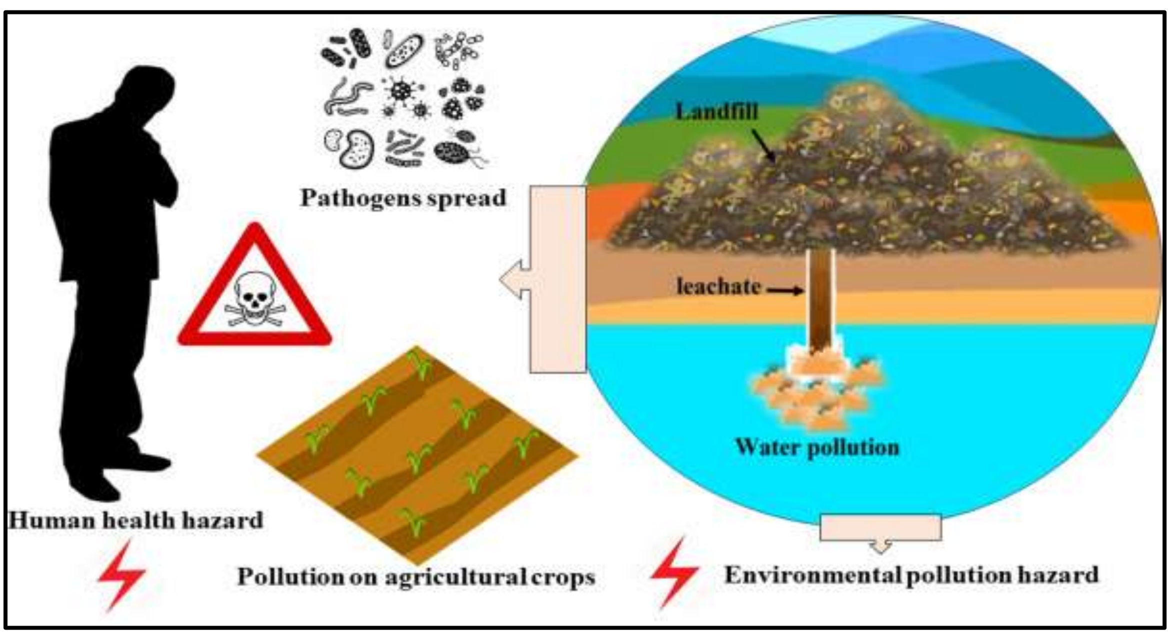

Fig. 1. Landfill leachate pollution constitutes environmental pollution and human health hazard.

There was a lack of understanding of the activities taking place in landfills, such as the generation of gases from LFL and the potential environmental damage; waste management at landfills continues to be a source of concern around the world. In the European Union (EU), the waste industry is blowing significant transformations because of the adoption of waste treatment laws and regulations. Regardless of these developments, trash output in the EU is on the rise, with an average of 2.5 billion tonnes generated yearly; of this, 36% was recycled, while the majority was landfilled or burned (Sáez and Osmani, 2019). Unfortunately, only 600 million tonnes of this material could be recycled, indicating a fundamental flaw in current waste disposal processes. Fundamentally, a shift is necessary for understanding waste as a problem that must be eliminated. Landfilling has its environmental challenges and concerns. Landfills emit off-gases such as methane (CH_4) and hydrogen sulfide, negatively affecting air quality. By enhancing waste management procedures, greenhouse gas emissions may be reduced, reducing environmental and health issues and landscape damage (Giersc et al., 2018).
The most significant problem associated with landfilling trash is the creation of LFL, a kind of hazardous wastewater (WW) that can cause groundwater pollution, which can be treated by aerobic granular sludge (AGS) technology. This method collects mature AGS from a running AGS bioreactor, stores it in different forms and environments (depending on the techniques used), and reactivates the stored AGS when needed (McCarthy et al., 2010, Yan et al., 2021, Tanavarotai et al., 2022). Also, MSW LFL is a highly concentrated organic WW of complex composition. It is a major source of pollution that threatens groundwater and surface water quality. Leachates must undergo rigorous treatment before being discharged into the environment (Chelliapan et al., 2020).
The available technologies developed for treating LFL can be classified as physical, chemical, or biological. The current major leachate method combines aerobic, chemical, and physical treatments (Tanavarotai et al., 2022). Biological treatments of LFL using aerobic ponds have some drawbacks, such as low removal of organic matter (OM) and some toxic pollutants. On the other hand, anaerobic treatment of WW has gained wide attention among researchers and sanitary engineers, mainly due to its economic merits over conventional aerobic methods (Chelliapan et al., 2020).
This review explores the nature and categorization of leachates, the variables influencing leachate generation, and the conventional and contemporary treatment methods for LFLs. Future landfilling paradigm, and the economic pointview on how to optimize LFL treatment methods (trash pretreatment, readiness level technology, and recycling sorting technology) are discussed. This review also shed lights on the possible applications of processed LFL, including green energy and natural gas.

2. LFL

LFL is a chemical mix that presents a grave threat to health and the ecosystem (Bove et al., 2015). LFL is created from liquids in the trash and percolates through landfills, collecting suspended and soluble elements originating from or byproducts of garbage degradation (McCarthy et al., 2010, Hussein et al., 2019). Infiltration of groundwater, precipitation, and rain through uncapped landfills significantly influences LFL production (Couto et al., 2017).
The Landfill Directive presents the principles and rules for controlling LFL inclusion at landfill places (Giersc et al., 2018). Specifically, the directive stipulates that a leachate treatment system must be constructed before the onset of young LFL. A leachate treatment procedure should guarantee that landfill-generated LFL is confined inside the site. Additionally, McCarthy et al. (2010) stated that the system should have established measures to decrease seepage from the

landfill via the base or side in order to reduce the interaction between the leachate and the landfill linear to prevent an increase in the liquid levels to the point where the leachate overflows and causes uncontrolled release into the surrounding environment. Therefore, information regarding the likelihood of leachate production is required for both the concept of the landfill and the following leachate treatment methods (McCarthy et al., 2010).

All MSW landfills should be designed with extensive leachate harvesting and disposal processes. Crucially, leachate harvesting should be performed throughout the landfill's lifetime, regardless of the liquid treatment approach employed. This has to contain (a) a drainage layer composed of ecological granular or artificial drainage substance, (b) perforated leachate collecting pipes, and (c) LFL harvesting sumps or header pipes from which leachate may be eliminated (Timoney, 2009, McCarthy et al., 2010). As waste decomposition continues inside this natural internal bioreactor, the LFL generated by a MSW dump varies significantly over time (Renou et al., 2008). Therefore, the categorization of LFL is crucial for determining the possible impacts of an unintentional release on the surrounding environment (EUR-Lex, 2000, Renou et al., 2008). Because of the ever-changing fact of waste degradation and leachate structures, landfill operators view LFL generation and management as a serious challenge (Wang, 2013, de Almeida et al., 2020, Ribera-Pi et al., 2021).

2.1. Classification of leachates

The categorization of LFL mostly relies on the age of the landfill. Certain variables, such as waste composition, rainfall penetration, and temperature, might influence LFL characteristics (Kjeldsen et al., 2002, Renou et al., 2008). Recent/young, middle/moderate, and mature/old LFLs are categorized according to the landfill age in which they were created (Osra et al., 2021).

2.1.1. Recent LFL

Recent/young "fresh" LFL (5 years old) is produced at landfills that have recently begun accepting garbage and where waste decomposes rapidly. During the breakdown of OM, the pH is near to neutral (6.5), and O_2 contained in the waste is eaten by native microorganisms.

The biochemical oxygen demand (BOD) and the chemical oxygen demand (COD) characterizes current LFL values greater than 0.3 (Wu and Li, 2021, Gutiérrez-Mosquera et al., 2022).

2.1.2. Middle LFL

A middle/moderate landfill (10 years) often includes substantial amounts of biodegradable OM, which promotes rapid anaerobic fermentation and the formation of vast quantities of volatile fatty acids (VFAs) (Idowu et al., 2019). Renou et al. (2008) and Kamaruddin et al. (2015) stated that middle LFL was characterized by elevated levels of COD, total organic carbon (TOC), and BOD where the BOD/COD rate was 0.7 (Abdul Halim et al., 2010) and pH was 7.5 (Lee and Nikraz, 2014).

2.1.3. Mature LFL

As the landfill ages (>10 years), the methanogenic phase of trash decomposition begins. Within this stage, methanogenic archaea convert VFAs into biogas composed of 50%–60% CH_4, 30%–40% CO_2, and 1% H2S (Buchroithner, 2015). Torretta et al. (2017) stated that mature LFLs have considerable pH > 7.5 and NH_3 N concentrations of >0.4 g L^{-1} and decreased COD (4 g L^{-1} O2). Because of the presence of humic acid (HA), fulvic acid (FA), and nonbiodegradable, recalcitrant OM, a BOD/COD ratio between 0.05 and 0.20 is also indicative of this stage (Abdul Halim et al., 2010). Table 1 summarizes other leachate management performances on the base of landfill age.

2.2. Composition of leachates

Leachate has varied physicochemical characteristics, and trash has a highly variable composition (Gutiérrez-Mosquera et al., 2022). In addition, climatic variables influence the accumulation degree of COD indices, where excess water helps attract OM and inorganic elements to the leachate, making their decomposition harder. At a given moment, the structure of a particular LFL is an excellent predictor of the biological processes' stage (Kjeldsen et al., 2002). LFL parameters include dissolved organic matter, inorganic macro material, heavy metals (HMs), and xenobiotic organic chemicals (XOCs). Other chemicals detectable in LFL include borate ($BO3\ 3-$), barium (Ba), arsenate ($AsO_4\ 3-$), mercury (Hg), lithium (Li), and cobalt (Co), albeit their minute quantities (Kjeldsen et al., 2002, Kamaruddin et al., 2015).

2.2.1. Dissolved OM

COD, BOD, and TOC are the quantifying dissolved OM metrics. These criteria encompass a variety of OM, including VFAs, FAs, and HA-like compounds. Dissolved OM influences leachate structure concerning other constituents via the complicated characteristics of its large molecular weight (MW) components. Kjeldsen et al. (2002) found that a low BOD/COD rate denotes leachates with small VFA and comparatively elevated levels of HA and FA–like substances. However, this depends on the landfill's stability and phase/age (Kamaruddin et al., 2015).

2.2.2. Inorganic macro compounds

This category includes sodium ($Na+$), ammonium ($NH4+$), calcium ($Ca+2$), potassium ($K+$), iron ($Fe+2$), manganese ($Mn+2$), magnesium ($Mg+2$), sulfate ($SO4-2$), and hydrogen carbonate ($HCO3-$) chloride ($Cl-$) compounds. Like dissolved OM, the existence of inorganic macro molecules relies on the waste phase. $Ca+2$, $Fe+2$, $Mn+2$, and $Mg+2$ concentrations are low in the methanogenic stage of LFL because of the increased pH and low dissolved organic material. At this point, $SO4-2$ content is also low due to the microbial conversion of $SO4-2$ to sulfide by communities in sequence batch reactors (SBR) (Kjeldsen et al., 2002, Kamaruddin et al., 2015).

2.2.3. Ammonia (NH3)

High amounts of NH_3 and organic N, collectively called total Kjeldahl N (TKN), are present in LFL. High amounts of NH_3 are typical of LFL, with values significantly higher than the national discharge regulations of 4 mg L^{-1} for WW to water bodies (McCarthy et al., 2010). This poisonous gas is prevalent throughout the early phases of waste decomposition and rises during the transition and acid production phases. However, NH_3 diminishes gradually over the metrogenic phases (Schiopu and Gavrilescu, 2010, Last et al., 2015). Kjeldsen et al. (2002) showed that LFL NH_3 typically varies among 500 and 2000 mg L^{-1}.

In general, NH_3 levels are greater than 600 mg L^{-1} may negatively impact microbial growth in the aerobic environments, lowering waste treatment's efficacy (Li et al., 1999). $NO2-$ and $NO3-$ levels are low in mature landfills due to the constant anaerobic conditions. Nonetheless, $NO3-$ concentrations are elevated during the methanogenesis phases of waste breakdown (Othman et al., 2010, Tsui et al., 2020). High $NO3-$ levels harm aquatic ecosystems and human health (Xu et al., 2010a, Tsakiris, 2015). Similar to NH_3, $NO3-$ is frequently observed in high quantities in MSW LFL (100 mg L^{-1}) and is strongly influenced by oxidizing circumstances, which can lead to volatilization and subsequent nitrification processes. Volatilization through processes generates free NH_3, which is changed to $NO3-$ through nitrification (Hassan and Ramadan, 2005). Under lowering anaerobic circumstances, however, this causes the conversion of $NO3-$ to NH_3 or N2 by dissimilatory $NO3-$ reduction, hence decreasing $NO3-$ levels and raising NH_3 levels (Hassan and Ramadan, 2005).

2.2.4. XOCs

XOCs are produced from artificial and ordinary chemicals, and are often present in trace amounts in LFL. Soluble XOCs can cause damage to soil and vegetation if they become contaminated (Kjeldsen et al., 2002). The most abundant substances that reach landfills are home items, such as aerosol cleaners and diapers (Galvão et al., 2020, Kwarciak-Kozłowska and Fijałkowski, 2021). The availability of hazardous material in landfills also contributes to the occurrence of XOCs in LFL. Even though this technique is no longer permitted, its lasting effects continue to be noticed. Other XOCs include hydrocarbons (toluene, benzene, and xylenes), halogenated hydrocarbons (chlorobenzene and tetrachloroethylene), phenols (4-chlorophenol and cresols) and aromatic pesticides such as hexazinone and bentazon (Kwarciak-Kozłowska and Fijałkowski, 2021).

2.2.5. HMs

The bioaccumulation of HMs can produce toxicity in living creatures and threaten human and animal health if the soil and surface become contaminated (Sulaimon et al., 2014, Vaverková et al., 2018). LFL typically contains HMs, such as copper (Cu^{+2}), zinc (Zn^{+2}), cadmium (Cd^{+2}), lead (Pb^{+2}), chromium (Cr^{+3}), and nickel (Ni^{+2}) (Baun and Christensen, 2004, Christensen, 2010, Abd El-Mageed et al., 2020). Generally, their levels stay low; the rise in pH, accompanied by the methanogenic phases of waste decomposition, enhances the soil's sorption capacity, leaving the metals immobile (Christensen et al., 2001). Multiple investigations have shown that the presence of HMs in LFL is of little relevance. However, their disposal or management is required when the HM concentration of LFL exceeds national discharge regulations (Christensen et al., 2001).

2.2.6. Microbiology of landfills

Due to the availability of abundant OM and diverse substrates in these places, landfills are dubbed "microbial pools" (Yang et al., 2022). Generally, changes in physicochemical characteristics describe landfill decomposition, with low or no connection to the altering microbial environment (Song et al., 2015a). Research has shown that a landfill's ecological circumstances and substrate specificity considerably impact its microbiota structure (Song et al., 2015b). Microorganisms transform organic waste into low MW molecules, primarily CO_2, H_2O, and HA–like chemicals, during the aerobic phase. The consequent high energy yields facilitate rapid microbial development (Antony et al., 2020, Mahtab et al., 2021). NH_3 is transformed to NO_2^- by NH_3^- oxidizing bacteria during nitrification, which is then lowered to NO_3^- by NO_2^- oxidizing microorganisms. Under anoxic circumstances, denitrifying bacteria use N_2O to convert nitrogen (N) to nitrogen gas (N_2). Without these aerobic and anaerobic processes, N cannot be successfully eliminated from landfills (Sang et al., 2012). Acidogens hydrolyze complex organic wastes such as carbohydrates, proteins, and fats into monosaccharides, amino acids, and fatty acids during the anaerobic phase. These bacteria convert these outputs into hydrogen, carbon dioxide (CO_2), and organic acids like lactate and acetate. These chemicals might be converted to CH_4 by acetoclastic and hydrogenotrophic methanogenesis under anaerobic conditions (Sang et al., 2012).

Numerous bacteria have been extracted from LFL to date. Wang et al. (2017) investigated the microbial composition of a municipal solid waste dump in Yangzhou City, East China, and discovered that Proteobacteria, Firmicutes, and Bacteroidetes were the major taxa. Similarly, Hale Boothe et al. (2001) discovered Gram-positive species (*Staphylococcus lentue*, *Staphylococcus delphini, Bacillus megatherium,* and *Bacillus pasteurii*) and Gram-negative species, the majority of which below the genera *Yersinia*, *Acinetobacter*, *Enterobacter*, and *Pseudomonas*. In addition, numerous Gram-positive bacteria, such as *Lysinibacillus, Bacillus, Brevibacillus, Staphylococcus,* and *Clostridium* spp., were recovered from LFL (Krishnamurthi and Chakrabarti, 2013).

3. Factors affecting leachate composition
The formation and composition of leachate are impacted by additional variables, such as meteorological conditions and waste makeup, which are briefly mentioned below. Other elements that impact leachate formation include the absence or the existence of hydrogen and oxygen gases, the retention period of MSW retention in the landfill, the covering material for the final LFL, and leachate recycling (Borglin et al., 2004, Hossain and Haque, 2009).
3.1. Weather differences
Alterations in room temperature and landfill equilibrium significantly impact the LFL features. For instance, landfills in hot, dry climates tend to accumulate tiny quantities of LFL, while those in tropical areas produce greater amounts of diluted LFL (Kamaruddin et al., 2015, Scandelai et al., 2020, Yuan et al., 2021). Tränkler et al. (2005) noted that landfill biodegradability is sluggish in regions with dry climates but significantly enhanced during wet seasons (Gautam and Kumar, 2021). More than 60% of LFL-generated precipitation can permeate landfill sites; this cannot be avoided while a landfill cell actively accepts garbage. The amount of moisture-rich OM wastes and liquids put into a landfill was also shown to boost LFL production (González et al., 2011, Cheng et al., 2020, Can-Güven et al., 2021). Due to increased microbial activity, biological decomposition happens more rapidly in warmer climes, accelerating the landfill's shift to acid production and methanogenic stages (Khattabi et al., 2002).
3.2. Waste structure
The structure of MSW and its consequent decay considerably influences the quantity and quality of LFL (Somani et al., 2019). Various garbage breakdown products combine with the moisture in the landfill during this process to create LFL. Because of the quantity of general rubbish and the organic and inorganic chemicals created during waste degradation, waste with a high organic and moisture inclusion, in particular, produces high-strength leachates (Barghash et al., 2021, Adaryani and Keen, 2022). However, researchers have reported that waste limitation tools, such as separation of the OM component of MSW, waste preprocessing, and reprocessing, can substantially affect the quality of LFL (Kamaruddin et al., 2017). Therefore, to reduce the production of high-strength LFL near the base of garbage structures, it is recommended that landfill operators examine the acceptance regulations at each site (Christensen, 2010). Scandelai et al. (2021) evaluated the intensification of super critical water oxidation (SCWO) process through ion-exchange with zeolite (SCWO/zeolite) for potential reuse of treated leachates depending on the municipal WW reuse rules. This technology was applied to raw leachate (RL) and the traditional techniques of pretreated leachate (PL) at the studied landfill. The continuous SCWO reactor operated at a temperature of 600 °C, a pressure of 23 MPa, and spatial time (τ) from 29–52 s without adding oxidants. A commercial zeolite (clinoptilolite) in a fixed-bed glass column was utilized for ion exchange. In addition, Scandelai et al. (2018) assessed LFL degradation using the innovative combination of SCWO with ozonation (O3/SCWO) and oxidation (SCWO/O_3). Ozonation occurred at different reaction times (30–120 min). Combining ozonation (30 min) with SCWO (O3-30′/SCWO) was considered the most efficient technique for LFL degradation, when compared with other treatments (Scandelai et al., 2018).
4. Impact of improper LFL management on the environment and public health
Determining and assessing the risks posed by LFL are essential to relieve the direct effects on the environment. It is important to stress the issue of proper landfill management to decrease leachate and landfill gas (LFG) emission. Although landfills are environmentally straining, precautions can be taken to alleviate negative impacts. According to the United States

Environmental Protection Agency (USEPA, 2016), the waste management company can close the landfill when ends and no longer can collect any waste. This is followed by a postclosure process of which the company is responsible for managing the landfill for another 30 years, taking into consideration that management proceeds because leachate and gases do not stop being produced. In addition, companies should have alternative uses for this land (USEPA, 2016). For example, companies can be engaged in the renewable energy industry by developing solar farms to help generate revenue. Enforcement of federal regulation of landfills can also lessen the effects of the landfill on air, soil and water (USEPA, 2016).

Well-designed and properly operated landfills ensure that the environment is contaminant-free; whereas proper construction and maintenance ensure that landfills are incorporated with environmental monitoring systems to track any gas release and groundwater contamination (USEPA, 2018). The United States Environmental Protection Agency Community-Focused Exposure and Risk Screening Tool (USEPA C-FERST, 2017) detailed proper designing/operating sustainable landfill management, and recommended that communities should be more involved in sharing this sustainable future by decreasing waste generation and alleviating the negative effects of landfills (USEPA C-FERST, 2017).

5. Regulations and considerations for hazardous waste management

Hazardous waste was first defined in the USA in the 1980s as unwanted materials that threaten humans and environment when poorly managed (Marinković et al., 2008). Rapid industrialization gives rise to hazardous waste generation associated with increased consumption of products and services (Duan et al., 2008). This requires great efforts to collect, transport, recycle, treat and dispose such waste. Thus, it is vital to take necessary safeguards and investigate novel methods for effective management (Marinković et al., 2008).

In Turkey, the expansion of its medical care infrastructure and the transformation of its industries into a developing country led to risks and challenges in hazardous waste management (HWM). As such, Turkey's environmental law in 1983, Basel convention on control of transboundary movement of waste in 1994 and HWM regulation in 1995 were introduced (Salihoglu, 2010). Although waste prevention and minimization have never been a priority at the national level for a long period of time, Turkey adopted regulatory practices in the European Waste Catalogue (2000/532). The transformation approach of the national waste management is based on prevention at source, reduction, recycling, incineration/energy production, and disposal (Salihoglu, 2010).

Safe transportation of hazardous waste from hazardous waste generators (HWG) to disposal sites, solving the costly and time-consuming hazardous waste transportation (HWT) problem and the selection of the hazardous waste carrier (HWC) are important subjects for decision makers (Zhao et al., 2016). Büyüközkan and Gocer (2017) indicated that on-site treatment can be done only for small quantities of hazardous waste compared to off-site treatment facilities. Large HWGs assign licensed HWCs to dispose hazardous wastes off-site. This requires special transportation, treatment facilities or designated hazardous waste storage and disposal sites (Orford, 2007).

Safe HWT, governed by strict HWT regulations, is vital to prevent potential contamination by hazardous waste and reduce threats to public health and environment (Orford, 2007). Governmental organizations inquire HWCs to have HWT licenses, to accept HWT with minimal spillage and risks. Therefore, HWG can assess different licensed HWC options and select the least cost one. This suggests that the selected HWC must operate legally to ensure quality, responsibility, safety and service (Ho, 2011).

Nowadays, multiple-criteria decision-making (MCDM) has become a "hot" topic, including different objectives or criteria and finding solutions to complicated problems (Lertprapai, 2013). MCDM methodologies such as analytic hierarchy process (AHP), the technique for order of preference by similarity to ideal solution (TOPSIS), and Višekriterijumsko kompromisno rangiranje (VIKOR), have been successfully applied to solve several types of decision-making problems (Lertprapai, 2013). However, most of these "standard MCDM techniques" are run with crisp numbers, and prove inadequate. The criteria for selecting the best alternative often rely on uncertain/fuzzy information. Therefore, these MCDM methods are used with fuzzy numbers (Lertprapai, 2013).

6. LFL different treatment technologies

LFL may differ depending on the content and age of landfill contents, the degradation procedure, and climate and hydrological conditions. Many treatments, including biological procedures (bioreactors, bioremediation, and phytoremediation), physicochemical approaches (advanced oxidation processes, adsorption, coagulation/flocculation, and membrane filtration (MF)), and others were used to treat LFL (Costa et al., 2019).

Membrane bioreactors and integrated biological techniques, including anaerobic NH4+ oxidation and nitrification/denitrification processes, have demonstrated high performance in NH_3 and N elimination, with >90% removal effectiveness (Costa et al., 2019). Moreover, improved elimination efficiency for suspended solids and turbidity has been achieved by coagulation/flocculation techniques. Combining different treatment techniques can improve removal effectiveness of metals by 40–100%. In addition, the combined techniques used for LFL treatment have been reported with good performance when they are linked with high COD and concentrations of NH_3 and low biodegradability (Costa et al., 2019). However, more research to improve treatment methods for maximum removal efficiency must be prioritized (Mojiri et al., 2021).

Various approaches are now employed to control LFL (Fig. 2), most of which have been modified from WW therapy procedures (Raghab et al., 2013). In general, a mixture of physiochemical and biological strategies is necessary to successfully manage LFL (Fig. 2), as it might be challenging to get satisfactory results with a single technique alone (Kargi and Pamukoglu, 2004). In addition, LFL produced by various facilities frequently differs substantially in structure and necessitates various forms of treatment. For instance, Gao et al. (2015) and Torretta et al. (2017) found that biological procedures can efficiently control LFL with an elevated organic inclusion, but LFL with a little organic inclusion is better fitting for physicochemical management (Gao et al., 2015, Torretta et al., 2017).

Conventional LFL treatments, including biological methods, physicochemical approaches or their combination, vary in their performance depending on the landfill age. In contrast, emerging treatments, such as C-based materials, are dependent on replacing activated C and other adsorbent materials using C-based materials (e.g., biochar or hydrochar). Photocatalytic, electrooxidation, and Fenton-like processes are an influential group of technologies that have been broadly tested for the degradation of different pollutants present in leachate (Bandala et al., 2021).

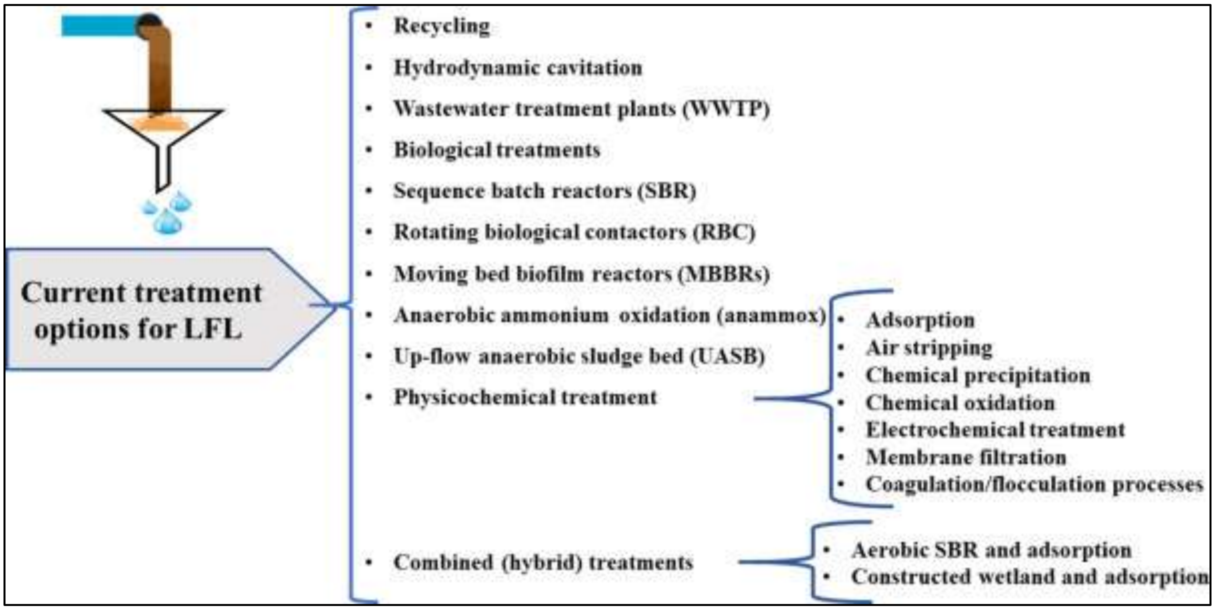

Fig. 2. Conventional and current treatment options for landfill leachates (LFL). Hybrids involve more than one treatment, mostly combining physiochemical and biological treatments.

6.1. Recycling

Since the 1990s, landfill sites have used LFL recirculation or recycling to manage LFL, generate and recover LFG, and generally improve the stability and sustainability of the dump (Lee and Jones-Lee, 1994, White et al., 2011). In general, this technique entails managing the biological, chemical, and physical processes of the landfill by reintroducing WW into the landfill, known as a landfill bioreactor at a specific time (Lema et al., 1988, Warith, 2002). After four years of LFL recirculation, the BOD/COD ratio of bioreactor-produced LFL reduced from 0.5 to 0.7 to 0.1, according to an analysis of the LFL's structure (Benson et al., 2007), which was studied by Warith (2002) in MSW landfill of Toronto, Canada, to improve organic waste decomposition over six months, including weekly leachate sample collection and analysis. Warith (2002) also noted a reduction in the LFL COD and BOD values from 9910 to 200 mg L^{-1} and 2055 to 200 mg L^{-1}, respectively.

In a 200 L bioreactor holding 2 tonnes of MSW trash and an LFL recirculation ratio of 2.2 L min^{-1}, Chugh et al. (1998) observed that high recirculation ratios boosted the solubilization of fresh garbage and accelerated the establishment of a methanogenic microbial culture, allowing waste stabilization within the bioreactor. Other investigations showed comparable outcomes (Šan and Onay, 2001). While the cycling of LFL at landfills has yielded favorable outcomes, extending the retention duration of LFL within landfills raises the risk of ground and surface infiltration and contamination, which are serious problems (Šan and Onay, 2001).

6.2. Hydrodynamic cavitation

The novel technical method of hydrodynamic cavitation can be employed to handle LFL WW. In the work conducted by Gutiérrez-Mosquera et al. (2022), all cavitation techniques resulted in a decrease of 23–51% in BOD, 30–53% in COD, 12–21% in TOC, and 99% exclusion in total suspended solids, with an elevation in the WW biodegradability indices from 0.16 to 0.25. In addition, they discovered that the ideal cavitation period was 30 min, which was sufficient to achieve chemical equilibrium and generate free radicals from water molecules separated by the cavitation pressure pulse. In addition, no more hydroxyl radicals were produced, which was necessary to stop the cavitation process since aromatic condensation and humification reactions might trigger the formation of new, difficult-to-decompose compounds (Gutiérrez-Mosquera et al., 2022).

In addition, a 200 mg L^{-1} concentration of oxidizing agent showed the greatest results in lowering the quality indices assessed. Hydrogen peroxide (H_2O_2) dissociation increased the production of hydroxyl radicals, accelerating the decomposition of organic and inorganic matter. However, the influence of other significant variables, such as the speed and flow of the fluid, the operating temperature, the geometry of the cavitation device, the number of steps or cycles in the reactor, and the exploration of other oxidizing agents, should be investigated in future research to obtain the greatest benefits from the hydrodynamic cavitation of LFL WW (Korniluk and Ozonek, 2011, Gutiérrez-Mosquera et al., 2022).

6.3. Waste water treatment plants (WWTPs)

Co-treatment of LFL with household sewage in WWTPs is one of the most prevalent options (Gao et al., 2015). This approach entails either a continuous feed or a batch load of LFL into the WWTP influent. LFL contributes to the N-containing portion of this sewage flow, whereas sewage fulfills the P requirements of WWTP (Gao et al., 2015). Numerous opponents have questioned the dependence on this strategy because of OM with low biodegradability (polycyclic aromatic hydrocarbons, polychlorinated biphenyls, HA, and FA) and HMs in LFL, which may diminish management efficacy and raise the effluent levels from these facilities (Renou et al., 2008).

In Ireland, the applicability and cotreatment of LFL in WWTPs were recently evaluated (Brennan et al., 2016, Brennan et al., 2017a). These assessments revealed that LFL (99%) was accomplished through direct discharge to sewers or tinkering to WWTPs, while only 1% was accomplished through onsite treatment, including six landfills, only three of which meet the requirements for direct disposal to receipt bodies (Brennan et al., 2017b). In addition, 30% of the Irish WWTPs rejected to take LFL from 2010 to 2014, highlighting this argument (Brennan et al., 2016). Consequently, treating LFL in WWTPs is judged unsuitable, and other options must be found. Previous studies have outlined the benefits and drawbacks of utilizing WWTPs for treating LFL (Teixeirae Junior and Marinheiros, 2014, Lippi et al., 2018).

6.4. Biological therapies

Biological approaches are frequently utilized for treating LFL because of their ease, dependability, and cost-effectiveness (Renou et al., 2008, Hamdan et al., 2021). Biological management successfully approaches lower COD and BOD concentrations in LFL. Young LFLs (5 years) with COD/BOD ratios in the range of (0.4–0.6) are suitable to bio-management for older LFLs because of biodegradable organics in the former (Torretta et al., 2017).

When the COD/BOD rate is 0.2 or less, the landfill becomes less biodegradable due to the accumulation of HA, FA, and other organic materials resistant to decomposition (Tchobanoglous and Kreith, 2002, Lee and Nikraz, 2014). Microorganisms, primarily aerobic and anaerobic bacteria, accomplish biodegradation by converting OM to CO_2 and CH_4 (Wang et al., 2017). Aerobic techniques are generally inexpensive and efficient for decomposing BOD and COD in addition to organic contaminants (Bove et al., 2015, Lam et al., 2020, Shao et al., 2021). Kamyab et al. (2015) used microalgae to remove pollutants from palm oil mill effluent (POME).

Most biological techniques are based on a suspended growth system, like SBR, activated sludge methods, and aerated lagoons, biofilm reactors, with several scientific studies describing their use to treat LFL (Xu et al., 2010b). On the other hand, anaerobic alternatives are more suited for addressing LFL with a high concentration. In contrast to aerobic, anaerobic treatments need small energy, generate fewer solids, and can work at reduced ambient temperatures (Christensen et al., 2011). Additionally, anaerobic methods may convert the garbage into biogas, a useful renewable energy source (H2, O2, N2, CO_2, CH_4, and H2S). The up-flow anaerobic sludge bed (UASB), the suspension growth method, and several SBR and anaerobic filters, such as hybrid and fluidized beds' reactors are examples of these anaerobic treatment alternatives (Kennedy and Lentz, 2000, Liang and Liu, 2008). Lin and Chang (2000) and Christensen (2010) outlined the benefits and drawbacks of both aerobic and anaerobic methods used to treat LFL. The benefits of aerobic biological treatment include the ability to achieve nitrification at low temperatures (5 °C), nutrient exclusion, simultaneous N2-fixation, denitrification onto microbial biofilms, high sludge retaining duration, low sludge production volumes, enhanced resistance to toxic shock, retention of active biological biomass inside the system, low hydraulic retention time (HRT), and active biological biomass retention within the system (Christensen, 2010). Table 2 summarizes advantages and disadvantages of aerobic and anaerobic biological treatments.

Denitrification in biofilm reactors requires high quantities of dissolved oxygen, forced aeration, mixing, high energy demand, and congestion difficulties associated with fixed bed systems (Table 2). Benefits associated with anaerobic biological treatment include small biomass production, low energy requirements, a small dose of P needed as a growth factor for anaerobic bacteria, increased organic loading rates (OLRs) combined with high removal of BOD and COD, and the ability to act as a net energy producer by producing CH_4 rich biogas. However, lengthy start-up period, temperature and pH variations susceptibility, and several hazardous chemicals, such as HMs and NH_3, are among of the drawbacks of the anaerobic biological treatment (Lin and Chang, 2000, Christensen, 2010, de Almeida et al., 2020). The efficacy of some organisms in removing leachate pollutants has already been documented (Salam and Nilza, 2021). Table 3 summarizes the removal rates of several leachate contaminants using aerobic biological treatment.

6.4.1. SBRs

Spagni et al. (2008) stated that SBRs are a technique for treating activated sludge that requires one or additional phases below nonstationary circumstances. Typically, the system utilizes aerobic circumstances and consists of four consecutive steps: filling, reacting, settling, and discharging. This management approach is very successful for the nitrification and denitrification of LFL because its functional approach permits simultaneous oxidation of organic carbon (C) and nitrification (Renou et al., 2008).

For instance, Diamadopoulos et al. (1997) demonstrated that using simultaneous LFL and municipal sewage treatment in an SBR removed 80, 95, and 85% of NH_3, COD, and BOD, respectively. In another study, treating LFL of solid wastes using SBR and membrane reactor removed NH_3, P, and BOD by 99.9%, 94%, and 82.2%, respectively (Timur and Özturk, 1999). They also studied the management of LFL utilizing SBRs below anaerobic circumstances. They employed this system to treat LFL from an MSW dump with COD levels ranging (from 3.8–15.9 g L^{-1}) and pH values (7.40–7.6). They reported that COD removal rates was found to be between 64–84%, with CH_4 conversion rate at 84% (Timur and Özturk, 1999). They demonstrated that SBR treatment was resilient and less susceptible to repeated variations in organic load or NH_3-N. This information is crucial for LFL management, as these concentrations might fluctuate substantially over time (Timur and Özturk, 1999). Nonetheless, this method has certain flaws. C consumption during the aeration stage might result in inadequate C resources during the subsequent anoxic stage, leading to low TKN removal rates (Wei et al., 2012). NH_3 levels can be reduced based on these systems' organic anaerobic decomposition ratios; to modify SBR by adding a pre- and post-treatment phase, which may be necessary to achieve higher disposal rates (Elleuch et al., 2020, Ghahrchi and Rezaee, 2021).

6.4.2. Rotating biological contactors (RBCs)

The RBC is an oxygenated, fixed-film biological management technique comprising closely separated plastic disks mounted on a polycarbonate or polystyrene plastic shaft. Rotation submerges the disks with a large biofilm-forming surface area in the system liquid. This treatment's efficacy is determined by the HRT administered, the disk's rotational speed, temperature, and submerged surface area (Castillo et al., 2007). RBC systems were widely used to treat biologically treatable WW, including industrial WW (Cortez et al., 2008).

The primary benefits of RBC techniques are their straightforward operation and preservation, but they need a substantial primary asset and have high operating expenses (Spuhler, 2018). Different data have assessed RBC systems for LFL treatment (Nazia et al., 2021). In addition to nitrification, the extended maintenance period of biomass in RBC and the substantial biofilm density allow N removal. Nitrification oxidizes NH_3 to NO_3 and NO_2, which was seen in LFL with little biodegradable OM. In addition, they obtained an N removal rate of 70% (Wang et al., 2009). Also, Cema et al. (2007) investigated the elimination of NH_3 and NO3− from LFL using an RBC anaerobic ammonium oxidation (anammox) method throughout a six-month experimental period. The greatest NH4+ and NO3− removal rates were 0.56 kg N m^{-3} d^{-1} and 0.76 kg N m^{-3} d^{-1}, whereas the maximum rate of inorganic N removal was 0.93 kg N m^{-3} d^{-1}. Their research shows that RBC might be employed in anammox procedures to reduce N compounds from LFL. Additionally, RBC may be utilized to eliminate COD from LFL (Cema et al., 2007).

Castillo et al. (2007) presented the application of a perforated acetate disk coated with a polyester mesh for COD elimination at varying rotating rates, flow rates, and HRTs. It was possible to attain high COD elimination rates of 69–74%; nevertheless, operational concerns surrounding disk detachment significantly influenced these rates. However, these investigations demonstrated the potential of RBC for LFL therapy. RBC systems should be integrated with other processes to achieve optimal elimination rates for organic and inorganic contaminants (Cortez et al., 2008, Hassard et al., 2015).

6.4.3. Moving-bed biofilm reactors (MBBRs)

MBBRs are oxygenated containers comprised of polymeric composite carriers with high surface area onto which microbial biofilms can grow. This carrier material normally occupies around 50% of the reactor's volume and is fluidized within the bioreactor using aeration (Bove et al., 2015). The literature summarizes the benefits of this approach above other typical suspended growth methods as follows: (1) the MBBR utilizes the full tank volume for biofilm formation, resulting in extremely reduced head loss; (2) unlike the sludge system activation, it does not need sewage sludge valorization as the process's usage of a screen at the top of the reactor, biomass is preserved; (3) it is less sensitive to toxic compounds; (4) this system can eliminate organic and high NH_3 concentrations with a single method; and (5) the reactor can be utilized for aerobic, anoxic, or anaerobic progression (Chen et al., 2008, Renou et al., 2008). Different data have detailed the application of MBBRs for LFL therapy with different outcomes (Chen et al., 2008).

Disposing organic N from LFL using two-stage aerobic and denitrification systems removes 90% of both inorganic and total N (Welander et al., 1998). Chen et al. (2008) employed an MBBR system with an anaerobic/aerobic configuration for the treatment of LFL, which removed high COD by >90% for applied OLRs of 4–15 kg COD m^{-3} d−l inside the anaerobic treatment, while the aerobic stage acted as a COD-polishing and NH4+ removal phase (Chen et al., 2008).
The effect of different carrier substances, free-floating polyurethane particles, and granular activated carbon (GAC) was investigated in LFL treatment in the MBBR system (Loukidou and Zouboulis, 2001). The GAC provided a porous medium capable of adsorbing OM and NH_3 and a good surface for developing microbial biofilm. Both processes were utilized to degrade N and remove organic pollutants. GAC particles are better than polyurethane particles in removing COD by 81% versus 65%. However, the BOD and N removal efficiencies were comparable (Loukidou and Zouboulis, 2001).

6.4.4. Anammox

Jetten et al. (2009) described anammox as a microbial procedure adopted by strictly anaerobic autotrophs that predominantly use NH4+ and NO2− as substrates for their catabolism. The procedure has been utilized in several previously described applications, such as the SBR, MBBR, and UASB (Klaus et al., 2017), for managing NH4+ enriched WW, such as LFL (Gao et al., 2015). For instance, Ruscalleda et al. (2010) described the effective removal of NH4+ and N from LFL utilizing anammox and heterotrophic denitrification processes. Before being fed into an anammox SBR, LFL was originally treated via a partial nitrification procedure. Overall, their investigation suggested N removal efficiency of 87%, with the anammox process accounting for around 85% of this elimination (Ruscalleda et al., 2010).
Comparatively, Selic et al. (2007) utilized an SBR to treat LFL from Chinese and German landfills and reported a decrease of 90% in both COD and NH_3 contents. Wang et al. (2011) got comparable findings when treating LFL using a combined SBR and anammox method. They reported a COD removal efficiency of 21 to 45% and a total N reduction of 62 to 80%. Anammox is an effective and long-lasting therapy option for LFL that may be combined with other treatment choices, as stated above. The anammox technique is also suitable for managing LFL from mature landfills with high NH_3 levels and low amounts of organic chemicals, like COD and BOD (Torretta et al., 2017).

6.4.5. UASB

The UASB is a single tank into which WW is driven upwards by hydraulic pressure through an intake at the foot of the bioreactor. The UASB bioreactors maintain metabolically active anaerobic granular sludge, allowing for the management of WW at high OLRs (Renou et al., 2008). The anaerobic granules include bacteria and methanogens, which lower influent WW anaerobic digestion (AD). The AD is the degradation of OM by bacteria without oxygen (Rogoff, 2014). This biological process generates a gas, commonly known as biogas, comprised mostly of CH_4 and CO_2 (Rogoff, 2014).
Kettunen and Rintala (1998) treated MSW LFL at low temperatures (13–20 °C) using UASB bioreactors injected with mesophilic biomass from a sewage treatment facility. The COD concentration of the LFL utilized in their investigation ranged from 630 to 2200 mg L^{-1}, and the bioreactors obtained COD removal efficiencies varying from 50 to 75%, with the maximum removal ratios recorded at 18 °C (Kettunen and Rintala, 1998).
Sun et al. (2010) examined the removal of COD from LFL utilizing two UASB bioreactors operating at low temperatures (10–15°C) and an SBR system. Both UASB bioreactors were inoculated with activated sludge from a WWTP in Heilongjiang, China, that treats brewery WW.

The COD content in the influent water varied between 7.856 and 22,500 mg L^{-1}. They showed a high COD removal efficiency (77%) in the first UASB bioreactor but a decrease in successive treatment systems. In addition, Lin et al. (2000) utilized UASB bioreactors to treat LFL mixed with sewage (septic tank sludge) in a 1:1 ratio. The applied OLR for these bioreactors was 7.73 kg COD m^{-3} d^{-1}, and HRT was 1.5 d. The total COD, NH_3, and P elimination efficiency percentages were 42, 47, and 44%, respectively (Lin et al., 2000).
Utilizing UASB bioreactors for LFL therapy has been practical, as seen in the cases above. However, the AD process's sensitivity to high NH_3 levels persists as a problem, rendering this therapy approach inappropriate as a stand-alone treatment for LFL (Torretta et al., 2017). In general, biological treatments are most potent in removing NH_3, BOD and COD from LFL (Torretta et al., 2017).

6.5. Physicochemical treatment

According to Kurniawan et al. (2006), when the concentration of pollutants significantly exceeds their national discharge regulations, physicochemical treatment of LFL is frequently required (Table 4). These treatment techniques are often employed to remove non-recyclable organics, such as HA and FA, HMs, and absorbable organic halogens (Abbas et al., 2009), and they are efficient for lowering color and colloidal particles (Torretta et al., 2017). Bio-refractory chemicals that typically hinder physicochemical procedures are, thus, most effective in pre- or post-treatment stages. Physicochemical therapies are frequently employed to treat mature LFL, with a BOD:COD rate of 0.2 and small concentrations of biodegradable chemicals (Kjeldsen et al., 2002). Adsorption, MF, chemical precipitation, and oxidation are the fundamental physicochemical approaches successfully used for LFL management.

6.5.1. Processes of coagulation and flocculation

In coagulation/flocculation, chemicals identified as coagulants or flocculants remove tiny, suspended particles from water by producing larger flocs, followed by physical separation or gravity settling. Despite its convenience, the approach is good for removing colloid pollutants, like trace metals, surfactants, and humic/fatty acids (An and Xu, 2013, Costa et al., 2019). The treatment of juvenile leachate using the lime coagulation technique significantly decreased many pollutants, including OM, turbidity, and mineral matter. This is because of Ca^{+2} coagulation, HM precipitation in carbonates, hydroxides, and phosphates (PO4−3), and their adsorption rates as pH rises (Hasna et al., 2020).

6.5.2. Retention

Adsorption is the process of precipitating (gas or liquid) solution-based compounds on a fit surface (Worch, 2012, Chen et al., 2021, Wang and Ding, 2022). Fixed bed adsorption has been a common physiochemical solution for many WW streams, mostly because of its ease and low cost (Biswas and Mishra, 2015). Activated carbon (AC) or another absorbent, like zeolite, pumice stone, and calcareous shells, can be adsorbed (Tsai et al., 2006, Abdul Halim et al., 2010, Chansuvarn, 2018). Few data have investigated the treatment of LFL (Huong et al., 2016); however, several adsorbents have been tried on WW (Song and Gao, 2013, Huong et al., 2016). In batch investigations, Martins et al. (2017) analyzed the usage of clinoptilolite (zeolite) to remove NH4+ from LFL. They predicted the adsorption characteristics using the Langmuir isotherm and found qmax (theoretical maximum adsorption capacity g^{-1} adsorbent) values between 2.61 and 17.68 mg L^{-1} ammoniacal N (NH4+-N) under various operating factors, like pH, adsorbent quantity, and LFL levels. The Langmuir isotherm is commonly used in adsorption tests to assess the type and quantity of adsorbed ions (Scandelai et al., 2020). Abdul Halim et al. (2010) evaluated the efficacy of AC, zeolite, and a composite material combining zeolite, limestone, AC, and rice husk C to eliminate COD and NH_3 from LFL in batch tests. Langmuir isotherm determined that COD and NH_3 adsorption rates differed significantly across the various adsorbent substances evaluated. However, it is essential to realize that research based on batch investigations – as explained above – may be inaccurate over longer periods (Callery et al., 2016). Therefore, adsorption substances should preferably be evaluated in a fixed bed column process to assess whether or not LFL can be treated for an extended period.

Aziz et al. (2010) improved the SBR treatment option using powdered AC (PAC) adsorption for LFL treatment. With a standalone SBR control, 1.2 g of PAC was applied to six 2 L beakers, including 1.08 L of sludge and 120 CC of LFL from Kulim Landfill, Kedah, Malaysia. The COD and NH4+-N removal efficiency were measured throughout the trial. In the SBR, COD (influent 1655 mg L^{-1}) had a removal rate of 69.8%; whereas NH_3 (influent 600 mg L^{-1}) had a removal ratio of 74% but rose to 76% when PAC was added. Adsorption successfully treated LFL; removal rates depend on the adsorption materials utilized. This method is simple and economical (Lofrano et al., 2020). In addition, adsorption can be employed as a standalone management strategy or in conjunction with biological treatment alternatives, as stated previously.

The intensified system substantially enhanced RL properties by eliminating 89% of COD and 99% of NH_3-N. Except of arsenic (As) and molybdenum contents, contaminants in PL were under the limits for discharge and reuse (Scandelai et al., 2021). Thus, the high As contents in RL and PL requires more investigation on hazardous As properties in leachates. On the other hand, the intensified procedure was carried out without including oxidants or auxiliary compounds, resulting in a cheaper and more environment-friendly process that could be applied to treat leachates (Scandelai et al., 2018, Scandelai et al., 2021).

6.5.3. Air stripping

NH4+-N levels are frequently elevated in LFL (Brennan et al., 2017b). Air stripping, the technique of separating organic contents from a waste stream by exposure to air, could be utilized effectively to remove this contaminant known to elevate WW toxicity (Christensen, 2010). Until equilibrium is reached, volatile organic molecules are released from the liquid state into the gaseous state during the process. This process takes place in a stripping tower holding an LFL-filled aerated medium (Renou et al., 2008). Leachate is collected at the base of the tower, while gases ascend to the top. Before being released, NH4+ polluted air is dealt with sulfuric or chloric acid. Nevertheless, treated leachate needs further treatments to meet NH_3 discharge limitations (Renou et al., 2008).

Ferraz et al. (2013) examined the pretreatment of LFL using air stripping and NH_3 recovery via absorption. The oxygenated packed tower removed 88% of the NH_3 from 100 L of LFL. At the same time, absorption units containing 0.4 mol L^{-1} of deionized sulfuric acid solution recovered 80% of the stripped NH_3. In addition, Marttinen et al. (2002) stated that removing NH_3 in LFL at low temperatures (6°C and pH 11) was 30% lower than using a temperature of 20 °C by air stripping method. In addition, they noted that independent of pH or temperature, COD elimination using air stripping ranged from 4 to 21%, showing that the treatment was unsuccessful at removing biodegradable organics (Ferraz et al., 2013).

Silva et al. (2004) evaluated the elimination of NH_3 from NH_3-rich LFL (750–800 mg L^{-1}) that had pretreatments of coagulation, flocculation, and membrane fractionation, although neither pretreatment reduced the baseline NH_3 content. At pH of 12, 2 h aeration was sufficient to remove NH_3 by 72%; however, an additional 84 h aeration was required to achieve a 5 mg L^{-1} NH4+ level (Silva et al., 2004). This treatment method has several drawbacks and limitations, most notably the management and destruction of exhaust air, like NH_3 gas. If discharged directly into the environment, these off-gases might create significant air pollution; hence, further treatment or NH_3 recovery stages are necessary (Abbas et al., 2009). In addition, the efficacy of this therapeutic method might vary depending on the composition of LFLs and may require further modification. This treatment method promises LFL therapy, but the technology must be developed further to obtain greater clearance rates of other current chemicals (Abbas et al., 2009).

6.5.4. Chemical precipitation

Chemical precipitation removes non-compostable OM, NH4+, and HMs from LFL because of the technique's ease and the necessary apparatus's low cost (Calli et al., 2005). During this process, chemical processes change dissolved ions into an insoluble solid state. Chemicals precipitate at varying pH levels (Kurniawan et al., 2010). Struvite formation, the most prevalent chemical precipitation process, is similarly influenced by pH, and Mg+2:NH4+:PO4−3 (MAP) ratios of WW. Struvite formation is highly effective in removing NH_3 from WW; however, it is less effective at treating LFL because of the lower amounts of Mg and P necessary to produce crystalline MAP. Consequently, LFL must be supplemented with these chemicals for MAP formation to be successful.

Li and Zhao (2001) employed struvite formation to treat LFL and studied the impact of three MAP combinations on effective precipitation: (I) disodium phosphate + magnesium chloride hexahydrate, (ii) magnesium oxide + phosphoric acid (85%), and (iii) monocalcium phosphate dihydrate + magnesium sulfate heptahydrate. The main level of NH4+-N in LFL was 5618 mg L^{-1}. When the first treatment was added to LFL samples, the quantity of NH4+-N decreased to 0.112 g L^{-1}, with Mg+2:NH4+:PO4−3 exhibiting a molecular ratio of 1:1:1 and MAP crystallizing in 15 min (Li and Zhao, 2001).

Meanwhile, Di Iaconi et al. (2010) assessed the effectiveness of MAP (1:1:1) precipitation in removing NH_3 from a mature LFL, which was accomplished by adding $H3PO_4$ and MgO as external sources of P and Mg, respectively. They reasoned that the poor solubility of MgO was attributed to the 67% NH_3 removal efficiency, which increased to 95% when the MgO ratio in the reaction mix was increased to (2:1:1) (Di Iaconi et al., 2010). Although this treatment approach is useful for NH_3 removal, it is insufficient for removing HMs, which render the effluent LFL and necessitate further treatment. When LFL was utilized to make struvite, the final product was impure and requires further purification (Song and Gao, 2013).

6.5.5. Organic oxidation

To manage polluted soil and groundwater, strong chemical oxidizers are immediately applied to a contaminated medium to destroy a wide variety of OM (Torretta et al., 2017). Oxidation and reduction reactions typically oxidize one of the reactants, reducing or gaining electrons in the other. Thus, the oxidizing agent reduces the dangerous pollutants into safe materials. Chemical oxidation is an extensively investigated process for treating WW effluents involving LFL (Rajasulochana and Preethy, 2016). The recent focus has been on advanced oxidation processes (AOPs) such as O3,H_2O_2, and electron or UV beams. However, the disadvantages of AOPs are briefed in the high energy requirements and the large amounts of antioxidants to degrade the pollutants, besides the high cost (Gao et al., 2015).

Fenton's reagent and ferrous sulfate catalyze $H2O_2$ more frequently than any other AOP. The Fenton procedure, which may eliminate hazardous chemical materials and enhance the decomposition of OM, is frequently utilized as either pretreatment or posttreatment for LFL (Torretta et al., 2017). The Fenton treatment of LFL involves reactions such as neutralization, oxidation, flocculation, solid–liquid separation, and reducing organic content, color, and odor

(Zhang et al., 2005). Zhang et al. (2006) found that the Fenton treatment removed 79% of COD (1 g L^{-1}) in a diluted LFL.
Furthermore, Lin and Chang (2000) examined the application of an electro-Fenton approach, a mix of an electrochemical process and Fenton's oxidation, for treating LFL produced in a MSW landfill in northern Taiwan. Two anodic and cathodic electrodes were put into LFL samples, and 500 and 1500 mg L^{-1} of H_2O_2 were added to the electrolytic cell before applying an electrical current. Fenton's reagent was produced by continuously adding $H2O_2$ to LFL; thus, a COD elimination efficiency of >85% was reported, with 67% attributable to the electro-Fenton method. The primary benefit of chemical oxidation is the quick and full mineralization of OM without producing hazardous byproducts. However, its downsides include high running costs and a need for the pH of the LFL to be lowered to 6 to be potent (Lin and Chang, 2000).
A previous study employing physicochemical techniques for LFL therapy has demonstrated promise (Braga et al., 2020). Nonetheless, this treatment approach is ideally suited for LFL formed in older landfills with low BOD:COD ratios and low amounts of biodegradable organics. In particular, physiochemical treatment techniques are optimal for removing NH_3, PO4−3, NO3−, and HMs from LFL but are inefficient for removing BOD and COD (Christensen, 2010, Kamaruddin et al., 2017).

6.5.6. Electrochemical procedure

Electrochemical therapy comprises electrocoagulation, chemical, and Fenton oxidation (Rahman et al., 2021). The metal plates used as sacrificial electrodes (anode and cathode) may comprise the same or various materials. Aluminum (Al+3) and iron (Fe+3) are the most typical electrode materials utilized in this method (Bouhezila et al., 2011).
The electrochemical dissolution of Al+3 or Fe+3 from their respective electrodes generates *in sit*u coagulating reagents. When no external chemicals are employed, the amount of sludge generated is decreased. The generated coagulants destabilize suspended particles and adsorb dissolved pollutants (Siddiqi et al., 2022).

6.6. MF

The selective permeability of ions and molecules via a thin film barrier supports MF. The selective barrier removes contaminants from bulk fluids by permitting only specific molecules to flow from a liquid containing harmful molecules and ions (Luo et al., 2020).
The microfiltration procedures are MF, reverse osmosis, ultrafiltration, and nanofiltration based on the tiny particle sizes tolerated (Cancino-Madariaga et al., 2011, Teng et al., 2021).
MF removes contaminants from pretreated leachate effluents (Vaccari et al., 2019). Nonetheless, the process faces various construction and operation challenges, like pretreatment requirements; enormous energy costs due to the pressure-driven pumping scheme; membrane fouling and scaling by retained organics, inorganics, colloidal particles, molecules, and ions of HMs; foulant cleaning and retrofitting of the membrane; improper removal of concentrated brine, which poses environmental risks; and inappropriate disposal of concentrated brine (Hube et al., 2020).

6.7. Combination therapies

In certain instances, a combination of biological and physiochemical treatments for LFL was successful (Torretta et al., 2017). In addition, recent experiments propose that removing diverse substances utilizing each procedure leads to overall high removal efficiency (Gao et al., 2015).
As the structure of LFL is significantly impacted by several factors, such as the age of the landfill, the number of stages/sections in the landfill, the sorts of trash, and the moisture content,

the LFL produced may fall into two or more classes (Babaei et al., 2021). Consequently, certain therapy choices are appropriate, and a mix of therapies is typically necessary. This section will focus on beneficial therapies (Torretta et al., 2017). Table 5 summarizes the efficacy of physiochemical therapies and biological treatments in eliminating LFL.

Table 5. Efficiency of the combined physiochemical and biological treatments on the removal of landfill leachate.

6.7.1. Anaerobic SBR (ASBR) and adsorption

Lim et al. (2016) studied the treatment of LFL utilizing ASBR paired with adsorption into the zeolite. Initial levels of NH4+-N and COD in the influent LFL utilized in this investigation were 1800 and 3200 mg L^{-1}, respectively. The influent LFL was taken from an MSW dump site in Johor, Malaysia. The ASBR was inoculated with *Brevibacillus panacihumi* strain ZB1 and aerated at 1.0–1.2 cm s^{-1} up-flow velocity. After 7 d of operation, the bioreactor effluent was put into a zeolite column containing 10% by weight. The findings of their investigation demonstrated a 65% and 30% removal efficiency for NH4+-N and COD, respectively, during the 7 d ASBR experiment. Moreover, adsorption decreased by 96% for NH4+-N and 43% for COD, respectively. The biological and physical treatments removed Al, Cr, Mg, and vanadium from LFL (Mojiri et al., 2021).

6.7.2. Man-made adsorption and swamps

The LFL cotreatment with municipal WW (1:5) utilizing a constructed wetland (CW) and two adsorbents—ZELIAC composite (Portland cement, zeolite, rice husk ash, limestone, and AC). Three young, fresh, and healthy plants were placed in the CW with two layers of adsorbent substrate. The analysis of influent LFL samples showed high amounts of COD (2301 mg L^{-1}), nickel (4.6 mg L^{-1}), NH4+-N (627 mg L^{-1}), and cadmium (2.5 mg L^{-1}). The biodegradability ratio was low (BOD5/COD = 0.20), and the influent BOD5 concentration was 461 mg L^{-1} (Mojiri et al., 2016). The composite of influent LFL and WW was put into the CW, and sewage samples were recovered post varying contact durations (12, 42, and 74 h). All the investigated components, including Ni, Cd, COD, and NH_3, exhibited high clearance rates of 88–99% after 50 h of response. This technique proved effective in treating LFL mixed with WW; however, the removal rates dropped as the leachate concentration grew. This procedure can repair low-strength leachate but might not affect high-strength LFL (Reshadi et al., 2020).

7. Potential uses of treated LFL

By 2030, the global population is anticipated to exceed 8.5 billion, and solid waste output will reach 2.59 billion tons. This will exacerbate the already precarious state of ecology and climate. There is a pivotal need for technologies to recycle solid waste. New treatment methods have been investigated to treat solid waste, and assess the economic feasibility of changing trash into useful energy *i.e.,* electricity. For instance, CH_4 extracted from municipal solid waste buried in landfills in Delhi, India may power 8–18 million homes and generate 7140 gigawatt h (gWh), with a potential of 31,346 and 77,740 gWh by 2030 and 2060, respectively (Siddiqi et al., 2022). As animal feeds, valorizing solid/food waste by anaerobic digestion systems may replace 61.46% of natural gas and 38.54% of coal in the United Kingdom and reduce land usage by 1.8 million hectares. In addition, the levelized cost of electricity using solid and anaerobic digestion waste-to-energy systems of \$0.04 kilowatt h^{-1} (kWh^{-1}) and \$0.07 kWh $^{-1}$, with a payback period of 0.73–1.86 years and 1.17–2.37 years, respectively. Nonetheless, landfill waste treatment systems are still ineffective, particularly for handling food waste containing over 60% water (Ahmed and Lan, 2012, Siddiqi et al., 2022).

8. Applications of LFL solid waste

Transforming municipal food waste into products with added value offers great promise. However, effective conversion technologies are still absent, and the technical constraints are mostly owing to the variety of waste (Sindhu et al., 2019). Future researchers may have to use other approaches to prevent the inconsistent impacts of waste heterogeneity.
Integrated pyrolysis machines for decomposing municipal solid waste are environmentally benign despite being expensive and requiring a great deal of thermal energy (Hasan et al., 2021) The high ash content of agricultural solid waste, such as soy straw, makes biomass furnaces challenging to run. However, co-firing more biomass with lower ash concentrations may overcome this problem (Gonzalez et al., 2022). Processing agricultural solid waste lignocellulose also has the potential to generate a wide range of chemical and biological substances that may be used in textile, materials, biomedical, and pharmaceutical sectors. Further investigation on excessive water usage, energy consumption, hazardous chemicals, and collection, transport, and disposal of lignocellulose should be taken into consideration (Gonzalez et al., 2022).
Industrial waste has a significant potential for recovery, and extracting rare precious metals from trash is one option to alleviate the resource crunch. When extracting precious metals from solid waste, care must be given to minimize secondary contamination by managing crucial technical parameters in order to lower the amount of newly valuable trash and simultaneously not to lose valuable metals (Wu et al., 2022). According to Haile et al. (2021), paper mill waste can manufacture essential engineering materials, such as C fibers, bioplastics and fibers, cellulose nanocrystals, and biocomposites. Biomass or biomass waste for different engineering applications and biomaterials manufactured using appropriate and practical techniques may also be used to develop multifunctional bio-based products for a wide range of conventional, high-performance, and intelligent applications (Akor et al., 2021).
Future uses of municipal solid waste with added value will need to overcome technological limits and build integrated solar heating systems. The value addition and use of agricultural solid waste must also investigate novel chemicals and technologies to prevent resource waste (Akor et al., 2021). In addition, the parameter control of the present value-added technology for industrial solid waste must be strengthened, and applying the direction of bacterial culture must be investigated (Akor et al., 2021).

8.1. Methodologies for evaluating the economic viability

Solis and Silveira (2020) assessed nine technologies for the chemical treatment of household plastics via a technological readiness assessment approach, eventually selecting three technologies based on extensive research and development centers to investigate economies of scale. The authors concluded that the readiness level technology might evaluate data about LFL wastes for economic feasibility analyses. Moreover, the economic viability of investing in waste technologies with added value may be established by evaluating the return on investment and net present value (Solis and Silveira, 2020).

8.2. Solid waste processing

Solid waste processing systems cover all actions that seek to reduce the negative impacts on health, environment and economy. Recycling and sorting is the first crucial stage in LFL waste valorization and application (Yang et al., 2023). Policies and infrastructure should enhance the rate of trash recovery and sorting precision. Consideration might include the duty to recycle and segregate garbage into the citizens’ code of conduct to achieve quick results and increase residents’ sense of ownership. The government can also enhance and support businesses to implement recycling programs for marketed items related to trash recycling behavior of customers. Thus, a particular type of industrial solid waste might be classified and characterized in depth (Yang et al., 2023).

According to Wiśniewska et al. (2022), the green desulfurization of scrap tires is consistent with the circular economy, and the manufacture of rubber-based materials for high-value end ground tire markets will increase according to current research trends. However, thorough sorting and characterization of scrap rubber before usage may considerably enhance the repeatability of the process and the performance qualities of the resulting recycled rubber products. Koskinopoulou et al. (2021) proposed that the installation of autonomous robotic systems for garbage recycling may be possible in the future with automated sorting and physical sorting of recyclables by material type. If artificial intelligence can be effectively applied to the field of waste recycling, the efficiency and accuracy of recycling will be greatly enhanced, and the garbage will be prepared for reuse (Wiśniewska et al., 2022).

9. Limitations and solutions for effective LFL management strategies

The operational difficulties associated with various LFL treatments are listed in Table 6.

9.1. Financial shortages and policy mismatches are limitations and major national challenges

All localities believe that the financial shortage and the mismatch of the policies are issues that encounter the policy implementation process in solid waste management. For examples, these represent limitations and major challenges for Vietnam. The population is densely populated, the per capita income is low, and a huge amount of waste is generated; therefore, extensive labor and operating costs are needed for the waste collection and the treatment system (Van Den Berg and Duong, 2018). Covering the full cost would be difficult for the government. Moreover, calling for investment is still limited in Vietnam. Localities also evaluate the policies as unsuitable to reality. National policies are announced and must be implemented throughout the territory of Vietnam. Each locality has; however, different developmental situations and natural, social, environmental, and human conditions. Vietnam is the 65th country in the world in terms of area, but its population ranks 15th.

In relation to the state management system, Vietnam is divided into 63 provinces or management units (Van Den Berg and Duong, 2018). Each province has its area, population, and management system. Therefore, using the same policy in various places offers practical challenges. It is essential to comprehend a locality's particular circumstances and create suitable programs. Nevertheless, implementing this strategy is particularly challenging in light of the current crisis. These limitations and challenges hinder the economic development of this country (Van Den Berg and Duong, 2018).

9.2. Policy implementation in certain countries

Some countries such as Malaysia and Brazil have been chosen to compare the implementation situation and draw lessons from the similarities in the context of waste management. Japan and Taiwan were also included to showcase how these countries overcome the difficulties associated with implementing waste management policies (Abas and Wee, 2014, Cetrulo et al., 2018).

Abas and Wee (2014) argued that Malaysia has not implemented an effective policy in environmental management; thus, leading to a number of environmental and human health issues. Among others, there are four major reasons of poor governance, lack of commitment among stakeholders, ineffectiveness in monitoring and enforcement, and neglected of social dimension which are behind the issues of ineffective solid waste management policy implementation in Malaysia. Therefore, policy implementation for solid waste management has become a priority in Malaysia. To tackle this crisis, an institutional capacity building framework which includes stakeholders of municipal authorities, administration, corporate bodies, non-governmental organizations and educational should be key players in implementing effective solid waste management policy (Abas and Wee, 2014).

In Brazil, waste management was ineffective because of inappropriate policies and national plans. Inefficiencies in policy implementation in Brazil can be related to several reasons, including financial shortage, inadequate professional training, lack of human resources, poor technology, weak awareness, low community participation, and mild punishment (Cetrulo et al., 2018). Strengthening strict supervision in policy implementation, providing adequate resources, enhancing the importance of the capacity of environmental managers, developing national programs for sanitary landfills, and waste minimization plans are some of the suggested solutions for Brazil (Cetrulo et al., 2018).

Japan has also experienced many years of deadlock in solving solid waste management. In the 1960s, swift industrialization occurred, and the waste increased rapidly. Immediately, Japan introduced strategies to improve this situation through the act on emergency measures concerning the development of living environment facilities (1963), the waste management act (1970), and the revision of the waste management act (1976) (Japan Ministry of the Environment, 2014).

These policies also faced various difficulties and had to be revised many times in the following years to suit the country's situation. Determining the right goals, direction, and implementation strategy from the government are important factors that helped Japan overcome the waste pollution crisis and become a model for sustainable development (Japan Ministry of the Environment, 2014).

10. Future development of LFL treatment

Leachates from landfills are a severe danger to the ecology. Numerous groundwater contamination instances have happened due to insufficient monitoring and effective remediation methods. The metrics tested at a landfill site are pH, alkalinity, electrical conductivity, COD, TDS, BOD, NH4+-N, chloride, nitrate, and sulfate. Additionally, it is essential to examine potentially dangerous compounds, such as polymers, halogenated chemicals, and poisonous HMs, among others (Lindamulla et al., 2022). In addition to their economic and ecological benefits, technologies based on microorganisms offer an attractive way to remove, treat, or detoxify leachate toxins (García-Pacheco et al., 2022, Lindamulla et al., 2022, Yan et al., 2023). Studying how microorganisms successfully remediate leachate has several advantages, but there is still much to learn in this area. Modern techniques (e.g., metagenomics) might be used to adapt new microorganisms that remediate leachate effectively. Biological remediation of leachates from middle-aged landfills has been proven to be effective in the past. Biological, physical, and chemical strategies can be used to boost the treatment efficiency of leachates from old landfill sites. In addition to phytoremediation and aerated lagoons, pollutants near landfills can be naturally eliminated through phytoremediation. Reuse and recycling must be closely studied to minimize the contemporary environment's solid and hazardous waste components (Fig. 3). Landfills must be meticulously designed. Proper monitoring, risk assessment, and leachate treatment utilizing cutting-edge technology are required to prevent major ecological harm and avoid leachate toxins from contaminating soil and groundwater (Yan et al., 2023).

Certain waste items must always be disposed of unless high concentrations of waste avoidance, reuse, biodegradable, safe goods, and recycling are attained (Fig. 3). Therefore, sustainable landfills should address this issue. When garbage is safely integrated into the ecosystem, it is considered that a landfill is entirely sustainable. Using this study's framework, built from a comprehensive literature review, all stakeholders engaged in the landfill industry may analyze the environmental consequences of landfills.

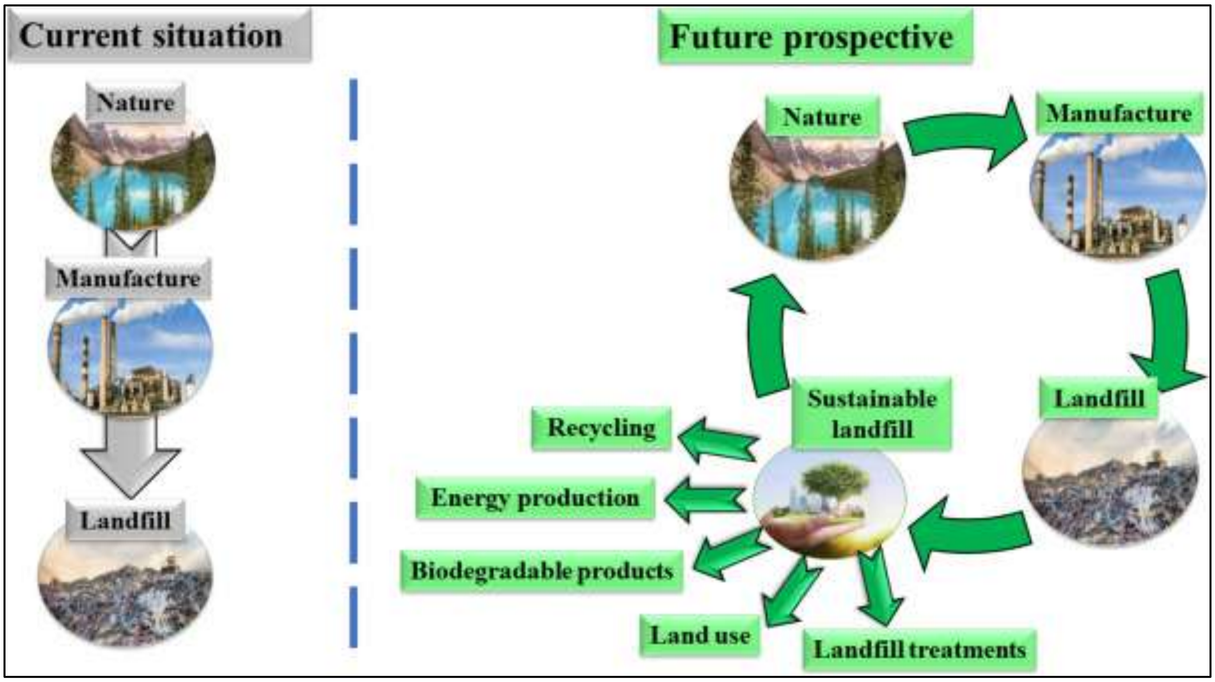

Fig. 3. Current situation and future prospects of landfilling. A sustainable landfill model can be implemented on waste materials safely assimilated into the surrounding environment after being reused, recycled or treated with biological, thermal or other processes and managing gas-related problems to minimize the environmental pollution impact.

11. Concluding remarks

Leachate pollution produces significant ecological and public health problems, especially in the developing countries with ineffective waste treatment strategies despite implementing minimum removal requirements. Leachate harms groundwater aquifers and terrestrial ecosystems, emitting dangerous pollutants and greenhouse gases. Therefore, innovative, cost-effective, sustainable, and eco-friendly leachate treatment technologies are required to reduce energy consumption, sludge production, and toxin formation and recover organic, inorganic, and xenobiotic chemicals in a harmless state to maximize their benefits for future use.

The optimal treatment for leachate is often decided by its characteristics, technical applicability, potential limits, needed effluent limit, cost-effectiveness, regulatory requirements, and long-term ecological effects. Using readiness level technology is essential for determining the recycling rate and net present value to undertake an exhaustive economic feasibility assessment of solid waste value addition and application. Moreover, recycling sorting technology can increase the rate of garbage recycling. A policy structure efficiently facilitates garbage recycling. Involvement of governmental, non-governmental and educational stakeholders is highly recommended for effective policy implementation in the ground.

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

https://www.sciencedirect.com/science/article/pii/S0048969723076568?utm_source=chatgpt.com
https://doi.org/10.1016/j.scitotenv.2023.169026
Science of The Total Environment. Volume 912, 20 February 2024, 169026
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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.

Abstract
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.

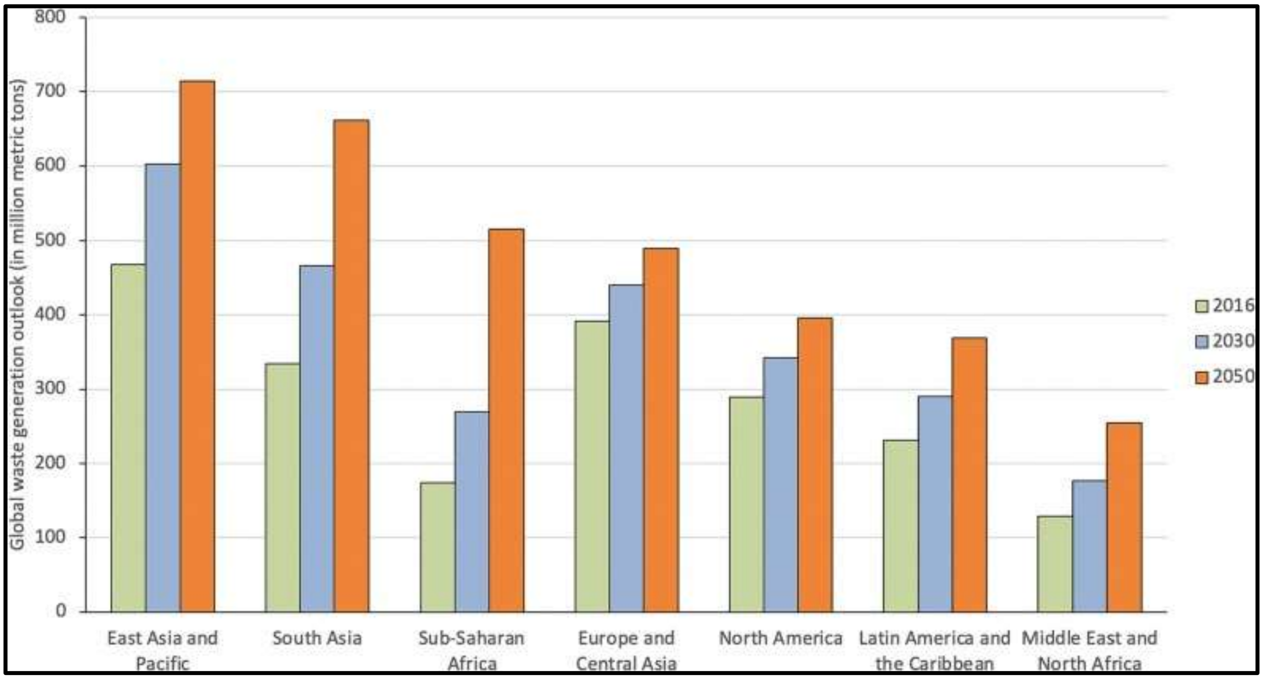

Fig. 1. Projection of waste generation worldwide in 2016, 2030, and 2050, by region (in million metric tons).

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 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).

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).

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).

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

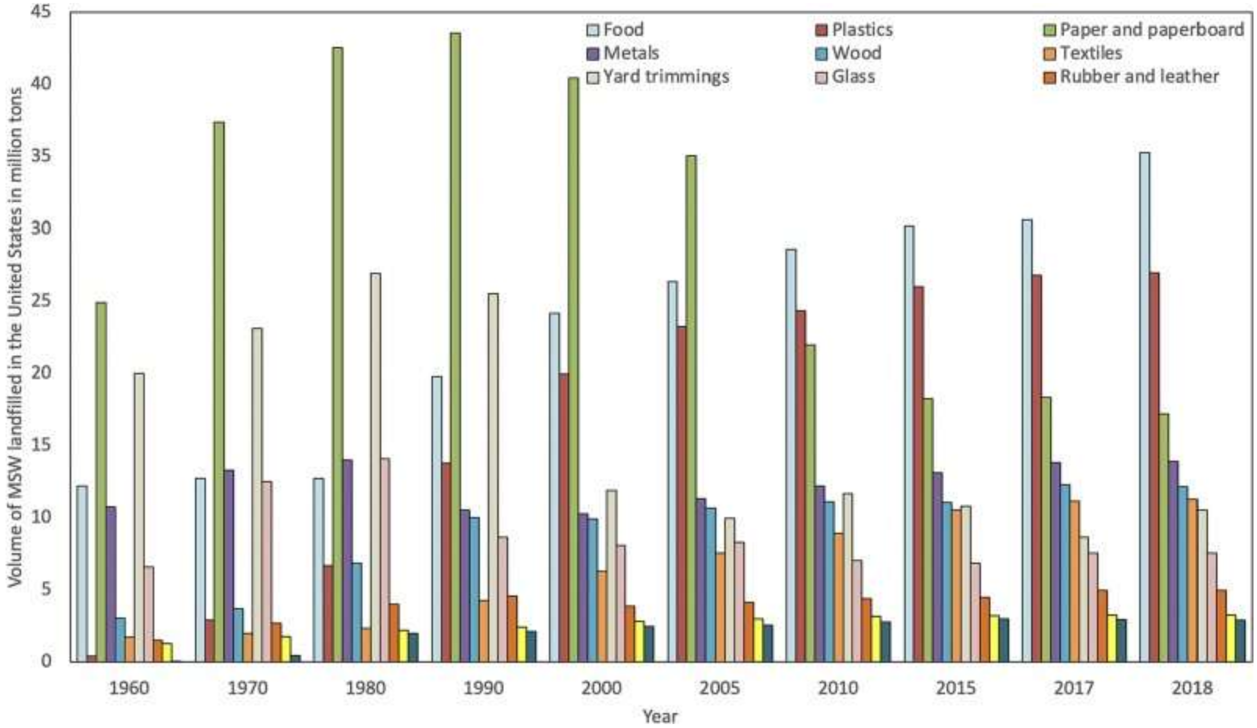

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Fig. 3. Volume municipal solid waste landfilled in the USA from 1960 to 2018, by material (Source: Environmental Protection Agency (Environmental Protection Agency, 2021)). It is possible to highlight the decrease in paper and paperboard waste generation, 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.

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 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 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 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 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 ≤ −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 microorganism 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.

7. Conclusions: This paper presents an 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. 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. However, none of these technologies are reduced to practice as of this date.