Springer Water



Salah Souabi
Abdelkader Anouzla *Editors*

Landfill Leachate Treatment Techniques

Springer

Springer Water

Series Editor
Andrey G. Kostianoy
Russian Academy of Sciences, P. P. Shirshov Institute of Oceanology, Moscow, Russia

Editorial Board
Angela Carpenter
School of Earth and Environment, University of Leeds, Leeds, West Yorkshire, UK

Tamim Younos
Green Water-Infrastructure Academy, Blacksburg, VA, USA

Andrea Scozzari
Institute of Information Science and Technologies (CNR-ISTI), National Research Council of Italy, Pisa, Italy

Stefano Vignudelli
CNR—Istituto di Biofisica, Pisa, Italy

Alexei Kouraev
LEGOS, Université de Toulouse, Toulouse Cedex 9, France

The book series Springer Water comprises a broad portfolio of multi- and interdisciplinary scientific books, aiming at researchers, students, and everyone interested in water-related science. The series includes peer-reviewed

monographs, edited volumes, textbooks, and conference proceedings. Its volumes combine all kinds of water-related research areas, such as: the movement, distribution and quality of freshwater; water resources; the quality and pollution of water and its influence on health; the water industry including drinking water, wastewater, and desalination services and technologies; water history; as well as water management and the governmental, political, developmental, and ethical aspects of water.

Editors
Salah Souabi and Abdelkader Anouzla

Landfill Leachate Treatment Techniques



Editors
Salah Souabi
Faculty of Science and Technology of Mohammedia,
University of Hassan II Casablanca, Casablanca, Morocco

Abdelkader Anouzla
Faculty of Science and Technology of Mohammedia,
University of Hassan II Casablanca, Casablanca, Morocco

ISSN 2364-6934 e-ISSN 2364-8198
Springer Water
ISBN 978-3-031-63156-6 e-ISBN 978-3-031-63157-3
https://doi.org/10.1007/978-3-031-63157-3

© The Editor(s) (if applicable) and The Author(s), under exclusive license to Springer Nature Switzerland AG 2024

This work is subject to copyright. All rights are solely and exclusively licensed by the Publisher, whether the whole or part of the material is concerned, specifically the rights of translation, reprinting, reuse of illustrations, recitation, broadcasting, reproduction on microfilms or in any other physical way, and transmission or information storage and retrieval, electronic adaptation, computer software, or by similar or dissimilar methodology now known or hereafter developed.

The use of general descriptive names, registered names, trademarks, service marks, etc. in this publication does not imply, even in the absence of a specific statement, that such names are exempt from the relevant protective laws and regulations and therefore free for general use.

The publisher, the authors and the editors are safe to assume that the advice and information in this book are believed to be true and accurate at the date of publication. Neither the publisher nor the authors or the editors give a warranty, expressed or implied, with respect to the material contained herein or for any errors or omissions that may have been made. The publisher remains neutral with regard to jurisdictional claims in published maps and institutional affiliations.

This Springer imprint is published by the registered company Springer Nature Switzerland AG
The registered company address is: Gewerbestrasse 11, 6330 Cham, Switzerland

Preface

Municipal solid waste landfills generate highly polluted landfill leachate, containing heavy metals, organic compounds, and hazardous chemicals. Landfill leachate is the liquid from water percolating through a waste stockpile, and its composition varies depending on the waste and civilization. Treatment of landfill leachate is crucial to prevent environmental pollution and ecotoxicity, but it is complex and costly due to high pollution loading and variable volume.

Waste management in general, and municipal solid waste in particular, is one of the significant challenges facing our societies. With ever-increasing and diversified consumption worldwide, waste production is steadily increasing in quantity, creating enormous risks for the environment and human health. The mechanisms involved in leachate production at landfill sites are biological and physicochemical. The liquid residues of these wastes, following their fermentation and bio-physical-chemical evolution, are a source of juices that penetrate the layer of waste and are enriched with soluble elements of the waste. Rain falling on the landfill and percolating through the thickness of the waste will increase the quantity of leachate. Reducing pollution from these discharges using various discharge treatment techniques offers several advantages: To mitigate the potential harm caused by contaminating the ecological system through overflowing or seepage, as well as the contamination of the groundwater reservoir, it is essential to take preventive measures and implement sustainable practices to safeguard the environment from adverse effects. To enhance the decomposition process of organic substances within the waste accumulation, consequently leading to a rise in the generation of biogas, a concerted effort can be

made through various means, such as optimizing environmental conditions, adjusting microbial populations, and implementing efficient waste management practices. To reduce the pollutants in leachates, one can implement strategies to enhance the degradation process of organic matter present in waste materials, thus decreasing the concentration of harmful substances seeping into the environment. The book's contents are comprised of a variety of chapters containing case studies that delve into topics such as leachate collection methods employed at landfill sites, the evaluation of pollution levels resulting from leachate, the environmental consequences associated with such pollution, and lastly, methods for treating pollution to maintain control over it. These case studies demonstrate the least expensive and most commonly utilized biological and physicochemical treatment approaches that effectively diminish pollution levels. Storing leachate in anaerobic conditions within landfill sites has shown its ability to decrease pollution, especially in regions with abundant sunshine. Furthermore, certain countries with elevated temperatures have used stored leachates to water solid waste that has undergone composting treatments. Identifying techniques specifically tailored to address the treatment of these discharges is essential, taking into account the initial conditions and the specific requirements of the surrounding environment. This responsibility is fundamental to our ongoing, urgent efforts to enhance waste management practices. Numerous measures are available to alleviate the negative impacts caused by pollution, although many of these solutions, such as evaporation, reverse osmosis, or mechanization, come with substantial costs. The book encompasses a diverse array of chapters featuring case studies on leachate collection procedures at landfill sites, evaluation of pollution levels stemming from leachate, environmental repercussions, and methods for pollution control. The case

studies showcase the most cost-effective and commonly employed biological and physicochemical treatment methods that successfully reduce pollution levels. Storing leachate in anaerobic conditions within landfill sites has effectively decreased pollution levels, particularly in regions with ample sunlight. Additionally, in certain nations with high temperatures, stored leachates are utilized to water solid waste treated through composting. In conjunction with anaerobic or aerobic biological treatment techniques, the book extensively covers physicochemical treatment approaches such as coagulation-flocculation, flotation, and chemical oxidation, providing in-depth real-life case studies. These techniques yield treated water that can be reused for irrigation purposes. Selecting the appropriate leachate treatment method is a multifaceted decision-making process that necessitates consideration of various factors, including waste quality, environmental impacts, costs, and feasibility of implementation. As a comprehensive resource, this book is designed to aid in selecting a treatment technology that offers the lowest capital and operational expenses, guiding individuals through this critical decision-making process and providing the necessary support. Effective treatment methods for landfill leachate include combining biological, chemical, and physicochemical processes to meet treatment standards and minimize environmental and health impacts. Therefore, suitable and efficient treatment for leachate is essential. Recent technologies for leachate treatment removal are highlighted.

Physicochemical methods like coagulation-flocculation, adsorption, and membrane filtration are effective for well-aged leachate while biological methods like membrane bioreactors and activated sludge processes work best for young leachate.

Combining treatment methods shows promise in removing metals and optimizing removal efficiency, but

further research is needed to enhance the effectiveness of these techniques.

This book will provide a comprehensive reference for up-to-date knowledge about leachate treatment. Consequently, individuals pursuing a master's or doctoral degree, along with academics, researchers, and students, will be able to comprehend the latest advancements in Landfill Leachate Treatment Techniques, thereby bolstering their investigations. Additionally, this publication will serve as a source of inspiration, guiding readers on practical approaches to address environmental pollution issues resulting from the contamination of freshwater and agricultural soils by leachate, utilizing a diverse array of technologies.

Abdelkader Anouzla
Salah Souabi
Casablanca, Morocco

Contents

1 Municipal Solid Waste Landfill Leachate Treatment: A Review
Anshu Gupta, Akanksha Verma and Paulraj Rajamani

2 An Overview of Existing Municipal Landfill Leachate Treatment Techniques: Opportunities and Challenges
Mohammad Toha, Sadia Sikder and Md. Mostafizur Rahman

3 Treatment Techniques Adopted to Developing Countries
Yasmin Cherni, Sarra Hechmi, Mohamed Ali Wahab, Samira Melki and Ismail Trabelsi

4 Physicochemical and Biological Processing of Municipal Landfill Leachates
Wafa Hassen, Bilel Hassen, Marwa El Ouaer, Solwen Kouki and Abdennaceur Hassen

5 Physicochemical Treatment Approaches for Leachate Treatment
S. R. M. A. K. D. Rajapaksha, E. M. Y. C. Ekanayake and K. G. N. Nanayakkara

6 Efficient Treatment of Landfill Leachate Via Coagulation-Flocculation: Optimizing Process Conditions
Roukaya Bouyakhsass, Safaa Khattabi Rifi, Younes Abrouki, Abdelaziz Madinzi, Abdelkader Anouzla, Nor-edine Abriak, Tonni Agustiono Kurniawan, Kastali Malika and Salah Souabi

7 Leachate Treatment Techniques: A Review of Oxidation and Physical Chemical Methods
Safaa Khattabi Rifi, Roukaya Bouyakhsass, Abdelaziz Madinzi, Alaa AEl Din Mahmoud,

Tonni Agustiono Kurniawan, Abdelkader Anouzla, Noredine Abriak, Younes Abrouki and Salah Souabi

8 Membranes in Landfill Leachate Treatment: Evaluation of the Generated Concentrate
Ronei de Almeida and Juacyara Carbonelli Campos

9 A Review of Strategies for Managing Membrane Concentrate from Landfill Leachate Treatment Facilities
Letícia Barbosa Alves, Ronei de Almeida and Juacyara Carbonelli Campos

10 Advanced Oxidative Processes as Treatment for Landfill Leachate
Siara Silvestri, Mariana Islongo Canabarro, Lizet Mirian Guevara Roman and Elvis Carissimi

11 Statistical Evaluation of the Effectiveness of Electrocoagulation Using Iron Electrodes for the Treatment of Leachate Effluent
Rosangela Bergamasco, Mayara Caroline Santos Godoi, Joseane Debora Peruco Theodoro and Paulo Sergio Theodoro

12 Electrochemical Processes for Stabilized Landfill Leachate Treatment: A Brief Review
Arseto Yekti Bagastyo and Anita Dwi Anggrainy

13 Mini-Review on Aged Refuse Bioreactors for Landfill Leachate Management
Ronei de Almeida and André Luís de Sá Salomão

14 Assessment of the Effectiveness of Ouled Berjal Landfill Leachate Treatment by Sequential Batch Reactor Technique: Elimination of Nitrogen Pollution, Phenolic Compounds, and Phytotoxicity
Hajar Bakraouy, Khalid Digua, Moncif Sabar, Roukaya Bouyakhsass, Abdelaziz Madinzi,

Tonni Agustiono Kurniawan, Abdelkader Anouzla, Younes Abrouki, Amane Jada and Salah Souabi

15 Filtration Technique for Leachate Using Recycled Coarse Aggregate

J. Logeshwari and T. Anstey Vathani

16 Performance Comparison of Conventional Method and Ultrafiltration in the Pretreatment of Landfill Leachate

Mohamed Zait and Mohamed Taky

Contributors

Nor-edine Abriak
IMT Lille Douai, GCE, Douai, France

Younes Abrouki
Department of Process Engineering and Environment Mohammedia, Hassan II University of Casablanca-Faculty of Sciences and Techniques of Mohammedia, Mohammedia, Morocco
Laboratory of Spectroscopy, Molecular Modeling, Materials, Nanomaterial, Water and Environment, CERNE2D, Faculty of Science, Mohammed V University in Rabat, Rabat, Morocco

Tonni Agustiono Kurniawan
College of the Environment and Ecology, Xiamen University, Xiamen, China

Anita Dwi Anggrainy
Research Center for Infrastructure and Sustainable Environment, Institut Teknologi Sepuluh Nopember, Surabaya, Indonesia

Abdelkader Anouzla
Faculty of Sciences and Techniques of Mohammedia, Department of Process Engineering and Environment Mohammedia, Hassan II University of Casablanca, Casablanca, Morocco

T. Anstey Vathani
Vel Tech Rangarajan Dr. Sagunthala R&D Institute of Science and Technology, Avadi, Chennai, India

Arseto Yekti Bagastyo

Department of Environmental Engineering, Faculty of Civil, Planning, and Geo Engineering, Institut Teknologi Sepuluh Nopember, Surabaya, Indonesia
Research Center for Infrastructure and Sustainable Environment, Institut Teknologi Sepuluh Nopember, Surabaya, Indonesia

Hajar Bakraouy
Process and Environmental Engineering Laboratory, Faculty of Science and Technics, Hassan II University, Mohammedia, Morocco

Letícia Barbosa Alves
School of Chemistry, Inorganic Processes Department, Universidade Federal do Rio de Janeiro, Rio de Janeiro, Brazil

Rosangela Bergamasco
Universidade Estadual de Maringá, Maringá, Brazil

Roukaya Bouyakhsass
Faculty of Sciences and Techniques of Mohammedia, Department of Process Engineering and Environment Mohammedia, Hassan II University of Casablanca, Casablanca, Morocco

Juacyara Carbonelli Campos
School of Chemistry, Inorganic Processes Department, Universidade Federal do Rio de Janeiro, Rio de Janeiro, Brazil

Mariana Islongo Canabarro
Universidade Federal de Santa Maria, Santa Maria, Brazil

Elvis Carissimi
Universidade Federal de Santa Maria, Santa Maria, Brazil

Yasmin Cherni
Laboratory of Wastewater Treatment and Valorization, Water Research and Technologies Center, Carthage University, CERTE, Technopark Tourist Route of Soliman Nabeul, Soliman, Tunisia

Ronei de Almeida
School of Chemistry, Universidade Federal do Rio de Janeiro, Rio de Janeiro, Brazil
Department of Sanitary and Environmental Engineering, Rio de Janeiro State University, Rio de Janeiro, Brazil

André Luís de Sá Salomão
Department of Sanitary and Environmental Engineering, Rio de Janeiro State University, Rio de Janeiro, Brazil

Khalid Digua
Process and Environmental Engineering Laboratory, Faculty of Science and Technics, Hassan II University, Mohammedia, Morocco

E. M. Y. C. Ekanayake
Department of Civil Engineering, University of Peradeniya, Peradeniya, Sri Lanka

Mayara Caroline Santos Godoi
Universidade Tecnológica Federal Do Paraná, Londrina, Brazil

Anshu Gupta
Department of Environmental Sciences, Jammu and Kashmir Higher Education Department (JKHED), UT of Jammu and Kashmir (J&K), India

Abdennaceur Hassen

Laboratory of Treatment and Wastewater Valorization, Center of Research and Water Technologies (CERTE), Soliman, Tunisia

Bilel Hassen
Laboratories of Microbiology and Pathology of Aquatic Organisms, National Institute of Navy Sciences and Technologies (INSTM), Salammbô, Tunisia

Wafa Hassen
Laboratory of Treatment and Wastewater Valorization, Center of Research and Water Technologies (CERTE), Soliman, Tunisia

Sarra Hechmi
Laboratory of Wastewater Treatment and Valorization, Water Research and Technologies Center, Carthage University, CERTE, Technopark Tourist Route of Soliman Nabeul, Soliman, Tunisia

Amane Jada
Institute of Materials Science of Mulhouse, Mulhouse, France

Safaa Khattabi Rifi
Faculty of Sciences and Techniques of Mohammedia, Department of Process Engineering and Environment Mohammedia, Hassan II University of Casablanca, Casablanca, Morocco

Solwen Kouki
Laboratory of Treatment and Wastewater Valorization, Center of Research and Water Technologies (CERTE), Soliman, Tunisia

J. Logeshwari

Vel Tech Rangarajan Dr. Sagunthala R&D Institute of Science and Technology, Avadi, Chennai, India

Abdelaziz Madinzi
Faculty of Sciences and Techniques of Mohammedia, Department of Process Engineering and Environment Mohammedia, Hassan II University of Casablanca, Casablanca, Morocco

Alaa AEl Din Mahmoud
Environmental Sciences Department, Faculty of Science, Alexandria University, Alexandria, Egypt

Kastali Malika
Department of Process Engineering and Environment Mohammedia, Hassan II University of Casablanca-Faculty of Sciences and Techniques of Mohammedia, Mohammedia, Morocco

Samira Melki
Laboratory of Wastewater Treatment and Valorization, Water Research and Technologies Center, Carthage University, CERTE, Technopark Tourist Route of Soliman Nabeul, Soliman, Tunisia

K. G. N. Nanayakkara
Department of Civil Engineering, University of Peradeniya, Peradeniya, Sri Lanka

Marwa El Ouaer
Laboratory of Treatment and Wastewater Valorization, Center of Research and Water Technologies (CERTE), Soliman, Tunisia

Md. Mostafizur Rahman

Department of Environmental Science, Bangladesh University of Professionals, Mirpur, Cantonment, Dhaka, Bangladesh
Laboratory of Environmental Health and Ecotoxicology, Department of Environmental Sciences, Jahangirnagar University, Savar, Dhaka, Bangladesh

Paulraj Rajamani
School of Environmental Sciences, Jawaharlal Nehru University (JNU), New Delhi, India

S. R. M. A. K. D. Rajapaksha
Department of Civil Engineering, University of Peradeniya, Peradeniya, Sri Lanka

Lizet Mirian Guevara Roman
Universidade Federal de Santa Maria, Santa Maria, Brazil

Moncif Sabar
Institute of Materials Science of Mulhouse, Mulhouse, France

Sadia Sikder
Department of Environmental Science, Bangladesh University of Professionals, Mirpur, Cantonment, Dhaka, Bangladesh

Siara Silvestri
Universidade Federal de Santa Maria, Santa Maria, Brazil
Universidade Federal de Pelotas, Pelotas, Brazil

Salah Souabi
Faculty of Sciences and Techniques of Mohammedia, Department of Process Engineering and Environment Mohammedia, Hassan II University of Casablanca, Casablanca, Morocco

Mohamed Taky
Laboratory of Advanced Materials and Process Engineering, Faculty of Sciences, Ibn Tofail University, Kenitra, Morocco

Joseane Debora Peruco Theodoro
Universidade Tecnológica Federal Do Paraná, Londrina, Brazil

Paulo Sergio Theodoro
Universidade Estadual Do Oeste Do Paraná Toledo, Toledo, Brazil

Mohammad Toha
Department of Environmental Science, Bangladesh University of Professionals, Mirpur, Cantonment, Dhaka, Bangladesh

Ismail Trabelsi
Laboratory of Wastewater Treatment and Valorization, Water Research and Technologies Center, Carthage University, CERTE, Technopark Tourist Route of Soliman Nabeul, Soliman, Tunisia

Akanksha Verma
Shyam Lal College (Evening), University of Delhi, Delhi, India

Mohamed Ali Wahab
Laboratory of Wastewater Treatment and Valorization, Water Research and Technologies Center, Carthage University, CERTE, Technopark Tourist Route of Soliman Nabeul, Soliman, Tunisia

Mohamed Zait

Laboratory of Advanced Materials and Process Engineering, Faculty of Sciences, Ibn Tofail University, Kenitra, Morocco

© The Author(s), under exclusive license to Springer Nature Switzerland AG 2024
S. Souabi, A. Anouzla (eds.), *Landfill Leachate Treatment Techniques*, Springer Water
https://doi.org/10.1007/978-3-031-63157-3_1

1. Municipal Solid Waste Landfill Leachate Treatment: A Review

Anshu Gupta[1], Akanksha Verma[2] and Paulraj Rajamani[3] ✉
(1) Department of Environmental Sciences, Jammu and Kashmir Higher Education Department (JKHED), UT of Jammu and Kashmir (J&K), India
(2) Shyam Lal College (Evening), University of Delhi, Delhi, India
(3) School of Environmental Sciences, Jawaharlal Nehru University (JNU), New Delhi, India

✉ **Paulraj Rajamani**
Email: paulrajr@yahoo.com

Abstract
One of the main problems posed by municipal solid waste landfills is the generation of leachate and the main issue related to landfill leachate is its treatment, especially in developing countries. Landfill leachate is highly polluted discharge containing high loads of heavy metals, organic compounds, ammonia (NH_3-N), chemical oxygen demand (COD), toxic chemicals, pathogens, and various other hazardous chemicals. It must be treated and appropriately disposed in order to avoid the environmental pollution and ecotoxicity caused by its disposal. The type of method selected for treatment depends upon the quality as well as quantity of leachate. The composition as well as quantity of leachate depends upon several factors like type of landfill, age of landfill, seasonal variations, degree of compaction, climatic conditions, precipitation, mode of operation etc. Therefore, the development of an efficient method for treatment of landfill leachate is very crucial. For the treatment process of landfill leachate, several methods including physicochemical, biological, chemical, and physical methods have been used. However, these methods are very complex and costly as they require various processes. Also, due to high pollution loading, complex chemical composition, and seasonally variable volume of leachate the treatment process become very difficult. Therefore, it becomes necessary to implement a combination of different methods including biological, chemical, and physicochemical processes for the effective treatment of leachate. The main objective behind treatment of landfill leachate is to make leachate meet the treatment standards so that it poses least impact on the environment as well as human health after its disposal.

Keywords Municipal solid waste – Landfill leachate – Leachate treatment – Ecotoxicity – Treatment standards

1.1 Introduction

Among the several methods of managing solid waste, landfilling is the most widely used and preferable method [34]. It is preferred due to several factors including technical feasibility, ease of operation, requirement of minimum supervisions and technology, as well as low operation expenditure [45]. The infiltrating water which passes through the solid waste landfill and transfers the contaminants from solid phase to liquid phase results in the formation of landfill leachate. Due to the heterogenous nature of the waste and because of the different compaction densities that will be encountered, water will be able to percolate through and appear as leachate at the base of the site [18]. The physical appearance of leachate is yellow or blackish colored and the smell is acidic and offensive when it emerges from a typical landfill. The quality of landfill leachate is affected by various factors, such as the waste type, operational conditions, climate, hydrogeology, and age of landfill [24]. The composition of landfill leachates varies greatly depending on the age of the landfill [1, 60]. The content of pollutants in leachate is usually high in the early years of operation and gradually decrease with the time [60]. In large parts of Asia, landfill characteristics are influenced by the monsoon climate, which includes the characteristic differences between the rainy season and dry seasons. Leachate in dry season is concentrated because of the evaporation whereas in the rainy season, large amount of leachate is produced with low concentration of pollutants [89, 93]. For the proper and efficient operation of leachate treatment, evaluation of seasonal variation plays an important role [59, 85].

1.2 Properties/Characteristics of Landfill Leachate

Landfill leachate is characterized by conventional parameters comprising of chemical oxygen demand (COD), total organic carbon (TOC), biochemical oxygen demand (BOD), suspended solids, pH, ammonia (NH_4^+-N) and heavy metal concentrations. The ratios of BOD_5/COD and COD/TOC are typical indicators for the biodegradability of organic compounds and the oxidized state of organic carbon [24]. The strength of leachate of one landfill differs from that of another. It is due to the variation in the composition of solid waste. A developing country experiences a higher organic content in the composition of MSW compared to a developed country [73].

Several factors affect the rate and characteristics of leachate from a landfill and exhibit considerable differences. These factors include variations in refuse composition and moisture content, degree of compaction, particle size, hydrology of site, the age of landfill as well as seasonal factors such as temperature, precipitation, and moisture content [23, 42, 56]. As the landfill age increases from young to old, the chemical oxygen demand (COD) (organic/inorganic) in the leachate declines whereas the ammonia nitrogen concentration starts rising. High ammonia concentration in the old landfill leachate is due to the hydrolysis and fermentation of nitrogen-containing fractions of biodegradable refuse substrates [1]. Gupta and Rajamani [33] found the leachate Pollution Index (LPI) values for all the three landfill sites in Delhi indicate that the leachates produced from these sites are highly polluted, hazardous and should be treated before

disposing them to the environment. The individual values of the pollutants should meet the standards set by the authorities for these pollutants. Concentration of Chromium, Pb, BOD_5 and COD in the leachate samples was found beyond the permissible limits set by as per the Gazette of India, for leachate disposal in all the leachate samples [31].

1.3 Environmental Issues Posed by Landfill Leachate

Landfill leachate has been generally found toxic, presenting potential threats to the surrounding environment and ecosystems [10]. The toxicity is generally determined based on its physicochemical properties, with ammonia, chemical oxygen demand (COD) and heavy metals being identified as the major contributors. Wdowczyk and Szymanska-Pulikowska [93] have reported that the toxic nature of the leachate may have resulted from the occurrence of high concentrations of ammonical nitrogen (AN), copper, and chromium. Gupta and Paulraj [33] have also reported a very high concentration of ammonical nitrogen in the landfill leachate samples. In the previous studies [31] have reported that the toxicity of landfill leachate was dependent on the concentration of heavy metals (Pb, Cu), conductivity and organic matter (COD and BOD_5). Gupta and Paulraj [32] have reported that landfill leachate collected in three seasons (summer, winter, and monsoon) caused toxicity in *Vicia faba* seedlings by inhibiting growth, antioxidant enzyme activities as well as a reduction in chlorophyll content. The Transmission Electron Microscopy (TEM) images of cells of *Vicia faba* root tips which were highly stressed by the contaminated leachate, provide new evidence for the presence of autophagic vacuoles in the cells [34]. Gupta et al. [35] have further found that their results evidently indicate the ability of landfill leachate to interrupt mitochondrial redox homeostasis, which might be a likely source for the immunotoxic consequences leading to plausible patho-physiological conditions in humans susceptible to such environmental exposures. If exposed for long term, it promotes an increase and accumulation of chromosomal aberrations affecting the cellular mechanisms, leading to a loss of cellular control, neoplasia, and cellular apoptosis [51]. Gupta et al. [34] have stated that that if leachate is released into the environment without treatment, it can lead to contamination of the aquatic environment in the vicinity of the landfill even at diluted concentrations. Their study also states that the long-term exposure to toxic agents that are present in leachate has a strong impact both on the environmental health and on the organisms living in the ecosystem posing a risk to the organisms exposed. Several methods are included to lessen the harmful health effects of wastewater on the surrounding biota, such as restricting inappropriate disposal of industrial and urban effluents and carrying out proper treatment of industrial and municipal solid waste before dumping it into the water body [91, 92]. Gupta and Rajamani [32] suggested that the most important aspect for the treatment of landfill leachate is controlling its concentration which varies with respect to different seasons so that the proper management of landfills is ensured. Following the findings of these earlier studies, efforts are being made to address the issue of leachate treatment to

reduce the impact on the environment and human health caused by percolation into groundwater and the contamination of surface water [7, 9, 48, 80].

1.4 Treatment of Landfill Leachate

The treatment methods for landfill leachate can be categorized as, biological, physical–chemical, and chemical methods, biochemical processes (Fig. 1.1). Besides this, conventional methods that include recirculation of landfill leachate and transfer of leachate to sewage treatment plants are also practiced.

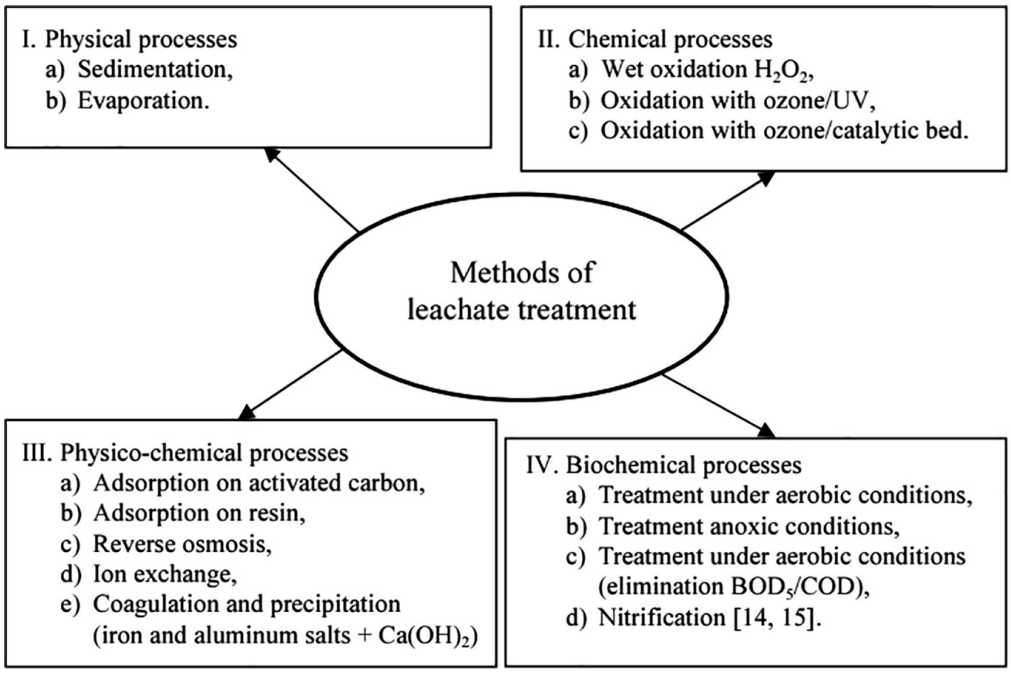


Fig. 1.1 Commonly used Leachate treatment methods for Landfill Leachate. *Source* Jelonek and Neczaj [41]

1.5 Conventional Treatment Methods for Landfill Leachate

Recirculation and transfer of leachate to sewage plants are the traditional techniques for landfill leachate treatment.

Landfill leachate recirculation: Due to ease and very less operational cost, landfill leachate recirculation has been extensively employed in previous decades. The key influencing factors in the effectiveness of recirculation are frequency of recirculation and volume. However, it may lead to accumulation of refractory substances in the leachate due to multiple cycles of recirculation. It affects the stability of the landfill system resulting in rise in difficulty of successive landfill leachate treatment [22].

Transfer of landfill leachate to sewage plants: Discharge of landfill leachate to wastewater treatment plants (WWTPs) is also a convenient and inexpensive

method for landfill leachate disposal. However, the effectiveness of downstream treatment processes is significantly affected and the overall treatment efficiency is reduced due to the addition of leachate containing high concentrations of Dissolved organic matter (DOM) [26].

Furthermore, recalcitrant organic compounds have been found resistant to biological degradation, allowing them to pass through WWTPs. In addition, effective pretreatment should be employed to remove UV-quenched substances as the non-degraded organic compounds, especially persistent UV-absorbing DOM, can significantly interfere with UV disinfection in WWTPs [99]. This will reduce the negative effect of landfill leachate on UV disinfection.

1.6 Integrated Treatment Technologies in Leachate Treatment

In order to fulfil and satisfy the strict discharge limit standards set up by the authorities, it is very crucial to develop and find a new alternative in leachate treatment. It is therefore common to integrate various treatment technologies either conventional or advanced processes. Figure 1.2 explains the diverse leachate treatment techniques that might be applied as an integrated treatment technologies in leachate treatment.

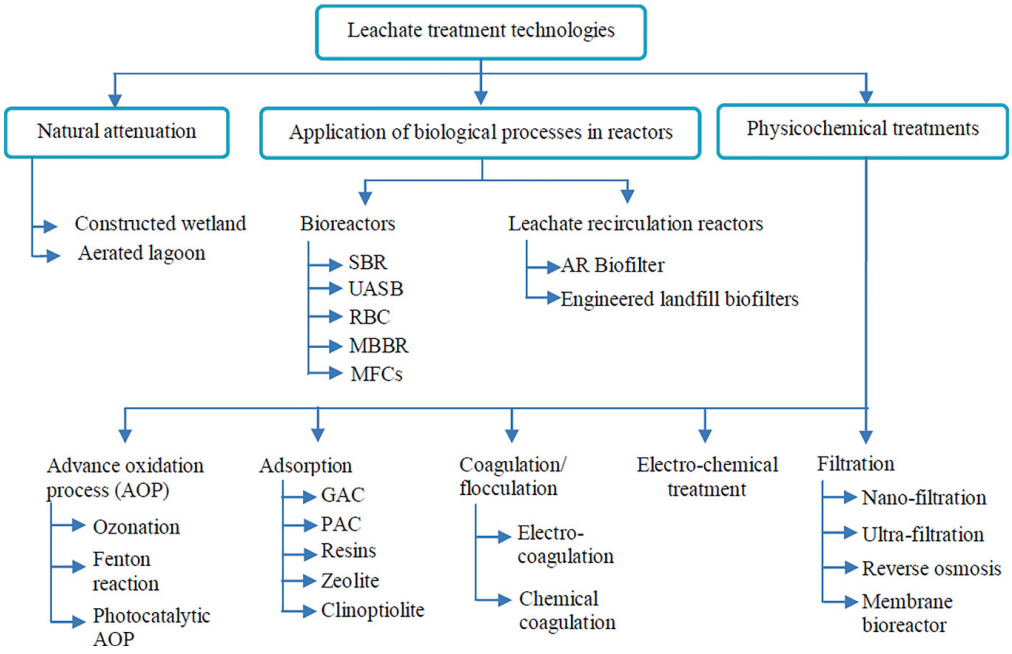


Fig. 1.2 Classification of Leachate treatment Technologies. *Source* Mukherjee et al. [66]

1.7 Natural Attenuation

Constructed wetlands (CWs): According to Verma et al. [90], the amalgamation of different physical, chemical and biological processes occurring in nature, which can efficiently reduce concentration, toxicity, and/or mobility of contaminants can be defined as natural attenuation. CWs are mainly of two

types, free surface water system and subsurface flow system, depending on the nature of wastewater flow. The treatment of wastewater in CWs involves a combination of biological and biochemical processes [94]. The wetlands provide suitable setting for speedy natural attenuation of organic contaminants. It happens due to the presence of large diversity of microorganisms, nutrients in the discharging groundwater, and a wide range of redox conditions in the surrounding groundwater or surface water interfaces [57, 87]. Microbial communities present in CWs can break down the complex organic compounds in wastewaters. The rate of attenuation of organic contaminants increases with age as the microbial population increases in a CW [20].

Phytoremediation: It is an attractive technology for landfill remediation and according to Kim and Owens [49], it can stabilize soil while simultaneously remediating landfill leachate. Plants planted in CWs influence the redox potential by supplying oxygen to the soil in the root rhizospheric zone. In this zone, enhanced nitrification by nitrifying bacteria takes place, thereby reducing the NH_4-N concentration in the landfill leachate [15]. CWs show high BOD_5, TN, and FCs removal efficiency of 91%, 96%, and more than 99%, respectively [17, 61, 76, 94]. Examples of leachate treatment in CWs and the achieved efficiency is tabulated in Table 1.1.

Table 1.1 Overview of leachate treatment techniques involving natural processes

Technology	Mechanism and process	Scope	Efficiency (%)	Country	Advantage	Disadvantage	Selected references
Constructed wetlands	Phytoremediation by cattail and in situ microorganisms	BOD_5	91				
		TN	96	Thailand	Low operation and maintenance cost	Buildup of excessive salts in soil due to poor understanding of soil plant system and improper management	[77]
		FC	> 99				
		Total P	98–99				
		Cd	99.70				
	Phytoremediation by reeds and cattail	BOD_5	50				
		COD	59	Slovenia	Low operation and maintenance cost	Slow operation in the initial phase	[17]
		NH_3-N	51				
		Total P	53				
		Fe	84				
		Chloride	35				
	Phytoremediation by cattail (*Typbalatifolia*)	COD	27.30				
		NH_4-N	62.30	Turkey	Low operation and maintenance cost	Low removal in the initial phase, Long stabilization period	[94]
		PO_4-P	52.60				
		Fe (III)	21				
	Phytoremediation by *Phragmitesaustralis* and Salix purpurea	SS	83.70	Slovenia	Leachate reuse as fertilizer for the growth of energy crops	Large amount of elements percolate back into the waste layers after irrigation	[43]
		BOD_5	65.50				
		NH_4-N	41.90				
		Total P	38.40				
		Phenols	61.70				
Aerated lagoons	Microbial oxidation, plant uptake	COD	75	United Kingdom	Low operation and maintenance cost	Long hydraulic retention time	[61]
		TN	80		Suitable for the removal of N		

Source Mukherjee et al. [66]

An aerated lagoon: It is a biological purification technique that uses free culture with an artificial oxygen supply [96]. The variance between an aerated lagoon and activated sludge is that the former does not sustain a fixed concentration of microorganisms [12, 97]. Luo et al. [58] have shown that for the on-site treatment of landfill leachate, aeration tanks are a simple and effective method that also has low-cost, where treatment is performed by biological oxidation. Researchers treated leachate having comparatively low COD (an average of 1740 mg L^{-1}) and high ammonium (an average of 1241 mg L^{-1}) levels

using four interconnected aerated lagoons. With a total nitrogen load removal of 80%, the COD removal efficiency averaged 75%. Other studies treated leachate having 5050 mg L^{-1} DOC (dissolved organic carbon) and 1670 mg L^{-1} TN using a two-stage anaerobic/optional lagoon system (i.e., 32 days for the anaerobic part and 240 days for the facultative part) and ultimately achieved removal efficiencies of 40% for COD, 64% for BOD, 77% for NH_4^+-N, 63% for NO_3-N, 77% for TN, 42% for P, 44% for SO_4^{2-}, and 44% for Mn [58]. Furthermore, the feasibility of treating phenolic and organic chemicals in an aeration tank was also explored. The tank was found to eliminate approximately 55–64% of the COD and 80–88% of the phenol from leachate [58]. The advantages of using aerated lagoons for leachate treatment include: the achievement of a correct bacteriological yield, limited operating constraints, the possibility of treating concentrated effluents, less sensitivity to hydraulic load variations, the possibility of adapting to seasonal variations in organic load, the storage of sludge for 10–15 years, and good landscape integration [96].

The potential disadvantages of the system are mainly related to the use of electromechanical equipment requiring a specialized agent for maintenance, possible noise pollution, and a relatively high operating cost (high energy consumption) [86].

1.8 Physicochemical Processes

The physical and chemical processes are the processes that help in reducing the suspended solids, colloidal particles, floating material, color, and toxic compounds by either flotation, coagulation-flocculation, adsorption, chemical oxidation or air stripping. Physical/chemical treatments for the landfill leachate are used in addition to the pre-treatment or to treat a specific pollutant (stripping for ammonia) [72]. For the treatment of landfill leachate, the common chemical methods used comprise of coagulation-flocculation, chemical precipitation, and chemical and electrochemical oxidations. Meanwhile, the major physical leachate treatment methods are air-stripping, adsorption and membrane filtration [95].

Electrocoagulation (EC): Another interesting method for the treatment of landfill leachate is by electrocoagulation. EC is an electrochemical method that treats different types of waste water by making use of electrical current without adding chemical coagulants. This method can efficiently remove small particles or colloidal pollutants content in wastewater [19]. It not only effectively removes pollutants from several wastewater such as slaughterhouse wastewater, dairy industry wastewater, vegetable oil refinery, nitrate-bearing wastewater, wastewater containing heavy metals and pesticides and phenolic compound, but it can also efficiently remove fluoride and humic acid content in leachate. The coagulants in situ are generated in EC method when aluminium or iron ions coming from aluminium or iron electrodes are electrically dissolved [11]. EC has become one of the reliable methods for landfill leachate treatment and has also received a substantial attention as it offers lots of advantages. Advantages of EC method include:

- less production of sludge, does not require chemical, easy to operate, low cost and operating in a short time [39].
- EC in leachate/wastewater treatment only needs a simple equipment, easy to work with and the process produces flocs similar to the chemical flocs. The flocs produced are expected to be larger, more stable and contain less bound water. They also stated that the TDS content after EC process is lesser than after the chemical treatment [65].
- Mohammadizaroun and Yusoff [64] found that the EC process requires less maintenance as the electrolytic processes in the EC cells are being controlled electrically and no moving parts even without the supply of electricity (such as rural area), the EC can still be operated using the sufficient energy available from the solar board attached to it.

Advanced Oxidation Process (AOP): In recent years, for the removal of organic materials and degradation of recalcitrant pollutants in wastewater, advanced oxidation processes (AOPs) have been recognised as an alternative method and the most promising procedures [1, 81]. Chemical oxidation process is the main treatment method in AOP techniques. Generally, there are two processes involved in AOP method, the first process is the formation of highly reactive free radical and the second process is the chemical reaction of radicals produced with an organic compound present in water [38]. The AOPs can be divided into two type of oxidation processes: chemical and photochemical oxidation [81]. The example of each type is listed in Table 1.2 [41].

Table 1.2 Examples of photochemical and chemical processes

Advanced oxidation processes-(AOPs)			
	Photochemical processes		**Chemical processes**
1	UV photolysis	1	Fenton reaction Fe^{2+}/H_2O_2
2	Reaction photo-fenton	2	Oxidation with ozone and hydrogen peroxide
3	Processes using UV/H_2O_2	3	Electrochemical oxidation
4	Processes using UV/O_3	4	Oxidation in supercritical conditions (supercritical water oxidation-SCWO
5	Processes using $UV/H_2O_2/O_3$	5	Wet air oxidation-(WAO)
6	Photocatalytic degradation in aqueous suspension semiconductors		
7	Processes with ultrasound		

Source Jelonek andNeczaj [41]

Adsorption: Adsorption process is considered as one of the most effective and promising approaches for removing DOM and NH_4^+-N in landfill leachate. It is recognized as one of the most efficient and extensively used fundamental approach in wastewater treatment processes [25, 50]. For the treatment of landfill leachate, the adsorbents with characteristics like a large surface area, microporous structure, surface reactivity, and thermostability have been applied. Activated carbon (AC) are the most used absorbents. Traditionally activated carbon has been used for leachate treatment due to its large porous surface

area, controllable pore structure, thermal stability, and low acid/base reactivity [52, 62]. Activated carbon has a superior ability to remove a wide variety of organic and inorganic pollutants dissolved in aqueous and gaseous environments [21, 83]. AC adsorption can improve the biodegradability of old landfill leachate, although the overall COD removal rate is low, with only 40% of organic matter removal achieved with 10 g/L AC, while the BOD_5/COD ratio increased from 0.18 to 0.56 [30]. AC adsorption preferentially removed the chromophoric DOM with hydrophobicity and microbial by-products in fluorescent DOM [26]. The addition of powdered activated carbon (PAC) improved the performance of biological treatment of leachate [46, 47]. Lim et al. [54] used EDTA modified rice husk in a sequencing batch reactor (SBR) and achieved better COD and nitrogen removal efficiency as compared to commercially available PAC. Activated carbons can be prepared from a large variety of carbon containing materials through pyrolysis. Large number of agricultural byproducts have been used to prepare inexpensive and renewable additional source of activated carbons such as sugarcane bagasse, rice straw, soybean hulls, rice hulls, peat moss, nutshells, and other lignocellulosic wastes [5, 44, 74]. Other low-cost adsorbents that have been successfully used for heavy metal (HM) removal are peat and rubber wood ash [36, 78]. These adsorbents may also be used for the treatment of leachate. A basic two stage process consisting of carbonization followed by activation is followed to produce activated carbons. In the first step the carbon content is enriched for the creation of an initial porosity and second activation stage helps in enhancing the pore structure [2, 3]. The combinations of hydrophilic and hydrophobic groups in the adsorbents make an excellent adsorption system which can remove both metallic ions and organic substances [68]. Addition of PAC to activated sludge reactors has shown to enhance the biological treatability of leachate [6]. The main drawbacks of AC adsorption include the requirement for AC regeneration and high levels of adsorbent consumption. Therefore, finding a low-cost and effective adsorbent has attracted much research attention. AC prepared from $ZnCl_2$ treatment of sewage sludge and cabbage has been applied to treat landfill leachate by adsorbing DOM, achieving a COD removal rate of 85.61% under optimal conditions [98]. Zeolite has also been applied for the treatment of landfill leachate as post-treatment with COD removal rates of 30% [69]. Recently, biochar has emerged as a good substitute for AC. Phosphoric acid activated biochar prepared from rice husk has been successfully used for the treatment of landfill leachate, resulting in about 80% removal of COD [58].

1.9 Biological Treatment

Due to its simplicity, dependability, and excellent cost-effectiveness, biological treatment is frequently employed for leachates containing significant amounts of organic compounds but retaining high BOD concentrations [27]. Biological treatments can be classified as aerobic or anaerobic, depending on the oxygen present. Organic substances are biodegraded by bacteria under aerobic conditions, producing carbon dioxide (CO_2) and sludge, whereas under anaerobic conditions, biogas (a mixture composed mainly of CO_2 and CH_4) is produced. Biodegradation is extremely successful at eliminating organic and nitrogenous materials from juvenile leachate with a high BOD/COD ratio (> 0.5) [63]. The

significant presence of refractory chemicals, primarily humic and fulvic acids, limits the effectiveness of such processes over time. Biological treatment of landfill leachate is broadly used due to its low economic costs and environmental impacts. However, a considerable number of refractory species remain in the effluent. In addition, organics, inorganic salts and metals have been found to pose inhibitory effects on activated sludge [30]. Therefore, the effectiveness of biological treatment largely depends on the type and composition of landfill leachate. Consequently, biological treatments are usually selected to treat young landfill leachate with high biodegradability.

Aerobic treatment: During aerobic treatment, the nitrification of ammonia nitrogen is carried out and organic contaminants that are only partially biodegradable are removed. Some aerobic biological processes including aerated lagoons, aerobic activated sludge, sequential batch reactors, rotating biological contactors, bacterial filters, moving bed biofilm reactors, fluidized bed biofilm reactors, membrane bioreactors (MBR), engineered wetlands, fungal treatments, and phytoremediation have all been successfully used to treat landfill leachate [88]. Aerobic biological treatments use microorganisms naturally present in the natural environment to degrade the pollutants. For example, in inactivated sludge-type treatment plants, the oxygen supply might be artificial (turbine or microbubble diffusion) or natural (wind or cascade system) in small lagoon facilities [67, 75]. Although aerobic techniques have successfully eliminated ammonia and organic pollutants, several disadvantages of these techniques have forced us to concentrate on other technologies [13, 84]. Among these drawbacks are [40, 40]:

- The need for a long aeration period
- A high energy demand and extra sludge generation
- Microbial suppression due to a high ammonia nitrogen content.

Fluidized bed biofilm reactor (FBBR): Due to its benefits—high biomass concentration, high removal performance, high shock tolerance, a stable ecosystem, good heterotrophic denitrification impact, and low energy consumption—the aerobic fluidized bed biofilm reactor (AFBBR) has gained much popularity in recent years. The amount of industrial wastewater produced has increased dramatically [14]. The primary goal of AFBBR technology, which has a density similar to wastewater, is to achieve homogeneous fluidization of the suspended media through aeration. This can enhance the amount of time that the biofilm on the suspension medium's surface touches the wastewater. The system ultimately accomplishes effective wastewater treatment [71]. FBBRs (fixed bed biofilm reactors) are a commonly used technology for BOD and ammonia removal in wastewater treatment. Compared to activated sludge, FBBRs are less sensitive to volatile flows, interrupted aeration, or grease leakages, making them a good choice for industrial applications, such as in the food and beverage, dairy, and chemical industries. FBBRs consist of two main components: plastic film holders and air diffusers. Different film-support designs are selected depending on the BOD and ammonia loading [53, 55, 70]. However, in most cases, a vertical flow channel design will provide sufficient surface area and reduce the risk of clogging. The submerged biofilm on the film's surface

requires oxygen for BOD and ammonium removal. The aeration diffusers provide oxygen. The first microorganisms settle after about 2 weeks on the film's surface. To continue the growth of the biofilm, the microorganisms digest the organic waste (BOD) and consume the oxygen supplied by the aeration diffusers. Only after all the BOD has been removed can the oxidation of ammonium NH_4^+ to nitrite NO_3^- begin. In the first step, microorganisms facilitate the oxidation of NH_4^+ to NO_2^- (nitrite), H+ acid, water, and ATP. Then, in the second step, nitrite is further oxidized to NO_3^- nitrate and more ATP [12, 29, 96].

An advantage of such a system is that the broad flow rate range of the aeration diffusers allows flexible operation [96]. This is very important when less BOD and ammonia are present. Reducing the oxygen supply will reduce energy costs but the same effluent results will be maintained. When more BOD and ammonia are present, increasing the oxygen supply will result in more significant removal of BOD and ammonia, so no further upgrading is required. If too much biofilm is produced, the filler material can become clogged, substantially increasing the oxygen supply will cause a scouring effect and clean the blocked channels. Per our instructions, the scouring process can also be automated by programming a VFD for specific intervals. Additionally, the oxygen input can be minimized for energy saving by monitoring the DO, BOD, and ammonia levels [29].

1.10 Combination of Methods

Activated sludge process has been reported to remove up to 52.5% of COD from landfill leachate with a BOD_5/COD ratio of 0.17 [79]. However, the disadvantage of activated sludge process is that it may produce large amounts of sewage sludge. Because of their ineffectiveness in the degradation of bio-refractory organics, standalone biological treatments generally fail to meet discharge standards. To overcome this limitation, biological treatments are often coupled with physicochemical processes. In a recent study, electrocoagulation-biofiltration hybrid techniques have been used to treat landfill leachate, with the electrocoagulation process achieving 37 ± 2% COD removal, mainly in the form of insoluble COD and HA, followed by 42 ± 7% COD removal in the subsequent biofiltration process [28]. The integration of the air-stripping, MBR and nanofiltration (NF) processes can provide highly efficient landfill leachate treatment, resulting in an overall reduction in COD, ammonia, color and toxicity of approximately 88, 95, 100 and 100%, respectively [8]. Recently, MBR technology has emerged as a promising method for the treatment of landfill leachate, utilizing a combination of membrane separation and biodegradation processes. MBR has many advantages in the treatment of landfill leachate, such as high effluent quality acquisition, high process stability, low environmental impacts, increased retention of mixed liquor suspended solids biomass and low sludge production. Depending on the landfill age and operational conditions, MBR has been adapted to treat landfill leachate, resulting in a wide variation in COD removal efficiencies, ranging from 23 to 90% [4]. Compared with conventional activated sludge processes, MBR has a higher loading rate and higher COD removal efficiency at a shorter hydraulic retention time.

The choice of one treatment technique over another is strongly linked to the composition of the leachate, its age, and the purification performance required by the standards in force [37, 82].

Biological processes are based on the activity of microorganisms contained in the leachate and the activity of external microorganisms, depending on the type of process used. They are generally very effective for the treatment of young leachates with a BOD/COD ratio exceeding a value of 0.5. The treatment results in reductions in NH_3, NH_4^+, iron, and biodegradable organic matter (Table 1.3). On the other hand, this treatment produces a large quantity of sludge which subsequently requires another treatment (Table 1.3). Physicochemical processes are more effective for the treatment of aged or stabilized leachates. Indeed, over time, the leachates are enriched with compounds refractory to biodegradation, which inhibits the activity of microorganisms and limits the effectiveness of biological processes. This treatment is known for its formation of sludge and the use of chemicals during treatment, depending on the type of treatment used. However, it does separate pollutants in suspension and colloidal particles [77, 82].

Table 1.3 Comparison of conventional treatment methods

Process	Features	Benefits	Disadvantages
Biological	Use of bacterial culture	Inexpensive Removes NH_3, NH_4^+, iron and biodegradable organic matter	Substantial production of sludge Ineffective in the presence of toxic and non-biodegradable pollutants
Physical (activated carbon filteration, membrane filteration)	Non-degrading	Separation of particulate or dissolved pollutants	High investment and energy costs Generation of concentrates
Physical–chemical	Fixation of pollutants by coagulation and separation of the flocs formed	Separation of suspended pollutants and colloidal particles	Use of chemicals Sludge formation
Chemical	Use of an oxidant Cl_2, ClO_2, O_3	Chemical oxidation of pollutants-little or no waste Increased biodegradability	Partial oxidation (formation of intermediates) Oxidant management

Source Bouaouda et al. [16]

1.11 Conclusion

As the population is growing, the quantity of solid waste generated is also increasing. It is further leading to continuous upsurge in the quantity of leachate generated from landfills. As the leachate is highly contaminated and polluted, it is pertinent that it must be treated before being discharged into the natural environment. Conventional as well as natural treatment methods are carried out for leachate treatment. Along with these, various techniques can be used for this purpose, such as biological, physical–chemical, and membrane methods. Most physical–chemical treatments are used as pre-treatments or post-treatments to

complement conventional treatment processes or remove specific contaminants. These treatments effectively remove organic compounds that are difficult to decontaminate biologically. The type of method or technique selected to treat the leachate depends upon several factors including type of landfill, age of landfill, seasonal variations, degree of compaction, climatic conditions, precipitation, mode of operation, type of waste, moisture content, etc. As young leachate is biodegradable; in contrast, biological processes lose their effectiveness with leachate aging as the aged leachates are characterized by high concentrations of ammonia nitrogen that inhibit their activity. For leachate treatment, the contemporary trend encompasses an amalgamation of biological, chemical, and physicochemical processes in multiple-stage treatment systems. Preferred methods depend on various technical considerations, which should be adequately evaluated because one method cannot generally be used for common applications. A number of research works have been carried out in order to come out with a treatment system that can improve performance and is cost effective. For developing sustainable treatment technologies for the treatment of landfill leachate, the main criteria are treatment efficiency, meet the treatment standards, reduced cost and minimising negative environmental consequences.

References

1. Abbas AA, Jingsong G, Ping LZ, Ya PY, Al-Rekabi WS (2009) Review on landfill leachate treatments. Am J Appl Sci 6(4):672–684
2. Acharya J, Sahu JN, Mohanty CR, Meikap BC (2009) Removal of lead (II) from wastewater by activated carbon developed from Tamarind wood by zinc chloride activation. Chem Eng J 149:249–262
3. Acharya J, Sahu JN, Sahoo BK, Mohanty CR, Meikap BC (2009) Removal of chromium (VI) from wastewater by activated carbon developed from Tamarind wood activated with zinc chloride. Chem Eng J 150:25–39
4. Ahmed FN, Lan CQ (2012) Treatment of landfill leachate using membrane bioreactors: a review. Desalination 287:41–54
5. Ahmedna M, Marshall WE, Rao RM (2000) Production of granular activated carbons from select agricultural by-products and evaluation of their physical, chemical and adsorption properties. Biores Technol 71:113–123
6. Aktas O, Cecen F (2001) Addition of activated carbon to batch activated sludge reactors in the treatment of landfill leachate and domestic wastewater. J Chem Technol Biotech 76:793–802
7. Alghamdi AG, Aly AA, Ibrahim HM (2021) Assessing the environmental impacts of municipal solid waste landfill leachate on groundwater and soil contamination in western Saudi Arabia. Arab J Geosci 14:1–12
8. Amaral MCS, Moravia WG, Lange LC, Zico MR, Magalhaes NC, Ricci BC, Reis BG (2016) Pilot aerobic membrane bioreactor and nanofiltration for municipal landfill leachate treatment. J Environ Sci Health Part A 51:640–649
9. Arabi M, Sbaa M, Vanclooster M, Darmous A (2020) Impact of the municipal solid waste typology on leachate flow under semi-arid climate–a case study. J Ecol Eng 21(6)
10. Baderna D, Caloni F, Benfenati E (2019) Investigating landfill leachate toxicity in vitro: a review of cell models and endpoints. Environ Int 122:21–30
11. Bazrafshan E, Moein H, Mostafapour FK, Nakhaie S (2013) Application of electrocoagulation process for dairy wastewater treatment. J Chem, 8pp. Article ID 640139 [online] https://doi.org/10.1155/2013/640139

12. Benyerou D (2022) Epuration et reutilisation des eaux residuaires. University of Sciences and Technology of Oran, Oran
13. Bernat K, Zaborowska M, Zielinska M et al (2021) Biological treatment of leachate from stabilization of biodegradable municipal solid waste in a sequencing batch biofilm reactor. Int J Environ Sci Technol 18:1047–1060. https://doi.org/10.1007/s13762-020-02915-6
14. Bhuyan SC, Kumar Swain A, Sahoo A, Bhuyan SK (2020) Nutrient (sulphate) removal from wastewater in inverse fluidized bed biofilm reactor. Mater Today Proceed 33:5476–5480. https://doi.org/10.1016/j.matpr.2020.03.306
15. Białowiec A, Davies L, Albuquerque A, Randerson PF (2012) Nitrogen removal from landfill leachate in constructed wetlands with reed and willow: redox potential in the root zone. J Environ Manage 97:22–27
16. Bouaouda S, Souabi S, Bouyakhsass R, Taleb A, Madinzi A, Anouzla A (2023) Techniques for treating leachate discharges: a critical review. Euro-Mediterranean J Environ Integrat 1–27
17. Bulc TG (2006) Long term performance of a constructed wetland for landfill leachate treatment. Ecol Eng 26(4):365–374
18. BRTC (2008) Solid waste management and construction of landfill at Matuail. Final report for Dhaka City Corporation prepared by Department of Civil Engineering, BUET, Dhaka, Bangladesh
19. Butler E, Hung Y-T, Yeh RY-L, Ahmad MSA (2011) Electrocoagulation in wastewater treatment. Water Environ J 3:495–525
20. Calli B, Mertoglu N, Roest K, Inanc B (2006) Comparison of long-term performances and final microbial compositions of anaerobic reactors treating landfill leachate. Biores Technol 97:641–647
21. Chingombe P, Saha B, Wakeman RJ (2005) Surface modification and characterisation of a coal-based activated carbon. Carbon 43:3132–3143
22. Chugh S, Clarke W, Pullammanappallil P, Rudolph V (2016) Effect of recirculated leachate volume on MSW degradation. Waste Manag. Res. 16:564–573
23. Costa AM, Alfaia RGDS, Campos JC (2019) Landfill leachate treatment in Brazil—an overview. J Environ Manage 232:110–116
24. Dabaghian Z, Peyravi M, Jahanshahi M, Rad AS (2018) Potential of advanced nano-structured membranes for landfill leachate treatment: a review. Chembioeng Rev. 5:119–138
25. Daifullah AAM, Girgis BS, Gad HMH (2004) A study of the factors affecting the removal of humic acid by activated carbon prepared from biomass material. Colloids Surf, A 235:1–10
26. Deng Y, Jung C, Zhao R, Torrens K, Wu L (2018) Adsorption of UV-quenching substances (UVQS) from landfill leachate with activated carbon. Chem Eng J 350:739–746
27. Detho A, Daud Z, Rosli MA et al (2022) Review on COD and ammoniacal nitrogen removal from landfill leachate using low-cost adsorbent. J Air Waste Manag Assoc 72:10–23. https://doi.org/10.1080/10962247.2021.1895366
28. Dia O, Drogui P, Buelna G, Dube R (2018) Hybrid process, electrocoagulation-biofiltration for landfill leachate treatment. Waste Manage 75:391–399
29. Doucet J (2022) Traitement simultane des nitrates et du methane des sites d'enfouissement a l'aide de bacteries methanotrophes par biofiltration. Laval University, Quebec, Thesis
30. Gotvajn AZ, Tisler T, Zagorc-Koncan J (2009) Comparison of different treatment strategies for industrial landfill leachate. J Hazard Mater 162:1446–1456
31. Gupta A, Paulraj R (2017) Leachate composition and toxicity assessment: an integrated approach correlating physicochemical parameters and toxicity of leachates from MSW landfill in Delhi. Environ Technol 38(13–14):1599–1605
32. Gupta A, Rajamani P (2015) Toxicity assessment of municipal solid waste landfill leachate collected in different seasons from Okhala landfill site of Delhi. J Biomed Sci Eng 8(06):357

33. Gupta A, Rajamani P (2016) Evaluation of the leachate composition and contamination potential of municipal solid waste landfill sites in Delhi. Int J Environ Waste Manage 18(4):285–302
34. Gupta A, Gaharwar US, Verma A, Rajamani P (2022) Municipal solid waste landfill leachate induced cytotoxicity in root tips of Vicia faba: Environmental risk posed by non-engineered landfill. IJBB. 59(11):1113–1125
35. Gupta P, Bhargava A, Kumari R, Lodhi L, Tiwari R, Gupta PK, Bunkar N, Samarth R, Mishra PK (2019) Impairment of mitochondrial-nuclear cross talk in lymphocytes exposed to landfill leachate. Environ Health Insights 13:1178630219839013
36. Hasan S, Hashim MA, Gupta BS (2000) Adsorption of Ni(SO_4) on Malaysian rubber-wood ash. Biores Technol 72:153–158
37. Hasnine MT, Anand N, Zoungrana A et al (2022) An overview of physicochemical and biological treatment of landfill leachate. In: Pathak P, Palani SG (eds) Circular economy in municipal solid waste landfilling: biomining & leachate treatment: sustainable solid waste management: waste to wealth. Springer International, Cham., pp 115–152
38. Huan KS, Fahmi MR, Abidin CZA, Soon-An O (2016) Advanced oxidation processes: process mechanisms, affecting parameters and landfill leachate treatment'. Water Environ Res 88(11):2047–2058
39. Huda N, Raman AA, Ramesh S (2017) Optimization of electrocoagulation process for the treatment of landfill leachate. IOP Conference. Materials Science and Engineering, Series
40. Hussain A, Kumari R, Sachan SG, Sachan A (2021) Chapter 8—Biological wastewater treatment technology: advancement and drawbacks. In: Shah M, Rodriguez-Couto S (eds) Microbial ecology of wastewater treatment plants. Elsevier, Amsterdam, pp 175–192
41. Jelonek P, Neczaj E (2012) The use of advanced oxidation processes (AOP) for the treatment of landfill leachate. In: 4th international conference on advances in sustainable sewage sludge management, Book of Extended Abstract, Szczyrk, Poland, 3–5 December
42. Jhamnani B, Singh SK (2009) Groundwater contamination due to Bhalaswa landfill site in New Delhi'. Int J Environ Sci Eng 1(3):121–125
43. Justin MZ, Zupancic M (2009) Combined purification and reuse of landfill leachate by constructed wetland and irrigation of grass and willows. Desalination 246:157–168
44. Kadirvelu K, Kavipriya M, Karthika C, Radhika M, Vennilamani N, Pattabhi S (2003) Utilization of various agricultural wastes for activated carbon preparation and application for the removal of dyes and metal ions from aqueous solutions. Biores Technol 87:129–132
45. Kamaruddin MA, Yusoff MS, Aziz HA, Hung YT (2015) Sustainable treatment of landfill leachate. Appl Water Sci 5(2):113–126
46. Kargi F, Pamukoglu MY (2003) Powdered activated carbon added biological treatment of pre-treated landfill leachate in a fed-batch reactor. Biotech Lett 25:695–699
47. Kargi F, Pamukoglu MY (2003) Simultaneous adsorption and biological treatment of pre-treated landfill leachate by fed-batch operation. Process Biochem 38:1413–1420
48. Kastali M, Mouhir L, Assou M, Anouzla A, Abrouki Y (2020) Diagnosis of leachate from a closed landfill, impact on the soil, and treatment by coagulation flocculation with alginate and ferric chloride. Desalin Water Treat 206:307–314
49. Kim KR, Owens G (2010) Potential for enhanced phytoremediation of landfills using biosolids–a review. J Environ Manage 91(4):791–797
50. Kurniawan TA, Lo W-H, Chan GYS (2006) Physico-chemical treatments for removal of recalcitrant contaminants from landfill leachate. J Hazard Mater 129:80–100
51. Leme DM, Marin-Morales MA (2009) *Allium cepa* test in environmental monitoring: a review on its application. Mutation Res/Rev Mutation Res 682(1):71–81

52. Li W, Zhang L-B, Peng J-H, Li N, Zhu X-Y (2008) Preparation of high surface area activated carbons from tobacco stems with K_2CO_3 activation using microwave radiation. Ind Crops Prod 27:341–347
53. Li Y, Xiang T, Liang H et al (2022) Achieving rapid mainstream deammonification through inoculating long-term refrigerated side stream sludge in plug-flow fixed-bed biofilm reactor. Sci Total Environ 806:151415. https://doi.org/10.1016/j.scitotenv.2021.151415
54. Lim P-E, Lim S-P, Seng C-E, Noor AM (2010) Treatment of landfill leachate in sequencing batch reactor supplemented with activated rice husk as adsorbent. Chem Eng J 159:123–128
55. Liu Z (2022) Resine echangeuse d'ions en mode biologique pour l'enlevement de matieres organiques naturelles des eaux de surface. Universite de Montreal, Montreal, Thesis
56. Longe EO, Balogun MR (2010) Groundwater quality assessment near a municipal landfill, Lagos, Nigeria'. Res J Appl Sci Eng Technol 2(1):39–44
57. Lorah MM, Cozzarelli IM, Bohlke JK (2009) Biogeochemistry at a wet- ¨ land sediment–alluvial aquifer interface in a landfill leachate plume. J Contam Hydrol 105:99–117
58. Luo H, Zeng Y, Cheng Y et al (2019) Recent advances in municipal landfill leachate: A review focusing on its characteristics, treatment, and toxicity assessment. J Pre-Proof/Sci Total Environ. https://doi.org/10.1016/j.scitotenv.2019.135468
59. Mangimbulude JC, Boris M, Breukelen V, Krave AS, Nico M, Wilfred FM (2009) Seasonal dynamics in leachate hydrochemistry and natural attenuation in surface run-off water from a tropical landfill. Waste Manage 29:829–838. https://doi.org/10.1016/j.wasman.2008.06.020
60. Maranon E, Castrillon L, Fernandez-Nava Y, Fernandez-Mendez A, Fernandez-Sanchez A (2010) Colour, turbidity and COD removal from old landfill leachate by coagulation-flocculation treatment. Waste Manage Res 28(8):731–737
61. Mehmood MK, Adetutu E, Nedwell DB, Ball AS (2009) In situ microbial treatment of landfill leachate using aerated lagoons. Biores Technol 100:2741–2744
62. Mendez-Dıaz JD, Abdel daiem MM, Rivera-Utrilla J, Sanchez-Polo M, Bautista-Toledo I (2012) Adsorption/bioadsorption of phthalic acid, an organic micropollutant present in landfill leachates, on activated carbons. J Colloid Interface Sci 369:358–365
63. Miao L, Yang G, Tao T, Peng Y (2019) Recent advances in nitrogen removal from landfill leachate using biological treatments—a review. J Environ Manage 235:178–185
64. Mohammadizaroun M, Yusoff MS (2014) Treatment of leachate by electrocoagulation technique using iron and hybrid electrodes. Int J Sci Res in Knowl 2(11):497–508
65. Mollah MY, Morkovsky P, Gomes J, Kesmez M, Parga J, Cocke DL (2004) Fundamentals, present and future perspectives of electrocoagulation. J Hazard Mater 114(1):199–210
66. Mukherjee S, Mukhopadhyay S, Hashim MA, Sen Gupta B (2015) Contemporary environmental issues of landfill leachate: assessment and remedies. Crit Rev Environ Sci Technol 45(5):472–590
67. Musa MA, Idrus S (2021) Physical and biological treatment technologies of slaughterhouse wastewater: a review. Sustainability 13:4656. https://doi.org/10.3390/su13094656
68. Okolo B, Park C, Keane MA (2000) Interaction of phenol and chlorophenols with activated carbon and synthetic zeolites in aqueous media. J Colloid Interface Sci 226:308–317
69. Poblete R, Oller I, Maldonado MI, Cortes E (2019) Improved landfill leachate quality using ozone, UV solar radiation, hydrogen peroxide, persulfate and adsorption processes. J Environ Manage 232:45–51
70. Reddy CV, Rao DS, Kalamdhad AS (2022) Combined treatment of high-strength fresh leachate from municipal solid waste landfill using coagulation-flocculation and fixed bed upflow anaerobic filter. J Water Process Eng 46:102554. https://doi.org/10.1016/j.jwpe.2021.102554
71. Ren J, Cheng W, Jiao M et al (2020) Characteristics of oxygen mass transfer and its impact on pollutant removal performance and microbial community structure in an aerobic fluidized bed biofilm reactor for treatment of municipal wastewater. Bioresource Technol 323:124552

72. Renou S, Givaudan JG, Poulain S, Dirassouyan F, Moulin P (2008) Landfill leachate treatment: review and opportunity. J Hazard Mater 150(3):468–493
73. Safari E, Jalili Ghazizade M, Shokouh A, Nabi Bidhendi GR (2011) Anaerobic removal of cod from high strength fresh and partially stabilized leachates and application of multi stage kinetic model. Int J Environ Res 5(1):255–270
74. Sahu JN, Acharya J, Meikap BC (2010) Optimization of production conditions for activated carbons from Tamarind wood by zinc chloride using response surface methodology. Biores Technol 101:1974–1982
75. Sanghamitra P, Mazumder D, Mukherjee S (2021) Treatment of wastewater containing oil and grease by biological method- a review. J Environ Sci Health Part A, 56:394–412. https://doi.org/10.1080/10934529.2021.1884468
76. Sawaittayothin V, Polprasert C (2007) Nitrogen mass balance and microbial analysis of constructed wetlands treating municipal landfill leachate. Biores Technol 98(3):565–570
77. Saxena V, Kumar Padhi S, Kumar Dikshit P, Pattanaik L (2022) Recent developments in landfill leachate treatment: Aerobic granular reactor and its future prospects. Environ Nanotechnol Monit Manage 18:100689. https://doi.org/10.1016/j.enmm.2022.100689
78. Sen Gupta B, Curran M, Hasan S, Ghosh TK (2009) Adsorption characteristics of Cu and Ni on Irish peat moss. J Environ Manage 90:954–960
79. Setiadi T, Fairus S (2003) Hazardous waste landfill leachate treatment using an activated sludge-membrane system. Water Sci Technol 48:111–117
80. Sharma A, Ganguly R, Kumar Gupta A (2020) Impact assessment of leachate pollution potential on groundwater: an indexing method. J Environ Eng 146(3):05019007
81. Sharu E, Sudha Kani M, Sajitha R, Latha CJ (2016) Leachate treatment by advanced oxidation process. Int J Adv Mech Civ Eng 3(3):117–120
82. Siddiqi SA, Al-Mamun A, Baawain MS, Sana A (2022) A critical review of the recently developed laboratory-scale municipal solid waste landfill leachate treatment technologies. Sustain Energy Technol Assess 52:102011. https://doi.org/10.1016/j.seta.2022.10201
83. Singh SK, Townsend TG, Mazyck D, Boyer TH (2012) Equilibrium and intra-particle diffusion of stabilized landfill leachate onto micro- and mesoporous activated carbon. Water Res 46:491–499
84. Sitterley KA, Silverstein J, Rosenblum J, Linden KG (2021) Aerobic biological degradation of organic matter and fracturing fluid additives in high salinity hydraulic fracturing wastewaters. Sci Total Environ 758:143622. https://doi.org/10.1016/j.scitotenv.2020.143622
85. Tatsi AA, Zouboulis AI (2002) A field investigation of the quantity and quality of leachate from a municipal solid waste landfill in a mediterranean climate (Thessaloniki, Greece). Adv Environ Res 6:207–219. https://doi.org/10.1016/S1093-0191(01)00052-1
86. Thompson M, Moussavi S, Li S et al (2022) Environmental life cycle assessment of small water resource recovery facilities: comparison of mechanical and lagoon systems. Water Res 215:118234. https://doi.org/10.1016/j.watres.2022.118234
87. Tobias CR, Macko SA, Anderson IC, Canuel EA, Harvey JW (2001) Tracking the fate of a high concentration groundwater nitrate plume through a fringing marsh: a combined groundwater tracer and in situ isotope enrichment study. Limnol Oceanogr 46:1977–1989
88. Torretta V, Ferronato N, Katsoyiannis IA et al (2017) Novel and conventional technologies for landfill leachates treatment: a review. Sustainability. https://doi.org/10.3390/su9010009
89. Trankler J, Visvanathan C, Kuruparan P, Tubtimthai O (2005) Influence of tropical seasonal variations on landfill leachate characteristics results from lysimeter studies. Waste Manage 25:1013–1020. https://doi.org/10.1016/j.wasman.2005.05.004
90. Verma A, Gaharwar US, Priyadarshini E, Rajamani P (2022) Metal accumulation and health risk assessment in wastewater used for irrigation around the Agra Canal in Faridabad, India. Environ Sci Pollut Res 1–15

91. Verma A, Gupta A, Rajamani P (2023) Application of wastewater in agriculture: benefits and detriments. In: River conservation and water resource management. Springer Nature Singapore, Singapore, pp 53–75
92. Visvanathan J, Trankler P, Kuruparan Q, Qin XN (2003) Influence of landfill operation and waste composition on leachate control: lysimeter experiments under tropical conditions. In: Proceedings of the 2nd Asia pacific landfill symposium, Seoul, pp 441–447
93. Wdowczyk A, Szymańska-Pulikowska A (2021) Analysis of the possibility of conducting a comprehensive assessment of landfill leachate contamination using physicochemical indicators and toxicity test. Ecotoxicol Environ Saf 221:112434
94. Yalili M, Kestioglu K, Yonar T (2007) Landfill leachate treatment by the combination of physicochemical methods with adsorption process. J Biol Environ Sci 1(1)
95. Yasmine G (2022) Wastewater treatment. University of Sciences and Technology of Oran
96. Yekani NH, Peyravi M, Jahanshahi M, Bagheri M (2022) Design and annual operation of full scale landfill leachate plant in Iran: focus on multistage membrane technology. Chem Pap. https://doi.org/10.1007/s11696-022-02108-0
97. Zeng F, Liao X, Pan D, Shi H (2019) Adsorption of dissolved organic matter from landfill leachate using activated carbon prepared from sewage sludge and cabbage by ZnCl2. Environ Sci Pollut Res 27:4891–4904
98. Zhao R, Novak JT, Goldsmith CD (2012) Evaluation of on-site biological treatment for landfill leachates and its impact: a size distribution study. Water Res 46:3837–3848
99. Zhu C, Huang H, Chen Y (2022) Recent advances in biological removal of nitroaromatics from wastewater. Environ Pollut 307:119570. https://doi.org/10.1016/j.envpol.2022.

© The Author(s), under exclusive license to Springer Nature Switzerland AG 2024
S. Souabi, A. Anouzla (eds.), *Landfill Leachate Treatment Techniques*, Springer Water
https://doi.org/10.1007/978-3-031-63157-3_2

2. An Overview of Existing Municipal Landfill Leachate Treatment Techniques: Opportunities and Challenges

Mohammad Toha[1], Sadia Sikder[1] and Md. Mostafizur Rahman[1, 2] ✉
(1) Department of Environmental Science, Bangladesh University of Professionals, Mirpur, Cantonment, Dhaka, 1216, Bangladesh
(2) Laboratory of Environmental Health and Ecotoxicology, Department of Environmental Sciences, Jahangirnagar University, Savar, Dhaka, 1342, Bangladesh

✉ **Md. Mostafizur Rahman**
Email: rahmanmm@juniv.edu

Abstract
The chapter discusses landfill leachate, a heavily contaminated liquid waste originating from landfills, exploring its composition, environmental impact, and treatment methods. This complex mixture contains organic compounds, inorganic salts, toxic gases, halogenated hydrocarbons, and heavy metals, necessitating treatment before release into natural water bodies. The challenge lies in selecting appropriate treatment methods. For well-aged leachate, physicochemical methods like coagulation-flocculation, adsorption, membrane filtration, ozone treatment, air stripping, and advanced oxidation work best, while biological methods like membrane bioreactors, activated sludge processes, anaerobic systems, and biological contactors work best for young leachate. This book chapter explored various treatments, including bioreactors, bioremediation, phytoremediation, advanced oxidation, adsorption, coagulation-flocculation, and membrane filtration. According to existing studies, membrane bioreactors and integrated biological techniques exhibit exceptional ammonia and nitrogen removal rates above 90%. Coagulation-flocculation effectively eliminated suspended solids and turbidity. It has been found that combining methods displayed promise in removing metals at 40–100% rates. Due to its high chemical oxygen

demand, ammonia concentrations, and low biodegradability, combined treatment techniques were successful for landfill leachate. Nonetheless, further research is needed to optimize methods and enhance removal efficiency. To summarize, effective treatment of hazardous landfill leachate is imperative before disposal, and the combination of biological and physicochemical approaches emerges as a promising strategy.

Keywords Landfill – Leachate – Biological treatment – Physical treatment – Chemical treatment

2.1 Introduction

Urban solid waste landfills are widely used for managing household, industrial, and commercial nonhazardous solid wastes and nonhazardous sludge. Despite being a typical waste management method with lower operational costs than alternatives like incineration, sanitary landfilling poses potential environmental hazards [23, 43]. Leachate, an extremely polluted wastewater produced after landfilling, contains high levels of chemical and biological oxygen demand (COD, BOD), organic contaminants, heavy metals, ammonia, and other harmful substances [46]. This leachate can contaminate nearby groundwater, surface water, and soil. Moreover, the composition of this leachate depends on landfill conditions and operating procedures [60, 65]. It is classified as hazardous or nonhazardous waste and requires suitable treatment. Leachates, which can contaminate soil and groundwater, and biogas, a source of air pollution, are the byproducts of landfilling [43, 9]. Modern landfills have reduced environmental impacts, but contaminated leachate remains a concern. Nevertheless, it can be inferred that the creation of leachate is an unavoidable outcome of landfilling. The associated risk posed by municipal solid waste is escalating, surpassing the confines of both local and regional areas. The matter has emerged as a global concern, exerting a growing influence on public health and various dimensions of society, the economy, and the environment.

Despite these severe ecotoxicological risks, municipal solid waste generation is being generated greatly per capita and overall. For example, waste production in Rio de Janeiro, Brazil, increased from 6200 tons per day to 8042 tons per day, with waste production in Norway and the USA increasing by 3 and 4.5% per year, respectively [53, 63, 64]. In more developed countries, annual waste production ranged from 300 to 800 kg per person, while in other countries, it was less than 200 kg per person [14, 45, 52]. By the year 2050, it is expected that it will have expanded by a factor of two in the world’s largest and most populous cities and by a factor of three in the countries that are the poorest in the world [15, 17, 20]. This is especially concerning when one considers that

countries with low incomes sometimes have restricted access to sanitary landfills, recycling facilities, and incinerator facilities. As a direct result of this problem, the vast majority of their trash is not adequately handled, does not undergo any treatment, and is disposed of in open dumps. Thus, this present view indicates that this large number of untreated wastes will produce massive landfill leachate, posing a potential threat to dispersed landfill surroundings. In these circumstances, protecting our ecosystem from the detrimental impacts of landfill leachate and other adsorbed pollutants is essential.

Various contaminants are present in high concentrations in landfill leachate. Thus, effective treatment systems are required before releasing leachate into receiving waters to prevent pollution and meet discharge criteria [9, 33]. Stringent regulations have been introduced to address the potential dangers of landfill leachates. Despite waste recovery and recycling efforts, controlled dumping stations and on-site pre-treatment of landfill effluent systems should be implemented.

However, several treatment methods, including biological and physicochemical processes, have already been utilized worldwide [3, 31, 38, 58]. While several treatment processes exist, a universally effective solution for leachate treatment has yet to be found. So, combining these methods is often necessary to achieve high removal efficiency and minimize environmental impacts. In addition, novel technologies address simultaneous carbon and nitrogen removal from landfill leachate.

Thus, this book chapter aims to review the landfill leachate treatment and provide a comparative evaluation of different treatment processes. Lastly, the overview of this book chapter will help to address the problems and encourage the search for new knowledge in landfill leachate treatment technologies.

2.2 Landfill Leachate: Features and Characteristics

Leachate is generated when moisture infiltrates waste materials within a landfill, transferring specific types of pollutants. The pollutants in leachate from municipal landfills can be classified into four primary categories: organic contaminants and substrates, inorganic compounds, heavy metals, and constituents like total dissolved solids (TDS) and color [46].

Depending on the duration of its presence, landfill leachate can be categorized into three main groups (Table 2.1): Young, Intermediate, and Old. For instance, in “Young” landfills (referred to as the acid phase), leachate tends to exhibit lower pH levels, elevated concentrations of volatile acids, and the presence of readily decomposable organic matter.

The defining attributes of a liquid effluent comprise its volumetric flow rate and composition [20, 51]. This correlation is especially evident in the context of leachate. A flowchart of the organic fraction balance of COD within a landfill is depicted in Fig. 2.1.

Table 2.1 Landfill leachate representation (on their deposition time)

Age (year)	Young	Intermediate	Old
	0–5 years	**5–10 years**	**> 10 years**
pH	< 6.5	6.5–7.5	> 7.5
COD (mg/L)	> 10,000	5000–10,000	< 5000
BOD5/COD	0.5−1.0	1.0–0.5	> 0.1
NH3-N (mg/L)	< 400	–	> 400
H.M	Medium to low	Low	Low
VFA/HFA	VFA (80%)	VFA (5–30%) + HFA	HFA (80%)
Biodegradability	High	Medium	Low

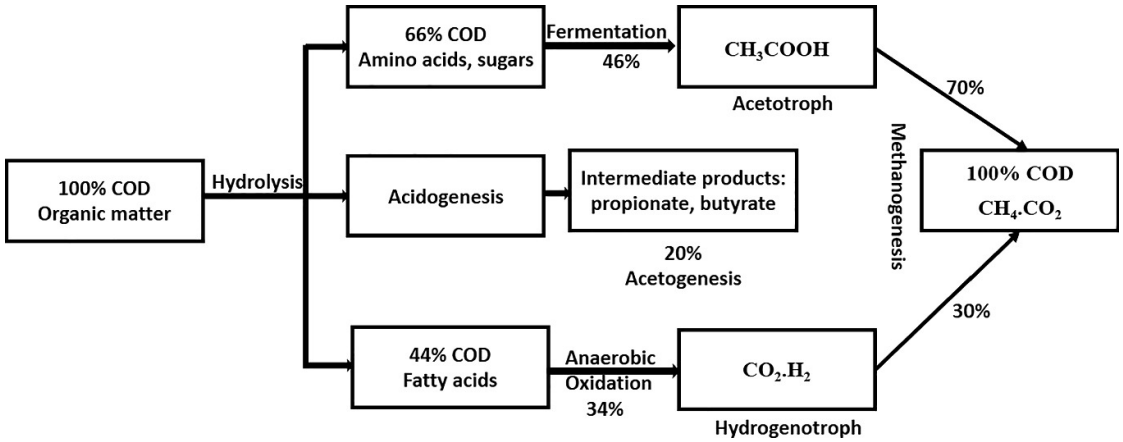


Fig. 2.1 The carbon–oxygen demand (COD) equilibrium of the organic component within a sanitary landfill

The methodologies employed in landfilling, including the use of impermeable covers and liners made of materials like clay, geotextiles, or plastics, play a crucial role in managing the influx of water into the landfill. This management, in turn, helps mitigate the potential for pollution. During the methanogenic phase in well-developed landfills, methane is made from leachate, and the pH and the composition of organic materials, mostly made up of humic and fulvic fractions, are high [9]. Nevertheless, variations can be observed in certain other research studies due to disparities in waste composition across different countries [53, 14, 31, 11]. The fundamental characteristics of landfill leachate are commonly encapsulated by critical parameters such as COD, BOD, the BOD/COD ratio, pH, suspended solids (SS, ammonium nitrogen (NH3-N, total Kjeldahl nitrogen (TKN, and heavy metal content [46, 64].

Literature reveals diverse leachate compositions from various sanitary landfills, indicating substantial discrepancies [45]. Referencing Table 2.2, which presents a global overview of landfill leachate attributes, it becomes evident that China exhibits the highest concentration of COD (28,000 mg/L), while Riyadh (Saudi Arabia) demonstrates elevated levels of Fe (167.6 mg/L) in concentrated landfill leachate [6, 61].

Table 2.2 Global characteristics of landfill leachate

Location	COD	BOD_5	BOD_5/COD	Ammonia (mg/L)	Heavy metals					References
					Fe	Mn	Zn	Cd	Ni	
Matuail landfill, Bangladesh	1343	96	0.07	NR	3.41	0.17	2.3	NR	0.17	[46]
Istanbul Kömürcüoda Landfill, Turkey	24,040	15,021	0.59	2281	10.37	NR	0.96	NR	0.95	[32]
MSW incineration plants, China	28,000	950	0.04	3.50	30.00	4.03	17.8	NR	3.70	[51]
Beijing, China	2305	105	0.04	1240	NR	NR	NR	NR	NR	[61]
Sao Carlos, Brasil	4,425–4,860	433–588	0.09–0.12	NR	NR	NR	NR	NR	NR	[58]
Riyadh City, Saudi Arabia	173	NR	NR	NR	167.61	10.83	0.18	NR	0.50	[6]

2.2.1 Color and TDS

Parvin and Tareq [46] present a common contamination found within leachate from landfills. The breakdown of specific organic compounds, including humic acid (HA), can potentially induce a shift in water coloration from yellow to a deep brown—the pivotal role of substances and particles in generating color and turbidity. The concept of total dissolved solids (TDS) incorporates the aggregate effect of particular cations and anions on the quality of all types of water. Furthermore, it can originate from even minor concentrations of dissolved organic matter, as noted by Tian et al. in 2022. The elevated TDS levels could potentially impede or diminish the natural breakdown of dissolved organic carbon, as emphasized by Renou et al. in 2019. This particular study articulated that heightened electrical conductivity and elevated TDS values might indicate the presence of dissolved organic and inorganic components within analyzed samples [3, 12, 13].

2.2.2 Organic and Inorganic Pollutants, and Heavy Metals

The composition of landfill leachate can vary depending on waste characteristics, landfill stage, and climate [6, 10]. Approximately 80% of the dissolved organic matter in leachate consists of refractory humic substances and volatile fatty acids. These refractory organics are challenging to degrade using conventional biological treatments [35, 54]. Persistent organic pollutants and various chlorinated compounds, have also been frequently observed in leachate [64]. Municipal landfills necessitate meticulous management due to the ever-growing contaminants, encompassing medicines, personal care products, plasticizers, and nanomaterials. The constituents of landfill leachate can also encompass a range of inorganic macro components which constitute both anionic and cationic elements [63], Ranjana et al. [50]. Moreover, this inorganic ions and cations constitute a significant portion of this mixture. Among the detrimental pollutants within landfill leachate, heavy metals encompass chromium, manganese, cadmium, lead, iron, nickel, and zinc [53, 5]. These heavy metals are recognized for their high toxicity levels.

Notably, in regions with developing economies, where segregating hazardous and nonhazardous waste before disposal is not widely adopted, landfill leachates frequently exhibit elevated concentrations of these hazardous heavy metals. The above-mentioned situation highlights the necessity of implementing efficient waste management tactics to minimize the dissemination of detrimental compounds into the surrounding ecosystem. It underscores the urgency of addressing waste segregation practices in these countries. Addressing heavy metal elimination from leachate presents notable difficulties. Metal levels are typically more elevated in young (acetogenic) than aged leachate. Thus, effective strategies are needed to successfully eradicate heavy metals during leachate treatment procedures. Given landfill leachate's diverse and potentially hazardous nature, effective treatment strategies are necessary to minimize its environmental impact. Proper management and removal of organic and inorganic contaminants, including heavy metals, are essential to ensuring the safe discharge of leachate into natural waters or public sewers. Continuous research and technological advancements are required to address the challenges posed by landfill leachate characteristics and develop sustainable and efficient treatment methods.

2.2.3 Characteristics of Landfill Leachate

1. **Organic matter**: Leachate contains important organic elements called humic-type components. Moreover, biodegradable and refractory organic materials may be found in leachate [46].

2. **Ammonia–nitrogen**: Landfill leachate often contains elevated levels of ammonia–nitrogen, which can harm the environment and aquatic life if discharged untreated [54].

3. **Heavy metals**: Leachate may contain heavy metals that are highly soluble in water and pose hazards to human health and the environment [68].

4. **Chlorinated organic and inorganic salts**: Leachate may contain chlorinated organic compounds and various inorganic salts [27, 37].

Landfill leachate treatment is necessary to remove these pollutants before discharging them into natural waters or public sewers. Regulations often require proper treatment to ensure that leachate does not pose significant environmental and human health hazards. Diverse treatment methods are employed to resolve the complex composition of landfill leachate and to reduce its environmental impact.

2.3 Methods of Landfill Leachate Treatment

The primary objective of landfill leachate treatment procedures is to effectively eliminate or minimize the diverse range of contaminants found in leachate, with the ultimate goal of ensuring its safe discharge into the surrounding environment. The treatment procedure commonly encompasses a synergistic application of physicochemical and biological methodologies in order to efficiently address the intricate composition of leachate [52, 55]. Figure 2.2 displays the schematic illustrating several strategies employed for the treatment of landfill leachate.

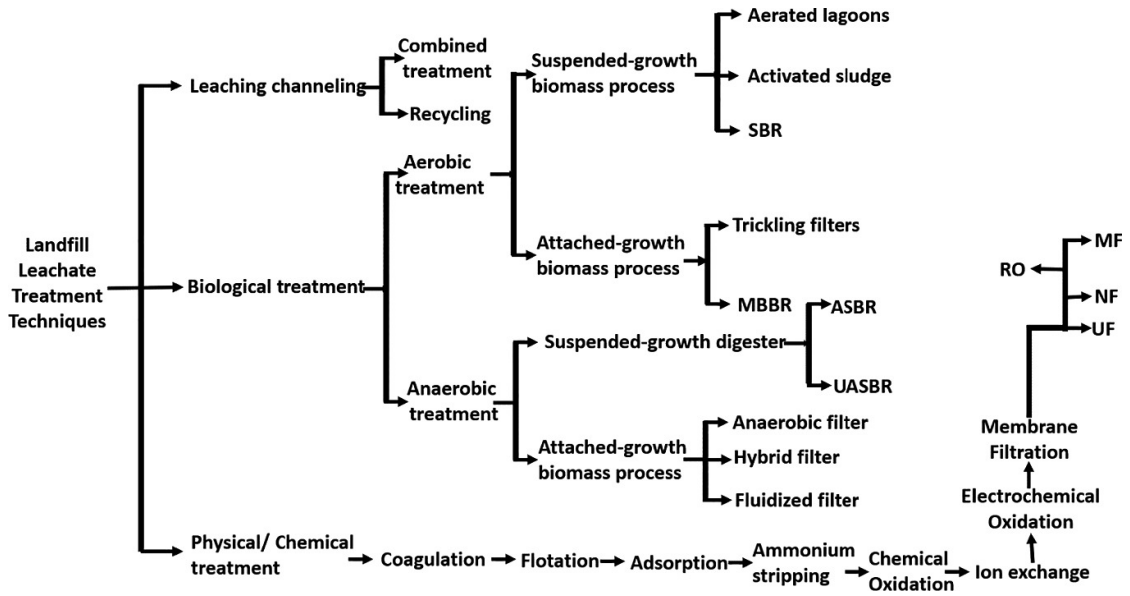


Fig. 2.2 Methods of landfill leachate treatment techniques

2.3.1 Biological Treatment Method

Biological treatment is an efficient technique for treating landfill leachate. Utilizing microbes to biodegrade and remove organic contaminants from leachate [39]. Typically, this process takes place in an aerobic environment, where oxygen is available to support the growth and activity of microorganisms. During biological treatment, microorganisms convert organic compounds into simpler, less hazardous substances, thereby reducing the pollutant burden in leachate [29]. The primary benefit of biological treatment is its ability to target and remove a broad spectrum of organic pollutants from landfill leachate [24]. In addition, the process is relatively economical and environmentally favorable when compared to chemical treatment methods.

However, biological treatment may have limitations, particularly when dealing with organic compounds that are difficult for microorganisms to degrade, such as those that are refractory or persistent. Therefore, it is common practice to combine biological and physicochemical treatment processes to obtain higher removal efficiencies for complex leachate contaminants.

2.3.1.1 Bioreactors

Bioreactors are commonly used to treat wastewater due to their simplicity, dependability, and cost-effectiveness [4, 66]. However, they have disadvantages, including sensitivity to temperature and leachate toxicity to microbial communities. Figure 2.3 depicts the membrane bioreactors' model configuration.

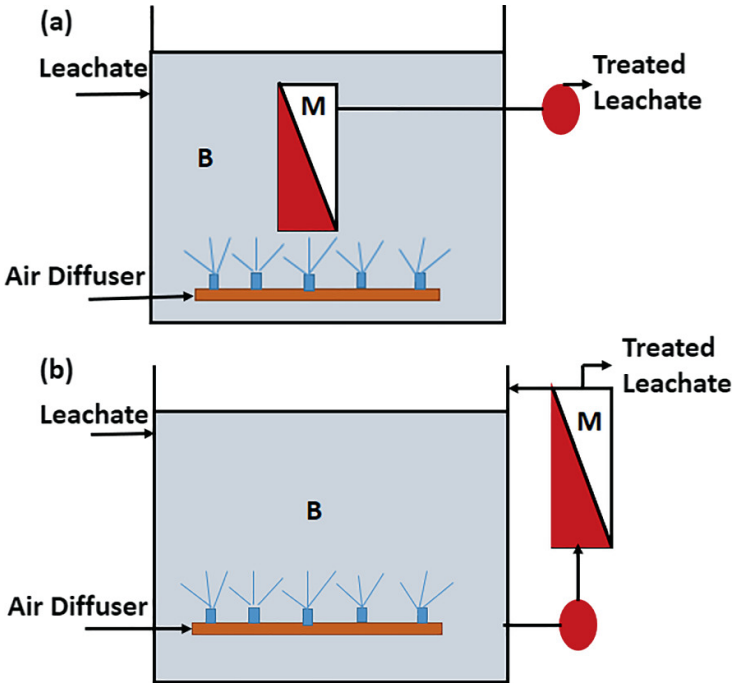


Fig. 2.3 A diagram of a membrane bioreactor's configuration. **a** The membrane module is installed within the reactor, **b** The membrane module is installed outside the reactor, M indicates membrane module, and B indicates bioreactor

Key factors impacting the effectiveness of membrane bioreactors encompass:

1. The ratio of BOD to COD;

2. The organic loading rate (OLR), with its influence varying based on the membrane reactor type;

3. Hydraulic and soild retention time (HRT).

This technique offers benefits, including resilience to leachate changes, easy sludge age control, decreased sludge production, and space efficiency [66]. Conversely, significant drawbacks hindering broader adoption comprise the expensive membranes, the need for cleaning, potential replacement due to clogging, and substantial power consumption.

Aerobic bioreactors are frequently utilized for biological treatments and involve sustained aeration of activated sediment composed of bacteria, fungi, and protozoa. The activated sludge process removes biodegradable organic matter from landfill effluent and converts it to carbon dioxide and water [19]. The sequencing batch reactor (SBR) is a popular method for landfill leachate remediation due to its batch operation, which improves process efficiency.

In contrast, anaerobic bioreactors perform better than aerobic techniques when treating landfill effluent. This is due to the elevated COD and BOD/COD ratio of leachates [4]. Anaerobic treatment has been demonstrated to be energy-efficient and ecologically sound, producing minimal amounts of sediment and biogas. However, limitations include extended retention periods, sensitivity to temperature fluctuations, and diminished elimination effectiveness. Anaerobic activated sludge processes may utilize upflow anaerobic sludge blanket (UASB) and expanded granular sludge blanket (EGSB) reactors for landfill leachate purification. These systems are distinguished by their increased microbial activity and cargo volume capacity. Principal advantages include substantial purification accomplishments in COD and BOD reduction and compact reaction volumes as a result of shortened hydraulic retention times.

Another effective process is anaerobic ammonium oxidation (anammox), where specific bacteria convert ammonium and nitrite into nitrogen gas, using CO_2 as a carbon source (Fig. 2.4) [66, 5]. The anammox process is represented by a specific equation (Eq. 2.1).

$$\begin{aligned} &NH_4 + 1.3\ H^+ + 1.32\ NO_2 + 0.06\ HCO_3 + 1.02\ N \\ &+ 0.26\ NO_3 + 2.03\ H_2O\ + 0.06\ CH_2O_{0.5}N_{0.15} \end{aligned} \tag{2.1}$$

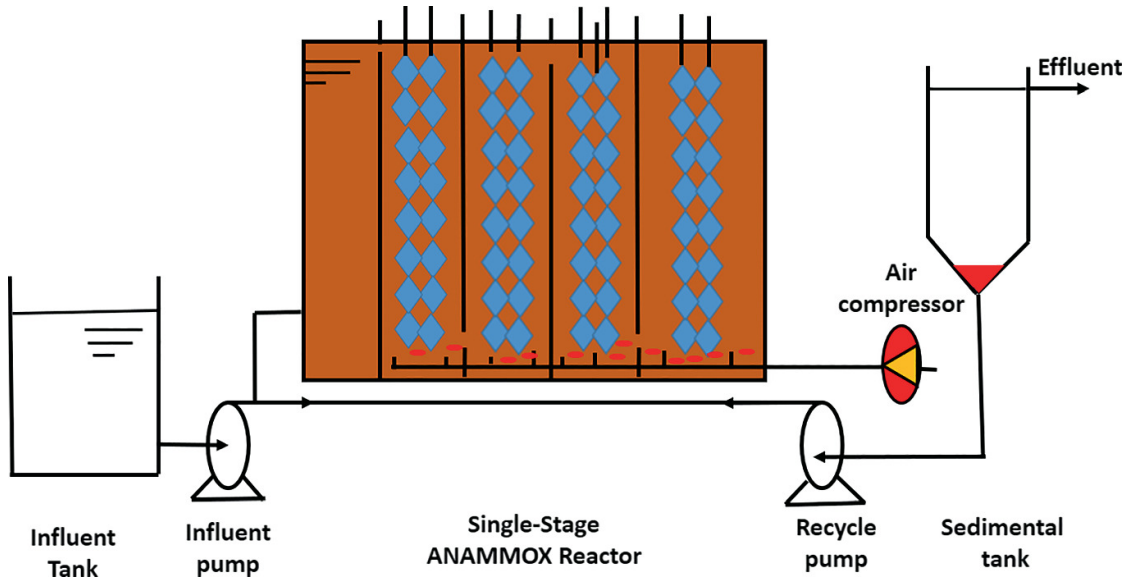


Fig. 2.4 ANAMMOX process described in flowchart (single-stage). Two rows formed the structure of the reactor, while peristaltic pumps were employed to introduce influent and recycled water

Anammox bacteria are known as monophyletic and include six candidate genera, namely *Candidatus jettenia*, *Candidatus anammoxoglobus*, *Candidatus brocadia*, *Candidatus scalindua*,

Candidatus anammoximicrobium, and Candidatus kuenenia [40]. Notably, anammox processes can be influenced by various categories of pollutants, including elevated COD levels and heavy metals.

As a result, it is common practice to integrate anammox reactors with supplementary treatment methodologies [64]. The Nitrification and denitrification processes for leachate treatment are depicted in Fig. 2.5.

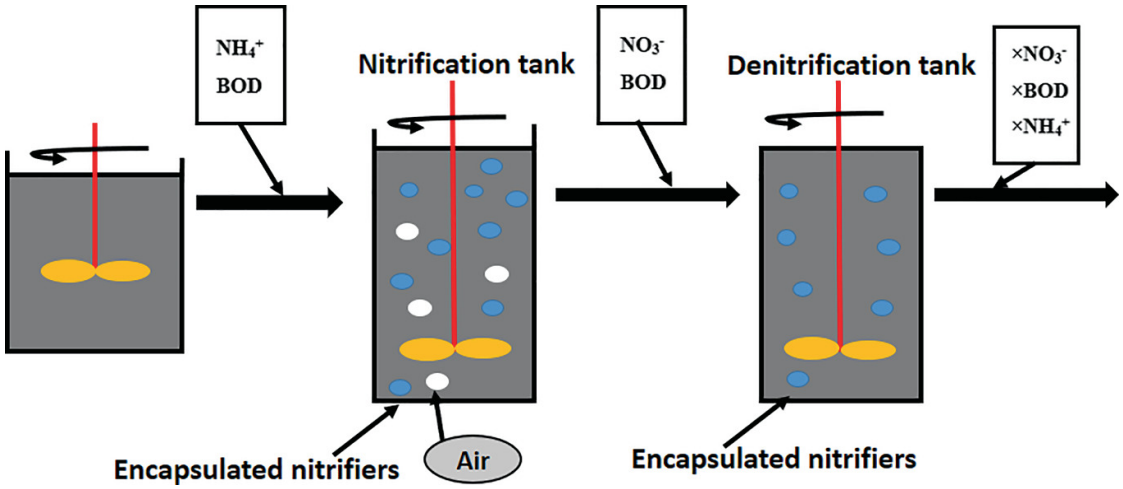


Fig. 2.5 Nitrification and denitrification processes for leachate treatment

The nitrification and denitrification processes in landfill leachate entail microbial transformations of ammonium. In an aerobic environment, ammonia is converted to nitrate. During a conventional nitrification–denitrification process, nitrate is converted into nitrogen gas (N_2) in an anoxic environment [41]. Ammonia is converted to nitrite by ammonia-oxidizing bacteria, and then nitrite is converted to nitrate by nitrite-oxidizing bacteria. In the final anaerobic phase, heterotrophic bacteria convert nitrate to nitrogen dioxide (N_2) [39]. Due to the influence of other pollutants on the process, denitrification is frequently combined with other treatment methods in landfill leachate treatment. The combination of nitrification and denitrification reduces ammonium concentrations and improves the overall treatment efficiency of landfill leachate.

2.3.1.2 Phytoremediation

Phytoremediation approaches exploit the capacity of plant-soil systems to mitigate and deactivate hazardous components present in leachate [1]. Phytoremediation delivers advantages like economical installation, low energy consumption, and efficient elimination of contaminants from landfill leachate. The successful treatment of landfill leachate using Lemna minor was demonstrated by Ramalho et al. [48]. During a contact period spanning 15 days, a significant proportion of metals, around 75%,

chemical oxygen demand (COD), approximately 42%, and biochemical oxygen demand (BOD), approximately 50%, were effectively eliminated. Abbas et al. [2] have also discovered that a range of aquatic plant species, such as Colocasia esculenta, Pistia stratiotes, Eichhornia crassipes, and others, have significant efficacy in the treatment of leachate.

$$\text{TF} = \frac{\text{Concentration of metals in aerial parts}}{\text{Concentration of metal in roots}} \tag{2.2}$$

$$\text{BCF} = \frac{\text{Concentration of metals in plant tissues}}{\text{Concentration of metal in substrate (water)}} \tag{2.3}$$

The success of phytoremediation relies on hyperaccumulator plants, which have the remarkable ability to accumulate metals. Hyperaccumulator plants can be discerned by evaluating their translocation factor (TF) and bioconcentration factor (BCF). The TF elucidates the plant's adeptness in transferring metals from root to shoot, whereas the BCF exemplifies the extent of metal accumulation within plant tissues. Specimens displaying BCF values surpassing two or TF values exceeding one are classified as hyperaccumulators [16, 18]. Incorporating hyperaccumulator plants into phytoremediation strategies enhances the effectiveness of leachate treatment and represents a promising approach for sustainable and eco-friendly remediation efforts.

2.3.1.3 Bioremediation

Bioremediation encompasses the utilization of biological processes to eliminate toxins from the environment, presenting economically viable and ecologically sustainable methodologies. The utilization of microalgae, algae, fungus, and bacteria for the purpose of bioremediating landfill leachate has been documented in many scholarly publications [59, 62]. Algae, including the genera Scenedesmus, Chlamydomonas, Chlorella, and cyanobacteria, have been utilized for the purpose of removing inorganic and simple organic molecules, while certain complex substances undergo biotransformation [22]. In addition, the utilization of notable bacterial species in the process of landfill leachate remediation includes Firmicutes, Actinobacteria, Proteobacteria, Brevibacillus panacihumi strain ZB1, and Pseudomonas putida [42].

2.3.1.4 Biological Co-treatment of Landfill Leachate and Municipal Wastewater

Researchers have investigated various approaches to enhance landfill leachate biodegradability and improve BOD/COD ratios. For example [40] studied the co-treatment of domestic wastewater with landfill

leachate, resulting in improved biodegradability. Ranjan et al. [49] used a Sequential Batch Reactor (SBR) for co-treatment of urban wastewater and landfill leachate, achieving significant removal efficiencies: 93% ammonia, 83% nitrite, 70% COD, and 83% turbidity removal with 20% v/v landfill leachate concentration and 6-day HRT. Addressing high COD and BOD/COD ratios in landfill leachate, as highlighted by Mojiri et al. [40], requires a combined treatment system. Chen et al. [20] employed denitrification/partial nitrification-anammox for nitrogen removal from intermediate landfill leachate, achieving a 96.7% TN elimination efficacy and a 0.45 m^3/d TN removal rate. The process improved degradable COD for nitrate removal and autotrophic bacteria growth. Miao et al. [39] utilized a combined aerobic-anaerobic/biogranular activated carbon SBR for high-performance biodegradable leachate treatment. These studies emphasize the importance of innovative technologies and combined treatment systems to improve landfill leachate treatment, especially concerning COD and nitrogen removal challenges, leading to more sustainable management practices.

2.3.2 Physical and Chemical Treatment Method

Physio-chemical methodologies are commonly employed with biological processes, primarily aimed at amplifying treatment efficacy or augmenting biodegradability. This integration is particularly pertinent when biological oxidation encounters limitations posed by recalcitrant biomaterials. A spectrum of physical and chemical treatment modalities, encompassing techniques such as coagulation, flocculation, flotation, chemical precipitation, adsorption, ammonium stripping, chemical oxidation, ion exchange, electrochemical oxidation, and membrane filtration, is engaged to effectually eliminate non-biodegradable constituents (e.g., humic and fulvic acids) and undesired substances (notably heavy metals) from landfill leachate [52, 1].

2.3.2.1 Adsorption and Ion-Exchange

The specific characteristics of the adsorbent surface facilitate the attachment of the adsorbate. Adsorption is reversible, allowing for desorption and releasing adsorbates back into the liquid [59, 30]. For instance, an illustrative example involves the utilization of modified activated carbon (MAC), a derivative product resulting from the immersion of granular activated carbon in a solution of $KMnO_4$. This innovation exhibits remarkable efficacy in the treatment of landfill leachate. MAC has demonstrated the capability to achieve a removal efficiency of approximately 99% for ammonia and 86% for zinc within 120 min. According to Erabee et al. [25], the Langmuir adsorption capacity of MAC for ammonia and zinc removal has been determined to

be 0.16 mg/g. In a distinct research endeavour conducted by Abuabdou et al. [4], the approach of employing an ion-exchange resin for landfill leachate treatment is showcased. Notably, this strategy showcases substantial adsorption capacities, as deduced from a pseudo-second-order kinetic model. The upper limits of adsorption capacities are notably recorded as 13.4 mg/g for Cr^{6+}, 13.5 mg/g for Al^{3+}, 14.2 mg/g for Cu^{2+}, 33,333.3 mg/g for COD, 10,000.0 mg/g for ammonia, and 50,000.0 mg/g for color removal. Erabee et al. [25] highlighted the broad application of adsorption for landfill leachate treatment due to its operational simplicity and versatility in removing contaminants.

Various adsorbents and their performance are outlined in Table 2.3. This method involves pollutants adhering to the surface of adsorbents through multiple mechanisms (Fig. 2.6).

Table 2.3 Adsorbents reported for landfill leachate treatment

Adsorbents	Pollution (landfill leachate)	Adsorption isotherm	Volume	Notes
Activated carbon (AC)	Pb Cu Ni	Pseudo-second order	0.03 0.01 0.01	The origin of AC stemmed from the utilization of sugarcane bagasse
Biochar	COD	Pseudo-second order	490	Biochar was procured from coconut shells under elevated temperatures, subsequently subjected to activation through microwave heating
		Freundlich	5.80	The production of biochar from Miscanthus occurred at a temperature of 450 degrees
Magnetic graphene oxide	FA HA Pb	Langmuir	82.16 106.50 45.50	
Bentonite modified by hexadecyl trimethyl ammonium bromide (HTAB)	Bisphenol A	Pseudo-second order	10.44	The HTAB-bentonite was synthesized by cation exchange with HTAB solution (20 mmol/L) over stirring
Red mud	Ni Zn Cd	Langmuir	11.06 12.04 12.57	Batch experiments were conducted under conditions of neutral pH, utilizing an adsorbent dosage of 10 g/L, and employing a stirring speed of 75 rotations per minute
Zeolites (Clinoptilolite)	Ammonia	Langmuir	17.45	

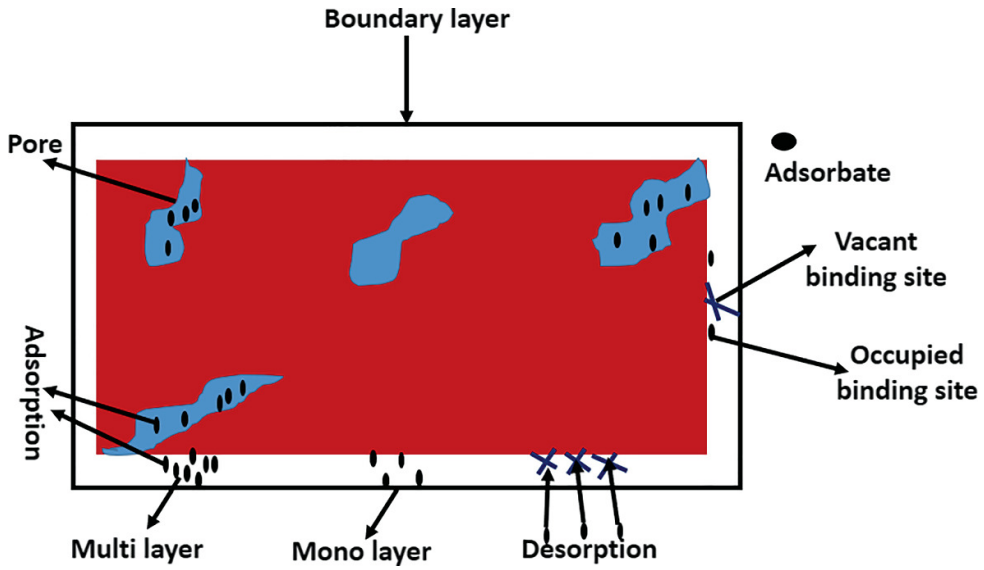


Fig. 2.6 Basic model of adsorption

These findings underscore the potential of adsorption methods using various adsorbents, such as modified activated carbon and ion-exchange resins, to effectively address the challenges posed by landfill leachate treatment, offering valuable insights for sustainable remediation strategies.

2.3.2.2 Advanced Oxidation Process

Advanced oxidation processes (AOPs) blend catalysts and oxidants to generate hydroxyl radicals (•OH) within solutions. Such processes encompass ultraviolet (UV) irradiation, Fenton reactions, ozonation, and electrochemical oxidation (EO) methods and have garnered significant attention for the breakdown of recalcitrant organic compounds or recalcitrant compounds in wastewater [9] (Fig. 2.7).

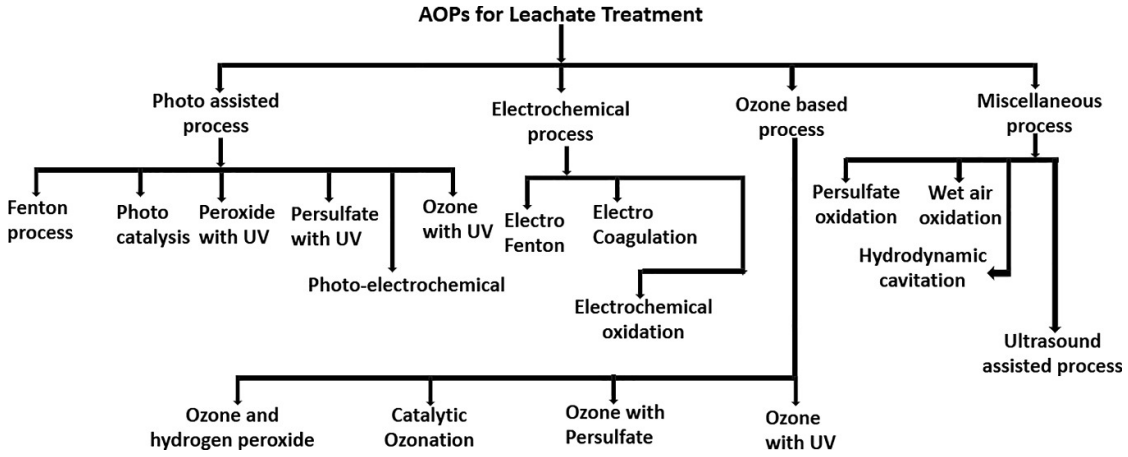


Fig. 2.7 Advanced oxidation processes for leachate treatment

Notwithstanding their efficacy, AOPs are impeded by the substantial expenses associated with their implementation and operation [26, 47]. In an electrochemical oxidation (EO) procedure, the expulsion of contaminants unfolds through either (a) direct EO, entailing the direct oxidation of organics by channelling electrons towards an anode, or (b) indirect EO, wherein specific electroactive species function as mediators and are generated to facilitate the degradation process. Principal focal points within catalyst-oriented Advanced Oxidation Processes (AOPs) for leachate treatment encompass the meticulous examination of structural attributes and the meticulous assessment of operational efficacy (Table 2.4).

Table 2.4 Advanced oxidation processes

AOPs	Method	Advantages	Disadvantages	Efficiency
Photo-assisted process	Light adsorption and radical generation $Fe^{2+} + H_2O_2 \rightarrow Fe^{3+} + OH^- + {}^*OH$	The system operates under standard atmospheric conditions, potentially harnessing solar irradiation. The absence of impediments in mass transfer ensures the efficient conversion of organic compounds into benign substances, notably carbon dioxide and water	A reduction in energy consumption can be achieved by moderating catalyst loading to counteract heightened opacity. The influence of pH on photocatalytic oxidation is multifaceted and intricate	The majority of degradable compounds underwent degradation, leading to a scarcity of biodegradable intermediates in the process
Ozone-based process	Molecular ozone reactions and OH radical reactions	Make ozone work better; Make ozonation work better by controlling how much radicals are made; Make more radicals at lower pH; Help reactions on the surface between ozone and pollutants stuck to it	Alone, it can't clean well enough; Using a lot of ozone needs lots of energy; Not much ozone goes from gas to liquid	Hange how things are built at a tiny level and make them easier to break down; Make leachate less dirty by reducing COD and BOD a lot; Use before or after regular leachate treatment
Electrochemical process	Indirect oxidation (oxidation through the mediator); direct oxidation (oxidation at anode surface)	Bipolar electrochemical reactors are simple to use and control. They work at regular temperature and pressure, which stops leftover waste from leaking or becoming gas. It doesn't use chemicals	This process uses a lot of energy and can make harmful chemicals. The cost is high due to pricey metal parts	It's better at getting rid of color than COD. It can change pollution into carbon dioxide and water

AOPs	Method	Advantages	Disadvantages	Efficiency
Miscellaneous process	Various advanced processes were used, like Wet air oxidation (WAO), ultrasound treatment, persulfate oxidation, and hydrodynamic cavitation (HC). The results showed that these processes helped: (a) Turn organic stuff in leachate into more stable forms, (b) Make tough substances easier for bacteria to break down later. Details about how each process works and its pros and cons were not discussed here			

The delineation of structural attributes can be manifested through spectroscopic or morphological methodologies. Paramount attention must be given to catalytic degradation effectiveness and sustained operational stability. Mechanistic intricacies predominantly encompass the discernment of intermediary and terminal byproducts, coupled with pertinent analyses of degradation paths.

2.3.2.3 Membrane Technology

Membrane technology is a procedure that separates fluids using a selective barrier [8]. Various membrane filtration techniques, including microfiltration, ultrafiltration, nanofiltration, and reverse osmosis, have been employed for landfill leachate treatment in recent years [67] (Fig. 2.8).

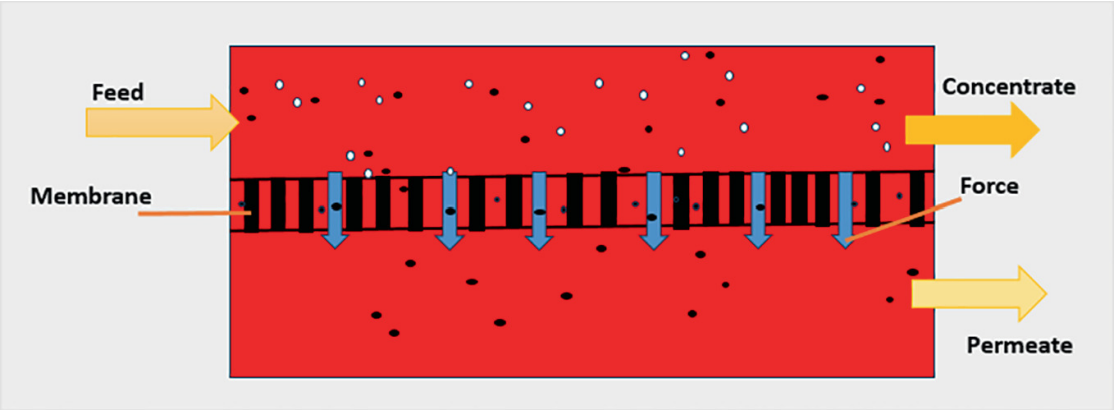


Fig. 2.8 Schematic diagram of the membrane filtration process

1. **Microfiltration**: With pore sizes of 0.05 to 10 microns, microfiltration captures microbial cells and particles. It is recommended as pretreatment or in combination with other processes. Zhang et al. [68] achieved COD removal of 25% to 35% using this as a pre-filtration stage.

2. **Ultrafiltration**: Operating at up to 10 bar, ultrafiltration effectively removes suspended matter. Hang et al. [9] suggested it as a reverse osmosis pretreatment. COD removal of 50% was achieved solely through ultrafiltration [36]. Full-scale membrane bioreactor plants have also effectively employed ultrafiltration [8].

3. **Nanofiltration**: Nanofiltration eliminates refractory organics and heavy metals positioned between ultrafiltration and reverse osmosis. COD and NH_3-N removals reached 60% to 70% and 50%, respectively [36]. Successful use requires effective membrane fouling control [68].

4. **Reverse osmosis**: This newer method efficiently treats mature landfill leachate by separating solutions of different concentrations [34, 56]. Kunwar et al. [34] improved treatment efficiency in Korea with reverse osmosis, achieving 96% COD and 97% NH_3-N removal. It is considered the most efficient among physical–chemical technologies for COD removal. However, challenges like membrane clogging and waste disposal need attention [34, 56]. These membrane techniques present promising options for effective and advanced landfill leachate treatment, offering solutions to manage contaminants and improve overall treatment efficiency.

2.3.2.4 Coagulation and Flocculation

Coagulation causes colloidal particles in wastewater to agglomerate, allowing them to form settleable flocs. Effective coagulation/flocculation is dependent on the choice of coagulants/flocculants, which are often trivalent-metal inorganic salts. According to Ndao et al. [44], this method efficiently removes organic matter, suspended solids, and humic acids. However, challenges include chemical costs and sludge management. Soares et al. [57] investigated natural chitosan as a landfill leachate coagulant, obtaining 80% color and 91.4% turbidity reduction at 960 mg/L chitosan dosage and pH 8.5. Cheng et al. [21] used pine bark as a natural coagulant and achieved an 85.2% turbidity reduction from landfill leachate at a pH of 7 with a 4 g/mL coagulant dosage.

2.3.2.5 Hybrid Physical/chemical Methods

Various physical/chemical methods have been combined to enhance removal efficiency and reduce energy consumption in landfill leachate treatment. Cheng et al. [21] proposed promising hybrid approaches, particularly AOPs, combined with other treatments. Four integrated systems have been identified:

1. **AOPs combined with membranes**: This integrated technique can efficiently address membrane fouling issues and enhance the separation performance. Ndao et al. [44] achieved 94–96% COD and 96–99% color removal using the Fenton, NF, and MF processes. Due to incomplete oxidation and chemical additions, membranes resolve high dissolved solids post AOP-Fenton.

2. **AOPs combined with coagulation**: [21] shown that by modifying residual organic structures, this technique can minimise organic pollutants and enhance biodegradability. Chen et al. [20] discovered flaws in electrocoagulation methods, however combining photoelectrooxidation with activated carbon eliminated 70.3% COD, 58.3% ammonia, and 58.4% TN. Ayala et al. [12] used coagulation-ozonation to remove 88.3% COD, 98.8% color, and 94.3% UV254.

3. **AOPs combined with adsorption**: Integrating AOPs with adsorption enhances pollutant removal efficiency, especially metals. Complex contaminants degraded by AOPs can generate intermediates. Integrated H2O2-granular activated carbon reduced 97.3% COD and increased biodegradability by 116%. Researchers highlighted enhanced peroxide decomposition and performance with H2O2-activated carbon. Chen et al. [20] removed 99.8% tetracycline using Heterogeneous Fenton: activated carbon–Fe_3O_4.

4. **Membrane filtration with coagulation or adsorption**: Coagulants or adsorbents administered sequentially to membranes remove suspended and colloidal particles, lowering organic load and fouling. Kamal et al. [31] demonstrated the role of coagulation in MBR systems in decreasing transmembrane pressure. Coagulation pre-treatment additionally postpones both reversible and irreversible fouling. Through the use of an integrated coagulation-membrane bioreactor [64] eliminated almost 90% of the Al. Combining membrane and coagulation, 99.2% COD, 100% suspended particles, and 97.3% total organic carbon were eliminated. 100% of the 4-chlorophenol and 78–100% of the oxidation intermediates were eliminated by catalytic oxidation and adsorption [20] (Table 2.5).

Table 2.5 Combined physicochemical treatments for landfill leachate treatment

Combined treatment methods	pH	Initial COD (mg/L)	COD removal efficiency (%)
Coagulation-flocculation/UV + PMS	7.95	5123 ± 281	91.5
Electrocoagulation/filtration			94
Filtration/adsorption	8.5	1101	94.6
Ozonation/oxidation			92
Fenton/UF	7.7	1750	80
Fenton/adsorption	8.31	10,193	> 90
Fenton/adsorption			> 90
Electrocoagulation/electro-oxidation/AOP	6.4	8580 ± 200	95.6

Combined treatment methods	pH	Initial COD (mg/L)	COD removal efficiency (%)
Electrocoagulation/flotation		13,600	96
Coagulation-flocculation/EAOP	8.37	1585	> 90
Filtration/adsorption/photo Fenton + ozone	8.9	11,950	95.1
Photo electro-oxidation/adsorption	7.73	1113	67.2
Coagulation-flocculation/solar photo Fenton	8.3	6200	73
Coagulation-flocculation/adsorption	8	6730	86
NF/coagulation/photo Fenton	8.5	1330	91.5
Coagulation/ozonation	8.2	4000	87
Ultrasonic/Fenton	7.8	14,000	92
Coagulation-flocculation/adsorption			43.24
Coagulation/sand filtration/MF/RO	8.44	2700	99.4
Fenton/adsorption	8.4	3135	82
Coagulation-flocculation/adsorption	8.03	8951	82
Fenton/Filtration/adsorption	8.3	5346	99.9
Fenton/coagulation-flocculation	8.3	704	85
Combined treatment methods	pH	Initial COD (mg/L)	COD removal efficiency (%)
Coagulation-flocculation/photo Fenton	8.0	2800	68
RO/electro-oxidation	7.94	2589	> 86
Coagulation-flocculation/adsorption	7.61	22,000	59

2.3.3 Hybrid Physical/chemical and Biological Methods

Landfill leachate treatment often involves biological approaches. However, relying solely on biological processes is insufficient for effectively eliminating most refractory contaminants in landfill leachate. Researchers propose combining biological and physical/chemical techniques to address this limitation, enhance biodegradability ratios, and optimize treatment performance. Five common combined treatment approaches have been identified.

2.3.3.1 Integrated Adsorption and Biological Treatment

Combining adsorption, such as activated carbon, with biological methods allows microbial growth to thrive in improved conditions due to reduced contaminants and leachate toxicity. This approach offers advantages like shielding microorganisms from inhibitory compounds, enhancing the degradation of refractory organics, and improving sludge characteristics. Moreover, integrating adsorption with biological methods reduces the required adsorbent quantity [59]. For instance, incorporating sawdust

into an SBR (Sequencing Batch Reactor) can eliminate 99% of COD and 95% of ammonia. Integrating adsorption and biodegradation achieves over 60% removal of ampicillin and more than 70% reduction of ammonia in landfill leachate [62].

2.3.3.2 Integrated Membrane and Biological Treatment

The membrane bioreactor innovation, especially the anaerobic membrane bioreactor (AnMBR), addresses the limitations of traditional methods like excessive sludge production and recalcitrant removal. AnMBR, designed to treat heavy wastewater like leachate, decouples HRT from SRT for efficient processing. However, initiating AnMBR at temperatures below 20 °C may hinder biomass growth. AnMBR has effectively removed over 90% of sulphonamides, tetracyclines, and COD from landfill leachate [22].

2.3.3.3 Integrated AOP and Biological Treatment

Combining Advanced Oxidation Processes (AOP) as pre-treatment generates readily biodegradable intermediates, facilitating biological post-treatment. AOP techniques like zone oxidation, photo catalysis, and EO enhance biodegradability. A hybrid reactor with ozone pre-treatment and bioreactor removes more than 70% of aromatic pollutants, while a photocatalytic pre-oxidation reactor combined with SBR reduces COD to less than 50 mg/L [57, 21]. Integrating ozonation and membrane bioreactors achieves high removal percentages for pharmaceuticals like Etodolac and various contaminants, including sulfadiazine and organic carbon [56].

2.3.3.4 Integrated Coagulation and Biological Treatment

Coagulation/flocculation, either pre-treatment or post-treatment, and biological approaches increase biodegradability and decrease COD, metals, and color in landfill leachate. This is known as an integrated coagulation and biological treatment. Refractory pollutants are removed by coagulation or flocculation as a post-treatment, with active biomass in rejected water acting as an additional means of removing dissolved organics. 72% of COD and 70% of total organic carbon can be removed when coagulation and an anaerobic bioreactor method are combined [44].

2.3.3.5 Constructed Wetlands

Constructed wetland (CW) systems, composed of permeable substrata and wetland plants, effectively remove degradable organic carbon, ammonia, nitrogen pollutants, and heavy metals from landfill leachate. Microbial activities, nitrification/denitrification processes, and plant root absorption contribute to nitrogen removal. Vertical flow-constructed

wetlands remove significant phenolic compounds, while diverse mechanisms, including adsorption and plant uptake, eliminate heavy metals. Notably, CWs have removed up to 92.2% of Zn, 96.8% of Ni, 99.5% of Cu, 87.5% of Cr, and 98.1% of Pb in studies [62, 8]. Thus, this extreme fluctuation of results can be due to the treatment time. Other combined physicochemical and biological treatment methods for landfill leachate are highlighted in Table. 2.6.

Table 2.6 Combined physicochemical and biological treatments for landfill leachate treatment

Combined treatment methods	pH	Initial COD (mg/L)	COD removal efficiency (%)
Photo Fenton/activated sludge	–	12,797	98
Oxidation/coagulation-flocculation	8.15	985	> 90
Aerobic biological oxidation/Fenton	7.6	6790	> 90
Electrocoagulation/anaerobic activated sludge	9	6400	92
Ozonation/aerobic oxidation	6.2	850	73
UASB/SBR/MF/RO	6.5–9	8000–20,000	99.8
Coagulation/hydrolysis acidification-SBR/electro-Fenton	7.1	9765	97.8
Electro-ozonation/SBR	7.34	3018	88.2
SBR/aeration cell-Fenton/adsorption	6.7	41,800	97.2
Fenton/UASB-downflow hanging sponge (DHS)	8.78	4358	85
Coagulation/Fenton/biological aerated filtering	5–8	600–700	88
SBR/coagulation/Fenton/upflow biological aerated filter	8.75	3000	97.3
Air stripping/aerobic and anaerobic processes	7.9	30,000	80
Coagulation-flocculation/SBR	7.11	53,199	98
Submerged anaerobic membrane bioreactor/MBR	7.5	570	96.1
SBR/electro-oxidation	8.6	3973	98
SBR/RO	7.84	2180	96.7
Fenton/aerated biomass	8.5	3909	83
SBR/Fenton-like/SBR	8.2	22,300	95
Electro-Fenton/MBR	8.1	23,200	94.5
Aerobic SBR/adsorption	9.66	3200	43
Constructed wetland/adsorption	7.95	2301	86.7
Photo Fenton/MBR	8.1	24,000	96.2
MBR/UF/electro-oxidation		2122	94
Trickling filters/electrocoagulation	8.21	765	80
Activated sludge/coagulation/photo Fenton	7.5	4084	96

Combined treatment methods	pH	Initial COD (mg/L)	COD removal efficiency (%)
Coagulation-flocculation/constructed wetland	5.82	902.5	92.6

In summary, combining AS-RO appears more efficient than MBR-RO, RBC-RO, and UASB-RO, leading to the complete removing COD and N-NH_4 [59]. Applying ammonia stripping with UASB-RO can boost landfill leachate treatment's effectiveness. NF is a viable polishing step for MBR in leachate treatment. Integrated MBR-NF systems typically achieve high pollutant removal, suitable for landfill reuse. However, high costs require justification via public health or environmental benefits. SBR-coagulation integration shows promise for COD removal, while plastic media increases color removal in SBR [22]. Activated sludge type, coagulant, treatment time, and filters impact removal rates. Process order also matters, coagulation as polishing achieves 85% COD removal, while pre-treatment reaches 60%. Energy-wise, anaerobic and phytoremediation methods excel, unlike energy-intensive aerobic methods. Aeration consumes 50–70% of energy in wastewater treatment; energy-efficient technologies like better blowers, diffusers, and oxygen sensors can improve this efficiency [28].

2.4 Combined Physicochemical Treatment Methods Versus Combined Physicochemical and Biological Treatment Method

2.4.1 Selective Effectiveness of Processes

- Different combined treatment methods have varying efficiencies in addressing specific issues such as color removal or heavy metal reduction. The selective effectiveness of process is shown in Fig. 2.9.

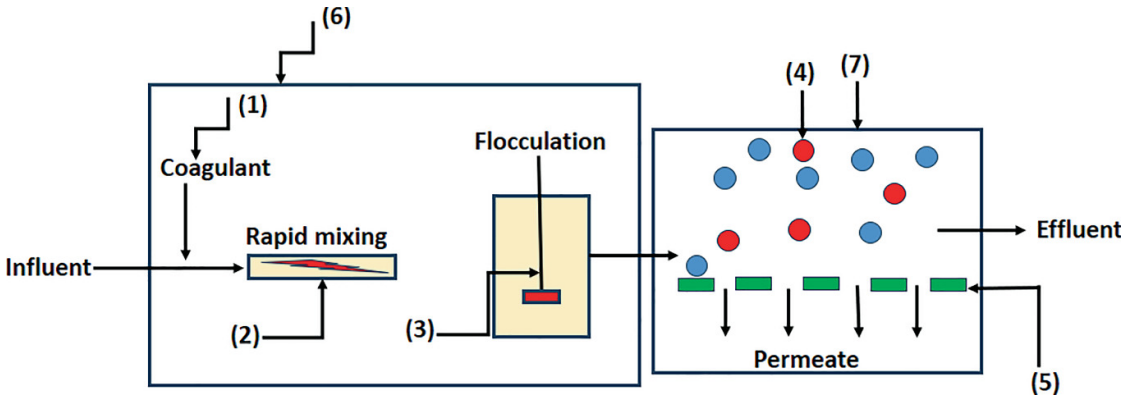


Fig. 2.9 Add coagulant in process (1), Coagulation (2) Stirrer (3), Pollutants (4), Membrane (5), Coagulation-flocculation configuration (6), and Nanofiltration configuration (7)

- The optimal treatment approach should be both cost-effective and capable of reducing all pollutants to permissible discharge levels.

2.4.2 Coagulation-Flocculation and Nanofiltration

- The combination of coagulation-flocculation and nanofiltration is an effective method for cleaning leachate. COD, TOC, humic compounds, color, and NH3-N levels can all be reduced with this method.
- Coagulation-flocculation can aid nanofiltration by preventing the membrane from becoming clogged.
- Tiny particles clump together in coagulation, and they form clumps in flocculation.
- Nanofiltration is effective in removing impurities, resulting in clean and safe water.

2.4.3 Activated Sludge and Reverse Osmosis

- For the treatment of landfill leachate, the combination of activated sludge and reverse osmosis (Fig. 2.10) stands out among various physical, chemical, and biological processes.

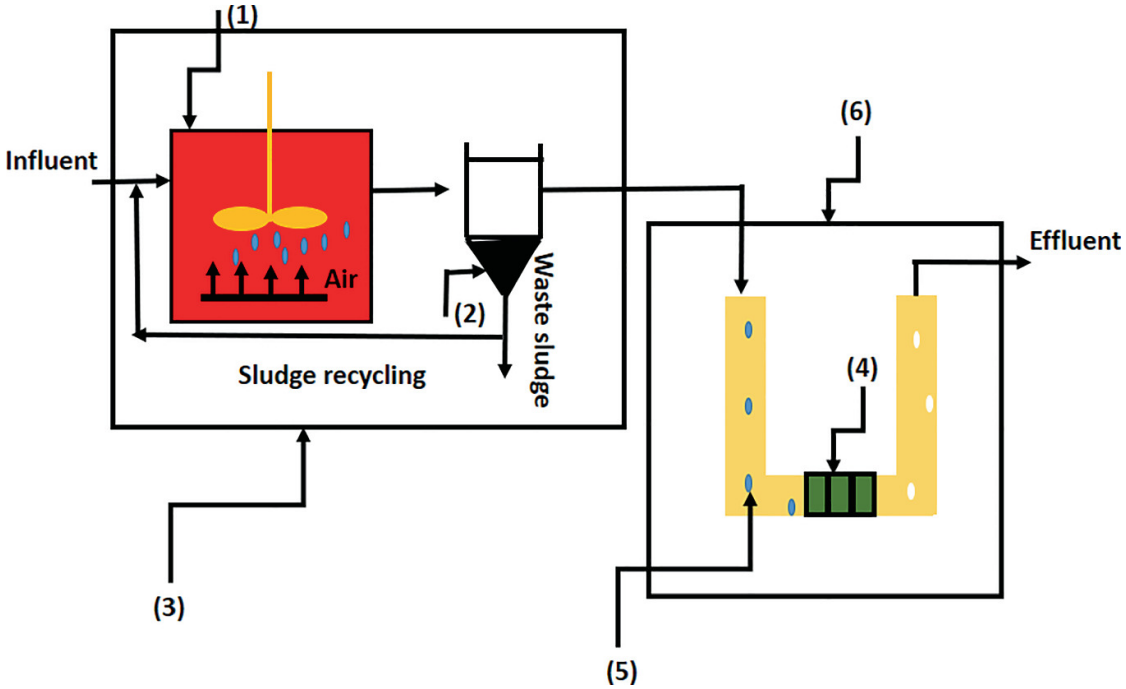


Fig. 2.10 Aeration tank (1), Clarifier (2), Activated sludge process (3), Semi-permeable membrane (4), Contaminants (5), and (6) Reverse osmosis process

- By combining these two processes, COD and NH_4-N emissions are reduced almost entirely. The activated sludge process uses microorganisms to break down organic waste, which calls for a constant supply of oxygen to maintain aerobic conditions.

- A part of the clarifier's settled solid activated sludge is recycled to the aeration tank.
- To separate particles and counteract osmotic pressure, reverse osmosis uses a semi-permeable membrane and pressure.

In summary, different treatment methods excel in different aspects, emphasizing the need for a holistic approach that considers cost-effectiveness and comprehensive pollutant reduction. The integration of coagulation-flocculation with nanofiltration shows promising results, and the combination of activated sludge with reverse osmosis demonstrates impressive potential for landfill leachate treatment.

2.5 Conclusions

Landfill leachate is a potent source of pollution due to its high levels of harmful substances. To effectively treat landfill leachate, it is recommended to use combined methods like membrane filtration and integrated biological processes. These methods have successfully removed nitrogen, ammonia, suspended solids, and turbidity. However, bioremediation methods have shown varying efficiency when it comes to reducing COD (chemical oxygen demand). Additionally, treatments involving physical and chemical processes effectively eliminate heavy metals. Combining different treatments is beneficial to achieve the best results, as it improves the quality of the treated water, reduces waste, and cuts down on costs. However, some processes like air stripping, ion exchange, and membrane need better fine-tuning to address specific pollution situations. Recently, advanced oxidation processes and biological treatments have gained attention, with many treatment plants globally using these methods. Around 60% of these plants start with a biological phase in their treatment process. Problematically, countries with low-income levels face significant challenges regarding their restricted availability of sanitary landfills, recycling facilities, and incinerators. The improper handling and disposal of waste materials, characterized by their lack of management, mistreatment, and disposal in open pits, have significant adverse effects on human health and the surrounding environment. Furthermore, the contamination has the potential to diffuse into Transboundary rivers, transforming this issue into a regional concern. This suggests that the implementation of cost-effective landfill leachate facilities is necessary on a global scale. Furthermore, it is imperative to implement a comprehensive and sustained monitoring and maintenance regimen and conduct further research to ascertain the efficacy of the implemented strategies and address any potential issues that may arise in the future.

References

1. Abbas Z, Arooj F, Ali S, Zaheer IE, Rizwan M, Riaz MA (2019) Phytoremediation of landfill leachate waste contaminants through floating bed technique using water hyacinth and water lettuce. Int J Phytorem 21(13):1356–1367
2. Abbas J, Hadi H (2023) A comparison of results among waste management producers: a case study for the process of waste management. Eng Technol J 41(4):586–591
3. Abood AR, Bao J, Du J, Zheng D, Luo Y (2014) Non-biodegradable landfill leachate treatment by combined process of agitation, coagulation SBR and filtration. Waste Manage 34(2):439–447. https://doi.org/10.1016/j.wasman.2013.10.025
4. Abuabdou SM, Ahmad W, Aun NC, Bashir MJ (2020) A review of anaerobic membrane bioreactors (AnMBR) for the treatment of highly contaminated landfill leachate and biogas production: effectiveness, limitations and future perspectives. J Clean Prod 255:120215
5. Agarwal A, Ramana GV, Datta M, Soni NK, Satyakam R (2022). Environmental assessment of unprocessed sand-sized construction & demolition waste for geotechnical reuse. J Clean Prod 363:132504
6. Alghamdi AG, Aly AA, Ibrahim HM (2021) Assessing the environmental impacts of municipal solid waste landfill leachate on groundwater and soil contamination in western Saudi Arabia. Arab J Geosci 14(5):350
7. Alghamdi AG, Aly AA, Ibrahim HM (2021) Assessing the environmental impacts of municipal solid waste landfill leachate on groundwater and soil contamination in western Saudi Arabia. Arab J Geosci 14(5):350
8. de Almeida R, Porto RF, Quintaes BR, Bila DM, Lavagnolo MC, Campos JC (2023) A review on membrane concentrate management from landfill leachate treatment plants: the relevance of resource recovery to close the leachate treatment loop. Waste Manage Res 41(2):264–284
9. Anqi T, Zhang Z, Hao S, Li X (2020) Review on landfill leachate treatment methods. IOP Conf Ser 565(1):012038. https://doi.org/10.1088/1755-1315/565/1/012038
10. Ashraf MA, Balkhair KS, Chowdhury AK, Hanafiah MM (2019) Treatment of Taman Beringin landfill leachate using the column technique. Desalin Water Treat 149(2019):370–387
11. Assou M, Fels LE, Asli AE, Fakidi H, Souabi S, Hafidi M (2016) Landfill leachate treatment by a coagulation–flocculation process: effect of the introduction order of the reagents. Desalin Water Treat 57(46):21817–21826. https://doi.org/10.1080/19443994.2015.1127779
12. Ayala J, Fernández B (2019) Treatment from abandoned mine landfill leachates. Adsorption technology. J Mater Res Technol 8(3):2732–2740. https://doi.org/10.1016/j.jmrt.2019.04.009
13. Azzouz L, Boudjema N, Aouichat F, Kherat M, Mameri N (2018) Membrane bioreactor performance in treating Algiers' landfill leachate from using indigenous bacteria and inoculating with activated sludge. Waste Manag 75:384–390
14. Babaei S, Sabour MR, Movahed SMA (2021) Combined landfill leachate treatment methods: an overview. Environ Sci Pollut Res 28(42):59594–59607. https://doi.org/10.1007/s11356-021-16358-0
15. Bandala ER, Liu A, Wijesiri B, Zeidman AB, Goonetilleke A (2021) Emerging materials and technologies for landfill leachate treatment: a critical review. Environ Pollut 291:118133

16. Bhagwat A, Ojha CSP, Pant A, Kumar R (2023) Interaction among heavy metals in landfill leachate and their effect on the phytoremediation process of Indian Marigold. J Hazardous, Toxic, and Radioact Waste 27(1):04022039
17. Bove D, Merello S, Frumento D, Arni SA, Aliakbarian B, Converti A (2015) A critical review of biological processes and technologies for landfill leachate treatment. Chem Eng Technol 38(12):2115–2126. https://doi.org/10.1002/ceat.201500257
18. Chan MY, Tee CS, Chai TT, Sim YL, Beh WL (2022) Evaluation of electro-assisted phytoremediation (EAPR) system for heavy metal removal from synthetic leachate using Pistia stratiotes. Int J Phytorem 24(13):1376–1384
19. Chelliapan S, Arumugam N, Din MFM, Kamyab H, Ebrahimi SS (2020) Anaerobic treatment of municipal solid waste landfill leachate. In Bioreactors. Elsevier, pp 175–193
20. Chen G, Wu G, Li N, Lu X, Zhao J, He M, Yan B, Zhang H, Duan X, Wang S (2021) Landfill leachate treatment by persulphate related advanced oxidation technologies. J Hazardous Mater 418:126355
21. Cheng SY, Show PL, Juan JC, Ling TC, Lau BF, Lai SH, Ng EP (2020) Sustainable landfill leachate treatment: Optimize use of guar gum as natural coagulant and floc characterization. Environ Res 188:109737
22. Choudhury AR, Singh N, Palani SG, Lalwani J (2023) Bioremediation of reverse osmosis concentrate generated from the treatment of landfill leachate. Environ Sci Pollut Res 1–18
23. Costa AM, Alfaia RGDSM, Campos JC (2019) Landfill leachate treatment in Brazil–an overview. J Environ Manage 232:110–116
24. Dhamsaniya M, Sojitra D, Modi H, Shabiimam MA, Kandya A (2023) A review of the techniques for treating the landfill leachate. Mater Today: Proc 77:358–364
25. Erabee IK, Ahsan A, Imteaz M, Alom MM (2019) Adsorption of hexavalent chromium using activated carbon prepared from garden wastes. Desalin Water Treat 164:293–299
26. Feng H, Mao W, Li Y, Wang X, Chen S (2021) Characterization of dissolved organic matter during the O3-based advanced oxidation of mature landfill leachate with and without biological pre-treatment and operating cost analysis. Chemosphere 271:129810
27. Gu Z, Feng K, Li Y, Li Q (2022) Microbial characteristics of the leachate contaminated soil of an informal landfill site. Chemosphere 287:132155
28. Han M, Duan X, Cao G, Zhu S, Ho SH (2020) Graphitic nitride-catalyzed advanced oxidation processes (AOPs) for landfill leachate treatment: a mini review. Process Saf Environ Prot 139:230–240
29. Hasnine MT, Anand N, Zoungrana A, Palani SG, Yuan Q (2022) An overview of physicochemical and biological treatment of landfill leachate. Circular economy in municipal solid waste landfilling: Biomining and leachate treatment: sustainable solid waste management: waste to wealth, pp 115–152
30. Ishaq A, Said MIM, Azman SB, Abdulwahab MF, Jagun ZT (2023) Optimizing total ammonia–nitrogen concentration for enhanced microbial fuel cell performance in landfill leachate treatment: a bibliometric analysis and future directions. Environ Sci Pollut Res 1–22
31. Kamal A, Makhatova A, Yergali B, Baidullayeva A, Satayeva A, Kim J, Arkhangelsky E (2022) Biological treatment, advanced oxidation and membrane separation for landfill leachate treatment: a review. Sustainability 14(21):14427

32. Kara N, Sari Erkan H, Onkal Engin G (2023) Characterization and removal of microplastics in landfill leachate treatment plants in Istanbul Turkey. Analyt Lett 56(9):1535–1548
33. Keyikoglu R, Karatas O, Rezania H, Kobya M, Vatanpour V, Khataee A (2021) A review on treatment of membrane concentrates generated from landfill leachate treatment processes. Separation and Purification Technol 259:118182
34. Kunwar P, Ahn J, Baek Y, Yoon J (2020) Technical and economical assessment of adsorption and reverse osmosis for removal of ammonia from groundwater of Kathmandu Nepal. Appropr Technol 6(2):174–182
35. Li S, Yang Y, Zheng H, Zheng Y, Jing T, Ma J, Nan J, Leong YK, Chang JS (2022) Advanced oxidation process based on hydroxyl and sulfate radicals to degrade refractory organic pollutants in landfill leachate. Chemosphere 297:134214
36. Lin J, Chen Q, Liu R, Ye W, Luis P, Van der Bruggen B, Zhao S (2021) Sustainable management of landfill leachate concentrate via nanofiltration enhanced by one-step rapid assembly of metal-organic coordination complexes. Water Res 204:117633
37. Lindamulla L, Nanayakkara N, Othman M, Jinadasa S, Herath G, Jegatheesan V (2022) Municipal solid waste landfill leachate characteristics and their treatment options in tropical countries. Current Pollut Rep 8(3):273–287
38. Mahtab MS, Islam DT, Farooqi IH (2021) Optimization of the process variables for landfill leachate treatment using Fenton based advanced oxidation technique. Eng Sci Technol Int J 24(2):428–435
39. Miao L, Yang G, Tao T, Peng Y (2019) Recent advances in nitrogen removal from landfill leachate using biological treatments–A review. J Environ Manage 235:178–185
40. Mojiri A, Zhou JL, Ratnaweera H et al (2020) Treatment of landfill leachate with different techniques: an overview. J Water Reuse and Desalinat 11(1):66–96. https://doi.org/10.2166/wrd.2020.079
41. Mu S, Chen X, Song B, Wu C, Li Q (2022) Enhanced performance and mechanism of the combined process of ozonation and a semiaerobic aged refuse biofilter for mature landfill leachate treatment. Chemosphere 308:136432
42. Muksy R, Kolo K (2023) Experimental observations and assessment of landfill leachate bioremediation by autochthonous fungi species and their effective geoactivities. J Ecol Eng 24(1)
43. Nath A, Debnath A (2022) A short review on landfill leachate treatment technologies. Mater Today: Proc
44. Ndao A, Drogui P, Tyagi RD (2022) Enhancement of landfill leachate treatment using extracellular polymeric substances as bio-flocculants. J Environ Sci Health B 57(1):62–70
45. Nivya TK, Pieus TM (2016) Comparison of photo ElectroFenton process (PEF) and combination of PEF process and membrane bioreactor in the treatment of landfill Leachate. Procedia Technol 24:224–231. https://doi.org/10.1016/j.protcy.2016.05.030
46. Parvin F, Tareq SM (2021) Impact of landfill leachate contamination on surface and groundwater of Bangladesh: a systematic review and possible public health risks assessment. Appl Water Sci 11(6):100
47. Pisharody L, Gopinath A, Malhotra M, Nidheesh PV, Kumar MS (2022) Occurrence of organic micropollutants in municipal landfill leachate and its effective treatment by advanced oxidation processes. Chemosphere 287:132216

48. Ramalho M, Jovanović T, Afonso A, Baía A, Lopes A, Fernandes A, Almeida A, Carvalho F (2023) Landfill leachate treatment by immediate one-step lime precipitation, carbonation, and phytoremediation fine-tuning. Environ Sci Pollut Res 30(4):8647–8656
49. Ranjan R, Sharma A, Verma MK (2021) Characterization of the second wave of COVID-19 in India. Curr Sci 121. https://doi.org/10.18520/cs/v121/i1/85-93
50. Ranjana P, Varsha S, Eliyas S (2021) IoT based smart garbage collection using RFID andsensors. In: Journal of Physics: conference series, vol 1818, no 1, p 012225. IOP Publishing
51. Ren X, Liu D, Chen W, Jiang G, Wu Z, Song K (2018) Investigation of the characteristics of concentrated leachate from six municipal solid waste incineration power plants in China. RSC Adv 8(24):13159–13166
52. Renou S, Givaudan JG, Poulain S, Dirassouyan F, Moulin P (2008) Landfill leachate treatment: review and opportunity. J Hazard Mater 150(3):468–493. https://doi.org/10.1016/j.jhazmat.2007.09.077
53. Ruoso AC, Dalla Nora M, Siluk JCM, Ribeiro JLD (2022) The impact of landfill operation factors on improving biogas generation in Brazil. Renew Sustain Energy Rev 154:111868
54. Shadi AMH, Kamaruddin MA, Niza NM, Emmanuel MI, Ismail N, Hossain S (2021) Effective removal of organic and inorganic pollutants from stabilized sanitary landfill leachate using a combined Fe_2O_3 nanoparticles/electroflotation process. J Water Proc Eng 40:101988
55. Siddiqi SA, Al-Mamun A, Baawain MS, Sana A (2022) A critical review of the recently developed laboratory-scale municipal solid waste landfill leachate treatment technologies. Sustainable Energy Technol Assess 52:102011
56. Singh S, Malyan SK, Maithani C, Kashyap S, Tyagi VK, Singh R, Malhotra S, Sharma M, Kumar A, Panday BK, Pandey RP (2023) Microplastics in landfill leachate: occurrence, health concerns, and removal strategies. J Environ Manage 342:118220
57. Soares R, Soares ACP, Maestá BC, Lippi M, Eduardo C, da Cunha SCP (2021) Evaluation of the efficiency of reverse osmosis in the treatment of sanitary landfill leachate in the metropolitan Region of the Rio de Janeiro
58. de Souza Pelinson N, Shinzato MPB, Morita AKM, Martins LGB, Wendland EC (2020) Innovative device to assay leachate production in non-sanitary landfills. J Mater Cycles Waste Manage 22:1985–1998
59. Tan B, He L, Dai Z, Sun R, Jiang S, Lu Z, Liang Y, Ren L, Sun S, Zhang Y, Li C (2022) Review on recent progress of bioremediation strategies in Landfill leachate-A green approach. J Water Proc Eng 50:103229
60. Teng C, Zhou K, Peng C, Chen W (2021) Characterization and treatment of landfill leachate: a review. Water Res 203:117525
61. Tian Z, Li G, Bai M, Hou X, Li X, Zhao C, Zhu Q, Du C, Li M, Liu W, Zhang L (2022) Microbial mechanisms of refractory organics degradation in old landfill leachate by a combined process of UASB-A/O-USSB. Sci Total Environ 848:157737
62. Viegas C, Nobre C, Mota A, Vilarinho C, Gouveia L, Gonçalves M (2021) A circular approach for landfill leachate treatment: Chemical precipitation with biomass ash followed by bioremediation through microalgae. J Environ Chem Eng 9(3):105187
63. Wijekoon P, Koliyabandara PA, Cooray AT, Lam SS, Athapattu BC, Vithanage M (2022) Progress and prospects in mitigation of landfill leachate pollution: Risk, pollution potential,

treatment and challenges. J Hazard Mater 421:126627
64. Wu C, Chen W, Gu Z, Li Q (2021) A review of the characteristics of Fenton and ozonation systems in landfill leachate treatment. Sci Total Environ 762:143131
65. Yalcuk A, Ugurlu A (2019) Treatment of landfill leachate with laboratory scale vertical flow constructed wetlands: plant growth modeling. Int J Phytorem 22(2):157–166. https://doi.org/10.1080/15226514.2019.1652562
66. Yavuzturk Gul B, Gulhan H, Soyel S, Kaya R, Ersahin ME, Ovez S, Koyuncu I (2023) Landfill leachate treatment with a full-scale membrane bioreactor: impact of leachate characteristics on filamentous bacteria. Environ Sci Pollut Res 1–13
67. Ye W, Hong M, Huang X, Chen T, Gu A, Lin X, Li X, Chen X, Seo DH, Zhao S, Chen X (2023) Towards effective recovery of humate as green fertilizer from landfill leachate concentrate by electro-neutral nanofiltration membrane. Sci Total Environ 896:165335
68. Zhang Z, Teng C, Zhou K, Peng C, Chen W (2020) Degradation characteristics of dissolved organic matter in nanofiltration concentrated landfill leachate during electrocatalytic oxidation. Chemosphere 255:127055

© The Author(s), under exclusive license to Springer Nature Switzerland AG 2024
S. Souabi, A. Anouzla (eds.), *Landfill Leachate Treatment Techniques*, Springer Water
https://doi.org/10.1007/978-3-031-63157-3_3

3. Treatment Techniques Adopted to Developing Countries

Yasmin Cherni[1] ✉, Sarra Hechmi[1], Mohamed Ali Wahab[1], Samira Melki[1] and Ismail Trabelsi[1]
(1) Laboratory of Wastewater Treatment and Valorization, Water Research and Technologies Center, Carthage University, CERTE, Technopark Tourist Route of Soliman Nabeul, B.P. 273, 8020 Soliman, Tunisia

✉ **Yasmin Cherni**
Email: yasmin.cherni@gmail.com

Abstract
Landfill leachate has several negative environmental effects and contaminates groundwater on a large scale. This leachate contains highly toxic contaminants such as heavy metals, solids, BOD, COD, ammonia, and TOC. Because of their versatility, pollutants cannot be treated using a single treatment technique. In fact, many treatment technologies are discussed, including physical and chemical processes, biological and advanced oxidation treatments. By providing an updated overview of the various alternatives that have been successfully applied in this field, this chapter highlights the benefits and

drawbacks of these technologies for landfill leachate treatment in developed countries. More research should be done to improve landfill leachate treatment in order to maximize removal performance.

Keywords Leachate – Treatment technologies – Advanced oxidation treatments – Combined processes

---

3.1 Introduction

Landfill leachate management involves controlling and treating the liquid generated during solid waste decomposition in a landfill [1]. Proper management of leachate, a mixture of water, organic and inorganic compounds, heavy metals, pathogens, and pollutants, is crucial to protect the environment and human health. To prevent soil and groundwater pollution, landfill leachate should be properly treated and disposed of, with a focus on minimizing generation and migration [2]. This requires a comprehensive understanding of the composition and characteristics of leachate, as well as appropriate technologies and treatment methods to remove the pollutants. Landfill leachate management is an important aspect of solid waste management and is critical for ensuring the protection of the environment and public health [3]. To effectively remove pollutants from leachate, it is important to have a comprehensive knowledge of its composition and characteristics, as well as the necessary technologies and treatment methods. Effective landfill leachate management is crucial for protecting the environment and public health while managing solid waste [4, 5]. In this frame, due to their dependability, simplicity, and high cost-effectiveness, biological treatments are typically preferred. However, due to their toxicity to microorganisms at high concentrations, some substances, like ammonia, are difficult to biodegrade [6]. Physical–

chemical processes provide attractive pretreatment options for lowering effluent toxicity, according to a number of studies. However, each type of landfill leachate treatment has benefits as well as disadvantages. Recent technological advancements have made it possible to treat LFL effectively using a combination of two or more physicochemical treatments or a combination of physicochemical and biological treatments [7]. This technology appears to be a practical and effective method for enhancing the efficacy of landfill leachate treatment. To sum up, the objective of the chapter is to summarize various physicochemical and biological technologies applied for leachate treatment in developed counteries, as well as to compare their capabilities and limitations. In fact, there are various physicochemical, biological and combined treatments in the literature that offer effective alternatives for dealing with high organic content in leachate.

3.2 Characteristics of Landfill Leachate in Developing Countries

In general, conventional parameters such as chemical oxygen demand (COD), total organic carbon (TOC), biochemical oxygen demand (BOD), suspended solids, pH, ammonia (NH_4^+-N), and trace metal concentrations characterize landfill leachate. BOD_5/COD and COD/TOC ratios are common indicators of organic compound biodegradability and the oxidised state of organic carbon. Various factors influence landfill leachate quality, including waste type, operational conditions, climate, hydrogeology, and landfill age [8].

The characteristics of landfill leachate (LFL) are critical in determining which methods and technologies to use before discharging to the environment. The primary

concern with LFL is that it contains a variety of pollutants at varying concentrations throughout the landfilling period. It is worth noting that the composition of disposed waste and the age of the landfill influence the composition and concentration of contaminants in leachate. As a result, landfll is generally classified as young for the first 5 years, medium for the next 5–10 years, and mature for the last 10 years [5]**.** Table 3.1 provides general information on the classification of landfill leachate.

Table 3.1 Classification of leachate from landfill

Parameter	**Young (< 5 years)**	**Intermediate (5-10 years)**	**Stabilised (> 10 years)**
PH	< 6.5	6.5–7.5	> 7.5
COD (mg/L)	> 10.000	4000–10.000	< 4000
BOD_5/COD	0.5–1.0	0.1–0.5	< 0.1
TOC/COD	< 0.3	0.3–0.5	> 0.5
Kjeldahl nitrogen (g/L)	0.1–0.2	NA*	NA*
Trace metals (mg/L)	Low to medium	Low	Low
Biodegradability	Important	Medium	Low

*NA is not available

Typically, contaminants found in LFL are classified as follows: dissolved organic matter (COD), trace metals, xenobiotic organic compounds such as aromatic hydrocarbons, phenols, and chlorinated aliphatic, persistent organic pollutants (POPs), and nitrogen (NH_3-N). The BOD/COD ratio, which represents the biodegradability of LFL, is commonly applied as an indicator for LFL age. It indicates the presence of recalcitrant or bio-refractory compounds in mature landfill leachate, on the other hand. This ratio decreases as time passes and the age of landfllling increases. As a result, the

high ratio is associated with young LFL, which contains a high concentration of biodegradable organic matter, such as a volatile fatty acid (VFA). The old LFL, on the other hand, has a low BOD/COD ratio because the non-biodegradable portion of COD will not be decomposed during the landflling period. The BOD/COD ratios of young, medium, and old LFL are generally in the ranges of 0.5–1.0, 0.1–0.5, and less than 0.1, respectively. Furthermore, stabilised leachate contains a high concentration of (NH_3-N), heavy metals, and recalcitrant organic matter, which severely limits the use of biological processes to properly treat LFL.

3.3 Common Treatment Techniques

Over the last five decades, lab-scale studies of the treatment process for landfill leachate have been published. Typically, landfill leachate treatment methods are classified as, biological, physical–chemical, and chemical and membrane treatments (Fig. 3.1).

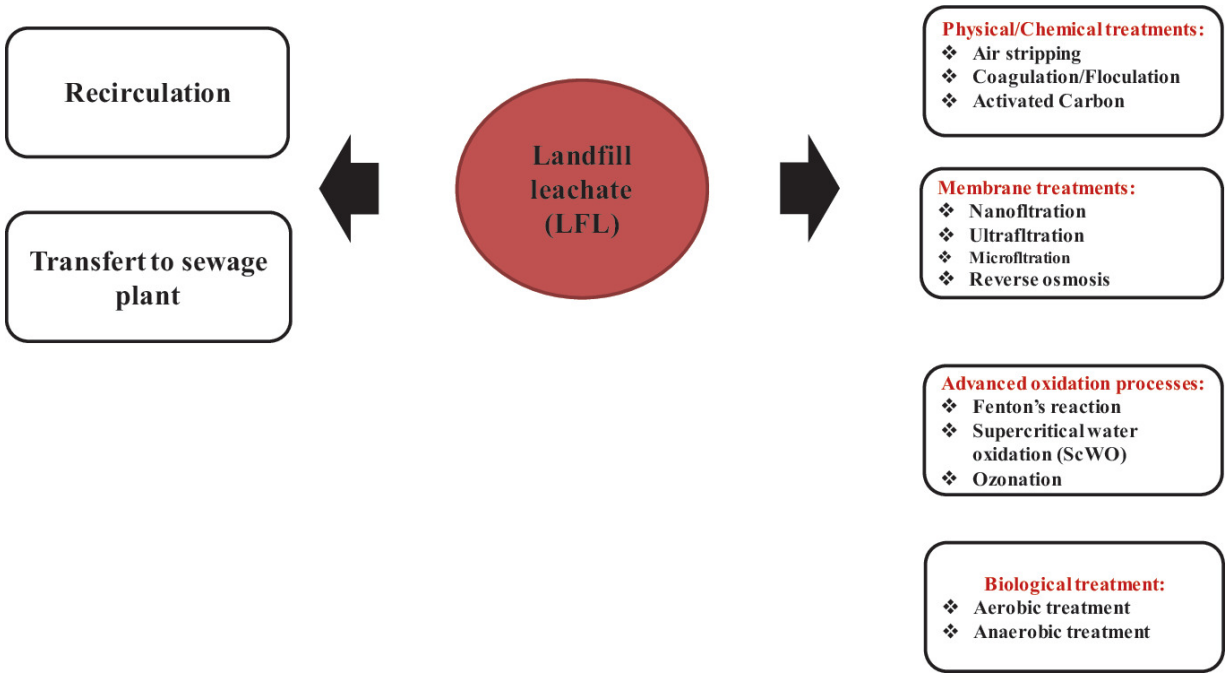


Fig. 3.1 Summary of the main technplogies applied for landfill leachate treatment in developed countries

A. Physical Treatment Techniques

Coagulation and Flocculation

The coagulation/flocculation process is a simple physical–chemical method for transferring contaminants from the liquid phase to the solid phase in the form of sludge. It has been widely used for landfill leachate pre-treatment and final polishing [9]. The coagulation/flocculation process effectively removes organic pollutants and trace metals while having no effect on NH_4^+-N. Coagulants like $Al_2(SO_4)_3$, $FeSO_4$, $FeCl_3$, and $Fe_2(SO_4)_3$ are widely used. Iron salts are generally more effective than aluminum salts because they are insoluble over a larger pH range [10].

Furthermore, coagulation has been shown to be more effective at removing high molecular weight hydrophobic organics than low molecular weight hydrophilic organics. Coagulation is typically followed by a focculation treatment, which increases particle size and converts unstable particles into bulky foccules for easier settlement. During coagulation-focculation, trace metal salts such as aluminum sulphate, ferrous sulphate, ferric chloride, and ferric chlorosulfate are added to the solution to produce high-valence cations and lower the zeta potential. In general, ferric ion salts outperform aluminum salts due to their insoluble nature across a wide pH range [11, 12]. Nonetheless, most research has focused on the use of coagulants such as aluminum sulphate, ferrous sulphate, ferric chloride, and polyaluminum chloride during the coagulation-focculation process. However, the use of these chemicals had a number of drawbacks, including negative effects on human health, high sludge production, and relatively high costs. Therefore, studies have focused on the use of composites or hybrid materials as another type

of coagulant that has demonstrated useful for wastewater treatment. In 2016, Nascimento et al. used natural chitosan as a coagulant for landfill leachate treatment. Using a chitosan dosage of 960 mg/L and a pH of 8.5 resulted in 80 and 91.4% color and turbidity removal rates, respectively. In the same vein, Nithya and Abirami reduced 85.2% of the turbidity from LFL using pine bark as a natural coagulant at a pH of 7 and a coagulant dosage of 4 g/mL. Righetto et al. recently investigated the efficacy of tannin-based materials as natural coagulants for treating landfill leachate under optimal conditions (pH: 7.3, concentration: 11.1 mL/L). 15% of the TOC, 4% of the TN, 47% of the TP, and 20% of the NH_3-N were extracted. Furthermore, the metals Fe, Ti, Cr, Al, Ba, and V were significantly reduced, while other metals were removed at a lower rate.

- **Air stripping**

Air stripping or ammonia stripping has drawn a great of interest as a practical alternative for lowering the NH_3 content of leachates [13]. The optimal pH for this physical process is 11. At this pH, all NH_3 forms transform into gases, which then react with acidic solutions to produce stable ammonium salts that can be used as mineral fertiliser. Several authors discovered that the effectiveness of ammonia removal was directly proportional to the applied superficial load, and that the removal of carbonaceous material was proportional to the amount of organic matter in the wastewater. The main disadvantage of this process is the inefficient degradation of organic matter and the release of NH_3 into the atmosphere, which can be harmful to the environment (Leite et al. 2018).

In 2021, Khoi et al. developed an ammonia-removing air stripper for LFL. The effects of various parameters, such as pH, hydraulic loading rate (HLR), gas/liquid ratio (G/L), and liquid recirculation, on ammonia stripping efficiency

were investigated. The findings showed that, regardless of G/L or HLR changes, increasing the pH from 9 to 12 significantly increased ammonia removal efficiency (up to 99.0% after 3 h). Increasing the G/L ratio increased removal efficiency at HLRs of 57.6 or 172.8 m^3/m^2/day, with the highest value (56%) obtained at an HLR of 172.8 m^3/m^2/day, pH 12, and a G/L of 728. The study demonstrated the effectiveness of air stripping as a pretreatment method for removing ammonia from landfill leachate.

- **Activated Carbon (AC)**

Adsorption is the most commonly used method for removing recalcitrant organic compounds from LFLs. Activated carbon, according to [14], is one of the most effective and practical adsorbents for removing various organic and inorganic contaminants, such as ammonium nitrogen and trace metals.

Granular activated carbon (GAC) and powdered activated carbon (PAC) are commonly used in leachate treatment due to their high area-to-volume ratio, large porous specific surface area, thermostability, and adsorption kinetics [15]. In 2019, Chávez et al. investigated the potential of AC made from coffee waste in the treatment of landfill leachate. They also reported that using 3 g/L of adsorbent, the adsorption process reduced organic matter in LFL by 97%. Activated carbon has a number of benefits,however, this adsorption method has high activated carbon consumption and regeneration cost, which leads to higher treatment costs and limits its application [16, 17] found that an activated carbon column was effective in treating LFL. Up to 74% of the COD, 47% of the BOD5, 90% of the ammonia, and 93% of the color were reduced, while the BOD_5/COD ratio rose from 0.3 to 0.9. Additionally, the AC adsorption procedure removed up to

60% of the trace metal content. Because of its potential for agroresidue disposal and environmental applications, the production of activated carbon (AC) from agricultural by-products has received a lot of attention in the research community. For the treatment of landfill leachate, other natural materials such as zeolite and clays are effective adsorbents comparable to AC [18]. Due to their high adsorption and ion exchange capacities and low price, natural minerals like zeolites have been suggested for use as adsorbents. Zeolites are composed of crystalline hydrated aluminosilicates and contain water-filled pores throughout their structure. The transferable cations (Ca^{2+}, K^{+}, and Mg^{2+}) in the physical structure of zeolite have been discovered to be easily replaceable. Clay minerals have a lot of potential because of their large family of adsorbents, high specific surface area, chemical and mechanical stability, and ionic exchange capacity. They are also relatively inexpensive and environmentally friendly [19].

B. Biological Treatment Techniques

- **Aerobic Treatment**

The most commonly used biological procedure is aerobic treatment. Aerobic reactors have continuous aeration and well-established bacterial populations. To remove organic materials from wastewater, the activated sludge process requires a high concentration of microorganisms, primarily bacteria, fungi, and protozoa. The activated sludge technique effectively removes biodegradable organic material by converting it entirely into carbon dioxide and water. SBRs are the most commonly used method of treating landfill leachate. An SBR has a number of time-oriented periodic steps and runs in batches, which may improve process efficiency [20, 21]. In this frame, The Hashemi et al. study demonstrated the efficacy of an SBR in leachate treatment. The average removal efficiencies for

COD, total nitrogen, phosphorus, and BOD_5 were 96%, 92.45%, 73.6%, and 96%, respectively. Numerous studies have generally demonstrated that when it comes to the breakdown of organic matter and other pollutants in landfill leachate, fungi and algae perform better than bacteria. According to Martínez-Ruiz et al., leachates can be treated by employing bioremediation in conjunction with microalgae cultivation to eliminate different nutrients and contaminants. This approach is simple to use, low maintenance, and requires little financial investment. One of the main purposes of using microalgae is to remove heavy metals and other environmental contaminants by growing them and producing biomass. The large volume of sludge produced and the expensive initial costs of the aeration equipment are the process's main drawbacks. Though, to increase the effectiveness of the primary treatment process, aerobic processes can actually be used as pretreatment or posttreatment steps. Typically, it is challenging to achieve the necessary removal efficiency of the different pollutants.

- **Anaerobic treatment**

Anaerobic treatment includes the biological degradation of organic or inorganic waste without the presence of oxygen molecules. Anaerobic techniques can be used to biotechnologically treat concentrated organic wastewater. These methods produce little sludge and biogas, making them environmentally benign and energy-efficient. Anaerobic methods frequently perform better at treating landfill leachates than aerobic methods because of the high COD and high BOD/COD ratio of these leachates. According to littérature, toxic pollutants like total ammoniacal nitrogen (TAN), sulphides, and trace metals, on the other hand, can inhibit these bacteria [22].

Ammonia is the most common pollutant in landfill leachate, according to the literature. A novel biological denitrification method identified as anammox, or anaerobic ammonium oxidation, has a high denitrification capacity and uses little energy [23]. Zhang et al. [23] recently investigated to the extent to which the anammox procedure worked for treating LFL. With an effluent quality of 10.3 mg/L TN, 86.2% nitrogen removal efficiency was achieved. Furthermore, the mass balance showed that 67.2% of the anammox nitrate could be recycled in situ as nitrite. The results of microbiological analyses showed that the nitrate reductase/nitrite reductase ratio and the genes of the anammox bacteria were highly detected.

Membrane filtration Techniques

Membrane separation technology is widely used in wastewater treatment, including landfill leachate. A membrane is a thin film or barrier that allows selective permeability. Membrane separation is the selective filtration of influent through different-sized pores. Depending on the pore size of membrane materials, the leading membrane processes can be classified into four groups: microfiltration MF, ultrafiltration UF, nanofiltration NF and reverse osmosis RO [24]. Membrane filtration is an efficient and cost-effective method for separating components from wastewater. It uses less energy than other separation technologies like distillation or evaporation because it uses pressure differences to push the feed solution through the membrane instead of heat. Membrane filtration technology's energy efficiency depends on factors such as membrane type, feed solution concentration, and fouling [25].

Microfiltration

MF is classified as a low-pressure, cross-flow membrane process. The membrane's large pore size limits its ability to

remove suspended solids and particles. Dissolved organic and inorganic impurities were not effectively treated. In 2018, Pertile and co-workers demonstrated the performance of the application of MF for LFL treatment. As result, they proved the importance of membrane processes in the reduction of 43% of organic matter and 63% of BOD_5. As a single process, no significant retention rate was obtained. MF is commonly used as a pre-treatment for other membrane processes like UF, NF, or RO, or to aid chemical treatments. MF has been recognized as an effective pre-treatment method for RO processes. This is due to its ability to retain microorganisms and suspended solids, which prevents fouling of RO membranes [26].

Reverse osmosis

RO is a highly effective method for treating landfill leachate. RO can effectively purify wastewater by concentrating low-MW dissolved and suspended solids. RO is capable of operating at high fluxes and across diverse temperature and pH ranges, with a rejection rate of 98–99% for both organic and inorganic contaminants. RO can effectively remove heavy metals, suspended materials, and dissolved solids from landfill leachate [27]. In 2022, Topal et al. demonstrated the efficiency of RO in leachate treatment with high removal rates such as 98% of COD, 99% of total nitrogen and suspended solids, and significant heavy metal removal. Typically, the RO process has demonstrated high rejection rates for both organic and inorganic contaminants in leachate. The main disadvantage of this technology is membrane fouling, which reduces treatment performance while increasing production, both of which are difficult to manage.

Ultrafiltration

UF is a pressure-driven membrane separation technique that uses MF and NF. UF is commonly used for sieving,

concentration, and fractionation. UF has a wider separation range than MF and can remove macromolecular substances, pathogens, particles, and colloids from landfill leachate. The efficiency of removal varies depending on the membrane material and pore size range [28]. Kulikowska et al. 2019 studied the feasibility of using UF as treatment of LFL. They discovered that UF is extremely effective in removing COD and dissolved organic carbon (removal rates of 46.5%, and 22.1%, respectively). Leachate treatment, like MF membranes, is commonly used as a pre-treatment for NF or RO. According to Anqi et al., UF can be used before RO or after biological treatment. UF is effective in removing larger MW components from leachate, which can foul NF and RO membranes. Full-scale plants have successfully used UF membranes with membrane bioreactors (MBRs) for pretreatment before NF or RO processes.

Nanofilltration

NF eliminates ions that contribute to osmotic pressure, requiring lower operating pressures than RO. NF can remove heavy metals and refractory organic substances from landfill leachate. NF uses a pressure of 5–15 bar to separate monovalent and bivalent ions in water. NF eliminates ions that contribute to osmotic pressure, requiring lower operating pressures than RO. NF can remove heavy metals and refractory organic substances from landfill leachate. NF uses a pressure of 5–15 bars to separate monovalent and bivalent ions in water [29]. According to Istirokhatuna et al., using an NF process, COD, TSS, and TDS were reduced by 96%, 100%, and 62%, respectively. When used as post-treatment, the NF process provides a versatile approach to meeting a variety of water quality objectives, including organic, inorganic, and microbial contamination control. Combining NF with

different processes can result in more complex but cost-effective processes.

D. Advanced Treatment Techniques

AOPs transform organic pollutants into carbon dioxide, water, and inorganic ions through oxidation reactions. AOPs are divided into two stages. The initial stage involves the oxidation process. At this point, strong oxidants or hydroxyl radicals are created. At the second stage, these oxidants interact with the organic pollutants in the effluent. The five types of AOPs are as follows: (1) heterogeneous photocatalysis; (2) H_2O_2-based processes; (3) electrochemical oxidation; (4) sonochemical oxidation and (5) ozone-based processes. These categories are based on the source of hydroxyl radicals [30].

Ozonation

Ozonation is an advanced oxidation technique that involves injecting ozone into water. Ozone (O_3) is a very powerful oxidant. Ozonation is an effective LFL treatment because it generates highly reactive oxygen species capable of degrading a wide range of organic and inorganic pollutants, as well as all microorganisms. Ozone is created by exposing oxygen (O_2) to energy (high voltage or UV radiation) [28]. In 2020, Hofmann et al. demonstrated the efficacy of ozonation in the treatment of raw LFL. The experiments were completed using a batch system. The pH was initially set to 7 and 10, and the contact times between gas and leachate were 20, 40, 60, 80, and 100 min. When the pH reached 7, ozonation demonstrated high removal efficiencies for COD, color, and UV abs. Turbidity degradation was more pronounced at pH 10. The maximum color removal (90%) and UV absorption (70%) occurred after 40 min of reaction time. Because of its high capacity

for organic compound removal, ozonation can be used as a pretreatment option for LFL.

TiO_2 photocatalysis

Due to their small size, high surface area, crystal form, and high reactivity, nanomaterials have emerged as a promising alternative for the purification and treatment of pollutants in order to convert them to non hazardous materials. Indeed, over the last few decades, nanotechnology has gained widespread attention. Various nanomaterials based on adsorption, photocatalytic, and antibacterial activities have been developed for treating and decontaminating water [31].

Although TiO_2 is a popular semiconductor photocatalyst, it may not be the only viable option. UV light is required to activate the photo-oxidation process. Solar energy is increasingly being used as a photon source with low-cost catalysts. However, pure TiO_2 has lower photocatalytic efficiency than UV sources. TiO_2 catalysis involves two types of reactions: adsorption–desorption equilibrium in dark conditions and photocatalysis during irradiation. Various researchers have applied TiO_2 for LFL treatment or pretreatment to improve the biodegradability of refractory organic compounds in leachate, as well as for LFL post-treatment to ensure effluent discharge standards are met [32].

Wang et al. [17] studied the effect of several factors such as the ultraviolet irradiation time, amount of the catalyst, and pH on the removal efficiency of several pollutants in the leachate treatment. Thus, the optimal process conditions were the ultraviolet irradiation time (500 min), amount of the catalyst (4 g/L), and pH (8.8). Under these conditions, the overall average removal rates for NH_4+–N and COD in the leachate were 98.8% and

32.5%, respectively, and the (NO_3-–N) and (NO_2–N) concentrations were maintained at 7–9 and 0.01–0.017 mg/L, respectively. Yet, TiO_2 photocatalysis has one significant disadvantage. UVA irradiation is the only way to activate catalysts. Only 3–5% of the solar radiation that reaches Earth's surface is UVA, so expanding TiO_2 absorbance into the visible region is critical. Several studies have shown that doping TiO_2 with metal ions is one of the most effective methods of improving TiO_2's photoelectrochemical properties when exposed to UV and sunlight. More recently [33] investigated the efficiency of using 0.1 mol% Ag/Fe co-doped TiO_2 nanoparticles in the photocatalytic treatment of leachate. The response surfacemethodology was applied to obtain the optimum conditions. Herein, three significant factors like photocatalyst concentration, pH, and reaction time were studied with respectto the reduction rates removal efficiency of severalpollutants. Theremoval efficiency for COD, PO_4^{3-}, NH_4^+, and NO_3^- were 57, 18.34, 8.76, and 21.08%, respectively.

Fenton's process

The Fenton process is widely used for leachate pretreatment, post-treatment, or full treatment due to its ability to oxidize refractory organic molecules, low cost, and ease of use [34]. This method uses a combination of ferrous ions (Fe^{2+}) and hydrogen peroxide (H_2O_2). The majority of studies on the removal of contaminants by Fenton reaction processes have shown that the parameters that most strongly influence the Fenton process are the ferrous ion and peroxide concentrations, the initial amount of the pollutant, the reaction time, and the pH and temperature. El Mrabet et al. investigated the suitabilityof Fenton process for the removal of toxic compounds from

LFL. As a result, the rates of COD, BOD_5 and color content were 73%, 99.5% and 94%, respectively. Several authors have argued that the Fenton process has advantages over other AOPs, such as high performance, ease of implementation, and no energy requirement for H_2O_2 activation [34]. The high amount of iron sludge produced at the end of the experiment, the risk of storing hydrogen peroxide, and the need to adjust the pH to the acidic range are major drawbacks of this method.

E. Combined treatments

Leachate treatment involves large fluctuations in volumetric flow and chemical composition due to its complex composition [35]. However, it has been demonstrated that combining different treatment methods yields better results, and this strategy has become a real possibility for the treatment of landfill leachate [4]. Combining treatments can maximize the advantages of each individual process used in the treatment, and for mature landfill leachate treatment, they have been shown to be more efficient and economical. A combination of biological treatment and coagulation pretreatment has already been used in many landfill leachate sites [35]. Most of the time, coagulation has been found to be more suitable not only for removing organic compounds and color from LFL, but also as a refining step prior to AOP [36]. A coagulation–ozonation method was developed to handle with the concentrate that remained after landfill leachate underwent membrane treatment, and [24] investigated the combined process to determine the effects of coagulant type and initial pH on the efficacy of treatment. The outcomes demonstrated how well the coagulation method removed a variety of organics. Subsequently, the organic materials resistant to coagulation were treated further using ozone. This remarkable outcome showed that the

combined coagulation-ozonation method was a viable means of removing the refractory organic materials found in landfill leachate concentrate, thereby enabling additional biological treatment. In other hand, the use of AOP procedures as a pre-treatment results in the production of easily biodegradable biological post-treatment intermediate pollutants. It therefore has a positive impact on wastewater treatment, including landfill leachate [25]. In order to mitigate the acute cytogenotoxicity associated with municipal landfills, Klauck et al. [37] examined the efficacy of a range of sophisticated oxidation methods and biological systems. Electrochemical oxidation in conjunction with ozonation treatment resulted in an effluent that met Brazilian legal requirements for physicochemical properties and was suitable for disposal in water bodies. The genotoxicity was eliminated by electrochemical oxidation in conjunction with ozonation and photo-electrochemical oxidation, despite the fact that none of the treatments could totally lessen the acute toxicity. The advanced oxidation technique employing heterogeneous photocatalysis was coupled with seeded bioreactors containing different kinds of inoculum for the treatment of landfill leachate. Heterogeneous photocatalysis yielded reductions of 50–84% of the initial COD at pH 5. This treated leachate, however, cannot be reused or distributed back into the environment without further processing. More recently, Elleuch et al. [38] investigated for a practical, low-cost method of removing toxic compounds from (LFL) by combining a biological pre-treatment with kefir grains with an Ag-doped TiO_2 photocatalytic process that removes various pollutants from LFL at the same time.

Consequently, TOC, COD, NH_4^+ −N, and PO_4^{3-} had overall removal efficiencies of 98, 96, 85, and 93%, respectively. In this regard [] demonstrated the

performance application of a combined process comprising a biological process employing *Candida tropicalis* for LFL treatment and a TiO_2/Ag photocatalytic pre-treatment. The removal rates of Pb and Cd were 95%, while the abatement rates of COD, NH_4^+ −N, TOC, Fe, Zn, and Cu were 90, 75, 84.61, 50, 63.8, and 83%, respectively. To sum up, the combination process is an excellent alternative method for leachate treatment.

3.4 Challenges and Limitations in Landfill Leachate Treatment

Implementing effective landfill leachate treatment techniques in developing countries presents a unique set of challenges that stem from a combination of socio-economic, environmental, and infrastructural factors. These challenges often intersect, compounding the difficulties in establishing comprehensive treatment systems. Understanding and addressing these hurdles is essential for devising sustainable and impactful solutions. Developing countries frequently struggle with inadequate waste management infrastructure, hindering the proper collection, containment, and transport of leachate. The absence of well-designed landfill cells and leachate collection systems exacerbates the risk of environmental contamination and groundwater pollution. Scarce financial resources pose a significant barrier to adopting advanced treatment technologies. The costs associated with setting up, operating, and maintaining treatment facilities, as well as training personnel, can be substantial. These financial limitations often impede progress in upgrading or implementing effective treatment methods. A shortage of qualified professionals with expertise in leachate treatment technologies can hinder the successful operation of treatment systems. The intricate nature of these techniques

requires skilled operators who understand the complexities of the processes and can troubleshoot issues as they arise. Cutting-edge treatment technologies might be inaccessible due to factors such as import restrictions, high costs, and a lack of local manufacturing capabilities. This limitation narrows the range of available options, making it challenging to adopt state-of-the-art solutions. Treatment methods that work well in developed countries might not be directly transferable to developing countries due to differences in waste composition, climate, and regulatory frameworks. Customizing treatment approaches to suit local conditions while maintaining effectiveness requires careful consideration. Ensuring the long-term operational sustainability of treatment facilities can be challenging in resource-constrained environments. Maintenance costs, availability of spare parts, and consistent energy supply can impact the reliability of treatment systems over time. Developing countries may face regulatory gaps or inconsistent enforcement of environmental laws. This can hinder the establishment of standardized treatment practices and create uncertainties for investors and stakeholders. he lack of awareness about the environmental and health implications of untreated leachate can undermine public support for implementing effective treatment methods. Engaging communities and stakeholders is crucial for garnering support and cooperation. Landfill leachate treatment should be integrated into broader waste management strategies that encompass waste reduction, recycling, and sustainable disposal. A fragmented approach can hinder the effectiveness of treatment efforts. Addressing these challenges requires a multi-faceted approach that involves capacity building, policy reforms, technology adaptation, and collaborative efforts among governments, private entities, NGOs, and international organizations. By acknowledging these limitations and devising context-

specific strategies, developing countries can overcome hurdles and forge a path towards sustainable and effective landfill leachate treatment.

3.5 Emerging Trends and Innovations

3.5.1 Advancements in Sustainable Leachate Treatment Approache

Developing countries often struggle with centralized wastewater treatment due to infrastructure limitations. Emerging trends focus on decentralized treatment systems, such as small-scale or anaerobic reactors, which can be implemented locally and require less infrastructure investment. Innovations are being explored to incorporate environmentally friendly and sustainable methods. This might involve using natural materials for adsorption, harnessing solar energy for treatment processes, or designing systems that mimic ecological processes for treatment. Phytoremediation involves using plants to absorb, accumulate, and detoxify contaminants from leachate. Research is ongoing to identify plant species that are efficient in removing specific pollutants from leachate, offering a cost-effective and low-energy treatment option. Many landfill leachates contain valuable nutrients like nitrogen and phosphorus. Emerging technologies aim to recover these nutrients from leachate to be repurposed as fertilizers or for other applications. This helps mitigate environmental impacts while creating a resource from waste. Electrochemical methods, such as electrocoagulation and electrooxidation, are gaining attention. These techniques use electricity to generate reactions that facilitate the removal of contaminants, and they can be adaptable to different scales and conditions. Enhancing the performance of biological treatment processes through microbial engineering and optimization

is an emerging focus. This includes selecting and modifying microorganisms to better adapt to the challenging conditions of leachate treatment. Advances in sensor technology and data analytics enable real-time monitoring of treatment processes. Smart systems can adjust operational parameters based on changing leachate quality and quantity, leading to more efficient and responsive treatment. Nanomaterials have the potential to improve adsorption, filtration, and oxidation processes in leachate treatment. Research is ongoing to understand the effectiveness and potential risks associated with using nanomaterials.

3.5.2 Collaborative Strategies for Effective Leachate Management

Collaborative Strategies for Effective Leachate Management encompass a dynamic interplay of Public–Private Partnerships and Policy and Regulation Innovations. Collaborations between governments, private companies, and NGOs are increasingly gaining traction, driven by the goal of leveraging expertise, funding, and technology. These partnerships stand as formidable solutions, adeptly addressing the financial and technical barriers that often hinder the implementation of advanced treatment techniques. By pooling resources and knowledge, these collaborations forge a pathway toward sustainable and impactful leachate management. Simultaneously, forward-thinking Policy and Regulation Innovations are reshaping the landscape. In response to the pressing need for sustainable treatment methods, numerous developing countries are charting new courses. Regulatory incentives, such as tax breaks for companies pioneering eco-friendly technologies, serve as catalysts for positive change. Moreover, the introduction of stricter regulations acts as a driving force, spurring innovation in the domain of leachate

treatment. This dynamic interplay of incentives and regulations not only accelerates the adoption of sustainable practices but also sets the stage for a transformative shift in how landfill leachate is managed. In unison, these Collaborative Strategies and Policy and Regulation Innovations represent a holistic approach to surmounting challenges. They encapsulate the concerted efforts of diverse stakeholders, unite the private and public sectors, and harness the power of regulation to steer advancements. As we witness this dynamic synergy, it becomes evident that effective leachate management is no longer an isolated endeavor but a collective journey toward a cleaner, healthier, and more sustainable future.

3.6 Environmental and Health Benefits of Effective Landfill Leachate Treatment

Effective landfill leachate treatment yields a host of positive outcomes that significantly benefit both the environment and public health. Through the mitigation of untreated leachate's adverse consequences, these treatment processes contribute to a more sustainable and healthier living environment. By preventing contaminated leachate from infiltrating groundwater and surface water bodies, water quality is improved, safeguarding drinking water sources and aquatic ecosystems while reducing the risk of waterborne diseases and preserving biodiversity. The treatment also curbs soil contamination risks, a critical consideration for regions where agriculture or residential development could suffer from leachate-induced soil pollution. Furthermore, certain treatment methods, such as biological degradation, reduce the release of volatile organic compounds (VOCs) and noxious odors associated with untreated leachate, thereby enhancing air quality and

positively impacting community health. Treated leachate minimizes human exposure to hazardous substances, subsequently reducing the likelihood of respiratory illnesses, skin disorders, and other pollution-related health conditions. The recovery of valuable resources through certain treatment methods not only lessens the strain on natural resources but also aligns with sustainable practices. Beyond tangible benefits, effective leachate treatment reinforces environmental compliance, fosters positive economic impacts, and contributes to community well-being. By addressing risks associated with landfill leachate, these treatment measures ensure the long-term sustainability of both ecosystems and human settlements, in alignment with global environmental goals and the overall aspiration for a healthier, more resilient planet.

3.7 Strategies for Enhancing Landfill Leachate Treatment in Developing Countries

Addressing the challenges of landfill leachate treatment in developing countries requires a multifaceted approach involving policymakers, researchers, and practitioners. To improve efficiency and sustainability, consider the following recommendations.

To bolster the efficiency and effectiveness of landfill leachate treatment in developing countries, a collaborative effort among policymakers, researchers, and practitioners is crucial. To this end, a series of targeted strategies are recommended. Firstly, governments and international bodies should allocate funding to support research and innovation tailored to the unique challenges of these regions. This investment will propel the development of cost-efficient and contextually relevant treatment solutions. Additionally, building technical capacity through training

programs and workshops for operators and practitioners is essential to ensure optimal system performance. Facilitating technology transfer between more developed regions and developing countries will accelerate the implementation of proven treatment methods. Incorporating local conditions and cultural factors into technology design is pivotal for sustainability.

Promoting integrated waste management approaches that prioritize waste reduction and recycling will alleviate the burden on landfills and subsequently mitigate leachate production. Public awareness campaigns highlighting the environmental repercussions of untreated leachate can foster community participation in waste management efforts. Public–private partnerships should be nurtured to combine resources, expertise, and funding for effective leachate treatment implementation. Comprehensive waste management policies need to be established and enforced, encompassing proper landfill design, leachate collection systems, and treatment specifications. To encourage the adoption of sustainable technologies, governments can introduce financial incentives such as tax benefits and subsidies.

Implementing monitoring and reporting systems ensures accountability and informs future decisions. Creating platforms for knowledge exchange among stakeholders promotes the sharing of best practices and lessons learned. Long-term planning that considers evolving population growth and waste generation rates is essential for sustainable leachate management strategies. By embracing these multifaceted recommendations, developing countries can effectively overcome challenges, enhance the efficiency of landfill leachate treatment, and pave the way for a healthier and more sustainable future.

3.8 Conclusion

In the pursuit of addressing the complex challenges posed by landfill leachate treatment in developing countries, a comprehensive and collaborative approach is imperative. The convergence of innovative technologies, policy reforms, public engagement, and knowledge sharing constitutes the bedrock of a sustainable solution. As these nations grapple with limited resources, rapid urbanization, and environmental degradation, the urgency to implement effective treatment techniques is undeniable. Equally paramount is the continuous pursuit of research and the fostering of cross-disciplinary collaboration. This ongoing dedication to advancing our understanding and refining our strategies will play an instrumental role in developing tailored and contextually relevant solutions. The intertwining of public–private partnerships, regulatory innovations, capacity building, local adaptability, and the tenacity of the global research community unveils a roadmap towards successful leachate management. By embracing these multifaceted strategies, developing countries can usher in an era of cleaner environments, improved public health, and responsible waste management practices. In forging ahead, the commitment to bridging gaps, nurturing partnerships, harnessing collective expertise, and fostering continuous research will undoubtedly pave the way for a more resilient, sustainable, and healthier global future.

References

1. Anand N, Palani SG (2022) A comprehensive investigation of toxicity and pollution potential of municipal solid waste landfill leachate. Sci Total Environ 838:155891
2. Yu D, Pei Y, Ji Z, He X, Yao Z (2022) A review on the landfill leachate treatment technologies and application prospects of three-dimensional electrode technology. Chemosphere 291:132895

3. Díaz AI, Laca A, Díaz M (2022) Approach to a fungal treatment of a biologically treated landfill leachate. J Environ Manage 322:116085
4. Guo Z, Zhang Y, Jia H, Guo J, Meng X, Wang J (2022) Electrochemical methods for landfill leachate treatment: A review on electrocoagulation and electrooxidation. Sci Total Environ 806:150529
5. Siddiqi SA, Al-Mamun A, Baawain MS, Sana A (2022) A critical review of the recently developed laboratory-scale municipal solid waste landfill leachate treatment technologies. Sustainable Energy Technol Assess 52:102011
6. Nath A, Debnath A (2022). A short review on landfill leachate treatment technologies. Materials today: proceedings.
7. Lindamulla L, Nanayakkara N, Othman M, Jinadasa S, Herath G, Jegatheesan V (2022) Municipal solid waste landfill leachate characteristics and their treatment options in tropical countries. Current Pollut Rep 8(3):273–287
8. Cherni Y, Elleuch L, Messaoud M, Kasmi M, Chatti A, Trabelsi I (2021) Recent technologies for leachate treatment: a review. Euro-Mediterranean J Environ Integrat 6:1–20
9. Bellouk H, El Mrabet I, Tanji K, Nawdali M, Benzina M, Eloussaief M, Zaitan H (2022) Performance of coagulation-flocculation followed by ultra-violet/ultrasound activated persulfate/hydrogen peroxide for landfill leachate treatment. Scientific African 17:e01312
10. De S, Hazra T, Dutta A (2022). Application of integrated sequence of air stripping, coagulation flocculation, electrocoagulation and adsorption for sustainable treatment of municipal landfill leachate. Cleaner Waste Systems 3:100033
11. Liu D, Yuan Y, Wei Y, Zhang H, Si Y, Zhang F (2022) Removal of refractory organics and heavy metals in landfill leachate concentrate by peroxi-coagulation process. J Environ Sci 116:43–51
12. Liu Y, Lv Y, Cheng H, Zou L, Li YY, Liu J (2022) High-efficiency anaerobic co-digestion of food waste and mature leachate using expanded granular sludge blanket reactor. Biores Technol 362:127847
13. De Melo Henrique JM, Isidro J, Saez C, Dos Santos EV, Rodrigo MA (2022) Removal of lindane using electrokinetic soil flushing coupled with air stripping. J Appl Electrochem 52(9):1317–1326
14. Sultana M, Rownok MH, Sabrin M, Rahaman MH, Alam SN (2022) A review on experimental chemically modified activated carbon to enhance dye and heavy metals adsorption. Cleaner Eng Technol 6:100382

15. Kumari M, Chaudhary GR, Chaudhary S, Umar A (2022) Transformation of solid plastic waste to activated carbon fibres for wastewater treatment. Chemosphere 294:133692
16. Rohers F, Dalsasso RL, Nadaleti WC, Matias MS, de Castilhos Júnior AB (2021) Physical–chemical pre-treatment of sanitary landfill raw leachate by direct ascending filtration. Chemosphere 285:131362
17. Wang H, Xu J, Liu X, Sheng L (2021) Preparation of straw activated carbon and its application in wastewater treatment: a review. J Clean Prod 283:124671
18. Augusto PA, Castelo-Grande T, Merchan L, Estevez AM, Quintero X, Barbosa D (2019) Landfill leachate treatment by sorption in magnetic particles: preliminary study. Sci Total Environ 648:636–668
19. Rahman ROA, El-Kamash AM, Hung YT (2022) Applications of nano-zeolite in wastewater treatment: An overview. Water 14(2):137
20. Ke X, Liu C, Tang SQ, Guo TT, Pan L, Xue YP, Zheng YG (2022) Characterization of acinetobacter indicus ZJB20129 for heterotrophic nitrification and aerobic denitrification isolated from an urban sewage treatment plant. Biores Technol 347:126423
21. Raychaudhuri A, Sahoo RN, Behera M (2022) Sequential anaerobic–aerobic treatment of rice mill wastewater and simultaneous power generation in microbial fuel cell. Environ Technology 1–7
22. Cai Y, Janke L, Meng X, Zheng Z, Zhao X, Pröter J, Schäfer F (2022) The absolute concentration and bioavailability of trace elements: two vital parameters affecting anaerobic digestion performance of chicken manure leachate. Biores Technol 350:126909
23. Zhang F, Peng Y, Liu Y, Zhao L (2021) Improving stability of mainstream Anammox in an innovative two-stage process for advanced nitrogen removal from mature landfill leachate. Biores Technol 340:125617
24. Chen W, Gu Z, Ran G, Li Q (2021) Application of membrane separation technology in the treatment of leachate in China: a review. Waste Manage 121:127–140
25. Mojiri A, Zhou JL, Ratnaweera H, Ohashi A, Ozaki N, Kindaichi T, Asakura H (2021) Treatment of landfill leachate with different techniques: an overview. Water Reuse 11(1):66–96
26. Yan Z, Zhu Z, Chang H, Fan G, Wang Q, Fu X, Liang H (2023) Integrated membrane electrochemical reactor-membrane distillation process for enhanced landfill leachate treatment. Water Res 230:119559

27. Gripa E, Daflon SDA, de Almeida R, da Fonseca FV, Campos, JC (2023) Landfill leachate treatment by high-pressure membranes and advanced oxidation techniques with a focus on ecotoxicity and by-products management: a review. Process Saf Environ Prot 173:747–764. https://doi.org/10.1016/j.psep.2023.03.074
28. Rathnayake WAPP, Herath GBB (2018) A review of leachate treatment techniques. In: The 9th international conference on sustainable built environment, vol 1, no 1, pp 97–106
29. Silva NCM, Moravia WG, Amaral MCS, Figueiredo KCS (2018) Evaluation of fouling mechanisms in nanofiltration as a polishing step of yeast MBR-treated landfill leachate. Environ Technol
30. Mahtab MS, Islam DT, Farooqi IH (2021) Optimization of the process variables for landfill leachate treatment using Fenton based advanced oxidation technique. Eng Sci Technol Int J 24(2):428–435
31. Pant M, Kharkwal H, Singh KP, Joshi HC (2017) Detection of rota virus with the help of nanomaterial based field effect transistor (BIO-FET). Biosens J 6(02)
32. Betancourt-Buitrago LA, Hernandez-Ramirez A, Colina-Marquez JA, Bustillo-Lecompte CF, Rehmann L, Machuca-Martinez F (2019) Recent developments in the photocatalytic treatment of cyanide wastewater: an approach to remediation and recovery of metals. Processes 7(4):225
33. Cherni Y, Messaoud M, Salem-Berrabah OB, Salhi R, Elleuch R, Kasmi M, Chatti A, Trabelsi I, Elleuch L (2023) A sustainable nanobioremediation approach for Tunisian landfill leachate using Ag/Fe co-doped TiO_2 nanoparticles combined with Saccharomyces cerevisiae. Euro-Mediterranean J Environ Integrat 1–16
34. Traid HD, Vera ML, Escalada G, López IE, Dwojak AN, Litter MI (2022) Application of a Fenton process after a biological nitrification treatment: a successful case for leachate treatment. Case Stud Chem mental Eng 5:100208
35. Reddy CV, Rao DS, Kalamdhad AS (2022) Combined treatment of high-strength fresh leachate from municipal solid waste landfill using coagulation-flocculation and fixed bed upflow anaerobic filter. J Water Proc Eng 46:102554
36. Cherni Y, Elleuch L, Kasmi M, Sghaier-Hammami B, Chourabi k, Chatti A, Trabelsi I, Kallel A (2022) Detoxification of leachate by coagulation treatment prior to fermentation and possible reuse in irrigation. CLEAN Soil Air Water 50(6), 2000395. https://doi.org/10.1002/clen.202000395

37. Klauck CR, Giacobbo A, Altenhofen CG, Silva LB, Meneguzzi A, Bernardes AM, Rodrigues MAS (2017) Toxicity elimination of landfill leachate by hybrid processing of advanced oxidation process and adsorption. Environ Technol Innov 8, 246–255. https://doi.org/10.1016/j.eti.2017.07.006
38. Elleuch L, Messaoud M, Djebali K, Attafi M, Cherni Y, Kasmi M, Elaoud A, Trabelsi I, Chatti A (2020) A new insight into highly contaminated landfill leachate treatment using Kefir grains pre-treatment combined with Ag-doped TiO_2 photocatalytic process. J Hazard Mater 382:121119. https://doi.org/10.1016/j.jhazmat.2019.121119

© The Author(s), under exclusive license to Springer Nature Switzerland AG 2024
S. Souabi, A. Anouzla (eds.), *Landfill Leachate Treatment Techniques*, Springer Water
https://doi.org/10.1007/978-3-031-63157-3_4

4. Physicochemical and Biological Processing of Municipal Landfill Leachates

Wafa Hassen[1] ✉, Bilel Hassen[2], Marwa El Ouaer[1], Solwen Kouki[1] and Abdennaceur Hassen[1] ✉

(1) Laboratory of Treatment and Wastewater Valorization, Center of Research and Water Technologies (CERTE), Tourist Road of Soliman, P.O. Box 273.8020, Soliman, Tunisia

(2) Laboratories of Microbiology and Pathology of Aquatic Organisms, National Institute of Navy Sciences and Technologies (INSTM), Street, March 2nd, 1934, 2025 Salammbô, Tunisia

✉ **Wafa Hassen**
Email: hassen.wafa@gmail.com

✉ **Abdennaceur Hassen (Corresponding author)**
Email: abdohas@gmail.com

Abstract
Landfill leachate (leachate in the trivial sense or percolate in physics) is the residual liquid that results from the percolation of water through a waste stockpile, of which a certain fraction may be soluble. The leachate comes from the interstitial water adhered to the waste matrix and from the water (usually rainwater) that percolates into the waste mass by gravity. The viscous, colored, and very turbid liquid mass collected will

constitute the landfill leachate. Its chemical and biological composition differs greatly depending on the waste deposited and changes closely with civilization and the lifestyle of a population. The considerable increase in the volumes of leachate produced at the scale of a landfill because of urban development, the increasing costs of waste management and treatment, and the effects and scarcity of fossil fuels have prompted research not only in leachate treatment but also in leachate valorization. In this chapter, we have briefly discussed the different strategies promoted and used in recent years to treat landfill leachate and the main techniques for its valorization as fertilizer to enrich agricultural soils, as a source of energy or biofuel for heating, electricity, and even as a nutrient matrix to cultivate specific organisms such as useful algae and bacteria.

Keywords Landfill leachate – Chemical and biological composition – Management – Treatment techniques – Valorization of these leachates

---

4.1 Introduction

Waste management on a global scale and, more particularly, in developing countries may appear at first sight as less problematic than most other commonly known environmental problems, such as water or air pollution. Thus, as for domestic waste, collection and treatment solutions may seem obvious and unproblematic: it would be enough to collect it, transport it with trucks to treatment sites by landfill, incineration, or in the best cases by composting [1]. In the same way, the incessant accumulation of this waste puts us in confrontation with the scale of the real problems and difficulties encountered in this field. These problems and difficulties are related to some significant observations, such as the exploitation of landfills with the enormous and increasing production of waste, leachates, sometimes inoperative treatments, inefficient compaction, and bogging down of the

machines [2, 3]. Additionally, great technical problems of functioning and feasibility of the composting installations can appear with sometimes the production of compost of bad quality and difficulties for its marketing and its use. The no-good combustion of some wastes put in incineration following a bad arrival of wastes or nonconform sorting and the generation of considerable masses of polluted and nauseating juices in the pits of the incineration plants. Finally, the low proportions of recyclable products and the difficult mobilization and sensitization of the populations for the acts of separation at the level of the projects of selective collections.

However, the major mass of waste generated in an urban population will be stored, stocked, and dumped in well-controlled sites. This infinite storage of urban waste generates a considerable and growing residual volume of leachate. This liquid product is an emerging and additional problem in the general management of solid waste and has an increasing tendency linked to urban development and an increase in the costs of management and treatment of these liquid residues that are too loaded and highly polluting of the water and atmospheric environment [4]. Hence, the emerging idea of valorizing these large volumes of leachates to reduce their potential nuisance, to reduce the cost of their treatment and especially to take advantage of their nutritive value by using them as fertilizer and energy as a source of biogas or biofuel.

4.2 Leachate: Definition, Composition, and Main Characteristics

Leachate (or percolate) is the residual liquid that results from the percolation of water through a solid or semisolid material. It comes from the Latin adjective “lixivium”, which means “of lye”, or “water used for washing”. This term is currently an official nomenclature strictly reserved for the liquid residual volume generated in a landfill of household waste or still

mediately called “landfill juice”. Thus, during the storage of waste and under the combined action of residual water specific to the waste, water from precipitation, and natural fermentation, the waste generates a residual liquid fraction known as “leachate”. Leachate is characterized by its foul odor and its high content of various organic materials, micropollutants, and trace elements. Thus, they cannot be directly discharged into the natural environment and must be imperatively and carefully collected, treated, and even valorized into biofertilizers and biofuels. It is now well recognized that landfills generate leachates characterized by a high organic load, often malodorous, and by the accumulation of certain compounds, sometimes dangerous, such as heavy metals and various organic micropollutants such as PAHs, PCBs, PBDEs, endocrine disruptors, etc. This residual liquid effluent and its danger can be reflected at the scale of fertilized soils, groundwater following infiltration on the site of landfills and accidental contacts with surface water courses such as Wadis, streams, rivers, etc. Thus, it is ultimately legitimate to protect the natural environment against this potentially dangerous product by recommending the appropriate recovery treatment. This protection of the natural environment requires the treatment of leachates before their valorization or their discharge in the natural environment. The leachate treatment consists mainly of eliminating diversified organic substances, characterized mainly by the measurement of chemical oxygen demand (COD), biological oxygen demand (BOD), and ammonium (NH_4^+) (Table 4.1).

Table 4.1 Average composition of landfill leachate [5]

Parameters	**Range**	**Unit**
pH	4.5–9	
Specific conductivity (uS/cm”)	2500–35000	mg/l
Total solids	2000–60000	
Organic matter		

Parameters	Range	Unit
Total organic carbon (TOC)	30–29000	mg/l
Biological oxygen demand (BOD_5)	20–57000	
Chemical oxygen demand (COD)	140–152000	
BOD < /COD (ratio)	0.02–0.80	
Organic nitrogen	14–2500	
Inorganic macro components		
Total phosphorous	0.1–23	mg/l
Chloride	150–4500	
Sulfate	8–7750	
Hydrogeno carbonate	610–7320	
Sodium	70–7700	
Potassium	50–3700	
Ammonium-N	50–2200	
Calcium	10–7200	
Magnesium	30–15000	
Iron	3–5500	
Manganese	0.03–1400	
Silica	4–70	
Inorganic trace elements		
Arsenic	0.01–1	mg/l
Cadmium	0.0001–0.4	
Chromium	0.02–1.5	
Cobalt	0.005–1.5	
Copper	0.005–10	
Lead	0.001–5	
Mercury	0.00005–0.16	
Nickel	0.015–13	
Zinc	0.03–1000	

The composition of leachate depends highly on the waste, its particle composition, and the geometry and management of the landfill, particularly on how water inputs are controlled. Similarly, the mechanisms involved in the waste degradation

and evolution process can intervene and intensively change the leachate composition. The chemical processes of neutralization, precipitation, oxidation−reduction, organo-mineral complexation, ionization, and acid−base reaction; the physical processes of ion exchange, adsorption-absorption, extraction-evaporation, and encapsulation filtration; and the biological processes of acid and alkaline hydrolysis, synthesis and catabolism, aerobic and anaerobic fermentation, nitrification–denitrification, sulfate reduction, acid−base reactions, and mineralization can change the composition of leachates. Broadly speaking, three main types of leachates can be distinguished, depending mainly on the age of the landfilled matrix waste. Leachates from recent sites with a relatively recent waste age of fewer than 5 years are rich in volatile fatty acids and easily metabolizable and biodegradable organic molecules, leachates of intermediate aged between 5 and 10 years, and leachates from old landfills containing old waste older than 10. These leachates are characterized by the presence and content of heavy and nonvolatile complex molecules, mainly humic and fulvic substances. In addition, the leachates showed main characteristics developing and changing according to time: the climate prevailing on the site, the temperature, the ventilation, and the pluviometry. The biological component of the leachate, in particular the microbial composition, largely influences its evolution and composition. The BOD5/COD ratio are a key parameter and orient the choice of leachate treatment toward a biological and/or a physico-chemical route. Similarly, some specific elements in high mass, such as the portion of ammoniacal nitrogen, toxic substances such as phenol or halogenated compounds, and heavy metals, can also guide the right choice of leachate treatment.

Diversified organic substances carried by some leachate could thus be discharged into the natural receiving and flowing water or reused directly in agriculture as an organic amendment [6, 7]. Faced with increasingly strict regulations, in terms of discharge, the traditional channels derived from

wastewater treatment, such as biological treatment, reverse osmosis, and nanofiltration, had to develop with the introduction of new processes and techniques that are less expensive and highly efficient.

In this context, significant research has focused on innovative and effective techniques for the biological and physico-chemical treatment of leachate. The main landfill leachate treatment techniques and methods will be briefly described, mentioning their performance and average efficiency, cost of implementation and operation, operating mechanism, etc. All these techniques and methods of landfill leachate treatment will be described. All these leachate treatment techniques and methods are grouped in Table 4.2.

Table 4.2 Main treatment processes for landfill leachate [8]

	Processes	**Advantages**	**Disadvantages**	**Byproducts**
Biological processes	Aerated lagoons	Removal of BOD5 and nitrogen	Denitrification is difficult, with a large footprint	Excessive biological sludge
	Activated sludge	Removal of BOD5 and overall nitrogen	Energy consumption for oxygen supply	Excessive biological sludge
	Membrane bioreactor	Removal of BOD5, nitrogen, residual COD, SS, bacteria, and viruses Small footprint	Energy consumption	Biological sludge in excess
	Fixed culture	Removal of BOD5 and nitrogen Low energy consumption	Low purification efficiency	

	Processes	Advantages	Disadvantages	Byproducts
Physical–chemical processes	Coagulation—flocculation	Treatment rates 40 to 50% reduction of hard COD Agglomeration of SS	Important production of sludge (20% of the treated volume)	Sludge
	Precipitation	Precipitation of metals, reduction of SS, elimination of part of the organic matter	Production of metal hydroxide sludge	Sludge
	Activated carbon filtration	Removal of residual COD, reduction of halogenated organs, and color	Renewal of absorbers	
	Ozonation (H_2O_2 and/or UV)	Removal of hard COD, decolonization, and increase of biodegradability	Sensitive to load variations Energy consumption	
Thermal processes	Evaporator	Maximum concentration of pollution Use of biogas	Partial dissipation to the atmosphere, Depends on the quality and quantity of the biogas	Concentrate
	Evapo-condenser	Maximum concentration of pollution and condensation of the vapor Use of the biogas	Depends on the quality and quantity of the biogas	Concentrate
	Isoflash dryer	Removal of pollution load from leachate, Use of the biogas	Depends on the quality and quantity of the biogas	Powdered dry extract

	Processes	Advantages	Disadvantages	Byproducts
Membrane processes	Nanofiltration	Retention of hard COD, salts, and metals	Energy consumption	Nanofiltration retentates
	Reverse osmosis	Retention of hard COD, salts, metals, and nitrates	Energy consumption	Osmosis residuals (brines)
Others	Treatment in the external treatment plant	No investment on site	Cost is sometimes high Dependence on a “third party caterer.”	Nonperennial solution

4.3 The Different Techniques and Methods Recognized to Treat Landfill Leachate

For in situ treatment, each process traditionally used in urban water treatment has its specificity and can claim to meet a particular treatment purpose. It is this treatment aim that should be considered when choosing the process to be implemented. This aim is determined by three criteria: the nature and quantities of leachate to be treated, the discharge constraints (discharge values, nature of the outlet...), and the cost of the installation. The variability of the leachate over time usually leads to the choice of a multipath system that combines various processes.

Table 4.2 summarizes the advantages and disadvantages of the most common leachate treatment processes and the byproducts that may be generated.

4.4 Biological Processes Commonly Used for Urban Wastewater Treatment

During this process, purification is performed naturally because of the proliferation of microorganisms and

microscopic algae.

4.4.1 Natural Lagoons and Aerated Lagoons

The biological purification process in lagoons is comparable to that which takes place in a natural watercourse; thus, it included making the pretreated effluent circulate slowly in a succession of shallow basins. This slow progress of the wastewater in the succession of lagoons results in a residence time exceeding 20 days and leads to the proliferation of microorganisms and microscopic algae. The latter produces oxygen through photosynthesis. The action of the sun's UV rays, which destroy many pathogenic microbial germs, also allows for significant bacterial decontamination of the effluent [9, 10].

Unlike activated sludge plants, the main process commonly used internationally for wastewater treatment, the active purifying biomass is a fixed mass implanted in the bottom of the ponds. The lagoon process takes place in several ponds of varying sizes or lagoons. Each lagoon fulfills a well-defined role. The conditions are related to the intensive biological degradation caused by the dissolved pollution conveyed in the water and the sufficient stabilization of the aerobically settled sludge, essentially also in anaerobiosis. The sand and heavy sludge settle in the inlet area, while the other solids are distributed throughout the primary basin. Floating matter is fixed by a dipping wall at the head of the primary lagoon. Thus, a sufficient supply of oxygen caused by insufflation of compressed air or by natural ventilation in small basins, effective circulation of the water mass ensured by a good dynamic flow materialized by a good distribution and homogenization of the wastewater in the basin to avoid the zones "known as dead" and to ensure exchange between the water to be treated and the biofilm implanted at the bottom of the basin. Most of the mineral and organic residues are well settled at the bottom or biologically degraded in the first or primary basin. A corresponding amount of excess sludge is deposited here with the settled sludge at the bottom of the

tank. The lagoons that follow have a primarily refining role in water treatment. The biological operators often differ according to the order of the lagoons; thus, in the first lagoon, these operators are mainly heterotrophic bacteria, and in the advanced order of the lagoons, the role of autotrophic photosynthetic bacteria and microalgae is crucial. In a certain choice, the system composed of at least three types of lagoons will be supplemented by a last deep basin with an anaerobic tendency to refine the water treatment. The sizing of aerated wastewater treatment lagoons is calculated with the volume load conditioned by the feed rate and the organic load in BOD5 and the programmed residence time.

Regarding the methods used for air insufflation in the lagoons, the differences lie in the method of introducing oxygen and its circulation pattern in the water mass. The depth of the lagoons is considered a good choice for the main air insufflation method in the water mass to be treated. The importance of autotrophic microalgae proliferation is controlled by the oxygenation level of the water, the degree of solar illumination, and especially the water temperature. Some gaseous emanations can take place, particularly during the summer period, with the development of assimilation processes for phosphates and nitrogen forms by the algae, of reduction and putrefaction of the organic matter connected with the growth of methanogenic and sulfurous bacteria (sulfur bacteria with the release of biomethane and hydrogen sulfide or H_2S). Additionally, poorly maintained lagoon processes can often lead to the development of insects and mosquitoes. Thus, the natural or aerated lagoons usually yielded treated wastewater almost in conformity with the standards in vigor from the bacteriological viewpoint, known as biological disinfection, and often not in conformity from the physico-chemical viewpoint considering their strong load of dissolved organic pollution as the DBO5 contributing to the phenomenon of eutrophication once rejected in the receiving natural environments.

As a result, the aerated or natural lagoon is considered an effective and financially beneficial technology to treat urban wastewater in rural or metropolitan regions, under reserve of availability of space and sufficient land for adequately implementing the treatment plant.

This process is easily integrated into the natural site, and the implementation and operating costs are reduced, as there is not enough expensive equipment to implement and not enough residual products to treat, contributing to the reduction of costly maintenance. The extended residence of the wastewater in the lagoon allowed for obtaining a constant quality of the treated effluent, even in case of pollution peaks or momentary hydraulic load.

4.4.2 Activated Sludge

The activated sludge process is one of the most common and largest processes used for secondary wastewater treatment [9]. The operation of activated sludge treatment is comparable to that of lagoons since both treatment techniques are based on the principle of free cultures operating the treatment. The installation essentially comprises a structure in which aerobic microorganisms are maintained fixed or not in a sufficiently oxygenated environment; these microbes ensure organic matter degradation and transformation by oxidation into mineral matter according to the process commonly known as mineralization.

Water treatment proceeds in an aeration tank in which a mixture of wastewater and aerobic bacteria, called “activated sludge” or “mixed liquor”, is kept in suspension. The phenomena involved are therefore the same as those that can be observed in a river or a lake, with the difference that the microbial operators are intensively concentrated in very large numbers in a small space called the activation basin. The oxygen supply and the mixing of the “wastewater bacteria” are ensured by mechanical systems such as brush bridges, turbines, or air blowers. Wastewater microorganisms that

degrade pollution grow, proliferate and agglomerate in complex functional biofilms. This biofilm develops intensively following the abundant availability of oxygen and nutrients; these various nutrients, coupled with vigorous mechanical stirring, ensure the formation of solid settling microstructures called settling flocs. This biological flocculation is comparable to the coagulation-flocculation caused by a chemical coagulant. The generation of a mass suspension involving liquid and solid phases is followed by a separation process of decantation-filtration in a clarifier. The solid part, rich in concentrated and settleable matter, is called sludge. This sludge is trapped in the clarifiers, and as they are rich in organic matter and therefore highly biologically reactive, the sludge must be stabilized, i.e., depleted in the metabolizable matter by oxidation, mainly biological, by the dominant microbial flora, and transformed into mineral matter according to the general mineralization process.

The process of purification by activated sludge mostly depends on microbial activity and its oxygen-dependent development. The oxygen supply and the suspension of bacterial floc development or sludge are ensured by a specific device, such as turbines, air boosters, brush bridges, etc. After pretreatment using bar screens, sand traps, and grease traps, the wastewater is admitted into an aeration tank, the biological reactor, allowing the development of bacteria and the degradation of pollution. The treatment continues with a sludge settling phase in a clarifier, allowing the separation of the treated water from the sludge. The settled sludge is partly returned to the biological reactor as the recirculation part, and the excess is directed to the sludge treatment process as the extraction part.

This activated sludge process is recognized as very efficient and controllable, ensuring good carbon and nitrogen pollution elimination between 60 and 75%; it is also a process that allows advanced phosphorus degradation after association with a complementary physical–chemical treatment of coagulation-flocculation. This treatment is

characterized by high production and operating costs (energy and reagent expenditure, equipment depreciation, sludge treatment, disposal, etc.). Both at once, it requires a high level of technical expertise, rigorous operational qualified and well-initiated personnel, with regular visits to the site several times a week for inherent adjustments, self-monitoring, equipment maintenance, upkeep of the works and surroundings, sludge extraction, etc. It requires the implementation of remote monitoring. The operation of the plant must integrate the sludge treatment and disposal process. This treatment of sewage sludge aims mainly at reducing its volume to limit the quantities to be stored (for spreading and landfill sites in particular), stabilizing its content to improve its chemical composition and its consistency to reduce its biological activity of biodegradation. Before being stabilized, sludge is rich in water, and its high content of fresh organic matter makes it particularly fermentable, which results in the generation of bad odors over large areas. Thus, the stabilization process allows the production of sludge with a high dryness, representing the dry matter rate and considerably limiting this environmental atmospheric nuisance.

The dewatering and stabilization treatment of sludge was performed in well-mechanized systems for mega plants and drying beds for small plants. This treatment of extracted sludge would closely depend on its destination and potential reuse as mainly landfill areas, incineration, and landfills and as fertilizing and restructuring materials of agricultural soils. This treatment of sludge can vary depending on the composition and type of sludge that leaves the treatment plants and their final use, which will not undergo the same treatment. Sewage sludge has variable compositions according to the adopted wastewater treatment process, and its dryness level is a key parameter in the choice of its treatment and its future destination; thus, as we have said above, this sludge can either be treated and valorized for agricultural fertilizing or restructuring use or energetic use

by biogas production, or it can be eliminated by incineration or be dumped as a restructuring landfill material. Finally, a study of the sludge treatment process is necessary depending on the capacity of the treatment works set up and especially on the wastewater to be treated, urban or industrial residual. This sludge treatment process includes the first step of the thickening and homogenization process of the content by mechanical mixing and high water loss. In the second step, dehydration allows an increase in the dryness of the sludge to make a pasty sludge or a solid sludge, and finally, the drying operation ensures that a dry solid sludge is obtained. All these last steps of sludge treatment aim essentially at gaining the volume of sludge to facilitate its storage, transport, and valorization. There are two kinds of drying: thermal drying can be direct or indirect via a heat transfer fluid: by evaporation of the sludge by convection or by heat exchange through a heated conduction wall. Notwithstanding the method of drying, the finished sludge showed a dryness of up to 95%. Thermal sludge drying techniques are demanding and energy-consuming, and solar drying, which is much more ecological, will be preferred to reduce the environmental footprint of the sludge process. The drying beds or solar drying are made of a draining surface to promote the dehydration of sludge by drainage and by natural evaporation following exposure to sunlight.

The last step of sludge treatment is its stabilization, which ensures a maximum reduction of the content of fresh organic matter in the sludge to make it as little fermentable as possible and therefore to reduce the bad and odorous nuisances likely to emanate from this sludge. This odor stabilization is of two types.

There are three main techniques of organic matter stabilization: (i) ***Biological stabilization*** includes two types of aerobic stabilization, ensuring an essential transformation of the sludge into compost for agricultural soil amendment. However, anaerobic stabilization, known as “methanization”, allowed sludge transformation into biogas; this biogas

production will be used as heat, electricity, fuel, etc. (ii) ***Chemical stabilization***: the neutralization of the biological activity of the sludge to reduce its organic matter content. To do this, the sludge is mixed with lime (quicklime or slaked lime), and the doses of lime are calculated according to the sludge's dryness. This stabilization technique can be performed only after sludge dehydration. All these last processes are known as sludge hygienization and aim to destroy all forms of life, particularly pathogenic microorganisms.

In addition, the agricultural use of sewage sludge represented approximately 70% of the treated sludge as agricultural soil fertilizer, and the rest of the sludge is destined for other uses, such as incineration (12%), biogas production (2%), storage in landfill sites (5–8%), etc. These percentages of sewage sludge utilization can significantly vary depending on the geographical location of the sewage treatment plant and the country's development status, whether it is developed or developing.

This paragraph briefly describes the activated sludge wastewater treatment process. This process is the most common and widely applied process worldwide for urban wastewater treatment, and it is used to treat landfill leachates in a unitary way or mixed with urban wastewater.

It is a successful procedure that has confirmed its effectiveness in several industrial wastewater treatment applications.

4.4.3 Membrane Bioreactors

The operating mechanism of membrane bioreactors is based on membrane filtration to separate sewage sludge from treated wastewater. This fine filtration, which is between micro- and ultrafiltration, is a compact and relatively complex technology compared to the processes usually used to treat domestic effluents. Many types of research on this innovative process have demonstrated its efficacy in reducing carbon pollution, suspended solids, and water disinfection. The

consumption of chemical reagents and energy is also higher than that usually registered for more conventional processes, which can achieve performances comparable to those often recorded for activated sludge associated with an adapted tertiary treatment by UV-C 254 irradiation, chlorine, its derivatives, etc.

In addition, the need for almost permanent aeration in the membrane system used to prevent fouling makes it difficult to treat and eliminate nitrogen. The installation of a specific basin called an anoxia basin is strongly recommended and advised in case of strong requirements on this parameter of nitrogen. In the same way, for an advanced treatment of phosphorus, it is useful to foresee an anaerobic tank to limit the injections of specific coagulation-flocculation reagents, whose use in excess increases the risks of membrane clogging.

This technology is mainly limited to the filtration rate of the membranes and is conditioned by the surface of the installed membranes, their wear and tear, and their clogging state. Thus, many factors or unforeseen events can contribute to making the effective filtration capacity insufficient, following hydraulic overload caused by intrusions of clear water in the sewerage network, nondomestic effluents, bad management of the reagents used on the plant, fall of the temperature of the effluents, washing of the membrane modules...). This is likely to cause effluent spills despite a storm basin that can collect some of this water.

The studies carried out on this process show that membrane bioreactors allow stable and efficient purification, with certain requirements and sensitivity. It thus appears that the additional costs related to the choice of membrane filtration to treat urban effluents must be justified by several factors: requirements of a thorough treatment of carbonaceous, particulate, and bacteriological pollution, important land constraints, and knowledge and control of the quantity and quality of the effluents arriving at the plant. The desire to reuse treated wastewater can be an additional

argument but is not sufficient to justify the choice of this method [11].

4.4.4 Biological Processes with Fixed Culture or Biofilters

The term biological filters or biofilters is commonly used in wastewater treatment and includes all processes that combine biological purification by fixed cultures on a specific support adapted synthetically as PVC or naturally as gravel with the retention of suspended matter generated and represented essentially by the microbial biomass operating in the biofilter, mineral materials such as clays and organic materials such as polysaccharide polymers, lipids, proteins, and fulvic and humic acids. These materials in suspension present solid cohesion reinforced by various bonds of electrostatic types and biophysical-chemical and mechanical mixing (Fig. 4.1).

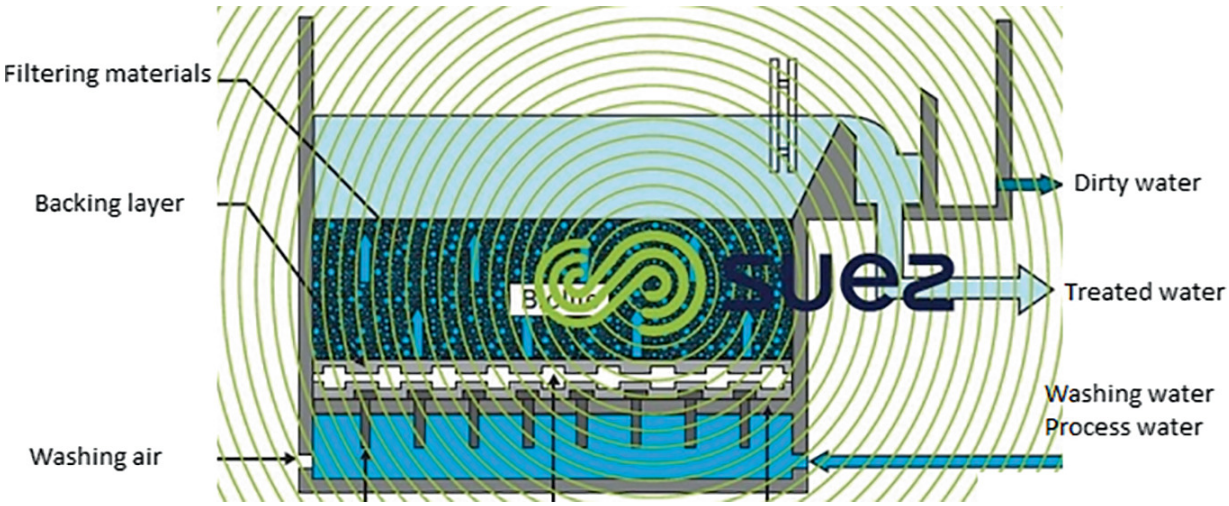


Fig. 4.1 Schematic diagram of wastewater treatment by biofiltration

This technique of wastewater treatment by bioffilters implements thin biological films regularly renewed by washing in cycles of 12–48 h. The result is a biomass of higher concentration and activity than in activated sludge. Its main advantage compared to activated sludge is space savings, due in particular to the suppression of the clarification stage. This

compactness facilitates the coverage of the works, the control of olfactory and sound nuisances, and the realization of esthetic stations; no risk of leaching since the biomass is fixed on a support, allowing face flow variations, and finally a modular construction with a certain facility of automation as filtration batteries.

Thus, the biofilter is a biological reactor with bacteria fixed on a specific material, natural or synthetic, as a fixed and immersed monolayer. This material is called BioLite, based on expanded clays or PVC, with a density higher than 1.2, and offers a strong irregularity and a large specific surface. This characteristic allows for keeping a bacterial film sufficient to restart the biological process quickly, even after a rather energetic air and water washing.

Oxygenation can be achieved by the prior dissolution of oxygen from the air or by direct injection of air into the reactor to achieve, in addition to eliminating suspended solids, a reduction in the residual BOD and nitrogen. This oxygenation is usually done at the current rate to remedy the clogging problem often observed in biofilters or by rotation of the filtering support through oxygenation with air and an immersion step in the water mass, better known as a rotary biofilter. The major disadvantage of this process is that it is not practicable to install in mega-treatment plants because of the high hourly and daily flow arriving at the plant site, but it is well adapted to the volume of leachate generated daily at a landfill site.

Bacterial beds are a slightly older variant of biofilters, the latter being a sort of update of the bacterial bed technique. The principle of operation of the treatment by bacterial beds is based on the realization of a work comprising support allowing the fixation and development of the microorganisms. The system promotes contact between the wastewater, the surrounding air, and the purifying microorganisms. The bacterial bed comprises materials with a large contact surface (pozzolan, plastic elements in PVC with enough partitions, etc.) piled up in a tank whose bottom is perforated to allow

the supply of atmospheric oxygen. The effluent pretreated by primary decantation is distributed on the surface of the bed by a sprinkler. After percolation through the lining material, the treated water is recovered at the base of the structure. Depending on the treatment process, the treated water is returned to the natural environment or is directed to a refining and clarification treatment and potentially to disinfection. For the biofilter case, trickling filters are a process that lends itself well to the treatment of landfill leachate.

4.4.5 Use of Chlorella Microalgae for Wastewater and Leachate Treatment

Biological operators such as microalgae have been tried for a double role: to purify landfill leachates and to take advantage of the residual products of this purification by producing useful substances such as diverse culture media and microbial growth or by generating compounds with energetic and pharmaceutical potential. Among the microalgae, we found Chlorella, which is a unicellular, nonmobile microalga of spherical or ellipsoidal shape. The size of cells measuring barely 2 to 12 µm in diameter. These algae are ubiquitous in freshwater streams, lakes, and oceans. They are deep green related to their chlorophyll content. The family of Chlorellaceae comprises 4 genera and 40 species; among these well-known genera, we distinguish the genus Chlorella, which presents approximately thirty species, mainly *Chlorella vulgaris* and *Chlorella pyrenoidosa*.

As a chlorophyll species, Chlorella is characterized by high chlorophyll contents (type and, b) that promote active photosynthesis, allowing and ensuring the production of high oxygen contents [12]. Analysis of the composition of Chlorella shows a significant richness in plant proteins on the order of 60% [13]. In addition, it has been shown that Chlorella is rich in vitamins A, B, and C, hence its high nutritional value for both fish and humans [14].

The only known mode of reproduction in the genus Chlorella is asexual reproduction through autospore formation [3, 15]. Autospores are produced because of cell division in Chlorella. Each cell can divide into 2, 4, or sometimes even 16 daughter cells. These daughter cells are then released, and they are identical to the mother cell but much smaller.

Many studies have shown that algae and Chlorella can fix a wide range of heavy metals in concentrated form from aquatic environments, such as Cu, Zn, Pb, and Cd [16–19].

In freshwater, a stimulating and nutritive effect of some soluble compounds originating from petroleum products has been found on various algae, such as Chlorella vulgaris, Anabaena flosaquae, *Ankistrodesmus convultus*, and *Tabellaria fenestrata*. Thus, Chlorella vulgaris can metabolize alkanes such as n-tridecane, n-tetradecane, and n-pentadecane in the presence of light and CO_2. The degradation of its hydrocarbons is so extensive that residual oxidation products such as alcohols and ketones are detected only in trace amounts in the culture medium [20].

Similarly, very stable organic micropollutants such as polychlorinated biphenyls (PCBs) can be absorbed and fixed with no metabolic effect by algae, and some can provide a concentration factor of over 1000 compared to that existing in the natural ambient environment. The binding capacity of these organic micropollutants can vary between algal species; for example, *Chlorella geitleri* absorbs more PCBs than *Scenedesmus acutus*. PCBs can be removed from the water even by dead algal cells. It can be concluded that the mechanism of fixation of such organic micropollutants is related to dissolution in cell lipids and/or even simple physical adsorption [20].

The disruptive effect of some pollutants on the enzymatic activity in Chlorella has been used to determine these pollutants in the environment. *Chlorella vulgaris* has two families of enzymes that are particularly interesting: phosphatases and esterases [21]. The localization of these

enzymes is membranous, allowing for fast enzymatic reactions.

—Phosphatases are enzymes whose function is to manage the level of phosphorus, an essential element for life and cell growth, during periods of deficiency. These enzymes hydrolyze phosphorus molecules to release hot phosphate. The latter can then passively diffuse through the cell membrane to start and serve as intracellular phosphate stocks. In microalgae, 3 types of phosphatases, acid, neutral or alkaline, can exist depending on the pH [21]. Alkaline phosphatase is inhibited in the presence of heavy metals such as Pb^{2+}, Cd^{2+}, and Zn^{2+}.

- Esterases are enzymes that hydrolyze the ester bonds of many organic molecules. Studies have shown the existence of nonneuronal cholinesterase in plants overall [22]. Wang et al. [23] suggest that these enzymes are involved in photosynthesis and the regulation of water uptake. Esterases are inhibited by carbamates and organophosphates [24] and the high conductivity of the environment [25].
- Bioassays or biosensors based on the detection of these enzymatic activities have been developed. The work of Chouteau in 2004 allowed the development of a bienzymatic conductimetric biosensor using Chlorella vulgaris cells immobilized in a membrane based on bovine serum albumin cross-linked in glutaraldehyde vapors. This biosensor can detect a family of pollutants, such as heavy metals and some organophosphorus or carbamate molecules, at low concentrations (0.1 ppb for Cd^{2+}, 1 ppb for Pb^{2+}, and 10 ppb for Zn^{2+} and carbamate).
- Studies have shown that Chlorella can fix mineral substances such as NH_4^+, NO_3^-, and PO_4^{3-}, which are considered pollutants, in wastewater effluents [16, 26, 27]. These same substances are nutrients for algae according to the work of [28], hence the great interest in the application of these organisms in the water treatment process.

- All these observations of laboratories or fields in the natural environment could lead to applications in new biological treatments of wastewater and landfill leachate, based on strains of Chlorella and implemented either in conventional lagoons [27] or in more intensive processes: immobilized algal cultures in agar beads [26, 29] or on rotating disks [30, 31].

Concerning wastewater treatment, the results of Chlorella's ability to remove dyes from textile wastewater by an HRAP (high algal level pond) system are reported in Table 4.3. The biosorption of textile dyes by Chlorella biomass was also investigated. The mechanism of color removal by C. vulgaris is biosorption, and it follows the Langmuir and Freundlich kinetic models. The HRAP using *Chlorella vulgaris* offers a good system to treat wastewater from textile mills (TWs) before final disposal [32].

Table 4.3 Various studies have used Chlorella for wastewater treatment

Pollutant	Chlorella strain	Living cell	Immobilized in alginate	Reduction rate	Algal growth rate	References
Textile dye: Supranol Red 3BW		+	−	50% 3BW 45.1% NH_4-N 33.3% PO4-P 62.3% COD	0.17 to 2.26 mg Chla/l	[32]
Biocide (Tributyltin: TBT)	*C. vulgaris*	+	+	90% TBT	1–2.7 and 3.3 (×106 cells)	[33]
	C. miniata *C. sorokiniana*	−/ + −/ +	− −	85% (−)/40% (+)	-	[34]
Ammonium (NH_4^+)	*C. srokiniana*	−	+	100% NH_4^+		[26]

In the case of the biocide TBT (tributyltin), the work of [33] shows that chlorella degrades this element into DBT (*dibutyltin*) and MBT (*monobutyltin*). Chlorella can grow in textile waters without dilution. Its purifying power decreases with increasing pollutant concentration. TBT is rapidly eliminated in the first 5 min of treatment with a percentage of 85% of elimination ensured by a dead Chlorella, with no difference in abatement between the first 5 min and the three days of exposure. The percentages of TBT removed by live Chlorella cells were significantly lower than those removed by dead cells of the same species. After three days of treatment, the percentages obtained for TBT removal by live Chlorella cells were only half that obtained by dead cells [34].

The treatment of landfill leachate with Chlorella cells has recently emerged as a new initiative to address and exploit the increasing volumes of leachate generated in landfills.

Thus, a great deal of work on using microalgae in wastewater treatment is readily available in the literature and industry, while that on the treatment of landfill leachate by Chlorella is almost nonexistent [35–37], and the working group of [38] pioneering this research in China has well introduced this strand of research on the treatment of landfill leachate by microalgae. Thus, two microalgae, *Chlorella pyrenoidosa* and *Chlamydomonas snowiae,* were isolated from a leachate pond with high ammonia levels in the Li Keng landfill in Guangzhou, China. Their growth and nutrient removal rates were determined in a dilution series of landfill leachate under laboratory operating conditions, and their growth rates were compared with those obtained for a *C. pyrenoidosa* strain isolated from an unpolluted river.

However, the growth density of *C. pyrenoidosa* (LK) and *C. pyrenoidosa* showed an increase on the order of 19.4 and 1.7 times, respectively, in the 30% leachate. For *C. snowiae,* negative growth was recorded with a significant decrease in cell density with 30, 50, and 80% leachates and crude leachates.

The relative removal rates of NH_4^+, ortho-P, and COD in leachates diluted by 10 and 30% appeared higher than those recorded for concentrated leachates of 50, 80, and 100%, respectively. Chlorella-treated leachates of 10 and 30% allow germination of *Brassica chinensis* seeds at 65 and 50%, respectively, while being lower than the germination rate recorded in distilled water at 97%.

The mass cultivation of Chlorella in wastewater or leachate treatment can be a first step to the valorization of the produced biomass in energy production as biofuels.

Within this important theme, our team has developed important research to propose a strategy to treat landfill leachates by green microalgae isolated from the natural environment, namely, *Chlorella sp*. Research on the optimal conditions for the growth of this alga in variable proportions of leachate has been undertaken, and the valorization of the algal biomass generated has been valorized in biofuels. The growth of this algal isolate was tested in a specific conventional algal culture medium, Bold medium, and compared with that obtained at different proportions of leachate. In the same sense, experiments were carried out to evaluate the growth of the algae, its main physiological parameters, and its purifying effect on the main leachate pollution parameters. Biomass harvesting after algal culture will be followed by a study of the yield of lipids and biofuels generated. The results showed mainly that this strain of microalgae *Chlorella sp*. is resistant and can grow well even in raw leachates, but its growth in its optimal development is recorded at 10% of leachates. The development of Chlorella in 10% leachate resulted in the reduction of pollution parameters such as salinity, COD, NH_4^+, etc. The harvested biomass has allowed the harvest of a considerable mass of lipids originally and exploited for the production of third-generation biofuels. In conclusion, in this innovative investigation on landfill leachates, the use of this *Chlorella sp.* strain is a good alternative to minimize the costs of leachate

treatment and an important source for biofuel production [35, 36].

4.5 Leachate Treatment for Biofuel Production

The interests of biofuel production cover all liquid, solid, or gaseous fuels produced from biomass and intended for energy recovery in transport, electricity, heating, or other uses. Biomass, a carbon-based renewable energy source for liquid fuels, has the potential to lessen dependence on fossil fuels and environmental problems [39]. Energy needs related to technological development, industrialization, and the growing population are increasing. However, fossil energy deposits are estimated to be depleted within 50 years at the current rate of consumption [40]. Burning oil releases trapped fossil carbon, increasing atmospheric CO_2, one of the greenhouse gases responsible for climate change. Biofuels have an advantage over fossil fuels because they limit greenhouse gas emissions given that their CO_2 emission balance is neutral [41], as the CO_2 emitted is compensated by the CO_2 fixed via photosynthesis, which can present a greener alternative energy source. In addition, the European Union aims to reduce its energy dependence on fossil resources monopolized by OPEC countries (Organization of the Petroleum Exporting Countries). It wishes to diversify and decentralize energy supplies [42].

Biofuels are used as additives or complement to fossil fuels such as gas oils, incorporated as biodiesel, gasoline, incorporated as ethanol, kerosene, and gaseous fuels such as methane.

Bélair [43] reports that the classification of biofuel generation is based on the biomass used, the process, and the final product obtained. Thus, a distinction is made between first-, second-, and third-generation biofuels.

The first generation of biofuels relied on crop reserve organs such as the seeds of cereals (wheat, maize) or oilseeds (rapeseed, sunflower, jatropha), roots of sugar beet or sugarcane, and fruits of oil palm [44, 45]. These reserve organs of plants, such as wheat, maize, beet, and cane, store sugar as starch or oil, such as rapeseed, sunflowers, palm, and jatropha. As these reserve organs are also used for human food, the production of biofuels competes with food production. In addition, first-generation biofuels require a large area of cultivable land, a large amount of water, agricultural machinery, fertilizer, and the use of pesticides.

In addition, biodiesel can be produced by transesterification of vegetable oil or ethanol by fermentation of synthetic plant sugars (Fig. 4.2).

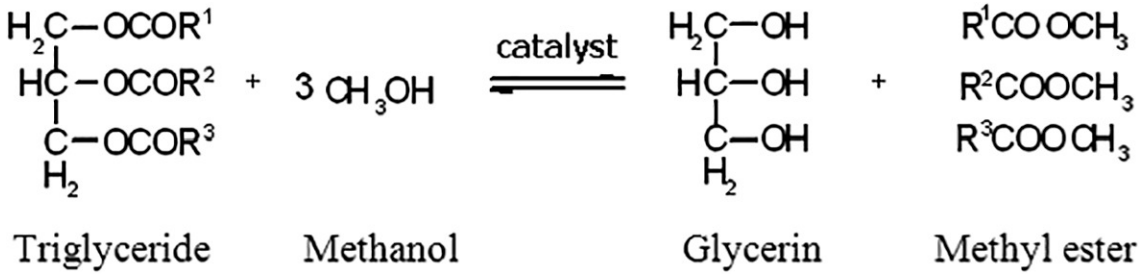


Fig. 4.2 Main pathways of triglyceride transesterification

The production of biofuels from microalgae is an emerging technology; via photosynthesis, Chlorella uses solar energy, water, and carbon dioxide (CO_2) to synthesize algal biomass and is accountable for approximately 40–50% of total CO_2 fixation on the scale of the planet. Some species accumulate up to 60% of their dry weight in oils as reserve lipids, including triglycerides and complex compounds containing long carbon chains, which represent an interesting resource for biodiesel synthesis.

The biosynthesis of lipids in microalgae cells would take place as soon as the culture conditions become unfavorable to their growth, such as the limitation of nitrogen, phosphate, and oxygen contents, various stresses, etc. Thus, microalgae, like other organisms, accumulate reserve compounds such as

starch or lipids as oils. The biosynthesis of oil triglycerides takes place in three main steps and involves several compartments in the cell. The biosynthesis of triglycerides in microalgae, in particular Chlorella, is first carried out in the chloroplasts from the sugars resulting from photosynthesis. After modification, three fatty acids are combined with a glycerol molecule to form a triglyceride in the endoplasmic reticulum or chloroplast. These triglycerides are then stored in lipid bodies in the cytoplasm and chloroplast called oleosomes (Fig. 4.3).

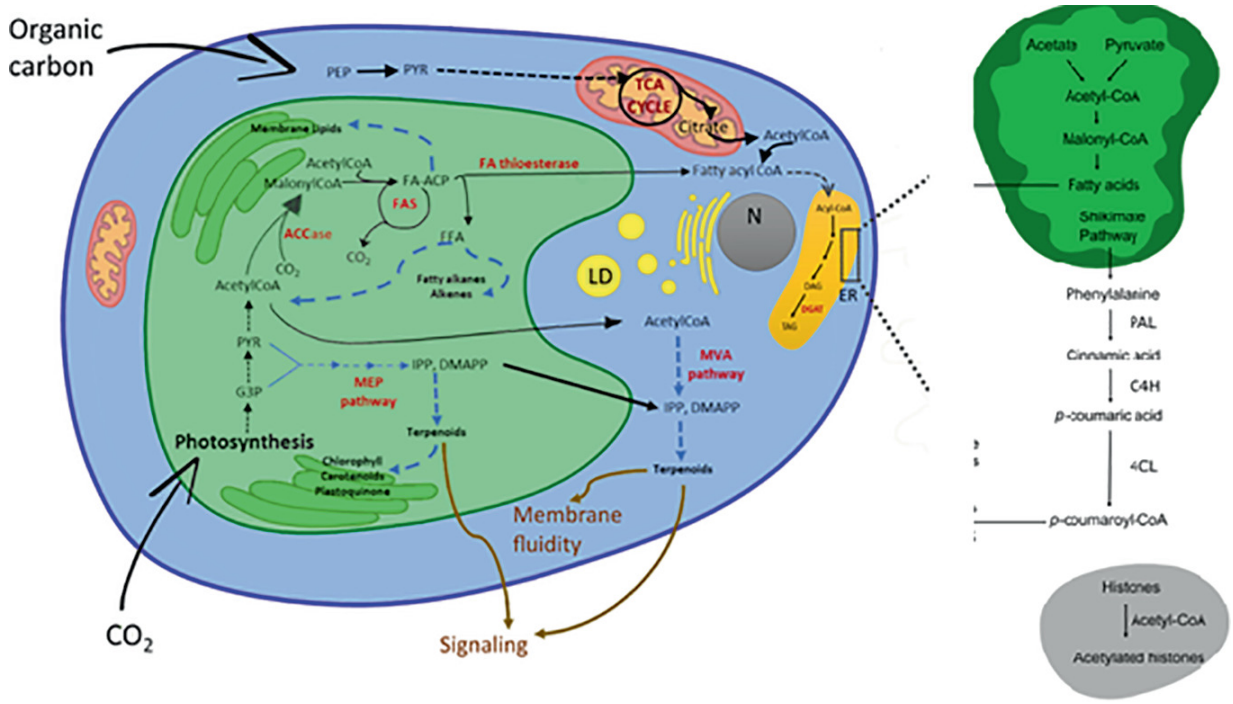


Fig. 4.3 Schematic view of lipid biosynthesis in a eukaryotic microalgae cell [46, 47]

Lipid biosynthesis by Chlorella occurs at the level of chloroplasts, and in this important organelle, the acetyl-CoA from the triose phosphate pool, or imported as acetate from the cytosol and mitochondria to the plastid, is carboxylated by ACCase to malonyl-CoA [46, 47]. The malonyl group is transferred to an ACP. Malonyl-ACP undergoes a series of condensation reactions with acetyl-CoA, catalyzed by a β-ketoacyl-ACP synthase (KAS). By route of the FAS

multisubunit complex, saturated C16:0-, C18:0-ACP, and acyl groups are synthesized.

The properties of microalgae oils are determined by the characteristics of the fatty acids constituting the triglycerides, including the length of the carbon chains and their saturation, defined by the number of carbon−carbon double bonds.

The transformation of the reserve lipids accumulated by the microalgae into biodiesel requires the harvesting of the biomass and the extraction of the oil, two steps that alone represent approximately 70% of the production cost.

Biodiesel is defined as a mixture of alkyl esters derived from oil and triglycerides of plant or animal origin. The viscosity of vegetable oil is 10–20 times higher than that of biodiesel. A modification technique, most often transesterification or esterification, can reduce the viscosity and generate a substitute for petroleum diesel.

Transesterification is a reaction of triglycerides with alcohol, in the presence or absence of a catalyst, to form fatty acid esters (biodiesel) and glycerol as a byproduct. Methanol and ethanol, which are the most suitable alcohols, form a methyl ester and an ethyl ester, respectively. Methanol is usually used because it is the cheapest of the alcohols. In terms of proportion, the reaction requires 3 mol of alcohol per mole of triglyceride to produce 3 mol of biodiesel and glycerol. Since the reaction is reversible, alcohol must be added in excess to promote the formation of fatty acid esters, and the ratio is typically 6:1 alcohol to oil (Fig. 4.3).

Transesterification can be performed under supercritical or subcritical conditions, assisted by ultrasound or microwave, in the presence of an enzyme, acid, or base catalyst.

Several pieces of research dealing with oil production by Chlorella have been cited in the literature, and the microalgae *Chlorella sp*. isolated from freshwater has been identified as a potential primary operator for oil production of variable fatty acid composition [10, 36, 48].

Chlorella sp. is a heterotrophic organism, and the relationship between lipid productivity and the culture process has been well studied using surface response methodology (SRM). According to this model procedure, glucose concentration was always considered a key factor up to 34.66%. Lipid productivity also increased to 247.16 mg L^{-1}. d^{-1} under optimal culture conditions; thus, the average composition of the medium would be glucose 26.2 g. L^{-1}, sodium nitrate 2.06 g. L^{-1} and a growth temperature of 28.18°C. The isolated microalgae had great potential for biofuel production considering their high lipid yield [49].

However, in the studies of Thangalazhy-Gopakumar et al. [39], the freshwater green alga “Chlorella vulgaris” was taken for the pyrolysis study. The results showed that the fixed bed pyrolysis of algae gave a bio-oil yield of 52.7 wt%. This represents a yield of 60.7 wt% carbon. In addition, pyrolysis analysis of *Chlorella vulgaris* was performed in a Py/CPG-MS to identify major compounds present in the bio-oil with and without a catalyst (H+ZSM-5). The study revealed that in catalytic pyrolysis, as the catalyst loading increased from zero to nine times the biomass, the carbon yield of aromatic hydrocarbons increased from 0.9 to 25.8 wt%.

Our recent research was carried out on this topic of lipid productivity from landfill leachate by a strain of *Chlorella sp*. selected from the natural environment to highlight the usefulness of this strain in biofuel production. The main results of this study are represented in Fig. 4.4.

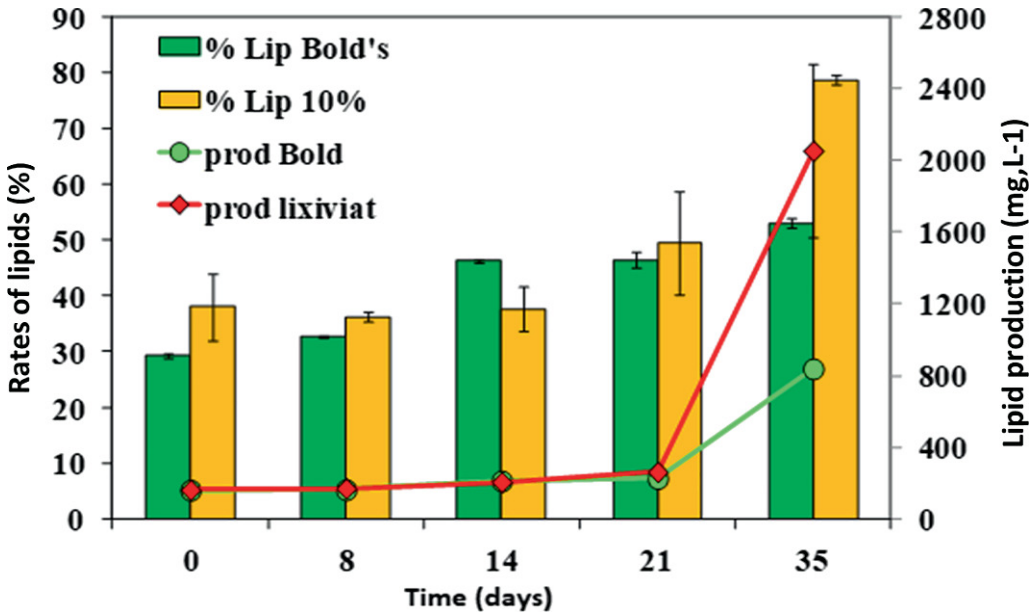


Fig. 4.4 Variation in percentage (histograms) and concentration (curves) of lipids in *Chlorella sp.* cells grown in Bold's culture medium and 10% leachates

Figure 4.4 shows that the initial lipid productivity of this strain of *Chlorella sp.* was important and can exceed 30% compared to other rates reported in the literature for other strains of Chlorella. For example, the study by Zhu et al. [50] showed a lipid production rate of approximately 34% for a strain of Chlorella sp. [51, 52] found that in *Chlorella vulgaris* strain cells, the registered lipid content did not exceed 20%. However, Fig. 4.5 shows that over time, the percentage of lipids varied weekly, especially during the three phases of growth, latency, exponential, and stationary. However, during the final lethal or decay phase obtained after 28 days, a significant increase in lipid content was recorded. It has been noticed that lipid production in leachates was of the same as in a Bold culture medium, the most commonly used medium for conventional Chlorella culture and growth, during the three common phases of growth. In addition, a significant increase in lipid production was recorded, especially during the lethal phase of the alga in both categories of growth medium used, but with a more significant rate in the 10% leachates. Thus, we reported in Table 10 the variation of lipid production of *Chlorella sp.* submitted to culturing in Bold's medium and at different dilution rates of landfill leachates

and during the three phases of growth, exponential, stationary, and lethal. The final percentage of lipids recorded was approximately 53% in Bold medium and 78% in leachates. Therefore, in the leachates, the lipid content showed a significant increase of approximately 1.5 times compared to that obtained in the Bold reference culture medium. This last result represented evidence to support the idea that landfill leachate can be a good culture medium for biodiesel production by Chlorella. This result is confirmed by the studies of [53, 54], who have shown that the most economical and environmentally friendly way is to produce an alternative to fossil fuels by installing an algae culture media preparation unit at sewage treatment plants and/or controlled landfills.

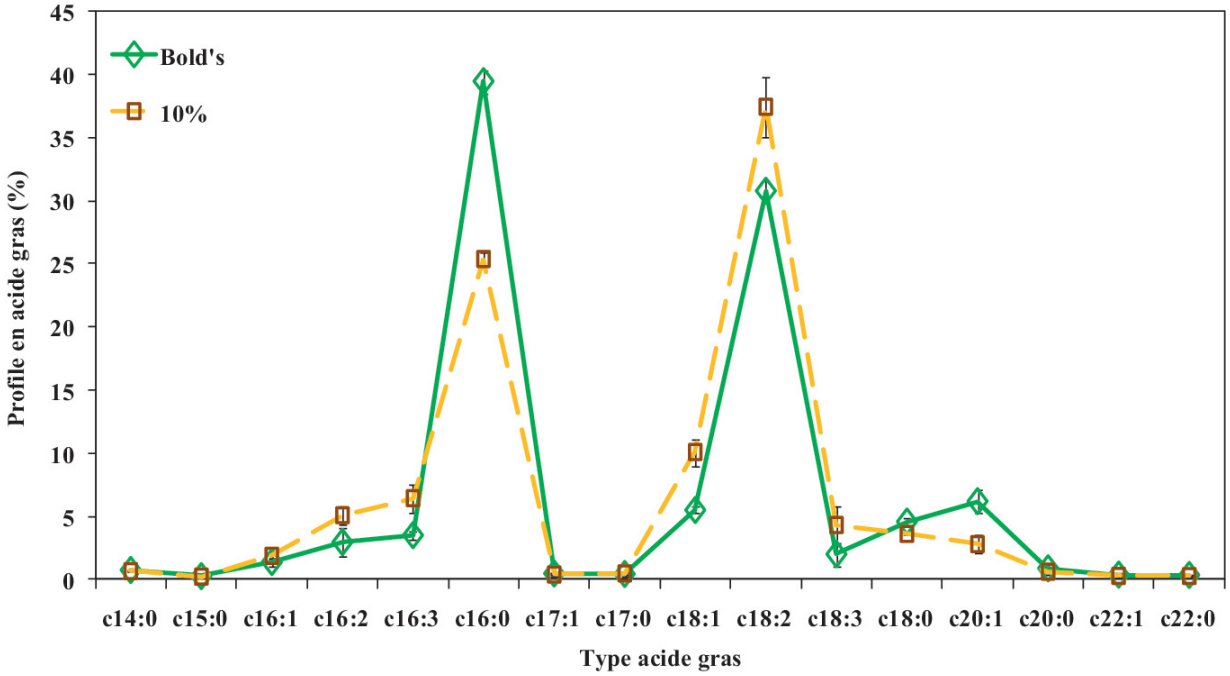


Fig. 4.5 Profile of fatty acids in (%) in *Chlorella sp*. after chloroform–methanol extraction (2:1) and culture in Bold's medium and a 10% dilution of the leachates

The increase in lipid production at the end of the growth cycle could be because of the deficiency of the medium in nitrogen, which is the main source of biosynthesis with the carbon of the biomass (Table 4.4). Widjaja et al. [55] confirmed that when the culture of Chlorella was higher than

20 days, a nitrogen deficiency of the medium was induced and reproduced. Thus, if the level of nitrogen in the medium becomes lower than 1 nM, the microalgae are in a stressful situation and produce lipids that will accumulate inside the cell. The storage of lipids results from a transient imbalance between the flow of carbon from photosynthesis and the flow of other elements for growth, such as phosphorus and nitrogen. The algal cell, having suffered a deficiency in one essential element, does not immediately stop the acquisition of CO_2 for photosynthesis, while it cannot use it to build proteins because of the lack of nitrogen, for example. The cell must therefore store this flow of elements, and in some Chlorella species, this storage would take place mainly as lipids [56].

Table 4.4 Relationship between biomass productivity, lipid content, and growth phase of Chlorella in Bold culture medium and leachate

Phase	**Exponential**		**Stationary**		**Lethal**	
Medium	**Bold's**	**10% leachate**	**Bold's**	**10% leachate**	**Bold's**	**10% Leachate**
Biomass Productivity (mg. $L^{-1}J^{-1}$)	1.37 ± 1.26	6.8 ± 2.15	18.12 ± 0.13	42.55 ± 3	131.97 ± 7.08	1046.55 ± 58.33
% of lipids	46.3 ± 0.22	37.64 ± 4	46,5 ± 1.31	49.5 ± 9.29	53.0 ± 0.94	78.7 ± 0.93
Rates of lipids (mg^{-1})	0.46 ± 0.01	0.36 ± 0.04	0.46 ± 0.01	0.49 ± 0,1	0.53 ± 0.01	0.78 ± 0.01
Lipid Productivity (mg $L^{-1}J^{-1}$)	0.63 ± 0.58	2.56 ± 0.8	8.62 ± 0.21	19.41 ± 0.99	69.62 ± 5.22	823.16 ± 45.88

Like terrestrial plants, some species of microalgae can accumulate, under certain culture conditions, fixed carbon as lipids called triglycerides. The stored lipids will be a carbon reserve for the microalgae. Under normal conditions, these contents remained low, between 20 and 50%. However, some species can accumulate up to 80% of their dry weight in lipids, as reported by Bigogno et al. [57] and Chisti [58].

Thus, algal families such as Chlorophyceae (e.g., *Chlorella vulgaris*), Chrysophyceae (e.g., Diatoms), and Bacillariophyceae (e.g., *Chaetoceros calcitrans*) are the most lipid-producing [57].

The characterization of fatty acid esters in the isolated *Chlorella sp.* strain or biodiesel was performed by gas chromatography equipped with an injector and a flame ionization detector (GC-FID). The fatty acids in *Chlorella sp.* grown in Bold's culture medium and leachates after 20 days of cultivation are regrouped in Table 4.5.

Table 4.5 Composition and nomenclature of fatty acids identified by GCMS-FID in *Chlorella sp.* grown in Bold's medium and the 10% dilution of leachate

Fatty acid	Formula
Capric	C10:0
Myrestic	C14:0
Pentadecyl	C15:0
Palmitoleic	C16:1
Palmitoleic	C16:2
Palmitoleic	C16:3
Palmitic	C16:0
Heptadecenoic	C17:1
Margaric	C17:0
Oleic	C18:1
Linoleic	C18:2
γ-Linoleic	C18:3
Stearic	C18:0
Gadoleic	C20:1
Arachidic	C20:0
Ericic	C22:1
Behenic	C22:0

From Fig. 4.5, it can be seen that the great majority of the lipids produced by the *Chlorella sp.* are 90% of C16 and C18 fatty acids, mainly palmitic (C16:0) and linoleic (C18:2) acids

with content exceeding 30% for each. This result was confirmed in the study of [59], in which they used the strain *Chlorella sorokiniana* to determine the different synthesized fatty acids, depending on the culture medium. They observed that *Chlorella sorokiniana* produced mainly unsaturated fatty acids such as oleic (C18:1), linoleic (C18:2), and linolenic (C18:3) acids. Chader et al. [59] also reported that the most dominant fatty acids are C16:0, C18:0, C18:1, C18:2, and C18:3, and this synthesis is independent of the culture medium used.

Concerning the quantity generated, it is shown that the percentage of these fatty acids (C16:0 and C18:2) hardly exceeded 23% for each one and was obtained in different culture media [59]. In the study of El Ouaer et al. [34], the percentage of lipids obtained was between 25 and 40% for these two types of fatty acids and in both Bold and 10% in each media. Thus, the production rate of fatty acids by the strain of *Chlorella sp.* selected in the study of El Ouaer et al. [34] exceeded that produced by *Chlorella sorokiniana* reported by Chader et al. [59].

However, the amount of linoleic acid produced by the Chlorella cultivated in the leachates appeared to be higher than that recorded in the Chlorella cultivated in the Bold medium. It can thus be concluded that the leachates act essentially on linoleic acid by increasing its production and inversely on palmitic acid, whose percentage appeared low compared to that recorded in the Bold medium.

Nevertheless, specify the solvents used for the extraction of lipids from Chlorella. Thus, chloroform has a strong affinity for polar lipids, mainly phospholipids, sterols and sphingolipids, and neutral lipids, mainly triglycerides, esters, esterified waxes sterols, and ketones. Hexane extracts mainly neutral lipids and is usually the solvent of choice for lipid extraction from Chlorella cultures because of its high affinity, yield, and low cost.

4.6 Physical–Chemical Processes for Leachate Treatment

4.6.1 Coagulation–Flocculation–Decantation

Coagulation–flocculation–decantation processes are physico-chemical processes aimed at eliminating suspended, colloidal, and dissolved matter likely to exist in water by transforming them and gathering them as more or less solid products called flocs. These flocs generated by complex physico-chemical action are comparable to those mentioned above in biofiltration generated because of intense microbial activity of interactions with the components of the water and a suspension. A separation operation is then carried out by decantation, flotation, and/or filtration systems.

This treatment constitutes a basic treatment applied to correct all or part of the defects of the water-related to inert particulate fractions such as silt, clay, organo-mineral colloids, or microbial fractions such as planktonic microalgae, microinvertebrates, in particular cysts of parasitic protozoa such as amoeba, Giardia, Cryptosporidium, etc.; bacteria, etc. It also ensures the elimination of the floculable fraction of organic matter, namely, macromolecules, in particular most of the humic acids responsible for color, certain heavy metals, and more normally the fraction of micropollutants associated with these suspended matter and colloidal macromolecules such as viruses, which are practically always carried by the suspended matter and colloids in the water. The colloids present in waste and natural water are recognized as usually being negatively charged because of imperfections in the crystal lattice and the ionization of peripheral chemical groups. To neutralize this overall negative surface charge, positive ions known as counter ions, present in the raw or added water (called coagulants), are attracted and come to form a layer around the colloid. Various theories have been proposed, mainly those of Helmholtz, Gouy-Chapman, and Stern, to explain this complex phenomenon of coagulation-

flocculation. Thus, coagulation results from the destabilization of colloidal particles by adding a chemical reagent called the coagulant, which brings to the medium multivalent cations, free or bound to an organic macromolecule known as cationic polyelectrolyte. These cations will be adsorbed and fixed on all the electronegative charges of the particles present in the water. However, flocculation is the phenomenon of agglomeration of the particles, previously “discharged”, into micro flocs by bridging, either by the hydroxides resulting from the hydrolysis of the mineral coagulant or by the macromolecules of the cationic polyelectrolyte. The micro flocs then aggregate into larger, settleable flakes, the floc. This flocculation can be enhanced and activated by the addition of another type of reagent, better known as a flocculant.

The formation of metal hydroxides results in the production of a large volume of sludge. This sludge can rarely be discharged into the natural environment and must be treated as mentioned above in the sludge treatment. Prechlorination and especially preozonation facilitate and speed up the coagulation-flocculation mechanisms.

This treatment by coagulation-flocculation and decantation is presented as an expensive and very efficient noble process [60, 61], its use for landfill leachate treatment can only be considered and used in somewhat special cases of leachate management.

4.6.2 Activated Carbon Filtration

The activated carbon process has been largely studied and used in recent decades for various water and wastewater treatments [62, 63]. It is a material essentially made up of carbonaceous material with a porous structure. It can be produced from any organic plant material rich in carbon, such as oak, beech, birch, poplar, pine or willow, coconut shells, etc. These materials are popular for making activated carbon. Bamboo, olive pits, coconut husks, and other similar plant sources also produce excellent-quality activated carbon. Once

activated physically or chemically by a carbonization-calcination-pyrolysis process or controlled oxidation following a thermal shock by airflow or water vaporization or a phosphoric acid or hydrochloric acid bath, macro- and microcells are created in the mass of material used, increasing the porosity. The exchange surface of the activated carbon or specific surface increases and can reach between 400 and 2000 m^2 g^{-1}. Thus, the carbon can be trapped by adsorption, and more precisely by physio-sorption, the molecules of gases for eliminating bad smells on the site of a wastewater treatment plant or a landfill and diversified micropollutants as in water intended for human consumption after preozonation or a noble industry such as the pharmaceutical industry activated carbon comes in two forms, namely, granular activated carbon (GAC) and fibrous activated carbon (FAC). Granular activated carbon (GAC) filters have granules up to 3 mm in diameter, have a longer life span and are lighter than fiber-activated carbon (FAC) filters. Fiber-activated carbon (FAC) filters comprise grains between 10 and 50 µm in size (100 times smaller than GAC). They are said to be easier to handle than GAC filters and have a faster adsorption capacity.

Thus, activated carbon is a clarifying purifier used for several centuries to filter water and air and for its medicinal properties to facilitate digestion in humans and prevent the production of gas. Filtration of activated carbon is currently recognized as the most widely used process for the purification of water intended for a specific use, such as drinking, pharmaceutical preparations, and laboratory analysis.

As technology has advanced, activated carbon has been used in the production of decontamination systems such as gas masks, kitchen hoods, and water clarification cartridges for culinary use to obtain purer water that is free of impurities and micropollutants.

Activated carbon filtration traps many elements as dissolved and gaseous compounds, such as chlorine and

derivatives, and various chemical residues, such as hydrocarbons and derivatives, but never does this filtration ensure the total removal of these residues. In a treatment process, the carbon filtration system must be placed at the end of the chain so that the organic waste does not saturate too quickly. However, the longer the water remains in reaction and contact with this activated material, the more efficient and safer the filtration will be. In water filtration, the quantity of activated carbon to be used depends greatly on the volume of water to be filtered and its pollutant load; thus, a mass of 500 g for 100 L of water to be treated seems to be the average ideal quantity.

Finally, this activated carbon filtration technique lends itself well as a complementary and refining treatment of leachates after a preliminary biological or chemical oxidation treatment. The regeneration of the carbon filter is required at a regular rate because of the rapid saturation during its operation. Microbial colonization contributes to the purifying power of the filter and is associated with its physical–chemical power. On average, the limit duration of activated carbon is 3 weeks. This frequency varies according to the level and load of pollution to be treated. The duration of 2 weeks seems to be the most effective. The quality of the leachates at the outlet of the carbon filters is often of an appreciable quality, which satisfies their discharge well in the receiving environments, their spreading, etc.

Activated carbon filtration is a complex and difficult technology that requires a certain amount of investigation to master well. It depends on many factors and is sensitive to temperature and humidity conditions, which can even lead to the release of trapped pollutants. Hence, an uncertain and sometimes variable efficiency would often require rigorous follow-up and permanent control, particularly in treatments of drug residues, pesticides, polluting harmful gases, and undesirable volatile organic compounds. A prechlorination (<5 mg Cl_2/L) or preozonation (< 10 mg O_3/L) step at low doses is often recommended at the beginning of the process

in the treatment of highly charged water intended for human consumption and, more recently, of landfill leachate.

4.6.3 Advanced Oxidation Processes by Ozonation, H_2O_2, and/or UV

Specific or advanced oxidation processes group together a set of physico-chemical treatment procedures designed to oxidize often organic and sometimes inorganic matter suspended or dissolved in loaded waters such as domestic, urban, or industrial effluent wastewater by oxidation via reactions with free hydroxyl radicals (HO^-) [64].

The best-known oxidation procedures include chemical processes based on ozone (O_3), hydrogen peroxide (H_2O_2), and/or UV light4 and, more rarely, on ultrasound (sonochemistry). Organic or semiorganic and inorganic contaminants, in suspension or dissolved, are converted to a large extent into stable inorganic compounds, mainly water and carbon dioxide, metal, or metalloid salts. It is therefore a faster process of transformation and destruction of matter, better known as mineralization. Oxidation processes aim to reduce the pollution and toxicity of a loaded effluent to where it can be safely drunk or reintroduced into the receiving environment. These processes alone cannot eliminate mineral and metallic toxins, which require complementary treatment, such as filtration on activated carbon, coagulation, flocculation, and decantation. These processes, however, can eliminate various recalcitrant pollutants during conventional water treatment, such as pesticide residues, antibiotics, other drug residues, and many endocrine disruptors. Advanced oxidation processes are based on the in situ (in water) production of hydroxyl radicals or free radicals (HO^-).

Under natural conditions, HO^-can be formed spontaneously in atmospheric water vapor under the effect of solar UV or ozone, but its strong reactivity makes it disappear almost immediately after its appearance. Free radical HO^- associated with other reactive oxygen species in a liquid such

as water, conditions a strong oxidation state that can be used directly for the transformation and fast degradation of the material in the water matrix. The efficiency of the process and, if correctly applied in water treatment and loaded liquids such as leachates, can reach and reduce the contaminant contents from several 100 ppm to less than 5 ppb and thus considerably reduce the COD and TOC.

Hydroxyl free radicals oxidize many organic compounds almost instantaneously with an oxidation rate 109 times higher than that of ozone, which is considered one of the most powerful oxidants in nature. They also react with C–C double bonds and attack aromatic rings, which are major components of many toxic, carcinogenic, and/or refractory mutagenic compounds [65–67].

Hydroxyl free radicals can be generated with one or more primary oxidants, such as ozone (O_3), hydrogen peroxide (H_2O_2), pure oxygen (O_2), chlorine and its derivatives (Cl_2, ClO^-, etc.), ultraviolet light, catalysts such as powdered titanium dioxide, and high-frequency ultrasound.

They are useful for destroying ecotoxic molecules or organic materials in the aqueous phase that are difficult to degrade by other means and the many emerging pollutants that are refractory or biorecalcitrant to biodegradation and that tend to bioaccumulate in the food web and the food chain, such as certain endocrine disruptors, contraceptive hormones, aromatic pollutants, most pesticides or their residues; the same is true for many petroleum compounds and derivatives and volatile organic compounds [68].

The treatment oxidizes nitrite, sulfite, cyanide, and bromate anions to other harmless compounds, such as NO_3^-, SO_4^{2-}, CO_2, and Br^-, respectively. In some oxidation models, trace metal elements may not be destroyed either, but the process may help remove some as a metal hydroxide (M [OH] x) precipitate. Finally, disinfection can also be achieved by destroying viruses, bacteria, Cyanophyceae, and all other

living organisms unless they are well protected in sufficiently resistant biofilm structures.

Oxidation processes have several advantages unequaled in water treatment, but they are not always perfect, since they can sometimes present shortcomings and disadvantages such as the need for careful and constant control of the oxidation reaction and its yield; likewise, this reaction is not selective and can generate even more dangerous elements and products, such as residual ozone molecules, UV, and various free radicals. These processes, therefore, require a great deal of technical expertise and involve significant costs.

To conclude, this last oxidation process, commonly known as an advanced oxidation process, creates powerful oxidizing agents, such as hydroxide (OH^-), but more precisely, as a neutral variant of the hydroxyl radical ($^\bullet OH$). Its oxidation potential is twice as strong as that of chlorine, a frequently used water disinfectant. Hydroxyl radicals are the driving forces behind many advanced oxidation processes. Ozone (O_3), hydrogen peroxide (H_2O_2), and ultraviolet light (UV) are often used in various combinations to produce $^\bullet OH$ in sufficient quantities to degrade various organic and inorganic residual compounds. This process can intensely reduce these compounds, from hundreds of parts per million (ppm) to a few parts per billion (ppb). Hydroxyl radicals ($^\bullet OH$) have disadvantages since they are nonselective; therefore, they attack most organic materials and release various compound intermediates. These residual compounds could react with the oxidants and mineralize into stable compounds. At last and despite this, this process of advanced oxidation process has proven its good usefulness and beneficial application for the removal of the large variety of compounds in water, in a dissolved or solid state.

4.6.4 Leachate Treatment by Reverse Osmosis

Reverse osmosis has become one of the most efficient processes to ensure the separation of solutes and solvents of

organic and inorganic, corrosive or inert nature from a fluid. The classical systems of wound membranes in cartridges or tubes have well-defined uses, such as in the food industry or some water/oil separation operations. Hence, for some industrial purposes, such as in landfill leachate treatment, the classical membrane systems are fragile. A solution to this problem has been made possible by new membrane technologies that employ the disk tube module system, the ROCHEM system, which is suited for this leachate treatment. Thus, the Schönberg site in Germany treats approximately 60 m^3/h of leachate in units using the Disk Tube Module process, 30–250 m^3/day with the ROCHEM system. Today, in Europe, many treatment sites use these new fast and very efficient techniques in leachate treatment. To ensure a further reduction in the leachate concentrate volume, the ROCHEM system integrates its concept of high-pressure reverse osmosis coupled with ultrafiltration and/or nanofiltration.

4.6.5 Thermal Processes: Evaporation

Several times above, the treatment of landfill leachate has become one of the greatest challenges in solid waste management. The reduction of leachate volumes has imposed the introduction of new techniques for the remediation of its strong and polluting liquid residues, such as evaporation. A maximum leachate concentration is sometimes needed, resulting in a significant reduction in leachate volumes. With advanced emerging technologies such as falling streams and forced circulation, new facilities ensure maximum concentrations and can reduce leachate volumes up to 100 times the treated leachate volume. The resulting leachate concentrate can be easily processed, neutralized, or stored.

Among the technologies proposed is a vacuum evaporation effluent treatment technology with mechanical vapor recompression coupled with reverse osmosis. This state-of-the-art technology is well adapted to the most efficient solution and allows the current standards of discharge into the natural environment, such as the one related to the

discharge of Kjeldahl nitrogen, to be reached. After osmosis, the quality of the discharge is such that it makes it possible to reuse the treated water in certain industrial processes. Similarly, to respond to energy and recovery issues, a vacuum evaporation leachate treatment process has been designed, coupled with the production of water for a motor or a biogas boiler. This technique requires a balance between the production of energy and the quantity of water to be evaporated to optimize energy valorization.

4.6.6 Treatment of Leachate by Photocatalysis

Landfill leachate is a complex liquid effluent with great variability in its physico-chemical characteristics, depending essentially on the landfilled waste and the level of the waste source. A good treatment of waste liquid, which is highly charged and foul smelling, therefore requires special attention. Among the wide range of solutions mentioned above is the procedure of leachate treatment by photocatalysis.

Photocatalysis is one of the advanced oxidation techniques and is another way of using photons to clean up air or water. Photons are absorbed by a photocatalyst, usually in a heterogeneous liquid−solid or gas−solid medium. Heterogeneous photocatalysis, as defined and detailed by Lacombe et al. [69], is based on the absorption of photons of energy equal to or greater than the band gap (Ec-Ev) by a semiconductor, which allows electrons to be transferred from the valence band (Ev) to the conduction band (Ec). Electronic gaps, commonly called holes (h^+), are thus generated in the valence band, and an oxidizing-reducing system is created. If their life allows it, electrons can be transferred to electron acceptors, while holes can be filled by electron donors.

Molecular oxygen O_2 is the most likely electron acceptor; it is reduced to the superoxide anion radical O_2^- or, depending on the pH, to its protonated form, the hydroperoxide radical HO_2^- ($pKa = 4.7$). These radicals are less reactive than the

HO^- radical toward organic compounds. Nevertheless, they can react with each other to form hydrogen peroxide, H_2O_2, or the hydroxyl radical HO^- by successive reductions. However, the most direct way of formation of the radical HO^- is the oxidation by a hole of H_2O or of an adsorbed OH^- ion. This radical, one of the most reactive and least selective, attacks most molecules by providing other radicals that react with O_2 [69].

Photocatalysis techniques have been used to treat urban wastewater, colored water from textiles, pharmaceutical and medical industries, and leachates. Some applications of photocatalysis have been used for their bactericidal and antiviral effects on wastewater, leachate, and self-cleaning materials, such as glazing. Performing this technique is usually good, and titanium dioxide (TiO_2) is the most widely used compound as a material for capturing light and solar energy [70].

Much research work is dedicated to the improvement of titanium dioxide effectiveness, either by improving its porosity, adsorption, and doping properties or by increasing its efficiency in visible light.

Advanced oxidation processes, or AOPs, offer several advantages unequaled in the treatment of water. Due to the remarkable reactivity of the HO- radical, which reacts immediately and indiscriminately with most aqueous pollutants, AOPs can be used in many scenarios (when and where many organic contaminants need to be removed simultaneously). They are useful for the destruction in the aqueous phase of ecotoxic molecules or organic materials that are difficult to degrade by other means, and many emerging pollutants are known to be refractory or biorecalcitrant to biodegradation.

Thus, photocatalysis can present many advantages for the degradation of many compounds recalcitrant to biodegradation in leachates and well-dimensioned systems. Photocatalytic mineralization of leachate pollutants is often

well suited for low loads (flow rates/low concentrations) and can be used in combination with other known treatment technologies, such as coagulation and flocculation. Photocatalytic reaction surfaces are in full development these days, with self-cleaning glazings and materials such as paints and varnishes, filters for eliminating bad smells (abatement of NOx and VOCs), and disinfection by the destruction of microorganisms carried in the liquids and air.

4.7 Conclusion

In conclusion, the treatment of landfill leachate is a priority, similar to the collection, landfilling, and treatment of waste, given their increasing volume on landfill sites and their highly polluting and toxic power for the natural environment. Many techniques and processes to treat these leachates are put on the market every day and are usually clean and costly. The performances also vary according to the process adopted, the nature of the waste collected, depending essentially on the standard of living of the population and the climate of the site, etc.

References

1. Teng C, Zhou K, Peng C, Chen W (2021) Characterization and treatment of landfill leachate—a review. Water Res 203:117525, ISSN 0043-1354. https://doi.org/10.1016/j.watres.2021.117525
2. Mojiri A, Zhou JL, Ratnaweera H, Ohashi A, Ozaki N, Kindaichi T, Asakusa H (2021) Treatment of landfill leachate with different techniques: an overview. J Water Reuse Desalin 11(1):66–96. https://doi.org/10.2166/wrd.2020.079
3. Yamamoto M, Kurihara I, Kawano S (2005) Late type of daughter cell wall synthesis in one of the Chlorellaceae, *Parachlorella kessleri* (*Chlorophyta, Trebouxiophyceae*). Planta 221(6):766–775
4. Chávez RP, Pizarro ECC, Galiano YL (2019) Landfill leachate treatment using activated carbon obtained from coffee waste. Eng Sanit Ambient 24:8330842. https://doi.org/10.1590/S1413-41522019178655
5. Christensen NTH, Kjeldsen P, Bjerg PL, Jensen DL, Christensen J, Baun A, Albrechtsen HJ, Heron G (2001) Biogeochernisfiry of landfill leachate plumes.

Appl Geochem I6:659–718

6. Białowiec A (2015) Transpiration as landfill leachate phytotoxicity indicators. Waste Manage 39:189–196

7. Kirmizakis P, Tsamoutsoglou C, Kayan B, Kalderis D (2014) Subcritical water treatment of lanfill leachate: application of response surface methodology. J Environ Manage 146:9–15

8. Billard H (2001) Waste storage center Operation

9. Scholz M (2006) Activated sludge processes in Wetland systems to control urban runoff

10. Akratos CS (2015) Book review: “Wetlands for Water Pollution Control,” 2nd edition, by Miklas Scholz (2015) Elsevier, Amsterdam, Netherlands. Environ Process 3 (Suppl 1):105–107. https://doi.org/10.1007/s40710-016-0170-2

11. Husson A, Stricker AE, Sénéchal M, Racault Y (2013) Synthesis document IRSTEA, Bordeaux Center, Adour Garonne Water Agency, France

12. Nora NF, Chong AMY, Wong YS (2002) Removal of tributyltin (TBT) by live and dead microalgal cells. Mar Pollut Bull 45(1):362–371

13. Travieso L, Canizares RO, Borja R, Benitez F, Dominguez AR, Dupeyron R, Valiente V (1999) Heavy metal removal by microalgae. Bull Environ Contam Toxicol 62(2):144–151

14. Kumudha A, Selvakumar S, Dilshad P, Vaidyanathan G, Thakur MS, Sarada R (2015) Methylcobalamin—a form of vitamin B12 identified and characterized in Chlorella vulgaris. Food Chem 170:316–320

15. Xie T, Sun Y, Du K, Liang B, Cheng R, Zhang Y (2012) Optimization of heterotrophic cultivation of *Chlorella sp*. for oil production. Biores Technol 118:235–242

16. Pore RS, Sorensen WG (1981) Kepone removal from aqueous solution by immobilized algae. J Environ Sci Health Part A: Environ Sci Eng 16(1):51–63

17. Przytocka-Jusiak M, Duszta M, Matusiak K, Mycielski R (1984) Intensive culture of *Chlorella vulgaris* as the second stage of biological purification of nitrogen industry wastewater. Water Res 18(1):1–7

18. Regel RH, Ferris JM, Ganf GG, Brookes JD (2002) Algal esterase activity as a bio measure of environmental degradation in a freshwater creek. Aquat Toxicol (Amsterdam, Netherlands) 59(3–4):209–223

19. Takashi H, Kiyoshi N, Shoichio K, Yotaro A, Akira Y, Yasunobu Y (2000) Chlorella vulgaris culture supernatant (CVS) reduces psychological stress-induced apoptosis in thymocytes of mice. Int J Immunopharmacol 22(11) :877–885

20. Mohan Viswanathan T, Hemapriya S, Sriga S, Bazeera Ferdhous P, Kathiresan T (2019) Oil extraction and compound analysis of microalgae chlorella vulgaris. Int J Recent Technol Eng (IJRTE) 8(4S4). ISSN: 2277–3878
21. Chouteau C (2004) Développement d'un biocapteur conductimétrique bienzymatique à cellules algales. Université Claude Bernard-Lyon 1
22. Gupta A, Gupta R (1997) A survey of plants for the presence of cholinesterase activity. Phytochemistry 46(5):827–831
23. Wang R, Hashimoto K, Fujishima A et al (1997) Light-induced amphiphilic surfaces. Nature 388:431–432. https://doi.org/10.1038/41233
24. Mouchet P (1986) Review of algal reactions to micropollutants, both mineral and organic; ecological consequences and possible industrial applications. Water Res 20(4):399–412
25. Stott R (2003) Fate and behavior of parasites in wastewater treatment systems. In: Handbook of water and wastewater microbiology. https://doi.org/10.1016/B978-012470100-7/50032-7
26. De-Bashan LE, Bashan Y (2010) Immobilized microalgae for removing pollutants: a review of practical aspects. Bioresource Technol 101(6):1611–1627
27. EL Halouani H, Picot B, Casellas C, Pena G, Bontoux J (1993) Élimination de l'azote et du phosphore dans un lagunage à haut rendement. Rev Sci Eau 6:47–61
28. Deviller G (2003) Traitement par lagunage à haut rendement algal (LHRA) des effluents piscicoles marins recyclés: évaluation chimique et écotoxicologiques. Université Montpellier II
29. Pittman J, Dean AP, Osundeko O (2011) The potential of sustainable algal biofuel production using wastewater resources. Biores Technol 102(1):17–25
30. Abdel-Raouf N, Al-Homaidan AA, Ibraheem IBM (2012) Microalgae and wastewater treatment. Saudi J Biol Sci 19(3):257–275
31. Priya M, Gurung N, Mukherjee K, Bose S (2014) Microalgae in removal of heavy metal and organic pollutants from soil. Microbial Biodegradation Bioremediation, Elsevier, Oxford, pp 519–537
32. Lim SL, Chu WL, Phang SM (2010) Use of *Chlorella vulgaris* for bioremediation of textile wastewater. Biores Technol 01(19):7314–7322
33. Luan TG, Jin J, Chan SMN, Wong YS, Tam NFY (2006) Biosorption and biodegradation of tributyltin (TBT) by alginate immobilized Chlorella vulgaris beads in several treatment cycles. Process Biochem 41(7):1560–1565

34. Nistor C, Emneus J (1999) Bioanalytical tool for monitoring polar pollutants. Waste Manage 19(2):147–170
35. El Ouaer M, Kallel A, Kasmi M et al (2017) Tunisian landfill leachate treatment using Chlorella sp.: effective factors and microalgae strain performance. Arab J Geosci 10:457. https://doi.org/10.1007/s12517-017-3241-4
36. El Ouaer M, Turki N, Ibrahim C et al (2022) Insights into the use of landfill leachate to grow *Chlorella sp.* for lipid and fatty acids production. Clean Techn Environ Policy. https://doi.org/10.1007/s10098-022-02460-5
37. Chang H, Quan X, Zhong N, Zhang Z, Lu C, Gang L, Cheng Z, Yang L (2018) High-efficiency nutrients reclamation from landfill leachate by microalgae *Chlorella vulgaris* in membrane photo bioreactors for biolipid production. Bioresour Technol 266:374–381, ISSN 0960-8524. https://doi.org/10.1016/j.biortech.2018.06.077
38. Lin L, Chan GYS, Jiang BL, Lan CY (2007) Use of ammoniacal nitrogen tolerant microalgae in landfill leachate treatment. Waste Manag 27(10):1376–1382
39. Saladini F, Patrizi N, Pulselli FM, Marchettini N, Bastianoni S (2016) Guidelines for emergy evaluation of first, second and third generation biofuels. Renew Sustain Energy Rev 66:221–227
40. Li-Beisson Y, Shorrosh B, Beisson F, Andersson MX, Arondel V, Bates PD, Baud S, Bird D, Debono A, Durret TP, Franke RB, Graham IA, Katayama K, Kelly AA, Larson T, Markham JE, Miquel M, Molin I, Nishida I, Rowland O, Samuels L, Schmid KM, Wada H, Welti R, Xu C, Zallot R, Ohlrogge J (2013) Acyl-lipid metabolism. Arabidopsis Book, vol 11, no e0161. Epub Jan 29, 2013. PMID: 23505340; PMCID: PMC3563272. https://doi.org/10.1199/tab.061
41. Gavrilescu M, Chisti Y (2005) Biotechnology-a sustainable alternative for the chemical industry. Biotechnol Adv 23(7–8):471–99. PMID: 15919172. https://doi.org/10.1016/j.biotechadv.2005.03.004
42. Demirbas MF, Balat M (2006) Recent advances on the production and utilization trends of biofuels: a global perspective. Energy Convers Manage 47:2371–2381. https://doi.org/10.1016/J.ENCONMAN.2005.11.014
43. Bélair V (2012) Développement de nouvelles techniques d'extraction des lipides à partir des microalgues en vue de leur valorization en biocarburant. Université Laval. Mémoire Faculté des études supérieures et postdoctorales de l'université Laval, Québec, 213 pages. Gavrilescu M, Chisti Y (2005) Biotechnology-a sustainable alternative for the chemical industry. Biotechnol Adv 23(7–8):471–99. https://doi.org/10.1016/j.biotechadv.2005.03.004. PMID: 15919172
44. Drabik D, Venus TJ (2017) Chapter 2.2.2—water use for biofuels in Europe A2—Ziolkowska, Jadwiga R. In: Peterson JM (ed) Competition for water resources. Elsevier, pp 14–159

45. Richards RG, Mullins BJ (2013) Using microalgae for combined lipid production and heavy metal removal from leachate. Ecological Modeling 249:59–67
46. Bogen C, Al-Dilaimi A, Albersmeier A, Wichmann J, Grundmann M, Rupp O, Lauersen KJ, Blifernez-Klassen O, Kalinowski J, Goesmann A, Mussgnug JH, Kruse O (2013) Reconstruction of the lipid metabolism for the microalgae *Monoraphidium neglectum* from its genome sequence reveals characteristics suitable for biofuel production. BMC Genomics 14:926
47. Nurachman Z, Hartini H, Rahmaniyah WR, Kurnia D, Hidayat R, Prijamboedi B, Suendo V, Ratnaningsih E, Panggabean LMG, Nurbaiti S (2015) Tropical marine *Chlorella sp*. PP1 as a source of photosynthetic pigments for desensitized solar cells. Algal Res 10:25–32
48. Gohain M, Chutia S, Deka D (2017) Microalga biomass production and oil extraction for algae biodiesel production—a review. J Energy Res Environ Technol (JERET) 4(1):53–57. p-ISSN: 2394-1561; e-ISSN: 2394-157X
49. Widjaja A, Chien C-C, Ju Y-H (2009) Study of increasing lipid production from freshwater microalgae *Chlorella vulgaris*. J Taiwan Inst Chem Eng 40(1):13–20
50. Zhu JB, Wang D, Gou YF, Wang WL, Gao JF (2014) Screening of microalgae rich in lipid in Xinjiang desert. China Oils Fats 39(12):68–71. https://doi.org/10.3969/j.issn.1003-7969.2014.12.020
51. Dos Santos RR, Moreira DM, Kunigami CN, Aranda DAG, Teixeira CMLL (2015) Comparison between several methods of total lipid extraction from Chlorella vulgaris biomass. Ultrason Sonochem 22:95–99
52. Feng Y, Li C, Zhang D (2011) Lipid production of Chlorella vulgaris cultured in artificial wastewater medium. Bioresour Technol 102(1):101–1 M. X.
53. Edmundson SJ, Wilkie AC (2013) Landfill lea” hate–a water and nutrient resource for algae-based biofuels. Environ Technol 34(13–14):1849–1857
54. Parsons S (2004) Advanced oxidation processes for water and wastewater treatment. IWA Publishing, Alliance House, London, England, 356 p
55. Wessler I, Kilbinger H, Bittinger F, Kirkpatrick CJ (2001) The biological role of nonneuronal acetylcholine in plants and humans. Jpn J Pharmacol 85(1):2–10
56. Lacombe S, Tran-Thi TH, Guillard C, Herrmann JM, Keller-Spitzer V, Keller N, Maurette MT, Pichat P, Pigot T, Pulgarin C, Rincon AG, Robert D (2007) Photocatalysis used against pollutants. L’environnement, l’actualité chimique - mai-juin 2007:308–309
57. Bigogno C, Khozin-Goldberg I, Boussiba S, Vonshak A, Cohen Z (2002) Lipid and fatty acid composition of the green oleaginous alga Parietochloris incisa, the richest plant source of arachidonic acid. Phytochemistry 60(5):497–503

58. Chisti Y (2013) Constraints to commercialization of algal fuels. J Biotechnol 167(3):201–214
59. Chader S, Mahmah B, Chetehouna K, Mignolet E (2011) Biodiesel production using *Chlorella sorokiniana* a green microalgae. Revue des Energies Renouvelables 14(1):21–26
60. El Fadel H, Merzouk M, Faouzi M, Chaouch M, Benlemlih M (2012) Biological and chemical treatment of the controlled public landfill leachate in Fez, Morocco. TSM 6:39–51. https://doi.org/10.1051/tsm/201206039
61. Khalil F, Bouaouine O, Chtioui H, Souabi S, Aboulhassan MA, Ouammou A (2015) Treatment of landfill Leachate by coagulation-flocculation. J Mater Environ Sci 6(5):1337–1342. ISSN: 2028–2508, CODEN: JMESCN
62. Ahmad A, Azam T (2019) Water purification technologies. In: Grumezescu AM, Holban AM (eds) Bottled and packaged water. Woodhead Publishing, pp 83–120. ISBN 9780128152720. https://doi.org/10.1016/B978-0-12-815272-0.00004-0
63. Johnson C (2014) Advances in pretreatment and clarification technologies, in comprehensive water quality and purification. In: Ahuj S (ed), vol 2, pp 60–74. Elsevier. ISBN 780123821836, https://doi.org/10.1016/B978-0-12-382182-9.00029-3
64. Elleuch L, Messaoud M, Djebali K, Attafi M, Cherni Y, Kasmi M, Trabelsi I, Chatti A (2020) A new insight into highly contaminated landfill leachate treatment using kefir grain pretreatment combined with Ag-doped TiO_2 photocatalytic process. J Hazard Mater 382:121119. https://doi.org/10.1016/j.jhazmat.2019.121119
65. Gogate P, Pandit AB (2004) A review of imperative technologies for wastewater treatment I: oxidation technologies at ambient conditions. Adv Environ Res 8:501–551
66. Gogate PR, Pandit AB (2004) A review of imperative technologies for wastewater treatment II: hybrid methods. Adv Environ Res 8:553–597. https://doi.org/10.1016/S1093-0191(03)00031-5
67. Martin GJO, Hill DRA, Olmstead IL, Bergamin A, Shears MJ, Dias, DA, Kentish SE, Scales PJ, Botté CY, Callahan DL (2013) Lipid profile remodeling in response to nitrogen deprivation in the microalga “Chlorella *sp.* (*Trebouxiophyceae*) and *Nannochloropsis sp.* (*Eustigmatophaceae*). Plos One 9(8):e103389. https://doi.org/10.1371/jounal.pone.0103389.
68. Brillasa E, Mur E, Sauleda R, Sànchez L, Peral J, Domènech X, Casado J (1998) Aniline mineralization by AOP’s: anodic oxidation, photocatalysis, electro-Fenton, and photoelectron-Fenton processes. Appl Catal B: Environ 16(1):31–42. https://doi.org/10.1016/S0926-3373[97]00059-3

69. Thangalazhy-Gopakumar S, Al-Nadheri WMA, Jegarajan D, Narayan Sahu J, Mubarak NM, Nizamuddin S (2015) Utilization of palm oil sludge through pyrolysis for bio-oil and biochar production. Bioresour Technol 178:65–69
70. Ursu A-V, Marcati A, Sayd T, Sante-Lhoutellier V, Djelveh G, Michaud P (2014) Extraction, fractionation and functional properties of proteins from the microalgae *Chlorella vulgaris*. Biores Technol 157:134–139

© The Author(s), under exclusive license to Springer Nature Switzerland AG 2024
S. Souabi, A. Anouzla (eds.), *Landfill Leachate Treatment Techniques*, Springer Water
https://doi.org/10.1007/978-3-031-63157-3_5

5. Physicochemical Treatment Approaches for Leachate Treatment

S. R. M. A. K. D. Rajapaksha[1], E. M. Y. C. Ekanayake[1] and K. G. N. Nanayakkara[1] ✉
(1) Department of Civil Engineering, University of Peradeniya, Peradeniya, Sri Lanka

✉ **K. G. N. Nanayakkara**
Email: nadeen@eng.pdn.ac.lk

Abstract
The generation of solid waste is anticipated to continue to increase globally. One of the most cost-effective methods for managing solid waste is landfilling, which is widely utilized. Leachate is a liquid that extracts dissolved and suspended materials as it flows through a landfill which is comprised of organic, inorganic, and heavy metal pollutants. As landfill leachate has a high level of toxicity, it is recommended to utilize truly effective techniques of purification to avoid contaminating the ecosystem. There are different methods available for treating landfill leachate which are based on biological processes (activated sludge, aerobic and anaerobic stabilization ponds and biological filters), physicochemical processes

(coagulation/flocculation, adsorption, electrochemical treatment, filtration and advanced oxidative processes) and natural attenuations (wetlands). The applications of treatment technique, however, have been constrained by a variety of criteria, including processing effectiveness, operational strategy, power requirements, and financial benefits. Among other factors, the relationship between landfill age and leachate composition should be taken into consideration when adopting the best technique. Even though biological processes are commended for their favorable, cost–benefit, and operational simplicity, effective utilization is only possible in landfills with BOD_5/COD ratio greater than 0.5 which shows higher biodegradability in the leachate. Moreover, temperature increases seem to facilitate faster stabilization of organic matter, resulting in a more recalcitrant leachate. Furthermore, the BOD_5/COD ratio decreases as landfill age, whereas the amount of non-biodegradable organic material increases in aged leachate. Excessive amounts of heavy metals in leachate may hinder the microorganisms responsible for ammonia removal. Thus, with the increase in landfills' age and ammoniacal nitrogen concentrations, conventional biological treatment approaches may become ineffectual in meeting the legislatively allowed threshold values for disposal into aquatic bodies. Due to bio recalcitrant substances that are difficult to be biologically treated, physicochemical treatment is the most suitable treatment for mature landfill. Hence, different treatments consisting of physicochemical approaches for leachate treatment were investigated and reviewed in this chapter.

Keywords Leachate treatment – Mature landfill – Physicochemical techniques

5.1 Introduction

Landfilling represents the prevailing approach and optimal available treatment methodology for solid waste management within developing nations owing to its technical viability, operational simplicity, minimal requirement for oversight, reduced technological complexity, and cost-effective operational expenses [6]. The generation of landfill leachate (LL) entrails a complex interplay of biogeochemical and hydrological reactions that functions as a mass transfer mechanism to establish a suitably elevated moisture level within the landfill. This process initiates liquid flow, which is primarily influenced by factors such as irrigation, precipitation, recirculation, surface runoff, snowmelt, decomposition of refuse, co-disposal of liquid waste, intrusion of groundwater, and the initial moisture content inherent to the landfill site [7]. According to the literature, the pollution content within a 1 m^3 of landfill leachate corresponds to the pollution content present in 100 metric tons of municipal wastewater [35]. Leachate composition is additionally affected by variables such as waste attributes, soil characteristics, landfill maturity, climatic conditions, liquid waste disposal practices, biochemical interactions within the landfill and at its interface with the surroundings, waste compaction levels as well as the complexity of landfill design [6, 34, 44]. Inadequate landfill designs and the application of excessive loads have contributed to the pollution of both soil and groundwater systems [36]. In the scenario of a heavily compacted landfill subject to an annual rainfall of 0.7 m per m^2 of waste coverage, the daily generation of leachate ranges from 0.278 to 0.487 dm^3. In the case of landfills characterized by low density, this quantity can be elevated by a factor of three [35]. LL typically exhibits significant amounts of organic pollutants, ammonia,

relatively high chemical oxygen demand (COD), heavy metals, toxic substances including xenobiotic organic compounds and inorganic salts. Additionally, there are refractory compounds present, such as fulvic-like fractions and humic substances, which are characterized by their resistance to breakdown and limited efficiency in biological treatment techniques such as anaerobic filtration, anaerobic lagoons and activated sludge [14, 34]. The proper management of non-biodegradable constituents within leachate is necessary to ensure their safe disposal. Failure to do so could potentially trigger eutrophication processes, leading to a decline in the levels of dissolved oxygen (DO) and an increase in the toxicity experienced by aquatic organisms within surface water systems [6]. Insufficiently drained leachate results in the buildup of leachate within the landfill. This situation can give rise to various consequences, such as the potential for landfill slopes to fail due to reduced shear strength of the waste materials and increased pore pressure. Landfill categories can be broadly categorized as aerobic, semi-aerobic, anaerobic, and anaerobic sanitary landfills. Within these classifications, semi-aerobic and anaerobic landfills are the most prevalent landfills encountered [6]. Nevertheless, the functionality of anaerobic landfills is constrained by high leachate concentrations, posing challenges for efficient treatment and adherence to required emission thresholds. Furthermore, the application of anaerobic landfills is constrained by emissions of greenhouse gases, predominantly encompassing carbon dioxide and methane, alongside the potential occurrence of fire-related incidents.

This chapter delves into the physicochemical approaches used in the treatment of landfill leachate, with a particular focus on advanced oxidation processes, adsorption, filtration, electrochemical treatments and coagulation, flocculation methods.

5.2 Characteristics of Landfill Leachate

The environment experiences a persistent influence from landfill leachate, which continues to be generated and discharged along with the ongoing release of biogas even long after the landfill has been closed. The leachate primarily contains larger molecular recalcitrant organic matter represented by compounds similar to fulvic and humic substances, small-molecule biodegradable organic matter like volatile fatty acids (e.g., butyric acid, propionic acid, and acetic acid), phenolic compounds, xenobiotic organic compounds (XOC), which include aromatic and aliphatic hydrocarbons, and phenoxy alkanoic acid herbicides which originate from industrial or household sources, heavy metals such as Hg, Pb, Cr, Cu, Zn, Cd, Ni and others. Among these, lead (Pb) is unique in that its concentration rises as pH increases, owing to its strong bonding with humic acids, forming stable compounds [34].

Generally, the qualities of the leachate are assessed based on its visual attributes, odor, total dissolved solids (TDS), pH, DO, total suspended solids (TSS), turbidity, hardness, biological oxygen demand (BOD_5), nitrate (NO_3^{-1}), ammonium nitrogen (NH_3–N), chloride content (Cl^{-1}), COD, BOD_5/COD ratio, total Kjeldahl nitrogen (TKN), total organic carbon (TOC), dissolved organic matter (DOM), total phosphorus (P), COD/TOC ratio, and sulfate (SO_4^{-2}) loading [6, 34, 44, 47, 50]. The elevated levels of NH_3–N and COD, which results from the breakdown of organic matter, are identified as the dominant contributors to acute toxicity in living organisms [47]. The molecular weight distribution stands as a critical parameter that provides insight into the composition of DOM within leachate [50]. In the characterization of landfill leachate, techniques

related to molecular weight include the utilization of membrane filtration with size exclusion chromatography (SEC) and various molecular weight cutoffs. Ultrafiltration (UF) membranes equipped with diverse molecular processing capacities have been employed to examine the distribution of molecular weights present in LL. Fourier transform ion cyclotron resonance mass spectrometry coupled with electrospray ionization (ESI-FT-ICR-MS) offers an exceptionally elevated level of resolution, enabling the differentiation of landfill leachate composition at the molecular scale [50].

The age of a landfill is a critical factor influencing the composition and characteristics of the leachate it produces. LL can be categorized into 4 groups based on the landfill age. As depicted in Table 5.1, leachate from landfills that are less than 5 years old (transition period) is termed “young leachate,” while “intermediate leachate” pertains to landfills aged between 5 and 10 years (acid formation period), “stabilized leachate” refers to leachate aged between 10 and 20 years (methane fermentation) and “old leachate” refers to leachate originating from landfills older than 20 years [33, 50]. During the acidogenic phase, the proportion of fatty acids within the organic content can attain levels ranging from 80 to 95%. A reduction in the pH of the leachate could be observed due to this significant presence of fatty acids. This results in the pH of leachate to be in the range between 4.5 and 7.5 [33]. The reduced pH as a result could increase the solubility of various compounds which includes heavy metals.

Table 5.1 Leachate characteristics during landfill lifetime [16, 33–35, 44, 48]

Parameters	Unit	Young	Intermediate	Stabilized	Old
Age	Years	< 5	5–10	10–20	> 20
Period		Transition	Acid formation	Methane fermentation	Final maturation

Parameters	Unit	Young	Intermediate	Stabilized	Old
Stability		Unstable	Moderately stable	Stable	
Color		Light yellow			Dark black–green
pH		3–7	6–8	> 7.5	> 7.5
Alkalinity		1000–20,000	500–6000	–	–
Conductivity	(μS/cm)	2000–50,000	1000–15,000	–	–
Organic compound		80% VFA*	5–30% VFA + Humic and Fulvic acids	Humic and Fulvic acids	Humic and Fulvic acids
BOD	mg/L	2000–50,000	500–15,000	50–1000	< 300
COD	mg/L	4000–90,000	1000–30,000	1000–5000	< 3000
TSS	mg/L	200–2000		100–400	> =72.33
TDS	mg/L	10,000–25,000	5000–10,000	2000–5000	< 1000
BOD/COD		0.5–1.0	0.1–0.5	< 0.1	< 0.1
TOC/COD		< 0.3	0.3–0.5	> 0.5	> 0.5
Heavy metals	mg/L	> 2	< 2	–	< 2
SO_4^{2-}	mg/L	300–4000	100–2000	20–200	< 100
P	mg/L	50–500	5–200	–	< 20
Cl	mg/L	500–6000	200–4000	50–500	< 200
Zn^{2+}	mg/L	50–400	20–200	5–50	< 20
Fe^{3+}	mg/L	200–3000	200–2000	50–500	< 200
Ca^{2+}	mg/L	1000–7000	200–4000	100–500	< 400
Na^{+}	mg/L	1000–7000	200–3000	50–500	< 200
Mg^{2+}	mg/L	300–3000	200–2000	50–500	< 200

Parameters	Unit	Young	Intermediate	Stabilized	Old
TKN	mg/L	500–4500	400–2000	50–2000	< 2000
NH_3–N	mg/L	< 400	NA	> 400	–
NH_4^+–N	mg/L	500–4500	–	–	< 1500
COD/NH_4^+–N(−)		5–10	3–4	< 3	
Kjeldahl nitrogen	mg/L	100–200	NA	NA	
Biodegradability		Important	Medium	Low	
Molecular size		Low molecular weight organics		High molecular weight organics	

The progression of waste transformation through successive aerobic, acidogenic, methanogenic, and stabilization phases [44] induces waste decomposition. The primary contaminant found in leachate is ammonia nitrogen. It appears in recent landfill sites due to the breakdown of organic compounds, leading to the deamination of amino acids [48]. Consequently, the aging process leads to a decline in BOD_5 content. This results in enhanced leachate stability due to the presence of refractory compounds, such as humic and fulvic acids, coupled with a decrease in the BOD_5/COD ratio. The anaerobic fermentation of the organic fraction of the waste generates fatty acids, contributing to elevated BOD_5 levels and low pH in young leachate [44]. Throughout the methanogenic stage, bacteria anaerobically convert the produced acids into carbon dioxide and methane, leading to a near-neutral pH level [44]. With ongoing landfill aging, the reduction of carboxylic acids results in a decrease in the BOD_5/COD ratio, while the non-biodegradable organic matter rises in aged leachate. Additionally, the concentrations of heavy metals tend to decrease as

leachate ages, due to the increase in pH which reduces the metals' solubility. Consequently, the effectiveness of biological treatment processes for aged leachate decreases. The coloration of aged leachate is characterized by a deep black-green hue, primarily attributed to the oxidation of ferrous iron and presence of humic substances. This differs from the yellowish tint observed in younger leachate [33], indicative of a higher concentration of biodegradable constituents. The humic substances, being recalcitrant compounds, exhibit limited biodegradability. Table 5.1 depicts the leachate characteristics during landfill lifetime.

5.3 Treatment Techniques

Due to the significant danger associated with highly contaminated leachate, it's necessary for it to adhere to stricter quality regulations when it comes to directly releasing the leachate (Talalaj et al. 2019). To comply with these quality standards, it is essential to treat the leachate on-site, and the choice of a suitable technology becomes even more critical. The determination of a treatment method for LL should be approached with careful consideration, only after gaining a comprehensive understanding of their composition and concentrations. Conventional strategies employed for the treatment of landfill leachate include practices such as internal recirculation within the landfill site and external conveyance to municipal sewage treatment facilities. The technique of landfill leachate recirculation has been extensively utilized in past years due to its practicality and cost-effectiveness [50]. The frequency and quantity of recirculation are key factors in determining the efficiency of this approach. Contemporary practices include a diverse range of leachate treatment methodologies, including natural processes (wetlands and lagoons), biological processes (biological filters, aerobic and anaerobic

stabilization lagoons, activated sludge and bioreactors), physical–chemical processes (advanced oxidation process, adsorption, coagulation/flocculation, electrochemical treatment, filtration), and hybrid processes [6, 16, 33]. Figure 5.1 demonstrates the different treatment technologies applied for LL treatment.

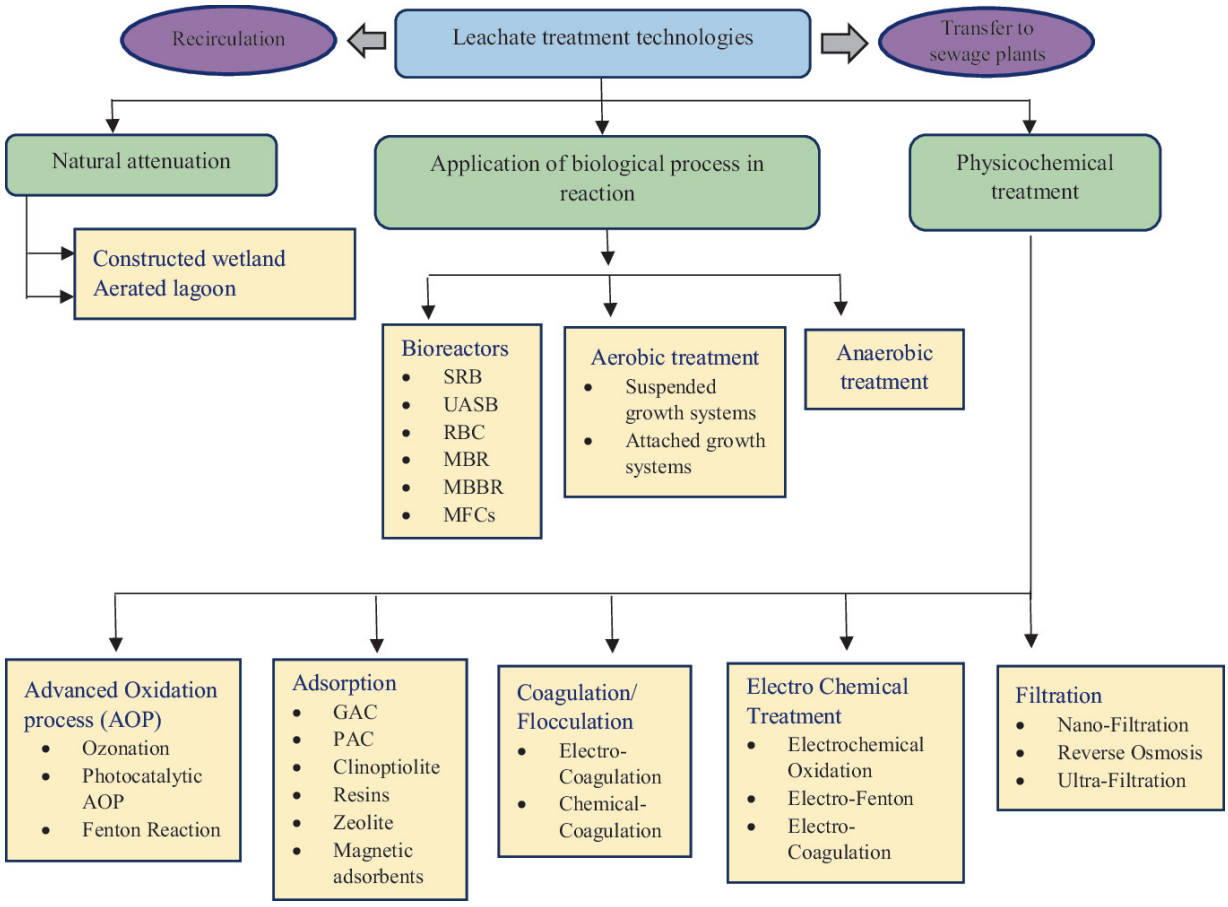


Fig. 5.1 Different leachate treatment technologies [6, 39, 40, 50]

The treatment process is varying with the leachate's strength characteristics. The diagram in Fig. 5.2 illustrates the parameters that need to be assessed when selecting a treatment method for landfill leachate.

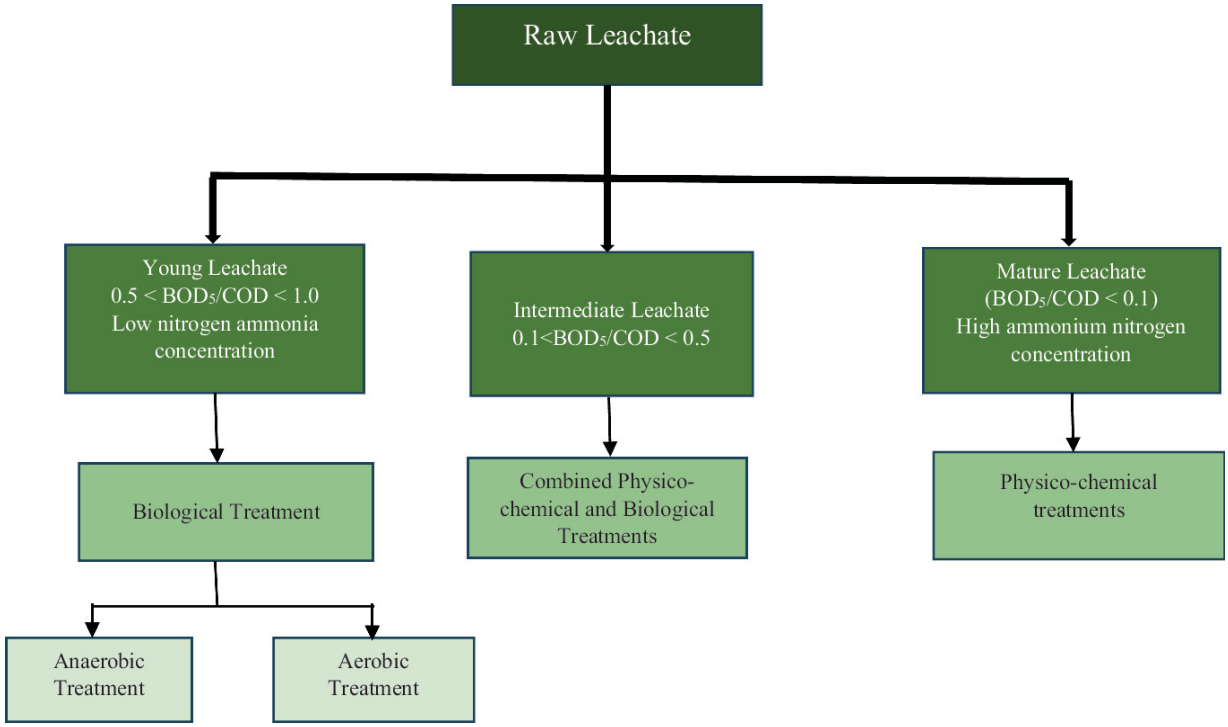


Fig. 5.2 Selection of best suitable treatment for LL [16, 49]

The chemical treatment of leachate often involves a standard practice of neutralization through the addition of acids or bases. As per the findings of research by Reshadi et al. [44], biological methods are best suited for treating early-stage leachates with substantial organic content, whereas physical–chemical techniques are more preferable for addressing mature leachates characterized by reduced biodegradability and elevated ammonia concentrations.

5.4 Physicochemical Treatment Techniques

It has been discovered that the most extensively researched aerobic techniques for treating leachate on a global scale consist of aerobic lagoons, accounting for 21%, followed by up flow anaerobic sludge blanket at 18%, activated sludge at 17%, and sequencing batch reactors at 15%. Membrane bioreactors contribute 8% to this list [48]. These approaches are likely the most effective and economical

methods for nitrogen removal from leachate. However, the efficacy of biological treatment is hindered by the presence of organic compounds or certain toxic substances that are resistant to biodegradation. Because of that, eliminating COD from aged leachate poses a challenge for traditional biological treatments. Hence, Physical–chemical methods are frequently employed when the effectiveness of biological processes is hindered by the presence of considerable refractory compounds in the leachate [51]. Different Physical–chemical approaches which are mainly used for the leachate treatment are described in this review.

5.4.1 Advanced Oxidation Processes (AOPs)

AOPs represent a highly promising approach due to their ability to rapidly break down various pollutants, leading to a substantial increase in leachate biodegradability [29]. Despite their higher efficiency in oxidizing organic pollutants and improving wastewater biodegradability, AOPs are considered a clean technology as they generate minimal concentrated residues. In general, AOPs are made up of two essential processes: the generation of highly reactive free radicals and the chemical interactions of these radicals with organic molecules found in water [34]. These free radicals effectively oxidise organic substances throughout the oxidation processes, changing them into more easily biodegradable compounds. As a result, the degraded compounds can undergo further oxidation, ultimately achieving complete mineralization. These commonly utilized reactive oxygen species (ROS) in different AOP types comprise hydroxyl radicals (•OH), sulfate radicals ($SO_4^{\cdot-}$), superoxide anions ($O^{\cdot-}$), and singlet oxygen ($^1O_2$) [29]. Among them, hydroxyl radicals (·OH) are the primary oxidizing agents utilized in AOPs because of their outstanding attributes. These include rapid

reaction rate constants ranging from 10^8 to 10^{10} M^{-1} s^{-1}, high oxidation potential and non-selective oxidation of organic matter. In general, a higher oxidation potential indicates stronger oxidants, leading to increased oxidation efficiency. Table 5.2 depicts the oxidation potentials of various oxidizing agents (E^0) (V).

Table 5.2 The oxidation potentials of various oxidizing agents (E^0) (V) [34, 50]

Oxidant/radical species	**Symbols**	**Oxidation potentials (E^0) (V)**
Flourine	F	3.06
Hydroxyl radical	·OH	2.8
Oxygen (diatomic)	O_2	2.42
Sulfate radical	$\cdot SO_4^-$	2.7
Ozone	O_3	2.07
Persulfate anion	$S_2O_8^{2-}$	2.01
Hydrogen peroxide	H_2O_2	1.78
Hydroperoxide radical	$\cdot HO_2$	1.7
Hypochlorite	ClO^-	1.49
Chlorine	Cl_2	1.36
Chlorine dioxide	ClO_2	1.27
Oxygen (molecular)	O	1.23
Superoxide radical	$\cdot O_2^-$	− 0.2

Besides utilizing stronger oxidants, the effectiveness of AOPs can be enhanced by adjusting the operating parameters to optimize the generation of free radicals. Techniques such as the incorporation of initiators in the process of treatment and pH adjustment can play a major role in increasing the production of free radicals, thereby improving the overall efficiency of the AOPs. These strategies enable better control and optimization of the

oxidation process for effective treatment of organic pollutants [34].

AOPs have the capability to transform large molecular organics into nontoxic small molecular organic substances or directly break them down into H_2O and CO_2 [50]. The fundamental reaction in AOPs is represented by Eq. (5.1).

$$\text{Oxidant} + \text{organic compound} \rightarrow \text{oxidation byproducts} \rightarrow CO_2 + H_2O \quad (5.1)$$

Researchers have studied various AOPs for treatment of leachate, which can be classified into different types. These include electrochemical techniques, ozone and chemical based processes, miscellaneous/other processes, and photochemical methods. Among the available AOPs, the Fenton process has been most employed due to its advantages and efficiency over other AOPs [36]. Figure 5.3 depicts the different Advanced oxidation processes for leachate treatment.

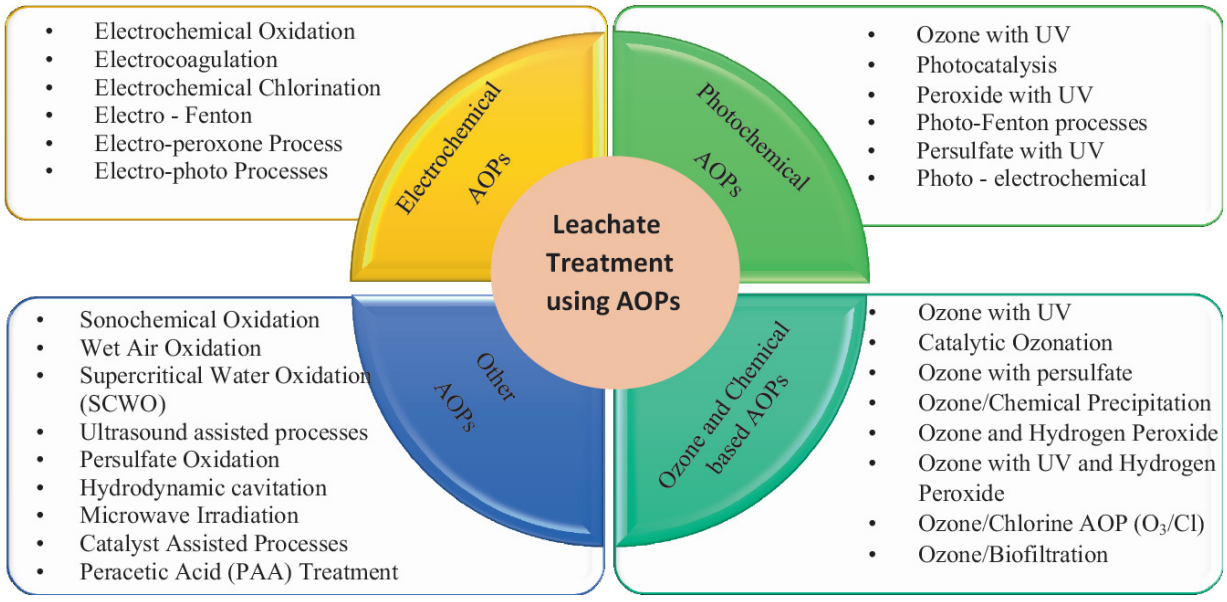


Fig. 5.3 Advanced oxidation processes for leachate treatment [25, 29, 40, 43]

Fenton-Related AOPs

Extensive research has been conducted on Fenton processes as a viable method for efficient LL treatment. These studies primarily focus on maximizing the removal percentage of COD and identifying the most favourable operational parameters for optimal performance. The COD removal rates in the Fenton process typically range from 35 to 90%. The pH and Fenton reagent dosages are main factors affecting dissolved organic matter (DOM) removal [50]. The Fenton process begins with the catalysis of hydrogen peroxide (H_2O_2) by Fe (II) to produce active oxidants, including hydroxyl radicals (·OH), which cause the degradation of organic compounds [36].

The process of Fenton oxidation is highly complex, involving multiple simultaneous chemical reactions. The essential reactions that are consistent across all Fenton systems (inorganic reactions) can be simplified as follows [36]:

Reactions	**k (M^{-1} s^{-1})**	
$Fe^{2+} + H_2O_2 \rightarrow Fe^{3+} + {}^{-}OH + OH^{\cdot}$	63	(2)
$Fe^{3+} + H_2O_2 \rightarrow Fe^{2+} + H^{+} + HO_2^{\cdot}$	0.01	(3)
$^{\cdot}OH + H_2O_2 \rightarrow HO_2^{\cdot} + H_2O$	2.7×10^7	(4)
$HO_2^{\cdot} \rightarrow O_2^{\cdot -} + H^{+}$	1.58×10^5	(5)
$O_2^{\cdot -} + H^{+} \rightarrow HO_2^{\cdot}$	1×10^{10}	(6)
$^{\cdot}OH + Fe^{2+} \rightarrow Fe^{3+} + OH^{-}$	3.2×10^8	(7)
$HO_2^{\cdot} + Fe^{2+} (+H^{+}) \rightarrow Fe^{3+} + H_2O_2$	1.2×10^6	(8)
$HO_2^{\cdot} + Fe^{3+} \rightarrow Fe^{2+} + H^{+} + O_2$	3.1×10^5	(9)
$O_2^{\cdot -} + Fe^{2+} (+2H^{+}) \rightarrow Fe^{3+} + H_2O_2$	1×10^7	(10)
$O_2^{\cdot -} + Fe^{3+} \rightarrow Fe^{2+} + O_2$	5×10^7	(11)
$^{\cdot}OH + {}^{\cdot}OH \rightarrow H_2O_2$	4.2×10^9	(12)

Reactions	k (M^{-1} s^{-1})	
$HO_2^{\cdot} + HO_2^{\cdot} \rightarrow H_2O_2 + O_2$	8.3×10^5	(13)
$^{\cdot}OH + HO_2^{\cdot} \rightarrow O_2 + H_2O$	1×10^{10}	(14)
$^{\cdot}OH + O_2^{\cdot -} \rightarrow O_2 + OH^-$	1×10^{10}	(15)
$HO_2^{\cdot} + O_2^{\cdot -} (+H^+) \rightarrow H_2O_2 + O_2$	9.7×10^7	(16)

AOPs hold significant advantages over conventional treatment methods such as not needing additional energy inputs, simple operation, minimal maintenance, rapid reaction rates, the presence of energy-efficient homogeneous reactions and potential for complete mineralization [3, 50]. Nevertheless, there are certain drawbacks that restrict the widespread use of this technique. These drawbacks include the necessity for substantial amounts of chemical reagents, leading to higher operational expenses, a limited operational pH range (typically between pH2 and 3), and the creation of iron-containing sludge, which contributes to secondary pollution [29, 50]. Furthermore, the reliance on energy-intensive resources such as ozonisers, ultraviolet light, and ultrasounds, along with the significant cost associated with reagents are another major drawback [3]. To address these limitations, Fenton-like processes have been explored for treating LL, incorporating both homogeneous and heterogeneous approaches. Additionally, energy-augmented Fenton processes such as photo-Fenton and electro-Fenton have been introduced to enhance the efficiency of landfill leachate treatment [50].

O_3-based AOPs

Ozonation at higher pH: Ozonation is a well-established ozone-based AOP that employs ozone (O_3) alone for the oxidation of substances in water Ozone undergoes a series of complex reactions with water molecules., generating

superoxide radicals ($O_2^{\cdot -}$) and hydroxyl radicals (OH·) [34]. The efficiency of ozonation could be significantly elevated when operating at higher pH levels, typically in the range of pH8–9. Formation of hydroperoxide ions (HO_2·) occur due to the reaction between O_3 and hydroxyl ions (OH^-) in the aqueous solution that is stimulated in this pH range. These resulting ions serves as initiators in the production of hydroxyl radicals (OH·).

When comparing to Fenton processes, O_3-based methods exhibit higher oxidative potential ($E0 = 2.08$ V) and reduced sludge generation for treating landfill leachate. O_3 has the capability to transform macromolecular substances into biodegradable compounds, resulting in improved biodegradability. These techniques operate through two distinct mechanisms [11]. The initial mechanism relies on direct electrophilic attacks of resilient contaminants, while the second mechanism involves indirect attack via the production of hydroxyl radicals (OH·), as depicted in Eq. (5.17) [31]:

$$3O_3 + H_2O \rightarrow 2\,^{\cdot}OH + 4O_2 \tag{5.17}$$

O_3 dose and ozonation time are the primary factors that significantly affect the treatment of LL. Typically, elevated O_3 doses correspond to heightened removal efficiency of DOM during the ozonation process. Removal efficiencies for humic acid (HA) and fulvic acid (FA) can reach levels as high as 88% and 83.3%, respectively [50].

The effective treatment of hazardous leachate comprises modifying the molecular structure of organic compounds and converting them into biodegradable substances by utilizing ozone's strong oxidative capabilities. However, ozone based AOPs encounter restrictions due to the inefficiency of transferring ozone from the gas phase to the liquid phase [25]. Additionally, there's a risk of ozone

potentially leaching into the surrounding environment, causing potential negative impacts. Given the complicated composition of leachate with a high organic content, substantial ozone dosing is necessary to attain the desired treatment efficiency. Nonetheless, the considerable energy consumption and substantial ozone emissions largely confine the utilization of ozone based AOPs to post-treatment or pre-treatment stages for leachate [29].

UV-Based AOPs

UV irradiation is recognized as an environmentally friendly and effective method for activating peroxides to form active oxidative species. These specific AOPs are progressively adopted for landfill leachate treatment, aiming to remove pollutants and enhance quality through the generation of extremely reactive free radicals like $SO_4^{\cdot -}$ and ·OH. Among UV-based AOPs, the UV/H_2O_2 process stands out as the most commonly employed technique in both laboratory and prototype-scale investigations due to its rapid oxidation rates, straightforward operational processes, and absence of sludge production.

Approaches centered around the sulfate radical ($SO_4^{\cdot -}$), like UV/peroxomonosulfate (HSO_5^-) and UV/persulfate ($S_2O_8^{2-}$), have also gained significant attention. $SO_4^{\cdot -}$, possessing a high standard redox potential between the range of 2.5–3.1 V, exhibits strong reactivity towards organic matter with rate constants in the range of 10^5–10^9 M^{-1} s^{-1} [50].

The utilization of TiO_2 for photocatalytic degradation has resulted in considerable interest for landfill leachate treatment owing to its non-toxic nature, affordability, environmental compatibility, and robust chemical stability. However, TiO_2's wide band gap only allows it to utilize UV

light, which constitutes a small fraction of sunlight (less than 5%). The exploration of visible light utilization in the Titanium dioxide (TiO_2) photocatalytic process is a pivotal area of study in the development of photocatalytic technology [50].

5.4.2 Adsorption

A multi-component fluid combination is drawn to the surface of a solid adsorbent and then forms attachments through a physical or chemical connection via the process of adsorption [46]. Adsorbent is the material that provides the solid surface whereas adsorbate is the material that is being removed from the liquid phase [47]. In a broad context, a wide range of materials is available for utilization as adsorbents. These materials consist of, but are not restricted to, the following categories: natural materials, synthetic materials like zeolites and resins, biological adsorbents like bacterial biomass and chitosan, enhanced natural materials such as activated alumina and activated carbon, agricultural and industrial residuals and by-products such as red mud and fly ash, and other diverse options [17]. The literature presents numerous classification systems. Table 5.3 demonstrates a simplified categorization of materials commonly employed as adsorbents [44].

Table 5.3 Simplified categorization of adsorbents

Adsorbents	Material
Natural materials	Clay, sawdust, bio-sorbent, chitin
Industrial by-products	Ash, tailings, sludge
Activated Carbon	From coal, cobs, wood, residues, nutshells
Inorganic–based materials	Activated alumina, activated bauxite, porous oxides

Adsorbents	Material
Advanced materials	Magnetic adsorbents, aerogels and xerogels, nanomaterials, metal–organic-frameworks

Activated Carbon

Among the various types of adsorbents, the adsorbent that stand out the most is activated carbon. Activated cardon is extensively employed due to its remarkable capacity for eliminating a diverse array of pollutants from both liquid and gas phases. This preference can be attributed to its expansive surface area, favorable porosity, thermal resilience, resistance to acidic and basic conditions, economical cost, and the capability to regulate pore structure [8, 44]. Utilizing activated carbon (AC) for the adsorption of pollutants in column setups yields more effective reductions in Chemical Oxygen Demand (COD) levels compared to chemical treatment approaches [47]. In the context of treating landfill leachate, activated carbon finds application in both granular (GAC) and powder (PAC) forms, typically possessing average particle sizes of approximately 0.2–5 mm and 15–25 µm, respectively. Nevertheless, granular activated carbon (GAC) is more commonly employed due to its advantage over powdered activated carbon (PAC), as GAC doesn't necessitate coagulation or flocculation for precipitation or separation, as is the case with PAC [38]. A wide range of carbon-rich materials can be utilized as feedstock for producing activated carbon. This includes materials like coal, peat, wood, nutshells, lignite, and tobacco [47]. As per literature, approximately 40% of commercial activated carbon originates from coal sources [47]. The primary constraints related to the application of AC in the leachate treatment procedure arise from the higher manufacturing expenses and the costly carbon-based substances [7]. Thus, Recently, as a result of the vast amount of landfill wastes that are

produced unconventional adsorbents have been created for the treatment of LL. Some of these adsorbents include activated carbons that is produced from bark husk, bamboo dust, chitin, lignite, coir pith, maize cob, palm fiber, palm shell, palm stone, rice husk, peat, pinewood, sago waste, sugarcane bagasse, tea leaves, and sawdust [14, 33, 47]. When compared with traditional activated carbons these unconventional materials are easily supplied at a lower cost and could be chemically modified. Nevertheless, the primary constraints affiliated with the utilization of activated carbon (AC) in the leachate treatment process stem from the increased production expenses and the costliness of carbon-rich materials. Activated carbons are typically produced using a continuous two-stage procedure involving activation and carbonization [33, 44]. The initial phase, carried out through pyrolysis at high temperatures, establishes the initial porosity, while the subsequent stage further refines the pore structure. Before this process, a preliminary treatment step is applied. This step encompasses activities such as crushing, milling, and sieving, aimed at achieving an appropriate size, which is crucial for the subsequent stages to be effective. The activation procedure is categorized into two types: chemical activation and physical activation. In the physical approach, the carbonized substance is exposed to an oxidizing gas, often heated steam or CO_2 (400–1000 °C). On the other hand, chemical activation involves blending the carbonized material with chemical activating agents, accompanied by heating in a non-reactive environment. Chemical activation demands lower temperatures and results in a product with a larger surface area and improved porosity compared to physical activation [44].

The adsorption process taking place on the adsorbent surface is frequently described by two proposed mechanisms. These are sorption through ion exchange, as

depicted by Reaction 5.1, and complex formation involving the negatively charged sites on the adsorbent, as depicted by reaction 2, This is particularly applicable in the context of activated carbon [33]

$$\text{Reaction 1: } \mathrm{xCOH + M^{y+} \rightarrow (CO)_x M^{(y-x)+} + xH^+} \tag{5.18}$$

$$\text{Reaction 2: } \mathrm{xCO^- + M^{y+} \rightarrow (CO)_x M^{(y-x)+}} \tag{5.19}$$

Biochar

In the past few years, Biochar (BC) has frequently been utilized as an adsorbent in its original form with minimal adjustments after treatment. It is typically less demanding in terms of resources, energy, and investment during the production process. The essential factors governing the sorbent properties of biochar include the temperatures used in pyrolysis, the duration of stay in the reactor, the rate of heating, and the specific kind of raw material employed. The efficacy of biochar in removing pollutants relies on specific surface area (SSA) of the adsorbent, which can vary from around 0.77–1160 $m^2\ g^{-1}$. The high temperatures used in the pyrolysis process generate a product that proves efficient as an adsorbent for organic contaminants in soil due to its expanded surface area, hydrophobic nature, and microporosity. Biochar produced at lower temperatures are better suited for eliminating inorganic or polar organic contaminants with oxygen functional groups through electrostatic forces [35].

Zeolite

In recent times, the utilization of naturally occurring minerals as absorbents for leachate treatment has proven to be efficient. One of the promising characteristics that sets zeolites apart from other porous materials is their

diverse range of pore sizes and shapes. Additionally, zeolites possess other properties, like intracrystalline pores and ion exchange capabilities, which create various robust acidic sites, accommodate molecules of different sizes, and act as active centers for metal-catalyzed reactions. These attributes have contributed to their extensive use in various industries. Zeolites containing large empty cages or spaces in their structures are especially valuable as inorganic cation exchangers. Significant cations like Na^+, Br^+, K^+, Ca^{2+}, and Mg^{2+}, as well as cationic groups like H_2O, NO_3^-, CO_3^{2-}, and NH_4^+ can be accommodated due to the structure of zeolites. Moreover, zeolites exhibit properties like ionizing radiation, high resistance to temperature, excellent resilience with the environment, and strong selectivity in cation exchange. Further, their enhanced adsorption and ion exchange capabilities, rendering them more attractive due to their simplified operational procedures and cost-effectiveness [4]. As per the research conducted by Aziz et al. [5] utilizing activated zeolite leads to enhanced removal percentages for COD, NH_3–N, and color. The removal rates increased from 24.3, 55.8, and 73.8% (with raw zeolite) to 46.3, 75.9, and 91.4%, respectively.

Magnetic Adsorption

Among the array of advanced adsorption techniques, magnetic adsorption has gained considerable attention as a novel technology. This interest can be attributed to its advantageous properties including low toxicity, chemical stability, and straightforward synthesis [44]. Additionally, magnetic adsorption has the potential to accelerate separation processes and enhance treatment plant capacities. In this context, iron oxide (Fe_3O_4) has emerged as a particularly well-suited material for this purpose.

Copper oxide (Cu_2O) is another promising compound capable of conferring magnetic properties to adsorbents. Also, a significant advantage of both iron and copper oxides is their ability to retention of magnetic characteristics during the thermal regeneration process. A magnetic adsorbent typically comprises three primary components: the shell, the magnetic core and an additional layer formed during the modification or functionalization process [44]. In many instances, the magnetic core is composed of iron oxides which is crucial for later separating of the adsorbent from wastewater [52]. The ideal protective layer for iron oxide is Silica (SiO_2) due to the ease of functionality and chemical stability. The final stage involves adding a tertiary layer. Examples of such materials include graphene oxide, amino groups, and thiols [44]. These latter processes serve the purpose of shielding the magnetic core from oxidation and erosion, while also providing additional active sites for the adsorption of pollutants. Furthermore, nanomaterials with diverse shapes (like wires, tubes, rods) have exhibited a range of advantages when employed as magnetic adsorbents. Research demonstrated the creation of magnetic adsorbents through the magnetization of carbon nanotubes with iron oxide, followed by functionalization with calcium. This process yielded a SSA of 160 $m^2\ g^{-1}$ and demonstrated remarkable efficacy in eliminating substances like humic acid (HA) as well as other contaminants including heavy metals, organic pollutants, dyes, pesticides, and radioactive elements [44, 52]. Furthermore, this adsorbent is capable of being regenerated and employed repeatedly. The effectiveness of magnetic adsorbents in removing pollutants fluctuates based on alterations in their structural components and the specific contaminants they target. Additionally, various factors, including process temperature, ionic strength, the isoelectric point of the adsorbent, the existence of other

pollutants or compounds, and pH, exert a significant influence [44].

5.4.3 Filtration

Filtration has gained attention as a viable method for leachate treatment because of its ability to selectively remove a range of pollutants while allowing the passage of smaller molecules. It is widely used in various stages of leachate treatment, including pre-treatment, as a standalone process, or as part of a multi-step treatment train [15]. These membrane-based technologies for leachate treatment are most promising due to relatively lower energy consumption and reduced chemical usage [48]. Additionally, it allows selective separation of contaminants and retention of essential ions and some beneficial minerals, which could be valuable in certain applications. This includes microfiltration, ultrafiltration, nanofiltration, and reverse osmosis, each with decreasing pore sizes. Figure 5.4 represents the filtration spectrum and the range that is applicable for each type of membrane process.

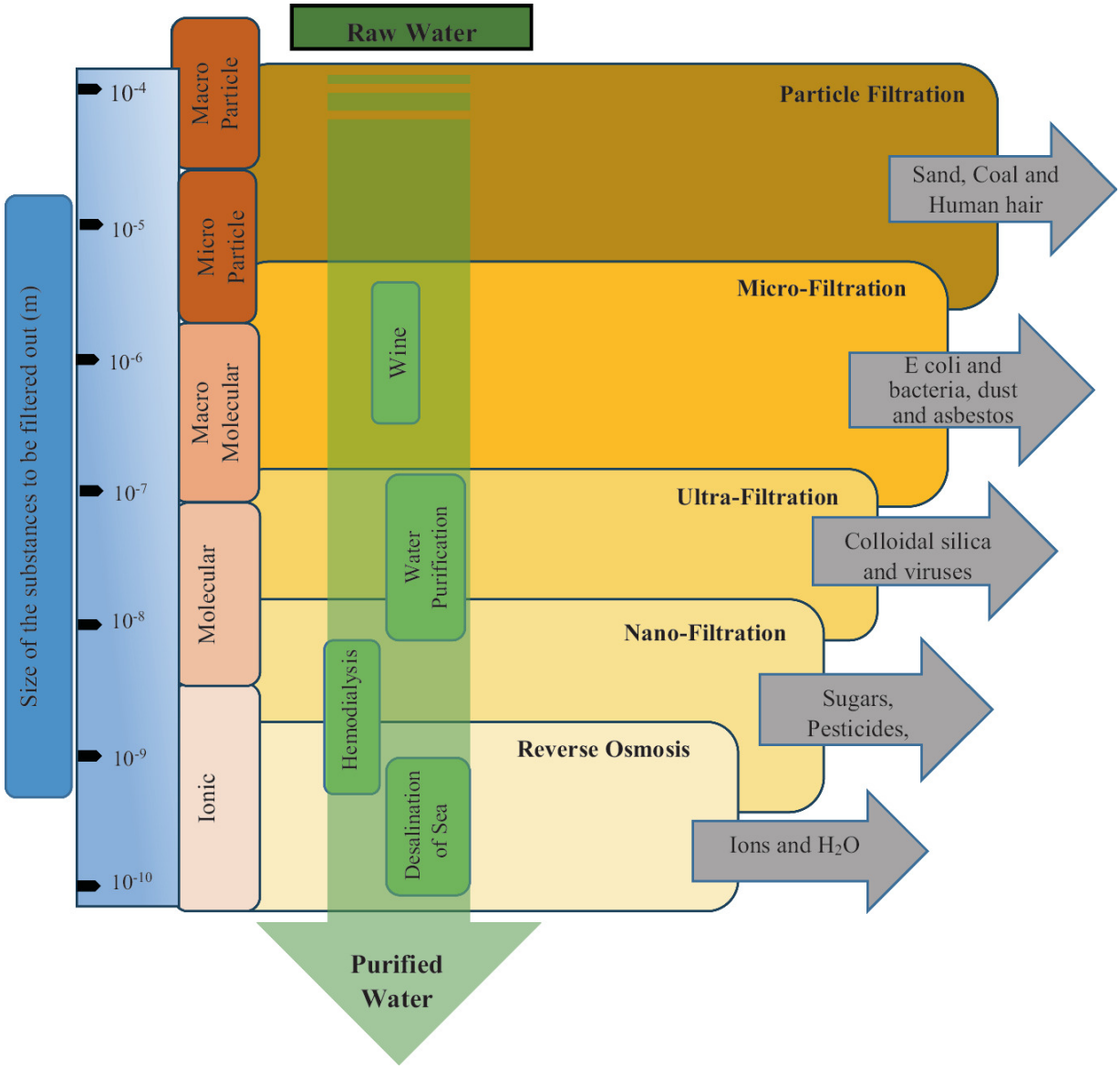


Fig. 5.4 Filtration spectrum [1, 21, 50]

Microfiltration (MF) is frequently employed as a pre-treatment modality in the treatment of LL, serving to selectively removal of suspended and colloidal particulate matter within the granularity range of 0.1–1 µm. Ultrafiltration (UF) proves effective in eliminating larger molecules and particles, featuring pores spanning from 2 nm to 0.1 µm. Employing this approach, the rates of COD removal fluctuate from 10 to 75% [50]. UF is often implemented as a preliminary stage for reverse osmosis (RO) to eliminate high molecular weight components

present in landfill leachate. These components can lead to membrane fouling in the course of the subsequent RO process. The nanofiltration (NF) process is adept at eliminating diverse contaminants—microbial, organic, and inorganic—via pores with molecular cut-off values ranging between 200 and 2000 Da [15]. The hierarchy of anion interception rates is sequenced as follows: $NO_3^- < Cl^- < OH^- < SO_4^{2-} < CO_3^{2-}$. For cations, the order is: $H^+ < Na^+ < Ca^{2+} < Mg^{2+}$. As highlighted by Ramaswami et al. [42] employing NF processes prior to RO stages is more energy-efficient, since both steps could operate under lower pressure conditions.

Even though membrane processes have demonstrated their capability to produce high-quality effluent, their widespread application is constrained by two intrinsic disadvantages. One disadvantage is the buildup of biological species, organic or inorganic, on the surface of the membrane and within its pores, resulting in a phenomenon known as membrane fouling. This undesirable outcome can drastically reduce the lifespan of membrane modules and boost the operational expenses [2]. The second concern involves the generation of concentrated landfill leachate through the membrane treatment process. It is worth noting that membrane technologies are not designed to degrade pollutants, but instead, they concentrate these pollutants into smaller volumes, resulting in what is termed as “membrane concentrated landfill leachate.“ This concentrated leachate, comprising approximately 20–30% of the initial volume, consists of elevated quantities of inorganic salts and recalcitrant organic compounds. In comparison to the original leachate, this concentrated variant is much more intricate and challenging to treat [50]. Additionally, during the membrane filtration process, the separation of ammonium is often inadequate [2].

Certain pretreatment techniques can mitigate membrane fouling, using strategies like eliminating suspended matter, avoiding the precipitation of insoluble salts, preventing scaling, and eliminating bacteria, organic matter and other microorganisms. Researchers have made strides in enhancing the performance and longevity filtration membranes through modifications and improvements in membrane materials and configurations [50].

5.4.4 Electrochemical Treatment

The basis of all electrochemical technologies is the transmission of electrons between an electrolytic solution and electrodes by generating an electric field between cathode and anode formed of materials that are unique to specific electrochemical technology [23]. Since electrons and in-situ formed oxidant agents are the principal species engaged in the process of oxidation, electrochemical technologies really have several benefits, one of which is their inherent ecological compatibility [18, 23]. Additional benefits include the utilization of basic equipment, easiness of application, durability, adaptability, and automation potential [23]. The electrochemical methods that have been the subject of the highest study to treat sanitary landfill leachate include electrocoagulation, electrochemical oxidation, electro-Fenton and electrochemical integrated processes [18, 23].

Electrocoagulation

In the process of electrocoagulation (EC), ions are added into the solution or suspension using consumable electrodes, causing pollutants that are emulsified, dissolved, or suspended to aggregate [23]. The electro-coagulation method, in contrast to the conventional chemical coagulation process, will not require any external

coagulants, thereby reducing the project's operating time and lowering the operational complexity [30]. There are three distinct stages to the in-situ production of the coagulating ions.

(i) the “sacrificial electrode” is electrolytically oxidized for the production of coagulants

(ii) breakdown of emulsions, destabilization of pollutants, and suspension of particulates

(iii) forming flocs out of the gathered destabilized phases.

Either precipitation and chemical reactions remove the pollutants from the solution, or colloidal materials produced by electrode corrosion allow the pollutants to physically and chemically bind and adhere to them [23]. It is possible to describe the fundamental electro-coagulation reaction through Eqs. (5.20), (5.21) and (5.23) [18].

$$\text{At the anode: } M_{(S)} \rightarrow M^{n+}_{(aq)} + ne^- \tag{5.20}$$

$$\text{At the cathode: } 2H_2O + 2e^- \rightarrow 2OH^- + H_2 \tag{5.21}$$

$$\text{In the solution: } M^{n+}_{(aq)} + nOH^- \rightarrow M(OH)_{n(S)} \tag{5.22}$$

The efficiency of EC treatment can be affected by several process factors, including conductivity, pH, electrode material, and current density, etc.

The most popular EC electrode materials are iron (Fe) and aluminum (Al) because they are affordable, easily accessible, and efficient [23]. Electrodes made of Al and Fe are about equally effective at lowering COD. Considering the cost and the toxicity to the environment, iron electrodes are considered as the most suitable electrode to be used [9, 19]. Since current density is the sole variable that directly controls how well electrocoagulation works, it has a significant impact on how effective this method is at

treating landfill leachate. An increment in current density accelerates the development of hydrogen gas, which affects how different contaminants and coagulants transfer mass. As a result, more collisions result in the production of flocs [28].

Additionally, the impact of mechanical stirring on the effectiveness of the EC on leachates treatment has been investigated. A more homogenous solution medium may be encouraged by mixing, which will greatly improve the efficiency of the interface between the pollutant and coagulants. As opposed to that, mixing may hinder the mobilization of ions and the disintegration of flocs. Although moderate stirring has drawbacks related to additional expenses, based on the experimental conditions used, it might still be regarded advantageous while treating leachates with the EC method [23]. The initial pH of the solution is also of significant importance when treating landfill leachate using electro-coagulation due to the impact of pH on the various types of flocculants currently in use. Elevated COD removal efficiencies were reported to be attained under neutral circumstances. COD removal was shown to decrease in acidic or alkaline medium [18]. Electrical conductivity is another crucial factor to the electrochemical processes because poor electrical conductivity results in decreased current efficiency and a requirement for additional applied potentials to prevent electrode passivation, which raises energy utilization [23].

Electro-Fenton (EF)

Fenton made the first observation of the Fenton reaction in 1984, which is hinged on the transfer of an electron between H_2O_2 and Fe^{2+} [23]. Iron catalyst is incorporated to the electrolyte to stimulate Fenton reaction to make high levels of hydroxyl radicals that supports the removal of

contaminants and efficient mineralization. H_2O_2 is formed at the cathode supplied with air or O_2 [18]. Both Fe^{2+} and H_2O_2 can be electrogenerated on-site during this procedure. Prior mentioned processes can be described by Eqs. (5.23)–(5.25).

$$O_2 + 2H^+ + 2e^- \rightarrow H_2O_2 \tag{5.23}$$

$$H_2O_2 + Fe^{2+} + H^+ \rightarrow Fe^{3+} + H_2O + \cdot OH \tag{5.24}$$

$$Fe^{3+} + e^- \rightarrow Fe^{2+} \tag{5.25}$$

One of the most important processes in the EF process is the choosing of the material for the anode. Using an unstable anode material can deteriorate the electrode in the electrolytic cells, and hydroxyl radicals would be produced in the EF process if a high-oxygen overvoltage anode was used. Due to its chemical stability and excellent conductivity at high potentials and in extremely corrosive environments, Pt is one of the frequently used anode materials in the EF process [23]. However, based on the research by Oturan et al. [41], it was revealed that Boron-Doped Diamond (BDD) anode was more effective than Pt anode at removing dissolved TOC since it produced more hydroxyl radicals.

One of the primary constraints limiting the efficiency of the EF process, among the many operating parameters, is the concentration of the Fenton reagents. Scavenging reactions happen when H_2O_2 or Fe (II), the Fenton reagents, are present in excess or insufficient amounts. Although the efficiency of H_2O_2 is diminished beyond a certain threshold, higher H_2O_2 doses often result in larger COD removal [23].

Another important element that impacts the effectiveness of the electro-Fenton reaction used to remediate landfill leachate is pH. It was discovered that the

electro-Fenton process’s high efficiency depends on a low pH, that seems to exclude the utilization of electro-Fenton technology for real world engineering applications. This appears to be as in most cases, landfill leachate does not have a particular pH range that is as acceptable, needing acidification prior to treatment [18, 22]. The applied current intensity in the EF process is another significant operational parameter. The electro-regeneration of Fe^{2+} from Fe^{3+} will accelerate with a boost in the applied current, which would predictably improve the effectiveness of Fenton chain reactions. High applied currents would, however, cause competing electrode processes such oxygen emission at the anode and the production of hydrogen at the cathode to intensify and hinder the primary reactions. The impact of electrical current on EF's ability in the removal of organic waste from landfill leachates has also been an investigated factor in the efficiency. Despite the increase in removal of organic load with increased current, it stabilizes or begins to decrease when additional increases in current is done. Most electro-Fenton investigations using landfill leachates showed that the optimum current densities are within the range of 300 and 500 Am^{-2}. Removal of pollutants in the EF process is also impacted by the separation between the electrodes. The ohmic drop across the electrolyte would be reduced when the distance between the electrodes decreases, which would also result in a reduction in the energy utilization and cell voltage. Large interelectrode spacing has an impact on the ferric ion's limiting mass transfer onto the cathode surface, which controls ferrous ion regeneration. To optimize COD elimination, an optimal distance range should be determined [23].

Electrochemical Oxidation

The frequently used electrochemical method for eliminating organic contaminants from wastewaters is electrochemical oxidation (EO). It is an electrolysis-based procedure that, in its most basic form, involves oxidizing contaminants in an electrolytic cell made up of two electrodes linked by a separate circuit to facilitate electrochemical processes [23]. Of all the benefits of EO, the absence of sludge generation, conversion of higher-molecular-weight organic molecules into lower-molecular-weight organic compounds, and total organic mineralization are arguably the most important ones. Ammonium removal by electrochemical oxidation was also found to be effective as well. Ammonium is thought to be the most major contaminant in Landfill Leachate over the long run. In the EO process, contaminants are either removed by direct EO or indirect EO as explained in Fig. 5.5 [37].

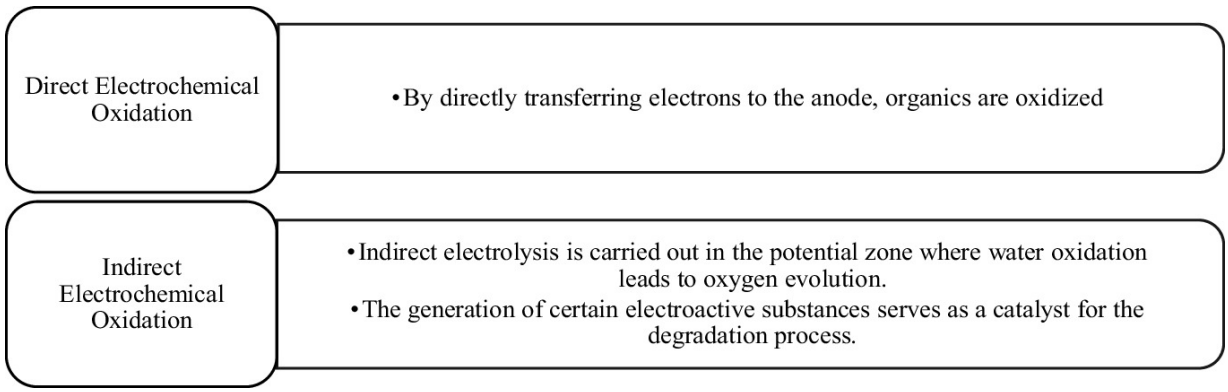


Fig. 5.5 Pollutant removal processes by electrochemical oxidation [37]

The oxidation processes that take place when an aqueous effluent is electrochemically oxidized might take one of two paths [23].

(I) Electrochemical conversion—This involves the transformation of organic chemicals into greater biodegradable reaction byproducts while still requiring a future treatment, typically a biological one.

(II) Electrochemical combustion—This requires no additional purification since organic chemicals are totally mineralized and converted into carbon dioxide, water, and various inorganic species.

An electrochemical oxidation system's effectiveness in terms of current efficiency and removal of pollutants depends on several of operational factors, including the material utilized for the anode, current density, influent chloride content, pH level, and concentration and type of the electrolyte used in the process of electrolysis [37].

In comparison with the other commonly used anode materials, the maximum organic oxidation rates and maximum current efficiencies were observed when BDD electrodes were used. Another significant aspect influencing electrochemical oxidation is the current density. The anode potential observed was lower than what was needed for a partial oxidation process when a reduced current density was supplied, which prevented oxidation. However, due to the fact that the enhancement of the reaction efficiency is mostly governed by the mass transfer process, the current efficiency significantly dropped when the current density exceeded the limit current density of electrochemical oxidation. The effectiveness of COD removal by electrochemical oxidation is also greatly influenced by the pH level. Hypochlorous acid and hypochlorite are the predominant forms of accessible chlorine when the pH is in the range of 6 and 9, which shows a strong oxidation performance. However, when the pH of solution is high, the chlorine that is available is transformed into chlorate, which has a weak oxidizing property [23].

5.4.5 Coagulation/Flocculation

The process of flocculation and coagulation allows electrical charges to be neutralized or reduced thereby

enabling colloidal particles to approach one another more closely before aggregating into flocs [20]. The coagulation/flocculation process is a very straightforward leachate treatment technique [26].

A chemical-based coagulant is required for the coagulation-flocculation process to effectively remove particles and solid contaminants by charge neutralizing negatively charged colloids. The coagulation-flocculation process due to the heavy use of chemicals typically produces an excessive volume of sludge. This has led to slowing down the process of settling the sludge disposal problem. In addition, the resulting sludge is considered to be poisonous, and the sludge might be hazardous to the environment [53]. The key parameters affecting the performance of coagulation are the pH, temperature, and time of the experiment, as well as the rate and kind of coagulant and/or flocculent used. Depending on the kind of contaminant present and coagulant/flocculant type used, this technique has been used successfully in order to remove nonbiodegradable organic compounds, colloidal particles, suspended solids, turbidity, heavy metals, and color [13]. Chemical coagulants can be classified into two distinct groups: Pre-hydrolyzing metallic salts and hydrolyzing metallic salts, as shown in Fig. 5.6.

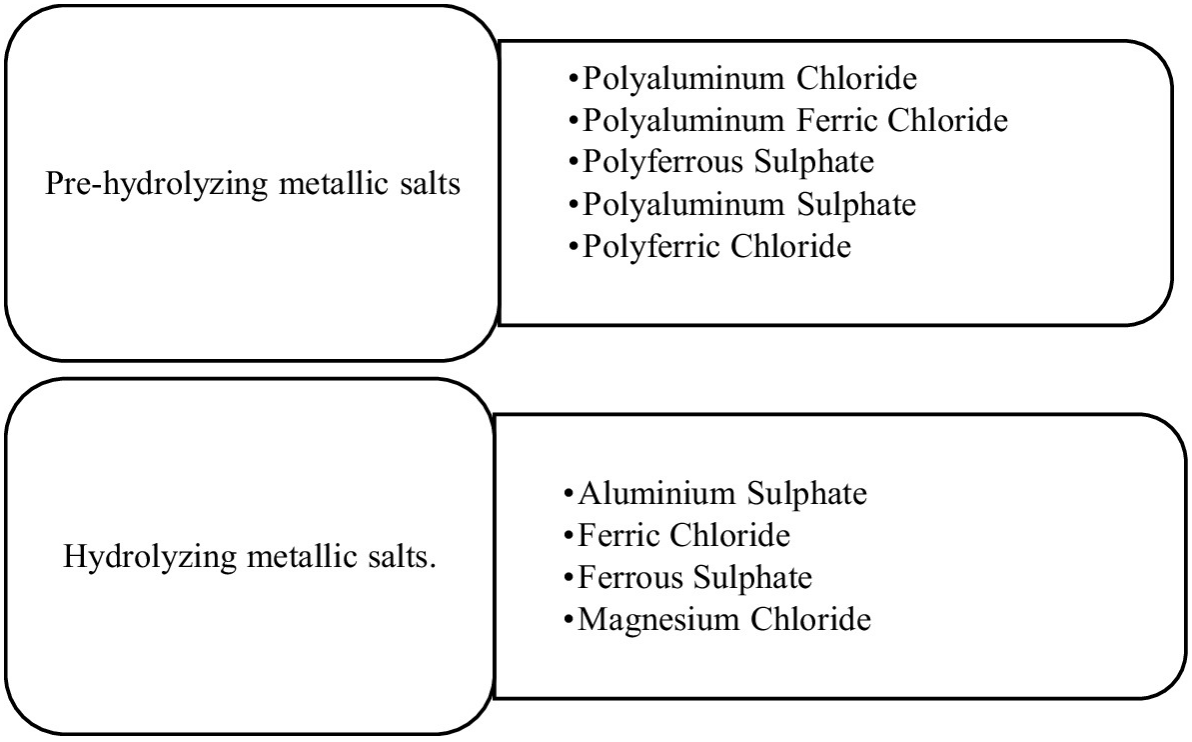


Fig. 5.6 Classification of chemical coagulants [33]

Typical coagulants that have been employed in previous research include lime, iron sulphate, iron sulphate, iron chloride, ferric chloride, and ferric sulphate [26].

Typically, coagulation is followed by a flocculation procedure to enlarge the particle size and turn the unstable particles into large floccules so that the particles can settle more readily [13]. Flocculation could be defined as a gentle mixing technique to amplify the size of the particles from micro flocs and sub-microscopic to noticeable suspended particles. The solution will then be stirred in a slow rotating mixing thereby allowing for a longer contact period and more opportunity for the particles to interact and destabilize. A shorter period of mixing during flocculation may result in collisions between colloids and flocculants which alter the precipitation of the suspended solids whereas an excessive or longer periods of mixing can cause the floc to break up. Swiping and bridging, two typical flocculation processes, will transpire to create micro flocs [54]. Flocculants can be derived from [33]

(i) Organic Synthetic Polymers

Organic flocculants were initially recovered as linear soluble polymers from non-renewable and oil-based materials with repeating monomer units based on acrylamide and acrylic acid. Organic Polymer- Based flocculants could be categorized into groups as anionic, cationic, non-ionic, and amphoteric.

(ii) Biopolymer

Environmentally friendly flocculants include biopolymers. Biopolymers have lately become an emerging alternative for the treatment of water and wastewater. Biopolymers are often classified into three groups: microorganism-based biopolymers, plant-based biopolymers, and animal-based biopolymers.

(iii) Grafted polymers

Grafted polymers are altered polymers created by attaching synthetic polymers to the structural core of natural polymers. With the use of this grafting method, the cationic charge and the active component of the polymers are then further enhanced.

When removing contaminated substances by coagulation and flocculation, pH is a crucial parameter to be considered [53]. The precipitation of metal hydroxides happens only at the desired pH followed after the incorporation of metal salts as coagulants are pH dependent. Contrary to coagulants, anionic or cationic polymers can function in a variety of pH conditions which includes acidic, alkaline, and neutral environments. The use of coagulants and flocculants is also significantly influenced by zeta potential. Cationic polymers are used if the zeta potential is below the

isoelectric point of the colloidal substances. These polymer chains adsorption onto the negatively charged colloidal surface will take place for charge neutralization and bridge formation. Anionic polymers will perform better and the polymer chains adsorption onto the positively charged colloidal surface will take place for charge neutralization and bridge formation when the zeta potential for colloidal particles is above the isoelectric point [33].

Coagulant Dosage is an important factor that has to be considered in the process of coagulation and flocculation. Based on studies done for coagulation and flocculation using different coagulants shows that the increase in dosage will increase removal efficiency (Turbidity, Ammoniacal-Nitrogen and COD) up until a specific dosage. Beyond that point the removal efficiency decreases which could be due to re-stabilization of colloids when excessive amount of coagulant is used [53]. Since it increases the likelihood that colloidal particles will cling together during the coagulation-flocculation process, the stirring speed plays a crucial role in the destabilization of the particles. As observed by Djeffal et al. [20] regardless of the coagulant used the clarity (turbidity reduction) increased until a certain speed and then has reduced beyond the optimum speed.

5.5 Integrated Routes for the Leachate Treatment

The integrated process reduces the drawbacks of each individual process which will then increase the overall efficiency treatment of leachate. Figure 5.7 illustrates the different integrated method used in landfill leachate treatment.

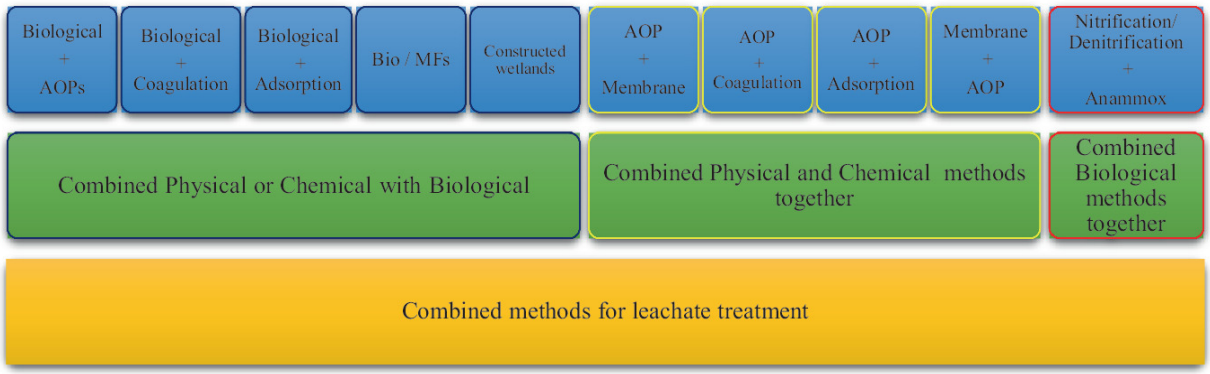


Fig. 5.7 Integrated methods for landfill leachate treatment [39]

Ghahrchi and Rezaee [27] stated that, to enhance the effectiveness of landfill leachate treatment, ozonation is frequently combined together with other AOPs, including electrochemistry/O_3, O_3/UV, $O_3/S_2O_8^{2-}$ and O_3/H_2O_2. As stated by Costa et al. [16], the utilization of a membrane treatment procedure together with biological methods of treatment, conducted within a membrane bioreactor (MBR), demonstrates noticeable effectiveness in eliminating COD, micro-pollutants, ammonia, and BOD, from leachate generated by mature landfills. Combining the air-stripping, nanofiltration, and MBR techniques can deliver exceptionally effective treatment for landfill leachate. This integrated approach leads to a collective decrease in toxicity, color, ammonia, and COD, by approximately 100, 100, 95 and 88%, respectively [50]. Additionally, the incorporation of the Fenton reaction alongside membrane separation proved highly beneficial in reducing membrane fouling. This approach offered supplementary treatment for leachate, resulting in effluent that contained contaminants meeting the necessary regulatory standards for environmental discharge [29].

Boumechhour et al. [10] performed combined treatment for leachate with coagulation and flocculation followed by Fenton process. The combined treatment method has shown a significant increase of COD removal of about 16% when compared with COD removal in the standalone

coagulation and flocculation process. This has been attributed to the recalcitrant organics removal that might not have been removed during the coagulation and flocculation process. Overall separation performance may be improved by effectively reducing membrane-fouling issues by the combination of membrane filtration with AOPs [39]. Integration of advanced oxidation processes based on AOP-Fenton and NF and MF membrane processes for landfill leachate treatment were studied by Santos et al. [45]. Results show that the combination of AOP-Fenton, NF and MF was discovered to be a successful method for treatment for landfill leachate with over 94% removal efficiencies of color and COD. Traditional electrochemical oxidation and ozonation can be made more effective using a technique called electro-catalytic ozonation [18]. According to Ghahrchi and Rezaee [27] compared to the EC and Single Ozone Process (SOP) procedures, the combined EC/SOP process shown excellent efficiency in lowering COD concentrations, which increased the landfill leachate's biodegradability index.

5.6 Conclusion

The most popular method of waste management is still landfilling. Water percolates through waste to create landfill leachate, which contains inorganic and organic particles like heavy metals, impurities, and other contaminated and hazardous materials. Untreated leachate discharge might cause serious issues for a very long period. This is because leachate is continually produced for 30–50 years after a landfill is closed [6]. The landfill's age, the amount of precipitation, the kind of waste, the volatility in the weather, and its composition all have an impact on the quality and quantity of leachate. A leachate treatment plant is always present at a working landfill site to handle harmful contaminants in the leachate. In order to properly

release treated leachate into the environment without causing any harm to the environment, landfill managers have long prioritized developing a sustainable approach for leachate treatment [32]. Leachate treatment techniques used today range widely, including natural processes, physical–chemical processes, biological processes, and hybrid processes. Among these physicochemical processes are the best to treat leachate of mature landfills. Every physicochemical process has its own advantages as well as drawbacks. Traditional treatments including biological and physicochemical procedures are increasingly ineffective due to the complex composition of leachate, which has led to the pursuit for new technologies [45].

In conclusion, the field of physicochemical treatment approaches presents a promising path for effectively addressing the challenges posed by leachate treatment. This review has highlighted their significant potential of treatments such as advanced oxidation processes, adsorption, filtration, electrochemical treatments and coagulation, flocculation methods in mitigating the environmental impact of landfill leachate. The complex interaction of numerous physicochemical processes provides a complete framework for targeting a wide range of contaminants, demonstrating the adaptability of these approaches. Further studies can be done on these integrated systems to improve the treatment of leachate.

References

1. Abdel-Fatah MA (2018) Nanofiltration systems and applications in wastewater treatment: review article. Ain Shams Eng J 9(4):3077–3092. https://doi.org/10.1016/j.asej.2018.08.001
2. de Almeida R, Bila DM, Quintaes BR, Campos JC (2020) Cost estimation of landfill leachate treatment by reverse osmosis in a Brazilian landfill. Waste Manage Res 38(10):1087–1092. https://doi.org/10.1177/0734242X20928411
[Crossref]

3. Antony J, Niveditha SV, Gandhimathi R, Ramesh ST, Nidheesh PV (2020) Stabilized landfill leachate treatment by zero valent aluminium-acid system combined with hydrogen peroxide and persulfate based advanced oxidation process. Waste Manage 106:1–11. https://doi.org/10.1016/j.wasman.2020.03.005
4. Aydın Temel F, Kuleyin A (2016) Ammonium removal from landfill leachate using natural zeolite: kinetic, equilibrium, and thermodynamic studies. Desalin Water Treat 57(50):23873–23892. https://doi.org/10.1080/19443994.2015.1136964
[Crossref]
5. Aziz HA, Noor AFM, Keat YW, Alazaiza MYD, Hamid AA (2020) Heat activated zeolite for the reduction of ammoniacal nitrogen, colour, and COD in landfill leachate. Int J Environ Res 14(4):463–478. https://doi.org/10.1007/s41742-020-00270-5
[Crossref]
6. Aziz HA, Ramli SF (2018) Recent development in sanitary landfilling and landfill leachate treatment in Malaysia. Int J Environ Eng 9(3/4):201. https://doi.org/10.1504/ijee.2018.10018737
[Crossref]
7. Azmi NB, Bashir MJK, Sethupathi S, Ng CA (2016) Anaerobic stabilized landfill leachate treatment using chemically activated sugarcane bagasse activated carbon: kinetic and equilibrium study. Desalin Water Treat 57(9):3916–3927. https://doi.org/10.1080/19443994.2014.988660
[Crossref]
8. Azmi NB, Bashir MJK, Sethupathi S, Wei LJ, Aun NC (2015) Stabilized landfill leachate treatment by sugarcane bagasse derived activated carbon for removal of color, COD and NH3-N—optimization of preparation conditions by RSM. J Environ Chem Eng 3(2):1287–1294. https://doi.org/10.1016/j.jece.2014.12.002
[Crossref]
9. Bouhezila F, Hariti M, Lounici H, Mameri N (2011) Treatment of the OUED SMAR town landfill leachate by an electrochemical reactor. Desalination 280(1–3):347–353. https://doi.org/10.1016/j.desal.2011.07.032
[Crossref]
10. Boumechhour F, Rabah K, Lamine C, Said BM (2013) Treatment of landfill leachate using Fenton process and coagulation/flocculation. Water Environ J 27(1):114–119. https://doi.org/10.1111/j.1747-6593.2012.00332.x
[Crossref]
11. de Brito RA, Filho HJI, Aguiar LG, de Alcântara MAK, Siqueira AF, da Rós PCM (2019) Degradation kinetics of landfill leachate by continuous-flow

catalytic ozonation. Ind Eng Chem Res 58(23):9855–9863. https://doi.org/10.1021/acs.iecr.9b01391
[Crossref]

12. Chen G, Wu G, Li N, Lu X, Zhao J, He M, Yan B, Zhang H, Duan X, Wang S (2021) Landfill leachate treatment by persulphate related advanced oxidation technologies. J Hazard Mater 418. https://doi.org/10.1016/j.jhazmat.2021.126355

13. Cherni Y, Elleuch L, Messaoud M, Kasmi M, Chatti A, Trabelsi I (2021) Recent technologies for leachate treatment: a review. Euro-Mediterr J Environ Integr 6(3). https://doi.org/10.1007/s41207-021-00286-z

14. Chávez RP, Pizarro ECC, Galiano YL (2019) Tratamento de lixiviado de aterro sanitário com carvão ativado obtido de resíduos de café. Engenharia Sanitaria e Ambiental 24:833–842. https://doi.org/10.1590/S1413-41522019178655
[Crossref]

15. Collado S, Núñez D, Oulego P, Riera FA, Díaz M (2020) Effect of landfill leachate ageing on ultrafiltration performance and membrane fouling behaviour. J Water Process Eng 36:101291. https://doi.org/10.1016/j.jwpe.2020.101291
[Crossref]

16. Costa AM, Alfaia RG, Campos JC (2019) Landfill leachate treatment in Brazil—an overview. J Environ Manage 232:110–116. https://doi.org/10.1016/j.jenvman.2018.11.006

17. Crini G, Lichtfouse E, Wilson L, Morin-Crini N, Wilson LD (2019) Conventional and non-conventional adsorbents for wastewater treatment. Environ Chem Lett 17(1):195–213. https://doi.org/10.1007/s10311-018-0786-8
[Crossref]

18. Deng Y, Zhu X, Chen N, Feng C, Wang H, Kuang P, Hu W (2020) Review on electrochemical system for landfill leachate treatment: performance, mechanism, application, shortcoming, and improvement scheme. Sci Total Environ 745. https://doi.org/10.1016/j.scitotenv.2020.140768

19. Dia O, Drogui P, Buelna G, Dubé R, Ihsen BS (2017) Electrocoagulation of bio-filtrated landfill leachate: fractionation of organic matter and influence of anode materials. Chemosphere 168:1136–1141. https://doi.org/10.1016/j.chemosphere.2016.10.092
[Crossref]

20. Djeffal K, Bouranene S, Fievet P, Déon S, Gheid A (2021) Treatment of controlled discharge leachate by coagulation-flocculation: influence of

operational conditions. Sep Sci Technol (Philadelphia) 56(1):168–183. https://doi.org/10.1080/01496395.2019.1708114
[Crossref]

21. Eyvaz M, Arslan S, Gürbulak E, Yüksel E (2017) Textiles for advanced applications textile materials in liquid filtration practices: current status and perspectives in water and wastewater treatment. InTech. https://doi.org/10.5772/intechopen.69462

22. Fernandes A, Labiadh L, Ciríaco L, Pacheco MJ, Gadri A, Ammar S, Lopes A (2017) Electro-fenton oxidation of reverse osmosis concentrate from sanitary landfill leachate: evaluation of operational parameters. Chemosphere 184:1223–1229. https://doi.org/10.1016/j.chemosphere.2017.06.088
[Crossref]

23. Fernandes A, Pacheco MJ, Ciríaco L, Lopes A (2015) Review on the electrochemical processes for the treatment of sanitary landfill leachates: present and future. Appl Catal B Environ 176–177:183–200. https://doi.org/10.1016/j.apcatb.2015.03.052

24. Gautam P, Kumar S, Lokhandwala S (2019) Advanced oxidation processes for treatment of leachate from hazardous waste landfill: a critical review. J Clean Prod 237:117639. https://doi.org/10.1016/j.jclepro.2019.117639
[Crossref]

25. Gautam P, Kumar S, Lokhandwala S (2019) Advanced oxidation processes for treatment of leachate from hazardous waste landfill: a critical review. J Clean Prod 237. https://doi.org/10.1016/j.jclepro.2019.117639

26. Ghaffariraad M, Ghanbarzadeh Lak M (2021) Landfill leachate treatment through coagulation-flocculation with lime and bio-sorption by Walnut-shell. Environ Manage 68(2):226–239. https://doi.org/10.1007/s00267-021-01489-4
[Crossref]

27. Ghahrchi M, Rezaee A (2020) Electro-catalytic ozonation for improving the biodegradability of mature landfill leachate. J Environ Manage 254. https://doi.org/10.1016/j.jenvman.2019.109811

28. Ghanbari F, Wu J, Khatebasreh M, Ding D, Lin KYA (2020) Efficient treatment for landfill leachate through sequential electrocoagulation, electrooxidation and PMS/UV/$CuFe_2O_4$ process. Sep Purif Technol 242. https://doi.org/10.1016/j.seppur.2020.116828

29. Han M, Duan X, Cao G, Zhu S, Ho SH (2020) Graphitic nitride-catalyzed advanced oxidation processes (AOPs) for landfill leachate treatment: a mini review. Process Saf Environ Protect 139:230–240. https://doi.org/10.1016/j.psep.2020.04.046

30. Huda N, Raman AAA, Bello MM, Ramesh S (2017) Electrocoagulation treatment of raw landfill leachate using iron-based electrodes: effects of process parameters and optimization. J Environ Manage 204:75–81. https://doi.org/10.1016/j.jenvman.2017.08.028 [Crossref]
31. Jung C, Deng Y, Zhao R, Torrens K (2017) Chemical oxidation for mitigation of UV-quenching substances (UVQS) from municipal landfill leachate: fenton process versus ozonation. Water Res 108:260–270. https://doi.org/10.1016/j.watres.2016.11.005 [Crossref]
32. Kamaruddin MA, Yusoff MS, Aziz HA, Hung YT (2015) Sustainable treatment of landfill leachate. Appl Water Sci 5(2):113–126. https://doi.org/10.1007/s13201-014-0177-7
33. Khoo KS, Tan X, Show PL, Pal P, Juan JC, Ling TC, Ho SH, Nguyen THP (2020) Treatment for landfill leachate via physicochemical approaches: an overview. Chem Biochem Eng Q 34(1):1–24. https://doi.org/10.15255/CABEQ.2019.1703
34. Kow S, Fahmi MR, Abidin CZA, Soon-An O (2016) Advanced oxidation processes: process mechanisms, affecting parameters and landfill leachate treatment. Water Environ Res 88(11):2047–2058. https://doi.org/10.2175/106143016x14733681695285 [Crossref]
35. Kwarciak-Kozłowska A, Włodarczyk R, Wystalska K (2019) Biochar compared with activated granular carbon for landfill leachate treatment. E3S Web Conf 100:0042. https://doi.org/10.1051/e3sconf/201910000042
36. Mahtab MS, Islam DT, Farooqi IH (2021) Optimization of the process variables for landfill leachate treatment using Fenton based advanced oxidation technique. Eng Sci Technol Int J 24(2):428–435. https://doi.org/10.1016/j.jestch.2020.08.013 [Crossref]
37. Mandal P, Dubey BK, Gupta AK (2017) Review on landfill leachate treatment by electrochemical oxidation: drawbacks, challenges and future scope. Waste Manage 69:250–273. https://doi.org/10.1016/j.wasman.2017.08.034
38. Mohammad-pajooh E, Turcios AE, Cuff G, Weichgrebe D, Rosenwinkel KH, Vedenyapina MD, Sharifullina LR (2018) Removal of inert COD and trace metals from stabilized landfill leachate by granular activated carbon (GAC) adsorption. J Environ Manage 228:189–196. https://doi.org/10.1016/j.jenvman.2018.09.020 [Crossref]

39. Mojiri A, Zhou JL, Ratnaweera H, Ohashi A, Ozaki N, Kindaichi T, Asakura H (2021) Treatment of landfill leachate with different techniques: an overview. J Water Reuse Desalin 11(1):66–96. https://doi.org/10.2166/wrd.2020.079
[Crossref]
40. Mukherjee S, Mukhopadhyay S, Hashim MA, Gupta BS (2015) Contemporary environmental issues of landfill leachate: assessment and remedies. Crit Rev Environ Sci Technol 45(5):472–590. https://doi.org/10.1080/10643389.2013.876524
[Crossref]
41. Oturan N, van Hullebusch ED, Zhang H, Mazeas L, Budzinski H, le Menach K, Oturan MA (2015) Occurrence and removal of organic micropollutants in landfill leachates treated by electrochemical advanced oxidation processes. Environ Sci Technol 49(20):12187–12196. https://doi.org/10.1021/acs.est.5b02809
[Crossref]
42. Ramaswami S, Behrendt J, Otterpohl R (2018) Comparison of NF-RO and RO-NF for the treatment of mature landfill leachates: a guide for landfill operators. Membranes 8(2). https://doi.org/10.3390/membranes8020017
43. Renou S, Givaudan JG, Poulain S, Dirassouyan F, Moulin P (2008) Landfill leachate treatment: Review and opportunity. J Hazard Mater 150(3):468–493. https://doi.org/10.1016/j.jhazmat.2007.09.077
44. Reshadi MAM, Bazargan A, McKay G (2020) A review of the application of adsorbents for landfill leachate treatment: focus on magnetic adsorption. Sci Total Environ 731. https://doi.org/10.1016/j.scitotenv.2020.138863
45. Santos AV, de Andrade LH, Amaral MCS, Lange LC (2019) Integration of membrane separation and Fenton processes for sanitary landfill leachate treatment. Environ Technol (UK) 40(22):2897–2905. https://doi.org/10.1080/09593330.2018.1458337
[Crossref]
46. Sasaki T, Iizuka A, Watanabe M, Hongo T, Yamasaki A (2014) Preparation and performance of arsenate (V) adsorbents derived from concrete wastes. Waste Manage 34(10):1829–1835. https://doi.org/10.1016/j.wasman.2014.01.001
[Crossref]
47. Shehzad A, Bashir MJK, Sethupathi S, Lim JW (2015) An overview of heavily polluted landfill leachate treatment using food waste as an alternative and renewable source of activated carbon. Process Saf Environ Protect 98:309–318. https://doi.org/10.1016/j.psep.2015.09.005

48. Tałałaj IA, Biedka P, Bartkowska I (2019) Treatment of landfill leachates with biological pretreatments and reverse osmosis. Environ Chem Lett 17(3):1177–1193. https://doi.org/10.1007/s10311-019-00860-6
49. Tejera J, Miranda R, Hermosilla D, Urra I, Negro C, Blanco A (2019) Treatment of a mature landfill leachate: comparison between homogeneous and heterogeneous photo-fenton with different pretreatments. Water 11(9):1849. https://doi.org/10.3390/w11091849
50. Teng C, Zhou K, Peng C, Chen W (2021) Characterization and treatment of landfill leachate: a review. Water Res 203. https://doi.org/10.1016/j.watres.2021.117525
51. Ukundimana Z, Omwene PI, Gengec E, Can OT, Kobya M (2018) Electrooxidation as post treatment of ultrafiltration effluent in a landfill leachate MBR treatment plant: effects of BDD, Pt and DSA anode types. Electrochim Acta 286:252–263. https://doi.org/10.1016/j.electacta.2018.08.019
[Crossref]
52. Wang J, Bi L, Ji Y, Ma H, Yin X (2014) Removal of humic acid from aqueous solution by magnetically separable polyaniline: adsorption behavior and mechanism. J Colloid Interf Sci 430:140–146. https://doi.org/10.1016/j.jcis.2014.05.046
[Crossref]
53. Zainol NA, Pin LB, Rashid NA, Ghani AA, Zailani SN, Rani ALA (2018) Treatment of landfill leachate by coagulation-flocculation process using red earth as coagulant. AIP Conf Proc 2030. https://doi.org/10.1063/1.5066684
54. Zakaria SNF, Abdul Aziz H, Mohamad M (2022) Comparison performance of coagulation flocculation process and combination with ozonation process of stabilized landfill leachate treatment. Water Environ Res 94(8). https://doi.org/10.1002/wer.10770

© The Author(s), under exclusive license to Springer Nature Switzerland AG 2024
S. Souabi, A. Anouzla (eds.), *Landfill Leachate Treatment Techniques*, Springer Water
https://doi.org/10.1007/978-3-031-63157-3_6

6. Efficient Treatment of Landfill Leachate Via Coagulation-Flocculation: Optimizing Process Conditions

Roukaya Bouyakhsass[1], Safaa Khattabi Rifi[1], Younes Abrouki[1, 2], Abdelaziz Madinzi[1], Abdelkader Anouzla[1] ✉, Nor-edine Abriak[3], Tonni Agustiono Kurniawan[4], Kastali Malika[1] and Salah Souabi[1]

(1) Department of Process Engineering and Environment Mohammedia, Hassan II University of Casablanca-Faculty of Sciences and Techniques of Mohammedia, Mohammedia, Morocco
(2) Faculty of Science, Mohammed V University in Rabat, Rabat, Morocco
(3) IMT Lille Douai, GCE, 764 Bd Lahure, BP 10838, 59508 Douai, France
(4) College of the Environment and Ecology, Xiamen University, Xiamen, 361102, China

✉ **Abdelkader Anouzla**
Email: aanouzla@gmail.com

Abstract
Landfill leachate, a complex and highly polluted liquid, represents a major environmental challenge, requiring

effective treatment methods. This study addresses the need for effective leachate treatment using a coagulation-flocculation approach. The research uses a response surface methodology to optimize key treatment factors, including pH, coagulant, flocculant, and agitation time. Optimal conditions were determined to be a pH of 6.87, a coagulant dose of 7.98 g/L, a flocculant dose of 12 ml/L, and an agitation time of 22.8 min. These conditions were obtained by mathematical modeling and were validated by experimental results. The study achieved remarkable removal percentages of 90.38% for turbidity, 85.50% for COD, and 55.53% for UV254, underlining the effectiveness of the optimized coagulation-flocculation treatment. The high R-squared values (0.97, 0.96, and 0.94 for turbidity, COD, and UV254, respectively) confirm the accuracy and reliability of the model. This research provides a sustainable solution for landfill leachate treatment. It illustrates the potential of response surface methodology in optimizing wastewater treatment processes. The results are expected to improve waste management practices and environmental preservation.

Keywords Leachate treatment – Coagulation-flocculation – Response surface methodology – Central composite design

---

6.1 Introduction

The world is experiencing an unprecedented increase in population, urbanization, and industrialization [28]. Alongside this progress, municipal solid waste (MSW) production has increased dramatically [25]. The relationship between population growth and waste production has led to a critical need for advanced and sustainable waste management practices. Various waste management techniques have been developed and implemented worldwide to meet this growing challenge.

Among these techniques, landfilling is the most widely utilized and established method [16, 15, 35]. Landfilling involves waste disposal in designated areas carefully designed to minimize environmental impacts.

Landfills have long been the preferred solution for MSW management, involving waste burial in designated areas. However, this simple method comes with a complex environmental challenge: the production of leachate [18, 19, 42]. Leachate is a complex, hazardous liquid formed by the decomposition of organic matter and the interaction of rainwater with waste [17]. It contains a variety of contaminants, including heavy metals, organic compounds, and pathogens, posing significant risks to the environment and human health. Its potential to infiltrate soil and migrate to groundwater sources has increased the importance of leachate treatment in waste management. Research conducted in Bangladesh has revealed the presence of various hazardous metals in surface and groundwater near several landfill sites. Levels of these substances exceed the acceptable limits established by the Bangladesh Ministry of Environment and the World Health Organization [35]. Similarly, in India, [30] found that groundwater quality near landfills is unsuitable for consumption or domestic use due to contamination from leachate infiltration. These investigations underline the imperative of identifying practical measures to mitigate the consequences of this wastewater.

In response to the consequences of untreated leachates, various treatment technologies have been developed to mitigate their impacts, such as biological treatment [29, 36], physicochemical treatment (Anouzla et al. 2022; [3, 8, 33]), membrane-based treatment, or integrated systems that combine severaltreatment methods [13, 22, 41]. Among these methods, coagulation-flocculation has emerged as a leading method for effectively removing contaminants from leachate. The process involves the

addition of coagulants to destabilize particles and colloids, followed by flocculation to form larger, settleable aggregates. Coagulation-flocculation offers numerous advantages, including -turbidity reduction, chemical oxygen demand (COD), and removal of heavy metals, making it a promising approach for leachate treatment [40].

In this context, integrating Response Surface Methodology (RSM) has proved highly effective in optimizing coagulation-flocculation processes. RSM provides a systematic framework for understanding and optimizing the complex relationships between multiple variables and responses [12]. A specific RSM design, the central composite design (CDD) stands out as one of the frequently employed experimental design methodologies [21, 34]. This approach identifies the ideal parameters for a specific process by comprehensively examining the interactions between variables and their impact on the desired outcome. CCD allows researchers to systematically explore the intricate landscape of coagulation-flocculation parameters, such as coagulant dosage, pH, mixing speed, and settling time [37].

This chapter aims to provide a comprehensive exploration of leachate treatment by coagulation-flocculation, focusing on the application of Central Composite Design within the framework of Response Surface Methodology. It aims to provide an in-depth understanding of the challenges posed by leachate and the potential advantages of coagulation-flocculation as a treatment method.

6.2 Coagulation Flocculation Treatment

Coagulation-flocculation treatment is a widely used water treatment technique designed to remove suspended

particles, dissolved organic matter, heavy metals, and other contaminants in water. This method relies on chemical and physical processes to aggregate suspended particles into larger flocs, which can be easily removed by sedimentation or flotation [7, 38].

The coagulation-flocculation process involves adding specific chemicals, usually coagulants such as aluminum sulfate (alum) or ferric chloride, to the treated water. These coagulants react with suspended particles and dissolved organic matter, neutralizing their electrical charges and causing them to aggregate into flocs. Subsequently, flocculating agents, such as polymers, are added to promote further floc growth by encouraging increased agglomeration [23]. The result is the formation of larger flocs that settle faster and are easier to separate from the treated water.

6.3 Experimental Design

Experimental design, an indispensable component of scientific research and industrial investigations, is a meticulously structured approach that enables researchers to analyze the complex relationships between multiple variables and their influence on desired outcomes. Its robust methodology facilitates pattern recognition, process optimization, and product quality improvement [9].

It is a statistical tool for creating, refining, and enhancing systems, processes, and products [11]. The choice of design depends on the specific scenarios faced by the experimenter. We have factorial designs, fractional designs, central composite designs, Box-Behnken designs, and Doehlert designs [14, 20, 26, 32].

Response surface methodology (RSM) is a statistical framework that extends the domain of experimental design to encompass complex multifactorial processes. RSM excels in its ability to fit mathematical models to empirical data,

providing researchers with a mathematical lens for understanding and optimizing complex processes influenced by multiple variables.

RSM, enhanced by central composite design (CCD), has applications in many disciplines, including chemistry, engineering, and environmental research. Its utility covers chemical reactions and product formulation processes to rationalizing manufacturing processes and environmental assessment [2, 10].

6.4 Application

This section examines a case study that summarizes the practical application of composite central design within the Response Surface Methodology to optimize the coagulation-flocculation treatment of leachate from the Mohammedia Benslimane landfill. This example provides a first-hand exploration of the potential of RSM to address the specific challenges posed by landfill leachate, highlighting its importance in real-world environmental remediation scenarios.

6.4.1 Sampling Procedure

The Mohammedia-Benslimane landfill, located around 8 km east of Beni Ykhlef in Morocco, is a critical waste management facility. Inaugurated in 2012, this landfill covers a significant area of 47 hectares. It plays an essential role in meeting the waste disposal needs of around 409,000 inhabitants in the region. It is strategically located in the douar Beni M'ghit Chaâba El Hamra and serves as a central hub for waste disposal. To better understand the characteristics of the landfill's leachates, 50 raw leachates were collected over 10 h at a constant rate of 5 L per hour. Samples were carefully stored in plastic containers at a controlled temperature of 4 °C until further analysis.

6.4.2 Leachate Treatment

Leachates are loaded with pollutants that can be carried away by runoff, polluting surface water, or infiltrating the landfill soil and contaminating the water table. Variations in the characteristics of different leachates between different studies are probably attributable to differences in waste composition, regional climate, landfill hydrogeology, and landfill age [43]. The samples used in this study are characterized by high pollution levels (Table 6.1).

Table 6.1 Leachate characterization

Parameter	Unit	Value
pH		7.41 ± 0.02
Conductivity	mS/cm	46.07 ± 0.1
Turbidity	NTU	1559 ± 3.47
TSS	g/L	26.9 ± 0.4
COD	mg/L	$10{,}560 \pm 8.04$
BOD_5	mg/L	5360 ± 5.9
UV_{254}		2.002 ± 0.001
Nitrates	mg/L	117.65 ± 3.23
Phenol	mg/L	263.31 ± 2.34

The leachate is described as having a dark brown color, neutral pH, and high conductivity of 46.07 ± 0.1 mS/cm, which is considerably higher compared to previous research [4, 31]. Increased turbidity, organic matter levels, and nitrogen compounds are also observed. Turbidity levels oscillate around 1559 NTU, with a TSS concentration of around 26.9 g/L. Although these results are lower than those reported in other studies for leachates from Fez and Marrakech [6, 31], they remain higher than the values obtained in Simpang Renggam, Malaysia [44]. Moreover, in this study, COD and BOD_5 concentrations show values of around 10,000 mg/L and 5360 mg/L, respectively, giving a

BOD5/COD ratio of around 0.51. These measurements imply the substantial existence of biodegradable organic material. In addition, leachate analysis indicates high levels of phenols (263.31 mg/L) and nitrates (117.65 mg/L), consistent with previous results from studies conducted by [5], concerning young leachates. Furthermore, absorbance at a wavelength of 254 nm gives a reading of 2.002, signifying the presence of phenolic compounds, as reported in the research by [27].

Initial experiments were conducted to reduce the range of various factors, including pH, coagulant and flocculant doses, and agitation duration. To investigate the impact of these factors and their interactions on the three responses (Turbidity, COD, and UV254), the central composite design was adopted (Table 6.2). The experimental design and mathematical modeling were carried out using Design-Expert 7.0.0. This study used the central composite design and the response surface method to optimize the flocculation-coagulation treatment. The factors investigated in this study are pH (A), coagulant dose (B), flocculant dose (C), and agitation duration (D). At the same time, the responses include turbidity, COD, and absorbance at a 254 nm wavelength.

Table 6.2 Levels of the experimental scale and independent test variables

Variables	**Symbol**	**Coded variables**				
		−2	**−1**	**0**	**+1**	**+2**
pH	A	6	7	8	9	10
Coagulant dose (g/L)	B	6.4	7.2	8	8.8	9.6
Floculant dose (ml/L)	C	4	6	8	10	12
Agitation duration (min)	D	10	15	20	25	30

Thirty-one experiments were executed, and the outcomes are detailed in Table 6.3.

Table 6.3 The experimental matrix and the respective experimental responses

Run N°	(A): pH	(B): Coagulant dose (g/L)	(C): Floculant dose (ml/L)	(D): Agitation duration (min)	Turbidity (NTU)	COD (mg/L)	UV_{254}
1	8	8	8	20	217 ± 3.45	480 ± 4.39	1.099 ± 0.003
2	7	8.8	10	15	157 ± 4.35	3360 ± 6.32	0.878 ± 0.002
3	9	7.2	6	25	174 ± 3.98	3840 ± 7.29	1.23 ± 0.007
4	9	8.8	10	15	250 ± 2.47	480 ± 5.12	1.12 ± 0.004
5	8	8	4	20	211 ± 4.20	960 ± 4.98	1.093 ± 0.003
6	9	8.8	10	25	198 ± 2.83	960 ± 5.01	1.11 ± 0.004
7	8	8	8	30	172 ± 4.28	2880 ± 5.38	1.046 ± 0.003
8	8	8	8	20	207 ± 1.97	480 ± 5.39	1.049 ± 0.003
9	9	7.2	6	15	184 ± 4.39	3360 ± 8.23	1.216 ± 0.006
10	7	8.8	6	25	170 ± 5.01	2880 ± 6.48	0.798 ± 0.001
11	8	8	8	20	209 ± 3.29	960 ± 5.23	1.081 ± 0.003
12	9	8.8	6	25	165 ± 4.74	2880 ± 7.23	1.094 ± 0.004
13	9	7.2	10	25	185 ± 3.23	1920 ± 6.06	1.218 ± 0.006
14	6	8	8	20	170 ± 3.47	1440 ± 5.83	1.013 ± 0.002
15	9	8.8	6	15	209 ± 3.25	1440 ± 4.93	1.151 ± 0.004
16	8	8	12	20	208 ± 3.94	960 ± 5.03	0.938 ± 0.001

Run N°	(A): pH	(B): Coagulant dose (g/L)	(C): Floculant dose (ml/L)	(D): Agitation duration (min)	Turbidity (NTU)	COD (mg/L)	UV_{254}
17	8	9.6	8	20	189 ± 3.16	3360 ± 7.39	1.035 ± 0.003
18	8	8	8	20	221 ± 4.73	480 ± 4.73	1.123 ± 0.004
19	7	7.2	6	25	254 ± 1.73	1440 ± 5.08	1.083 ± 0.003
20	7	7.2	10	15	171 ± 2.78	2400 ± 6.03	0.924 ± 0.002
21	10	8	8	20	184 ± 1.96	1440 ± 5.47	1.351 ± 0.007
22	9	7.2	10	15	199 ± 4.38	2880 ± 6.27	1.182 ± 0.005
23	7	8.8	10	25	150 ± 2.29	2400 ± 6.94	0.917 ± 0.001
24	8	6.4	8	20	224 ± 1.93	3840 ± 7.59	1.169 ± 0.005
25	7	8.8	6	15	199 ± 3.75	1920 ± 7.06	0.945 ± 0.002
26	7	7.2	6	15	242 ± 2.19	1440 ± 6.64	1.077 ± 0.003
27	7	7.2	10	25	210 ± 4.20	1440 ± 6.94	1.051 ± 0.003
28	8	8	8	20	226 ± 4.23	960 ± 5.16	1.122 ± 0.006
29	8	8	8	10	209 ± 3.46	3360 ± 7.95	1.103 ± 0.005
30	8	8	8	20	218 ± 2.39	960 ± 5.03	1.037 ± 0.003
31	8	8	8	20	222 ± 3.48	480 ± 4.21	1.08 ± 0.003

The highest turbidity removal rate was 90%. This result was obtained with a combination of factors: a pH set at 7, a coagulant dose of around 8.8 g/L, a flocculant dose of 10 ml/L, and an agitation duration of 25 min. However,

COD removal did not exceed 77% and UV_{254} removal was around 54% under these conditions.

On the other hand, maximum COD removal reached 95% under a pH of 8, a coagulant dose of around 8 g/L, a flocculant dose of 8 ml/L, and agitation durations of 15–20 min. Concerning UV_{254} removal, the highest efficiency was 60% when the pH was 7, the coagulant dose was 8.8 g/L, the flocculant dose was 6 ml/L, and the agitation duration was 25 min. In this particular test, turbidity and COD removal percentages were 89% and 73%, respectively.

The evaluation of model quality and its fit is carried out using statistical techniques such as the analysis of variance (ANOVA) and various statistical parameters, including the coefficient of determination (R^2, adjusted R^2, and predicted R^2), as well as lack-of-fit tests [24, 1]. The results of the ANOVA are presented in Table 6.4.

Table 6.4 p-values for the studied responses (Turbidity, COD, UV_{254})

	Turbidity p-value	**COD p-value**	**UV_{254} p-value**
Model	< 0.0001	< 0.0001	< 0.0001
A-pH	0.2534	0.7142	< 0.0001
B-Coagulant dose	< 0.0001	0.0190	< 0.0001
C-Flocculant dose	0.0227	0.0190	0.0153
D-Agitation duration	< 0.0001	0.7142	0.5758
AB	< 0.0001	< 0.0001	0.1553
AC	< 0.0001	< 0.0001	0.6411
AD	0.0001	0.0365	0.7852
BC	0.0017	0.6541	0.0960
BD	< 0.0001	0.0056	0.0311
CD	0.1876	0.0001	0.0246
A^2	< 0.0001	0.0028	0.0072

	Turbidity p-value	COD p-value	UV_{254} p-value
B^2	0.0382	< 0.0001	0.5399
C^2	0.1161	0.2919	0.0129
D^2	< 0.0001	< 0.0001	0.4851
Lack of fit	0.5737	0.4845	0.10

For a 95% confidence level, a smaller p-value indicates a greater significance of the corresponding term (p-value < 0.05). The data presented in Table 6.4 demonstrate that, at a 95% confidence level, the linear effects of coagulant dose (B), flocculant dose (C), and agitation duration (D) have a significant impact on turbidity removal. These results agree with those reported by [39] and [20], highlighting the influence of FeCl3 and cationic polymer concentrations on turbidity. It should be noted that the interactions (AB, AC, AD, BC, and BD) and quadratic terms (A^2, B^2, and D^2) also significantly affect turbidity removal.

At the 95% confidence level, only the linear terms related to coagulant and flocculant doses significantly affect COD removal. However, all the interaction and quadratic terms significantly affect COD reduction, except the BC and C^2 terms.

On the other hand, concerning UV_{254}, we observe that the linear terms A (pH), B (coagulant dose), and C (flocculant dose) have a significant impact on UV_{254} reduction. However, only the CD interaction term and the quadratic terms A^2 and C^2 significantly influence this response. Based on the results presented in Table 6.4, it is evident that the interaction and quadratic terms exert the most significant influence on leachate pollution reduction.

In our study, the R^2 values for the three responses are close to 1, in the order of 0.97, 0.96, and 0.94 for turbidity, COD, and UV254, respectively. Furthermore, the difference between the predicted and the adjusted R^2 is less than 0.2

for all three responses, confirming the model's accuracy, as outlined by [24] (Table 6.5).

Table 6.5 Values of R-squared, Adj R-squared and Pred R-squared for the studied responses

	Turbidity	**COD**	**UV 254**
R-squared	0.9657	0.9702	0.9407
Adj R-squared	0.9358	0.9440	0.8889
Pred R-squared	0.8625	0.8750	0.7035

Second-order polynomial equations represent the responses for turbidity, COD, and UV254:

$$\begin{aligned} Turbidity = \; & +828.18452 - 46.09524 \times pH - 96.50298 \times Coagulantdose \\ & - 98.81548 \times Floculantdose + 40.97381 \times Agitation\ duration \\ & + 21.95313 \times pH \times Coagulantdose + 8.65625 \times pH \times Floculantdose \\ & - 1.68750 \times pH \times Agitation\ duration + 3.94531 \times Coagulantdose \\ & \times Floculantdose - 2.48438 \times Coagulantdose \times Agitation\ duration \\ & + 0.23125 \times Floculantdose \times Agitation\ duration \\ & - 10.21280 \times pH^2 - 4.43406 \times Coagulantdose^2 \\ & - 0.52195 \times Floculantdose^2 - 0.27351 \times Agitation\ duration^2 \end{aligned} \tag{6.1}$$

$$\begin{aligned} COD = \; & +37305.71429 + 4742.85714 \times pH - 12614.28571 \times Coagulantdose \\ & + 2325.71429 \times Floculantdose - 1349.71429 \times Agitation\ duration \\ & - 787.5 \times pH \times Coagulantdose - 225 \times pH \times Floculantdose \\ & + 30 \times pH \times Agitation\ duration - 18.75 \times Coagulantdose \times Floculantdose \\ & + 52.5 \times Coagulantdose \times Agitation\ duration - 33 \times Floculantdose \\ & \times Agitation\ duration + 173.57143 \times pH^2 + 1114.95536 \times Coagulantdose^2 \\ & + 13.39286 \times Floculantdose^2 + 23.74286 \times Agitation\ duration^2 \end{aligned} \tag{6.2}$$

$$\begin{aligned} UV_{254} = \; & +2.85692 - 0.40092 \times pH - 0.071771 \times Coagulantdose - 0.08 \\ & \times Floculantdose + 0.037367 \times Agitationduration + 0.017656 \times pH \\ & \times Coagulantdose + 2.25 \times 10^{-3} \times pH \times Floculantdose - 5.25 \times 10^{-4} \times pH \\ & \times Agitationduration + 0.010469 \times Coagulantdose \times Floculantdose - 5.59375 \\ & \times 10^{-3} \times Coagulantdose \times Agitationduration + 2.35 \times 10^{-3} \times Floculantdose \\ & \times Agitationduration + 0.021812 \times pH^2 - 6.93359 \times 10^{-3} \times Coagulantdose^2 - 4.95313 \\ & \times 10^{-3} \times Floculantdose^2 - 2.025 \times 10^{-4} \times Agitationduration^2 \end{aligned} \tag{6.3}$$

To minimize turbidity, COD, and UV_{254}, a numerical optimization analysis was conducted using the Design

Expert software. This analysis involves selecting the desired objectives for each factor and response.

All four variables were considered to determine optimal coagulation-flocculation conditions. The optimal conditions were a pH of 6.87, a coagulant dose of 7.98 g/L, a flocculant dose of 12 ml/L, and an agitation time of 22.8 min. To validate these optimal conditions, an experiment was conducted to assess the suitability of the proposed central composite design model. The results closely match the predicted values, giving turbidity, COD, and UV254 removal percentages of 92.57%, 86.36%, and 54.87%, respectively, for the three responses studied. This model is applicable in the analyzed range, allowing for the prediction and optimization of flocculation-coagulation in leachate treatment.

6.5 Conclusion

In conclusion, this study demonstrated the effectiveness of the coagulation-flocculation approach for landfill leachate treatment. Using response surface methodology and central composite design, we determined optimal conditions for pH, coagulant dose, flocculant dose, and agitation time to maximize turbidity, DOC, and UV254removal. Experimental results showed strong agreement with predicted values, confirming the reliability of our model. This approach has great potential for improving landfill leachate management by significantly reducing turbidity, DOC, and UV254 concentration. In addition, it can be generalized to other similar wastewater treatment applications, offering an effective and sustainable solution for mitigating the environmental impact of landfill leachates.

References

1. Adelodun B, Ajibade FO, Ogunshina MS, Choi KS (2019) Dosage and settling time course optimization of Moringa oleifera in municipal wastewater treatment using response surface methodology. Desalin Water Treat 167:45–56. https://doi.org/10.5004/dwt.2019.24616
2. Alismaeel ZT, Abbar AH, Saeed OF (2022) Application of central composite design approach for optimization of zinc removal from aqueous solution using a flow-by fixed bed bioelectrochemical reactor. Sep Purif Technol 287:120510. https://doi.org/10.1016/j.seppur.2022.120510
3. Anouzla A, Souabi S, Safi M, Abrouki Y, Loukili H, Rhbal H (2014) Waste to treat waste of landfill 250 leachates. Int J Environ Prot Policy 2:50–53. https://doi.org/10.11648/j.ijepp.20140202.12
4. Asouam S, Arabi M, Faik F, MorjaniZEAE, Abioui M, Diaz LF, Kostyuchenko Y (2021) Physicochemical characterization of the leachate of the tamelast landfill site, grand agadir (Morocco). Ecol Eng Environ Technol 22(5):65–72. https://doi.org/10.12912/27197050/139338
5. Bakraouy H, Souabi S, Digua K, Dkhissi O, Sabar M, Fadil M (2017) Optimization of the treatment of an anaerobic pretreated landfill leachate by a coagulation–flocculation process using experimental design methodology. Process Saf Environ Prot 109:621–630. https://doi.org/10.1016/j.psep.2017.04.017
6. Benaddi R, Ferkan Y, Bouriqi A, Ouazzani N (2022) Impact of landfill leachate on groundwater quality—a comparison between three different landfills in morocco. J Ecol Eng 23(11):89–94. https://doi.org/10.12911/22998993/153006
7. Bolto B, Xie Z (2019) The use of polymers in the flotation treatment of wastewater. Processes 7(6). https://doi.org/10.3390/pr7060374
8. Chaouki Z, Hadri M, Nawdali M, Benzina M, Zaitan H (2021) Treatment of a landfill leachate from Casablanca city by a coagulation-flocculation and adsorption process using a palm bark powder (PBP). Sci Afr 12:e00721. https://doi.org/10.1016/j.sciaf.2021.e00721
9. Curtis MJ, Alexander S, Cirino G, Docherty JR, George CH, Giembycz MA, Hoyer D, Insel PA, Izzo AA, Ji Y, MacEwan DJ, Sobey CG, Stanford SC, Teixeira MM, Wonnacott S, Ahluwalia A (2018) Experimental design and analysis and their reporting II: updated and simplified guidance for authors and peer reviewers. Br J Pharmacol 175(7):987–993. https://doi.org/10.1111/bph.14153

10. Dalhat Mu'azu N, Olusanya Olatunji S (2023) K-nearest neighbor based computational intelligence and RSM predictive models for extraction of Cadmium from contaminated soil. Ain Shams Eng J 14(4):101944. https://doi.org/10.1016/j.asej.2022.101944
11. Durakovic B (2017) Design of experiments application, concepts, examples : state of the art. Periodicals Eng Nat Sci 5(3). https://doi.org/10.21533/pen.v5i3.145
12. El-taweel RM, Mohamed N, Alrefaey KA, Husien S, Abdel-Aziz AB, Salim AI, Mostafa NG, Said LA, Fahim IS, Radwan AG (2023) A review of coagulation explaining its definition, mechanism, coagulant types, and optimization models; RSM, and ANN. Curr Res Green Sustain Chem 6:100358. https://doi.org/10.1016/j.crgsc.2023.100358
13. Elmaadawy K, Hu J, Guo S, Hou H, Xu J, Wang D, Liang T, Yang J, liang S, Xiao K, Liu B (2020) Enhanced treatment of landfill leachate with cathodic algal biofilm and oxygen-consuming unit in a hybrid microbial fuel cell system. Bioresour Technol 310:123420. https://doi.org/10.1016/j.biortech.2020.123420
14. Gonçalves BR, Borges W, Machado AEH, Trovó AG (2017) Biodiesel wastewater treatment by coagulation-flocculation : evaluation and optimization of operational parameters. J Braz Chem Soc 28:800–807. https://doi.org/10.21577/0103-5053.20160231
15. Gupta P, Bhargava A, Kumari R, Lodhi L, Tiwari R, Gupta PK, Bunkar N, Samarth R, Mishra PK (2019) Impairment of mitochondrial-nuclear cross talk in lymphocytes exposed to landfill leachate. Environ Health Insights 13:1178630219839013
16. Gupta A, Gaharwar US, Verma A, Rajamani P (2022) Municipal solid waste landfill leachate induced cytotoxicity in root tips of Vicia faba: environmental risk posed by non-engineered landfill. IJBB 59(11):1113–1125
17. Gupta A, Paulraj R (2017) Leachate composition and toxicity assessment: an integrated approach correlating physicochemical parameters and toxicity of leachatesfrom MSW landfill in Delhi. Environ Technol 38(13–14):1599–1605
18. Gupta A, Rajamani P (2015) Toxicity assessment of municipal solid waste landfill leachate collected in different seasons from Okhala landfill site of Delhi. J Biomed Sci Eng 8(06):357
19. Gupta A, Rajamani P (2016) Evaluation of the leachate composition and contamination potential of municipal solid waste landfill sites in Delhi. Int J Environ Waste Manage 18(4):285–302

20. Haddaji C, Ennaciri K, Driouich A, Digua K, Souabi S (2022) Optimization of the coagulation-flocculation process for vegetable oil refinery wastewater using a full factorial design. Process Saf Environ Prot 160:803–816. https://doi.org/10.1016/j.psep.2022.02.068
21. Hassan SS, El-Shafie AS, Zaher N, El-Azazy M (2020) Application of pineapple leaves as adsorbents for removal of rose bengal from wastewater : process optimization operating face-centered central composite design (FCCCD). Mol (Basel, Switzerland) 25(16):3752. https://doi.org/10.3390/molecules25163752
22. He L, Du P, Chen Y, Lu H, Cheng X, Chang B, Wang Z (2017) Advances in microbial fuel cells for wastewater treatment. Renew Sustain Energy Rev 71:388–403. https://doi.org/10.1016/j.rser.2016.12.069
23. Iwuozor K (2019) Prospects and challenges of using coagulation-flocculation method in the treatment of effluents. Adv J Chem Sect A 2:105–127. https://doi.org/10.29088/SAMI/AJCA.2019.2.105127
24. Kakoi B, Kaluli JW, Ndiba P, Thiong'o G (2017) Optimization of maerua decumbent bio-coagulant in paint industry wastewater treatment with response surface methodology. J Cleaner Prod 164:1124–1134. https://doi.org/10.1016/j.jclepro.2017.06.240
25. Kaur A, Bharti R, Sharma R (2023) Municipal solid waste as a source of energy. Mater Today Proc 81:904–915. https://doi.org/10.1016/j.matpr.2021.04.286
26. Kumar SS, Bishnoi NR (2017) Coagulation of landfill leachate by $FeCl_3$: process optimization using Box-Behnken design (RSM). Appl Water Sci 7(4):1943–1953. https://doi.org/10.1007/s13201-015-0372-1
27. Liu Z, Wu W, Shi P, Guo J, Cheng J (2015) Characterization of dissolved organic matter in landfill leachate during the combined treatment process of air stripping, Fenton, SBR and coagulation. Waste Manage 41:111–118. https://doi.org/10.1016/j.wasman.2015.03.044
28. Maja MM, Ayano SF (2021) The impact of population growth on natural resources and farmers' capacity to adapt to climate change in low-income countries. Earth Syst Environ 5(2):271–283. https://doi.org/10.1007/s41748-021-00209-6
29. Miao L, Yang G, Tao T, Peng Y (2019) Recent advances in nitrogen removal from landfill leachate using biological treatments—a review. J Environ Manage 235:178–185. https://doi.org/10.1016/j.jenvman.2019.01.057
30. Mishra S, Tiwary D, Ohri A, Agnihotri AK (2019) Impact of municipal solid waste landfill leachate on groundwater quality in Varanasi. India

Groundwater Sustain Dev 9:100230. https://doi.org/10.1016/j.gsd.2019.100230

31. El Mrabet I, Benzina M, Valdés H, Zaitan H (2020) Treatment of landfill leachates from Fez city (Morocco) using a sequence of aerobic and Fenton processes. Sci Afr 8:e00434. https://doi.org/10.1016/j.sciaf.2020.e00434

32. Muniz GL, Borges AC, da Silva TCF, Batista RO, de Castro SR (2022) Chemically enhanced primary treatment of dairy wastewater using chitosan obtained from shrimp wastes: optimization using a Doehlert matrix design. Environ Technol 43(2):237–254. https://doi.org/10.1080/09593330.2020.1783372

33. Oloibiri V, Chys M, De Wandel S, Demeestere K, Van Hulle SWH (2017) Removal of organic matter and ammonium from landfill leachate through different scenarios: operational cost evaluation in a full-scale case study of a Flemish landfill. J Environ Manage 203:774–781. https://doi.org/10.1016/j.jenvman.2016.09.055

34. Ozturk D, Dagdas E, Fil BA, Bashir MJK (2021) Central composite modeling for electrochemical degradation of paint manufacturing plant wastewater: one-step/two-response optimization. Environ Technol Innov 21:101264. https://doi.org/10.1016/j.eti.2020.101264

35. Parvin F, Tareq SM (2021) Impact of landfill leachate contamination on surface and groundwater of Bangladesh: a systematic review and possible public health risks assessment. Appl Water Sci 11(6):100. https://doi.org/10.1007/s13201-021-01431-3

36. Payandeh PE, Mehrdadi N, Dadgar P (2017) Study of biological methods in landfill leachate treatment. Open J Ecol 07(09). https://doi.org/10.4236/oje.2017.79038

37. Rifi SK, Souabi S, El Fels L, Driouich A, Nassri I, Haddaji C, Hafidi M (2022) Optimization of coagulation process for treatment of olive oil mill wastewater using Moringa oleifera as a natural coagulant, CCD combined with RSM for treatment optimization. Process Saf Environ Prot 162:406–418. https://doi.org/10.1016/j.psep.2022.04.010

38. Saritha V, Srinivas N, Srikanth Vuppala NV (2017) Analysis and optimization of coagulation and flocculation process. Appl Water Sci 7(1):451–460. https://doi.org/10.1007/s13201-014-0262-y

39. Singh B, Kumar P (2020) Pre-treatment of petroleum refinery wastewater by coagulation and flocculation using mixed coagulant: optimization of process parameters using response surface methodology (RSM). J Water Process Eng 36:101317. https://doi.org/10.1016/j.jwpe.2020.101317

40. Sonal S, Mishra BK (2021) Role of coagulation/flocculation technology for the treatment of dye wastewater : trend and future aspects. In: A. Singh, Agrawal M, Agrawal SB (éds.). Water Pollut Manage Practices, pp 303–331. Springer. https://doi.org/10.1007/978-981-15-8358-2_13
41. Spiniello I, De Carluccio M, Castiglione S, Amineva E, Kostryukova N, Cicatelli A, Rizzo L, Guarino F (2023) Landfill leachate treatment by a combination of a multiple plant hybrid constructed wetland system with a solar photoFenton process in a raceway pond reactor. J Environ Manage 331:117211. https://doi.org/10.1016/j.jenvman.2022.117211
42. Wijekoon P, Koliyabandara PA, Cooray AT, Lam SS, Athapattu BCL, Vithanage M (2022) Progress and prospects in mitigation of landfill leachate pollution: risk, pollution potential, treatment and challenges. J Hazard Mater 421:126627. https://doi.org/10.1016/j.jhazmat.2021.126627
43. Yang N, Damgaard A, Kjeldsen P, Shao L-M, He P-J (2015) Quantification of regional leachate variance from municipal solid waste landfills in China. Waste Manage 46:362–372. https://doi.org/10.1016/j.wasman.2015.09.016
44. Zailani LWM, Amdan NSM, Zin NSM (2018) Characterization of Leachate at Simpang Renggam Landfill Site, Johor, Malaysia. IOP Conf Ser Earth Environ Sci 140(1):012053. https://doi.org/10.1088/1755-1315/140/1/012053

© The Author(s), under exclusive license to Springer Nature Switzerland AG 2024
S. Souabi, A. Anouzla (eds.), *Landfill Leachate Treatment Techniques*, Springer Water
https://doi.org/10.1007/978-3-031-63157-3_7

7. Leachate Treatment Techniques: A Review of Oxidation and Physical Chemical Methods

Safaa Khattabi Rifi[1], Roukaya Bouyakhsass[1], Abdelaziz Madinzi[1], Alaa AEl Din Mahmoud[4], Tonni Agustiono Kurniawan[5], Abdelkader Anouzla[1, 1] ✉, Nor-edine Abriak[2], Younes Abrouki[3] and Salah Souabi[1]

(1) Faculty of Sciences and Techniques of Mohammedia, Department of Process Engineering and Environment Mohammedia, Hassan II University of Casablanca, Casablanca, Morocco
(2) IMT Lille Douai, GCE, 764 Bd Lahure, BP 10838, 59508 Douai, France
(3) Laboratory of Spectroscopy, Molecular Modeling, Materials, Nanomaterial, Water and Environment, CERNE2D, Faculty of Science, Mohammed V University in Rabat, Rabat, Morocco
(4) Environmental Sciences Department, Faculty of Science, Alexandria University, Alexandria, 21511, Egypt
(5) College of the Environment and Ecology, Xiamen University, Xiamen, 361102, China

✉ **Abdelkader Anouzla**
Email: aanouzla@gmail.com

Abstract
This review examines the physicochemical treatments and advanced oxidation techniques (AOT) used for landfill leachate. The treatments studied include coagulation-flocculation, oxidation, adsorption, precipitation, and combined treatments. The advantages and limitations of each treatment are evaluated based on criteria such as pH, hydraulic retention time, organic loading rate, leachate characteristics, and treatment performance. Coagulation-flocculation processes using chemicals such as ferric chloride and aluminum sulfate have shown effective COD reduction, particularly for young leachates. Good management of chemicals and sludge is essential to maximize the benefits of this method. In conclusion, choosing the most appropriate treatment for leachate depends on its specific characteristics, technical applicability, potential constraints, regulatory requirements, cost-effectiveness, and long-term environmental impacts.

Keywords Municipal solid waste – Coagulation-flocculation – Oxidation – Precipitation – Adsorption

---

7.1 Introduction

Waste management is one of the main challenges facing our societies [1]. With ever-increasing and more diverse consumption worldwide, the quantity of waste produced continues to rise, creating enormous environmental and human health problems [2]. This situation is much more worrying in developing countries [3]. Most municipal waste is disposed of in landfills to reduce and regulate its effects on the surrounding environment [4]. To this end, landfills

became a source of soil, air, and human health pollution instead of a waste disposal solution [5, 6]. Several constraints are to be reported for solid waste management, such as the regulations, the lack of financial means, the inadequacy of the material means mobilized in some cities, and the low collection rate [7].

Today, many controlled landfills are operational in Morocco, and others are being rehabilitated. All these landfill sites are facing the problem of leachate management. Leachate is generated by the sanitary landfill process and is considered a complex organic wastewater [8, 9]. The amount of leachate generated is closely linked to various factors, such as precipitation levels, evapotranspiration, surface water runoff, infiltration, groundwater intrusion into the landfill, and the degree of waste compaction [10–13]. Leachate impacts soil, surface water, and groundwater [14–16]. Alemayehu et al. [17] show that the impact of leachates on water resources increases during the wet season when contaminant dispersion is accelerated by maximum infiltration and surface runoff. It has been identified as a possible cause of contamination of soil, groundwater, and surface water if it is not collected, treated, and disposed of properly [18–20]. In addition, groundwater and surface water can degrade slowly due to the infiltration of these contaminants into groundwater or their passage into watercourses [19, 21]. Unlike groundwater contamination, which results from the infiltration and diffusion of leachates in the permeable or fissured subsoil, surface water pollution by leachates can occur through overflow and liquid flow in the hydrographic network, either abruptly or gradually [22]. Developed countries have established rigorous standards and regulations for collecting, containing, treating, and disposing of leachates. These standards also emphasize regular monitoring of surface and groundwater near landfill sites.

Leachate characteristics are influenced by several factors, including the solid waste collection system, the age of the landfill, which affects waste stabilization, the type and composition of the discarded waste, the ability of solid waste components to dissolve, etc. [23]. A high proportion of low-molecular organic matter, such as volatile organic acids, characterizes young leachates. They have a high COD, a significant amount of total organic carbon (TOC), a high biological oxygen demand (BOD_5), and a BOD_5/COD ratio greater than 0.6. On the other hand, stabilized leachates (more than 10 years old) are characterized by a relatively low chemical oxygen demand (COD) (less than 4000 mg/L), a slight basicity (pH greater than 7.5), and low biodegradability (BOD5/COD ratio less than 0.1) [24, 25].

Leachate must be treated appropriately to avoid environmental damage. To comply with the criteria for leachate discharge into receiving water bodies, removing organic matter, ammoniacal nitrogen, and other toxic components [26].

Leachates can be treated in various ways, including biological techniques such as aerobic or anaerobic biodegradation [27], physical techniques such as filtration, activated carbon adsorption, ozonation or membrane separation [28], Chemical techniques such as coagulation-flocculation [29–31], chemical precipitation [32], and advanced chemical oxidation [33]. It has been demonstrated that treating landfill leachate with two or more treatment technologies in combination is more successful and efficient than doing so alone [34, 35]. This is explained by the ability of two- and three-step treatments to combine the advantages of a single treatment while getting around their limitations [36, 37]. It is well recognized that the three primary factors to be taken into account when recommending a sustainable treatment technology or combination of landfill leachate treatment

technologies are (1 treatment efficiency, (2 cost of the treatment system, and (3 environmental effect of the treatment system [38, 39].

This study aims to comprehensively review the various physico-chemical and oxidation techniques for leachate treatment. This article highlights the importance of these methods in addressing the complex challenges associated with leachate management and pollution control. By exploring and evaluating these techniques in detail, we aim to provide valuable information to guide researchers and practitioners in making informed decisions about the most appropriate and practical approaches to leachate treatment.

7.2 Types of Pollutants Present in Leachate

Landfills represent a potential source of pollution that can contaminate water, air, soil and the whole environment [40]. Once waste is deposited in landfills, it undergoes degradation processes characterized by complex biological and physico-chemical reactions [41, 42]. When water infiltrates these areas, it produces leachates of organic and mineral elements, which mainly cause organic and metallic pollution due to the natural biodegradation of the deposited waste. This process releases numerous toxic substances into the ecosystem [22]. The major types of pollutants contained in the leachates are as follows.

7.2.1 Pollution by Organic Matter

Improperly managed municipal solid waste (MSW) landfills decompose organic material and generate enormous amounts of leachate, which contains synthetic chemicals and decaying organic matter in the form of dissolved organic matter (DOM) or particulate organic matter [43].

7.2.2 Pollution by Heavy Metals

In developing countries, landfills are reliable recipients of various industrial wastes. Industrial waste is a significant source of heavy metals, which can cause cancer. If not handled properly, they can have irreversible environmental and human health effects [44]. The main heavy elements in waste disposal leachate are arsenic, copper, zinc, lead, cadmium, chromium, and mercury [45]. When different items, including batteries, consumer electronics, ceramics, lightbulbs, and glass, are disposed of in landfills, these pollutants are created [46].

7.2.3 Pollution by Microorganisms

Landfills provide a suitable environment for the growth and development of diverse microbial populations due to the variety of substrates available to meet their physiological needs [47]. Leachate from waste-related moisture in landfills contains microbial pathogens [48]. The source of the landfill, the amount of water available, the age of the landfill, and the toxicity of the waste components present all affect the complexity and composition of the microbial community in any landfill leachate. Numerous fungi, coliform and fecal coliform bacteria, and aerobic, psychrophilic, and mesophilic bacteria can also contaminate leachate.

7.2.4 Pollution by Nitrogen

Leachate containing harmful pollutants, particularly NH_4^+, can leak into the underlying groundwater, endangering the ecosystem and the general public's health, according to [49]. NH_3–N is one of the main contaminants, among others. Untreated NH_3–N harms organisms due to its high concentration, which can hinder nitrification and deplete dissolved oxygen.

7.2.5 Fatty Acids

Fatty acids are essential components of the animal and plant world. They are divided into significant families according to the number of ethylenic double bonds they possess saturated (no double bonds) branched or unbranched, monounsaturated (single, double bond), and polyunsaturated (at least two double bonds) fatty acids [39].

7.3 Treatment Techniques

Leachate, a complex and potentially hazardous liquid generated by landfill sites, represents a worldwide environmental challenge. Leachates require effective treatment to prevent their harmful effects on ecosystems and public health. It is, therefore, necessary to explore various treatment techniques capable of effectively eliminating or attenuating the multiple pollutants present in leachates.

7.3.1 Coagulation-Flocculation

Coagulation-flocculation treatment is a fundamental process for treating wastewater by aggregating impurities into easily removable flocs. This essential technique finds wide application in ensuring the quality and safety of water resources and wastewater discharge.

Amor et al. [50, 51] studied the effect of coagulation/flocculation with calcium hydroxide, $Ca(OH)_2$; aluminum sulfate, $Al_2(SO_4)^{3-}18H_2O$; ferrous sulfate, $FeSO_4{}^{-}7H_2O$ and ferric chloride, $FeCl_3$–$6H_2O$, assessing the role of different initial pH and chemical concentration. The results show a higher COD reduction at pH 6 using 4 g/L $FeCl_3$. Calcium hydroxide showed low COD removal capacity. Aluminum sulfate achieved a 39% reduction in

COD, 26% in total polyphenols, and 6% in organic matter [51]. Maleki et al. [52] studied the effect of treating leachate with a coagulation-flocculation process. The effects of pH and different ferric chloride and aluminum sulfate concentrations were compared. Results showed that aluminum sulfate removed 18% of COD and 90% of heavy metals at a pH of 6.5. Ferric chloride showed 28% COD removal and 86% heavy metal removal at pH 10 (optimum for ferric chloride). Samadi et al. [53] show that coagulation-flocculation can effectively treat leachates from the Hamadan landfill. Various coagulants were tested with different dosages and pH values to assess their efficiencies in removing chemical oxygen demand (COD) and suspended solids. The results showed that at pH = 12, COD removal efficiencies were in the order of 60, 62.33, and 70.66% using polyaluminum chloride with a concentration of 2500 mg/L, alum with a dose of 1000 mg/L and ferrous sulfate with a dose of 1500 mg/L respectively. For suspended solids, the results showed that at pH = 12 the removal efficiency was 39.14, 58.37, and 35.58%, respectively, using poly aluminum chloride with a concentration of 2500 mg/L, alum with a concentration of 1500 mg/L and ferrous sulfate with a concentration of 2500 mg/L [53].

Based on these results, it was concluded that ferrous sulfate is the best coagulant for COD removal. The coagulation-flocculation process can be used effectively, particularly for young leachates, before post-treatment (polishing) for partially stabilized leachates.

The coagulants $FeCl_3$ and $Al_2(SO_4)_3$ were tested, and it was found that the $FeCl_3$ treatment was effective at a pH of 6.5, while the optimum pH for $Al_2(SO_4)_3$ was 5.3 [54]. Assou et al. [54] have shown that coagulation-flocculation with $FeCl_3$ and Al_2 $(SO_4)_3$ is highly effective in reducing turbidity, with removal rates reaching 95 and 98%,

respectively. COD removal with $FeCl_3$ and $Al_2(SO_4)_3$ reached 67% and 60%, respectively, at optimum concentrations of 18.5 mmol/L Fe^{3+} and 5.82 mmol/L Al_3^+.Aluminum sulfate produced less sludge than ferric chloride, with a sludge volume of around 230 ml/L, while the sludge volume produced by $FeCl_3$ was around 800 ml/L. The physico-chemical coagulation-flocculation process has proved effective and economically viable for treating leachate from landfills [54]. Cheng et al. [55] have optimized the coagulation-flocculation leachate treatment process. A central composite design method optimized operating parameters such as alum dosage, pH and mixing speed. The results showed that optimum chemical oxygen demand (COD) removal of 54% was achieved with a low alum dosage, a pH of 8.5 and a mixing speed of 100 rpm. Zakaria et al. [56] examined the effectiveness of zirconium tetrachloride ($ZrCl_4$) and tin tetrachloride ($SnCl_4$) as coagulants for leachate treatment. Color and chemical oxygen demand (COD) were used as performance indicators. Results showed that $SnCl_4$ is a good coagulant, with color and COD removal of 97 and 77%, respectively. Campos et al. [57] studied the combined treatment of leachate from the Morro do Céu landfill in Niterói, RJ, and wastewater samples from the Icaraí wastewater treatment plant with different proportions. Physico-chemical treatment followed by expulsion through an underwater outfall was simulated. Different coagulants, such as ferric chloride, aluminum sulfate, Tanfloc SG, Tanfloc SL and Panfloc™ Hiper Plus, were used at different concentrations. Statistical analysis showed that Tanfloc SG performed better than the other coagulants, without requiring the addition of polyelectrolytes for flocculation. Toxicity tests were also carried out, showing that the toxicity of the samples was below the limit set by current legislation.

Coagulation-flocculation is an effective method for removing suspended particles and other pollutants from leachates. Still, it requires the use of chemicals, generates sludge, and can be sensitive to variations in water quality. Proper management of chemicals and sludge and optimization for specific conditions are essential to maximize coagulation-flocculation benefits.

7.3.2 Precipitation

Precipitation treatment is vital in water and wastewater treatment, effectively removing contaminants by forming insoluble particles that settle out. This method is crucial in purifying water sources and ensuring environmental compliance.

Reynier et al. [58] treated leachates returning from alkaline soil washing contaminated with organic (pentachlorophenol) and inorganic (arsenic, chromium and copper) compounds using the precipitation technique. The results show that initial contaminant concentrations in the alkaline leachate are high, but copper, pentachlorophenol and arsenic concentrations were reduced thanks to precipitation. However, precipitation did not effectively remove polychlorinated dioxins and furans (PFDs) beyond 17% [58]. Tests with activated carbon and resins failed to remove arsenic and chromium effectively in highly alkaline environments. However, the Lewatit SR7 anion resin proved to be the most effective for the simultaneous removal of copper, pentachlorophenol, and DFs, with high recovery yields [58]. Zhang et al. [59] used chemical precipitation to remove ammonium (NH_4^+) from leachates. Results showed that the optimum pH for ammonium removal was 9.5. A controlled molar ratio of Mg_2^+:NH_4^+:PO_4^{3-} of 1.15:1:1 enabled effective ammonium removal while avoiding a high PO43 concentration in the effluent. Different chemicals generated varying salt

concentrations, adding MgO and H_3PO_4 reducing salt concentration while maintaining good ammonium removal. Addition of $Ca(H_2PO_4)_2$–H_2O and $MgSO_4$–$7H_2O$ gave the lowest removal rates [59]. Huang et al. [60] studied the technique of ammonia removal from landfill leachates by struvite precipitation, using waste phosphoric acid as the phosphate source. The results show that the presence of Al^{3+} ions in waste phosphoric acid has a significant impact on struvite precipitation. Still, an ammonia removal rate was obtained, similar to pure phosphate salts. However, a large amount of NaOH was required to neutralize the acid in the spent phosphoric acid. To solve this problem, inexpensive magnesium was proposed as an alkaline reagent in struvite precipitation [60]. Using a Mg:N:P molar ratio of 3:1:1 with inexpensive MgO, an ammonia removal rate of 83% was achieved, with a residual phosphate concentration of 56 mg/L [60]. An economic analysis showed that using phosphoric acid and inexpensive MgO reduced chemical costs by 68% compared to pure chemicals. In addition, using an anaerobic biological filter in post-treatment demonstrated that the high concentration of Mg^{2+} in the struvite precipitation effluent had no inhibiting effect on the performance of the biological treatment [60]. The use of chemical precipitation and biosorption has been studied by [61] to recover NH_4^+–N from landfill leachate. Optimum conditions for chemical precipitation were determined with a pH of 10, a contact time of 30 min, and an Mg/N/P molar ratio of 1:1:1, achieving approximately 88% NH_4^+–N removal with an initial concentration of 610–640 mg/L. Next, banana peels (BP) and modified biosorbents (NMBP) were investigated to purify further water containing NH_4^+–N. Results showed that NMBP significantly enhanced NH_4^+–N removal by sorption [61]. Zeng et al. [62] studied the effect of

microbiologically induced calcite precipitation (MICP) in mineralizing toxic metals (lead and cadmium) present in leachates. Results show that in treatment trials of Pb^{2+} −contaminated synthetic leachate, 100% of the 20 mg/L Pb^{2+} was removed, with a maximum urease activity of only 20.96 U/ml. Microscopic and spectroscopic analysis of the precipitates indicated the formation of agglomerated square particles, 76.9% of which ranged in size from 33.93 to 57.06 µm. Spectroscopic and diffraction analyses confirm that the precipitates are composed mainly of calcite crystals, while lead is fixed in hydrocerussite [62]. Pb–MICP precipitates are stable even in continuous acid degradation, with only 1.76% of lead released after 15 days [62]. In a test treatment of real landfill leachate, all the Pb^{2+} and Cd^{2+} present were simultaneously mineralized, thus confirming the feasibility of PCM for treating toxic metals in landfill leachates [62]. Kim et al. [63] studied the effect of carbonates in reducing alkaline leachate production from basic oxygen furnace (BOF) slags. BOF slag was treated with $NaHCO_3$ solutions at different concentrations for 48 h. The treatment lowered the pH of the leachate by reducing the dissolution of free CaO present in the slag. Spectroscopic and thermogravimetric analysis showed the formation of $CaCO_3$ precipitates on the surface of the treated slag. Carbon dioxide as an alternative carbonate source enhanced $CaCO_3$ precipitation and reduced leachate pH [63]. Electron microscopy observations confirmed a greater quantity of precipitated $CaCO_3$ on slag treated with CO_2 gas [63]. This study highlights the mechanism of leachate pH reduction and the effect of the carbonate source, which may contribute to a more environmentally safe management of oxygen furnace slag [63]. A comparative study of hydroxide precipitation, sulfide precipitation, carbonate precipitation and the

struvite formation process for the removal of manganese and ammoniacal nitrogen from electrolytic manganese residue leachate (EMMRL) was carried out by [64]. The chemical equilibrium model—Visual MINTEQ—was applied to simulate chemical reactions and optimize chemical dosages in the removal of manganese and ammoniacal nitrogen. The phase transition, morphology and valence state of the precipitates were characterized by X-ray diffraction (XRD), scanning electron microscopy (SEM), Fourier transform infrared spectroscopy (FT-IR) and X-ray photoelectron spectroscopy (XPS). The results indicate that carbonate precipitation is more effective than the other two methods in removing manganese and ammoniacal nitrogen [64].

7.3.3 Adsorption

Adsorption is a treatment technique relying on the adherence of contaminants to solid surfaces, typically activated carbon or other adsorbent materials. This method is pivotal in removing various pollutants and achieving water quality standards.

A series of low-cost conventional leachate treatment processes have been proposed to minimize the environmental consequences of leachate ingress [65]. The main objective was to improve the removal of chemical oxygen demand (COD) and heavy metals while reducing the volume of sludge produced and the consumption of materials [66]. Walnut shell adsorption was used with lime treatment to remove heavy metals and low molecular weight organic compounds. Samples taken from a landfill in Iran were used to study the influence of lime and walnut shell dosages and pH on removal efficiency. The results showed a COD removal rate of 43.24% under optimum conditions. Concentrations of heavy metals, including nickel, cadmium, zinc and lead, were also effectively reduced [67]. In addition, using the inactive surface of

crushed walnut shells was preferred to reduce costs compared with using the activated surface [67]. Halim et al. [68] examined the adsorption of ammonia nitrogen and chemical oxygen demand (COD) in semi-aerobic leachate from a specific landfill site. Three types of adsorbent, namely zeolite, activated carbon and new composite support, were studied regarding adsorption isotherms and kinetics. The results show that all adsorbents agree well with the Langmuir and Freundlich isotherm models ($R_2 >$ 0.9) for ammonia adsorption [68]. A comparison reveals that the composite adsorbent has a better adsorption capacity for ammonia nitrogen than zeolite and activated carbon, and a capacity comparable to that of activated carbon for COD. Kinetic studies show that ammonia adsorption on the new composite adsorbent and zeolite follows different kinetic models such as pseudo-first order, pseudo-second order, the Elovich model and the intra-particle diffusion model, with pseudo-second order as the dominant model [68]. COD adsorption mainly corresponds to the pseudo-second-order kinetic model, while activated carbon follows the pseudo-first-order and intra-particle diffusion models. These results highlight the adsorption properties of the different adsorbents and provide useful information for landfill leachate treatment, particularly for removing ammoniacal nitrogen and COD [68]. Walnut shells were studied as adsorbent media to treat stabilized landfill leachate. Adsorption isotherm results showed that the Langmuir isotherm was more suitable than the Freundlich isotherm [69]. Mohammad-Pajooh et al. [70] compared the chemical oxygen demand (COD) and trace metal adsorption performance of different types of granular activated carbon in landfill leachate. They found that granular activated carbon (GAC) 1 was unsuitable for treating landfill leachate, while GAC 2 was able to reduce COD in leachate pretreated by dissolved air flotation (DAF).

GAC 3 can be used either in the biological stage alone, or in the tertiary stage after the DAF unit. In addition, GAC 3 has also demonstrated trace metal adsorption efficiency, removing up to 66% of copper, 64% of cobalt, 48% of chromium, 47% of manganese, 43% of nickel and 25% of zinc [70]. The results of this study underline the importance of choosing the right type of granular activated carbon for landfill leachate treatment, particularly for COD and trace metal removal. Pedro-Cedillo et al. [71] showed that the adsorption process with granular activated carbon was capable of removing over 99% of the color, chemical oxygen demand (COD), total nitrogen and total organic carbon (TOC) present in raw leachate samples. Adsorption treatment offers several significant advantages, such as high efficiency, broad spectrum of contaminants, selectivity, ease of use, reversibility and relatively low cost. However, it is important to note that adsorption can have some limitations, such as saturation of adsorbent materials, degradation of adsorbents over time and the need to properly manage and dispose of adsorbed materials once used.

7.3.4 Oxydation

Oxidation treatment is a vital process in water and wastewater management, harnessing the power of chemical reactions to break down and remove pollutants. This technique is instrumental in addressing organic contaminants, pathogens, and odorous compounds.

Ozonation is widely used as a leachate treatment method as a polishing step after a biological process [72]. Ozone is a powerful oxidant capable of rapidly transforming recalcitrant organic compounds into lower molecular weight compounds, thus preventing the accumulation of toxic residues [73]. Compounds present in leachates determine ozone reactivity, with a particular affinity for compounds containing aromatic rings and C–C bonds. pH

plays an important role in ozone decomposition, with direct electrophilic attack or generating –OH radicals. At acid pH, ozone specifically targets organic compounds, while at alkaline pH, –OH radicals rapidly oxidize most target compounds. Alkaline pH is, therefore, a key parameter in AOT-based ozonation [74]. Amr et al. [75] used persulfate in advanced oxidation to treat stabilized landfill leachate in an ozone reactor. A central composite response design (CMR) was used to determine optimal operating conditions, considering ozone and persulfate dosages, pH, and reaction time. The results showed that the quadratic models for the measured responses (COD, color, NH_3–N, and ozone consumption) were significant. Optimal conditions included a reaction time of 210 min, 30 g/m^3 ozone, 1 g/1 g persulfate per initial COD (S_2O_8), and a pH of 10. Experimental results agreed with predicted models, showing COD, color, and NH_3–N removal rates of 72, 96, and 76%, respectively, and ozone consumption of 0.60 kg O_3/kg COD. Amor et al. [76] showed that high pH values and hydrogen peroxide as a co-oxidant enhanced ozonation efficiency. These conditions led to the production of hydroxyl radicals, resulting in better organic load removal and improved biodegradability. The O_3/H_2O_2 process with 4 g H_2O_2 per liter showed the best performance, with 45% chemical oxygen demand (COD) removal, 89% color removal and an increase in the BOD5/COD ratio of the leachate. Ozonation alone also showed significant removal of recalcitrant organic matter (43% of COD) and increased leachate biodegradability, making it a viable option as a pre-treatment for biological biodegradation.

7.3.5 Evaporation

Evaporation treatment is a valuable method employed in water and wastewater management, utilizing heat-driven

processes to separate water from contaminants. Parameters such as pH, temperature and evaporation rate have been studied by [77] to understand their impact on the properties of the leachate condensate liquid. Experimental results have shown that the initial pH of the concentrate affects the concentration of chemical oxygen demand (COD) and ammoniacal nitrogen (NH_3–N) in the condensate. Acidic pH leads to high COD concentration, while alkaline pH results in high NH_3–N concentration. In addition, operating temperature influences the condensate's pH, total dissolved solids (TDS), ammoniacal nitrogen, and UV254 absorbance. TDS and NH_3–N in the condensate increase with temperatures up to 70 °C, then decrease [77]. This study evaluates a pilot system using natural evaporation techniques to treat sanitary landfill leachate. Generally, TDS is mainly attributable to ammonia evaporating into the condensate and is closely related to NH_3–N concentration. As the evaporation rate increases, TDS and NH_3–N decrease in the middle of the evaporation stage, then increase slightly in the last phase [78]. Alcântara et al. [79] have installed a pilot system in a region with a semi-arid climate, and the results obtained have been compared with a similar experiment carried out in a humid subtropical climate. The results show that in Juazeiro do Norte, CE, between October 2010 and September 2011, average daily leachate evaporation was 3.2 times higher than that observed in Florianópolis, SC, between January 2008 and September 2008. These results confirm the potential of solar energy for leachate treatment in semi-arid climates.

7.3.6 Combined Treatments

Combined treatment methods represent a multifaceted water and wastewater treatment approach, integrating various techniques to address complex contamination

challenges. By synergizing different treatment processes, such as coagulation-flocculation, adsorption, oxidation, and more, combined treatments offer comprehensive solutions to ensure water quality and environmental compliance.

Amor et al. [50, 51] studied the effect of the Fenton and solar photo-Fenton processes in combination with coagulation/flocculation pretreatment. The results show greater reductions in DOC (75%) than solar photo-Fenton alone (54%). In the combined treatment (coagulation/flocculation and solar photo-Fenton), a COD reduction of 50% was noted after chemical oxidation, with 110 kJ/L of accumulated UV energy and H_2O_2 consumption of 116 mM.

The combination of polyaluminium chloride (PAC) coagulation followed by the microelectrolysis-Fenton (MEF) process resulted in excellent decontamination performance, with predicted chemical oxygen demand (COD) removal of 90.27% and humic acid (HA) removal of 93.79% [80]. de Almeida et al. [81] focused on evaluating a combined treatment process (coagulation-flocculation with lime and nanofiltration) to treat leachates from the Seropédica landfill in Brazil. Laboratory tests were carried out to determine the optimum lime dosage, and it was found that a high pH favored the removal of ammoniacal nitrogen during the coagulation-flocculation process. The mixture was then decanted for 6 h with slow agitation to facilitate ammonia stripping before applying the nanofiltration step at a pressure of 8 bar. After ammonia stripping, the NH_3–N concentration was reduced from 1236 to 353 mg/L, corresponding to a removal efficiency of 71%. At the end of the combined process, the removal percentages for total organic carbon (TOC), humic substances (HS) and chemical oxygen demand (COD) were 89, 80 and 94%, respectively. These results demonstrate the effectiveness of the combined process in removing recalcitrant compounds

and ammonia. Treatment with lime coagulation-flocculation followed by nanofiltration has proved promising for treating leachate from the landfill studied. Bellouk et al. [82] treated leachates by coagulation-flocculation combined with advanced oxidation processes. Primary pretreatment by coagulation-flocculation, using 20 g/L $FeCl_3$ (optimum coagulant dose), removed 71% of chemical oxygen demand (COD) and 95% of turbidity. Next, a secondary treatment was applied using different combinations of two inorganic oxidants (potassium persulfate, $K_2S_2O_8$, and hydrogen peroxide, H_2O_2) used simultaneously or separately with UV-A irradiation and/or ultrasound (US). The results showed that oxidation systems in the presence of H_2O_2 were the most effective compared with the other methods. In particular, the H_2O_2-PS-US-UV-A system achieved the highest COD removal, at 89%, while combinations involving potassium persulfate were the least effective (PS alone 19%, PS-UV-A 22%, PS-US 32%, and PS-US-UV-A 35%). Kwarciak-Kozłowska and Fijałkowski [83] studied the effectiveness of advanced oxidation (AOP) and adsorption (BC) for landfill leachate treatment. Two AOP systems were used: I with H_2O_2 and II with SPC (sodium percarbonate). The systems applied showed significant reductions in chemical oxygen demand (COD) and total organic carbon (TOC). Mojiri et al. [84] treated concentrated leachates using a combination of electro-ozonation and a sequential batch reactor (SBR) reinforced with composite adsorbents. Results showed high efficiencies in COD, color, and nickel removal. Ozonation was carried out at different reaction times (30–120 min), while supercritical water oxidation (ScWO) was performed at a temperature of 600 °C, a pressure of 23 MPa, and a space–time of 29–52 s. The results showed that the combination of ozonation (30 min) and the supercritical water oxidation process (O_3–30'/ScWO) was the most

effective for leachate degradation evaluated [85]. Afsharnia et al. [86] treated the leachate using an evaporation-distillation process. Results showed that 90% of the leachate was distilled and that chemical oxygen demand (COD) was reduced by 83.56% during evaporation-distillation. Zielińska et al. [87] showed that leachate treatment efficiency was highest when the membrane process was combined with the addition of powdered-activated-carbon (PAC), indicating that activated carbon adsorption played a dominant role in leachate treatment. In particular, adsorption with a PAC dose of 3 g/L and an adsorption time of 30 min contributed most to COD and color removal from the leachate, with removal efficiencies of 73.1 and 94.4% respectively.

7.4 Conclusion

Leachate, a complex organic effluent produced by landfill processes, poses environmental dangers. Numerous elements, such as the amount of precipitation, the compaction of waste, and groundwater intrusion, impact the amount of leachate produced. Before being released into bodies of water, leachate must be treated to eliminate hazardous substances, organic materials, and ammoniacal nitrogen. Leachate treatment has been researched using physico-chemical methods like precipitation and coagulation-flocculation. The effective COD reduction has been demonstrated by coagulation-flocculation methods utilizing ferric chloride and aluminum sulfate, particularly for young leachates. Proper chemical and sludge management is essential to get the most out of this approach. To eliminate certain pollutants from leachate, precipitation approaches such as chemical precipitation and microbiologically induced calcite precipitation have been investigated. These techniques have successfully lowered metal, ammonium, and other chemical

concentrations. Each method has strengths and weaknesses. Although coagulation-flocculation produces sludge and necessitates chemicals, it removes suspended particles. Variations in water quality may impact precipitation processes, which can be used to remove particular contaminants selectively. Overall, physico-chemical treatment approaches provide potential solutions for leachate treatment. Still, optimization and consideration of site-specific variables are required to assure their efficacy and environmental sustainability—improved results from ongoing study and development in this area.

References

1. Bouyakhsass R, Bouaouda S, Taleb A, Madinzi A, Baudu M, Souabi S (2023) Critical review for the quantification of leachate production: evaluation example for the “Oum Azza” public landfill in Rabat city, Morocco. Environ Qual Manag
2. Bouaouda S, Souabi S, Bouyakhsass R, Taleb A, Madinzi A, Anouzla A (2023) Techniques for treating leachate discharges: a critical review. Euro-Mediterranean J Environ Integr 1–27
3. Bakraouy H, Souabi S, Digua K, Dkhissi O, Sabar M, Fadil M (2017) Optimization of the treatment of an anaerobic pretreated landfill leachate by a coagulation–flocculation process using experimental design methodology. Process Saf Environ Prot 109:621–630
4. El Fadili H, Ali MB, Touach N, El Mahi M (2022) Ecotoxicological and pre-remedial risk assessment of heavy metals in municipal solid wastes dumpsite impacted soil in morocco. Environ Nanotechnol Monit Manag 17:100640
5. Blair J, Mataraarachchi S (2021) A review of landfills, waste and the nearly forgotten nexus with climate change. Environments 8:73
6. Ozbay G, Jones M, Gadde M, Isah S, Attarwala T (2021) Design and operation of effective landfills with minimal effects on the environment and human health. J Environ Public Health
7. Ebekozien A, Aigbavboa C, Emuchay FE, Aigbedion M, Ogbaini IF, Awo-Osagie AI (2022) Urban solid waste challenges and opportunities to

promote sustainable developing cities through the fourth industrial revolution technologies. Int J Build Pathol Adapt

8. Teng C, Zhou K, Peng C, Chen W (2021) Characterization and treatment of landfill leachate: a review. Water Res 203:117525

9. Gupta A, Paulraj R (2017) Leachate composition and toxicity assessment: an integrated approach correlating physicochemical parameters and toxicity of leachates from MSW landfill in Delhi. Environ technol 38(13–14):1599–1605

10. Bhalla B, Saini MS, Jha MK (2013) Effect of age and seasonal variations on leachate characteristics of municipal solid waste landfill. Int J Res Eng Technol 2:223–232

11. Bouyakhsass R, Souabi S, Rifi SK, Taleb A, Pala A, Madinzi A (2023) Optimization of coagulation-flocculation for landfill leachate treatment: an experimental design approach using response surface methodology. Environ Nanotechnol Monit Manag 20:100841

12. Gupta A, Rajamani P (2015) Toxicity assessment of municipal solid waste landfill leachate collected in different seasons from Okhala landfill site of Delhi. J Biomed Sci Eng 8(6):357

13. Gupta A, Rajamani P (2016) Evaluation of the leachate composition and contamination potential of municipal solid waste landfill sites in Delhi. Int J Environ Waste Manag 18(4):285–302

14. Ololade OO, Mavimbela S, Oke SA, Makhadi R (2019) Impact of leachate from northern landfill site in Bloemfontein on water and soil quality: implications for water and food security. Sustainability 11:4238

15. Gupta A, Gaharwar US, Verma A, Rajamani P (2022) Municipal solid waste landfill leachate induced cytotoxicity in root tips of Vicia faba: environmental risk posed by non-engineered landfill. IJBB 59(11):1113–1125

16. Gupta P, Bhargava A, Kumari R, Lodhi L, Tiwari R, Gupta PK, Bunkar N, Samarth R, Mishra PK (2019) Impairment of mitochondrial-nuclear cross talk in lymphocytes exposed to landfill leachate. Environ Health Insights 13:1178630219839013

17. Alemayehu T, Mebrahtu G, Hadera A, Bekele DN (2019) Assessment of the impact of landfill leachate on groundwater and surrounding surface water: a case study of Mekelle city Noerthern Ethiopia. Sustain Water Resour Manag 5:1641–1649

18. Al Raisi SAH (2022) The generation, composition and fate of landfill leachate: a review. Eur J Environ Earth Sci 3:105–109

19. Udosen NI (2022) Geo-electrical modeling of leachate contamination at a major waste disposal site in south-eastern Nigeria Model. Earth Syst Environ 8:847–856
20. Wang H, Ding K (2022) Effect of self-made TiO2 nanoparticle size on the performance of the PVDF composite membrane in MBR for landfill leachate treatment. Membranes 12:216
21. Rajoo KS, Karam DS, Ismail A, Arifin A (2020) Evaluating the leachate contamination impact of landfills and open dumpsites from developing countries using the proposed Leachate Pollution Index for Developing Countries (LPIDC). Environ Nanotechnol Monit Manag 14:100372
22. Parvin F, Tareq SM (2021) Impact of landfill leachate contamination on surface and groundwater of Bangladesh: a systematic review and possible public health risks assessment. Appl water Sci 11:100
23. Banch TJH, Hanafiah MM, Alkarkhi AFM, Salem AM (2019) Statistical evaluation of landfill leachate system and its impact on groundwater and surface water in Malaysia. Sains Malays 48:2391–2403
24. Kamaruddin MA, Yusoff MS, Aziz HA, Hung Y-T (2015) Sustainable treatment of landfill leachate. Appl Water Sci 5:113–126
25. Zineb H, Latifa M, Laila S, Salah S (2019) Impact of the Mohammedia benslimane landfill on the quality of groundwater and surface water. In: Advanced intelligent systems for sustainable development (AI2SD'2018), vol 3, Advanced Intelligent Systems Applied to Environment. Springer, pp 280–288
26. Costa AM, de Alfaia RG, S.M., Campos, J.C., (2019) Landfill leachate treatment in Brazil–an overview. J Environ Manage 232:110–116
27. El-Gohary FA, Kamel G (2016) Characterization and biological treatment of pre-treated landfill leachate. Ecol Eng 94:268–274
28. Babaei S, Sabour MR, Moftakhari Anasori Movahed S (2021) Combined landfill leachate treatment methods: an overview. Environ Sci Pollut Res 1–14
29. Anouzla A, Malika K, Azoulay I, Bencheikh A, Ghizlane F, Jamal A et al (2022) Multi-response optimization of coagulation-flocculation process for stabilized landfill leachate treatment using a coagulant based on an industrial effluent. Desal Water Treat 254:71–79. https://doi.org/10.5004/dwt.2022.28388
30. Anouzla A, Souabi S, Safi M, Younes Abrouki Y, Loukili H, Rhbal H (2014) Waste to treat waste of landfill leachates. Int J Environ Protect Policy 2(2):50. https://doi.org/10.11648/J.IJEPP.20140202.12

31. Khalil F, Bouaouine O, Chtioui H, Souabi S, Aboulhassan MA, Ouammou A (2015) Traitement des lixiviats de décharge par coagulation-floculation (treatment of landfill leachate by coagulation-flocculation). J Mater Environm Sci 6:1337–1342
32. Viegas C, Nobre C, Mota A, Vilarinho C, Gouveia L, Gonçalves M (2021) A circular approach for landfill leachate treatment: Chemical precipitation with biomass ash followed by bioremediation through microalgae. J Environ Chem Eng 9:105187
33. Mahtab MS, Islam DT, Farooqi IH (2021) Optimization of the process variables for landfill leachate treatment using Fenton based advanced oxidation technique. Eng Sci Technol Int J 24:428–435
34. Reddy CV, Rao DS, Kalamdhad AS (2022) Combined treatment of high-strength fresh leachate from municipal solid waste landfill using coagulation-flocculation and fixed bed upflow anaerobic filter. J Water Process Eng 46:102554
35. Tałałaj IA (2022) Performance of integrated sequencing batch reactor (SBR) and reverse osmosis (RO) process for leachate treatment: effect of pH. J Environ Heal Sci Eng 1–11
36. Kucharska MA, Mirehbar S, Ładyńska JA (2022) Novel combined IME-$O3/OH-/H_2O_2$ process in application for mature landfill leachate treatment. J Water Process Eng 45:102441
37. Siddiqi SA, Al-Mamun A, Baawain MS, Sana A (2022) A critical review of the recently developed laboratory-scale municipal solid waste landfill leachate treatment technologies. Sustain Energy Technol Assess 52:102011
38. Luo H, Cheng Y, He D, Yang E-H (2019) Review of leaching behavior of municipal solid waste incineration (MSWI) ash. Sci Total Environ 668:90–103
39. Wijekoon P, Koliyabandara PA, Cooray AT, Lam SS, Athapattu BCL, Vithanage M (2022) Progress and prospects in mitigation of landfill leachate pollution: risk, pollution potential, treatment and challenges. J Hazard Mater 421:126627
40. Vaverková MD, Zloch J, Adamcová D, Radziemska M, Vyhnánek T, Trojan V, Winkler J, Đorđević B, Elbl J, Brtnický M (2019) Landfill leachate effects on germination and seedling growth of hemp cultivars (Cannabis Sativa L.). Waste Biomass Valoriz 10:369–376
41. Alshehrei F, Ameen F (2021) Vermicomposting: a management tool to mitigate solid waste. Saudi J Biol Sci 28:3284–3293

42. Iravanian A, Ravari SO (2020) Types of contamination in landfills and effects on the environment: a review study. In: IOP Conference Series: earth and environmental science. IOP Publishing, pp 12083
43. Ma S, Zhou C, Pan J, Yang G, Sun C, Liu Y, Chen X, Zhao Z (2022) Leachate from municipal solid waste landfills in a global perspective: characteristics, influential factors and environmental risks. J Clean Prod 333:130234
44. Naveen BP, Sumalatha J, Malik RK (2018) A study on contamination of ground and surface water bodies by leachate leakage from a landfill in Bangalore India. Int J Geo-Eng 9:1–20
45. Chu Z, Fan X, Wang W, Huang W (2019) Quantitative evaluation of heavy metals' pollution hazards and estimation of heavy metals' environmental costs in leachate during food waste composting. Waste Manag 84:119–128
46. Odom F, Gikunoo E, Arthur EK, Agyemang FO, Mensah-Darkwa K (2021) Stabilization of heavy metals in soil and leachate at Dompoase landfill site in Ghana. Environ Challen 5:100308
47. Wdowczyk A, Szymańska-Pulikowska A, Domańska M (2022) Analysis of the bacterial biocenosis of activated sludge treated with leachate from municipal landfills. Int J Environ Res Public Health 19:1801
48. Gu Z, Feng K, Li Y, Li Q (2022) Microbial characteristics of the leachate contaminated soil of an informal landfill site. Chemosphere 287:132155
49. Kurniawan TA, Singh D, Xue W, Avtar R, Othman MHD, Hwang GH, Setiadi T, Albadarin AB, Shirazian S (2021) Resource recovery toward sustainability through nutrient removal from landfill leachate. J Environ Manage 287:112265
50. Amor C, De Torres-Socías E, Peres JA, Maldonado MI, Oller I, Malato S, Lucas MS (2015) Mature landfill leachate treatment by coagulation/flocculation combined with Fenton and solar photo-Fenton processes. J Hazard Mater 286:261–268
51. Amor C, Lucas MS, García J, Dominguez JR, De Heredia JB, Peres JA (2015) Combined treatment of olive mill wastewater by Fenton's reagent and anaerobic biological process. J Environ Sci Heal Part A 50:161–168
52. Maleki A, Zazouli MA, Izanloo H, Rezaee R (2009) Composting plant leachate treatment by coagulation-flocculation process. Am J Agric Environ Sci 5:638–643
53. Samadi M, Saghi M, Rahmani A, Hasanvand J, Rahimi S, Syboney MS (2010) Hamadan landfill leachate treatment by coagulation-flocculation process. J Environ Heal Sci Eng 7:253–258

54. Assou M, El Fels L, El Asli A, Fakidi H, Souabi S, Hafidi M (2016) Landfill leachate treatment by a coagulation–flocculation process: effect of the introduction order of the reagents. Desalin Water Treat 57:21817–21826
55. Cheng SY, Show PL, Juan JC, Chang JS, Lau BF, Lai SH, Ng EP, Yian HC, Ling TC (2021) Landfill leachate wastewater treatment to facilitate resource recovery by a coagulation-flocculation process via hydrogen bond. Chemosphere 262:127829. https://doi.org/10.1016/j.chemosphere.2020.127829
56. Zakaria SNF, Abdul Aziz H, Mohamad M (2022) Comparison performance of coagulation flocculation process and combination with ozonation process of stabilized landfill leachate treatment. Water Environ Res 94:e10770
57. Campos JC of coagulation/flocculation process in the landfill leachate treatment at the M.W.T.P., Machado BS, Blonski MED, Bila DM, Ferreira JA (2013) Evaluation of coagulation/flocculation process in the landfill leachate treatment at the Municipal Wastewater Treatment Plant. Rev Ambient Água 8:43–53
58. Reynier N, Coudert L, Blais J-F, Mercier G, Besner S (2015) Treatment of contaminated soil leachate by precipitation, adsorption and ion exchange. J Environ Chem Eng 3:977–985
59. Zhang T, Ding L, Ren H (2009) Pretreatment of ammonium removal from landfill leachate by chemical precipitation. J Hazard Mater 166:911–915
60. Huang H, Xiao D, Zhang Q, Ding L (2014) Removal of ammonia from landfill leachate by struvite precipitation with the use of low-cost phosphate and magnesium sources. J Environ Manage 145:191–198
61. Chen Y, Liu C, Nie J, Luo X, Wang D (2013) Chemical precipitation and biosorption treating landfill leachate to remove ammonium-nitrogen. Clean Technol Environ Policy 15:395–399
62. Zeng Y, Chen Z, Du Y, Lyu Q, Yang Z, Liu Y, Yan Z (2021) Microbiologically induced calcite precipitation technology for mineralizing lead and cadmium in landfill leachate. J Environ Manage 296:113199
63. Kim SH, Jeong S, Chung H, Nam K (2020) Mechanism for alkaline leachate reduction through calcium carbonate precipitation on basic oxygen furnace slag by different carbonate sources: application of $NaHCO_3$ and CO_2 gas. Waste Manag 103:122–127
64. Shu J, Wu H, Chen M, Peng H, Li B, Liu R, Liu Z, Wang B, Huang T, Hu Z (2019) Fractional removal of manganese and ammonia nitrogen from electrolytic metal manganese residue leachate using carbonate and struvite precipitation. Water Res 153:229–238

65. Jagaba AH, Kutty SRM, Lawal IM, Abubakar S, Hassan I, Zubairu I, Umaru I, Abdurrasheed AS, Adam AA, Ghaleb AAS (2021) Sequencing batch reactor technology for landfill leachate treatment: a state-of-the-art review. J Environ Manage 282:111946
66. Zanona VRCM, Barquilha CER, Braga MCB (2023) Removal of recalcitrant organic matter of landfill leachate by adsorption onto biochar from sewage sludge: a quali-quantitative analysis. J Environ Manage 344:118387
67. Ghaffariraad M, Ghanbarzadeh Lak M (2021) Landfill leachate treatment through coagulation-flocculation with lime and bio-sorption by walnut-shell. Environ Manage 68:226–239
68. Halim AA, Aziz HA, Johari MAM, Ariffin KS (2010) Comparison study of ammonia and COD adsorption on zeolite, activated carbon and composite materials in landfill leachate treatment. Desalination 262:31–35
69. Daud Z, Abubakar MH, Kadir AA, Latiff AAA, Awang H, Halim AA, Marto A (2017) Adsorption studies of leachate on cockle shells. Geomate J 12:46–52
70. Mohammad-Pajooh E, Turcios AE, Cuff G, Weichgrebe D, Rosenwinkel K-H, Vedenyapina MD, Sharifullina LR (2018) Removal of inert COD and trace metals from stabilized landfill leachate by granular activated carbon (GAC) adsorption. J Environ Manage 228:189–196
71. Pedro-Cedillo S, Méndez-Novelo RI, Rojas-Valencia MN, Barceló-Quintal M, Castillo-Borges ER, Sauri-Riancho MR, Marrufo-Gómez JM (2015) Evaluation of adsorption and Fenton-adsorption processes for landfill leachate treatment. Rev Mex Ing química 14:745–755
72. de Melo Franco Domingos J, de Alencar Neves T, de Sousa Maia DL, Carvalho Siqueira R, Araújo Marques MV, Alves OL, Guimarães JR, Antunes Nolasco M, Rosa AH (2023) Effect of the association of coagulation/flocculation, hydrodynamic cavitation, ozonation and activated carbon in landfill leachate treatment system. Sci Rep 13:9502
73. Nabavi E, Shamskilani M, Dezvareh GA, Darban AK (2023) ANN-Based modeling of combined O3/H2O2 oxidation, and activated carbon adsorption treatment system: forest polluting site leachate. Water Air Soil Pollut 234:86
74. Kurniawan TA, Lo W, Chan GYS (2006) Radicals-catalyzed oxidation reactions for degradation of recalcitrant compounds from landfill leachate. Chem Eng J 125:35–57
75. Amr SSA, Aziz HA, Adlan MN (2013) Optimization of stabilized leachate treatment using ozone/persulfate in the advanced oxidation process. Waste Manag 33:1434–1441

76. Amaral-Silva N, Martins RC, Castro-Silva S, Quinta-Ferreira RM (2016) Ozonation and perozonation on the biodegradability improvement of a landfill leachate. J Environ Chem Eng 4:527–533
77. Wei S, Yuru W (2017) Treatment of landfill leachate concentrate by low temperature vacuum evaporation. Chinese J Environ Eng 11:5417–5423
78. Yuan Q, Huang Y, Qi E, Liu C, Wang X, Wu Y (2023) Experimental study on treatment of leachate concentrate from disk-tube reverse osmosis treatment in a waste incineration plant by gas-liquid contact evaporation
79. Alcântara PB, de Castilhos Junior AB (2014) Treatment of leachates by evaporation in the semiarid region of the Brazilian northeast. J. Solid Waste Technol Manag 40:44–56
80. Luo K, Pang Y, Li X, Chen F, Liao X, Lei M, Song Y (2019) Landfill leachate treatment by coagulation/flocculation combined with microelectrolysis-Fenton processes. Environ Technol 40:1862–1870
81. de Almeida R, Moraes Costa A, de Almeida Oroski F, Carbonelli Campos J (2019) Evaluation of coagulation–flocculation and nanofiltration processes in landfill leachate treatment. J Environ Sci Heal Part A 54:1091–1098
82. Bellouk H, El Mrabet I, Tanji K, Nawdali M, Benzina M, Eloussaief M, Zaitan H (2022) Performance of coagulation-flocculation followed by ultra-violet/ultrasound activated persulfate/hydrogen peroxide for landfill leachate treatment. Sci Afric 17:e01312
83. Kwarciak-Kozłowska A, Fijałkowski KL (2021) Efficiency assessment of municipal landfill leachate treatment during advanced oxidation process (AOP) with biochar adsorption (BC). J Environ Manage 287:112309
84. Mojiri A, Ziyang L, Hui W, Ahmad Z, Tajuddin RM, Amr SSA, Kindaichi T, Aziz HA, Farraji H (2017) Concentrated landfill leachate treatment with a combined system including electro-ozonation and composite adsorbent augmented sequencing batch reactor process. Process Saf Environ Prot 111:253–262
85. Scandelai APJ, Cardozo Filho L, Martins DCC, de Souza Freitas TKF, Garcia JC, Tavares CRG (2018) Combined processes of ozonation and supercritical water oxidation for landfill leachate degradation. Waste Manag 77:466–476
86. Afsharnia M, Torabian A, Mousavi GR, Abduli MA (2012) Landfill Leachate treatment by sono-evaporation. Desalin Water Treat 48:344–348
87. Zielińska M, Kulikowska D, Stańczak M (2020) Adsorption-Membrane process for treatment of stabilized municipal landfill leachate. Waste Manag 114:174–182

© The Author(s), under exclusive license to Springer Nature Switzerland AG 2024
S. Souabi, A. Anouzla (eds.), *Landfill Leachate Treatment Techniques*, Springer Water
https://doi.org/10.1007/978-3-031-63157-3_8

8. Membranes in Landfill Leachate Treatment: Evaluation of the Generated Concentrate

Ronei de Almeida[1] ✉ and Juacyara Carbonelli Campos[2] ✉
(1) Department of Sanitary and Environmental Engineering, Rio de Janeiro State University, Rio de Janeiro, Brazil
(2) School of Chemistry, Universidade Federal Do Rio de Janeiro, Rio de Janeiro/RJ, Brazil

✉ **Ronei de Almeida**
Email: ronei.almeida@eng.uerj.br

✉ **Juacyara Carbonelli Campos (Corresponding author)**
Email: juacyara@eq.ufrj.br

Abstract
The current chapter explores different aspects of the reverse osmosis (RO) process for landfill leachate management. As the foremost or polishing step in a leachate treatment chain, the RO must meet the most restrictive water discharge or reuse standards. Indeed, high-pressure-driven membranes (e.g., RO and nanofiltration) have been used in several landfill sites to treat wastewater efficiently. However, concentrate management is critical for closing the leachate treatment cycle. The management of membrane concentrate streams and related issues are analyzed in this context, following our early experimental case study. Experimental landfill cells were constructed to investigate the impacts of

concentrate recirculation practice on the leachate quality and treatment performance. The findings from an eight-month monitoring experiment are briefly presented. Finally, prospects for the disposal of concentrate streams from the RO system are discussed.

Keywords Concentrate streams – Experimental landfill cells – Landfill leachate – Municipal solid waste – Reverse osmosis – Waste-to-resources

---

8.1 Introduction

The landfill behaves like a dynamic reactor, whose chemical and biological reactions produce biogas, liquid effluent (leachate) and mineralized waste (humus) from the decomposition of organic matter. Landfill leachate is one of the main environmental risk factors due to its high concentrations of organic matter and ammonia nitrogen, among other pollutants.

As a result, it is not easy to standardize a treatment system that generates effluents with standards compatible with current legislation. Furthermore, due to the complexity of the composition of landfill leachate and its extreme variability, its treatment often requires a combination of processes [1, 2].

Conventional treatments consist mainly of the biological degradation of leachate, including stabilization ponds, activated sludge and combined treatment with sanitary sewage in Sewage Treatment Plants (STPs) [3, 4]. Biological treatments are effective for leachate from young landfills, whose organic fraction mainly comprises volatile fatty acids. Other techniques that can be used to treat leachate are the ammonia stripping process, to remove ammonia nitrogen, chemical precipitation, which consists of adding compounds that react with each other to form a precipitate; and coagulation/flocculation, which causes the destabilization of colloidal particles through the action of a coagulating agent.

Treatment techniques using membranes have been widely used due to their operational flexibility because the system is more compact and produces a better quality treated effluent [5]. Their growing application is also justified by their ability to remove contaminants that conventional processes cannot, such as recalcitrant organic matter. Among the main Membrane Separation Processes (MSP) applicable to the treatment of landfill leachate are Reverse Osmosis (RO) and Nanofiltration (NF) due to their high capacity to retain organic matter and inorganic contaminants [6, 7].

The RO process produces a high-quality permeate and a membrane concentrate stream. High concentrations of dissolved organic matter characterize the leachate concentrate but have a low ratio between biochemical oxygen demand (BOD) and chemical oxygen demand (COD), indicating poor biodegradability. In addition, the high salinity of the leachate concentrate harms the effectiveness of biological degradation [8].

Concerning the concentrate from the membrane separation process, this stream is generally recirculated to the landfill cells at no additional cost to the landfill managers. However, the consequences of this practice are not fully understood, and the information available in the literature presents divergent conclusions.

This chapter aims to shed light on the main advantages and disadvantages of using the reverse osmosis process in leachate treatment and discusses solutions for the disposal of concentrate streams from the RO system.

8.2 Landfill Leachate Treatment: A Brief Overview

Leachate from municipal solid waste landfills, generated from the decomposition of solid waste, is a complex mixture of organic and inorganic substances, the composition of

which causes damage to the environment due to the toxicity present.

The leachate generated needs to be drained, collected and sent for treatment since, in addition to the risk of surface water contamination, there is also the risk of groundwater contamination and soil and vegetation degradation. Although landfills are lined, once this effluent has been drained, it cannot be discharged directly into a receiving body due to its risk. Given the complexity of its composition, there is a consensus that there is no simple, universal solution for treating landfill leachate.

Depending on the age of the landfill and the degree of degradation of the waste it contains, the characteristics of the leachate can vary [6]:

- Leachate from a new landfill presents biodegradable organic compounds easily removed by the microorganisms involved in biological treatment, a low concentration of ammonia nitrogen, which is toxic to active biomass in high concentrations, and a high concentration of low molecular mass volatile fatty acids. In this case, biological treatment is appropriate;
- Leachate from older landfills, a significant decrease in its biodegradable organic fraction is expected. In these cases, the expected characteristics are high ammonia nitrogen concentrations and a recalcitrant organic matter content. Aerobic and anaerobic biological treatment can be limited in removing organic matter. This problem can be reversed by combining biological and physical–chemical treatments.

In general, treatment processes need to be combined, taking into account cost and legal requirements for disposal in water bodies. Figure 8.1 shows a scheme for the choice of treatment for landfill leachates.

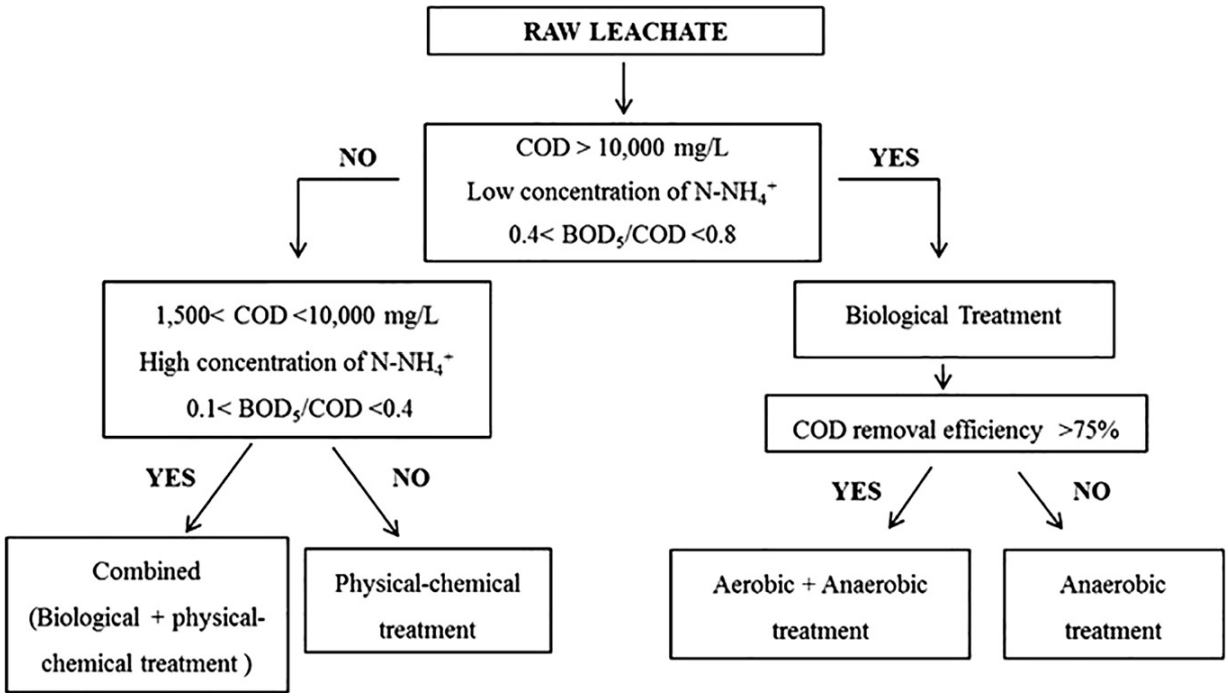


Fig. 8.1 Parameters considered in selecting the appropriate treatment for leachate.

Source Costa et al. [6] with permission of Elsevier (license # 5,615,681,394,123)

A brief comparison of landfill leachate treatment techniques is shown in Table 8.1.

Table 8.1 Comparison of landfill leachate treatment techniques

Method	Principles	Advantages	Disadvantages
Recirculation	Return of leachate collected from the landfill to landfills. The landfill system serves as a biofilter	The stabilization of landfilled waste is accelerated, the waste degradation is increased, and the landfill leachate volume is reduced	Dissolved organic matter accumulates in a stream that is more difficult to treat
Co-treatment	Landfill leachate is treated together with sewage	Cheaper	It makes the disinfection process ineffective

Method	Principles	Advantages	Disadvantages
Biological treatments	Degradation of organic pollutants in leachate by aerobic or anaerobic reactions	Simplicity; reliability; highly cost-effective	Ineffective with non-biodegradable organic matter
Chemical precipitation	Chemical reagents react with soluble substances to form precipitates	Strong capacity, simplicity, low operating costs	Low COD removal efficiency; pH-dependent; high precipitant dosage; excessive sludge generation
Advanced oxidative processes	Free radicals destroy dissolved organic matter and degrade it into small molecules or mineralize it directly into CO_2 and H_2O	Strong oxidation capacity; high efficiency; mild operating conditions; ease of operation and low maintenance; fast reaction speed	High dosage of chemical reagents; high energy consumption; non-renewable
Coagulation/flocculation	Destabilization of colloidal or by reducing repulsion forces. Collusion between particles and consequent flocculation	Low cost; high efficiency	Generation of excessive sludge; secondary pollution; pH-dependent
Adsorption	Pollutants are transferred from the liquid phase to the surface of a solid through physical and/or chemical interactions	High efficiency	Regeneration of adsorbents; high consumption of adsorbents
Air stripping	Air or water vapor passes through the	Simple process with strong	NH_3 leakage and low COD removal

Method	Principles	Advantages	Disadvantages
	leachate, transferring NH_3–N from the liquid to the gas phase	adaptability, low investment costs	efficiency; calcium carbonate fouling in the stripping tower
Membrane filtration	Size exclusion and electrostatic interactions between membranes and molecules	High efficiency; low cost; simple equipment requirements; mild conditions	Membrane fouling and production of a concentrated stream

Source Adapted from Teng et al. [9]

The conventional processes for landfill wastewater treatment are biological and physicochemical systems or their combination in integrated or sequential schemes. Traditional biological technologies alone cannot significantly treat methanogenic leachates, which contain contaminants resistant to biodegradation. Additionally, high ammonia concentrations cannot be removed successfully by biological treatment, such as activated sludge, aerated lagoons, sequence batch reactors, or trickling filters [10]. Therefore, the option of advanced technologies should be considered.

Among the membrane separation processes (MSP), reverse osmosis have been considered the most suitable approach for treating leachates to meet legislation quality criteria. On the other hand, the concentrate stream is an issue. Therefore, an effective and feasible method should be developed to treat membrane concentrates because they pose high environmental risk.

8.3 RO Process

8.3.1 Fundamental Concepts

Reverse osmosis (RO) is a high-pressure-driven membrane filtration technology. This process generates two output streams during its operation: permeate and concentrate. The

permeate is the purified effluent, and the concentrate, also known as retentate or brine (for water desalination), is the residual stream (Fig. 8.2). Flowrate and mass balances are considered during filtration (Eqs. 8.1 and 8.2) [11].

$$Q_f = Q_p + Q_c \tag{8.1}$$

$$Q_f \times C_{f,i} = Q_p \times C_{p,i} + Q_c \times C_{c,i} \tag{8.2}$$

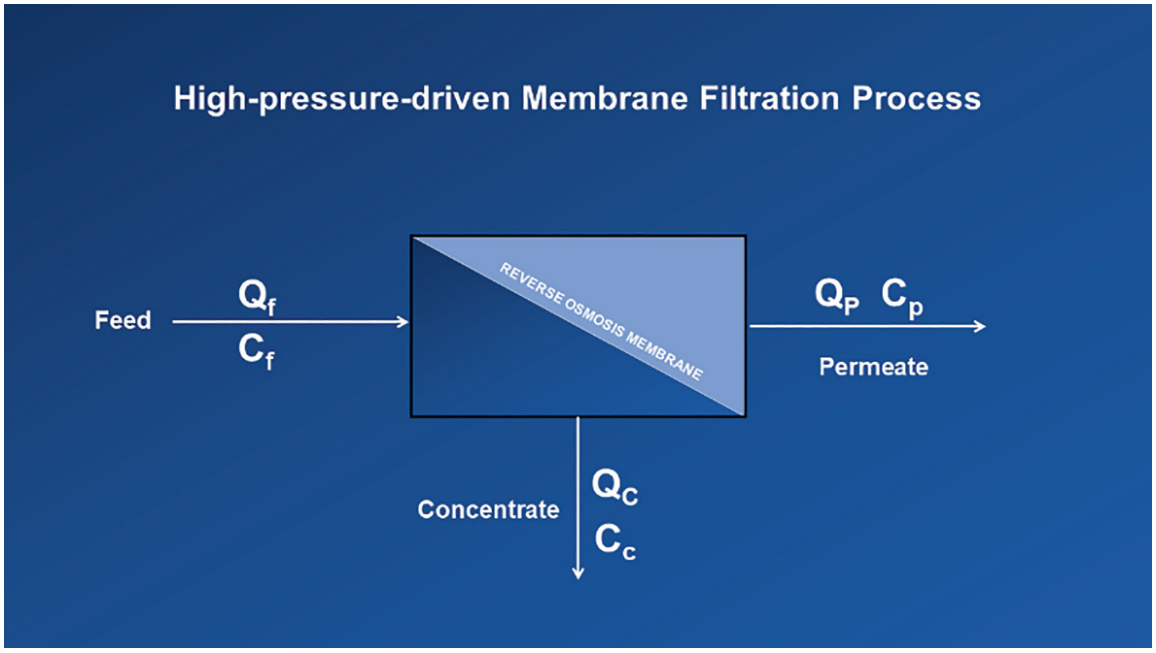


Fig. 8.2 Schematic representation of the membrane filtration process

where: Q_f, Q_p, and Q_c (L h^{-1}) are feed, permeate, and concentrate flowrates, respectively. The $C_{f,i}$, $C_{p,i}$, and $C_{c,i}$ (mg L^{-1}) are feed, permeate, and concentrate concentrations of the solute "i".

Two essential parameters determine the performance of membrane-based processes: solvent permeability or flux through the membrane, rejection of solutes, and yield or recovery of the membrane-based unit. The permeate flux (J [L m^{-2} h^{-1}, LMH]) is defined as the permeate flow rate per membrane's permeation area (Eq. 8.3). In contrast, the

membrane permeance (Lp [L m^{-2} h^{-1} bar^{-1}]) is the permeate flux per unit of applied pressure (Eq. 8.4) [11].

$$J = Qp/Am \tag{8.3}$$

$$Lp = J/TMP \tag{8.4}$$

where: A_m represents the membrane's permeation area (m^2), and TMP is the transmembrane pressure (bar).

The rejection of solute “i” is defined by Eq. 8.5.

$$\text{Rejection}\,(\%) = (1 - C_{p,i}/C_{f,i}) \times 100\% \tag{8.5}$$

Rejection is a dimensionless parameter, and its value varies between 100% (complete rejection of the solute) and 0% (solute and solvent pass freely through the membrane) [12].

The RO recovery (R [%]) is the ratio of the permeate stream to the feed stream flow rate (Eq. 8.6). It should be noted that the recovery parameter is linked to the design of an industrial application rather than a membrane characteristic [12].

$$R\,(\%) = \left(Q_p/Q_f\right) \times 100\% \tag{8.6}$$

The RO process uses a physical mechanism in which the operating pressure must be kept higher than the solution osmotic pressure ($\Delta P > \Delta\pi$) (Fig. 8.3). The transport mechanism across the RO membrane follows the dissolution/diffusion model, where both solvent and solute dissolve in the dense surface layer of the membrane and diffuse separately due to the chemical potential gradient of each species. In wastewater treatment systems, RO membranes are primarily used to remove low molecular mass solutes such as salts and heavy metals [11, 13].

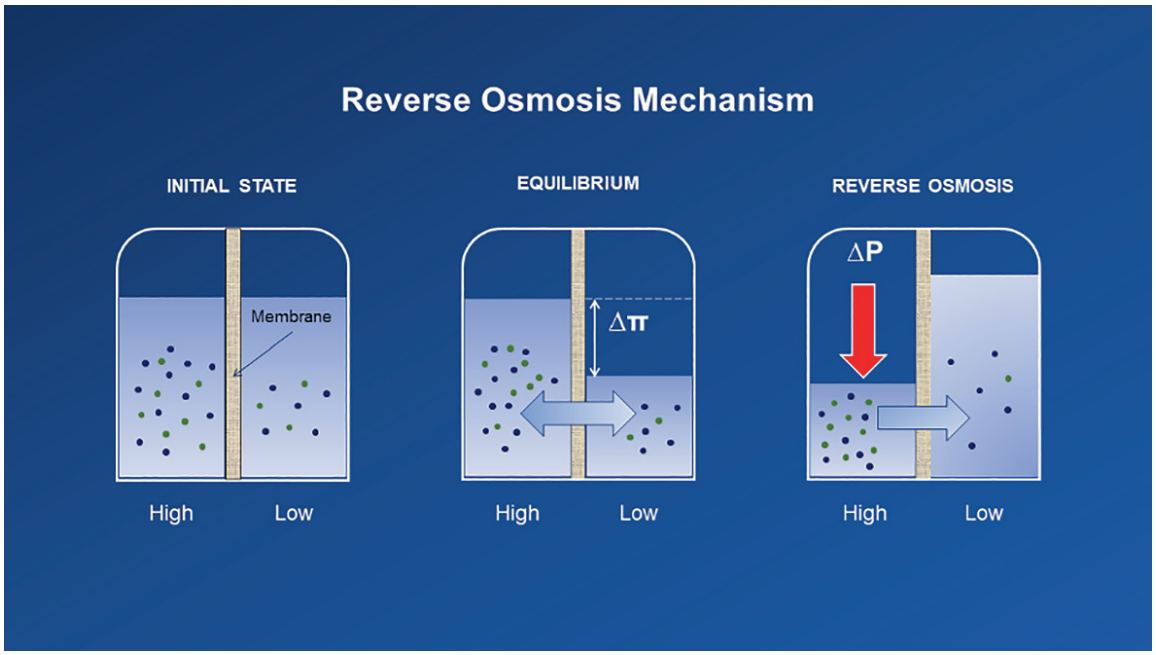


Fig. 8.3 Schematic representation of reverse osmosis mechanism principles

The standard operational mode of the full-scale RO system is cross-flow with high internal flow rate concentrate recirculation. Typical membrane modules include tubular, spiral wound, hollow fiber, and disc tubes [14]. The higher cost–benefit of the spiral wound module makes it the most used configuration. Spiral wound modules are available in standardized design, in standard diameters (2.5, 4, and 8 inches) to fit pressure vessels that can hold several modules in series connected by O-rings. The module uses flat sheet membranes wrapped around a central tube. The membranes are glued along three edges over a permeate spacer to form “leafs”. The permeate spacer supports the membrane and conducts the permeate to the central permeate tube [15].

RO modules are available in containers from different suppliers. Thanks to their modular configuration, they can be adapted to suit specific conditions. RO spiral wound systems are commonly used in a stand-alone mode, and depending on the wastewater requirements, multiple steps can be combined. In this approach, the affluent undergoes filtration in two or more steps before reaching its final discharge point [16].

Figure 8.4 shows the leachate treatment facility at Seropédica Landfill in Rio de Janeiro, Brazil. This plant handles over 22,400 pounds of waste daily and generates approximately 2000 m^3 per day of landfill wastewater (estimated in 2022).

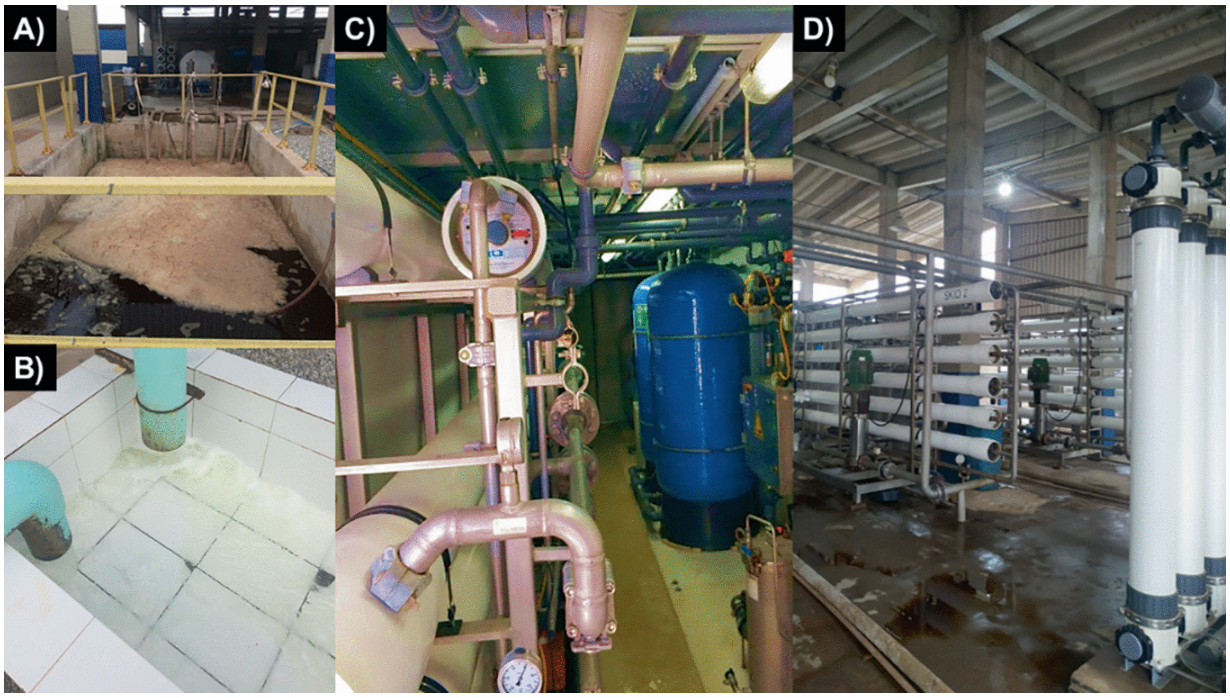


Fig. 8.4 High-pressure-driven membrane process in Seropédica landfill, Rio de Janeiro—Brazil. **a** Raw landfill leachate. **b** Treated landfill leachate after the membrane filtration process. **c** RO process installed in a container. **d** High-pressure-driven membrane modules.

Source Authors' collection

8.3.2 RO Performance in Landfill Wastewater Purification

The reverse osmosis (RO) process is indispensable for leachate treatment due to its cost-effectiveness, i.e., producing high-quality water at competitive costs [17]. This system has been used either as a single treatment step or in combined trains [18, 19].

Several studies have documented RO in full-scale landfill leachate treatment plants (LLTP) (Table 8.2), which demonstrates the membrane technology maturity. In Brazil, a

250 m^3 d^{-1} LLTP has performed a three-stage RO system at a permeate flux of 10 LMH and recovery of 70% [20, 21]. Overall, treatability results show removal efficiencies of BOD_5, COD, and NH_3–N greater than 98% [21]. In recent studies, RO systems have also been pointed as a highly efficient process for removing contaminants of emerging concerns like per- and polyfluoroalkyl substances (PFAS) and antibiotic resistance genes [22, 23]. Chen et al. [22] performed an investigation and showed that RO systems remove more than 98% of several types of PFAS in the raw leachate.

Table 8.2 RO process in full-size mode applications for landfill wastewater treatment

Treatment scheme	**Location**	**Module type**	**Operating conditions**		**Performance (% removal)**		**References**
			Permeate flux (LMH)	**R (%)**	**COD**	**NH_3-N**	
Coagulation + sand filtration + MF + **RO**	Thailand	Spiral wound	NA	50	87.5	> 88	[24]
Coagulation + sand filtration + **RO**	Thailand	Spiral wound	NA	50	>98	NA	[25]
Two-stage RO	Czech Republic	Spiral wound	42	94	97.3	94.0	[26]
Sand and cartridge filtration + **two-stage RO**	Romania	Disc-tube	12–22	NA	94–96	>94	[27]
Buffer tank + **RO**	Poland	Disc-tube	Up to 50	75	98.8	99.1	[28]
UF + three-stage **RO**	Italy	Disc-tube	13–32.5	91–95	92.4–99.2	46.2–95.8	[29]

Treatment scheme	Location	Module type	Operating conditions		Performance (% removal)		References
			Permeate flux (LMH)	R (%)	COD	NH_3-N	
Two-stage RO + Air stripping	China	Disc-tube	N.A	>83%	>99	>99	[30]
Coagulation + aerobic biological tank + MBR + **RO**	Iran	Spiral wound	NA	NA	>96	NA	[31]
MBR + **RO**	Spain	Spiral wound	32–64	62 ± 9	>99	>97	[32]
Coagulation + MF + **Two-stage RO**	USA	NA	NA	85	NA	NA	[22]
Three-stage RO	Brazil	Spiral wound	10	70	>98	>98	[21]

COD chemical oxygen demand. *MBR* membrane bioreactor. *MF* microfiltration. *NA* Not available. *NH_3–N* ammonia nitrogen. *RO* reverse osmosis. *UF* ultrafiltration. *USA* United States of America

8.4 Challenges and Prospects

8.4.1 Membrane Fouling and Concentrate Management

Two significant issues are identified as shortcomings of implementing RO systems: (i) membrane fouling, which decreases the permeate flux and water quality, and (ii) RO concentrate management. Membrane fouling can be represented by a resistances-in-series model (Eq. 8.7). Resistances in series represent the theoretical resistance to permeate flux due to a build-up of a layer of dirt or other fouling substances onto the membrane surface [33]. In

addition, a gel layer arises from the precipitation of some salts that reach the solubility product at the membrane-liquid interface [34].

$$J = TMP/\mu \times R_t \quad (8.7)$$

where: TMP is the transmembrane pressure (kg m^{-1} s^{-2}), μ is the dynamic viscosity (kg m s^{-1}), and R_t is the total fouling resistance (m^{-1}).

Membrane fouling is an inherent phenomenon in RO operation. However, it can be mitigated through various approaches, including pretreatment of the fed leachate, cleaning protocols, optimization of operational conditions, and modification of the membrane properties [10]. As discussed in the previous section, RO modules are often combined in two or three stages to achieve up to 90% water recovery rates. Nevertheless, these systems typically require extensive feed pretreatment to minimize fouling issues and significant anti-scaling and cleaning procedures. Unfortunately, fouling remains a bottleneck problem in membrane promotion and application, as it can lead to increased operational costs [35].

Finally, a challenge that must be dealt with is managing membrane concentrate streams. The development of technologies and process breakthroughs in the water desalination field have been helping to tackle the concentrate management issue in other areas. There is a pool of patented technologies to improve the overall feed water recovery, aiming to reduce concentrate volume and achieve minimal liquid discharge (MLD) or zero liquid discharge (ZLD) [36, 37]. However, their high costs linked mainly to energy requirements limit the implementation of a full-size ZLD mode in landfills. Furthermore, it is essential to note that MLD/ZLD systems are associated with unintended environmental impacts due to their high energy demand and carbon footprint [38]. Our early study comprehensively

reviewed membrane concentrate management from leachate treatment plants [39, 40].

8.4.2 Membrane Concentrate Management—a Case Study Using Experimental Landfill Cells

Several methods have been employed to manage leachate membrane concentrate, including recirculation in landfills. Figure 8.5 illustrates the membrane concentrate pounds and reinjection operation from two Brazilian landfills where concentrate recirculation practices are implemented. Treatment strategies can be categorized into advanced techniques (e.g., thermal and advanced oxidation processes) and engineered combinations of physico-chemical methods in hybridized trains [41–43].

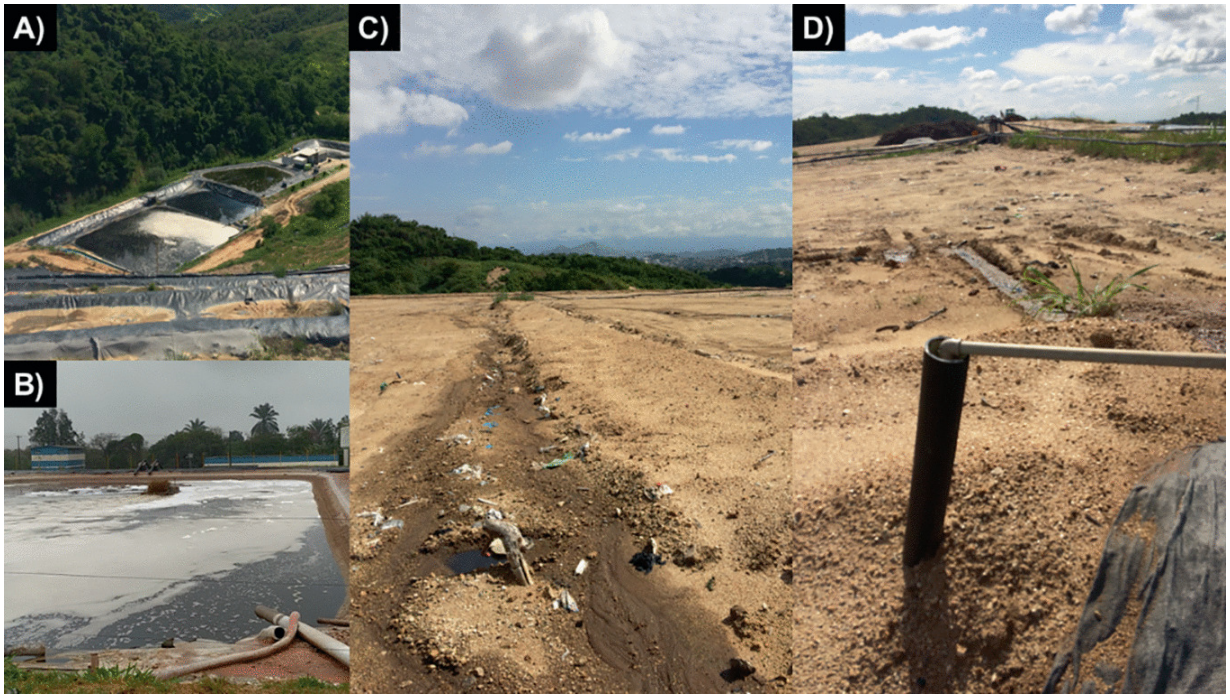


Fig. 8.5 Leachate concentrate recirculation in Brazilian landfills. **a** Membrane concentrate pound in the Brazilian landfill A. **b** Membrane concentrate pound in the Brazilian Landfill B. **c** and **d** Operation of the membrane concentrate infiltration in the Brazilian Landfill B.

Source Authors' collection

The characteristics of leachate concentrate, such as salinity, biodegradability, and ecotoxicity, play a crucial role

in determining the most suitable approach for handling concentrate streams. For more in-depth information, readers are referred to the works of [44, 45].

As mentioned before, the leachate concentrate reinjection into landfill waste mass is the conventional approach, similar to leachate recycling, to adjust moisture content and degrade organics in landfills. However, the impacts of this management tactic should be more discussed, especially in emerging nations. In this context, we conducted a study to assess the reverse osmosis concentrate (ROC) infiltration effects on leachate quality and treatment performance, monitoring experimental landfill cells over eight months. Our research focused on changes in leachate quality, membrane fouling, and treated leachate ecotoxicity.

The median values of pollution parameters like chemical oxygen demand (COD) and absorbance at 254 nm (ABS254) were significantly higher in leachates from cells with ROC recirculation compared to those without (p-value < 0.05) [46]. This result indicated that ROC infiltration increased pollution levels due to increased organic content (COD and ABS254) and recalcitrance in the leachate (higher BOD_5/COD ratio). Furthermore, the membrane fouling resistance was sixfold higher in the RO system treating methanogenic leachates from cells that operated with ROC recirculation. Therefore, organic fouling was suggested to play a significant role in membrane permeability decline. In full-size operation mode, membrane fouling increases treatment expenses by demanding intensive cleaning and replacement of membrane modules. Additionally, the ecotoxicological evaluation using *Vibrio fischeri* bacteria showed that the concentrate reinjection increased leachate ecotoxicity and decreased its removal efficiency during the RO treatment [46].

In summary, higher organic content in the leachate and increased membrane fouling and ecotoxicity indicate that ROC recirculation worsens environmental concerns and lowers RO performance. These findings emphasize the need

for careful management of ROC in landfill operations to minimize its adverse impact on wastewater quality and ensure successful leachate purification. For more detailed information, readers are referred to [46].

Nowadays, incineration and solidification/stabilization (S/S) have been used to handle concentrated leachate, and some projects in full mode are ongoing in China [47]. However, the high operating costs might hinder their implementation in small and mid-sized landfill sites. Meier et al. [48] documented that treatment costs of incineration and S/S range from 51 to 151 and 41 to 67 EUR per metric cubic meter of treated concentrate, respectively. Therefore, developing cost-effective options to manage leachate membrane concentrate is necessary.

8.4.3 Prospects: Waste-to-Resource Approach

Landfill leachate that has undergone reverse osmosis (RO) treatment permits its consideration for reuse applications. However, it is essential to note that RO does not destroy pollutants but produces a membrane concentrate that requires proper management. Depending on the RO recovery rate, the concentrate stream can account for up to 50% of the incoming landfill leachate (Table 8.2).

Researchers have shifted their focus towards a waste-to-resource approach to tackle the issue of membrane concentrate management. Table 8.3 presents an overview of the most favorable characteristics and disadvantages of concentrate recovery strategies investigated in the contemporary literature.

Table 8.3 Advantages and disadvantages of recovery technologies

Resource/applications	Recovery technology	Advantages	Disadvantages	Reference
Humic-like substances (soil conditioner)	Loose NF + MD	High retention of humic-like substances	It requires further product purification	[49]

Resource/applications	Recovery technology	Advantages	Disadvantages	Reference
		Low rejection of inorganic salts	Product toxicity should be tested	
Nitrogen (fertilizer)	Air stripping—absorption	Commercially available High ammonia removal and ammonia phosphate recovery	High recovery costs (estimated at > 50 US$ m^{-3}) Public acceptance	[50]
Energy	Pyrolysis (Py)	The quantity and quality of Py-oil and gas products are improved Py-gas with a heating value of 12.29 MJ kg^{-1} for co-pyrolysis—leachate concentrate: sanitary sludge (30:70 wt%)	It may need an external energy source to keep the Py-process Product upgrading may be required	[51]
Biochar (soil conditioner, C-sink, and energy storage devices)	Slow-Py of concentrate residue and spent coffee ground (1:1 wt%)	Biochar with 23.25 wt% water content and 22.05 MJ kg^{-1} heating value	An external energy source to keep the Py-process may be needed Full-size mode investigation is needed Product toxicity should be tested	[40]

NF nanofiltration. *MD* membrane distillation

Energy recovery and material extraction for soil conditioning show the most promising results among the

tested methods. Liu et al. [49] integrated loose nanofiltration (NF) and membrane distillation (MD) for membrane concentrate valorization. The integrated loose NF + MD process achieved a remarkable 95.5% organics recovery and produced 82% high-quality water. In another study, [50] successfully recovered ammonia phosphate from leachate concentrate streams. Organics and ammonia components obtained from these processes have potential applications as agricultural fertilizers. Recent literature explores using RO concentrate in pyrolytic operations to recover energy as syngas and bio-oil while producing biochar. The biochar can serve as a C-sink and be utilized as a soil amendment [40, 51].

Combining these methods shows excellent promise for sustainable concentrate treatment and resource recovery. The products' recovery could generate revenues and lessen leachate treatment costs. However, further research is needed to implement continuous monitoring and performance evaluation of full-scale systems to understand their long-term effectiveness. Additionally, it is crucial to address the potential environmental implications of leachate concentrate management. Studies should focus on assessing the ecological impacts of concentrate treatment schemes and identifying best practices for concentrate safe disposal or reuse [39].

Moreover, the waste-to-resource approach faces significant commercial, environmental, and regulatory challenges during practical implementation. The strategies for recovering resources from concentrate streams need more specific legislation encouraging the landfill industry to explore viable, cost-effective solutions. Currently, most studies on this topic are conducted at the lab-scale stage and are not continuous processes [45]. Therefore, to unlock the full potential of waste-to-resources in the landfill industry and advance sustainable leachate management practices, it will be essential to focus on scientific exploration, technological innovation, and the implementation of supportive policies.

8.5 Concluding Remarks

This chapter provided an overlook of the technological aspects of the RO system for landfill leachate purification. The following conclusions are drawn:

- Landfill leachate is a complex and polluting wastewater, necessitating the application of advanced technologies for its treatment. Reverse Osmosis (RO) is employed to achieve contaminant removal from leachate beyond what conventional processes like biological and oxidation treatments can achieve.
- Generally, RO demonstrates removal efficiencies for BOD_5, COD, and NH_3–N, exceeding 98%. This technology has also garnered attention for its ability to address emerging contaminants.
- Managing RO leachate concentrate, the residual stream from RO systems, poses a critical challenge for landfill managers. Its recirculation into the waste mass has adversely affected leachate quality and treatment performance. Solidification/stabilization and thermal treatment have been used in full-scale projects. Conversely, these technologies might lack cost-effectiveness, hindering their implementation, especially in small and medium-sized landfills. Hence, future research and engineering practices are necessary to achieve environmental benefits and economic feasibility in the landfill industry.
- The waste-to-resource approach is promising for future studies. Scientific exploration, technological innovation, and supportive policies will be crucial to advancing sustainable leachate management practices and realizing the full potential of resource recovery in the landfill operation.

References

1. Hong M, Lu G, Hou C, She S, Zhu L (2017) Advanced treatment of landfill leachate membrane concentrates: performance comparison, biosafety and toxic residue analysis. Water Sci Technol 76(11):2949–2958. https://doi.org/10.2166/wst.2017.469
2. Kallel A, Ellouze M, Trabelsi I (2017) Co-management of landfill leachate concentrate with brick waste by solidification/stabilization treatment. Arab J Geosci 10(4):1–12. https://doi.org/10.1007/s12517-017-2871-x
3. Yang BQ, Yang JM, Yang H, Liu YM, Li XK, Wang QZ, Pan XJ (2018) Co-bioevaporation treatment of concentrated landfill leachate with addition of food waste. Biochem Eng J 130:76–82. https://doi.org/10.1016/j.bej.2017.11.010
4. Zhang X, Fan L, Roddick FA (2013) Influence of the characteristics of soluble algal organic matter released from Microcystis aeruginosa on the fouling of a ceramic microfiltration membrane. J Membr Sci 425:23–29. https://doi.org/10.1016/j.memsci.2012.09.033
5. Marian T, Nghiem LD (2010) Landfill leachate tretment using hybrid coagulation-nanofiltration processes. Desalination 250(2):677–681. https://doi.org/10.1016/j.desal.2009.03.024
6. Costa AM, Alfaia RGDSM, Campos JC (2019) Landfill leachate treatment in Brazil–an overview. J Environ Manage 232:110–116. https://doi.org/10.1016/j.jenvman.2018.11.006
7. Gao M, Liang F, Yu A, Li B, Yang L (2010) Evaluation of stability and maturity during forced-aeration composting of chicken manure and sawdust at different C/N ratios. Chemosphere 78(5):614–619. https://doi.org/10.1016/j.chemosphere.2009.10.056
8. Ye W, Liu H, Jiang M, Lin J, Ye K, Fang S, Xu Y, Zhao S, Van der Bruggen B, He Z (2019) Sustainable management of landfill leachate concentrate through recovering humic substance as liquid fertilizer by loose nanofiltration. Water Res 157:555–563. https://doi.org/10.1016/j.watres.2019.02.060
9. Teng C, Zhou K, Peng C, Chen W (2021) Characterization and treatment of landfill leachate: a review. Water Res 203:117525. https://doi.org/10.1016/j.watres.2021.117525
10. Gripa E, Daflon SDA, de Almeida R, Fonseca FV, Campos JC (2023) Landfill leachate treatment by high-presssure membranes and advanced oxidation techniques with a focus on ecotoxicity and by-products management: a review. Process Saf Environ Prot. https://doi.org/10.1016/j.psep.2023.03.074
11. Baker RW (2012) Membrane technology and applications. In: Membrane technology and applications, vol 3rd ed. Wiley
12. Uragami T (2017) Nanofiltration principle of nanofiltration fundamental analysis of nanofiltration, pp 297–323

13. Wilf M (2014) The reverse osmosis process. In: Jane Kucera (ed) Desalination, vol 9781118208. John Wiley and Sons, Inc., pp 155–204. https://doi.org/10.1002/9781118904855.ch3
14. Yang J, Monnot M, Ercolei L, Moulin P (2020) Membrane-based processes used in municipal wastewater treatment for water reuse: state-of-the-art and performance analysis. Membranes 10(6):131. https://doi.org/10.3390/membranes10060131
15. Maynard E, Whapham C (2020) Quality and supply of water used in hospitals. In: Decontamination in hospitals and healthcare (Second Edi, pp. 45–69). Elsevier. https://doi.org/10.1016/B978-0-08-102565-9.00003-0
16. de Almeida R, de Souza Couto JM, Gouvea RM, de Almeida Oroski F, Bila DM, Quintaes BR, Campos JC (2020) Nanofiltration applied to the landfill leachate treatment and preliminary cost estimation. Waste Manag Res J Sustain Circ Econ 38(10):1119–1128. https://doi.org/10.1177/0734242X20933333
17. Chen W, Gu Z, Ran G, Li Q (2021) Application of membrane separation technology in the treatment of leachate in China: a review. Waste Manage 121:127–140. https://doi.org/10.1016/j.wasman.2020.12.002
18. Bouchareb R, Isik Z, Ozay Y, Karagunduz A, Keskinler B, Dizge N (2022) A hybrid process for leachate wastewater treatment: evaporation and reverse osmosis/sequencing batch reactor. Water Environ Res 94(4):1–10. https://doi.org/10.1002/wer.10717
19. Elfilali N, Elazhar F, Dhiba D, Elmidaoui A, Taky M (2022) Performances of various hybrids systems coagulation—ultrafiltration / nanofiltration-reverse osmosis in the treatment of stabilized landfill leachate. 257:55–63. https://doi.org/10.5004/dwt.2022.28360
20. de Almeida R, Bila DM, Quintaes BR, Campos JC (2020) Cost estimation of landfill leachate treatment by reverse osmosis in a Brazilian landfill. Waste Manag Res J Int Solid Wastes Public Cleans Assoc ISWA 38(10):1087–1092. https://doi.org/10.1177/0734242X20928411
21. Soares R, Soares CPA, Maestá BC, Lippi M, Cunha CESCP (2022) Evaluation of the efficiency of reverse osmosis in the treatment of sanitary landfill leachate in the metropolitan region of the Rio de Janeiro. Revista Virtual de Química 1–11. https://doi.org/10.21577/1984-6835.20220059
22. Chen Y, Zhang H, Liu Y, Bowden JA, Tolaymat TM, Townsend TG, Solo-Gabriele HM (2022) Concentrations of perfluoroalkyl and polyfluoroalkyl substances before and after full-scale landfill leachate treatment. Waste Manage 153(May):110–120. https://doi.org/10.1016/j.wasman.2022.08.024
23. Yang J, Xiang J, Xie Y, Yu K, Li J, Wang H, Li P, Gin KY-H, He Y (2022) Removal behavior and key drivers of antibiotic resistance genes in two full-

scale leachate treatment plants. Water Res 226(July):119239. https://doi.org/10.1016/j.watres.2022.119239

24. Theepharaksapan S, Chiemchaisri C, Chiemchaisri W, Yamamoto K (2011) Removal of pollutants and reduction of bio-toxicity in a full-scale chemical coagulation and reverse osmosis leachate treatment system. Biores Technol 102(9):5381–5388. https://doi.org/10.1016/j.biortech.2010.11.091

25. Rukapan W, Khananthai B, Chiemchaisri C, Chiemchaisri W, Srisukphun T (2012) Short- and long-term fouling characteristics of reverse osmosis membrane at full scale leachate treatment plant. Water Sci Technol 65(1):127–134. https://doi.org/10.2166/wst.2011.844

26. Šír M, Podhola M, Patočka T, Honzajková Z, Kocurek P, Kubal M, Kuraš M (2012) The effect of humic acids on the reverse osmosis treatment of hazardous landfill leachate. J Hazard Mater 207–208:86–90. https://doi.org/10.1016/j.jhazmat.2011.08.079

27. Şchiopu AM, Piuleac GC, Cojocaru C, Apostol I, Mămăligă I, Gavrilescu M (2012) Reducing environmental risk of landfills: leachate treatment by reverse Osmosis. Environ Eng Manag J 11(12):2319–2331. https://doi.org/10.30638/eemj.2012.286

28. Talalaj IA (2015) Removal of nitrogen compounds from landfill leachate using reverse osmosis with leachate stabilization in a buffer tank. Environ Technol (United Kingdom) 36(9):1091–1097. https://doi.org/10.1080/09593330.2014.982207

29. Cingolani D, Fatone F, Frison N, Spinelli M, Eusebi AL (2018) Pilot-scale multi-stage reverse osmosis (DT-RO) for water recovery from landfill leachate. Waste Manage 76:566–574. https://doi.org/10.1016/j.wasman.2018.03.014

30. Wu C, Li Q (2021) Organic matter characteristics removed from highly saline mature landfill leachate by an emergency disk tube-reverse osmosis treatment system. Chemosphere 263:128347. https://doi.org/10.1016/j.chemosphere.2020.128347

31. Yekani NH, Peyravi M, Jahanshahi M, Bagheri M (2022) Design and annual operation of full scale landfill leachate plant in Iran: focus on multistage membrane technology. Chem Pap 76(7):4509–4521. https://doi.org/10.1007/s11696-022-02108-0

32. García-Pacheco R, Galizia A, Toribio S, Gabarró J, Molina S, Landaburu-Aguirre J, Molina F, Blandin G, Monclús H, Rodríguez-Roda I, Comas J (2022) Landfill leachate treatment by using second-hand reverse osmosis membranes: long-term case study in a full-scale operating facility. Membranes 12(11):1170. https://doi.org/10.3390/membranes12111170

33. Collado S, Núñez D, Oulego P, Riera FA, Díaz M (2020) Effect of landfill leachate ageing on ultrafiltration performance and membrane fouling

behaviour. J Water Process Eng 36(March):101291. https://doi.org/10.1016/j.jwpe.2020.101291

34. Habert AC, Borges CP, Nobrega R (2006) Membrane separation processes, 1st edn. E-papers Serviços Editoriais Ltda, Rio de Janeiro In Portuguese

35. Alfaia RGSM, de Almeida R, Nascimento KS, Campos JC (2023) Landfill leachate pretreatment effects on nanofiltration and reverse osmosis membrane performance. Process Saf Environ Prot 172(January):273–281. https://doi.org/10.1016/j.psep.2023.02.038

36. Joo SH, Tansel B (2015) Novel technologies for reverse osmosis concentrate treatment: a review. J Environ Manag. https://doi.org/10.1016/j.jenvman.2014.10.027

37. Subramani A, Jacangelo JG (2014) Treatment technologies for reverse osmosis concentrate volume minimization: a review. Sep Purif Technol 122:472–489. https://doi.org/10.1016/j.seppur.2013.12.004

38. Voutchkov N, Kaiser GN (2020) Zero-liquid discharge concentrate disposal systems. In: Management of concentrate from desalination plants. Elsevier, pp 187–208. https://doi.org/10.1016/B978-0-12-818045-7.00009-9

39. de Almeida R, Porto RF, Quintaes BR, Bila DM, Lavagnolo MC, Campos JC (2023) A review on membrane concentrate management from landfill leachate treatment plants: the relevance of resource recovery to close the leachate treatment loop. Waste Manag Res J Int Solid Wastes Public Cleans Assoc ISWA 41(2):264–284. https://doi.org/10.1177/0734242X221116212

40. de Almeida R, Lanero F, Lavagnolo MC, Sgarbossa P, Bertani R, Vianna MM, Campos JC (2023) Thermal characterization of biochars produced in slow co-pyrolysis of spent coffee ground and concentrated landfill leachate residue. ECP 2023:12. https://doi.org/10.3390/ECP2023-14614

41. Hao W, Gao T, Shi W, Zhao M, Huang Z, Ren H, Ruan W (2022) Coagulation removal of dissolved organic matter (DOM) in nanofiltration concentrate of biologically treated landfill leachate by ZrCl4: Performance, mechanism and coagulant recycling. Chemosphere 301(April). https://doi.org/10.1016/j.chemosphere.2022.134768

42. Sun X, Wang X, Liu Y, Lian Y, Meng L, Su Z (2022) Removing refractory organic matter from nanofiltration concentrated landfill leachate by electrooxidation combined with electrocoagulation: characteristics and implication for leachate management. J Water Process Eng 47(March):102747. https://doi.org/10.1016/j.jwpe.2022.102747

43. Tejera J, Hermosilla D, Miranda R, Gascó A, Alonso V, Negro C, Blanco Á (2020) Assessing an integral treatment for landfill leachate reverse osmosis concentrate. Catalysts 10(12):1389. https://doi.org/10.3390/catal10121389

44. Keyikoglu R, Karatas O, Rezania H, Kobya M, Vatanpour V, Khataee A (2021) A review on treatment of membrane concentrates generated from landfill leachate treatment processes. Sep Purificat Technol 259(October 2020):118182. https://doi.org/10.1016/j.seppur.2020.118182
45. Li H, Liu H (2021) Treatment and recovery methods for leachate concentrate from landfill and incineration: a state-of-the-art review. J Clean Prod 329(November):129720. https://doi.org/10.1016/j.jclepro.2021.129720
46. de Almeida R, Porto RF, Hinojosa MAG, Sanches LCM, Monteiro BB, Conde ALFM, Costa AM, Quintaes BR, Bila DM, Campos JC (2022) Monitoring of experimental landfill cells with membrane concentrate infiltration: a systematic assessment of leachate quality and treatment performance. Process Saf Environ Prot 168:1155–1165. https://doi.org/10.1016/j.psep.2022.11.003
47. Li Q, Cui H, Li Y, Song X, Liu W, Wang Y, Hou H, Zhang H, Li Y, Wang F, Song J, Ye H, Song S, Che T, Shao S, Kong D, Liang B (2023) Challenges and engineering application of landfill leachate concentrate treatment. Environ Res 231(P1):116028. https://doi.org/10.1016/j.envres.2023.116028
48. Meier J, Remmen K, Wintgens T, Melin T (2021) Nanofiltration in landfill leachate treatment. In: Nanofiltration. Wiley, pp 663–690. https://doi.org/10.1002/9783527824984.ch15
49. Liu G, Han M, Luo W, Liu Y, Fang J, Tian M, Wang L, Zhao S, Meng F (2023) Loose nanofiltration—membrane distillation integrated process for resource recovery from leachate MBR effluent: performance and membrane fouling. Desalination, 545:116185. https://doi.org/10.1016/j.desal.2022.116185
50. dos Santos HAP, de Castilhos Júnior AB, Nadaleti WC, Lourenço VA (2020) Ammonia recovery from air stripping process applied to landfill leachate treatment. Environ Sci Pollut Res 27(36):45108–45120. https://doi.org/10.1007/s11356-020-10397-9
51. Hassen-Trabelsi AB, Kallel A, Ben Amor E, Cherbib A, Naoui S, Trabelsi I (2020) Up-grading biofuel production by co-pyrolysis of landfill leachate concentrate and sewage sludge mixture. Waste Biomass Valoriz 11(1):291–301. https://doi.org/10.1007/s12649-018-0399-2

© The Author(s), under exclusive license to Springer Nature Switzerland AG 2024
S. Souabi, A. Anouzla (eds.), *Landfill Leachate Treatment Techniques*, Springer Water
https://doi.org/10.1007/978-3-031-63157-3_9

9. A Review of Strategies for Managing Membrane Concentrate from Landfill Leachate Treatment Facilities

Letícia Barbosa Alves[1] ✉, Ronei de Almeida[2] ✉ and Juacyara Carbonelli Campos[1] ✉
(1) School of Chemistry, Inorganic Processes Department, Universidade Federal do Rio de Janeiro, Rio de Janeiro, Brazil
(2) Department of Sanitary and Environmental Engineering, Rio de Janeiro State University, Rio de Janeiro, Brazil

✉ **Letícia Barbosa Alves**
Email: leticiabalves@eq.ufrj.br

✉ **Ronei de Almeida (Corresponding author)**
Email: ronei.almeida@eng.uerj.br

✉ **Juacyara Carbonelli Campos**
Email: juacyara@eq.ufrj.br

Abstract
Nowadays, the landfill leachate treatment chain is characterized by different methods arranged in integrated

schemes to ensure proper landfill wastewater purification. Nanofiltration (NF) and reverse osmosis (RO) have been used as tertiary steps to purify landfill wastewater efficiently. However, a challenge that must be dealt with is managing membrane-concentrated streams. NF and RO concentrate have high salinity and organic loads. Against this backdrop, there is an urgent need to address leachate treatment and membrane concentrate management. The current chapter assesses NF/RO concentrate characteristics (1), reviews the management strategies to handle landfill leachate membrane concentrate (2), and summarizes the advantages and disadvantages of existing treatment processes (3). The membrane concentrate disposal into the landfill is the most convenient management option. However, the consequences of this practice are not entirely understood. Integrated treatment systems combining thermal process, ozonation, and solidification/stabilization are the most effective solutions for concentrate management. However, operating costs and secondary pollution can hinder their full promotion. Future studies on developing low-cost systems and cost–benefit analysis in pilot and full-scale are recommended.

Keywords Concentrated leachate – Landfill wastewater – Membrane concentrate – Nanofiltration – Reverse osmosis

---

9.1 Introduction

Landfill leachate (LL) is a prominent environmental risk factor due to its high organic and ammonia concentrations [72]. Identifying treatment schemes that generate effluents with standards compatible with disposal requirements is challenging. The treatment of LL often requires a combination of biological degradation processes, such as stabilization ponds, activated sludge and combined treatment with sanitary sewage in sewage treatment plants

[119]. Biological treatments are effective for LL management produced in young landfills, whose organic fraction mainly comprises volatile fatty acids, but insufficient when handling LL from mature landfill sites [113]. Other techniques used in the leachate treatment chain comprise air stripping, chemical precipitation, and coagulation-flocculation. These processes are specially employed to remove ammonia, metals, and colloids, respectively [22, 43].

Due to their cost-effectiveness and landfill wastewater composition complexity, membrane techniques (e.g., nanofiltration and reverse osmosis) have been used as integral or polishing treatment [26]. On the other hand, generating landfill leachate membrane concentrate (LLMC) —a dark-colored solution containing refractory pollutants and high salinity—represents one of the main disadvantages of membrane technology [115]. The concentrated leachate produced in membrane treatments accounts for 13–50% of the incoming leachate [26, 39]. The LLMC composition depends on the incoming LL and the used membrane treatment. Overall, the LLMC is typically characterized by a high level of salts and low biodegradability ($BOD/COD < 0.3$) [115], making this stream practically untreatable by conventional treatments such as biological systems [17, 25, 39].

Membrane concentrate management strategies include natural evaporation, solidification/stabilization (S/S), chemical coagulation, electrocoagulation (EC), ozonation, and advanced oxidation processes (AOPs) like Fenton, photo-Fenton, anodic oxidation, as well as thermal treatment [19, 39]. The injection into landfills remains a common strategy for dealing with LLMC, especially in developing countries. However, the osmotic pressure might increase sharply, leading to a decrease in the membrane treatment efficiency and, in turn, inhibiting the operation of LL treatment [50]. Therefore, finding a proper LLMC

management solution is a challenge faced by landfill owners and practitioners.

Against this backdrop, the present chapter reviews the management strategies to handle LLMC. Specific objectives include discussing concentrate characteristics (1), reviewing the management strategies to handle LLMC streams, and summarizing the advantages and disadvantages of existing treatment processes.

9.2 Composition of LLMCs

High-pressure membrane systems, such as nanofiltration and reverse osmosis, are the most common membrane technologies in the leachate treatment chain. As mentioned, membrane concentrate from these systems accounts for 13–50% of the incoming leachate [26, 39]. Table 9.1 shows the typical nanofiltration concentrate (NFC) and reverse osmosis concentrate (ROC) pollution parameters found in the recent literature.

Table 9.1 The main characteristics of LLMCs from RO and NF systems [9, 38, 83, 95, 119]

Parameter	NFC	ROC
pH	5.6–8.9	6.2–8.3
COD (g/L)	1.28–9.50	1.65–49.52
NH_3-N (mg/L)	14.20–3276	62.90–8300
TN (mg/L)	592–3045	244.5–4575
BOD_5	2.6–288	28.5–4800
BOD_5/COD	<0.09	0.01–0.55
Chloride (g/L)	0.23–20.12	1.82–30.77
Sodium (mg/L)	890–4206	15,400
Potassium (mg/L)	210–2806	9600
Copper (mg/L)	0.01–9,26	0.022–1.49
Lead (mg/L)	0.006–56.97	0.05–0.54

Parameter	NFC	ROC
Nickel (mg L^{-1})	0.08–3.182	0.20–1.59
Sulfate (mg/L)	52.8–3409	20–4030
TOC (mg/L)	454–2365	818–3600
EC (mS/cm)	3.1–79.4	10.5–98
HS (mg/L)	910–1501.2	1838
HA (mg/L)	264.6–856	94.9
FA (mg/L)	921–1671	441.7

COD Chemical oxygen demand; BOD_5: 5-day Biochemical oxygen demand; NH_3-N: Ammoniacal nitrogen; *TOC* Total organic carbon; *EC* Electrical conductivity; *HS* Humic substances; *HA* Humic acid; *FA* Fulvic acid; *NFC* Nanofiltration concentrate; *ROC* Reverse osmosis concentrate

NFC and ROC streams are characterized by high concentrations of organic matter, salts, and total volatile solids, for example, are identified through the characterization. The organic matter reported as COD ranges from 1.28 to 49.52 g/L. It was also demonstrated that humic substances accounted for a higher fraction of organic matter (60–70%) [119]. High COD values associated with low BOD5 values also demonstrate the concentrate's low biodegradability, which can be attributed to the fact that readily biodegradable fractions could be decomposed into carbon dioxide and water in aerobic/anaerobic biological reactions. At the same time, high-molecular-weight and non-biodegradable compounds can be concentrated during the membrane filtration process and accumulated in the concentrated leachate [57].

Salts, including chlorine, sodium, and potassium, are found in concentrations up to 30, 15, and 9 g/L, respectively. These values are in the same range as those found in brines produced in seawater desalination plants

[67]. As reverse osmosis usually rejects salts, salts will be presented in high concentrations in the concentrate stream, which impairs the disposal of concentrate streams without undergoing a treatment to manage the concentrated leachate [60, 75]. The increased salt concentration can also be linked to the electrical conductivity, ranging from 3.1 to 98 mS/cm. In addition to the high amount of organics and salts, NFC and ROC can be high-rich in nutrients like nitrogen. For instance, values of ammonia nitrogen were reported in the range of 14.20–3276 and 62.90–8300 mg/L for NFC and ROC, respectively. In some cases, nitrogen can be removed by biological processes in the leachate treatment chain, which reduces its concentration in the membrane concentrate [19].

It is important to note that different from the incoming leachate, NFC and ROC can be polluted by heavy metals. For example, Ni, Pb, and Cu were found in levels (3183, 56.97, and 9.26 mg/L, respectively) higher than that established by the Brazilian legislation for effluent discharge (Conama n. 430/2011). The concentrations of As, Cd, Cr, Co, Cu, Ni, Pb, and Zn, among others, depend directly on the type of waste deposited in landfills. According to Carvajal-Flórez and Santiago-Alonso Cardona-Gallo [10], the concentrations of heavy metals in the raw leachate generally decrease as the age of the landfill increases.

Nowadays, more attention has been directed to contaminants of emerging concern (CEC), identified in several wastewaters, including NFC and ROC [26]. CECs are bioactive and bioaccumulative pollutants that can have widespread occurrence and persistence to a greater extent in industrialized regions and even in remote areas. The main CECs identified are polyaromatic hydrocarbons (PAHs), per/polyfluoroalkyl substances (PFASs), micro- and nanoplastics, antibiotics/antibiotic-resistant genes (ARGs), and pharmaceuticals and personal care products (PPCPs),

which have been associated with the changing composition of municipal solid waste as a result of technological development and uncontrolled urbanization. Emerging pollutants can enter the aquatic environment through leachate runoff and soil and groundwater contamination. CECs are mixtures that can increase overall toxicity and produce synergistic effects. Thus, toxicity can be a function of the sum of the concentrations of individual compounds. Therefore, it can occur even when they are present at no-observed-effect concentration (NOEC) levels [26]. Therefore, more studies are needed on this topic.

9.3 Overview on Strategies for LLMC Management

9.3.1 Infiltration of Concentrate Stream into the Landfill

When the reverse osmosis (RO) or nanofiltration treatment is adopted, one of the most debated options is landfill leachate membrane concentrate (LLMC) recirculation (Fig. 9.1). This process is similar to the recirculation of the raw leachate to adjust moisture and nutrient content in landfills due to the good cost–benefit ratio of the recirculation of leachate concentrate in landfills, even in the face of the pros and cons that will be presented here, this remains one of the leading choices for LLMC destination [9, 51, 54, 75].

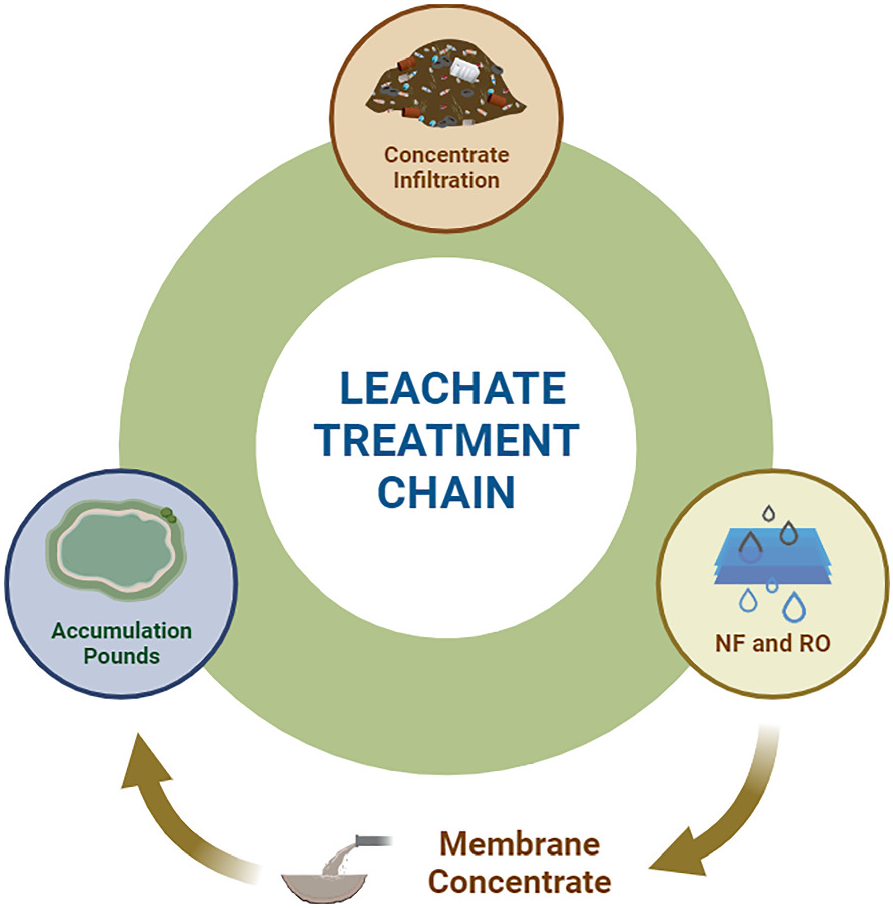


Fig. 9.1 Diagram of membrane concentrate management in the leachate treatment chain

Talalaj and Biedka [90] evaluated the impact of this recirculation in the short term (one year) in a landfill located in the province of Warminsko-Mazurskie, Poland. In the first few months of its application, infiltration caused a decrease in the pH value of the generated leachate and increased ammonia nitrogen, chlorides, electroconductivity, and chemical oxygen demand. According to the authors, there was an increase in the concentration of organic compounds (COD and BOD) until the first six months, after which there was a decrease and subsequent stabilization, indicating that the recirculation of the concentrate accelerates the decomposition of organic matter, which was particularly noticeable in the landfill cell with the freshest waste.

According to [63], the impact of recirculating a leachate concentrate on leachate quality is negligible or limited to the period immediately after re-injection. On the other hand [88] argue that recirculation of the concentrate is unsuitable in the long term due to its adverse effects on leachate production. Calabrò et al. [9] show that the only change observed after the start of concentrate recirculation is a moderate increase in COD, nickel, and zinc. Since the recirculated concentrate leachate contains a high amount of sulfates, the issues related to the presence of sulfate-reducing bacteria are discussed both in terms of metal concentration control through the precipitation of metal sulfides and possible partial inhibition of methanogens due to the competition of the same sulfate-reducing bacteria.

Džolev and Vujić [21] documented that recirculation of concentrate from leachate treatment by reverse osmosis can generate additional amounts of landfill gas and methane from degraded waste under anaerobic methanogenic conditions. Numerical results obtained by laboratory analysis of the organic matter in the concentrate and by measuring the composition of the landfill gas in the wells showed that 20% of the total organic matter in the concentrate participates directly in methane production. The most important influence is the moisture supplied to the landfill, especially in climatic areas with low precipitation.

Robinson [78] presented monitoring data of a German landfill that operated an RO system for one year, returning the concentrate to the landfill. He showed that ROC infiltration increased the ammonia and salinity of the generated leachate, which immediately affected the treatability performance of the RO system. Similar results were found by Talalaj and Biedka [90]. On the other hand, 15-year monitoring data of an Italian landfill revealed a moderate change in leachate quality (slight increase in

NH_4^+, Cl^-, and SO_4^{-2}) and an increase in leachate quantity. However, RO treatment performance was unaffected [9]. The literature shows that the consequences of concentrate disposal to landfill are site-specific. If adopted as a management strategy, suitable engineering design must be followed by monitoring of site conditions [19].

Table 9.2 shows the advantages and disadvantages of the concentrate recirculation practice presented in the literature.

Table 9.2 Advantages and disadvantages of recirculating LLMC streams in landfills

Advantages	Disadvantages	References
Distribution of nutrients and enzymes	Accumulation of chloride and ammonium ions	[90]
pH buffering	Increase in chemical oxygen demand	[63]
Reduces the cost for concentrated leachate treatment		[94] [28]
Increases waste biodegradation		[63]

Some advantages of infiltrating this concentrate include nutrient and enzyme distribution, pH buffering, liquid storage, evaporation opportunities, enhanced biodegradation of organic matter, and stabilizing the waste quickly. Also, increased methane production rate and reduced leachate treatment costs are reported in published studies [63, 94]. On the other hand, the disadvantages of leachate recirculation, such as the accumulation of chloride and ammonium ions, were highlighted by Talalaj and Biedka [90]. It can lead to an inhibition of methane genesis due to a high concentration of organic acids and a low pH [63]. Landfill slope failures and related issues are also reported as a possible consequence of the infiltration practice [19].

9.3.2 Leachate Membrane Concentrate Treatment

Landfill leachate membrane concentrate treatment options include thermal, physicochemical, and biological processes (Fig. 9.2). This section summarizes the main aspects, advantages, and limitations of management options to handle concentrated leachate.

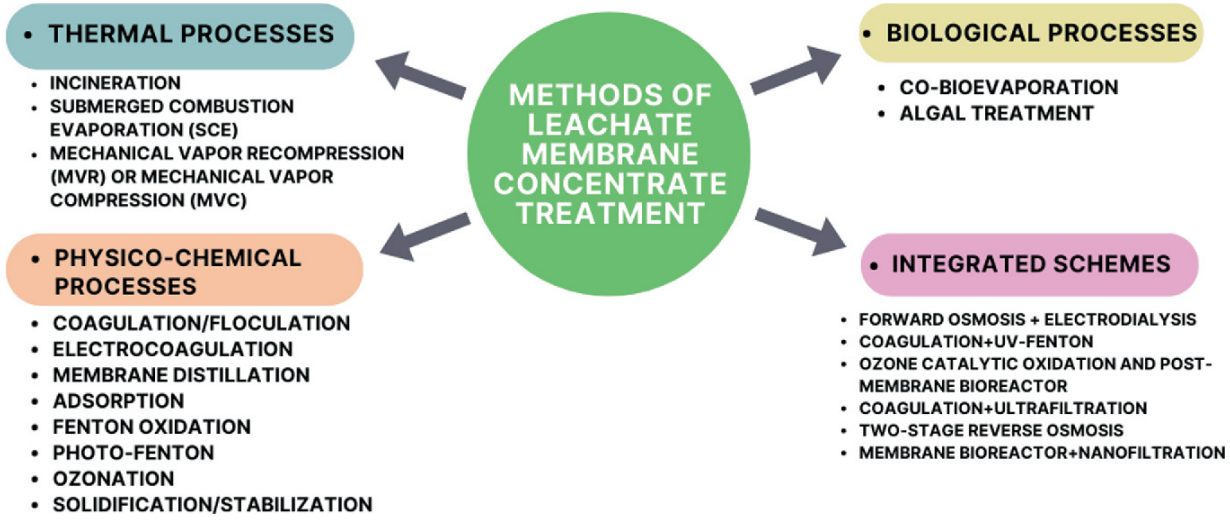


Fig. 9.2 Methods for leachate membrane concentrate treatment

Thermal processes

Incineration. Typically, incineration is carried out in a reactor containing primary and secondary chambers. In these chambers, the waste is initially heated and volatilized to temperatures > 800 °C (~1500 °F) in the first chamber. The by-products of incomplete combustion are further treated in the second chamber at temperatures > 1100 °C (~2000 °F) with excess oxygen. Liquid residues can also be sprayed into a secondary chamber [96]. Some factors influence the efficiency of thermal reactors, such as the properties of the waste and the presence of phosphates and silicates [96]. Operating conditions (e.g., residence time, operating temperature, mixing/turbulence, and presence of

excess oxygen) or addition of additives (CaO, KOH, Al_2O_3, La_2O_3, Na_2FeO_4, $Na_2S_2O_8$, etc.) is also essential [89, 108].

Spraying the membrane concentrate into the incinerator is a simple, effective, and commonly used method. However, the interior wall of this equipment might suffer severe corrosion due to the high concentration of chlorides and sulfates in the concentrated leachate [12]. Also, the management of byproduct like incineration ashes is a issue that need to be deal with. Therefore, LLMC spraying is being gradually phased out in most incineration plants.

In the United States of America, an estimated 13% of MSW is managed through incineration, resulting in approximately 7.5 million metric tons of ash being disposed of in landfills yearly [115], this percentage is even higher in many European and Asian counties [105]. Another issue is Per/polyfluoroalkyl substances (PFASs) presents in concentrated leachate. These compounds are considered contaminants of emerging concern (CEC) and tends to bioaccumulate in membrane concentrates [26]. PFASs in leachate affects the wastewater treatment plant performance and receiving water bodies if discharged without removal [86].

Submerged Combustion Evaporation (SCE). Submerge combustion evaporation (SCE) is an evaporation process in which hot flue gases bubble through the sub-evaporated liquid. Direct heat transfer occurs between the hot gases and the liquid without any fixed interface [45]. The process has been considered one of the most effective for concentrating membrane concentrate, where the leachate volume can be condensed more than ten times by SCE technology [45, 110]).

Yue et al. [117] developed an innovative SCE-based evaporation system consisting of two-stage evaporators. This system is applied in a full-scale facility to treat membrane concentrate from a membrane biological reactor

followed by an RO train. The leachate membrane concentrate contains a high COD value of up to 13,200 mg/L. The treatment capacity of the full-scale two-stage SCE system is 20 m^3/d; 1 m^3 of concentrated leachate can be directly treated using 150 m^3 of landfill gas (which contains 60% of methane). In the SCE process, the steam from the evaporator in the second stage is used to preheat the leachate, reducing energy consumption in the first chamber. The condensate is discharged directly into the irrigation canal in batches. At the same time, solid residues are disposed of in the landfill after solidification with a volume of 0.5–1 m^3 once per batch every six hours. The SCE system has operated since October 2004 at the Beishenshu Landfill in Beijing, China. Initially, there was no other leachate treatment except the recirculation method in the landfill. The combined process of MBR and RO, as well as evaporation, was designed to reduce the threat of leachate accumulation in concentrate pounds [117].

Physico-chemical processes

Coagulation/Flocculation. Coagulation is a simple and cost-effective technique for treating leachate membrane concentrates. It effectively removes dissolved organic carbon (DOC) and improves leachate membrane concentrates' biodegradability [16, 103]. Aluminum salts, iron salts, and poly aluminum chloride are common coagulants used in wastewater treatment [77]. Zhou et al. [121] found that coagulation with $FeCl_3$ removed 26% of DOC in ROC produced in a leachate treatment plant.

Coagulant type and dosage and operating conditions affect the efficiency of COD removal, so the selection of a suitable chemical, the determination of optimal experimental conditions, the evaluation of the effect of pH, and the determination of the optimal reagent dosage are

necessary for the optimization of performance [4]. Long et al. [57] used different types of coagulants to treat leachate concentrate streams. $FeCl_3$ obtained the highest organic matter removal efficiency compared to the other coagulants (i.e., $FeCl_2$, $FeSO_4$, and polyaluminum chloride). The COD concentration of the leachate concentrate was reduced from 4000 mg/L to 718 mg/L by coagulation with $FeCl_3$ at pH 4.0. These results demonstrated that coagulation with $FeCl_3$ is feasible for LLMC treatment. Similarly, in another study, $FeCl_3$ showed higher DOC removal efficiency (52%) than alum (42%) at the optimum dose of 0.4 mM [20].

Electrocoagulation. Electrocoagulation destabilizes suspended, emulsified, or dissolved contaminants in an aqueous medium by introducing an electric current into the medium [23]. Various materials can be used for the anodes, such as Pt, TiO_2, SnO_2, Al, and Fe. For example, aluminum anodes have the same effect as adding Al-based coagulants in conventional treatment systems [23]. Electrocoagulation is an alternative technology for water and wastewater treatment systems and is more effective in removing inorganic contaminants and pathogens [23, 33]. Compared to conventional chemical coagulation, electrocoagulation has many advantages, such as simple equipment, easy operation, and automation, a short retention time, low sludge production, and no chemical need [69]. Regarding membrane treatment of concentrate, electrochemical treatment has several advantages, such as moderate to high salinity, which ensures excellent electrical conductivity that can reduce energy consumption, and high chloride concentration, which improves indirect oxidation by the electrogenerated ion of strong oxidants [69].

Top et al. [95] characterized and investigated the treatment capacity of NF concentrate from a full-scale

leachate treatment plant. The performance of the electrocoagulation process for the treatment of NF concentrate was reported under different current densities and treatment times. The efficiency of the Al electrodes in treating the nanofiltration concentrate from the leachate treatment plant was investigated through the removal of color, COD, and phosphorus. The results indicated high removal efficiencies of around 45%, 60%, and > 90% for COD, color, and total phosphorus. The final concentration of NH_3-N in the treated concentrate ranged from 30 to 200 mg/L.

Forward Osmosis: Forward Osmosis (FO) technology capitalizes on the osmotic pressure gradient to move freshwater across a selectively permeable membrane. In this mechanism, a draw solution (DS) possessing elevated salinity establishes a discrepancy in osmotic pressure, prompting water to move from the feed effluent across the semipermeable FO membrane. Following the FO process, additional steps are necessary to separate the diluted DS, yielding reclaimed water as the end product [19, 82].

The prevalent draw solutes used in FO applications are those derived from ammonia and carbon dioxide, such as NH_4HCO_3. After the FO filtration process is completed, the DS is reclaimed and reintroduced into the FO system for recycling. Meanwhile, the permeate might necessitate additional treatment before its potential discharge or reuse [1, 107].

FO systems that lack DS regeneration exhibit notably reduced energy consumption (0.2–0.9 kWh/m^3) compared to systems with DS regeneration (10–14 kWh/m^3) [100]. [47] conducted an investigation involving an FO system employing NH_4HCO_3 (3 mol/L) as the DS for landfill leachate treatment. The FO process achieved a recovery rate exceeding 90%, yielding water that adhered to regulatory standards suitable for agricultural fertigation. In

another study [70] suggested the utilization of recovered NH_4HCO_3 (2 mol/L) as the DS in a hybrid FO system designed for water reclamation from landfill wastewater.

Membrane Distillation (MD). Membrane distillation (MD) is a promising new desalination technology. Unlike other pressure separation technologies, MD is driven by the vapor pressure difference between the two sides of the hydrophobic microporous membrane [13]. Among its advantages, MD can operate at atmospheric pressure. The solution does not need to heat up to the boiling point, which requires less energy and a less expensive membrane such as plastic, thus avoiding the problem of corrosion [2].

In addition, MD also has a high rejection of salt and non-volatile components [85], indicating that it is a process with potential for concentrate streams in many aspects, such as concentrated coal gasification brine and brewery wastewater [5, 37, 74], and can also be an effective method to treat LLMC [13], processing wastewater from livestock [105], olive mill wastewater [22], biologically treated coking wastewater, as well as other organic wastewaters [54, 87].

However, challenges in treating leachate concentrate by MD are the high concentration of refractory organic pollutants and metal ions such as Ca^{2+} . The fouling layer caused by the refractory organic foulants can produce significant heat and mass transfer resistance, thus contributing to unexpected flow decline [91]. Calcium ions can form $CaCO_3$ and $CaSO_4$, aggravating severe membrane fouling. This problem might be even worse at elevated temperatures, which is the normal operating condition of the MD process [103]. Besides, organic fractions (e.g., humic acids) might interact with Ca^{2+} ions, forming organic complexes and causing more severe fouling due to coagulation/precipitation of organic complexes on the membrane surface [46].

Adsorption. The adsorption process—a surface phenomenon by which a fluid mixture (gas or liquid) of various components is attracted to the surface of a solid adsorbent and forms bonds through physical or chemical linkages—is recognized as the most efficient and promising approach in wastewater treatment [18]. A notable trend in the development of activated carbon, an adsorbent with its sizeable porous surface area, controllable pore structure, thermostability, and low acid/base reactivity, has been promulgated due to its superior ability to remove a wide variety of organic compounds and dissolved inorganic pollutants in aqueous media, even in a gaseous environment [14, 51]. In most cases, activated carbon adsorption revealed the prominence of removing an essential amount of organic compounds and ammonia nitrogen from leachate samples [14].

In Malaysia, a comparative study for ammonia nitrogen removal was conducted by Aziz et al. [7] using granular activated carbon (GAC) and limestone at the Burung Island landfill. Approximately 40% of ammonia nitrogen with an initial concentration of 1909 mg/L was eliminated with 42 g/L GAC. Meanwhile, 19% removal was achieved using 56 g/L limestone at the exact dosage.

A substantial amount of simultaneous adsorption and biological treatment investigations have been practiced, offering several advantages, including increased nitrification efficiency (activity of nitrifiers), improved sludge dewaterability (filterability), and removal of refractory organic compounds [35, 66, 76, 109]. Under the co-treatment with biological processes, activated carbons are believed to contribute a synergy effect to provide a binding surface for bio regeneration (microorganisms) and serve as a nucleus for floc formation [11]. Furthermore, it has been associated with beneficial biodegradation as a support medium in biofilm reactors and with leachate

buffering effects when combined in domestic wastewater and landfill leachate treatment systems [35].

In the study proposed by Mojiri et al. [62], concentrated landfill leachate was treated using a combined treatment technique that included electro-ozonation and a sequencing batch reactor (SBR) process augmented with composite adsorbent. After treatment with optimum electro-ozonation performance—optimum reaction time (96.9 min) and ozone dosage (120.0 mg/L), COD, color, and nickel removal efficiencies of 64.8%, 90.4%, and 52.9%, respectively—the concentrated leachate was transferred to the SBR reactor augmented with powdered composite adsorbent (P-BAZLSC) (PB-SBR). The PB-SBR improved COD, color, and nickel removal efficiency from 64.8% to 88.2%, 90.4% to 96.1%, and 52.9% to 73.4%, respectively.

In recent years, activated carbon adsorption processes have prevailed as one of the most satisfactory and viable options for treating contaminated landfill leachate combined with ozonation oxidation [62]. After decomposition, ozone oxidizes organic substances to their more stable oxidation states, producing water and carbon dioxide. Activated carbon can also remarkably accelerate the kinetic rate by forming hydroxyl radicals ($^{\bullet}OH$) [76].

Fenton oxidation. The catalysis of hydrogen peroxide by ferrous sulfate, which is Fenton's reagent, is one of the most common advanced oxidation processes. The Fenton reagent catalyzes $^{\bullet}OH$ from the chain reaction between ferrous ions and hydrogen peroxide. The primary reaction in the Fenton process is the one between H_2O_2 and ferrous ions to produce hydroxyl radicals, as described in Eq. (9.1). Many secondary reactions take place during the Fenton reagent, including the scavenging reactions of produced hydroxyl radicals by both ferrous ions (Eq. 9.2) and hydrogen peroxide (Eq. 9.3) in excess, which makes reactants dosage a critical parameter [61, 81].

$$H_2O_2 + Fe^{2+} \rightarrow Fe^{3+} + HO^- + {}^{\cdot}OH \quad (9.1)$$

$${}^{\cdot}OH + Fe^{2+} \rightarrow Fe^{3+} + HO^- \quad (9.2)$$

$${}^{\cdot}OH + H_2O_2 \rightarrow H_2O + {}^{\cdot}O_2H \quad (9.3)$$

Extensive research has been increasingly focused on treating landfill leachate by the Fenton process, which was used as a post-treatment or pretreatment since biological treatment alone cannot remove leachate's refractory organics [79]. Compared with other advanced oxidation techniques, the Fenton process is simpler and easier to operate. In addition, Fenton process reagents are readily available, easy to store, and relatively safe to handle [68].

He et al. [28] studied Fenton oxidation to improve the biodegradability of concentrated leachate from the membrane process, evaluated in terms of the BOD_5/ COD ratio. The COD and BOD_5 concentrations and the BOD_5/COD ratio of the raw concentrated leachate from the membrane process were 3060 mg/L, 288 mg/L, and 0.094, respectively. The COD removal efficiency of the concentrated leachate increased from 26.1 to 63.5%, with the Fe^{2+} concentration increased from 100 to 1000 mg/L at pH 3, the H_2O_2 /Fe^{2+} mass ratio of 9, and the mass reaction time of 2 h. When the Fe^{2+} concentration was increased to 800 mg/L, the COD removal efficiency increased to 63.3%, close to that of the Fe^{2+} concentration of 1000 mg/L (63.5%). The BOD_5/COD ratio of the concentrated leachate increased from 0.12 to 0.54 when the Fe^{2+} concentration was increased from 100 to 600 mg/L. At the same time, it showed a decreasing trend when the Fe^{2+} concentration was higher than 600 mg/L.

The BOD_5/COD ratio was 0.53 at the Fe^{2+} concentration of 400 mg/L, close to that at the Fe^{2+} concentration of

600 mg/L (0.54). Considering the cost of the Fe^{2+} reagent and the load imposed by the iron sludge, 400 mg/L was selected as the ideal Fe^{2+} concentration for the Fenton process to improve the biodegradability of the concentrated leachate [53].

However, the toxicity of some intermediates after the Fenton process may be higher than that of the initial compounds. For example, the toxicity of hydroquinone and p-benzoquinone, which are formed in the Fenton oxidation of phenol, is several orders of magnitude higher than that of phenols present in the wastewater [81].

Photo-Fenton. The photo-Fenton method combines the Fenton process with ultraviolet–visible radiation to produce additional •OH through the decomposition of H_2O_2. By applying an ultraviolet (UV) source, the extra •OH is produced in the presence of Fe2 + and regeneration of the catalyst occurs via photo-reduction from Fe^{3+} to Fe^{2+} [65]. Therefore, UV radiation integrated into the Fenton process has been shown to achieve COD removal efficiencies higher than the Fenton process. For example, the application of photo-Fenton increased the COD removal efficiency of the Fenton reagent from 73 to 92.8% when treating concentrated leachate [48]. In another study, the photo-Fenton removed 94.75% of COD, while the Fenton process removed 89.60% under the same conditions [103]. In addition, only the UV-Fenton process could remove the genotoxicity from the membrane entirely concentrates.

An integral three-step treatment process for reverse osmosis concentrate from leachate treatment plants was proposed and evaluated by Tejera et al. [93] to reduce organic matter content and conductivity and increase biodegradability. The first stage consisted of a coagulation/flocculation process, the best results of which were obtained using a dosage of 5 g/L of ferric chloride at an initial pH = 6 (removal of 76% of COD, 57% of the

specific ultraviolet absorption [SUVA] and 92% of color). The second stage applied the photo-Fenton process, which resulted in more excellent biodegradability (i.e., the ratio between BOD_5 and COD increased from 0.06 to 0.4), and a further 43% of COD was removed under the best reaction conditions tested of $[H_2O_2]/COD = 1.06$, $pH = 4$ and $[H_2O_2]/[Fe]mol = 45$. Overall, the complete treatment achieved the removal of 99.9% color, 90% COD, 90% sulfate, 90% nitrogen, 86% Al, 77% Zn, 84% Mn, 99% Mg, and 98% Si, and significantly increased biodegradability up to $BOD_5/COD = 0.4$.

Ozonation. The ozonation process is based on the ozone (O_3) oxidative potential, a powerful oxidant ($E0 = 2.08$ V). The effectiveness of this process can be significantly improved by the secondary formation of •OH through the decomposition of O_3 in an alkaline environment [99]. The characteristics of wastewater, such as pH, alkalinity, and natural organic matter, strongly influence ozone's stability. Ozone decomposition can also be promoted by external interventions such as increasing pH or adding H_2O_2 [79]. The ozonation process can decompose complex organic molecules into low molecular weight substances and, therefore, may be attractive to be applied to leachate concentrates. It is mainly used as a pretreatment step rather than a stand-alone process because it can increase wastewater's biodegradability by altering the refractory organics' molecular structure. However, carbonate ions can negatively affect ozonation, which can act as •OH radical scavengers [101].

Wang et al. [103] investigated the decomposition of humic substances and transformation behaviors during the treatment of nanofiltration concentrate (NFC) and reverse osmosis concentrate (ROC) streams by ozonation. The COD removal efficiency of the process was obtained as 55.5% for

ROC and 43.2% for NFC. These values did not meet the effluent discharge limits (COD limit of 100 mg/L according to pollution control regulations in China). Therefore, additional biological treatment was required. Still, the process was able to degrade large molecules into small molecular weight molecules and remove the non-degradable humic acids altogether.

The microbubble ozonation (MB-O_3) process was developed to treat NFC [101]. MB-O_3 is intended to boost ozone transfer mass into the bulk solution. Using an ozone dosage of 2.4 g/L, the process achieved 76% COD and 69.96% TOC removal efficiency in 120 min of reaction time. In addition, the MB-O_3 process was compared with a conventional ozonation system, which generally has medium or large bubble sizes. While the TOC removal efficiency of the ozonation processes with medium and large bubble sizes was 26.7% and 29.3%, the efficiency of the MB-O_3 process was 62.7% in a reaction time of 75 min [101].

In addition to modifying conventional ozonation, utilizing it as a co-oxidant or UV irradiation can increase the effectiveness of the ozonation by promoting the formation of additional $^{\bullet}OH$ [79]. The performance of three ozone-based processes (i.e., O_3, O_3/H_2O_2, and O_3/UV) was compared in the treatment of NFC [101]. The COD removal efficiency of a single O_3 process (41.7%) was improved by the introduction of H_2O_2 (51.7%) and UV (56.7%). In a similar study, adding H_2O_2 to the ozonation process increased the effectiveness of O_3 [3]. The O_3/H_2O_2 process increased the COD removal efficiency of simple ozonation (34%) to 44%.

Solidification/Stabilization(S/S).
Solidification/stabilization(S/S) is the transformation and

enclosure of contaminated material into a physically durable and slightly permeable form to prevent the transport of hazardous pollutants into the environment [29]. The S/S process involves mixing hazardous waste with hydraulic binders to produce a stabilized solid material to ensure safe transportation and storage [29]. Zeolite, sand, or gravel are used as additional aggregate materials in the solidification/stabilization.

Hunce et al. [31] used Portland cement as a solidifying agent to solidify ROC. It was reported that only about 1% of TOC, DOC, TDS, and ammonium from the leachate concentrate remained in the eluate water, and the remaining pollutants were successfully solidified. Various binders can be used for solidification stabilization processes. Hendrych et al. [29] conducted a study investigating S/S treatment using Portland cement and possible substitutes for solidifying agents. As binder, fly ash, reactivated gypsum, ground blast furnace slag, and dastit® were selected. Different mixing ratios and cost analyses were carried out for various binders. Therefore, regarding cost and efficiency, strength tests should be preferred to apply cheap and highly qualified binders such as Portland cement. It has been shown that solids formed from alternative binders can also be applied for reuse.

The S/S of LLMC has also been investigated using brick waste as a substitute for Portland cement. It was reported that brick waste can be used as an alternative to Portland cement when the mixing ratio does not exceed 25% [36]. The solid products of the solidification process passed compressive and flexural strength tests, therefore, it would be possible and safe to cast valuable masonry units. Concerning environmental impact, the concrete produced was suitable and effectively reduced most contaminants in the LLMC below regulatory limits. However, estimating the useful life of such solidified waste for storage or reuse is very difficult [31]. Solidification and stabilization can be

chosen for LLMCs that are rich in heavy metals and contain a low amount of biodegradable substances. However, after the S/S processes, an increase in the volume and mass of the final waste is observed [36].

Biological processes

Co-bioevaporation. The co-evaporation process has been recently investigated to treat landfill leachate membrane concentrate (LLMC). This technology adds an organic substrate, such as food waste, as a supplementary carbon source to handle wastes [114] jointly. With the degradation of organic matter and the accumulation of metabolic heat, the process temperature can reach 60–70 °C. The rapid increase in temperature affects co-bioevaporation in two opposite ways [112]. With increasing temperature, chemical and enzymatic reactions in the cells occur faster, and degradation of organics becomes more efficient, at the same time, water evaporation is promoted. Above a specific temperature, however, denaturation of one or more essential cellular components, such as a critical enzyme, hinders metabolic activity and decreases the rate of co-bioevaporation [114]. At the correct temperature, thermophilic bacteria can be amplified 1000-fold and mesophilic bacteria 100-fold [112].

Extracellular enzymes such as protease, urease, and phosphatase play a crucial role in co-bioevaporation as they catalyze the hydrolysis of insoluble organic matter into small, soluble molecules that can quickly enter cells and be digested by microorganisms [15, 116]. Protease participates in the hydrolysis of proteolysis [64]. Urease hydrolyzes urea to form carbonic acid and ammonia [27], and phosphatase promotes the hydrolysis of organic phosphorus compounds into different inorganic forms by microorganisms [104]. In addition, dehydrogenase as oxidoreductase catalyzes ATP production by oxidizing

organic substances, and adenosine triphosphate (ATP) fuels microbial activity. These enzymes strongly influence the degradation rate [6].

Yang et al. [114] studied the treatment of LLMC from the two-stage disk-tube reverse osmosis (DTRO) process by the co-bioevaporation process with the addition of food waste (FW) as a supplementary carbon source. A series of experiments were conducted to explore the feasibility and critical factors of co-bioevaporation treatment of LLMC and FW. The results indicated that the simultaneous removal of water and organic matter in the LLMC and FW mixture was effectively achieved in the co-bioevaporation process. At LLMC: FW mixture ratio 1:1.10 m/m%, 85.67% of water was removed. The optimum operating conditions were initial moisture content of 33.9%, dosage of 90%, and airflow rate of 0.035 m^3/kgTSh. At optimal operating conditions, removals of water and volatile solids higher than 96% were obtained after five cycles of co-bioevaporation.

As the organics present in the LLMC are insufficient to produce the heat necessary for the complete removal of the water itself, aeration utilizing continuous and intermittent ventilation could aid the co-bioevaporation [113]. Although continuous ventilation (CV) achieved a higher temperature than intermittent aeration [24], it was not suitable for heat preservation and energy-saving (He et al. 2018). Intermittent ventilation (IV) as an energy-efficient strategy in composting and aeration can meet the requirements of both aerobic metabolism and water removal [84], so it is a promising and favorable type of aeration for co-bioevaporation. Liu et al. [55] investigated IV for co-bioevaporation. They observed significant differences in the archaeal and fungal communities but had less effect on the bacterial community. IV reduced energy consumption in aeration and consumed a similar amount of heat for water

evaporation with almost half the ventilation time, suggesting a potentially favorable ventilation strategy in the practical application of the co-bioevaporation process [55].

Algal treatment. Microalgae can play an active role in purification processes, including removing heavy metals and nutrients and the assimilation and degradation of organic matter [92]. Microalgae increase the removal of nutrients and heavy metals and provide oxygen to aerobic heterotrophic bacteria to mineralize organic pollutants and use the carbon dioxide released by bacterial respiration [42]. Photosynthetic aeration is important for reducing operating costs and limiting the risks of pollutant volatilization under mechanical aeration. Thus, microalgae and bacteria synergistic relationships can be applied for efficient leachate treatment [42]. Several authors have examined algal species to determine their adaptability in the context of phytoremediation, selected the best candidate based on various criteria, and applied it to treat a wide range of refractory wastewater, including concentrate streams [41, 59, 97, 98, 121].

Maeng et al. [59] demonstrated that treating synthetic reverse osmosis concentrate (ROC) with *Scenedesmus quadricauda* is highly feasible to remove inorganic nutrients, reduce enteric bacteria, and degrade recalcitrant compounds. Similarly, other studies have corroborated the effectiveness of algal-mediated treatment in removing inorganic nutrients, organic substances, metal ions, and emerging contaminants from a wide range of wastewater, including real ROC [73].

Separation of biosolids is essential before the direct discharge of algae-treated ROC to meet the regulatory standard or minimize the concentration volume before disposal [106]. In particular, membrane bioreactor (MBR) is an up-and-coming technology for treating organic-rich

wastewater. It is advantageously combined with photobioreactors for microalgae cultivation and wastewater treatment [52]. The membrane photobioreactor (MPBR) removes organic carbon in the MBR in parallel to reduce nutrients in the photobioreactor. MPBRs have focused on algal biomass cultivation to primarily harvest a high density of feedstock to produce bioenergy and value-added materials [52].

However, using membranes is typically limited by fouling, which decreases permeability and, in extreme cases, can result in a loss of membrane integrity (membrane failure). Fouling mechanisms in MPBRs are similar to those observed in conventional MBRs [58]. Fouling can be caused by biological growth, deposition of inorganic precipitates or particles on or within the membrane pores, or physicochemical attachment of organic matter. It is generally agreed that the impact of algal cells on membrane fouling is less than that of algal-derived organic matter [71].

Woo et al. [106] studied membrane fouling by EfOM (wastewater effluent organic matter), commonly called EPS (extracellular polymeric substances), and SMP (soluble microbial products) derived from biological activities in wastewater treatment. Many attempts have been made to minimize fouling or maximize the restoration of membrane permeability, which include hydrophilization of the membrane surface, optimization of operating parameters, manipulation of algal biomass properties, and physicochemical cleaning of fouled membranes [52]. Feed water pretreatment before algae treatment is also applicable to improve membrane filtration performance. Woo et al. [106] utilized ozone for this purpose. Ozonation also tends to increase the hydrophilicity and biodegradability of organic matter, and only small portions of the organics can be completely mineralized [41]. Pretreatment of feedwater using oxidation techniques has

effectively reduced membrane fouling and cake resistance during algae filtration [30, 32, 118].

Woo et al. [106] observed that algae treatment as a stand-alone process alleviated fouling to some extent in microfiltration (MF), while a better performance was obtained when ozonation was combined with algae treatment. Ozone pretreatment and algae treatment using *Scenedesmus quadricauda* significantly attenuated the flux decline. It increased the reversibility of fouling in MF. This combination suggests enhanced degradation of recalcitrant organic matter and nutrient removal during algal growth under light illumination with aeration (5 and 10% CO_2). Spectroscopic diagnostics of membrane fouling indicated that increasing algae cultivation time for ROC treatment promoted deformation and/or production of undesirable organic matter to the MF, strongly associated with accelerated flux decline and deterioration of permeability recovery. The authors suggest long-term studies to determine the fouling effects of allogenic organic matter, especially the fluorescent protein fractions that may be produced in algal growth. However, it is important to note that the combination of algae-based MBR and ozonation is scarce and still in its infancy.

Integrated schemes

Due to concentrated streams' complex and recalcitrant characteristics, integrated treatments have been investigated to handle LLMC streams. Table 9.3 presents integrated systems studied in the contemporary literature.

Table 9.3 Integrated schemes for membrane concentrate management from landfill wastewater treatment plants

Integrated scheme	Main features	Main findings	References
CF + photo-Fenton + Lime treatment	Dosage of 5 g/L of ferric chloride at an initial pH = 6 $[H_2O_2]/COD$ = 1,06, pH = 4 and $[H_2O_2]/[Fe]$ Mmol = 45 An ultra violet-A light emitting diode (UVA-LED) lamp 30 g/L lime treatment	Overall, the integral treatment of LLMC achieved the removal of 99.9% color, 90% COD, 90% sulfate, 90% nitrogen, 86% Al, 77% Zn, 84% Mn, 99% Mg, and 98% Si	[93]
Fenton Oxidative coagulation + UV-Fenton	pH: 5 H_2O_2: 20 mM (680 mg/L)	For both > 90% COD and TOC removal at H_2O_2/Fe^{2+} dosages of 35 mM/8 mM and 90 mM/10 mM	[103]
Anodic oxidation + photo-Fenton	pH = 3, 16 mg/L ferric, current intensity of 0.3 A; 8 h	The removal rates of COD, NH_4^+ and total nitrogen was 60%, 33% and 22%, respectively	[44]
Coagulation using two aluminum-(alum and aluminum chlorohydrate (ACH)) and two ferric-based coagulants + UVC/H_2O_2	pH = 5 Irradiation was conducted using an annular reactor with a centrally-mounted UV lamp H_2O_2 (3 mM)	Ferric-based coagulants removed a greater proportion of most of the DOC fractions. The reduction of DOC was comparable (46–49%) for alum and ferric chloride at a similar metal dosage (1 mM) for UV fluence of 32×103 mJ/cm^2, whereas ferric sulphate performed significantly better (58%)	[98]

DOC Dissolved organic carbon; *CF* Coagulation/flocculation; *COD* Chemical oxygen demand; *TOC* Total organic carbon; NH_4^+: ammonium; *UV* ultraviolet; *UVA* Ultraviolet A; *UVC* ultraviolet C; *LED* Light emitter diode

As discussed in our previous work, ozone-based technologies (e.g., O_3, O_3/H_2O_2, O_3/UV, and O_3/H_2O_2/UV) are promising for concentrate treatment, as they can convert complex molecules into simpler and smaller ones, increasing biodegradation [26]. Ozonation is applied in current cases due to its effectiveness, small footprint, automated operation, and absence of solid waste production [34]. It could, therefore, be used downstream in an RO process to treat the membrane concentrate, reducing its polluting potential.

As for energy improvement, using landfill gas and developing support systems based on solar energy are suggested to work alongside existing technologies, promoting more economical solutions [8, 111]. In this direction, procedures based on solar energy with heat localization appear as a viable strategy to reduce the amount of membrane concentrate [8]. This practice would save energy and minimize membrane concentrate volume for ozonation.

Another possibility, after reducing the volume, is to direct the concentrate to a solidification/stabilization (S/S) process, which uses materials, promoting zero liquid emissions (Route B: MBR + RO → solar-thermal system → S/S). Solar treatment before solidification/stabilization of the concentrate would minimize the burdens of S/S technology, such as prolonged treatment times and significant volume of material for final disposal [26].

9.4 Summarising the Management Practices: Advantages and Bottlenecks

Conventional LLMC management can be grouped into two main categories: final disposal and treatment. The disposal includes the concentrate accumulation in natural pounds and subsequent recirculation to the landfill body. As discussed in previous sections, treatment options comprise technologies that aim to degrade/reduce concentrate pollutants before LLMC disposal.

Disposing LLMC streams into landfills is the most convenient, simplest, and cheapest method. The concentrate is injected into the waste mass through vertical and/or horizontal drains. Concerning the treatment of leachate concentrate streams, several technologies have been proposed and assessed in laboratory investigations; however, data on full-scale projects are scarce, and research is needed to cover this aspect. Some of the LLMC treatment procedures, advantages, and disadvantages are summarized in Table 9.4.

Table 9.4 Advantages and disadvantages of proposed treatment technologies for LLMCs

	Treatment technology	Pros	Cons
Physico-chemical	Coagulation/flocculation	Low cost	Insufficient removal efficiencies Addition of chemicals is required Sludge management is required

	Treatment technology	Pros	Cons
	Electrocoagulation	Good removal efficiencies High-tech and automated system No chemical needed	Energy input is required
	Membrane distillation	High Separation efficiency Versatility and scalability	Complex design Energy input is required Concentration of residues Limited industrial experience
	Forward osmosis	Compatibility with high wastewater salinity Reduced energy consumption Resource recovery opportunity	Draw solution management Fouling and scaling of membranes Limited commercial experience
	Adsorption	Low cost	Insufficient removal efficiencies Adsorbent regeneration is needed
	Fenton oxidation	Low cost	Addition of chemicals is required Possible change in ecotoxicity
	Photo-fenton	Concentrate biodegradability is increased	Sludge management is required Possible change in ecotoxicity
	Ozonation	Concentrate biodegradability is	High cost

	Treatment technology	Pros	Cons
		increased	Possible changes in ecotoxicity
	Solidification/stabilisation	Low cost	Time-consuming process Non-destructive technique The volume of treated concentrate is increased
Biological	Co-bioevaporation	Efficient removal of water and organics	Energy input is required Time-consuming process Gaseous emissions
	Algal treatment	High nutrients removal Requires low energy Low cost	Time-consuming process
	Incineration	Simple and commonly used method	High energy demand Equipment corrosion Flue gas treatment system and management of the residual stream are required
Thermal	Submerge combustion evaporation	High reduction of concentrate volume	High energy demand Management of the residual stream is required

Based on [12, 19, 36, 39, 40, 49, 56, 95, 106]

Physico-chemical processes are among the most investigated treatment technologies. Coagulation/flocculation, electrocoagulation, and adsorption are low-cost techniques with good removal efficiencies. However, due to their low salinity removal, these processes are primarily applied as a pretreatment step rather than a stand-alone technology. Although advanced methods such as chemical Fenton and ozonation can produce a high-quality treated effluent, they have high installation and operational expenses, restricting full-scale implementation. Biological techniques stand out in terms of simplicity and low cost. However, due to the high salinity and poor concentrate biodegradability (low BOD_5/COD ratio) (Table 9.1), they are not recommended for treating LLMCs.

9.5 Conclusions

The current article reviewed the management practices for membrane concentrate from leachate treatment facilities. The recirculation of concentrate streams into the landfill body is the most convenient and cheapest management option. However, this practice can have several negative impacts, including accumulating pollutants in treatment units and reducing leachate's treatability efficiency. Integrated treatment systems combining thermal process, ozonation, and solidification/stabilization are the most effective solutions for concentrate management. However, although these technologies can produce high-quality treated effluent, they have high installation and operational expenses, restricting full-scale implementation, especially in developing countries. Future studies on developing low-cost systems and cost–benefit analysis in pilot and full-scale are recommended.

References

1. Achilli A, Cath TY, Childress AE (2010) Selection of inorganic-based draw solutions for forward osmosis applications. J Membr Sci 364(1–2):233–241. https://doi.org/10.1016/j.memsci.2010.08.010
[Crossref]

2. Alkhudhiri A, Darwish N, Hilal N (2012) Membrane distillation: a comprehensive review. Desalination, vol 287, pp 2–18. https://doi.org/10.1016/j.desal.2011.08.027

3. Amaral-Silva N, Martins RC, Castro-Silva S, Quinta-Ferreira RM (2016) Ozonation and perozonation on the biodegradability improvement of a landfill leachate. J Environ Chem Eng 4(1):527–533. https://doi.org/10.1016/j.jece.2015.12.002
[Crossref]

4. Amor C, Torres-Socías ED, Peres JA, Maldonado MI, Oller I, Malato S, Lucas MS (2015) Mature landfill leachate treatment by coagulation/flocculation combined with Fenton and solar photo-Fenton processes. J Hazard Mater 286:261–268. https://doi.org/10.1016/j.jhazmat.2014.12.036
[Crossref]

5. Anwar N, Rahaman MS (2020) Membrane desalination processes for water recovery from pre-treated brewery wastewater: performance and fouling. Separation and Purification Technology, 252.https://doi.org/10.1016/j.seppur.2020.117420

6. Awasthi MK, Pandey AK, Bundela PS, Khan J (2015) Co-composting of organic fraction of municipal solid waste mixed with different bulking waste: characterization of physicochemical parameters and microbial enzymatic dynamic. Biores Technol 182:200–207. https://doi.org/10.1016/j.biortech.2015.01.104
[Crossref]

7. Aziz HA, Yusoff MS, Adlan MN, Adnan NH, Alias S (2004) Physico-chemical removal of iron from semi-aerobic landfill leachate by limestone filter. Waste Manage 24(4):353–358. https://doi.org/10.1016/j.wasman.2003.10.006
[Crossref]

8. Bah A, Chen Z, Bah A, Qian Q, Tuan PD, Feng D (2023) Systematic literature review of solar-powered landfill leachate sanitation: challenges and research directions over the past decade. J Environ Manage 326. https://doi.org/10.1016/j.jenvman.2022.116751

9. Calabrò PS, Gentili E, Meoni C, Orsi S, Komilis D (2018) Effect of the recirculation of a reverse osmosis concentrate on leachate generation: a case study in an Italian landfill. Waste Manage 76:643–651. https://doi.org/10.1016/j.wasman.2018.03.007 [Crossref]
10. Carvajal-Flórez E, Cardona-Gallo S-A (2019) Technologies applicable to the removal of heavy metals from landfill leachate. Environ Sci Pollut Res 26(16):15725–15753. https://doi.org/10.1007/s11356-019-04888-7
11. Cecen F, Erdincler A, Kiliç K, Kiliç K (2003) Effect of powdered activated carbon addition on sludge dewaterability and substrate removal in landfill leachate treatment. Adv Environ Res 7
12. Chen W, Gu Z, Wen P, Li Q (2019) Degradation of refractory organic contaminants in membrane concentrates from landfill leachate by a combined coagulation-ozonation process. Chemosphere 217:411–422. https://doi.org/10.1016/j.chemosphere.2018.11.002 [Crossref]
13. Chen L, Chen Z, Wang Y, Mao Y, Cai Z (2021) Effective treatment of leachate concentrate using membrane distillation coupled with electrochemical oxidation. Separat Purification Technol 267.https://doi.org/10.1016/j.seppur.2021.118679
14. Chingombe P, Saha B, Wakeman RJ (2005) Surface modification and characterisation of a coal-based activated carbon. Carbon 43(15):3132–3143. https://doi.org/10.1016/j.carbon.2005.06.021 [Crossref]
15. Choińska-Pulit A, Łaba W, Rodziewicz A (2019) Enhancement of pig bristles waste bioconversion by inoculum of keratinolytic bacteria during composting. Waste Manage 84:269–276. https://doi.org/10.1016/j.wasman.2018.11.052 [Crossref]
16. Comstock SEH, Boyer TH, Graf KC (2011) Treatment of nanofiltration and reverse osmosis concentrates: comparison of precipitative softening, coagulation, and anion exchange. Water Res 45(16):4855–4865. https://doi.org/10.1016/j.watres.2011.06.035 [Crossref]
17. Costa AM, de Alfaia RGSM, Campos JC (2019) Landfill leachate treatment in Brazil—an overview. J Environ Manage 232:110–116. https://doi.org/10.1016/j.jenvman.2018.11.006
18. Daifullah AAM, Girgis BS, Gad HMH (2004) A study of the factors affecting the removal of humic acid by activated carbon prepared from

biomass material. Colloids Surf, A 235(1–3):1–10. https://doi.org/10.1016/j.colsurfa.2003.12.020
[Crossref]

19. de Almeida R, Porto RF, Quintaes BR, Bila DM, Lavagnolo MC, Campos JC (2023) A review on membrane concentrate management from landfill leachate treatment plants: the relevance of resource recovery to close the leachate treatment loop. Waste Manage Res 41(2):264–284. https://doi.org/10.1177/0734242X221116212

20. Dialynas E, Mantzavinos D, Diamadopoulos E (2008) Advanced treatment of the reverse osmosis concentrate produced during reclamation of municipal wastewater. Water Res 42(18):4603–4608. https://doi.org/10.1016/j.watres.2008.08.008
[Crossref]

21. Džolev NM, Vujić GV (2016) Impact assessment of concentrate recirculation on the landfill gas production. Therm Sci 20(4):1283–1294. https://doi.org/10.2298/TSCI160401137D
[Crossref]

22. El-Abbassi A, Hafidi A, Khayet M, García-Payo MC (2013) Integrated direct contact membrane distillation for olive mill wastewater treatment. Desalination 323:31–38. https://doi.org/10.1016/j.desal.2012.06.014
[Crossref]

23. Emamjomeh MM, Sivakumar M (2009) Review of pollutants removed by electrocoagulation and electrocoagulation/flotation processes. J Environ Manage 90(5), 1663–1679. https://doi.org/10.1016/j.jenvman.2008.12.011

24. Febrisiantosa A, Ravindran B, Choi HL (2018) The effect of co-additives (Biochar and FGD Gypsum) on Ammonia volatilization during the composting of livestock waste. Sustainability (Switzerland), 10(3). https://doi.org/10.3390/su10030795

25. Gao M, Liang F, Yu A, Li B, Yang L (2010) Evaluation of stability and maturity during forced-aeration composting of chicken manure and sawdust at different C/N ratios. Chemosphere 78(5):614–619. https://doi.org/10.1016/j.chemosphere.2009.10.056
[Crossref]

26. Gripa E, Daflon SDA, de Almeida R, da Fonseca FV, Campos JC (2023) Landfill leachate treatment by high-pressure membranes and advanced oxidation techniques with a focus on ecotoxicity and by-products management: a review. Proc Safety Environ Prot 173:747–764. Institution of Chemical Engineers. https://doi.org/10.1016/j.psep.2023.03.074

27. Guo H, Yao J, Cai M, Qian Y, Guo Y, Richnow HH, Blake RE, Doni S, Ceccanti B (2012) Effects of petroleum contamination on soil microbial numbers, metabolic activity and urease activity. Chemosphere 87(11):1273–1280. https://doi.org/10.1016/j.chemosphere.2012.01.034 [Crossref]
28. He R, Tian BH, Zhang QQ, Zhang HT (2015) Effect of Fenton oxidation on biodegradability, biotoxicity and dissolved organic matter distribution of concentrated landfill leachate derived from a membrane process. Waste Manage 38(1):232–239. https://doi.org/10.1016/j.wasman.2015.01.006 [Crossref]
29. Hendrych J, Hejralová R, Kroužek J, Špaček P, Sobek J (2019) Stabilisation/solidification of landfill leachate concentrate and its residue obtained by partial evaporation. Waste Manage 95:560–568. https://doi.org/10.1016/j.wasman.2019.06.046 [Crossref]
30. Heng L, Yanling Y, Weijia G, Xing L, Guibai L (2008) Effect of pretreatment by permanganate/chlorine on algae fouling control for ultrafiltration (UF) membrane system. Desalination 222(1–3):74–80. https://doi.org/10.1016/j.desal.2007.01.126 [Crossref]
31. Hunce SY, Akgul D, Demir G, Mertoglu B (2012) Solidification/stabilization of landfill leachate concentrate using different aggregate materials. Waste Manage 32(7):1394–1400. https://doi.org/10.1016/j.wasman.2012.03.010 [Crossref]
32. Hung MT, Liu JC (2006) Microfiltration for separation of green algae from water. Colloids Surf, B 51(2):157–164. https://doi.org/10.1016/j.colsurfb.2006.07.003 [Crossref]
33. Ilhan F, Kurt U, Apaydin O, Gonullu MT (2008) Treatment of leachate by electrocoagulation using aluminum and iron electrodes. J Hazard Mater 154(1–3):381–389. https://doi.org/10.1016/j.jhazmat.2007.10.035 [Crossref]
34. Issaka E, AMU-Darko JNO, Yakubu S, Fapohunda FO, Ali N, Bilal M (2022) Advanced catalytic ozonation for degradation of pharmaceutical pollutants—a review. Chemosphere 289. Elsevier Ltd. https://doi.org/10.1016/j.chemosphere.2021.133208
35. Kalderis D, Koutoulakis D, Paraskeva P, Diamadopoulos E, Otal E, del Valle JO, Fernández-Pereira C (2008) Adsorption of polluting substances

on activated carbons prepared from rice husk and sugarcane bagasse. Chem Eng J 144(1):42–50. https://doi.org/10.1016/j.cej.2008.01.007 [Crossref]

36. Kallel A, Ellouze M, Trabelsi I (2017) Co-management of landfill leachate concentrate with brick waste by solidification/stabilization treatment. Arab J Geosc 10(4). https://doi.org/10.1007/s12517-017-2871-x

37. Kamranvand F, Davey CJ, Williams L, Parker A, Jiang Y, Tyrrel S, McAdam EJ (2020) Ultrafiltration pretreatment enhances membrane distillation flux, resilience and permeate quality during water recovery from concentrated blackwater (urine/faeces). Separat Purification Technol 253. https://doi.org/10.1016/j.seppur.2020.117547

38. El Kateb M, Trellu C, Darwich A, Rivallin M, Bechelany M, Nagarajan S, Lacour S, Bellakhal N, Lesage G, Héran M, Cretin M (2019) Electrochemical advanced oxidation processes using novel electrode materials for mineralization and biodegradability enhancement of nanofiltration concentrate of landfill leachates. Water Res 162:446–455. https://doi.org/10.1016/j.watres.2019.07.005 [Crossref]

39. Keyikoglu R, Karatas O, Rezania H, Kobya M, Vatanpour V, Khataee A (2021) A review on treatment of membrane concentrates generated from landfill leachate treatment processes. Separat Purif Technol 259. https://doi.org/10.1016/j.seppur.2020.118182

40. Khan D, Kumar A, Samadder SR (2016) Impact of socioeconomic status on municipal solid waste generation rate. Waste Manage 49:15–25. https://doi.org/10.1016/j.wasman.2016.01.019 [Crossref]

41. Kim HC, Choi WJ, Maeng SK, Kim HJ, Kim HS, Song KG (2014) Ozonation of piggery wastewater for enhanced removal of contaminants by S. quadricauda and the impact on organic characteristics. Biores Technol 159:128–135. https://doi.org/10.1016/j.biortech.2014.02.061 [Crossref]

42. Kumari M, Ghosh P, Thakur IS (2016) Landfill leachate treatment using bacto-algal co-culture: An integrated approach using chemical analyses and toxicological assessment. Ecotoxicol Environ Saf 128:44–51. https://doi.org/10.1016/j.ecoenv.2016.02.009 [Crossref]

43. Kurniawan TA, Lo W, Chan G, Sillanpää MET (2010) Biological processes for treatment of landfill leachate. J Environ Monit 12(11), 2032–2047. https://doi.org/10.1039/c0em00076k

44.
Labiadh L, Fernandes A, Ciríaco L, Pacheco MJ, Gadri A, Ammar S, Lopes A (2016) Electrochemical treatment of concentrate from reverse osmosis of sanitary landfill leachate. J Environ Manage 181:515–521. https://doi.org/10.1016/j.jenvman.2016.06.069
[Crossref]
45.
Lage PL da C, Campos FB (2004) Advances in direct contact evaporator design. Chem Eng Technol 27(1):91–96. https://doi.org/10.1002/ceat.200401760
46.
Lee WJ, Ng ZC, Hubadillah SK, Goh PS, Lau WJ, Othman MHD, Ismail AF, Hilal N (2020) Fouling mitigation in forward osmosis and membrane distillation for desalination. Desalination 480. https://doi.org/10.1016/j.desal.2020.114338
47.
Li J, Niu A, Lu CJ, Zhang JH, Junaid M, Strauss PR, Xiao P, Wang X, Ren YW, Pei DS (2017) A novel forward osmosis system in landfill leachate treatment for removing polycyclic aromatic hydrocarbons and for direct fertigation. Chemosphere 168:112–121. https://doi.org/10.1016/j.chemosphere.2016.10.048
[Crossref]
48.
Li J, Zhao L, Qin L, Tian X, Wang A, Zhou Y, Meng L, Chen Y (2016) Removal of refractory organics in nanofiltration concentrates of municipal solid waste leachate treatment plants by combined Fenton oxidative-coagulation with photo—fenton processes. Chemosphere 146:442–449. https://doi.org/10.1016/j.chemosphere.2015.12.069
[Crossref]
49.
Li H, Liu H (2021) Treatment and recovery methods for leachate concentrate from landfill and incineration: a state-of-the-art review. J Cleaner Prod 329. https://doi.org/10.1016/j.jclepro.2021.129720
50.
Li Q, Cui H, Li Y, Song X, Liu W, Wang Y, Hou H, Zhang H, Li Y, Wang F, Song J, Ye H, Song S, Che T, Shao S, Kong D, Liang B (2023) Challenges and engineering application of landfill leachate concentrate treatment. Environ Res 231. https://doi.org/10.1016/j.envres.2023.116028
51.
Li W, Zhang Bo L Peng Hui J, Li N, Zhu X, Yun (2008) Preparation of high surface area activated carbons from tobacco stems with K_2CO_3 activation using microwave radiation. Ind Crops Prod 27(3):341–347.https://doi.org/10.1016/j.indcrop.2007.11.011
52.
Liao Y, Bokhary A, Maleki E, Liao B (2018) A review of membrane fouling and its control in algal-related membrane processes. Bioresource Technol 264:343–358. https://doi.org/10.1016/j.biortech.2018.06.102

53. Lima VN, Rodrigues CSD, Madeira LM (2020) Simultaneous treatment of toluene-containing gas waste and industrial wastewater by the Fenton process. Sci Total Environ 749.https://doi.org/10.1016/j.scitotenv.2020.141497
54. Liu C, Chen L, Zhu L (2018) Fouling mechanism of hydrophobic polytetrafluoroethylene (PTFE) membrane by differently charged organics during direct contact membrane distillation (DCMD) process: an especial interest in the feed properties. J Membr Sci 548:125–135. https://doi.org/10.1016/j.memsci.2017.11.011
[Crossref]
55. Liu Y, Wang Q, Pan Q, Zhou X, Peng Z, Jahng D, Yang B, Pan X (2022a) Ventilation induced evolution pattern of archaea, fungi, bacteria and their potential roles during co-bioevaporation treatment of concentrated landfill leachate and food waste. Chemosphere 289. https://doi.org/10.1016/j.chemosphere.2021.133122
56. Liu Y, Wang Q, Zhou X, Pan Q, Zhu H, Yang B, Pan X (2021) Evaporation efficiency and important microorganisms under different ventilation strategies of co-bioevaporation process. Environ Technol Innovat 21. https://doi.org/10.1016/j.eti.2021.101374
57. Long Y, Xu J, Shen D, Du Y, Feng H (2017) Effective removal of contaminants in landfill leachate membrane concentrates by coagulation. Chemosphere 167:512–519. https://doi.org/10.1016/j.chemosphere.2016.10.016
[Crossref]
58. Low SL, Ong SL, Ng HY (2016) Characterization of membrane fouling in submerged ceramic membrane photobioreactors fed with effluent from membrane bioreactors. Chem Eng J 290:91–102. https://doi.org/10.1016/j.cej.2016.01.005
[Crossref]
59. Maeng SK, Khan W, Park JW, Han I, Yang HS, Song KG, Choi WJ, Kim S, Woo H, Kim HC (2018) Treatment of highly saline RO concentrate using Scenedesmus quadricauda for enhanced removal of refractory organic matter. Desalination 430:128–135. https://doi.org/10.1016/j.desal.2017.12.056
[Crossref]
60. Mahmoudkhani R, Hassani AH, Torabian A, Borghei SM (2012) Study on high-strength anaerobic landfill leachate treatability by membrane bioreactor coupled with Reverse Osmosis. Int J Environ Res 6(1):129–138. www.SID.ir

61. Meng G, Wang Y, Li X, Zhang H, Zhou X, Bai Z, Wu L, Bai J (2020) Treatment of landfill leachate evaporation concentrate by a modified electro-Fenton method. Environ Technol (United Kingdom), 1–36. https://doi.org/10.1080/09593330.2020.1795931
62. Mojiri A, Ziyang L, Hui W, Ahmad Z, Tajuddin RM, Abu Amr SS, Kindaichi T, Aziz HA, Farraji H (2017) Concentrated landfill leachate treatment with a combined system including electro-ozonation and composite adsorbent augmented sequencing batch reactor process. Process Saf Environ Prot 111:253–262. https://doi.org/10.1016/j.psep.2017.07.013
[Crossref]
63. Morello L, Cossu R, Raga R, Pivato A, Lavagnolo MC (2016) Recirculation of reverse osmosis concentrate in lab-scale anaerobic and aerobic landfill simulation reactors. Waste Manage 56:262–270. https://doi.org/10.1016/j.wasman.2016.07.030
[Crossref]
64. Nazdar N, Imani A, Noori F, Sarvi Moghanlou K (2018) Effect of silymarin supplementation on nickel oxide nanoparticle toxicity to rainbow trout (Oncorhynchus mykiss) fingerlings: pancreas tissue histopathology and alkaline protease activity. Iranian J Sci Technol, Trans A: Sci 42(2):353–361. https://doi.org/10.1007/s40995-016-0052-5
[Crossref]
65. Neyens E, Baeyens J (2003) A review of classic Fenton's peroxidation as an advanced oxidation technique. J Hazardous Mater 98
66. Özgür Akta FÇ (2001) Addition of activated carbon to batch activated sludge reactors in the treatment of landfill leachate and domestic wastewater. J Chem Technol Biotechnol 76(8):793–802. https://doi.org/10.1002/jctb.450
67. Panagopoulos A (2021) Beneficiation of saline effluents from seawater desalination plants: fostering the zero liquid discharge (ZLD) approach—a techno-economic evaluation. J Environ Chem Eng 9(4). https://doi.org/10.1016/j.jece.2021.105338
68. Primo O, Rivero MJ, Ortiz I (2008) Photo-fenton process as an efficient alternative to the treatment of landfill leachates. J Hazard Mater 153(1–2):834–842. https://doi.org/10.1016/j.jhazmat.2007.09.053
[Crossref]
69. Pérez G, Fernández-Alba AR, Urtiaga AM, Ortiz I (2010) Electro-oxidation of reverse osmosis concentrates generated in tertiary water treatment. Water Res 44(9):2763–2772. https://doi.org/10.1016/j.watres.2010.02.017
[Crossref]

70. Qin M, Molitor H, Brazil B, Novak JT, He Z (2016) Recovery of nitrogen and water from landfill leachate by a microbial electrolysis cell-forward osmosis system. Biores Technol 200:485–492. https://doi.org/10.1016/j.biortech.2015.10.066
[Crossref]

71. Qu F, Liang H, Tian J, Yu H, Chen Z, Li G (2012) Ultrafiltration (UF) membrane fouling caused by cyanobateria: fouling effects of cells and extracellular organics matter (EOM). Desalination 293:30–37. https://doi.org/10.1016/j.desal.2012.02.020
[Crossref]

72. Ramakrishnan A, Blaney L, Kao J, Tyagi RD, Zhang TC, Surampalli RY (2015) Emerging contaminants in landfill leachate and their sustainable management. Environ Earth Sci 73(3):1357–1368. https://doi.org/10.1007/s12665-014-3489-x
[Crossref]

73. Rawat I, Gupta SK, Shriwastav A, Singh P, Kumari S, Bux F (2016) Microalgae applications in wastewater treatment, pp 249–268. https://doi.org/10.1007/978-3-319-12334-9_13

74. Razaqpur AG, Wang Y, Liao X, Liao Y, Wang R (2021) Progress of photothermal membrane distillation for decentralized desalination: a review. Water Res 201. https://doi.org/10.1016/j.watres.2021.117299

75. Renou S, Givaudan JG, Poulain S, Dirassouyan F, Moulin P (2008) Landfill leachate treatment: Review and opportunity. J Hazardous Mater 150(3):468–493. https://doi.org/10.1016/j.jhazmat.2007.09.077

76. Rivas FJ, Beltrán FJ, Gimeno O, Frades J, Carvalho F (2006). Adsorption of landfill leachates onto activated carbon. Equilibrium and kinetics. J Hazardous Mater 131(1–3):170–178. https://doi.org/10.1016/j.jhazmat.2005.09.022

77. Rizzo L, Gernjak W, Krzeminski P, Malato S, McArdell CS, Perez JAS, Schaar H, Fatta-Kassinos D (2020) Best available technologies and treatment trains to address current challenges in urban wastewater reuse for irrigation of crops in EU countries. In *Science of the Total Environment* (Vol. 710). Elsevier B.V. https://doi.org/10.1016/j.scitotenv.2019.136312

78. Robinson, A. H. (2005). *Landfill leachate treatment*.

79. Rosenfeldt EJ, Linden KG, Canonica S, von Gunten U (2006) Comparison of the efficiency of •OH radical formation during ozonation and the advanced oxidation processes O_3/H_2O_2 and UV/H_2O_2. Water Res 40(20):3695–3704. https://doi.org/10.1016/j.watres.2006.09.008

[Crossref]

80. Ruiz-Sarmiento JR, Monroy J, Moreno FA, Galindo C, Bonelo JM, Gonzalez-Jimenez J (2020) A predictive model for the maintenance of industrial machinery in the context of industry 4.0. Eng Appl Artif Intell 87. https://doi.org/10.1016/j.engappai.2019.103289

81. Santos MSF, Alves A, Madeira LM (2011) Paraquat removal from water by oxidation with Fenton's reagent. Chem Eng J 175(1):279–290. https://doi.org/10.1016/j.cej.2011.09.106
[Crossref]

82. Shaffer DL, Werber JR, Jaramillo H, Lin S, Elimelech M (2015) Forward osmosis: where are we now? Desalination 356:271–284. https://doi.org/10.1016/j.desal.2014.10.031

83. Shah TM, Ramaswami S, Behrendt J, Otterpohl R (2017) Simultaneous removal of organics and ammonium-nitrogen from reverse osmosis concentrate of mature landfill leachate. J Water Proc Eng 19:126–132. https://doi.org/10.1016/j.jwpe.2017.07.024
[Crossref]

84. Shao LM, He X, Yang N, Fang JJ, Lü F, He PJ (2012) Biodrying of municipal solid waste under different ventilation modes: drying efficiency and aqueous pollution. Waste Manage Res 30(12):1272–1280. https://doi.org/10.1177/0734242X12462278
[Crossref]

85. Silva TLS, Morales-Torres S, Esteves CMP, Ribeiro AR, Nunes OC, Figueiredo JL, Silva AMT (2018) Desalination and removal of organic micropollutants and microorganisms by membrane distillation. Desalination 437:121–132. https://doi.org/10.1016/j.desal.2018.02.027
[Crossref]

86. Solo-Gabriele HM, Jones AS, Lindstrom AB, Lang JR (2020) Waste type, incineration, and aeration are associated with per- and polyfluoroalkyl levels in landfill leachates. Waste Manage 107:191–200. https://doi.org/10.1016/j.wasman.2020.03.034
[Crossref]

87. Song X, Luo W, McDonald J, Khan SJ, Hai FI, Price WE, Nghiem LD (2018) An anaerobic membrane bioreactor—membrane distillation hybrid system for energy recovery and water reuse: Removal performance of organic carbon, nutrients, and trace organic contaminants. Sci Total Environ 628–629:358–365. https://doi.org/10.1016/j.scitotenv.2018.02.057
[Crossref]

88. Stegmann R, Heyer K-U, Cossu R (2005) Leachate Treatment. Proc Sardinia

89. Sui Q, Zhao W, Cao X, Lu S, Qiu Z, Gu X, Yu G (2017) Pharmaceuticals and personal care products in the leachates from a typical landfill reservoir of municipal solid waste in Shanghai, China: Occurrence and removal by a full-scale membrane bioreactor. J Hazard Mater 323:99–108. https://doi.org/10.1016/j.jhazmat.2016.03.047 [Crossref]
90. Talalaj IA, Biedka P (2015) Impact of concentrated leachate recirculation on effectiveness of leachate treatment by reverse osmosis. Ecol Eng 85:185–192. https://doi.org/10.1016/j.ecoleng.2015.10.002 [Crossref]
91. Tan YZ, Chew JW, Krantz WB (2016) Effect of humic-acid fouling on membrane distillation. J Membr Sci 504:263–273. https://doi.org/10.1016/j.memsci.2015.12.051 [Crossref]
92. Tchobanoglous G, Burton FL, David Stensel H (2003) Wastewater engineering treatment and reuse, 4th edn
93. Tejera J, Hermosilla D, Miranda R, Gascó A, Alonso V, Negro C, Blanco Á (2020) Assessing an integral treatment for landfill leachate reverse osmosis concentrate. Catalysts 10(12):1–17. https://doi.org/10.3390/catal10121389 [Crossref]
94. Tomasini HR, Hacifazlioglu MC, Kabay N, Bertin L, Pek TO, Yuksel M (2019) Concentrate management for integrated MBR-RO process for wastewater reclamation and reuse-preliminary tests. J Water Proc Eng 29. https://doi.org/10.1016/j.jwpe.2017.07.020
95. Top S, Sekman E, Hoşver S, Bilgili MS (2011) Characterization and electrocaogulative treatment of nanofiltration concentrate of a full-scale landfill leachate treatment plant. Desalination 268(1–3):158–162. https://doi.org/10.1016/j.desal.2010.10.012 [Crossref]
96. Tow EW, Ersan MS, Kum S, Lee T, Speth TF, Owen C, Bellona C, Nadagouda MN, Mikelonis AM, Westerhoff P, Mysore C, Frenkel VS, Desilva V, Walker WS, Safulko A K, Ladner DA (2021) Managing and treating per-and polyfluoroalkyl substances (Pfas) in membrane concentrates. AWWA Water Sci 3(5). Inc. https://doi.org/10.1002/aws2.1233
97. Umar M, Roddick F, Fan L (2016a) Comparison of coagulation efficiency of aluminium and ferric-based coagulants as pretreatment for UVC/H2O2 treatment of wastewater RO concentrate. Chem Eng J 284:841–849. https://doi.org/10.1016/j.cej.2015.08.109

98. Umar M, Roddick F, Fan L (2016b) Impact of coagulation as a pretreatment for UVC/H_2O_2-biological activated carbon treatment of a municipal wastewater reverse osmosis concentrate. Water Res 88:12–19. https://doi.org/10.1016/j.watres.2015.09.047
99. Von Gunten U (2003) Ozonation of drinking water: Part I. Oxidation kinetics and product formation. Water Res 37
100. Voutchkov N, Kaiser GN (2020) Zero-liquid discharge concentrate disposal systems. In: Management of concentrate from desalination plants (pp 187–208). Elsevier. https://doi.org/10.1016/B978-0-12-818045-7.00009-9
101. Wang H, Wang Y, Lou Z, Zhu N, Yuan H (2017) The degradation processes of refractory substances in nanofiltration concentrated leachate using micro-ozonation. Waste Manage 69:274–280. https://doi.org/10.1016/j.wasman.2017.08.048 [Crossref]
102. Wang G, Lu G, Yin P, Zhao L, Jimmy Yu Q (2016) Genotoxicity assessment of membrane concentrates of landfill leachate treated with Fenton reagent and UV-Fenton reagent using human hepatoma cell line. J Hazardous Mater 307:154–162. https://doi.org/10.1016/j.jhazmat.2015.12.069
103. Wang H, Wang YN, Li X, Sun Y, Wu H, Chen D (2016a) Removal of humic substances from reverse osmosis (RO) and nanofiltration (NF) concentrated leachate using continuously ozone generation-reaction treatment equipment. Waste Manage 56:271–279https://doi.org/10.1016/j.wasman.2016.07.040
104. Wei K, Bao H, Huang S, Chen L (2017) Effects of long-term fertilization on available P, P composition and phosphatase activities in soil from the Huang-Huai-Hai Plain of China. Agr Ecosyst Environ 237:134–142. https://doi.org/10.1016/j.agee.2016.12.030 [Crossref]
105. Woo YC, Tijing LD, Shim WG, Choi JS, Kim SH, He T, Drioli E, Shon HK (2016) W5ater desalination using graphene-enhanced electrospun nanofiber membrane via air gap membrane distillation. J Membr Sci 520:99–110. https://doi.org/10.1016/j.memsci.2016.07.049 [Crossref]
106. Woo H, Yang HS, Timmes TC, Han C, Nam JY, Byun S, Kim S, Ryu H, Kim HC (2019) Treatment of reverse osmosis concentrate using an algal-based MBR combined with ozone pretreatment. Water Res 159:164–175. https://doi.org/10.1016/j.watres.2019.05.003 [Crossref]

107. Wu S, Zou S, Liang G, Qian G, He Z (2018) Enhancing recovery of magnesium as struvite from landfill leachate by pretreatment of calcium with simultaneous reduction of liquid volume via forward osmosis. Sci Total Environ 610–611:137–146. https://doi.org/10.1016/j.scitotenv.2017.08.038
[Crossref]
108. Wu Y, Tao Y, Deng Z, Zhou J, Xu C, Zhang B (2020) A fuzzy analysis framework for waste incineration power plant comprehensive benefit evaluation from refuse classification perspective. J Cleaner Prod 258. https://doi.org/10.1016/j.jclepro.2020.120734
109. Xu J, Wu S, Liu M, Wang F, Liu C, Wu C, Sun Q, Zheng Y (2023) The adsorption mechanism of sludge-based biochar toward highly concentrated organic membrane concentrates from landfill leachate. Environ Sci Pollut Res 30(18):54149–54159. https://doi.org/10.1007/s11356-023-26069-3
[Crossref]
110. Xu YD, Yue DB, Zhu Y, Nie YF (2006) Fractionation of dissolved organic matter in mature landfill leachate and its recycling by ultrafiltration and evaporation combined processes. Chemosphere 64(6):903–911. https://doi.org/10.1016/j.chemosphere.2006.01.039
[Crossref]
111. Yaashikaa PR, Kumar PS, Nhung TC, Hemavathy RV, Jawahar MJ, Neshaanthini JP, Rangasamy G (2022) A review on landfill system for municipal solid wastes: Insight into leachate, gas emissions, environmental and economic analysis. Chemosphere 309. https://doi.org/10.1016/j.chemosphere.2022.136627
112. Yang B, Li X, Lin Z, Hu D, Liu Y, Pan X (2020) Evolution of enzyme activity, heavy metals bioavailability and microbial community in different temperature stages of the co-bioevaporation process. Waste Manage 102:751–762. https://doi.org/10.1016/j.wasman.2019.11.044
[Crossref]
113. Yang F, Li GX, Yang QY, Luo WH (2013) Effect of bulking agents on maturity and gaseous emissions during kitchen waste composting. Chemosphere 93(7):1393–1399. https://doi.org/10.1016/j.chemosphere.2013.07.002
[Crossref]
114. Yang BQ, Yang JM, Yang H, Liu YM, Li XK, Wang QZ, Pan XJ (2018) Co-bioevaporation treatment of concentrated landfill leachate with addition of food waste. Biochem Eng J 130:76–82. https://doi.org/10.1016/j.bej.2017.11.010
[Crossref]

115. Ye W, Liu H, Jiang M, Lin J, Ye K, Fang S, Xu Y, Zhao S, Van der Bruggen B, He Z (2019) Sustainable management of landfill leachate concentrate through recovering humic substance as liquid fertilizer by loose nanofiltration. Water Res 157:555–563. https://doi.org/10.1016/j.watres.2019.02.060
[Crossref]
116. Yuan J, Zhang D, Li Y, Chadwick D, Li G, Li Y, Du L (2017) Effects of adding bulking agents on biostabilization and drying of municipal solid waste. Waste Manage 62:52–60. https://doi.org/10.1016/j.wasman.2017.02.027
[Crossref]
117. Yue D, Xu Y, Mahar RB, Liu F, Nie Y (2007) Laboratory-scale experiments applied to the design of a two-stage submerged combustion evaporation system. Waste Manage 27(5):704–710. https://doi.org/10.1016/j.wasman.2006.04.017
[Crossref]
118. Zhang X, Fan L, Roddick FA (2015) Effect of feedwater pretreatment using UV/H_2O_2 for mitigating the fouling of a ceramic MF membrane caused by soluble algal organic matter. J Membr Sci 493:683–689. https://doi.org/10.1016/j.memsci.2015.07.024
[Crossref]
119. Zhang QQ, Tian BH, Zhang X, Ghulam A, Fang CR, He R (2013) Investigation on characteristics of leachate and concentrated leachate in three landfill leachate treatment plants. Waste Manage 33(11):2277–2286. https://doi.org/10.1016/j.wasman.2013.07.021
[Crossref]
120. Zhang L, Wang X, Yue D (2020) Effect of submerged combustion evaporation on Cd complexation potential of organic matter in municipal solid waste landfill leachate. Environ Pollut 267.https://doi.org/10.1016/j.envpol.2020.115573
121. Zhou T, Lim TT, Chin SS, Fane AG (2011) Treatment of organics in reverse osmosis concentrate from a municipal wastewater reclamation plant: Feasibility test of advanced oxidation processes with/without pretreatment. Chem Eng J 166(3):932–939. https://doi.org/10.1016/j.cej.2010.11.078

© The Author(s), under exclusive license to Springer Nature Switzerland AG 2024
S. Souabi, A. Anouzla (eds.), *Landfill Leachate Treatment Techniques*, Springer Water
https://doi.org/10.1007/978-3-031-63157-3_10

10. Advanced Oxidative Processes as Treatment for Landfill Leachate

Siara Silvestri[1, 2] ✉, Mariana Islongo Canabarro[1], Lizet Mirian Guevara Roman[1] and Elvis Carissimi[1]
(1) Universidade Federal de Santa Maria, Santa Maria, Brazil
(2) Universidade Federal de Pelotas, Pelotas, Brazil

✉ **Siara Silvestri**
Email: siarasilvestri@gmail.com

Abstract
Sanitary landfill leachate is an effluent characterized by great heterogeneity in its composition and may contain the most diverse types of organic and inorganic pollutants. Therefore, leachate needs efficient treatment before being released into the environment. Sanitary landfill leachate represents a high environmental risk and may result in the contamination of groundwater and/or surface water and the eutrophication of water bodies, if disposed of improperly, that is, without treatment. A study carried out at a simple landfill in southeastern China detected more than 50 types of highly hazardous pollutants, while a similar study carried out in southern Brazil detected more than 70 polluting compounds. With the development of technology, several treatment techniques have emerged with the aim of improving treatment efficiency and adapting the leachate to standard release regulations for later disposal. Knowing that the polluting compounds present in the leachate are recalcitrant and that advanced oxidative processes are a set of highly recommended procedures for chemical treatment aimed at degrading pollutants in water and effluents via oxidation through radical reactions, this chapter concentrates information on the application of advanced oxidative processes used as a treatment option for leachate.

10.1 Introduction

Sanitary landfill leachate is a liquid waste with different compositions and high levels of pollutants: degradable organic matter, inorganic macrocomponents, heavy metals and xenobiotic organic compounds; that are generated by the decomposition of urban solid waste discarded in sanitary landfills. Different landfill leachate treatment processes such as biological and combined (batch sequencing reactor; membrane bioreactor; constructed wetland system; anaerobic/nitrification process; nitrification/anaerobic/ozonation; and oxidation trench/granular activated carbon) are applied all around the world. However, the AOP has advantages over other treatments because it is capable of generating oxidative radicals that are strongly responsible for the degradation of many pollutants present in the leachate.

Data from 2010 from Eurostat, the EU statistics group, indicate that in Europe 38% of waste ended up in landfills. In Bulgaria, for example, all rubbish ends up in landfills. In Romania, Lithuania and Latvia, this rate is higher than 90%. In the United States, for example, landfills currently account for 53% of all public waste sent for treatment and final disposal, while in Brazil this percentage rises to 58% (Fig. 10.1).

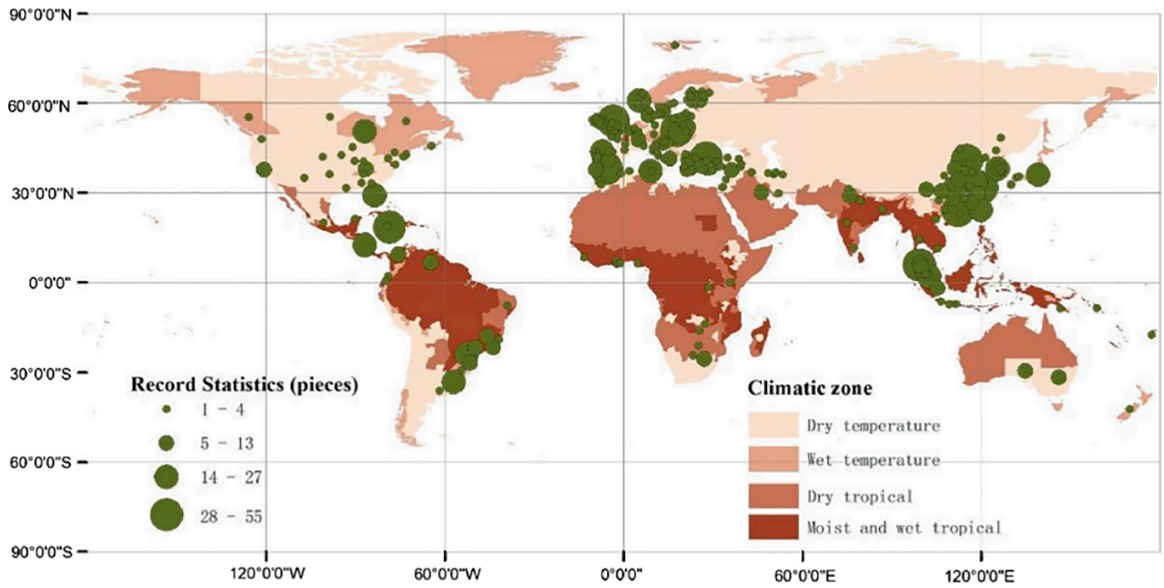


Fig. 10.1 Global distribution of the numbers of landfill leachate pollutant records from collected literature (reproduced with permission from Ma et al. [61])

10.2 Advanced Oxidative Processes

Advanced oxidative processes (AOPs) were initially proposed to treat drinking water [33]. AOPs generate and use mainly hydroxyl radicals

(·OH) to oxidize organic compounds. The main characteristics of hydroxyl radicals are their short life span, simple form of production, high oxidation power, high reactivity, and non-selectivity.

Research on AOPs is focused on the reduction of recalcitrant compounds and toxic elements present in leachate, and further research will increase the efficiency of existing AOPs and the development of new AOPs [31].

Conventionally used AOPs are classified as homogeneous and heterogeneous processes, depending on whether they occur in a single-phase or use heterogeneous catalysts such as carbon materials, metal-supported catalysts, or semiconductors such as titanium, zinc, and tungsten oxides [10]. Some examples of homogeneous processes are ozonation (O_3), O_3 and H_2O_2, hydrogen peroxide (H_2O_2, oxidation, photo-Fenton, and others; while heterogeneous processes involve catalytic oxidation of H_2O_2, catalytic ozonation, and others [31].

One of the main challenges in the treatment of landfill leachate is the highly bio-recalcitrant nature of organic matter (generally dissolved matter, BOD/ COD ratio < 0.1), making treatment by conventional biological and physicochemical processes difficult. Ozone (O_3) is a strong oxidizing agent ($E° = 2.07$ V), particularly effective in removing colors, and has high reactivity and selectivity toward organic pollutants such as humic substances [36].

In recent decades, advanced oxidative processes (AOPs) have been considered an attractive means for color removal, organic load reduction, and improved biodegradability of recalcitrant contaminants from old leachates. These processes involve the production of free radical species, mainly the hydroxyl radical (·OH). The hydroxyl radical is produced from single oxidants, such as ozone (O_3), or a combination of strong oxidants, such as O_3 and hydroxide (OH^-), O_3 and hydrogen peroxide (H_2O_2), or ferrous ions with H_2O_2 [78, 80], entre outros.

10.2.1 Fenton Process

The Fenton process consists in the generation of hydroxyl radicals (·OH) through the activation of hydrogen peroxide (H_2O_2) in the presence of Fe_2^+ ions in an acidic medium [30]. In the Fenton process, hydrogen peroxide is added to the wastewater in the presence of ferrous salts, generating species that are strongly oxidative toward organic compounds. The ·OH radicals are traditionally considered to be the main oxidizing species in Fenton processes. The classical

Fenton free radical mechanism in the absence of organic compounds mainly involves the sequence of reactions below (Eqs. 10.1–10.7) [22]:

$$Fe^{2+} + H_2O_2 \rightarrow Fe^{3+} + \cdot OH + OH^- \quad (10.1)$$

$$Fe^{2+} + H_2O_2 \rightarrow Fe^{2+} + HO_2^{\cdot} + H^+ \quad (10.2)$$

$$\cdot OH + H_2O_2 \rightarrow HO_2^{\cdot} + H_2O \quad (10.3)$$

$$\cdot OH + Fe^{2+} \rightarrow Fe^{2+}OH^- \quad (10.4)$$

$$Fe^{3+} + HO_2^{\cdot} \rightarrow Fe^{2+} + O_2H^+ \quad (10.5)$$

$$Fe^{2+} + HO_2^{\cdot} + H^+ \rightarrow Fe^{3+} + H_2O_2 \quad (10.6)$$

$$2HO_2^{\cdot} \rightarrow H_2O_2 + O_2 \quad (10.7)$$

In general, the Fenton process is composed of four stages: (i) pH adjustment, (ii) oxidation reaction, (iii) neutralization and coagulation, and (iv) precipitation [22]. Organic substances are removed in the oxidation and coagulation stages [50]. Chemical coagulation predominates at lower H_2O_2/Fe^{2+} ratios, while chemical oxidation predominates at higher ratios [72]. Figure 10.2 presents a schematic of the batch Fenton experiment.

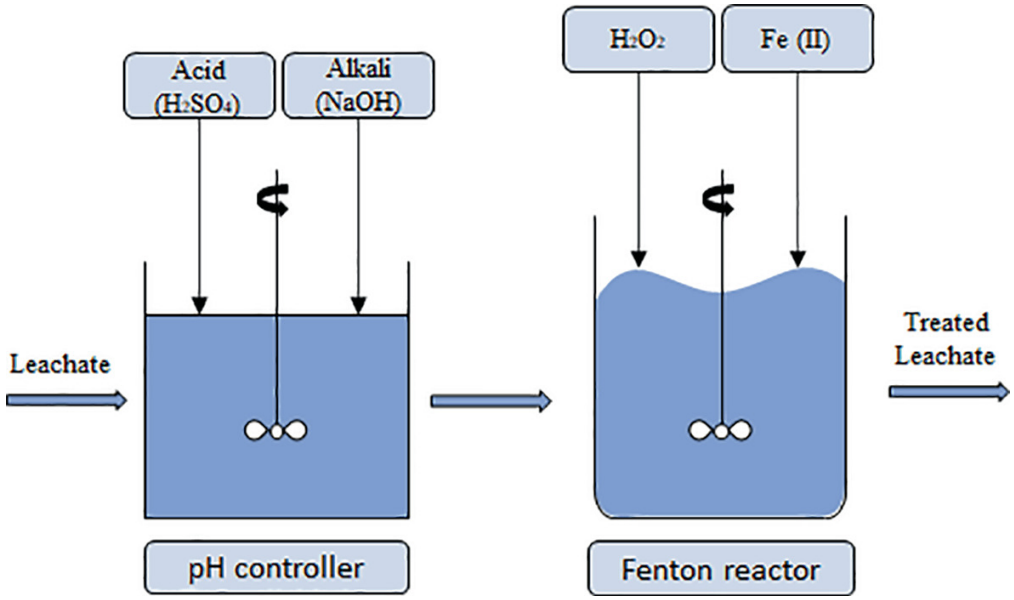


Fig. 10.2 Schematic representation of the batch Fenton process. Adapted from Zhang et al. [106]

The treatment efficiency of the Fenton process depends on the pH and the dosage of H_2O_2. This reagent dosage is the main operational cost item for wastewater treatment. The removal of organic contaminants is enhanced as the concentration of Fe^{2+} and H_2O_2 increases. However, an excessive increase becomes insignificant once a certain limit is exceeded. According to some researchers, the optimal molar ratio of H_2O_2 to Fe^{2+} is 1.5–3.0 [41, 104, 106]. Regarding pH, it is a very important parameter because it controls the production of hydroxyl radicals and the concentration of iron ions.

The Fenton process has been widely studied and applied because it is one of the best cost-effective treatment alternatives. Besides this advantage, the activation of hydrogen peroxide does not require energy, it is an easy-to-operate system and requires less maintenance, the reactants (iron and hydrogen peroxide) are cheap and non-toxic, it does not involve mass transfer, and the reaction time is short. However, some disadvantages are the narrow operating range for pH (between 2 and 3) and the need for a large number of chemicals and labor to treat the sludge generated at the end of the treatment [31, 60].

Roudi et al. [81] obtained maximum COD removal (94.41%) using pH 3, $[Fe^{+2}] = 1000$ mg L^{-1}, $[H_2O_2] = 6000$ mg L^{-1}, ratio H_2O_2:Fe^{2+} = 6 and a reaction time of 32.5 min. These removal rates decreased as the pH was increased. Thus, they were able to conclude that the pH value should be in the acidic range to generate the greatest amount of hydroxyl radicals and to obtain maximum oxidation of organic compounds. While Singa et al. [88, 89] achieved maximum COD removal of 56.49% under optimal experimental conditions of pH equal to 3, $[Fe^{2+}] = 0.04$ mol $L^{-1} = 1360$ mg L^{-1}, $[H_2O_2] = 0.12$ mol L^{-1} = 4080 mg L^{-1}, ratio H_2O_2:Fe^{2+} = 3 and a reaction time of 150 min.

10.2.2 Photo-Fenton Process

The photo-Fenton process consists of combining hydrogen peroxide and UV radiation with Fe^{2+} or Fe^{3+}. This combination is capable of producing more hydroxyl radicals compared to the conventional Fenton method or photolysis and, in turn, increases the rate of degradation of organic pollutants [34, 83, 91]. The Fenton reaction accumulates Fe^{3+} ions in the system and the reaction does not proceed until all Fe^{2+} ions are consumed. Photochemical regeneration of ferrous ions (Fe^{2+}) by photo-reduction of ferric ions occurs in the photo-Fenton reaction [28]. The newly generated ferrous ions react

with H_2O_2 and generate hydroxyl radicals and ferric ions, and the cycle continues (Eq. 10.8):

$$\mathrm{Fe(OH)}^{2} + h\upsilon \rightarrow \mathrm{Fe}^{2+} + \cdot\mathrm{OH} \tag{10.8}$$

Direct photolysis of H_2O_2 produces hydroxyl radicals that can be used to degrade organic compounds. However, in the presence of iron complexes, which strongly absorb radiation, this reaction contributes only to a minor degree to the photodegradation of organic contaminants (Eq. 10.9):

$$\mathrm{H_2O_2} + h\upsilon \rightarrow 2 \cdot \mathrm{OH} \tag{10.9}$$

The conventional photo-Fenton process generally occurs at pH 3, the molar ratio of H_2O_2/iron between 2 and 150, and the molar ratio of H_2O_2/COD between 1 and 3.5 [2, 25, 26]. This process has been classified as being more efficient than conventional Fenton treatment. In some cases, the use of sunlight instead of UV radiation reduces costs. However, this offers a lower level of pollutant degradation [34]. It is worth noting that the iron concentration in water can be considerably lower when the photo-Fenton process is applied compared to the conventional Fenton process (in the absence of adequate light) due to the importance of rapid recycling of iron. One of the main disadvantages of the conventional photo-Fenton process is the low pH of the water. Adjusting the pH (around pH 3) is a relatively easy operation on the laboratory scale, but is generally considered not economically feasible for application on the industrial scale (especially considering subsequent neutralization of the water and separation of the generated sludge). In addition, when sunlight is not used as an energy source, medium or high pressure (200–300 nm) UV lamps are required, which generates high energy consumption [82].

By applying the photo-Fenton process to landfill leachate, Singa et al. [89] obtained a COD removal of 68%. In that study, pH 3, $[Fe^{2+}] = 0.03$ mol $L^{-1} = 1675.35$ mg L^{-1}, ratio H_2O_2:Fe^{2+} $= 3$ and reaction time of 90 min were used. In turn, Colombo et al. [19] applied the photo-Fenton process as post-treatment of a landfill leachate previously subjected to biological treatment and obtained a COD removal of 89%. In their study, pH of 2.4, $[H_2O_2] = 3400$ mg L^{-1}, $[Fe^{2+}] = 80$ mg L^{-1} and reaction time of 120 min were used.

The following is a summary table (Table 10.1) of the treatments and efficiencies discussed within topics 10.2.3.1 and 10.2.3.2.

Table 10.1 Application of Fenton reagent treatment in landfill leachate

Process	UV (W)	$[H_2O_2]$ mg L^{-1}	$[Fe^{2+}]$ mg L^{-1}	Ratio H_2O_2:Fe^{2+}	pH	Reaction time (min)	COD (%)	References
Fenton	–	6000	1000	6	3	32.5	94.41	[81]
Fenton	–	4080	1360	3	3	150	56.49	[88]
Photo-Fenton	16	3060	1020	3	3	90	98	[89]
Photo-Fenton	250	3400	80	–	2.4	120	89	[19]

10.2.3 Ozone-Based Processes

The use of ozone has been shown to be a promising and effective technique for the treatment of highly contaminated effluents, such as landfill leachate, which is produced when water loses through solid waste in the landfill, dissolving toxic substances. The application of O_3 in landfill leachate treatment offers greater odor control, color removal, oxidation of organic and inorganic compounds which consequently reduces the toxicity of the effluent, making it more suitable for safe disposal or reuse in the environment. In addition, ozone is one of the chemical processes receiving considerable attention concerning landfill leachate treatment due to its ability to reduce the amount of non-biodegradable organic compounds present in the leachate [93, 102].

In the liquid medium, ozone undergoes a series of reactions, decomposing into various oxidative species, including the hydroxyl radical, which is an even stronger oxidant than molecular ozone. There are several ways to increase the production of hydroxyl radicals in the treatment of landfill leachate with ozone, such as the addition of hydrogen peroxide, UV radiation, activated carbon, catalysts, ultrasound that depends mainly on oxidation with hydroxyl radicals. Because based on the use of ozone are able to oxidize various organic compounds, pesticides, drugs, emerging contaminants, metals, pharmaceuticals and personal hygiene, solvents present in landfill leachate [74].

The main difference between ozonation processes and advanced ozone-based oxidative processes (AOPS) lies in the oxidation approach used. While ozonation is mainly based on direct oxidation (electrolytic attack) employing ozone dissolved in water, AOPS are based on indirect attack performed by hydroxyl radicals generated during the ozone decomposition process [23, 101]. Typically, under acidic conditions ($pH < 4$), the direct pathway prevails; above pH 10, the transition to the indirect pathway occurs. In mature landfill leachates

(> 10 years), pH values reach above 8, at which time both pathways—direct and indirect—should be considered. A more comprehensive explanation of these ozone reactions in water treatment, specifically in relation to landfill leachates, was discussed by [99]. Ozone-based POAs potentiate hydroxyl radical generation by combining ozone with several complementary techniques such as UV radiation, addition of hydrogen peroxide, Fe (II), metal oxide catalysts, activated carbon, ultrasound, Fenton among others [68]. A radical-type chain reaction that occurs in the solution usually follows these initiations and further increases the transformation of O_3 into OH radicals, as described by [42].

Ozone can alter the molecular structure of organic compounds and subsequently oxidize them into more biodegradable compounds to be removed by biological treatments, for example. Ozone-based AOPs largely reduce the COD and BOD of leachate [31]. The ozone reaction mechanism is a complex process that can occur in two ways [21]:

- Direct reaction with dissolved ozone (O_3).
- Indirect oxidation through the formation of hydroxyl radicals (·OH).

AOP using O_3 are extensively employed for landfill leachate treatment, both in laboratory environments and on a full scale. Ozone, whether in a high pH environment or when combined with hydrogen peroxide, can effectively activate the production of hydroxyl radials. In addition to pH, factors such as temperature, ozone concentration and the amount of H_2O_2 used also influence the process [54]. When AOP is used in ideal conditions, contaminants can be degraded from about 100 ppb to about 5 ppb [86]. In ozonation systems, the pH generally exerts a positive effect on the COD removal rate. This effect is due to two factors: the presence of dissociating compounds that react quickly with ozone, characterizing direct ozonation (phenolic and aromatic compounds), and, in the absence of high concentrations of these compounds, the increase in ozone decomposition to generate hydroxyl radicals, which characterizes indirect ozonation. Thus, it can be deduced that any increase in COD removal during wastewater ozonation with increased pH can be due to either direct or indirect ozone reactions. Direct reactions may be responsible for the increase in COD removal due to the increase in the steady rate of reactions between ozone and the dissociating compounds in the wastewater. In wastewater containing phenol, for example, at pH 4 the rate constant is about 10,000 M^{-1} s^{-1}, while at pH 9 the rate constant increases to approximately, 10^9 M^{-1} s^{-1}. In the absence of compounds that react

rapidly with ozone, the increase in pH gives rise to the appearance of hydroxyl radicals, because ozone decomposes in wastewater (there are no compounds to react directly with ozone) [11]. Figure 10.3 shows a typical ozonation system.

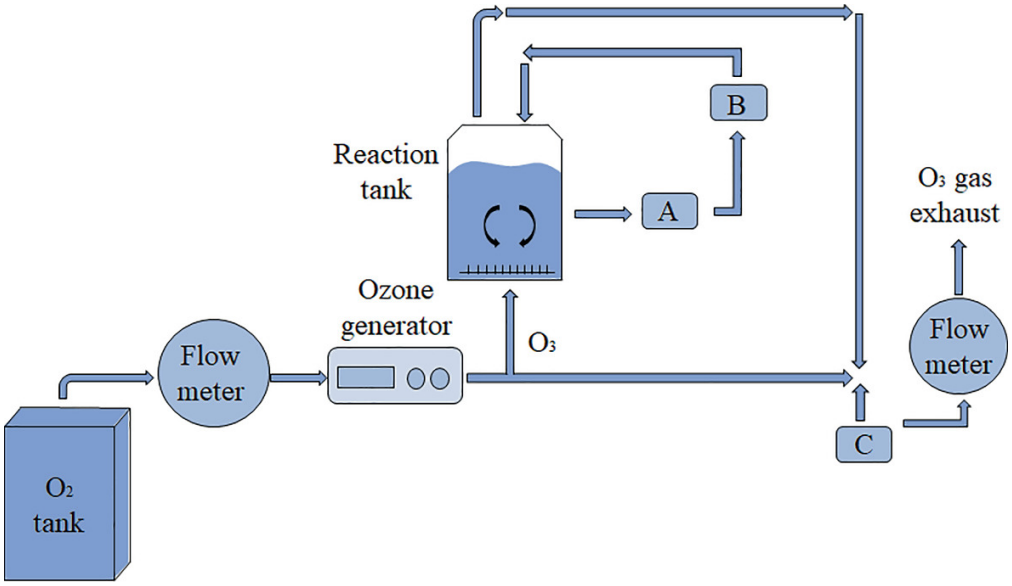


Fig. 10.3 Typical system for ozonation (A = liquid circulation pump; B = ozone liquid analyzer; C = ozone gas analyzer).
Adapted from Mishra et al. [69]

Thus, ozone oxidation depends on the pH value of the effluent to be treated. Generally, in acidic conditions (pH < 4), direct ozonation prevails according to the equation (Eq. 10.10):

$$3O_3 + OH^- + H^+ \rightarrow 2OH + 4O_2 \tag{10.10}$$

On the other hand, when pH > 9, the indirect oxidation pathway prevails. As a rule, the degradation rates in ozonation processes increase with increasing pH, since high pH favors the decomposition of ozone into free radicals as shown in Eq. 10.16. Other chemical reactions involved in indirect oxidation with ozone are as shown below. In the pH range between 4 and 9, direct and indirect reactions can occur simultaneously (Eqs. 10.11–10.16).

$$O_3 + OH^- \rightarrow O_2 + HO_2^- \tag{10.11}$$

$$O_3 + HO_2^- \rightarrow HO_2 + O_3^- \tag{10.12}$$

$$HO_2 \rightarrow H^+ + O_2^- \tag{10.13}$$

$$O_2^- + O_3 \rightarrow O_2 + O_3^- \tag{10.14}$$

$$O_3^- + H^+ \rightarrow HO_3 \tag{10.15}$$

$$HO_3 \rightarrow HO \cdot + O_2 \tag{10.16}$$

Using ozone alone, Tizaoui et al. [94] obtained 27 and 87% removal rates for COD and color, respectively, during the ozonation process for 60 min and using a dosage of 80 g m^{-3} O_3 and pH 8.7. [40] obtained a 22% reduction of COD using 4 g L^{-1} O_3 dosage and pH 8–9, while [79] achieved a 30% reduction of COD during 1 h ozonation process using 1.3–1.5 gO_3 g^{-1} COD degradation and pH 8.2.

10.2.3.1 Ozonation Combined with Persulfate ($O_3/S2O8^{2-}$)

The use of ozone in AOPs presents a low efficiency in the removal of ammonia, which exists in large quantities in already stabilized leachates (from old landfills > 10 years old) and is extremely toxic to the environment and aquatic organisms. Persulfate ($S_2O_8^{2-}$) is a relatively recent oxidant that has come into use in AOPs and can also be used in the process of chemical oxidation of groundwater and in soil cleaning. Chemical or thermal activation of persulfate can produce free sulfate radicals ($SO_4^{\cdot}-$) [96]. Moreover, it has an oxidation potential E° = 2.01 V comparable to ozone (E° = 2.07 V) and due to this, it has received a lot of attention to be used in leachate treatment because of its effectiveness in removing organic compounds and ammonia. Although the persulfate anion can act as a direct oxidant, its removal rates are limited for more recalcitrant contaminants. Therefore, oxidation with persulfate in combination with another treatment process is more efficient for the treatment of stabilized leachates [7].

The reaction mechanism involved in the process of ozone combined with peroxide is as shown in the following equations (Eqs. 10.17–10.19). The hydroxyl radicals formed during the ozonation process act as an initiator in the production of $SO_4^{\cdot -}$. At pH ranging from 2 to 7, persulfate ions react with water to form hydrogen sulfate and hydrogen peroxide ions which lead to increased HO_2^- production of more hydroxyl radicals in the system.

$$S_2O_8^{2-} + 2H_2O \rightarrow 2HSO_4^- + H_2O_2 \tag{10.17}$$

$$S_2O_8^{-2} + OH^- \rightarrow SO_4^{-2} + SO_4^{\cdot -} + 1/2O_2 + H^+ \tag{10.18}$$

$$SO_4^{2-} + OH^- \rightarrow SO_4^{\cdot -} + OH^- \quad (10.19)$$

In the study by [7], in which the performance of the advanced oxidative process combining persulfate and ozone was evaluated, maximum removal efficiencies for COD, color, and NH_3-N of 72, 93, and 55%, of 72, 93, and 55%, respectively, were obtained under ideal operating conditions of 80 g m^{-3} O_3, 35 g $Na_2S_2O_8$ dosage, 35 g $Na_2S_2O_8$ persulfate dosage, pH 10, and reaction time of 210 min. Furthermore, the biodegradability of the stabilized leachate was improved, suggesting a biological treatment process after the O_3/persulfate process. Whereas in the study by [53], the combined process of ozone with persulfate was used to evaluate the degradation of the drug acetaminophen. It was possible to obtain 91.4% removal of the drug under ideal operating conditions that were initial acetaminophen concentration of 150 mg L^{-1}, 60 g m^{-3} O_3, pH 10, dosage, pH 10, the reaction time of 30 min, and 3 mmol L^{-1} persulfate concentrations.

10.2.3.2 Ozonation Combined with Hydrogen Peroxide (O_3/H_2O_2)

The efficiency of using ozone alone in leachate treatment is low. However, its effectiveness can be improved by using advanced oxidation materials and techniques (such as O_3/H_2O_2, O_3/UV, O_3/Fe^{2+}/H_2O_2, etc.) [1]. The hybrid method O_3/H_2O_2 is widely recognized in water and effluent purification. H_2O_2 is colorless, odorless and has a high oxidation potential (1.78). When combined with O_3, the concentration of hydroxyl radicals increases compared to the exclusive use of ozone, which increases the oxidation potential of the system and improves oxidation by free radicals. As a common key reaction for direct dosages of H_2O_2, we must consider that deprotonated H_2O_2 (i.e., HO_2^-) transfers an electron to O_3 and produces deprotonated O_3^- and HO_2/O_2^- which act as charge carriers for the subsequent chain reaction of the type. When pH is increased in one unit, the deprotonated H_2O_2 and H_2O fractions reacting with O_3 increase tenfold, making H_2O_2/HO_2^- a more important reagent for O_3 than the hydroxide ion (OH^-) [11].

Ozone processes can be made more effective, for example, at alkaline pH (O_3/OH^-) and by the addition of hydrogen peroxide

(O_3/H_2O_2). This system favors the production of hydroxyl radicals (·OH), which are highly reactive species. OH^- and H_2O_2 initiate a series of radical reactions that accelerate the decomposition of ozone to produce ·OH [94].

In the system O_3/OH^-, the hydroxide ion reacts with ozone to produce superoxide anion radicals ($\cdot O_2^-$), which in turn are involved in a series of reactions that produce ·OH. In the system O_3/H_2O_2, when H_2O_2 is dissolved in water, partially dissociates into hydroperoxide ion (HO_2^-, the conjugate base of hydrogen peroxide), which reacts rapidly with ozone to initiate chain-less reactions that lead to hydroxyl radicals.

Ozone and hydrogen peroxide react according to reactions (10.20)–(10.25) in a complex set of reactions to form hydroxyl radicals (·OH) [55]. The end product of these reactions is two hydroxyl radicals, involving one molecule of H_2O_2 and two molecules of O_3 [85].

$$H_2O_2 + H_2O \leftrightarrow HO_2^{\cdot -} + H_2O^+ \tag{10.20}$$

$$O_2 + HO_2^{\cdot -} \rightarrow \cdot OH + \cdot O_2^- + O_2 \tag{10.21}$$

$$\cdot O_2^- + H^+ \leftrightarrow HO_2^{\cdot} \tag{10.22}$$

$$\cdot O_2^- + O_3 \rightarrow \cdot O_3^- + O_2 \tag{10.23}$$

$$\cdot O_2^- + H^+ \leftrightarrow HO_3^{\cdot} \tag{10.24}$$

$$H_2O_2 + 2O_3 \rightarrow 2HO \cdot + 3O_2 \tag{10.25}$$

Using hydrogen peroxide (H_2O_2) in AOPs, Tizaoui et al. [94] improved the removal efficiency from 27 to 50% and from 87 to 94% for COD and color, respectively, using the dosage of 2 g L^{-1} of H_2O_2. [40] obtained an improvement in COD removal rate from 22% (ozone only) to 50%, using the dosage of 1 g L^{-1} of H_2O_2 in pH 8.

However, excessive use of H_2O_2 can lead to a reduction in the effectiveness of the organic matter degradation process. The excess of H_2O_2 tends to capture hydroxyl free radicals, as described by [45], resulting in a decrease in the oxidation potential of the process. According to [90], at low peroxide concentrations (between 10^{-5} and 10^{-4} M), it accelerates the decomposition of ozone in water, which leads to an increase in hydroxyl radical formation. Therefore, it is evident that the increase in the dosage of hydrogen peroxide does not

necessarily result in an increase in the oxidant power of the treatment. Each matrix studied has a specific optimal dose.

10.2.3.3 Ozonation with Radiation in the Ultraviolet (UV) Region

In the O_3/UV process, there is a synergistic effect between ozone and UV radiation, which promotes the decomposition of ozone and the production of both direct and indirect (Eqs. 10.26 and 10.27) OH radicals [38]. Ozonation combined with UV radiation is an efficient catalytic system for the degradation of refractory pollutants present wastewater, such as landfill leachates.

$$O_3 + H_2O + h\nu \rightarrow H_2O_2 \tag{10.26}$$

$$H_2O_2 + h\nu \rightarrow 2^{\cdot}OH \tag{10.27}$$

Ozone absorbs UV radiation in the wavelength range (λ) from 200 to 300 nm, with maximum absorption occurring at 253.7 nm, according to [55], while [42] reports that it occurs at 258 nm. Thus, we can deduce, if aqueous ozone is exposed to UV light of one (λ) less than 300 nm, the dissolved ozone molecules when irradiated by UV light, undergo photolysis reactions, resulting in the formation of hydrogen peroxide. This hydrogen peroxide can initiate ozone decomposition. In turn, a hydrogen peroxide molecule may undergo a further photolysis reaction to form two hydroxyl radicals. Therefore, photolysis with ozone increases the generation of hydroxyl ions, and this leads to the formation of hydrogen peroxide (H_2O_2), which proves that a combination of UV radiation with ozonation is more effective than using ozone only.

Thus, this method can be used to degrade compounds that are resistant to treatment by only ozonation. Chen [15] achieved very effective mineralization of total organic carbon (TOC) with UV radiation of 96 W and ozone dosage of 3.8 g h^{-1}. Wang et al. [98] also used this process (O_3/UV) applied for landfill leachate treatment and compared the results to the treatment process using ozone (O_3) only. As a result, he obtained better COD and color removal efficiency in the combined O_3/UV process, with approximately 10–15% improvement in COD removal (reaching around 57% removal) and 15–20% in color removal (reaching 55% removal). van Aken et al. [97] applied the O_3/UV process intending to remove recalcitrant compounds from landfill leachate, using the following operational conditions: 16 g h^{-1} ozone (O_3) concentration, 425 W UV radiation intensity, pH 9, and

reaction time of 240 min. Under these conditions, it was observed that the combination of UV radiation with O_3 is not satisfactorily efficient to remove recalcitrant compounds, but it was possible to obtain a reduction of COD of 37%.

10.2.3.4 Combined Ozonation with Hydrogen Peroxide and UV Radiation (O_3/H_2O_2/UV)

In the O_3/H_2O_2/UV system, there is a wide variety of individual processes that can give rise to the generation of hydroxyl radicals (·OH). Therefore, this ternary system can be considered as the result of the integration of different unitary or binary systems, such as direct photolysis, ozonation only, UV photolysis of O_3, the combination between O_3 and H_2O_2, and UV photolysis of H_2O_2 [21].

All these processes result in the formation of hydroxyl radicals (·OH) through a sequence of reactions that, directly or indirectly, are related to the formation of H_2O_2, as shown in the following reactions (Eqs. 10.28 and 10.29) [21]:

$$O_3 + H_2O \rightarrow 2 \cdot OH + O_2 \rightarrow H_2O_2 + O_2 \tag{10.28}$$

$$2H_2O_2 + O \rightarrow 2HO + O_3^- \tag{10.29}$$

The main advantage of this ternary system is that ozone decomposition is accelerated by the simultaneous presence of hydrogen peroxide and UV irradiation, which results in an increasing rate of ·OH radical generation. In addition, this process can be applied under mild temperature and pressure conditions. Regarding the disadvantages, the high costs of the three elements that constitute the system (i.e., ozone, hydrogen peroxide, and UV radiation) represent the main limitation for the application of this system in a wider process. Consequently, the use of this ternary system is generally restricted to the treatment of highly polluted effluents to achieve adequate degradation and mineralization of recalcitrant pollutants [21].

According to [58], the ternary O_3/H_2O_2/UV system was applied to treat a winery effluent at pH 10, and it was possible to achieve 57% COD removal during 300 min of the experiment, obtaining higher degradation at pH 10 compared to pH 7 and 4. [75] used the O_3/H_2O_2/UV_{solar} process intending to improve the quality of landfill leachate. To this end, experiments were performed under optimal operating conditions with an ozone concentration of 3 gO_3 L^{-1},

hydrogen peroxide concentration of 0.67 g L^{-1} UV_{solar} radiation intensity of 28 kJ L^{-1} $Q_{uv,}$ and pH 8.7. As a result, it was possible to obtain a 17.3 and 56% reduction of COD and color, respectively. Gomes et al. [36] applied the O_3/H_2O_2/UV system to treat biologically pretreated landfill leachate. Adopting this treatment option, it was possible to obtain a COD removal of 67% with pH ranging from 7 to 9.

Table 10.2 shows the summary of treatments and efficiencies discussed within topics 10.2.3.1 to 10.2.3.4.

Table 10.2 Application of ozone treatments to sanitary landfill leachate

Process	Dosage O_3	Dosage H_2O_2	Dosage $S_2O_8^{2-}$	UV	pH	Time (min)	DQO (%)	Color (%)	References
O_3	0.08 (g L^{-1})	–	–	–	8.7	60	27	87	[94]
O_3	4.0 (g L^{-1})	–	–	–	8–9	(?)	22	–	[40]
O_3	1.3–1.5 (g L^{-1})	–	–	–	8.2	60	30	–	[79]
O_3	0.80 (g L^{-1})	–	–	–	8.2–8.5	240	23	69	[18]
O_3/H_2O_2	0.08 (g L^{-1})	2.0 (g L^{-1})	–	–	8.7	60	50	94	[94]
O_3/H_2O_2	4.0 (g L^{-1})	1.0 (g L^{-1})	–	–	8–9	(?)	50	–	[40]
O_3/H_2O_2	5.6 (g h^{-1})	0.4 (g L^{-1})	–	–	7	60	72	–	[20]
O_3/H_2O_2	0.18 (g L^{-1})	0.5 (g L^{-1})	–	–	7–9	180	57	–	[36]
O_3/	0.08 (g L^{-1})	–	35 (g)	–	10	210	72	93	[1]
O_3/UV	16.0 (g h^{-1})	–	–	425 (W)	9	240	37	–	[97]
O_3/UV	3.4 (mg g^{-1}DQO)	–	–	600 (W)	–	90	57	55	[98]
$O_3/H_2O_2/UV_{solar}$	3.0 (g h^{-1})	0.67 (g L^{-1})	–	28 (KJ L^{-1} Q_{uv})	8.7	180	17.3	56	[75]
O_3/H_2O_2/UV	0.18 (g L^{-1})	0.5 (g L^{-1})	–	6 (W)	7–9	180	67	–	[36]

Process	Dosage O_3	Dosage H_2O_2	Dosage $S_2O_8^{2-}$	UV	pH	Time (min)	DQO (%)	Color (%)	References
O_3/H_2O_2/UV	6.0 (g h^{-1})	1.16–2.32 (g L^{-1})	–	36 (W)	10	300	57	–	[58]

10.2.3.5 Catalytic Ozonation

Despite the remarkable advantages of AOP, they present the disadvantage of generating toxic intermediates, since they promote only partial oxidation of organic compounds [52]. To overcome this limitation, catalysts are used, making room for advanced catalytic ozonation technique. Initial studies indicate its effectiveness in the removal of all types of organic and inorganic compounds present in landfill leachate with high degradation efficiency [70].

A variety of metals, metal oxides, activated carbon and minerals are used as catalysts to improve efficiency in the removal of pollutants [47]. According to the catalyst used, catalytic ozonation can be divided into homogeneous catalytic ozonation and heterogeneous catalytic ozonation. Transition metal ions (Fe^{2+}, Cu^{2+}, Cr^{2+}, Mn^{2+}, Ni^{2+}, Co^{2+}, Cd^{2+}, Ag^{+}, Zn^{2+}) are often used as catalysts in the homogeneous catalytic process, to disintegrate persistent organic pollutants found in wastewater. These metal ions influence the reaction speed, the selectivity of oxidation by ozone and the effectiveness in the application of ozone. Homogeneous catalytic ozonation significantly improves the removal of most organic pollutants in water [57, 100]. However, the addition of metal ions results in the production of secondary contaminants in several situations. Thus, significant research has been directed towards heterogeneous catalytic ozonation as an ideal process to remove recalcitrant organic and inorganic pollutants [29].

Nowadays, there has been extensive research on various heterogeneous catalysts in solid form in the context of the ozonation system. Compared with homogeneous ozone based AOPs, heterogeneous catalytic ozonation based on solid catalysts attracted great interest due to the advantages of low operating cost, high efficiency, and low chemical consumption [48]. However, so far, only a few studies of heterogeneous catalytic ozonation have been reported for the treatment of landfill leachates and development of an efficient and stable process for advanced leachate treatment is promising, although challenging [77].

10.2.3.6 Photocatalytic Ozonation

Among the advanced oxidation methods currently employed with UV radiation to treat landfill leachate, heterogeneous photocatalysis using titanium dioxide (UV/TiO_2) is highlighted as a high efficiency technique [14, 16, 71]. This process acts in the complete mineralization and degradation of pollutants present in waters and effluents, transforming toxic or recalcitrant compounds into biodegradable substances or easily removable by other processes such as biological or physicochemical [5, 65, 92], however, represents a slow process due to the low oxidation rate. Similarly, the oxidation process by molecular O_3, due to its high selectivity, results only in partial degradation of organic compounds, forming carboxylic acids and end products that cannot be completely mineralized.

For these reasons, the simultaneous approach of these two methods, resulting in photocatalytic ozonation, can provide a robust complementation of processes, increasing the rate of degradation and leading to a more comprehensive and efficient treatment [63]. This process is considered a topic of recent research, and the literature suggests that the free electrons generated by TiO_2, in its surface layer, interact with molecules of O_3 or O_2 generating superoxide radicals ($O_2^{-}\cdot$). Subsequent reactions originate the chemical species called ozonide radical ($O_3^{-}\cdot$), capable of decomposing into molecules $\cdot OH$ and O_2, characterizing a synergism between processes (UV/TiO_2 and O_3).

Various combinations of ozone, oxygen, and hydrogen peroxide, homogeneous and heterogeneous catalysts, as well as photocatalysts, have been studied by several researchers around the world. One such combination employs semiconductor photoactivation (photocatalysis) in combination with ozone, which results in a new advanced oxidation method called photocatalytic ozonation [6, 64]. This new method is considered a promising technique for the decomposition of refractory microorganisms and organic compounds. [103], for example, showed that among six different advanced oxidation processes, photocatalytic ozonation was the most efficient for completing the mineralization of 4-chloronitrobenzene.

The presence of photocatalysts (other than ozone) in the oxidation medium and the adsorption of ozone and pollutants on its surface can essentially alter the oxidation mechanisms, which indicates that photocatalytic ozonation is a different process than ozonation in the absence of a photocatalyst. A schematic of the photocatalytic ozonation process is shown in Fig. 10.4. The photocatalytic reactions

begin by photo-exciting the photocatalyst surface with UV–Vis radiation, which can provide the energy in the appropriate band range to generate photoactivated hollow-electron pairs (Eq. 10.30). In parallel, ozone molecules can adsorb onto the photocatalyst surface through three different interactions: (i) physical adsorption, (ii) formation of weak hydrogen bonds with surface hydroxyl groups, and (iii) molecular or dissociative adsorption on acid sites [13] each interaction resulting in the production of active oxygen (·O) radicals (Eq. 10.31). [44] showed that these active oxygen radicals react with water molecules to produce hydroxyl radicals, which play a key role in photocatalytic ozonation processes.

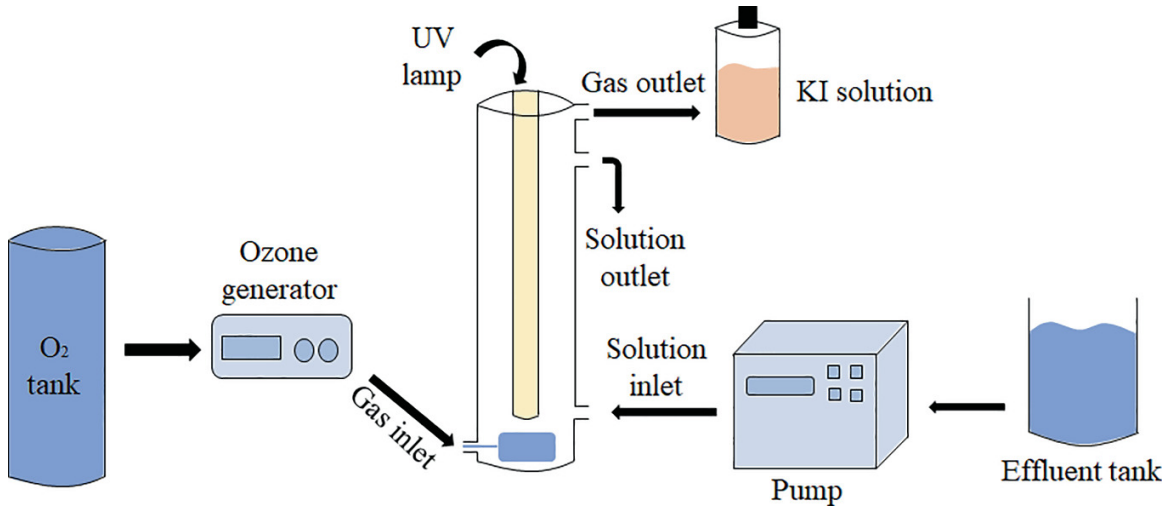


Fig. 10.4 Schematic representation of the photocatalytic ozonation process. Kang Adapted from et al. [49]

In addition, using wavelengths shorter than 300 nm, the ozone and hydrogen peroxide molecules will also absorb this wavelength, producing active oxidizing reagents (Eqs. 10.32 and 10.33) [39].

The photogenerated electrons on the photocatalyst surface (Eq. 10.30) react with the adsorbed oxygen and ozone molecules (Eqs. 10.34 and 10.35), and these reactions are important to decrease the recombination rate of the hollow-electron pairs. This recombination of electrons and gaps negatively affects the reduction and oxidation reactions on the photocatalyst surface, quantitatively decreasing the effective interactions and consequently reducing the performance of this technology. The electronic affinity of ozone (2.1 eV) is higher than that of oxygen (0.44 eV), which ensures that photocatalytic reactions occur more easily in the presence of ozone than in the presence of oxygen. The hydrogen peroxide molecules generated can also react with the electrons photo-excited on the photocatalyst surface, forming hydroxyl radicals (Eq. 10.36) [66].

The photogenerated hollow-electron (Eq. 10.30) are assumed to either directly attack the pollutant molecules adsorbed on the photocatalyst surface (Eq. 10.37) or react with water molecules (under acidic conditions) or, more efficiently, with hydroxide anions (under alkaline conditions), producing hydroxyl radicals (Eq. 10.38), which are responsible for oxidizing the pollutant molecules. Other oxidizing species, such as oxygen atom radicals (·O) and superoxide radical and ozonide anions ($O_2{\cdot}^-$ and $O_3{\cdot}^-$), are believed to be generated as reactive intermediates in the middle of the photocatalytic ozonation process and can potentially oxidize the contaminants (directly) or produce hydroxyl radicals through different chain reactions (Eqs. 10.39–10.49). In addition, some pollutants, such as phenols, amines, and humic substances, are presumed to act as initiators of ozone decomposition into radical anions of the ozonide ($O_3{\cdot}^-$) and then hydroxyl radicals, which consequently accelerate the degradation of these pollutants (Eq. 10.50) [12, 87]. The hydroxyl radicals produced in the aforementioned processes attack pollutant molecules and non-selectively decompose them into less harmful substances (Eq. 10.51). Three possible mechanisms are assumed for the decomposition of pollutants by hydroxyl radicals: electron transfer, radical addition, and hydrogen abstraction.

Initial reactions in the presence of photocatalyst, ozone, and light energy source:

$$Photocatalyst + h\upsilon \rightarrow Photocatalyst + e^- + h^+ \tag{10.30}$$

$$O_3 + Photocatalyst\,(activesurface) \rightarrow {\cdot}O + O_2 \tag{10.31}$$

$$\mathrm{O_3} + h\upsilon \rightarrow {\cdot}\mathrm{O} + \mathrm{O_2} \tag{10.32}$$

$$\mathrm{H_2O_2} + h\upsilon \rightarrow 2\mathrm{HO}{\cdot} \tag{10.33}$$

Possible photogenerated electron reactions on the photocatalyst surface:

$$\mathrm{O_2}\,(ads) + e^- \rightarrow {\cdot}\mathrm{O_2^-} \tag{10.34}$$

$$\mathrm{O_2}\,(ads) + e^- \rightarrow {\cdot}\mathrm{O_3^-} \tag{10.35}$$

$$\mathrm{H_2O_2}\,(ads) + e^- \rightarrow {\cdot}\mathrm{OH} + \mathrm{OH^-} \tag{10.36}$$

Possible reactions of photogenerated gaps (h^+) on the photocatalyst surface:

$$Polluent\,(ads) + h^+ \rightarrow [Pollutant]^+_{ads} \rightarrow^{multi-steps} \mathrm{CO}_x, \mathrm{NO}_x, \mathrm{SO}_x,\ etc. \tag{10.37}$$

$$\mathrm{H_2O\,(ads)} + h^+ \rightarrow \cdot\mathrm{OH} + \mathrm{H^+}\,(pH < 7)\ or\ \ \mathrm{OH^-_{(ads)}} + h^+ \rightarrow \cdot\mathrm{OH}(pH > 7) \quad (10.38)$$

Possible additional chain reactions in the oxidation medium:

$$\cdot\mathrm{O} + \mathrm{H_2O} \rightarrow 2\mathrm{HO}\cdot \quad (10.39)$$

$$\cdot\mathrm{O_2^-} + \mathrm{H^+} \leftrightarrow \mathrm{HO_2^{\cdot}} \quad (10.40)$$

$$\cdot\mathrm{O_3^-} + \mathrm{H^+} \leftrightarrow \mathrm{HO_3^{\cdot}} \quad (10.41)$$

$$\mathrm{HO_3^{\cdot}} \rightarrow \mathrm{O_2} + \cdot\mathrm{OH} \quad (10.42)$$

$$\mathrm{O_3} + \mathrm{OH^-} \rightarrow \cdot\mathrm{O_2^-} + \mathrm{HO_2^{\cdot}} \quad (10.43)$$

$$\mathrm{O_3} + \cdot\mathrm{OH} \leftrightarrow \mathrm{HO_4^{\cdot}} \rightarrow \mathrm{O_2} + \mathrm{HO_2^{\cdot}} \quad (10.44)$$

$$2\mathrm{HO_2^{\cdot}} \rightarrow \mathrm{H_2O_2} + \mathrm{O_2} \quad (10.45)$$

$$\mathrm{H_2O_2} + \cdot\mathrm{O_2^-} \rightarrow \cdot\mathrm{OH} + \mathrm{OH^-} + \mathrm{O_2} \quad (10.46)$$

$$\mathrm{O_3} + \mathrm{HO_2^{\cdot}} \rightarrow 2\mathrm{O_2} + \cdot\mathrm{OH} \quad (10.47)$$

$$\mathrm{O_3} + \cdot\mathrm{O_2^-} \rightarrow \cdot\mathrm{O_3^-} + \mathrm{O_2} \quad (10.48)$$

$$\mathrm{O_3} + \mathrm{H_2O_2} \rightarrow \cdot\mathrm{OH} + \mathrm{HO_2^{\cdot}} + \mathrm{O_2} \quad (10.49)$$

$$Pollutant + \mathrm{O_3} \rightarrow [Pollutant]^+ + \cdot\mathrm{O_3^-} \quad (10.50)$$

$$Pollutant_{(ads)} + \cdot\mathrm{OH} \rightarrow [Pollutant]^{\cdot}_{(ads)} + \mathrm{H_2O} \rightarrow^{multi-steps} \mathrm{CO}_x, \mathrm{NO}_x, \mathrm{SO}_x, etc. \quad (10.51)$$

Photocatalytic ozonation is considered one of the most expensive treatment technologies, and its use in removing biodegradable pollutants from water is not economically justifiable. The particular importance of this oxidation method applies to the destruction of poorly biodegradable organic compounds or to improve the biodegradability of wastewater samples containing these compounds [27, 76].

In the work of [46], the catalytic ozonation process was applied to treat sanitary landfill leachate, which had previously undergone a biological treatment step. In this study, activated carbon was used as the catalyst of the process and, as a result, it was possible to obtain a reduction of the COD of the leachate of 89.87% under the following optimal conditions: temperature of 25 °C, the reaction time of 120 min, ozone concentration of 60 g m^{-3} and 350 g dosage of

activated carbon. The results obtained in this combination of ozone and a catalyst were better than those obtained for the treatment of the same leachate only with ozone or only with activated carbon.

In [49], an efficient approach to improve mass transfer in the photocatalytic ozonation process using a helical photocatalytic module (HPM) and an annular UVC reactor was presented, applied to wastewater treatment. The optimal conditions used in the experiment were: radiation intensity of 11 W, ozone concentration between 0 and 100 $gO_3\ m^{-3}$, and hydraulic retention time of 19 min. Under these conditions, it was possible to obtain a TOC reduction of 91.5% using the helical photocatalytic module (HPM) and 58.1% without using the HPM. Regarding COD, the reduction obtained was 63% using a hydraulic retention time of 57 min.

In the studies of [32] the combination of UV radiation with ozone and TiO_2 was used to investigate the potential of this combination in the treatment of winery effluents. The parameters used were photocalyzer (TiO_2) concentration of 1.5 g L^{-1}, ozone concentration of 50 mg L^{-1}, pH 3, and duration of 120 min, and radiation intensity of 30 W. Under these conditions, it was possible to obtain an 80% reduction of COD.

The research of [27] investigated the efficiency of ozonation and ozonation combined with catalyst (TiO_2 and activated carbon) in the presence and absence of UV radiation for the treatment of automotive gas station effluent. The ideal conditions for performing the tests were as follows: ozone concentration of 15 mg L^{-1}, pH 7.2, catalyst concentration of 0.3 g L^{-1}, reaction time of 60 min, and temperature of 20ºC. Under these conditions, it was possible to obtain COD removal levels of 50% for the O_3/CA (activated carbon) system, 38% for the O_3/TiO_2 system, and 30% for the system with only ozone.

Table 10.3 below presents some applications of the catalysis/photocatalysis processes in the treatment of effluents.

Table 10.3 Application of catalysis/photocatalysis processes to wastewater treatment

Process	Effluent	Catalyst	Dosage of O_3	UV (W)	COD (%)	TOC (%)	pH	Time (min)	References
O_3/AC[a]	Landfill leachate	Activated charcoal	60 g/m^3	–	89.8	–	8.4	120	[46]
O_3/UV	Wastewater	HPM[b]	0–100 g/m^3	11	63	91.5	7.4	19/57	[49]

Process	Effluent	Catalyst	Dosage of O_3	UV (W)	COD (%)	TOC (%)	pH	Time (min)	References
O_3/UV/TiO_2	Winey effluent	TiO_2	50 g/m^3	30	80	–	3	120	[32]
O3/UV/cat[c]	Gas station effluent	TiO_2/AC[a]	15 g/m^3	(?)	30/50	–	7.2	60	[27]
UV/TiO_2	Refinery effluent	TiO_2	–	400	78	–	3	120	[84]
UV/TiO_2	Landfill leachate	TiO_2	–	(?)	96	98	3–8	120/360	[24]
UV/TiO_2–W–C	Landfill leachate	TiO_2–W–C	–	40	32/84	–	(?)	3600	[9]
UV/H_2O_2-PMS[d]	Landfill leachate	H_2O_2-PMS	–		28/38	–	7.5	60	[37]

[a] *AC* Activated Charcoal
[b] *HPM* Helical Photocatalytic Module
[c] *cat* Catalyst
[d] *PMS* Peroxymonosulfate

10.2.3.7 Combined Ozonation with Ultrasound

Ultrasound (US) is one of the advanced oxidation technologies that can also be used to produce reactive radicals, such as hydroxyl radicals, through the collapse of cavitation bubbles. Existing revisions are usually divided into two types. As several studies have shown that the frequency of ultrasound used in wastewater treatment ranges from low frequency [67], medium frequency [51] to high frequency [62], the ultrasound discussed in this review refers to general ultrasound. However, to date, there are few reviews focused on the use of ultrasound-based AOPs to treat landfill leachate [8, 43, 56, 95].

$$H_2O\,US \rightarrow \cdot OH + \cdot H \tag{10.52}$$

$$\cdot OH + \cdot OH \rightarrow H_2O_2 \tag{10.53}$$

$$2 \cdot H \rightarrow H_2 \tag{10.54}$$

$$\cdot H \rightarrow O_2^{\cdot} \rightarrow HO_2^{\cdot} \tag{10.55}$$

$$2HO_2^{\cdot} \rightarrow H_2O_2 + O_2 \tag{10.56}$$

The combination of O_3 and ultrasound (US) can strengthen the oxidation process by O_3 from the following aspects: (i) US can break

the O_3 bubbles and increase the surface area of the exposure of the atmospheric liquid, thereby improving the mass transfer coefficient of O_3; (ii) Strengthening the decomposition of O_3 to produce [105]. O_3 promotes the formation of and H_2O_2 through the pyrolysis of cavitation bubbles [59]. Studies have shown that although the use of ultrasound can improve the efficiency of degradation of organic matter, ammonia, and pathogenic microorganisms in landfill leachate, it has no effect on the degradation of heavy metals, increases the turbidity of the leachate and changes the electrical conductivity in the system [3, 4] (Mojtaba 2018). Along with its low energy utilization and high cost [56], ultrasound is usually combined with other AOP to overcome these shortcomings and improve the degradation effect.

10.2.4 Heterogeneous Photocatalysis

Among the existing AOPs, heterogeneous photocatalysis employs semiconductors as catalysts (TiO_2, ZnO, Fe_2O_3, CdS, GaP, and ZnS) and shows good efficiency in degradation of a wide range of refractory organic compounds [17]. Photocatalysis is defined as the acceleration of a photochemical reaction by the presence of a semiconductor, which is activated by absorbing radiation with energy above its bandgap. The term “heterogeneous” refers to the fact that the contaminants are present in a fluid phase, while the catalyst is present in a solid phase. The most commonly used catalyst is titanium dioxide (TiO_2) due to its high chemical stability, low cost, and excellent results [34, 35].

The first reaction of the photocatalysis process is the absorption of UV radiation by the catalyst, followed by the formation of hollow-electron (h^+/e^-), pairs, according to reaction (10.57). In environmental applications, photocatalytic processes are carried out under aerobic conditions and oxygen can be absorbed on the surface of the catalyst. Therefore, the mentioned electrons, due to their high reducing power, reduce the absorbed oxygen on the TiO_2 surface, initiating the generation of superoxide radical ions ($\cdot O_2^-$), as indicated in reaction (10.58). On the other hand, the gaps can cause the oxidation of the effluent and/or HO– absorbed species into ·OH radicals, according to reactions (10.59) and (10.60), which subsequently oxidize the organic compounds. When organic matter is also absorbed in the catalyst surface, it can be oxidized directly by transferring an electron from the catalyst [73].

In the presence of redox species adsorbed on the semiconductor particle and under irradiation, oxidation and reduction reactions occur

simultaneously on the catalyst surface. The photogenerated reductants generate photo-oxidation reactions while the electrons in the conduction band generate photo-reduction reactions (Eqs. 10.57–10.60) [21]:

$$\mathrm{TiO_2} + h\upsilon \rightarrow \mathrm{TiO_2}\left(e^-\right) + \mathrm{TiO_2}\left(h^+\right) \tag{10.57}$$

$$\mathrm{TiO_2}\left(e^-\right) + \mathrm{O_2} \rightarrow \mathrm{TiO_2} + \mathrm{O_2^-} \tag{10.58}$$

$$\mathrm{TiO_2}\left(h^+\right) + \mathrm{H_2O} \rightarrow \mathrm{TiO_2} + \mathrm{O_2^-} \tag{10.59}$$

$$\mathrm{TiO_2}\left(h^+\right) + \mathrm{HO^-} \rightarrow \mathrm{TiO_2} + \mathrm{OH} \tag{10.60}$$

The main advantages of this AOP are that it can be operated under ambient temperature and pressure, the possibility of using sunlight as a source of irradiation of the catalyst, and the low cost and reuse of the catalyst. In addition, this system can achieve the complete mineralization of many compounds. However, it also has some disadvantages, such as the difficulties of achieving uniform irradiation over the entire catalyst surface on a large scale or the need for a subsequent separation treatment to recover the catalyst in suspension, which makes the process more expensive [21].

Saien and Shahrezaei [84] subjected a real oil refinery wastewater containing a variety of aliphatic and aromatic organic compounds to treatment using TiO_2 particles as a photocatalyst in the UV/TiO_2 process. A catalyst concentration of only 100 mg L^{-1}, pH 3, and temperature of 45 °C were used, these were considered to be ideal operating conditions. Thus, a degradation efficiency of more than 78% of organic pollutants, as determined by the COD criterion, was achieved by applying irradiation for 120 min.

In the work of [24] a combination of biological and photocatalysis processes was used to treat landfill leachate. The leachate was first subjected to the biological treatment process with Kefir grains and after 5 days, with the controlled temperature at 37 °C, it was possible to obtain removals for TOC, COD, NH_4^+-N and PO_4^{-3} of, respectively, 93, 83.33, 70, and 88.25%. In the photocatalysis step, the concentration of the catalyst (TiO_2) used varied between 0.5 and 2 g L^{-1}, as well as the pH varied between 3 and 8, and the reaction time between 120 and 360 min. As a result, photocatalysis was able to increase the removal efficiency to 98, 96, 85, and 93% of TOC, COD, NH_4^+–N, and PO_4^{-3}, respectively.

In the research of [9], it was investigated the efficiency of COD removal from landfill leachate through the process of photocatalysis, using as catalyst titanium dioxide nanoparticles (TiO_2) with tungsten (W) and carbon (C). For leachate with initial COD of 550 mg L^{-1}, it was possible to obtain a removal of 84% of COD, using an irradiation intensity of 40 W and leachate flow of 1 L min^{-1}. For leachate with initial COD of 10,100 mg L^{-1}, it was obtained the removal of 32% of COD in 60 h of treatment, under the same conditions. In the studies by [37] photocatalysis was used combining ultraviolet radiation (UV) with hydrogen peroxide (UV/H_2O_2), peroxymonosulfate (UV/PMS), and with both oxidants (UV/H_2O_2/PMS), aiming at reducing COD from landfill leachate. As a result, the treatment efficiency followed the following order: UV/PMS treatment with removal of 37.39% > $UV/PMS/H_2O_2$ treatment with removal of 30.51% > UV/H_2O_2 treatment with removal of 28.59%.

10.3 Conclusion

In most countries, landfilling is the most common way to dispose of municipal solid waste. Despite the many advantages, the generation of highly polluted leachates, showing significant variations in both volume flow and chemical composition, constitutes a major disadvantage. The highly polluting issue of landfill leachates is indisputable. Understanding the leachate profiles of landfills can be useful to promote the sustainable management of municipal solid waste, however, there are few studies at a global level. Each year recognition of the impact of landfill leachate on the environment has forced authorities to set increasingly stringent requirements for pollution control. This chapter brought the advantages of the advanced oxidative processes applied for the oxidation of the leachate and were discussed in the items: (i) Fenton and photo-Fenton, (ii) ozone-based processes and the combination with other processes, and, (iii) heterogeneous photocatalysis.

We can conclude at the end of this chapter that, as a highly complex liquid in its composition and diversity is leached, the advanced oxidative processes applied to the treatment of these polluting residues become so difficult and diverse. Most techniques applied in oxidative processes are combined to enhance the oxidation process.

The application of AOP to landfill leachate has been shown to be effective, efficient, and advantageous, making leachate less

environmentally toxic. From future perspectives, research is sought that uses combined methods to make leachate a non-toxic liquid that can be reused for irrigation, recreation, and industrial processes.

References

1. Abu Amr SS, Aziz HA, Adlan MN (2013) Optimization of stabilized leachate treatment using ozone/persulfate in the advanced oxidation process. Waste Manag 33:1434–1441. https://doi.org/10.1016/j.wasman.2013.01.039
2. Affam AC, Kutty SRM, Chaudhuri M (2012) Solar photo-fenton induced degradation of combined chlorpyrifos, cypermethrin and chlorothalonil pesticides in aqueous solution. Int J Chem Environ Eng 6:167–173
3. Afsharnia M, Torabian A, Mousavi GR et al (2012) Landfill Leachate treatment by sono-evaporation. Desalin Water Treat 48(1–3):344–348. https://doi.org/10.1080/19443994.2012.702959
4. Afsharnia M, Biglari H, Rasouli SS et al (2018) Sono-electrocoagulation of fresh leachate from municipal solid waste; Simultaneous applying of iron and copper electrodes. Int J Electrochem Sci 13:472–484. https://doi.org/10.20964/2018.01.22
5. Ahmad R, Ahmad Z, Khan AU et al (2016) Photocatalytic systems as an advanced environmental remediation: recent developments, limitations and new avenues for applications. J Environ Chem Eng 4(4):4143–4164. https://doi.org/10.1016/j.jece.2016.09.009
6. Agustina TE, Ang HM, Vareek VK (2005) A review of synergistic effect of photocatalysis and ozonation on wastewater treatment. J Photochem Photobiol C Photochem Rev 6:264–273. https://doi.org/10.1016/j.jphotochemrev.2005.12.003
7. Amr SSA, Aziz HA, Adlan MN, Bashir MJK (2013) Pretreatment of stabilized leachate using ozone/persulfate oxidation process. Chem Eng J 221:492–499. https://doi.org/10.1016/j.cej.2013.02.038
8. Asaithambi P, Sajjadi B, Abdul Aziz AR, Daud WMABW (2017) Ozone (O3) and sono (US) based advanced oxidation processes for the removal of color, COD and determination of electrical energy from landfill leachate. Sep Purif Technol 172:442–449. https://doi.org/10.1016/j.seppur.2016.08.041
9. Azadi S, Karimi-Jashni A, Javadpour S, Amiri H (2020) Photocatalytic treatment of landfill leachate using cascade photoreactor with immobilized W-C-codoped TiO2 nanoparticles. J Water Process Eng 36:101307. https://doi.org/10.1016/j.jwpe.2020.101307
10. Babuponnusami A, Muthukumar K (2013) A review on Fenton and improvements to the Fenton process for wastewater treatment. J Environ Chem Eng 557–572. https://doi.org/10.1016/j.jece.2013.10.011
11. Beltrán FJ (2005) Ozone reaction kinetics for water and wastewater systems
12. Beltrán FJ, Aguinaco A, Rey A, García-Araya JF (2012) Kinetic studies on black light photocatalytic ozonation of diclofenac and sulfamethoxazole in water. Ind Eng Chem Res 51:4533–4544. https://doi.org/10.1021/ie202525f

13. Bulanin KM, Lavalley JC, Tsyganenko AA (1995) IR spectra of adsorbed ozone. Colloids Surf A Physicochem Eng Asp 101:153–158. https://doi.org/10.1016/0927-7757(95)03130-6
14. Carard RF, Junior Schiavon G, De Castro TM et al (2021) Photocatalytic ozonation performance in landfill leachate treatment. An Acad Bras Cienc 93:1–20. https://doi.org/10.1590/0001-3765202120190137
15. Chen WS, Juan CN, Wei KM (2007) Decomposition of dinitrotoluene isomers and 2,4,6-trinitrotoluene in spent acid from toluene nitration process by ozonation and photo-ozonation. J Hazard Mater 147:97–104. https://doi.org/10.1016/j.jhazmat.2006.12.052
16. Cherni Y, Elleuch L, Messaoud M et al (2021) Recent technologies for leachate treatment: a review. Euro-Mediterr J Environ Integr 6:1–18. https://doi.org/10.1007/s41207-021-00286-z
17. Chong MN, Jin B, Chow CWK, Saint C (2010) Recent developments in photocatalytic water treatment technology: a review. Water Res 44:2997–3027. https://doi.org/10.1016/j.watres.2010.02.039
18. Chys M, Oloibiri VA, Audenaert WTM et al (2015) Ozonation of biologically treated landfill leachate: efficiency and insights in organic conversions. Chem Eng J 277:104–111. https://doi.org/10.1016/j.cej.2015.04.099
19. Colombo A, Módenes AN, Góes Trigueros DE et al (2019) Treatment of sanitary landfill leachate by the combination of photo-Fenton and biological processes. J Clean Prod 214:145–153. https://doi.org/10.1016/j.jclepro.2018.12.310
20. Cortez S, Teixeira P, Oliveira R, Mota M (2011) Evaluation of Fenton and ozone-based advanced oxidation processes as mature landfill leachate pre-treatments. J Environ Manage 92:749–755. https://doi.org/10.1016/j.jenvman.2010.10.035
21. Cuerda-Correa EM, Alexandre-Franco MF, Fernández-González C (2020) Advanced oxidation processes for the removal of antibiotics from water: an overview. Water (Switzerland) 12. https://doi.org/10.3390/w12010102
22. Deng Y, Englehardt JD (2006) Treatment of landfill leachate by the Fenton process. Water Res 40:3683–3694. https://doi.org/10.1016/j.watres.2006.08.009
23. Deng Y, Zhao R (2015) Advanced Oxidation Processes (AOPs) in wastewater treatment. Curr Pollut Rep 1:167–176. https://doi.org/10.1007/s40726-015-0015-z
24. Elleuch L, Messaoud M, Djebali K et al (2020) A new insight into highly contaminated landfill leachate treatment using Kefir grains pre-treatment combined with Ag-doped TiO2 photocatalytic process. J Hazard Mater 382:121119. https://doi.org/10.1016/j.jhazmat.2019.121119
25. Elmolla ES, Chaudihuri M (2010) Effect of photo-Fenton operating conditions on the performance of photo-Fenton-SBR process for recalcitrant wastewater treatment. J Appl Sci 10:3236–3242
26. Elmolla ES, Chaudhuri M (2010) Photo-Fenton treatment of antibiotic wastewater. Nat Environ Pollut Technol 9:365–370
27. Emam EA (2012) Effect of ozonation combined with heterogeneous catalysts and ultraviolet radiation on recycling of gas-station wastewater. Egypt J Pet 21:55–60. https://doi.org/10.1016/j.ejpe.2012.02.008

28. Faust BC, Hoigné J (1990) Photolysis of Fe(III)-Hydroxy complexes as sources of OH radicals in clouds fog and rain. Atmos Environ 24:79–89
29. Feng C, Zhao J, Qin G, Diao P (2021) Construction of the Fe3+-O-Mn3+/2+ hybrid bonds on the surface of porous silica as active centers for efficient heterogeneous catalytic ozonation. J Solid State Chem 300:1–30
30. Fenton HJH (1894) Oxidation of tartatic acid in presence of iron. J Chem Soc Trans 65:899–910
31. Gautam P, Kumar S, Lokhandwala S (2019) Advanced oxidation processes for treatment of leachate from hazardous waste landfill: a critical review. J Clean Prod 237:117639. https://doi.org/10.1016/j.jclepro.2019.117639
32. Gimeno O, Rivas FJ, Beltrán FJ, Carbajo M (2007) Photocatalytic ozonation of winery wastewaters. J Agric Food Chem 55:9944–9950. https://doi.org/10.1021/jf072167i
33. Glaze WH, Kang JW, Chapin DH (1987) The chemistry of water treatment processes involving ozone, hydrogen peroxide and ultraviolet radiation. Ozone Sci Eng 9:335–352. https://doi.org/10.1080/01919518708552148
34. Gogate PR, Pandit AB (2004) A review of imperative technologies for wastewater treatment I: oxidation technologies at ambient conditions. Adv Environ Res 8:501–551. https://doi.org/10.1016/S1093-0191(03)00032-7
35. Gogate PR, Pandit AB (2004) A review of imperative technologies for wastewater treatment II: hybrid methods. Adv Environ Res 8:553–597. https://doi.org/10.1016/S1093-0191(03)00031-5
36. Gomes AI, Soares TF, Silva TFCV et al (2020) Ozone-driven processes for mature urban landfill leachate treatment: organic matter degradation, biodegradability enhancement and treatment costs for different reactors configuration. Sci Total Environ 724. https://doi.org/10.1016/j.scitotenv.2020.138083
37. Guo S, Wang Q, Luo C et al (2020) Hydroxyl radical-based and sulfate radical-based photocatalytic advanced oxidation processes for treatment of refractory organic matter in semi-aerobic aged refuse biofilter effluent arising from treating landfill leachate. Chemosphere 243:125390. https://doi.org/10.1016/j.chemosphere.2019.125390
38. Gottschalk C, Libra JA, Saupe A (2000) Ozonation of water and waste water, a practical guide to understanding ozone and its application
39. Gurol MD, Akata A (1996) Kinetics of ozone photolysis in aqueous solution. AIChE J 42:3283–3292. https://doi.org/10.1002/aic.690421128
40. Hagman M, Heander E, Jansen JLC (2008) Advanced oxidation of refractory organics in leachate—potential methods and evaluation of biodegradability of the remaining substrate. Environ Technol 29:941–946. https://doi.org/10.1080/09593330801985057
41. Hermosilla D, Cortijo M, Huang CP (2009) Optimizing the treatment of landfill leachate by conventional Fenton and photo-Fenton processes. Sci Total Environ 407:3473–3481. https://doi.org/10.1016/j.scitotenv.2009.02.009
42. Hoigné J (1998) Quality and treatment of drinking water II
43. Hori H, Nagano Y, Murayama M et al (2012) Efficient decomposition of perfluoroether carboxylic acids in water with a combination of persulfate oxidant and ultrasonic irradiation. J Fluorine Chem 141:5–10. https://doi.org/10.1016/j.jfluchem.2012.05.012

44. Huang H, Li W (2011) Destruction of toluene by ozone-enhanced photocatalysis: performance and mechanism. Appl Catal B Environ 102:449–453. https://doi.org/10.1016/j.apcatb.2010.12.025
45. Huang C-R, Shu H-Y (1995) The reaction kinetics, decomposition pathways and intermediate formations of phenol in ozonation, UVO3 and UVH2O2 processes. J Hazard Mater 41(1):47–64. https://doi.org/10.1016/0304-3894(94)00093-V
46. Huang X, Cheng W, Quan X et al (2019) Catalytic ozonation of biologically treated leachate from municipal solid waste in a microbubble reactor. Ozone Sci Eng 41:415–426. https://doi.org/10.1080/01919512.2018.1555027
47. Issaka E, AMU-Darko JNO, Yakubu S, Fapohunda FO, Ali N, Bilal M (2022) Advanced catalytic ozonation for degradation of pharmaceutical pollutants—a review. Chemosphere 289
48. Jothinathan L, Cai QQ, Ong SL, Hu JY (2022) Fe-Mn doped powdered activated carbon pellet as ozone catalyst for cost-effective phenolic wastewater treatment: mechanism studies and phenol by-products elimination. J Hazard Mater 424:1–40
49. Kang W, Chen S, Yu H et al (2021) Photocatalytic ozonation of organic pollutants in wastewater using a flowing through reactor. J Hazard Mater 405:124277. https://doi.org/10.1016/j.jhazmat.2020.124277
50. Kang YW, Hwang KY (2000) Effects of reaction conditions on the oxidation efficiency in the Fenton process. Water Res 34:2786–2790. https://doi.org/10.1016/S0043-1354(99)00388-7
51. Kewalramani JA, Wang B, Marsh RW, Meegoda JN, Rodriguez Freire L (2022) Coupled high and low-frequency ultrasound remediation of PFAS-contaminated soils. Ultrason Sonochem 88:106063. https://doi.org/10.1016/j.ultsonch.2022.106063
52. Khan I, Saeed K, Ali N et al (2020) Heterogeneous photodegradation of industrial dyes: an insight to different mechanisms and rate affecting parameters. J Environ Chem Eng 8. https://doi.org/10.1016/j.jece.2020.104364
53. Khashij M, Mehralian M, Goodarzvand Chegini Z (2020) Degradation of acetaminophen (ACT) by ozone/persulfate oxidation process: experimental and degradation pathways. Pigment Resin Technol. https://doi.org/10.1108/PRT-11-2019-0107
54. Kidak R, Qadir MN, Aziz FH (2020) Optimization of h2 o2 and ozone-based advanced oxidation processes as pretreatment of sanitary landfill leachate. Desalin Water Treat 203:119–128. https://doi.org/10.5004/dwt.2020.26238
55. Langlais B, Reckhow DA, Brink DR (1991) Ozone in water treatment application and engineering
56. Lei Y, Hou J, Fang C et al (2023) Ultrasound-based advanced oxidation processes for landfill leachate treatment: energy consumption, influences, mechanisms and perspectives. Ecotoxicol Environ Saf 263
57. Li W, Gu K, Yu Q et al (2021) Leaching behavior and environmental risk assessment of toxic metals in municipal solid waste incineration fly ash exposed to mature landfill leachate environment. Waste Manag 120:68–75. https://doi.org/10.1016/j.wasman.2020.11.020
58. Lucas MS, Peres JA, Li Puma G (2010) Treatment of winery wastewater by ozone-based advanced oxidation processes (O3, O3/UV and O3/UV/H2O2) in a pilot-scale bubble

column reactor and process economics. Sep Purif Technol 72:235–241. https://doi.org/10.1016/j.seppur.2010.01.016

59. Lu X, Qiu W, Peng J, Xu H et al (2021) A review on additives-assisted ultrasound for organic pollutants degradation. J Hazard Mater 403:123915. https://doi.org/10.1016/j.jhazmat.2020.123915

60. Ma S, Zhou C, Pan J, Yang G, Sun C, Liu Y, Chen X, Zhao Z (2022) Leachate from municipal solid waste landfills in a global perspective: characteristics, influential factors and environmental risks. J Clean Prod 333:130234

61. Matouq M, Al-Anber Z, Susumu N et al (2014) The kinetic of dyes degradation resulted from food industry in wastewater using high frequency of ultrasound. Sep Purif Technol 135:42–47. https://doi.org/10.1016/j.seppur.2014.08.002

62. Mecha AC, Onyango MS, Ochieng A et al (2016) Synergistic effect of UV–vis and solar photocatalytic ozonation on the degradation of phenol in municipal wastewater: a comparative study. J Catal 341:116–125

63. Mehrjouei M, Müller S, Möller D (2012) Synergistic effect of the combination of immobilized TiO2, UVA and ozone on the decomposition of dichloroacetic acid. J Environ Sci Heal Part A Toxic/Hazard Subst Environ Eng 47:1073–1081. https://doi.org/10.1080/10934529.2012.668026

64. Mehrjouei M, Müller S, Möller D (2015) A review on photocatalytic ozonation used for the treatment of water and wastewater. Chem Eng J 263:209–219. https://doi.org/10.1016/j.cej.2014.10.112

65. Mena E, Rey A, Acedo B et al (2012) On ozone-photocatalysis synergism in black-light induced reactions: oxidizing species production in photocatalytic ozonation versus heterogeneous photocatalysis. Chem Eng J 204–206:131–140. https://doi.org/10.1016/j.cej.2012.07.076

66. Meng X, Liu Z, Qian X et al (2023) Enhanced degradation mechanism of anticancer drug irinotecan through low-frequency ultrasound assisted reactive electrochemical membrane. J Clean Prod 383

67. Miklos DB, Remy C, Jekel M et al (2018) Evaluation of advanced oxidation processes for water and wastewater treatment—a critical review. Water Res 139:118–131. https://doi.org/10.1016/j.watres.2018.03.042

68. Mishra N, Reddy R, Kuila A et al (2017) A review on advanced oxidation processes for effective water treatment. Curr World Environ 12:469–489. https://doi.org/10.12944/cwe.12.3.02

69. Mishra S, Sundaram B (2022) Efficacy of nanoparticles as photocatalyst in leachate treatment. Nanotechnol Environ Eng 7:173–192. https://doi.org/10.1007/s41204-021-00209-x

70. Nawaz M, Shahzad A, Tahir K, Kim J, Moztahida M, Jang J, Alam MB, Lee SH, Jung HY, Lee DS (2020) Photo-fenton reaction for the degradation of sulfamethoxazole using a multi-walled carbon nanotube-NiFe2O4 composite. Chem Eng J 382:123053

71. Neyens E, Baeyens J (2003) A review of classic Fenton's peroxidation as an advanced oxidation technique. J Hazard Mater 98:33–50. https://doi.org/10.1016/S0304-3894(02)00282-0

72. Pirkanniemi K, Sillanpää M (2002) Heterogeneous water phase catalysis as an environmental application: a review. Chemosphere 48:1047–1060. https://doi.org/10.1016/S0045-6535(02)00168-6
73. Pisharody L, Gopinath A, Malhotra M et al (2022) Occurrence of organic micropollutants in municipal landfill leachate and its effective treatment by advanced oxidation processes. Chemosphere 287:132216. https://doi.org/10.1016/j.chemosphere.2021.132216
74. Poblete R, Oller I, Maldonado MI, Cortes E (2019) Improved landfill leachate quality using ozone, UV solar radiation, hydrogen peroxide, persulfate and adsorption processes. J Environ Manage 232:45–51. https://doi.org/10.1016/j.jenvman.2018.11.030
75. Rajeswari R, Kanmani S (2009) A study on synergistic effect of photocatalytic ozonation for carbaryl degradation. Desalination 242:277–285. https://doi.org/10.1016/j.desal.2008.05.007
76. Ren T, Zhang X, Chen S et al (2022) Hydrogen peroxide and peroxymonosulfate intensifying Fe−doped NiC−Al_2O_3−framework−based catalytic ozonation for advanced treatment of landfill leachate: performance and mechanisms. Sci Total Environ 843
77. Renou S, Givaudan JG, Poulain S et al (2008) Landfill leachate treatment: review and opportunity. J Hazard Mater 150:468–493. https://doi.org/10.1016/j.jhazmat.2007.09.077
78. Rivas FJ, Beltrán F, Gimeno O et al (2003) Stabilized leachates: ozone-activated carbon treatment and kinetics. Water Res 37:4823–4834. https://doi.org/10.1016/j.watres.2003.08.007
79. Rosenfeldt EJ, Linden KG, Canonica S, von Gunten U (2006) Comparison of the efficiency of {radical dot}OH radical formation during ozonation and the advanced oxidation processes O_3/H_2O_2 and UV/H_2O_2. Water Res 40:3695–3704. https://doi.org/10.1016/j.watres.2006.09.008
80. Roudi AM, Chelliapan S, Mohtar WHMW, Kamyab H (2018) Prediction and optimization of the Fenton process for the treatment of landfill leachate using an artificial neural network. Water (Switzerland) 10. https://doi.org/10.3390/w10050595
81. Rueda-Márquez JJ, Levchuk I, Manzano M, Sillanpää M (2020) Toxicity reduction of industrial and municipal wastewater by advanced oxidation processes (Photo-fenton, UVC/H_2O_2, electro-fenton and galvanic fenton): a review. Catalysts 10. https://doi.org/10.3390/catal10060612
82. Ruppert G, Bauer R, Heisler G (1993) The photo-Fenton reaction - an effective photochemical wastewater treatment process. J Photochem Photobiol A Chem 73:75–78. https://doi.org/10.1016/1010-6030(93)80035-8
83. Saien J, Shahrezaei F (2012) Organic pollutants removal from petroleum refinery wastewater with nanotitania photocatalyst and UV light emission. Int J Photoenergy 2012:1–6. https://doi.org/10.1155/2012/703074
84. Schulte P, Bayer A, Kuhn F, Luy T (2008) H_2O_2/O_3, H_2O_2/UV and H_2O_2/Fe processes for the oxidation of hazardous wastes. Ozone Sci Eng 17:119–134. https://doi.org/10.1080/01919519508547541

85. Seibert D, Henrique Borba F, Bueno F et al (2019) Two-stage integrated system photo-electro-Fenton and biological oxidation process assessment of sanitary landfill leachate treatment: an intermediate products study. Chem Eng J 372:471–482. https://doi.org/10.1016/j.cej.2019.04.162

86. Sein MM, Zedda M, Tuerk J et al (2008) Oxidation of diclofenac with ozone in aqueous solution. Environ Sci Technol 42:6656–6662. https://doi.org/10.1021/es8008612

87. Singa PK, Isa MH, Ho YC, Lim JW (2018a) Treatment of hazardous waste landfill leachate using Fenton oxidation process. E3S Web Conf 34:4–9. https://doi.org/10.1051/e3sconf/20183402034

88. Singa PK, Isa MH, Ho YC, Lim JW (2018b) Mineralization of hazardous waste landfill leachate using photo-fenton process. E3S Web Conf 65:1–8. https://doi.org/10.1051/e3sconf/20186505012

89. Staehelln J, Holgné J (1982) Decomposition of ozone in water: rate of initiation by hydroxide ions and hydrogen peroxide. Bull Environ Contain Toxicol 16(32). https://pubs.acs.org/sharingguidelines

90. Sun Y, Pignatello JJ (1993) Photochemical reactions involved in the total mineralization of 2,4-D by Fe3+/H_2O_2/UV. Environ Sci Technol 27:304–310. https://doi.org/10.1021/es00039a010

91. Szczepanik B (2017) Photocatalytic degradation of organic contaminants over clay-TiO2 nanocomposites: a review. Appl Clay Sci 141:227–239. https://doi.org/10.1016/j.clay.2017.02.029

92. Taylor P, Rice RG (2010) Applications of ozone for industrial wastewater treatment—a review. Ozone Sci Eng 18:477–515

93. Tizaoui C, Bouselmi L, Mansouri L, Ghrabi A (2007) Landfill leachate treatment with ozone and ozone/hydrogen peroxide systems. J Hazard Mater 140:316–324. https://doi.org/10.1016/j.jhazmat.2006.09.023

94. Torkashvand J, Rezaei Kalantary R, Heidari N et al (2021) Application of ultrasound irradiation in landfill leachate treatment. Environ Sci Pollut Res 28(35):47741–47751. https://doi.org/10.1007/s11356-021-15280-9

95. Usman M, Cheema SA, Farooq M (2020) Heterogeneous Fenton and persulfate oxidation for treatment of landfill leachate: a review supplement. J Clean Prod 256. https://doi.org/10.1016/j.jclepro.2020.120448

96. van Aken P, Lambert N, Degrève J et al (2011) Comparison of different oxidation methods for recalcitrance removal of landfill leachate. Ozone Sci Eng 33:294–300. https://doi.org/10.1080/01919512.2011.583170

97. Wang HW, Li Xy, Hao ZP et al (2017) Transformation of dissolved organic matter in concentrated leachate from nanofiltration during ozone-based oxidation processes (O_3, O_3/H_2O_2 and O_3/UV). J Environ Manage 191:244–251. https://doi.org/10.1016/j.jenvman.2017.01.021

98. Wdowczyk A, Szymańska-Pulikowska A (2021) Analysis of the possibility of conducting a comprehensive assessment of landfill leachate contamination using physicochemical indicators and toxicity test. Ecotoxicol Environ Saf 221. https://doi.org/10.1016/j.ecoenv.2021.112434

99. Wei Z, Liu J, Shangguan W (2020) A review on photocatalysis in antibiotic wastewater: pollutant degradation and hydrogen production. Chin J Catal 41:1440–1450. https://doi.org/10.1016/S1872-2067(19)63448-0
100. Wu C, Chen W, Gu Z, Li Q (2021) A review of the characteristics of Fenton and ozonation systems in landfill leachate treatment. Sci Total Environ 762. https://doi.org/10.1016/j.scitotenv.2020.143131
101. Wu JJ, Wu CC, Ma HW, Chang CC (2004) Treatment of landfill leachate by ozone-based advanced oxidation processes. Chemosphere 54:997–1003. https://doi.org/10.1016/j.chemosphere.2003.10.006
102. Ye M, Chen Z, Liu X et al (2009) Ozone enhanced activity of aqueous titanium dioxide suspensions for photodegradation of 4-chloronitrobenzene. J Hazard Mater 167:1021–1027. https://doi.org/10.1016/j.jhazmat.2009.01.091
103. Zhang H, Choi HJ, Huang CP (2006) Treatment of landfill leachate by Fenton's reagent in a continuous stirred tank reactor. J Hazard Mater 136:618–623. https://doi.org/10.1016/j.jhazmat.2005.12.040
104. Zhang ML, Fan YT, Xing Y et al (2007) Enhanced biohydrogen production from cornstalk wastes with acidification pretreatment by mixed anaerobic cultures. Biomass Bioenerg 31:250–254
105. Zhang H, Heung JC, Huang CP (2005) Optimization of Fenton process for the treatment of landfill leachate. J Hazard Mater 125:166–174. https://doi.org/10.1016/j.jhazmat.2005.05.025
106. Zhou Q, Li W, Hua T (2010) Removal of organic matter from landfill leachate by advanced oxidation processes: a review. Int J Chem Eng 2010. https://doi.org/10.1155/2010/270532

© The Author(s), under exclusive license to Springer Nature Switzerland AG 2024
S. Souabi, A. Anouzla (eds.), *Landfill Leachate Treatment Techniques*, Springer Water
https://doi.org/10.1007/978-3-031-63157-3_11

11. Statistical Evaluation of the Effectiveness of Electrocoagulation Using Iron Electrodes for the Treatment of Leachate Effluent

Rosangela Bergamasco[1] ✉, Mayara Caroline Santos Godoi[2], Joseane Debora Peruco Theodoro[2] ✉ and Paulo Sergio Theodoro[3]
(1) Universidade Estadual de Maringá, Maringá, Brazil
(2) Universidade Tecnológica Federal Do Paraná, Londrina, Brazil
(3) Universidade Estadual Do Oeste Do Paraná Toledo, Toledo, Brazil

✉ **Rosangela Bergamasco**
Email: rosangela@deq.uem.br

✉ **Joseane Debora Peruco Theodoro (Corresponding author)**
Email: jtheodoro@utfpr.edu.br

Abstract
Electrocoagulation is a technique that utilizes electric current to induce oxidation–reduction reactions, forming

hydroxides that act as coagulating agents for contaminant removal. In this study, electrocoagulation was applied to treat leachate effluent. Leachate contains high organic and inorganic loads and is the liquid phase formed by water percolating through decomposing waste accumulated in a landfill. This study aimed to evaluate the effectiveness of electrocoagulation in removing certain pollutants from leachate effluent. A statistical experimental design was developed, employing a 3^2 factorial design in duplicate. The independent variables were treatment time (min.) and initial pH, while the dependent variables were apparent color (uH), turbidity (uT), final pH, and conductivity (mS). At the end of the treatments, the results indicated an 89% reduction in turbidity and a 99% reduction in apparent color. The conductivity remained around 20 mS, and the final pH was slightly basic. Based on the presented results, it can be observed that electrocoagulation shows potential as an effective technique for treating leachate effluent.

Keywords Iron electrode – Wastewater – Occupational safety – Experimental design

---

11.1 Introduction

The demand for sanitary landfills has been growing along with population growth, and with it comes the issue of leachate generation and the necessary precautions to prevent soil and natural water contamination, whether underground, in rivers, or lakes.

The production and generation of solid waste are directly linked to population growth. Brazil alone has more than 200 million inhabitants and a modern consumerist mindset constantly demanding consumer goods [7].

In Brazil, which has a population of approximately 200 million inhabitants, 79 million tons of waste are produced annually [2]. Organic waste is commonly

disposed of in sanitary landfills and dumpsites, causing direct and indirect environmental damage through soil contamination and groundwater pollution by leachate, affecting nearby residents or using the contaminated source for consumption [8].

According to [6], the solution to the final disposal of urban solid waste, which is the sanitary landfill, has turned from a solution into a problem due to the contamination that can occur if there is no effective management of the waste generated by leachate formation.

Leachate is a dark-colored liquid waste with a high organic and inorganic load. It is one of the important factors to be studied and monitored in operational sanitary landfills or during project development. Additionally, leachate contains high concentrations of suspended solids, organic and inorganic compounds, and fats [1]. Leachate can be defined as the liquid phase of the landfill mass, originating from the percolation of water through the waste, carrying dissolved or suspended materials. The water comes from sources such as rainfall, groundwater, urban drainage, springs, and the decomposition of waste [3]. The composition of the waste is related to the type of stored waste, applied or manipulated methods, exposure time, and climate (precipitation, temperature).

Therefore, it is necessary to treat leachate in a way that reduces environmental impacts on water bodies, fauna, and flora, as well as promotes the preservation of the health of residents in areas near landfills. Among the treatment methods currently available, we highlight the electrocoagulation method. Electrocoagulation requires an electric current and sacrificial metallic electrodes that release metallic ions, forming compounds with coagulant properties [4]. Coagulant agents destabilize contaminants and promote the formation of quickly settling flocs. Among the advantages of this alternative are reduced operating

costs, easy-to-operate equipment, no need for the addition of chemical components, and low sludge generation [5].

Thus, the objective of this work is to apply the electrocoagulation process to treat leachate using iron electrodes to evaluate the removal efficiency of apparent color, turbidity, and chemical oxygen demand parameters and monitor pH and electrical conductivity parameters.

11.2 Methodology

11.2.1 Description of the Study Site

The collection of leachate samples was conducted at a company that receives and provides the final disposal of urban and industrial solid waste. This company is responsible for waste collection in the Northern Region of the State of Paraná–Brazil. Figure 11.1 shows the leachate pond where sampling was carried out.



Fig. 11.1 Leachate pond where the sampling was conducted

The tests were performed to obtain data related to the parameters of apparent color (uH), turbidity (uT), pH, electrical conductivity (mS), and electric current (μA), and these procedures were conducted in the laboratory located at the Federal Technological University of Paraná–Campus Londrina. A laboratory-scale module was used for the electrocoagulation tests, as depicted in the schematic diagram shown in Fig. 11.2.

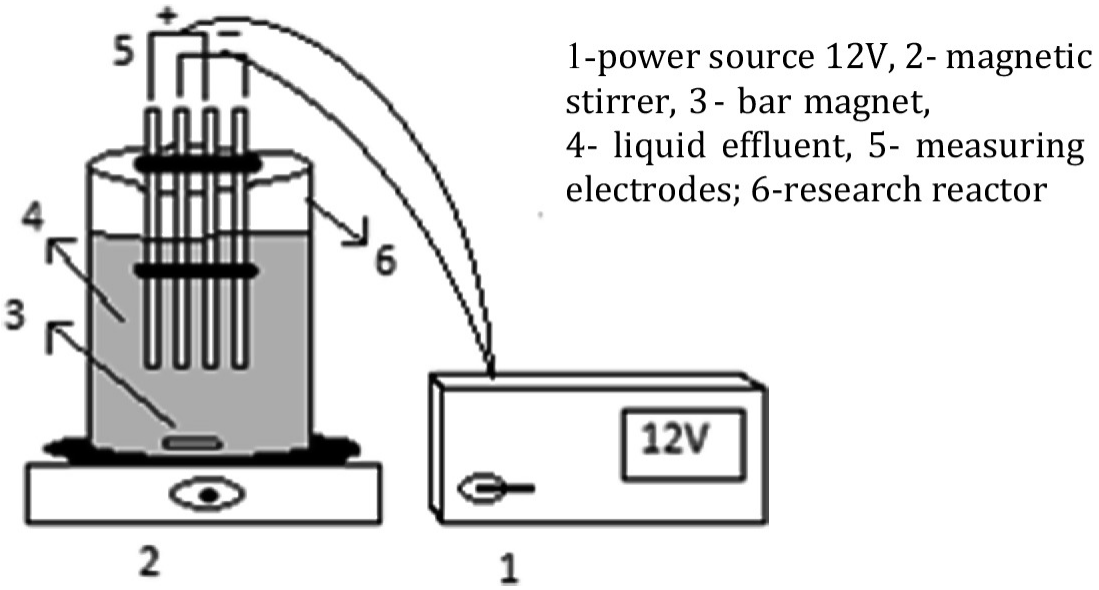


Fig. 11.2 Schematic of the module used to carry out the tests, electrocoagulation reactor at bench level. *Source* [9]

In the leachate treatments, samples were collected at the following times: 40, 60, and 80 min. The electrocoagulation tests were carried out using a factorial experimental design with three levels (−1, 0, 1). The experiments consisted of 3^2 tests with replicates, totaling 18 tests, with the independent variables being the initial pH and the electrocoagulation time. Table 11.1 presents the matrix of the statistical design applied to the leachate treatment using electrocoagulation.

Table 11.1 Experimental planning

Test	Time (min)	Initial pH
1	40	3
2	60	3
3	80	3
4	40	6
5	60	6
6	80	6
7	40	9
8	60	9
9	80	9
10	40	3
11	60	3
12	80	3
13	40	6
14	60	6
15	80	6
16	40	9
17	60	9
18	80	9

To assess electrode wear and energy consumption, calculations were performed to determine the feasibility of the experiment. The calculation of electrode wear for the tests is determined according to Eq. (11.1), while energy consumption was obtained using Eq. (11.2).

$$Mel = i.t.M/F.n \tag{11.1}$$

where:

m_e is the amount of dissolved electrode mass (g);i is the current (A);t is the processing time (s);M is the molar mass of the predominant electrode element(g.mol^{-1});n is the number of electrons in the anode electrode oxidation reaction; and$F$ is Faraday's constant (F = 9,65 × 10^4 C.mol^{-1}).

$$C = U.i.t.v \tag{11.2}$$

where: C = energy consumption ($Wh.m^{-3}$);U = applied electrical voltage (V);i = electric current (A);t = current application time or process time (h);v = volume of treated effluent (m^3).

In this way, the system's operating cost is calculated using Eq. (11.3).

$$Coper = a.Cenerg + b.Celet \tag{11.3}$$

where: C_{oper} = total system operating cost ($R\$.m^{-3}$);a = electrical energy cost ($R\$.kWh^{-1}$);$C_{energ}$ = electrical energy consumption ($kWh.m^{-3}$);b = cost of electrode (R$.kg aluminum^{-1});$C_{elet}$ = electrode consumption ($kg.m^{-3}$ reated water).

11.3 Results and Discussions

11.3.1 Effluent Characterization Before Treatment

Table 11.2 presents the values of the evaluated parameters of the leachate before treatment and the values of the ideal parameters regulated by environmental law and applied by CONAMA 357/2005.

Table 11.2 Gross values and maximum values allowed by CONAMA resolution 357/2005

Parameters	Gross	Maximum values allowed (VMP)
True color (uH)	3336	75 uH
Turbidity (uT)	445	100 uT
pH	8,32	6.0 a 9.0
Electric conductivity (mS)	19,54	–
DQO($mg.L^{-1}$)	4952	–

It can be observed that the values of the raw parameters of the leachate wastewater are in disagreement with the standards established by the Brazilian standard defined by the NATIONAL ENVIRONMENT COUNCIL by CONAMA Resolution 357/05, which provides for the classification of bodies of water and guidelines environmental issues for its framework, as well as establishing the conditions and standards for releasing effluents and providing other measures. It shows that it is necessary to treat wastewater before the company releases it into the receiving body.

11.3.2 Result of the Evaluated Parameters

The results obtained after treating wastewater (slurry) by applying electrocoagulation using iron electrodes are presented in Table 11.3, and the response surfaces that show the behavior of each response variable as a function of the independent variables.

Table 11.3 Results obtained in the tests

	% removal				Follow-up
Tests	**DQO**	**Apparent Color**	**Turbidity**	**pH**	**Electric conductivity mS**
1	96	95	74	5.7	20
2	95	92	63	5.99	20
3	92	90	65	6.02	20
4	93	91	84	9.25	20
5	92	92	83	9.31	18.35
6	94	93	88	9.79	17.17
7	99	98	70	9.74	16.8
8	94	92	55	10.03	15.8
9	91	93	84	10.19	17.02
10	98	99	85	5.83	20
11	98	98	69	6.07	20
12	99	96	76	6.26	20
13	92	90	68	8.85	16.37

	% removal				Follow-up
Tests	**DQO**	**Apparent Color**	**Turbidity**	**pH**	**Electric conductivity mS**
14	97	96	88	9.11	15.8
15	95	94	89	9.21	15.92
16	92	91	58	9.78	16.74
17	95	93	84	10.06	17.87
18	97	96	80	10.06	18.2

11.3.3 Analysis of the Response Surface of the Dependent Variables

The removal behavior generated by the mathematical models can be visualized through the response surfaces presented in Figs. 11.3, 11.4, 11.5, 11.6 and 11.7, as a function of the independent variables treatment time (min.) and initial pH.

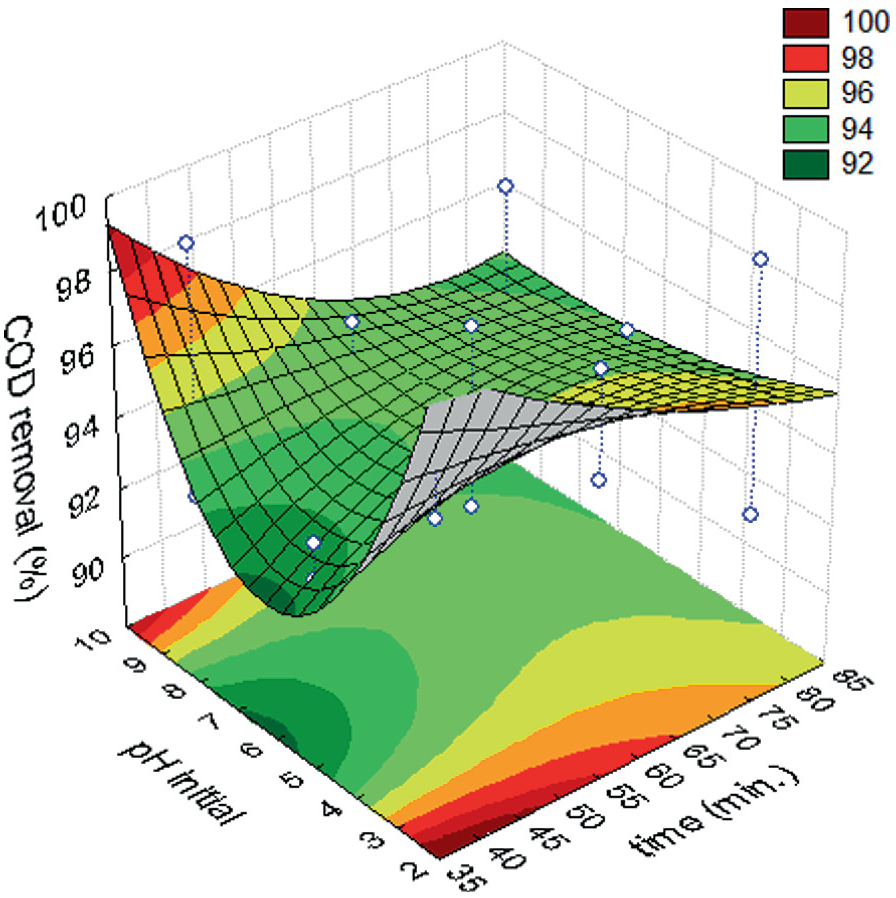


Fig. 11.3 COD removal response surface

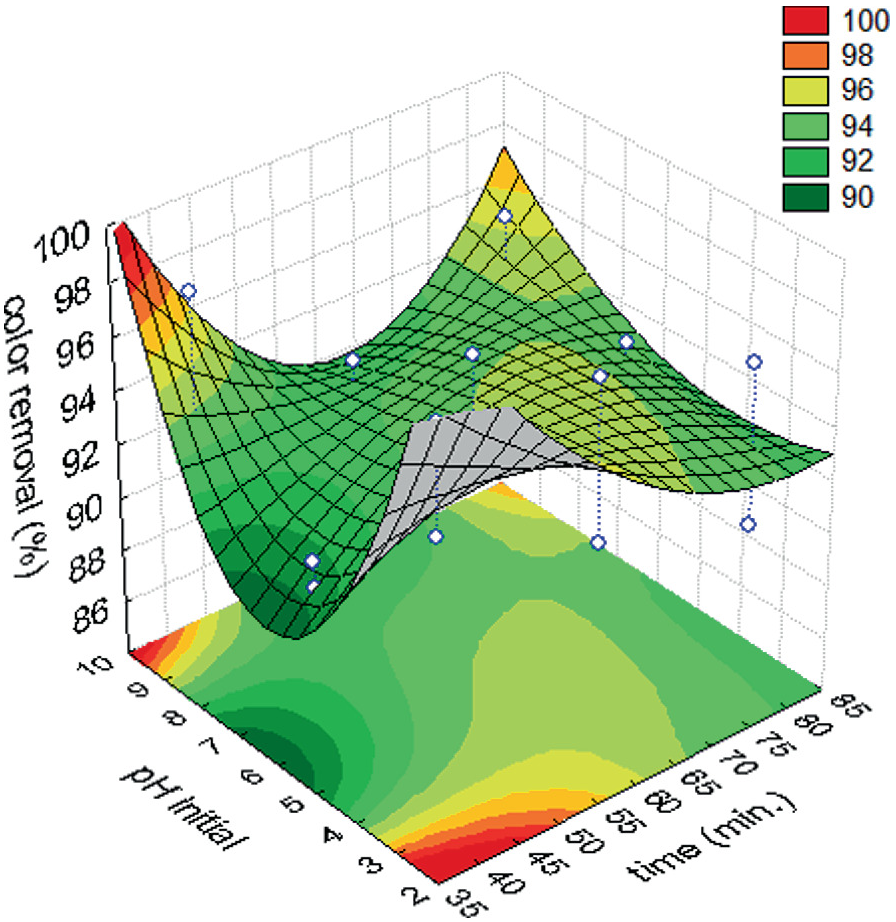


Fig. 11.4 Apparent color removal response surface

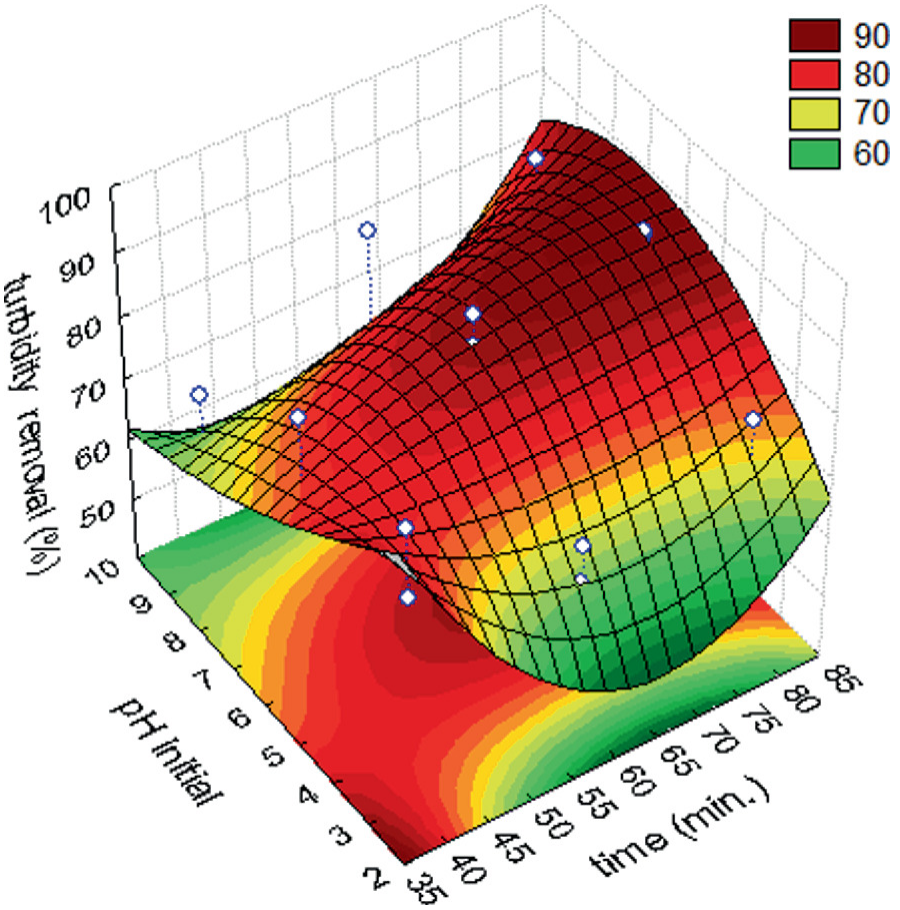


Fig. 11.5 Response surface for removing the turbidity parameter

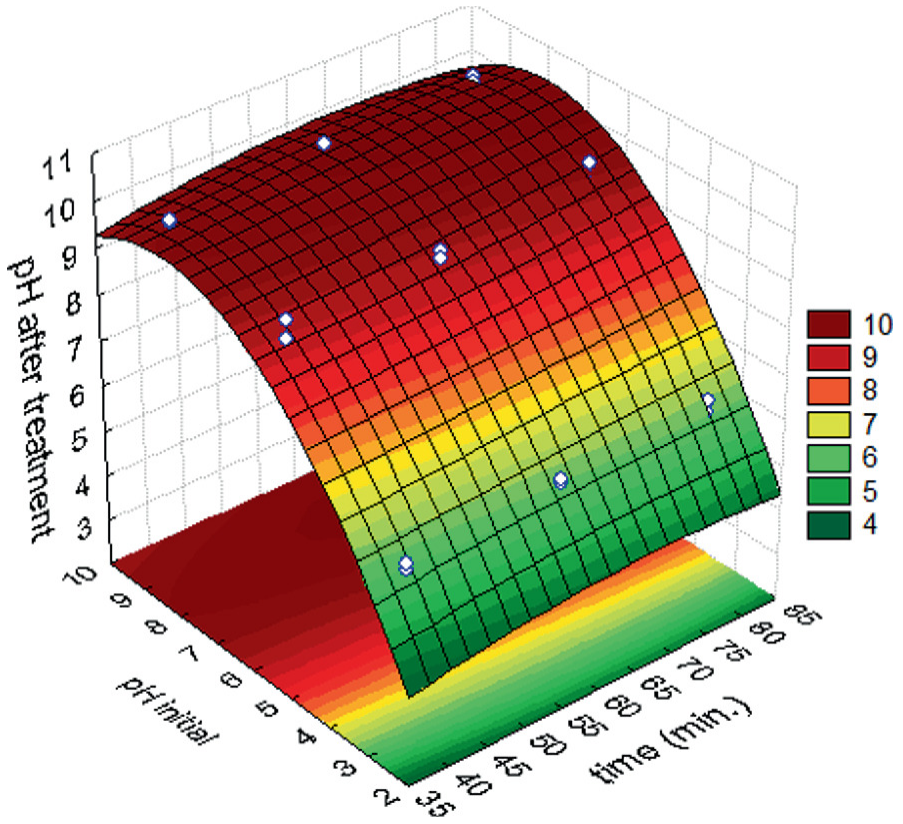


Fig. 11.6 Superfície de resposta do pH após tratamento

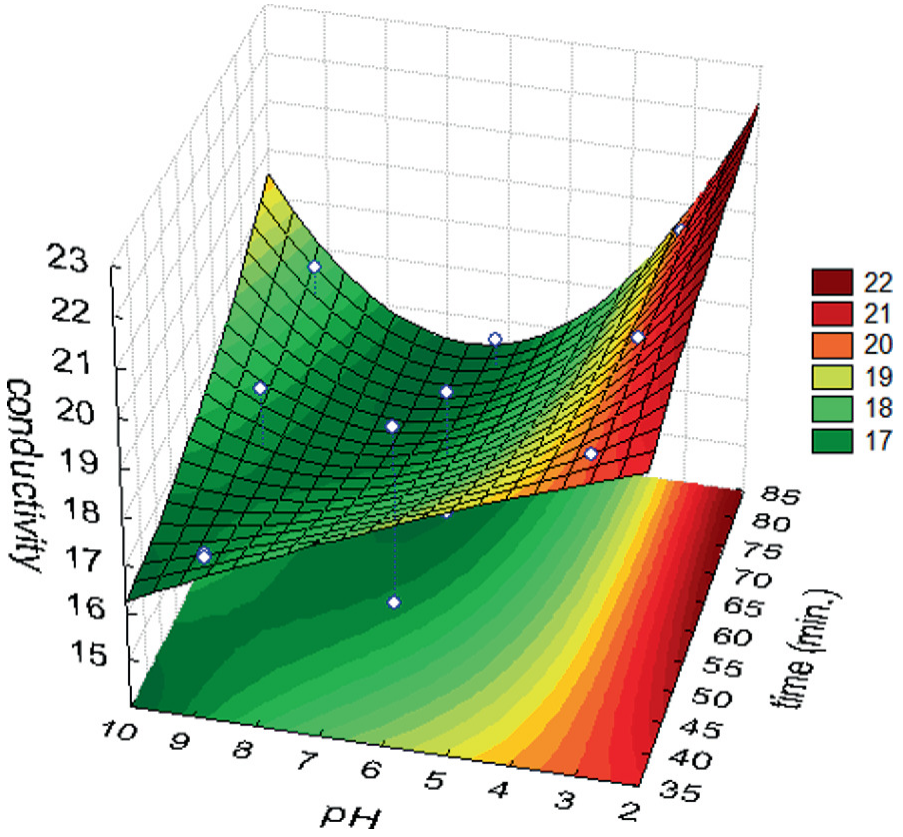


Fig. 11.7 Response surface of electrical conductivity behavior

According to the response surface presented in Fig. 11.3, it is observed that the percentage of removal of chemical oxygen demand (COD) showed greater significance for initial pH at acidic values below 3 and at basic values above 9 for storage time. Treatment takes around 50 min.

Figure 11.4 shows the response surface of the apparent color removal percentage. According to this surface, it is observed that the color was removed in a more significant percentage for initial pH at acid values below 3 and at basic values above 9.5 for a treatment time of around 40 min.

The response surface presented in Fig. 11.5 shows that the percentage of turbidity removal was maximized by:

– Initial pH varying from 2 to 4 for a treatment time of up to 40 min;

– Initial pH around 6 for a treatment time ranging from 50 to 85 min;
– Initial pH above 9 for treatment time above 80 min.

Figure 11.6 shows the response surface of pH behavior after electrocoagulation treatments. It is observed that the final pH is not influenced by the treatment time, it follows the profile of the initial pH variation

According to this response surface presented in Fig. 11.7, it can be seen that the conductivity values (mS) are minimized at pH above 9 and for treatment times below 40 min. With the commitment to comply with environmental laws, the conductivity value should be as low as possible.

11.3.4 Assessment of Electrocoagulation Operating Parameters.

In this stage of evaluating the operating parameters of the electrocoagulation process, the wear of the iron electrode in each test, the energy consumption, and also the operating cost, i.e., the amount invested in the treatment, are presented. Table 11.4 presents the values of the parameters that were considered essential for calculating the financial costs of each test performed in the experiment.

Table 11.4 Electrode wear, Energy consumption and cost of treatment operation

pH	Test	Electrode wear (mg)	Energy consumption (kWh)	Operation cost (US$)
Acid	1 (40 min)	1.70	0.087	1.44
Acid	2 (60 min)	2.36	0.121	2.41
Acid	3 (80 min)	2.56	0.131	2.32
Neutral	4 (40 min)	1.66	0.085	1.51
Neutral	5 (60 min)	1.99	0.102	1.81
Neutral	6 (80 min)	2.66	0.136	2.41

pH	Test	Electrode wear (mg)	Energy consumption (kWh)	Operation cost (US$)
Basic	7 (40 min)	0.97	0.050	0.88
Basic	8 (60 min)	1.42	0.073	1.29
Basic	9 (80 min)	1.44	0.074	1.31
Acid	10 (40 min)	1.86	0.095	1.68
Acid	11 (60 min)	2.68	0.137	2.43
Acid	12 (80 min)	3.36	0.172	3.05
Neutral	13 (40 min)	0.75	0.038	0.67
Neutral	14 (60 min)	1.14	0.058	1.03
Neutral	15 (80 min)	1.57	0.080	1.42
Basic	16 (40 min)	0.91	0.047	0.82
Basic	17(60 min)	1.49	0.076	1.35
Basic	18 (80 min)	1.65	0.085	1.50

According to Table 11.4, it can be seen that the most significant parameter, that is, the one that had the most influence on the cost of the leachate treatment operation using electrocoagulation with iron electrodes, was the treatment time.

11.4 Conclusion

Using the response surfaces generated by statistical analyses, it was possible to observe the behavior of the dependent variables as a function of the independent variables. According to the results obtained, the removal percentage of the parameters apparent color, turbidity, and chemical oxygen demand were respectively close to 100,

89, and 98%. The pH values after treatment were influenced by the initial pH value. Conductivity (µS/cm) did not undergo significant variations. The costs of using electrocoagulation to treat leachate were significantly low, depending on the parameters evaluated, the investment value for all tests was in the range of 0.67–3.05 US$.

References

1. Celere MS, Oliveira AS, Trevilato TMB, Muñoz SIS (2007) Metals present in leachate collected at the ribeirão preto landfill, São Paulo, Brazil, and their relevance for public health. Cad. Saúde Pública, Rio de Janeiro 23(4):939–947
2. Dutra NF, Souza FR, Gontijo TS, Rodrigues AC, Andrade ICM (2018) The impact of national solid waste policy on scientific publications on reverse logistics. Braz J Prod Eng São Mateus. Publisher CEUNES/DETEC 4(1):66–82
3. Hamada J (1997) Estimates of generation and characterization of lease in sanitary landfills. Abes–Brazilian Association of sanitary and environmental engineering, Bauru, p 1801–1810
4. Mella B (2013) Removal of chromium from tanning waste baths through chemical precipitation and electrocoagulation. In: 2013. 106f. Thesis (Master's degree. Federal University of Rio Grande do Sul, Porto Alegre
5. Meneses JM, Vasconcelos RF, Fernandes TF, Araújo GT (2012) Treatment of biodiesel effluent using electrocoagulation/flotation: investigation of operational parameters. Quim Nova 35(2):235–240
6. Moraes CL et al (2017) Assessment of the physical-chemical and biological parameters of water and leachate in the area of influence of the deactivated landfill in the municipality of Lages-SC. Environ Manag Sustain Mag Florianópolis 6(3):149–164
7. Sabiá RJ, Silva DL, Barros GDT, Santos YTC, Junior FAVS, Lima AFO (2015) Contamination of the salt river basin by the influence of leachate and possible treatment through biosorbents. Cult Sci Notebook 14(1)
8. Santos DCB (2013) Mechanized bioprocessor prototype for processing organic waste by the composting method with fertilizer production in a reduced space and draining the resulting manure. In: 2013. 77f. Completion of course work. Federal Institute of Education, Science and Technology Fluminense, Campos dos Goytacazes-Rio de Janeiro

9. Theodoro JDP, Ferrarez CNM, Iwamura VC, Pereira ER, Theodoro PS (2017) Water treatment through the electrocoagulation process associated with the use of an organic coagulant (Tanfloc Sg). In: Eutrophication causes, mechanisms and ecological effects. Editora Nova Science Publishers, New York. pp 141–153

© The Author(s), under exclusive license to Springer Nature Switzerland AG 2024
S. Souabi, A. Anouzla (eds.), *Landfill Leachate Treatment Techniques*, Springer Water
https://doi.org/10.1007/978-3-031-63157-3_12

12. Electrochemical Processes for Stabilized Landfill Leachate Treatment: A Brief Review

Arseto Yekti Bagastyo[1, 2] ✉ and Anita Dwi Anggrainy[2]
(1) Department of Environmental Engineering, Faculty of Civil, Planning, and Geo Engineering, Institut Teknologi Sepuluh Nopember, Surabaya, Indonesia, 60111
(2) Research Center for Infrastructure and Sustainable Environment, Institut Teknologi Sepuluh Nopember, Surabaya, Indonesia, 60111

✉ **Arseto Yekti Bagastyo**
Email: bagastyo@enviro.its.ac.id

Abstract
Stabilized leachate from the old landfill has caused major concern due to its toxic nature and poor biodegradability. Although their composition may vary in each region, the characteristics are similar, e.g., pH above 7, BOD_5/COD ratio below 0.2, high ammoniacal nitrogen concentration, etc. In the last two decades, many studies have been conducted to search for the best alternative technology to treat this type of wastewater. The electrochemical process has gained more interest due to its effectiveness, simplicity, versatility, easy automation, less chemical consumption, and low sludge production. This technology typically utilizes a compact reactor; thus, it is suitable for a landfill with a space shortage. Moreover, integrating the biological process with electrochemical treatment can be a better option to increase treatment efficiency and minimize operational costs from energy consumption. This chapter briefly describes the electrochemical treatment, which focuses on the electrocoagulation, electro-Fenton, and electrochemical oxidation processes. The process mechanisms and crucial factors that may affect their performances are explained. Their performance and application were compiled as a single treatment, pre-treatment, or post-treatment. The advantages and limitations of each treatment were also summarized. The objective is to provide a useful reference for researchers to design a more effective and efficient treatment and help decision-makers select the most appropriate technology based on the process efficacy and cost-effectiveness.

Keywords Electrochemical oxidation – Electrocoagulation – Electrode materials – Electro-Fenton – Stabilized leachate

List of Abbreviations

BDD Boron-doped diamond
BOD Biological oxygen demand
COD Chemical oxygen demand
DOC Dissolved organic carbon
EC Electrocoagulation
EF Electro-Fenton
EO Electrochemical oxidation
TN Total nitrogen

TOC Total organic carbon
TSS Total suspended solid
TP Total phosphate

12.1 Introduction

12.1.1 Characteristics of Landfill Leachate

All waste disposal practices, i.e., open dumpsites, controlled dumps, and sanitary landfills [127], inevitably generated a dark liquid from waste decomposition mixed with moisture (i.e., rainfall, run-off, and irrigation) percolated through solid waste. This effluent (leachate) production is estimated to be between 0.2 and 1.0 m^3 per ton of waste landfilled [19, 28] and continues for many years, even after landfill closure [128]. Leachate contains high concentrations of pollutants, such as organic compounds, ammonium-nitrogen, heavy metals, chlorinated-organic compounds, inorganic salts, refractory organics, and xenobiotic compounds [36, 147]. Due to its composition and ability to infiltrate the soil and contaminate groundwater and surface water, proper leachate control and treatment are mandatory to minimize detrimental effects on public health and the environment [57, 115].

Leachate compositions vary considerably from place to place, attributable to waste types and composition, landfill age, waste degradation stage, climate and rainfall patterns, hydrogeology conditions, etc. [82, 126]. Leachate generated from young landfills (up to 5 years old) typically has a low pH (less than 6.5) and high biodegradability [50, 71]. However, as the landfill ages, the pH of leachate gradually increases while its biodegradability rapidly decreases. When it reaches the mature or stabilized stage (landfill age > 10 years), leachate generally has pH > 7.5, a low BOD_5/COD ratio (< 0.1), high ammonium nitrogen concentration (over 400 mg L^{-1}), and a high amount of recalcitrant organic matter [50]. In this stage, single biological processes are ineffective due to leachate's poor biodegradability. Table 12.1 describes the characteristics of stabilized leachate from several landfills worldwide.

Table 12.1 Characteristics of stabilized leachate from various landfills

Location	Operation	pH	EC	COD	BOD_5	BOD_5/COD	NH_4^+-N	NH_3-N	NO_3^--N	TN (as TKN)	PO_4^{3-} (as TP)	Cl^-
			(mS cm^{-1})	(mg L^{-1})	(mg L^{-1})		(mg L^{-1})	(mg L^{-1})	(mg L^{-1})	(mg L^{-1})	(mg L^{-1})	(mg L^{-1}
Open dump												
Veneto, Italy	> 25 years	8.56	n.a	n.a	400	n.a	1700	n.a	< 1	(2060)	(9.6)	n.a
Harbil, Morocco	Since 1966	7.9	20.04	1398.55	99.33	0.07	224.33	n.a	n.a	(597.89)	2.56	n.a
Ariyamangalam, India	> 50 years	8.8	14.18	2880	65	0.02	n.a	27.67	n.a	3020	58.5	242
Ranchi, India	> 20 years	7.8	8.9	4000	70	0.02	n.a	238.82	30.26	n.a	61	n.a
Chennai, India	> 15 years	6.61	n.a	2240	80	0.04	n.a	1888.35	19.51	2380	n.a	429
Vientiane, Laos	Since 2008	7.6	8.6	770	170	0.22	n.a	n.a	n.a	n.a	n.a	650
Perak, Malaysia	> 20 years	8.12	15.93	4202	181	0.05	n.a	1360	n.a	n.a	n.a	n.a
Landfill (controlled dump and sanitary landfill)												
Jebel Chakir, Tunisia	Since 1999	7.37	45	24,000	2160	0.09	269.5	n.a	13.05	n.a	n.a	n.a

Location	Operation	pH	EC	COD	BOD_5	BOD_5/COD	NH_4^+-N	NH_3-N	NO_3^--N	TN (as TKN)	PO_4^{3-} (as TP)	Cl^-
			(mS cm^{-1})	(mg L^{-1})	(mg L^{-1})		(mg L^{-1})	(mg L^{-1})	(mg L^{-1})	(mg L^{-1})	(mg L^{-1})	(mg L^{-1}
Fez City, Morocco	Since 2004	n.a	n.a	5198	500	0.09	n.a	n.a	8.13	n.a	n.a	684
Zdounsky, Czech Rep	Since 1995	8.4	10.37	364	n.a	n.a	317.8	n.a	n.a	n.a	n.a	n.a
Łubna-Poland	Since 1978	7.8	13.32	2570	514	0.20	408.6	n.a	n.a	n.a	n.a	n.a
Northeast-Poland	> 16 years	8.12	n.a	788	53	0.07	n.a	359	n.a	(418)	(59)	n.a
Northwest-Poland	Since 1983	7.84	12.17	2180	135	0.06	400	n.a	n.a	450	n.a	234
Komürcüoda, Turkey	> 25 years	7.44	21.8	13,000	n.a	n.a	n.a	22.18	60	1020 (950)	n.a	586
Heimifeng, China	Since 2003	8.57	n.a	2033	150.7	0.07	n.a	2556	n.a	n.a	n.a	n.a
Laogang, China	Since 2005	9.11	n.a	4530.21	n.a	n.a	2980.69	n.a	60.12	32,000	(75.35)	n.a
Chongqing, China	Since 2003	8.2	n.a	3676	570	0.16	n.a	1682	n.a	1824	(14)	n.a
Sudokwon, South Korea	Since 1992	8.2	10.2	536	n.a	n.a	n.a	n.a	n.a	1240.7	39.3	n.a
Deir El Balah, Palestine	Since 1996	8.42	40.8	20,233	1821	0.09	n.a	2040.70	813.35	n.a	n.a	695
Dompe, Srilanka	Since 2014	7.9	1.97	2443	43.03	0.02	n.a	n.a	3.75	62	5.8	n.a
Phnom Penh, Cambodia	Since 2009	8.1	38	2900	n.a	n.a	n.a	757.65	n.a	n.a	n.a	325
Nonthaburi, Thailand	Since 2004	7.8	35.2	3380	760	0.22	n.a	872.94	n.a	(1190)	n.a	441
Nam Son, Vietnam	Since 1999	7.6	n.d	2760	232	0.08	968.33	n.a	288.23	n.a	13.63	n.a
Jeram, Malaysia	Since 2007	8.1	25.98	3583.33	834.27	n.a	n.a	1394.51	12.13	n.a	62.33	n.a
Sahom, Malaysia	Since 1992	8.3	18	1188	89	0.07	n.a	313	n.a	n.a	n.a	n.a
Pulau Burung, Malaysia	Since 1991	8.4	22.36	1819	181	0.10	n.a	1627	20.33	n.a	(28.58)	994
Kuala Sepetang, Malaysia	> 16 years	8.7	9.68	1456	257	0.18	n.a	564	9.83	n.a	(44.63)	899
Gresik, Indonesia	Since 2002	7.8	n.a	5426.9	n.a	n.a	1532.81	n.a	30.67	n.a	132	197
Parana, Brazil	Since 2011	7.9	7.94	4212	923	0.22	1648	n.a	n.a	n.a	n.a	186

Data is on average; n.a: data unavailable

12.1.2 Physicochemical Treatment of Stabilized Landfill Leachate

Physicochemical processes have been frequently used to increase the treatment efficiency of stabilized landfill leachate. These processes may include coagulation-flocculation [24, 65], adsorption [156], ion-exchange [18, 147], advanced oxidation process (AOP) [140]

and membrane-based process [73, 111]. Despite their effectiveness, these processes have drawbacks, such as high chemical requirements, high sludge production, membrane fouling, high energy consumption, and so on [156].

12.1.3 Electrochemical Process for Stabilized Leachate

The electrochemical process is one of the physicochemical treatments that offer more advantages in terms of versatility, simplicity, easy automation, short operation time, high energy efficiency, fewer chemical requirements, and low sludge production [54, 121, 153]. This technology has gained more interest in recent decades due to its high effectiveness in degrading organic materials, ammonium nitrogen, hazardous materials, and bio-refractory organic compounds [41, 146]. Nonetheless, a single unit of electrochemical process for leachate treatment may require high energy consumption and higher operational costs; thus, integrating the electrochemical process with other treatment systems may become a better solution to minimize the operational cost and achieve more efficient treatment [114].

To date, electrochemical processes such as electrocoagulation (EC), electro-Fenton (EF), and electrochemical oxidation (EO) have been widely studied for the treatment of landfill leachate [4, 55, 74]. The number of publications is shown in Fig. 12.1. The literature search is performed by searching in the ScienceDirect database using queries TITLE-ABS-KEY (“Terms”). Between 2000 and 2023, there were only 8 of 206 publications that mentioned specific terms of “stabilized leachate”, “mature leachate”, or “old leachate” in their title or keywords. But it does not mean the other publications did not study those terms.

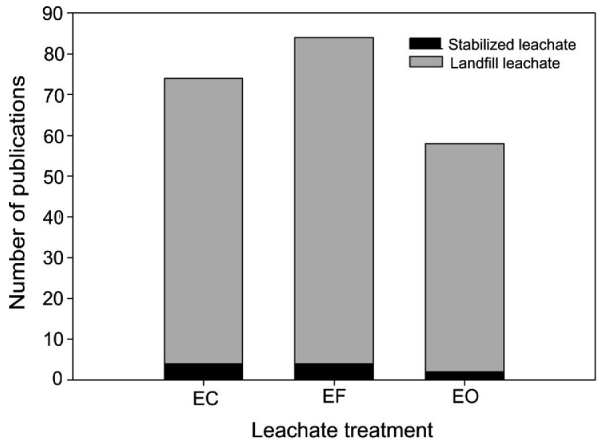


Fig. 12.1 The number of publications related to the electrochemical process for leachate treatment

This chapter briefly describes the mechanisms of those electrochemical-based processes for pollutant removal. Several factors that may influence the performance efficacy were detailed. The advantages and disadvantages of electrode materials that are crucial for these processes were also explained. Additionally, the performance of each treatment was compiled based on its application (as a single treatment, pre-treatment, or post-treatment). At the end of this chapter, we summarized the advantages and limitations of each treatment and the challenges that may occur during the process. Future perspectives were also added to give some outlook for the process improvement.

12.2 Electrocoagulation

12.2.1 Principles and Mechanism of Electrocoagulation

Electrocoagulation (EC) is an electrochemical-based process that has been proven effective in removing color, suspended solids, organic matter, heavy metals, etc. This

process relies on the generation of coagulant from the in-situ electrochemical reduction of the metal anode and the formation of flocs and gas bubbles that enhance the floatation of pollutants [30]. During electrolysis, the electrochemical reaction suddenly occurs after the anode and cathode are connected to an electric current. The mechanisms are as follows [62, 64]:

Anodic reaction:

$$M_{(s)} \rightarrow M^{Z+}_{(aq)} + Ze^{-} \tag{12.1}$$

Cathodic reaction:

$$2H^{+}_{(aq)} + 2e^{-} \rightarrow H_{2(g)} \quad \text{(acidic environment)} \tag{12.2}$$

$$2H_2O_{(l)} + 2e^{-} \rightarrow 2OH^{-}_{(aq)} + H_{2(g)} \quad \text{(neutral or alkaline environment)} \tag{12.3}$$

Overall reaction in the bulk solution:

$$M^{Z+}_{(aq)} + zH_2O_{(aq)} \rightarrow MOH_{z(s)} + zH^{+}_{(aq)} \quad \text{(acidic environment)} \tag{12.4}$$

$$M^{Z+}_{(aq)} + zOH^{-}_{(aq)} \rightarrow MOH_{z(s)} \quad \text{(neutral or alkaline environment)} \tag{12.5}$$

The oxidation process at the anode induces the dissolution of a sacrificial metal electrode (M) to its cation (M^{z+}) (Eq. 12.1), where z refers to the number of electrons transferred per mole of metal. Simultaneously, water's electrochemical reduction promotes hydrogen evolution at the cathode (Eqs. 12.2 and 12.3). The overall reactions (Eqs. 12.4 and 12.5) resulted in the in-situ generation of metal hydroxide, which acts as a strong coagulant, allowing destabilization of colloids, suspended particles, and emulsions, which later agglomerated into flocs that can be removed by flotation or sedimentation [47, 63, 74]. The schematic process of EC is presented in Fig. 12.2.

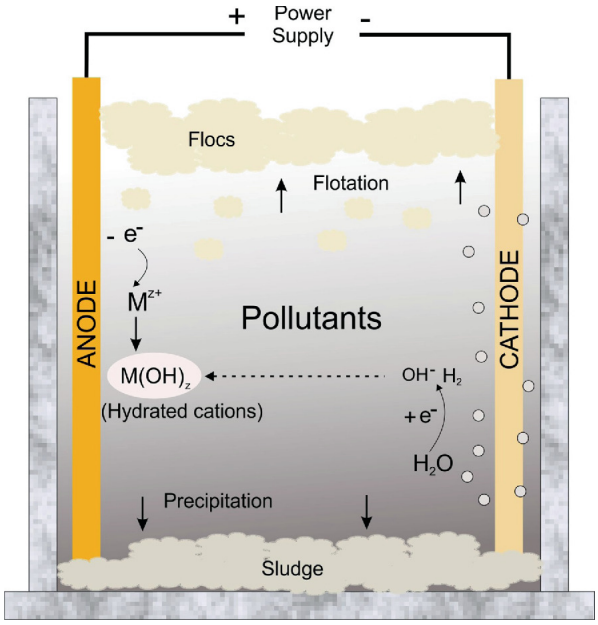


Fig. 12.2 Schematic diagram of the electrocoagulation process. Adapted from [47]

12.2.2 Factors Influencing the Performance of Electrocoagulation Process

The performance of the EC process could be affected by electrode material, inter-electrode distance, pH, conductivity, current density, mechanical stirring, and temperature. The influences of each factor on the electrocoagulation process are described below.

Electrode materials

The electrochemical reactions that take place in the electrocoagulation process are determined by the electrode materials. Aluminum (Al) and iron (Fe) are the most common electrode materials used as anodes due to their low prices, availability, and efficacy in removing pollutants. Aluminum dissolves as Al^{3+}, but depending on the pH, iron may dissolve as Fe^{2+} or Fe^{3+} [40, 110]. Nonetheless, applying stainless steel as cathode is highly recommended if the solution contains Ca^{2+} or Mg^{2+} ions in high concentration [81]. Table 12.2 presents several anode materials used in the EC process for leachate treatment. The active nature of electrodes significantly influences their dissolution rate during electrochemical reactions. The ability of metal to corrode can be seen through the corrosion potential (E_{corr}). Lower E_{corr} means the metal is highly active and could easily corrode or dissolve in galvanic couples (Table 12.3).

Table 12.2 Anode material that has been used in the EC process

Electrode	Advantages	Disadvantages	References
Aluminum	Abundant availability High COD, color, total nitrogen (TN), and turbidity removal efficiency compared to iron	More expensive than iron High electrode passivation Higher sludge ratio and treatment cost than iron The sludge is toxic to the environment	[17, 38, 104]
Iron	Cheap Abundant availability In some cases, it is more efficient than aluminum Low toxicity to the environment Low operating cost	Low removal efficiency compared to aluminum	[17, 47]
Magnesium	Cheap Abundant availability Eco-friendly	Require high operational pH (pH > 8.5) to achieve better removal efficiency	[37, 98]
Ti/RuO_2–IrO_2	Complete mineralization of organic matter	Expensive	[75]

Table 12.3 Typical corrosion potential (E_{corr}) of anode materials [117]

Metal	$E_{corr}(V_{SCE})$
Mg	− 1.65
Al-7075	− 0.88
Al-1xxx	− 0.73
Fe	− 0.50

Many researchers have evaluated the efficacy of some electrode materials in removing pollutants. The application of aluminum as a sacrificial electrode in the EC process has achieved approximately 40–81% COD [14, 52], 82% color [52], and 59% NH_4^+ removal [14]. Similar color removal efficiency was also obtained using the iron electrode [62], but the COD removal was lower, i.e., 39–45% [7, 62]. Another study [75] reported very high COD and NH_3-N using Ti/RuO_2–IrO_2, i.e., 83.7% and 100%, respectively.

On the other hand, a passivation mechanism always follows the dissolution process to reduce the deterioration of materials. In the EC process, electrode passivation has been known to harm the reactor performances (Fig. 12.3). Aluminum electrodes, in particular, could form an oxide layer on the electrode surface that will hamper metal dissolution and electron transfer and limit coagulant production [81]. As the passivation layer is increased, the efficacy of EC will decrease. Nonetheless, the anions in the solution could slow down the passivation process or break up the passivation layers. The sequence order is as follows: $Cl^- > Br^- > I^- > F^- > ClO_4^- > OH^-$ and SO_4^{2-} [81].

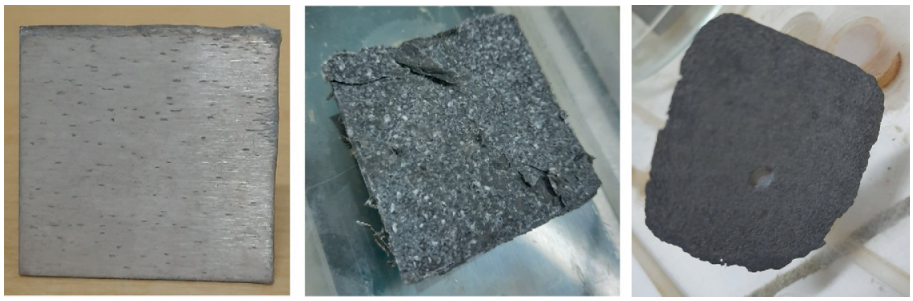


Fig. 12.3 Build-up of the passivation layer and metal corrosion during the electrochemical process

Inter-electrode distance

Inter-electrode distance could determine the number of electrodes placed in the electrocoagulation cell [56]. It also defines the residence time between the electrodes in the continuous electrocoagulation system and the electrolysis time for a batch system to reach desirable electrocoagulation efficacy. A previous study [17] showed that increasing the inter-electrode distance between 0.5 and 2.8 cm could slightly enhance COD removal by 10%. However, this result may differ due to leachate composition and the volume of leachate treated.

Increasing the distance between electrodes could increase the uniformity of current density distribution between electrodes. However, it could also raise the electrolyte's ohmic resistance (IR-drop), decrease current density, and reduce metal dissolution [56, 89]. The electrical potential must be increased to maintain higher current density, which leads to high energy requirements. The IR-drop and its relation to power consumption can be calculated according to Eqs. 12.6–12.8:

$$IR = \frac{I \times d}{A \times k} \tag{12.6}$$

$$U = I \times IR \tag{12.7}$$

$$P = \int_0^t U \times I \times dt \tag{12.8}$$

where IR is the ohmic resistance of the electrolyte (Ω), I is the applied current (A), d is the inter-electrode distance (cm), A is the surface area of the cathode and anode (m^2), k is the solution conductivity (mS m^{-1}), U is the cell voltage (V), and P is the energy consumption.

In some cases, a shorter inter-electrode distance may be better in the electrocoagulation process, as it enhances the turbulent hydrodynamics in the cells, promotes shorter electrolysis time, and reduces energy consumption [88]. In a solution with high electrolytic conductivity, a short inter-electrode distance increases the variation in current density distribution on the electrode geometry [89]. Uneven electrode

dissolution can be noticed since the areas with greater current density were rapidly exhausted, providing a greater capacity to remove pollutants. [89] obtained a significant increase in natural organic matter (NOM) and dissolved organic carbon (DOC) removal as the inter-electrode gap decreased from 10 to 1 mm. Nonetheless, the selection of inter-electrode distances must be carefully considered, as it may induce faster electrode passivation, particularly at a high organic load [47].

pH

The removal of pollutants in the EC process is pH-dependent. It involves two mechanisms, i.e., (i) binding of metal species to anionic sites of the organic molecules, neutralizing their charge, and reducing their solubility; (ii) adsorption of organic matter on the amorphous metal hydroxide precipitate [23]. The pH could affect the current efficiency and electrode dissolution [40]. It could also control the metal solubility via hydrolysis and polymerization reaction (Fig. 12.4). The pH value tends to change during electrocoagulation. The pH increase is typically quite high for influent with pH < 4 and relatively small for influent with an initial pH value around 6–8. In contrast, the effluent pH value decreases if the initial pH is over 8. This condition indicates a buffering effect due to the balance between the production and consumption of hydroxyl ions during electrocoagulation [70].

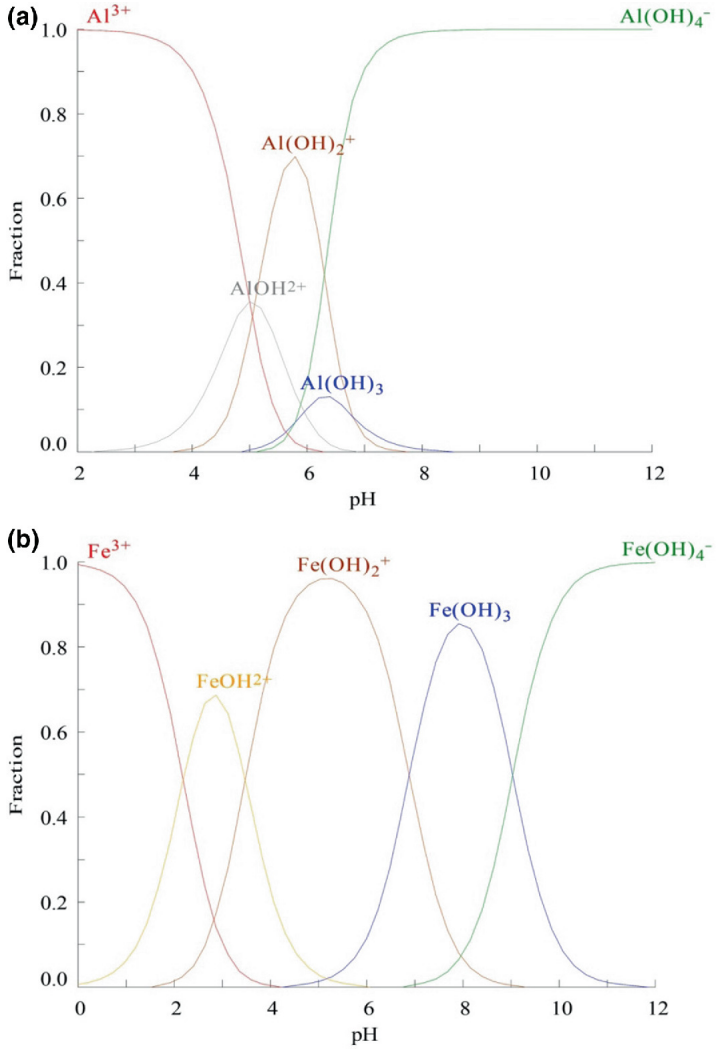


Fig. 12.4 Speciation diagrams of the mononuclear hydrolysis products for **a** Aluminum and **b** Iron [40]

Stabilized leachate typically has a neutral to nearly alkaline pH (7–9). In this pH range, aluminum may exist in the forms of $Al(OH)_2^+$, $Al(OH)_3$, and mostly $Al(OH)_4^-$. The high $Al(OH)_4^-$formation is unfavorable since it decreases the electrocoagulation performance. The pH of leachate has to be reduced to around 5.5–7 to promote the formation of insoluble $Al(OH)_3$ and increase the coagulation performance. On the contrary, since Fe^{3+} is effective in a wider pH range (pH 5–9), the pH reduction is unnecessary.

The influence of pH on the EC process was also confirmed by Galvao et al. (2020). It revealed that the increase of pH induces a higher release of an electrode and higher production of OH^-, leading to a higher amount of coagulant. [14] obtained 81% COD removal using an aluminum electrode when experiments were performed at pH 4 and a current density of 29.17 mA cm^{-2}. However, this removal efficiency dropped to 71% when the pH was raised to 8. Likewise, another study attained 62% COD removal at pH 5, which was higher compared to COD removal at pH 7 [7].

Conductivity

High electrical conductivity is beneficial to increase the current efficiency and inhibit the electrode passivation process [47]. Otherwise, supporting electrolytes (such as NaCl, Na_2SO_4, or $NaNO_3$) need to be added to the solution to maintain high electrolytic conductivity and current efficiency and reduce the applied cell potential for the EC process. The removal efficiency of NOM was reported at about 96.84% and 99.64% with the addition of Na_2SO_4 and NaCl, respectively [144]. Adding Cl^- is also beneficial to increase the ionic strength of water, decrease the negative effect of CO_3^{2-} and SO_4^{2-}, and reduce the deposition of Mg^{2+} and Ca^{2+} on the electrode surface, which could rapidly decrease the current efficiency. [81] recommended that 20% of Cl^- be present in the solution to ensure a normal operation of the EC process. However, since stabilized landfill leachate generally has high electrical conductivity, adding Cl^- may not be necessary.

Current density

Current density plays an important role in the EC process. The increase of current density stimulates mass transfer at the electrode, higher release of metal ions, and metal hydroxide production, leading to bigger flocs formation and higher removal of pollutants. The generation rate of H_2 bubble gas also increased considerably with increasing current density, which is favorable for the flotation process [36, 53]. Nonetheless, the increasing current density may not always be beneficial for the EC process because (i) increases the precipitation of hydroxyl ions in the electrode surface, preventing metal dissolution and reducing electrocoagulation efficacy; (ii) enhances the power consumption and operational cost; (iii) overdose the coagulant production, reversing the charge of the colloids and redisperse them into the solution; as a result, the coagulation efficiency and electrode life span decreased considerably [47, 107]. For those reasons, the current density selection should consider the operating pH, temperature, and flow rate to ensure high current efficiency.

Theoretically, the quantity of metal dissolved or deposited can be calculated through Faraday's laws [81]:

$$w = \frac{itM}{nF} \tag{12.9}$$

where w is the amount of dissolved electrode material (g cm^{-2}); i is the current density (A cm^{-2}); t is the electrolysis time (s); M is the molar mass of the electrode (g mol^{-1}); n is the number of an electron; F is the Faraday's constant (96,500 C mol^{-1}).

However, it has been reported that the actual dissolution rate may differ from the theoretical value [110]. At lower pH (pH 5), higher iron dissolution (twice the theoretical value calculated by Faraday's law) indicates the existence of another mechanism of iron dissolution that is independent of the electrochemical process. While at higher pH (pH > 9), lower iron dissolution (half the theoretical value) may be attributed to the secondary reactions that occur near the anode, including oxygen evolution. To evaluate the electrocoagulation performance, faradaic efficiency (FE) is typically used (Eq. 12.10) [64].

$$FE(\%) = \frac{Observed\ metal\ dissolution}{Theoretical\ metal\ dissolution} \times 100\% \tag{12.10}$$

A previous study [81] showed that the faradaic efficiency for aluminum electrodes could be between 120 and 140%. High faradaic efficiency (over 100%) is typically attributed to pitting corrosion, particularly when chloride ions are present in the solution. In contrast, the faradaic efficiency of iron electrodes is generally around 100%. However, lower faradaic efficiency of iron was frequently reported in some studies [64]. Sasson et al. [110] also observed lower iron dissolution (almost half of the theoretical iron dissolution calculated by Faraday's law of electrolysis) in the experiments conducted at pH 9. Although the cause is unclear, this phenomenon may be attributed to secondary reactions other than iron dissolution. The secondary reactions (such as O_2 evolution) stimulate the formation of γ-FeOOH that induces passivation. Moreover, higher current densities could also strengthen anodic polarization, which attracts particles on the electrode surface [64].

Mechanical stirring

Mechanical stirring during electrocoagulation could increase anode dissolution, reduce passivation, promote a homogeneous solution, and stimulate efficient contact between coagulants and pollutants [47, 92]. On the other hand, higher stirring speed may disturb ion mobilization and break up agglomerated flocs into small particles that are difficult to collect. [92] obtained more than 90% COD, TSS, and TDS removal at a mixing speed of 150 rpm. Moreover, a gradual reduction of removal efficiency due to flocs breakage was detected as the speed increased to 200–250 rpm.

Temperature

Increasing the temperature during the EC process promotes faster destruction of the oxide membrane and increases the current efficiency, leading to higher pollutant removal efficiency [81]. [92] also found that the increase in temperature could enhance the hydrolysis of Al^{3+} to $Al(OH)_3$. Their study confirmed this theory; between 1.5 and 2.5% increase in COD, TSS, and TDS removal was achieved during experiments at 25–45 °C. A further temperature rise (> 60 °C) is unrecommended, as it gradually decreases the current efficiency.

12.2.3 Performances and Applications of Electrocoagulation for Leachate Treatment

The performance and application of electrocoagulation in several studies are compared in Table 12.4.

Table 12.4 The performance and application of EC for stabilized leachate treatment

Experimental scale	Feed characteristics	Operational parameters	Removal efficiency	Energy consumption	References
Single treatment (Lab-scale)	pH: 8.42 COD: 3160 mg L^{-1}	Electrodes: Al (anode), SS (cathode) Electrode gap: 2 cm	81% COD 59% NH_4^+	10.47 Wh g^{-1} COD	[14]

Experimental scale	Feed characteristics	Operational parameters	Removal efficiency	Energy consumption	References
	BOD_5: 157 mg L^{-1} NH_4^+: 202 mg L^{-1}	Feed vol: 0.5 L Flow rate: 5 mL s^{-1} pH: 4 I: 0.7 A *j*: 29.17 mA cm^{-2} t: 360 min Mixing speed: 200 – 400 rpm		105.33 Wh g^{-1} NH_4^+	
Single treatment (Lab-scale)	pH: 7.9 COD: 4212 mg L^{-1} BOD_5: 923 mg L^{-1} NH_4–N: 1648 mg L^{-1} Cl^-: 1862 mg L^{-1}	Electrodes: Al (anode & cathode) Electrode gap: 1 cm Feed vol: 2 L *j*: 12.8 mA cm^{-2} t: 90 min Mixing speed: n.a	82% Color 40% COD 90% BOD_5	n.a	[52]
Single treatment (Lab-scale)	pH: 8.1 – 8.8 COD: 7560 – 8892 mg L^{-1}	Electrodes: Fe sheets (anode and cathode) Electrode gap: 4 cm Feed vol: 0.25 L pH: 3 *j*: 40 mA cm^{-2} t: 15 min No mixing	39% COD	n.a	[7]
Single treatment (Lab-scale)	pH: 8.1 – 8.8 COD: 7560 – 8892 mg L^{-1}	Electrodes: Al sheets (anode and cathode) Electrode gap: 4 cm Feed vol: 0.25 L pH: 5 *j*: 30 mA cm^{-2} t: 25 min No mixing	62% COD	n.a	[7]
Single treatment (Lab-scale)	pH: 7.82 COD: 2130 mg L^{-1} BOD: 850 mg L^{-1} TSS: 270 mg L^{-1} TDS: 4397 mg L^{-1}	Electrodes: Al (rotating anode) Electrode gap: 1 cm Feed vol: 10 L pH: 7 I: 2 A t: 40 min Mixing speed: 150 rpm	> 90% COD, TSS, TDS	n.a	[92]
Single treatment (Lab-scale)	pH: 8.43 COD: 7230 mg L^{-1} BOD: 540 mg L^{-1} Color: 14,750 ADMI	Electrodes: Fe sheets (anode and cathode) Electrode gap: 1.16 cm Feed vol: 0.50 L pH: 7.73 t: 60 min Mixing speed: 100 rpm	45.1% COD 82.7% Color	n.a	[62]
Single treatment (Lab-scale)	pH: 8.04 COD: 3968 mg L^{-1} BOD_5: 480 mg L^{-1}	Electrodes: Ti/RuO_2-IrO_2 (anode); Al (cathode) Electrode gap: n.a Feed vol: 0.50 L pH: 6.37	83.7% COD 100% NH_3–N	61.59 Wh g^{-1} COD 106.91 Wh g^{-1} NH_3–N	[75]

Experimental scale	Feed characteristics	Operational parameters	Removal efficiency	Energy consumption	References
	NH_3–N: 2000 mg L^{-1} Cl^-: 2500 mg L^{-1}	*j*: 100 mA cm^{-2} [Cl^-]: 6.5 g L^{-1} t: 150 min Mixing speed: 200 rpm			
Pre-treatment (Lab-scale)	pH: 7.8 COD: 3850 mg L^{-1} BOD: 60 mg L^{-1} TSS: 4400 mg L^{-1} NH_3: 638 mg L^{-1} NO_3^-: 63 mg L^{-1} PO_4^{3-}: 40 mg L^{-1}	Electrodes: Al (anode), SS (cathode) Electrode gap: 5 cm Feed vol: 0.75 L pH: no adjustment *j*:38. 6 mA cm^{-2} t: 30 min leachate: municipal wastewater ratio of 1:5 Mixing speed: n.a	73% COD 53% TSS 87% NH_3 95% NO_3^- 85% PO_4^{3-}	n.a	[133]
Post-treatment (Lab-scale)	pH: 7.74 COD: 10,000 mg L^{-1} TOC: 3719 mg L^{-1} Color: 14,850 Pt–Co Cl^-: 7200 mg L^{-1}	Electrodes: Fe (anode and cathode) Electrode gap: 1 cm Feed vol: 0.5 L pH: 7 I: 3.5 A t: 20 min Mixing speed: 200 rpm	57.4% COD 77.1% Color 45% TOC	n.a	[55]
Post-treatment (Lab-scale)	pH: 4.62 COD: 1840 mg L^{-1} TOC: 698.78 mg L^{-1} NH_3–N: 636 mg L^{-1} TN: 1290 mg L^{-1}	Electrodes: Al (anode), graphite (cathode) Electrode gap: n.a pH: 7 *j*: 6 mA cm^{-2} t: 120 min Mixing speed: n.a	51.8% COD 66.1% TOC 31.5% NH_3-N 36.2% TN	33.31 Wh L^{-1}	[118]

n.a: no data available

12.3 Electro-Fenton

12.3.1 Principles and Mechanism of Electro-Fenton

Electro-Fenton (EF) can be defined as an electrochemically assisted Fenton process. It has higher removal efficiency, simplicity, easy regulation of the H_2O_2 production, and lower operational cost compared to the traditional Fenton process [29, 78]. Moreover, ferric sludge production is low due to the continuous regeneration of ferrous ions [77]. In the EF process, Fe^{2+} ions are obtained from sacrificial iron anodes (Eq. 12.11). This reaction is followed by simultaneous reduction of water at the cathode (Eq. 12.12). The electro-generation of H_2O_2 (Eq. 12.13), which controls the rate of Fenton's reaction, immediately occurs through cathodic reduction of O_2 or O_2 introduction (injected as compressed air). It then reacted with Fe^{2+} to mass produce the highly reactive hydroxyl radicals (Eq. 12.14). Fe^{2+} instantly regenerates at the cathode through cathodic reduction of Fe^{3+} (Eq. 12.15). Higher applied voltage stimulates faster reduction of Fe^{3+} and increases the production of $OH^{\bullet}$. Additionally, $OH^{\bullet}$ is also generated at the anode from water oxidation (Eq. 12.16) [46, 97, 130].

$$Fe \rightarrow Fe^{2+} + 2e^- \tag{12.11}$$

$$2H_2O + +2e^- \rightarrow H_2 + 2OH^- \tag{12.12}$$

$$O_2 + 2H^+ + 2e^- \rightarrow H_2O_2 \tag{12.13}$$

$$H_2O_2 + Fe^{2+} + H^+ \rightarrow Fe^{3+} + H_2O + OH^\bullet \tag{12.14}$$

$$Fe^{3+} + e^- \rightarrow Fe^{2+} \tag{12.15}$$

$$H_2O \rightarrow OH^\bullet + H^+ + e^- \tag{12.16}$$

Hydrogen radicals obtained in the EF process are the primary oxidizing agents for pollutant degradation (Fig. 12.5).

$$OH^\bullet + organic\,matter \rightarrow intermediates + CO_2 + H_2O \tag{12.17}$$

$$Fe^{2+} + H_2O_2 + organic\,matter \rightarrow intermediates + Fe^{3+} + CO_2 + H_2O + OH \tag{12.18}$$

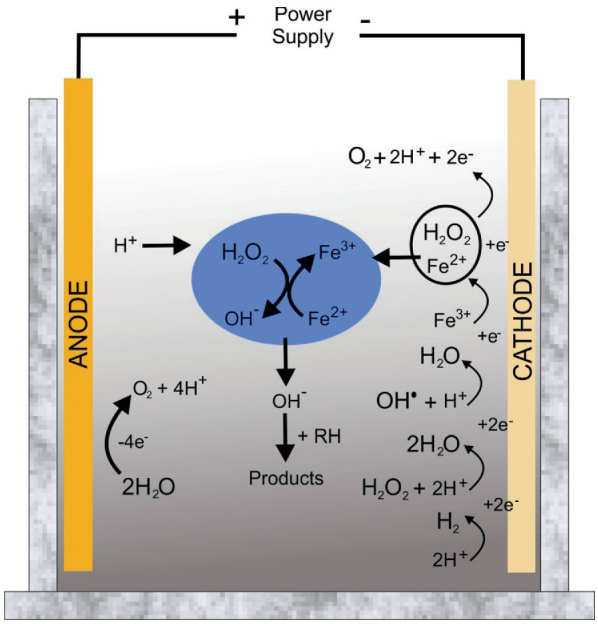


Fig. 12.5 Schematic diagram of electro-Fenton.
Adapted from [95] © Elsevier

12.3.2 Factors Influencing the Performance of Electro-Fenton Process

The performances of the electro-Fenton process can be affected by electrode materials, electrode distance, pH, applied current, oxygen sparging rate, supporting electrolytes, catalyst concentration, H_2O_2 concentration, and temperature [60, 95, 97].

Electrode materials

Selecting appropriate electrode materials is fundamental in the EF process. Proper anode materials contribute to the high generation rate of hydroxyl radicals and better treatment efficiency. The anode materials that have been used in the EF process include Pt, BDD, $Ti/IrO_2/RuO_2$, Ti/RuO_2, $Ti/IrO_2/SnO_3$, graphite, carbon felt, Fe, stainless steel, and carbon nanotubes [95]. Among those, BDD was superior in terms of current efficiency, formation rate of hydroxyl radicals, energy consumption, and pollutant degradation. Furthermore, cathode materials selection is crucial for the efficiency of EF reaction since it could promote oxygen reduction reaction (Eq. 12.19) and the in-situ generation of H_2O_2 [60]. An

ideal cathode (working electrode) for the EF process should have good chemical stability, low catalytic activity for H_2O_2 decomposition, and high overpotential for H_2 evolution [20].

Carbon-based materials are typically used as cathodes due to their low cost, availability, good conductivity, and the possibility to modulate their characteristics to favor the selectivity toward oxygen reduction reaction (ORR) [90]. [119] compared different carbon-based materials based on their ability to generate H_2O_2. The result showed that carbon sponge cathode (particularly with 45 pores per linear inch) has a higher H_2O_2 production due to their high specific surface area and high porosity for mass transport. In contrast, carbon felt has a dense structure that could hamper the transport of O_2 and H_2O_2 in the cathode mass and lead to lower H_2O_2 generation [119]. Modification of cathode materials is often applied to produce more active sites at the cathode surface to increase the production of H_2O_2, reduce energy consumption, and widen the working pH [91]. The materials that have been studied include functionalized carbon black (FCB), porous carbon nanotube (CNT), and carbon aerogel [136, 137, 142]. Several electrode materials for the EF process are compiled in Table 12.5. In order to measure the oxidation efficiency of cathode material, mineralization current efficiency (MCE) is often used (Eq. 12.20) [119].

$$O_2 + 4e^- + 4H^+ \rightarrow 2H_2O \tag{12.19}$$

$$MCE(\%) = \frac{nFV_s \cdot \Delta(TOC)_{\exp}}{4.32 \times 10^7 mIt} \times 100 \tag{12.20}$$

Table 12.5 Advantages and disadvantages of electrode materials for the EF process

Electrode materials	Advantages	Disadvantages	References
Pt	• High stability • High conductivity	• Expensive • Robust interaction with $OH^\bullet$ hinders the $OH^\bullet$ availability for pollutant degradation	[95, 114]
BDD	• High oxidizing power • Higher mineralization current efficiency • Larger O_2 overvoltage • A high oxidation window allows the direct oxidation of pollutants • High formation of $OH^\bullet$ • Lower specific energy consumption	• Expensive	[20, 116]
Gas diffusion electrode (GDE)	• Highly porous structure • High catalytic activity • Efficient three-phase contact between solid, liquid, and gas • Effective for large H_2O_2 generation and oxygen reduction reaction process	• The electrode materials used to prepare GDE influence the amount of H_2O_2 production • Traditional GDE requires aeration using an external device, which adds energy consumption and leads to physical erosion of conventional electrodes	[120, 152]
Carbon black	• Low cost • High electronic conductivity	• Low faradaic efficiency (particularly when large electrodes are used)	[90]
Activated carbon fiber	• Low cost • High specific surface area (typically ~ 1000 $m^2\ g^{-1}$) • Good chemical stability • High conductivity	• Low strength; their fragmentation and peeling may cause secondary pollution during usage	[68, 79, 91]

Electrode materials	Advantages	Disadvantages	References
	• High catalytic activity for H_2O_2 generation (12 times faster than graphite) • Great adsorption capability		
Reticulated vitreous carbon	• Low cost • Chemically and electrochemically inert • High specific surface area • High fluid permeability • Stands high current density • High corrosion resistance • Low density and thermal expansion • Easily shaped as required by cell design • Good selectivity for H_2O_2 electrosynthesis	• Fragile • Lower electrical conductivity compared to other carbonaceous materials • Requires specific assembly for industrial application	[10, 51, 131]
Carbon sponge	• High surface area • High porosity • High H_2O_2 generation	• The amount of pores per linear inch (ppi) influences the H_2O_2 generation. Higher pores lead to low H_2O_2 production	[119]
Carbon/graphite felt	• Relatively low cost • High specific surface • Stable physicochemical properties • Good conductivity	• Inadequate wettability and electrochemical activity in aqueous solution • Hydrophobic surface • Dense structure • Poor kinetics for redox reaction • Low H_2O_2 generation	[72, 91, 119]
Carbon nanotube	• Low cost • Porosity > 99% • Highest electrogeneration of H_2O_2 • High selectivity for ORR • Good mass transfer rate • High electrochemical active reaction area • High structural flexibility and robustness • High electrical conductivity and wettability to organics	• Have a strong tendency to agglomerate and are difficult to disperse in solvent	[136]
Carbon aerogels	• Large specific surface area • High porosity • High electrical conductivity • Good corrosion resistance • Good thermal stability • High mechanical robustness	• Integration with other materials is required to enhance their electrosorption ability and catalytic activity in the electrosorption EF process	[91, 137]
Graphite/graphene	• High discoloration rate • High electrogeneration of H_2O_2	• Low surface area • Poorly malleable property	[91, 137]
Stainless steel	• Cheap	• Small electrochemical active reaction area • Low production of H_2O_2 • Very low $OH^{\bullet}$ production, poor oxidation efficiency	[119]

where n is the number of electrons consumed during the mineralization process, F is the Faraday constant (96,487 C mol^{-1}), V_s is the solution volume (L), $\Delta(TOC)_{exp}$ is the experimental TOC decay (mg L^{-1}), 4.32×10^7 is a conversion factor to homogenize unit, m is the number of carbon atoms of the molecules, I is the applied current (A), and t is the electrolysis time (h).

Electrode distance

The distance between electrodes highly influences the overall EF performance. Short electrode distances reduce the resistance between electrodes, increase the current flow and mass transfer rate, reduce electrolysis time, and enhance the treatment efficiency [60]. [151] reported that the optimal inter-electrode distance for COD removal is between 1.3 – 2.1 cm. Lower inter-electrode distance would accelerate the oxidation of Fe^{2+} to Fe^{3+}. While longer inter-electrode distances hinder the mass transfer of Fe^{3+} to the cathode, inhibit Fe^{2+} regeneration, decrease the EF efficacy, and increase energy consumption [8, 95].

pH

pH is an important factor in the EF process, as it influences the formation of hydroxyl radicals, electro-dissolution of Fe^{2+} catalyst, and reduction reaction on the surface of electrodes [116]. Electro-Fenton works more effectively in acidic environments, i.e., pH 2.5–3.5 (Oturan and [97]). In the lower pH, hydroxyl radicals may be scavenged by excessive hydrogen ions to produce a stable $H_3O_2^+$ that is less reactive toward Fe^{2+}. Consequently, the reaction rate between H_2O_2 and Fe^{2+} decreases, and $OH^{\bullet}$ production is limited (Eq. 12.21).

$$H_2O_2 + H^+ \rightarrow H_3O_2^+ \tag{12.21}$$

At pH 3–4, ferrous ions existed in the form of $[Fe(OH)(H_2O)_5]^+$, $Fe(OH)^+$, and $Fe(OH)_2$ [93]. Due to its catalyst nature, the formation of $Fe(OH)_2$ in the EF process is desirable as it enhances the production of $OH^{\bullet}$. Meanwhile, higher pH ($pH > 4$) is not favorable since (i) ferric ions are prone to precipitate (Eq. 12.22), decreasing the total iron in the solution, hindering the regeneration of Fe^{2+}, and reducing the overall performance of electro-Fenton; (ii) H_2O_2 self-decomposition increased in the alkaline environment (Eq. 12.23); (iii) the absence of H^+ hinder the decomposition of H_2O_2 and reduce the production of $OH^{\bullet}$; (iv) under alkaline environment, CO_3^{2-} and HCO_3^- are considered as $OH^{\bullet}$ scavengers [34].

$$Fe^{3+} + OH^- \rightarrow Fe(OH)_3 \tag{12.22}$$

$$2H_2O_2 \rightarrow 2H_2O + O_2 \tag{12.23}$$

The effect of lower pH on electro-Fenton performances was proved by many researchers. [46] achieved a higher COD removal at pH 3 than at pH 5 (i.e., ~ 40% and ~ 20%, respectively) during 8 h experiments at an applied current of 1.4 A and initial dissolved iron concentration of 61 mg/L. Similar results were obtained by [8], by which the COD removal was around 60% at pH 3 and reduced to approximately 40% when the pH rose to pH 4.

Applied current

Applied current is the key factor in the EF process since it controls the generation rate of H_2O_2, Fe^{2+}, and $OH^{\bullet}$ [116]. Increasing the applied current raises the current density, the $OH^{\bullet}$ production and physisorbed on electrodes, and the efficiency of the Fenton reaction. Moreover, a reduction of electrolysis time may also occur with the increase in current density. After a limit of applied current is achieved, the EF process efficiency decreases due to the occurrence of competitive side reactions, including anodic oxygen discharge (Eq. 12.24), hydrogen evolution at the cathode (Eq. 12.25), reduction of H_2O_2 at the anode (Eq. 12.26) and cathode (Eq. 12.27), parasitic reaction of $OH^{\bullet}$, leading to undesirable consumption of H_2O_2 (Eqs. 12.28 and 12.29), and recombination of $OH^{\bullet}$ (Eq. 12.30) [95],Oturan and [55, 97]. Massive alkalinization would occur in the high current density, inducing $Fe(OH)_3$ precipitation that hampers the EF process [20]. Although increasing the applied current may reduce the degradation rate, it could also increase energy consumption. Thus, the adjustment of applied current is necessary for balancing the targeted efficiency removal and energy cost.

$$2H_2O \rightarrow 4H^+ + O_2 + 4e^- \tag{12.24}$$

$$2H^+ + 2e^- \rightarrow H_2 \tag{12.25}$$

$$H_2O_2 + 2H^+ + 2e^- \rightarrow 2H_2O \tag{12.26}$$

$$H_2O_2 \rightarrow O_2 + 2H^+ + 2e^- \tag{12.27}$$

$$H_2O_2 + OH^{\bullet} \rightarrow H_2O + HO_2^{\bullet} \tag{12.28}$$

$$HO_2^{\bullet} + OH^{\bullet} \rightarrow H_2O + O_2 \tag{12.29}$$

$$2OH^{\bullet} \rightarrow H_2O_2 \tag{12.30}$$

Oxygen sparging rate

Increasing the oxygen sparging rate significantly increases dissolved oxygen concentration, mass transfer rate, and the production of H_2O_2. Furthermore, the oxygen sparging rate also helps to remove NH_3–N and N_2 from the solution. [135] obtained approximately 94% NH_3–N, 100% NO_2–N, and 51% total nitrogen removal at the end of the EF process.

Supporting electrolytes

Electrolyte addition is often required to increase the conductivity of solutions for efficient reactions. Na_2SO_4 is commonly used in the EF system due to its high ionic strength and low interference in the solution. Other electrolytes that have been used include KCl, NaCl, $NaNO_3$, and $NaClO_4$. Previous studies showed that H_2O_2 production is faster in the presence of NO_3^- than SO_4^{2-}, while the mineralization efficiency was higher with ClO_4^{2-} than SO_4^{2-} [32, 60, 106]. The addition of Cl^- could also accelerate H_2O_2 production, promote removal efficiency, and shorten the electrolysis time by producing Cl_2 or HClO (Eqs. 12.31 and 12.32), which are strong oxidation agents. However, the formation of toxic by-products may also occur due to the reaction of organics and OCl^-.

$$2Cl^- \rightarrow Cl_2 + 2e^- \tag{12.31}$$

$$Cl_2 + H_2O \rightarrow H^+ + Cl^- + HClO \tag{12.32}$$

On the other hand, Cl^- may act as an $OH^\bullet$ scavenger (Eqs. 12.33 and 12.34). A previous study [112] reported a reduction in $OH^\bullet$ with increased NaCl concentration. Moreover, the oxidation of Cl^- leads to the formation of $Cl^\bullet$ and $Cl_2^{-\bullet}$, which is less reactive than $OH^\bullet$ [32, 112].

$$OH^\bullet + Cl^- \rightarrow HOCl^{-\bullet} \tag{12.33}$$

$$HOCl^{-\bullet} + H^+ \rightarrow Cl^\bullet + H_2O \tag{12.34}$$

Catalyst concentration

The best catalyst for Fenton's reaction is Fe^{3+}/Fe^{2+} due to its non-toxic nature and efficiency at low concentrations (typically around 0.1 mM) [116]. A high concentration of Fe^{2+} enhances the ionic strength of the solution, improves the current efficiency of the EF process, and increases the degradation rate of pollutants. However, at long treatment times, the effectiveness of the EF process decreases with the excess of Fe^{2+}. The overdose concentration of Fe^{2+} may scavenge $OH^\bullet$ in the solution and reduce their ability to oxidize organic matter (Eq. 12.35) [93].

$$Fe^{2+} + OH^\bullet \rightarrow Fe^{3+} + OH^- \tag{12.35}$$

H_2O_2 concentration

H_2O_2 concentration affects the amount of $OH^\bullet$ production in the EF system [34]. The overall degradation efficiency usually increases with the rise of H_2O_2 concentration. Nevertheless, an excess of H_2O_2 may not only reduce the concentration of $OH^\bullet$ in the solution and decrease their ability to oxidize organic pollutants, but it also creates detrimental effects to the following treatment (particularly when EF is used as a pretreatment to biological processes). Guvenc et al. [55] noticed that COD removal increased considerably as the ratio of H_2O_2/COD raise from 0.5 to 1.25 (i.e., 53.6% to 68.9%), then slightly decreased to 61.6% at the H_2O_2/COD ratio of 2 due to $OH^\bullet$ scavenging reaction (Eqs. 12.28 and 12.29). Therefore, a proper H_2O_2 concentration needs to be determined to maximize the EF performance and minimize the drawbacks related to excess concentration [60].

Temperature

The temperature of a reaction system could affect the reaction rate. However, their effect on the EF process may be insignificant compared to other influencing factors [60]. The EF is typically performed at an ambient temperature. Increasing temperature may enhance the reaction rate between H_2O_2 and Fe^{2+}, the $OH^\bullet$ production, and the efficiency of pollutant degradation. A previous study [3] achieved higher mineralization efficiency when the experiments were conducted at 35–40° C. As the temperature increased further, the efficacy of the EF process gradually decreased due to a reduction in O_2 solubility and rapid decomposition of H_2O_2. Jones et al. [89] stated that a 10 °C increase in temperature would speed up the H_2O_2 decomposition by 2.3 times.

12.3.3 Performances and Applications of Electro-Fenton for Leachate Treatment

The performance and application of electro-Fenton in several studies are compared in Table 12.6.

Table 12.6 The performance and application of EF for stabilized leachate treatment

Application (scale)	Leachate characteristics	Operational parameters	Removal efficiency	Energy consumption	References
Single treatment (Lab-scale)	pH: 9.5 COD: 5120 mg L^{-1} BOD: 535 mg L^{-1} Color: 2767 Abs Cl: 720 mg L^{-1}	Electrodes: Fe (anode and cathode) Feed vol: 4 L pH: 3 U: 4.5 V H_2O_2 dosage: 5000 mg L^{-1} t: 90 min	COD: 99% Color: 100%	n.a	[67]
Pre-treatment (Lab-scale)	pH: 8.1 TSS: 498 mg L^{-1} COD: 23,200 mg L^{-1} BOD: 4872 mg L^{-1} NH_3-N: 2196 mg L^{-1} Phosphate: 185 mg L^{-1} Cl: 2670 mg L^{-1}	Electrodes: cast iron Feed vol: 1 L pH: 2.85 H_2O_2 dosage: 55.76% *j*: 12.95 mA cm^{-2} t: 40 min	TSS: 45.2% COD: 78.2% BOD: 55.4% NH_3-N: 41% Phosphate: 60.9% Cl: 62.2%	1.8 kWh m^{-3}	[124]
Post-treatment (Lab-scale)	pH: 6.95 – 7.06 COD: 792 mg L^{-1} Cl^-: 4648 mg L^{-1}	Electrodes: Ti/RuO_2 mesh (anode and cathode) Feed vol: 0.72 L pH: 4.0 U: 5.5 V H_2O_2/COD_0: 1.5 H_2O_2/Fe^{2+}: 2.5 I: 2.75 A t: 50 min	COD: 94%	3.44 kWh kg^{-1} COD	[77]
Post-treatment (Lab-scale)	pH: 7.95 COD: 5250 mg L^{-1} TSS: 122.5 mg L^{-1} Cl: 7285 mg L^{-1}	Electrodes: Fe (anode and cathode) Feed vol: 0.5 L pH: 2.9 H_2O_2/COD: 1.42 I: 2.27 A t: 30.3 min	COD: 67.1%	n.a	[130]
Post-treatment (Lab-scale)	pH: 7.74 COD: 10,000 mg L^{-1} TOC: 3719 mg L^{-1} Color: 14,850 Pt–Co Cl: 7200 mg L^{-1}	Electrodes: Fe (anode and cathode) Feed vol: 0.5 L pH: 3.5 I: 2.75 A t: 30 min	COD: 69.4% Color: 87.6% TOC: 60.2%	n.a	[55]
Post-treatment (Lab-scale)	pH: 7.9 COD: 42,000 mg L^{-1} TKN: 9100 mg L^{-1} NH_3–N: 8300 mg L^{-1} Cl: 11,100 mg L^{-1}	Electrodes: graphite felt (anode), BDD (cathode) Feed vol: 0.25 L pH: 3 t: 480 min	COD: 40%	n.a	[46]
Post-treatment (Lab-scale)	pH: 7.70 COD: 3896 mg L^{-1}	Electrodes: Pt (anode) carbon PTFE (cathode) Cathode area: 20 cm^2	TOC: 82% TN: 51%	n.a	[135]

Application (scale)	Leachate characteristics	Operational parameters	Removal efficiency	Energy consumption	References
	TOC: 1347 mg L^{-1} TN: 592 mg L^{-1}	Feed vol: 0.2 L pH: 3 $FeSO_4$: 10 mM *j*: 30 mA cm^{-2} t: 360 min			
Post-treatment (Lab-scale)	pH: 8.36 COD: 2350 mg L^{-1} Color: 1143 Abs PO_4–P: 10.25 mg L^{-1} NH_4–N: 310 mg L^{-1} Cl: 5484 mg L^{-1}	Electrodes: Fe (anode and cathode) Feed vol: 0.5 L pH: 3 [H_2O_2]: 2000 mg L^{-1} I: 2 A t: 20 min	COD: 72% Color: 90% PO_4–P: 87% NH_4–N: 26%	n.a	[8]

n.a. data is not available

12.4 Electrochemical Oxidation

12.4.1 The Principles and Mechanisms of Electrochemical Oxidation

The EO process involves direct current (DC) from a power supply, electrode materials, and the electrolyte that provides the ion transport mechanism within an electrolytic cell [35, 48]. The feasibility and efficacy of this process depend on the production of physically or chemically adsorbed hydroxyl radicals, the characteristic nature of electrode materials, and the competition with the oxygen evolution reaction [86]. In general, organic contaminants can be removed through two different mechanisms, i.e., direct and indirect oxidation processes.

Direct oxidation process via intermediate oxygen evolution

The direct oxidation or “anodic oxidation” process does not require large amounts of chemical addition or O_2 feeding to the cathode. Thus, the reactor will be much simpler, with no secondary pollution. Generally, electrochemical oxidation can be achieved by producing physically adsorbed hydroxyl radicals ($OH^{\bullet}$) or chemically adsorbed oxygen in the oxide lattice (MO). The reactions are as follows [86]:

First, adsorbed hydroxyl radicals (OH•) are formed from the oxidation of water molecules on the electrode surface (Eq. 12.36).

$$M + H_2O \rightarrow M(OH^{\bullet})_{ads} + H^+ + e^- \tag{12.36}$$

where M is the electrode, and $(OH^{\bullet})_{ads}$ are the adsorbed hydroxyl radicals. The reactivity of adsorbed $OH^{\bullet}$ reaction strongly depends on the nature of electrode materials. These electrode materials can be classified into *active electrodes* and *non-active electrodes*. The description of the *active and non-active electrodes* will be explained further in Sect. 4.2.

In the case of the *active electrode*, there is a strong interaction between the electrode and the hydroxyl radicals. The adsorbed $OH^{\bullet}$ may interact with the anode and form MO with a higher oxide (Eq. 12.37). Since the MO acts as a mediator of organic (R) oxidation (Eq. 12.38), the oxidative reaction will be more selective. There is a competition between the side reaction of oxygen evolution and the chemical decomposition of higher oxide MO, which leads to partial oxidation of contaminants (Eq. 12.39).

$$M(OH^{\bullet})_{ads} \rightarrow MO + H^+ + e^- \tag{12.37}$$

$$MO + R \rightarrow M + RO \tag{12.38}$$

$$MO \rightarrow M + \frac{1}{2}O_2 \tag{12.39}$$

The *non-active electrode* has a weak interaction between hydroxyl radicals and the electrode surface. Here, organic contaminants can be fully oxidized into CO_2 or other oxidation products because the oxidation process is mediated by hydroxyl radicals (Eq. 12.40)

$$M(OH^{\bullet}) + R \rightarrow M + CO_2 + H_2O + H^+ + e^- \tag{12.40}$$

In general, the electrochemical activity (related to the overpotential of oxygen evolution) and chemical reactivity (oxidation rate of organics) of adsorbed $OH^{\bullet}$ are strongly correlated to the strength of the M $OH^{\bullet}$ interaction. Weaker interaction will produce a fast chemical reaction and higher anode reactivity for organics oxidation.

Indirect oxidation process

Meanwhile, indirect oxidation processes are carried out by forming Cl_2/OCl^- electrochemically generated from the oxidation of Cl^- that naturally exists or is added to the solution [35]. The reactions are shown in Eqs. 12.41–12.43 [100]. The indirect oxidation process significantly affects contaminant removal more than the direct process because contaminants can be oxidized in the bulk solution [27, 87].

$$2Cl^- \rightarrow Cl_2 + 2e^- \tag{12.41}$$

$$Cl_2 + H_2O \rightarrow HOCl + H^+ + Cl^- \tag{12.42}$$

$$HOCl + R \rightarrow CO_2 + H_2O + H^+ + Cl^- \tag{12.43}$$

12.4.2 Factors Influencing the Performance of Electrochemical Oxidation Process

In this section, we described several major factors that significantly influence the removal of contaminants in the EO system.

Electrode materials

Electrode materials have a significant effect on the EO process. Each type of electrode material has characteristics that affect the production of reactive oxygen species ($OH^{\bullet}$), chlorine species (Cl_2 and HOCl), and sulfate species (SO_4^{2-}) involved in the degradation of contaminants [85]. The properties of electrode materials are explained in Table 12.7. *Active electrodes* are typically highly conductive and can act as an electro-catalyst. However, due to a low overpotential for oxygen evolution and strong adsorption properties against OH radical ($OH^{\bullet}$), the utilization of these materials will lead to partial oxidation of organic compounds [86]. Carbon, graphite, platinum, and metal oxides such as IrO_2 and RuO_2 are an example of *active electrodes* [11, 102].

Table 12.7 Properties of several electrode materials that have been used in the EO process

Electrode category	Electrode materials	Advantages	Disadvantages	Oxygen potential (V)	References
Active	Carbon and graphite	• Very cheap • High electric conductivity	• Prone to electrochemical corrosion that reduces their service life	1.7	[83, 99, 102]

Electrode category	Electrode materials	Advantages	Disadvantages	Oxygen potential (V)	References
		• Able to combine adsorption and electrochemical degradation of contaminants • Relatively effective for COD removal	• Ineffective in raising the biodegradability index		
	Pt	• High conductivity • High chemical stability • Long service life	• Expensive • Selective oxidation • Low current efficiency	1.3–1.6	[83, 102]
	IrO_2	• Active electro-catalyst for chlorine evolution • Lower dissolution rate compared to RuO_2	• Expensive due to metal scarcity • Short lifetime, susceptible to corrosion under the strongly acidic and anodic conditions	1.52–1.6	[26, 83, 101, 102]
	RuO_2	• Higher chlorine evolution efficiency than IrO_2 • Higher electrocatalytic activity than IrO_2 at a higher current density • High ammonium and COD removal	• Expensive due to metal scarcity • Short lifetime, susceptible to corrosion under the strongly acidic and anodic conditions • Unstable for oxygen evolution	1.47	[25, 83, 101, 102]
Non-active	PbO_2	• Inexpensive • Have high oxygen overpotential • High electrical conductivity • Good stability • Higher organic contaminants removal compared to other electrodes, except for BDD	• Poor electrocatalytic oxidation performance • May cause electrochemical corrosion, which leads to severe water pollution • Likely release toxic ions that are harmful to human health	1.9	[83, 99, 102]
	SnO_2	• Inexpensive • Low toxicity • Relatively high oxygen evolution potential • High electrochemical oxidation activity • Slow resistivity and better electrochemical stability • Promote full oxidation of organic compounds	• Not commercially available • Relatively short lifetime due to the weak combination between Ti substrate and SnO_2 layer	1.9	[59, 99, 102, 113]
	BDD	• High oxygen overpotential • High chemical and electrochemical stability under extreme conditions • High resistivity against anodic corrosion • High current efficiency • High oxidation rate • Less energy consumption	• Expensive • Lower removal efficiency of ammonium nitrogen compound compared to Pt • May generate high toxic by-products (e.g., chlorate, perchlorate, or halogenated organic compounds) when applied at a certain current	2.3–2.7	[100, 102]

On the other hand, *the non-active electrode* has a high overpotential for oxygen evolution that allows the production of highly reactive physisorbed $OH^{\bullet}$ on the electrode surface. This high overpotential for oxygen evolution enhances the EO process efficiency by promoting the complete degradation of organic compounds [99, 100]. Examples of the

non-active electrode are boron-doped diamond (BDD) and other metal oxides such as PbO_2 and SnO_2.

In practice, two or three different materials can be fabricated to get more desired properties. For example, the fabrication of titanium coated with RuO_2 or IrO_2 (Ti/RuO_2 or Ti/IrO_2), which is commonly called the dimensionally stable anode (DSA), is used to increase the electrode catalytic activity for chlorine evolution and enhance the electrode lifetime by rising their chemical and mechanical stability against anodic corrosion [99, 102]. Additionally, the fabrication of Ti/SnO_2 can be made to increase the electrode stability, their overpotential for oxygen evolution, and their effectiveness on phenol removal. Moreover, the combination of Ti/SnO_2–Sb_2O_5 could increase the chemical and electrochemical stability, conductivity, current efficiency, their overpotential for oxygen evolution, and the production of hydroxyl radicals [83].

High percentages of contaminant removal have been achieved previously. Between 63–84% of COD and 80–100% of NH_4-N were successfully removed using Ti/Pt electrodes [45, 134]. Another study reported 83% of COD and 100% NH_3-N removal using Ti/RuO_2–IrO_2 [75]. Approximately 90% of COD and 56–70% of NH_4-N removal were achieved using BDD electrode [6, 12, 49, 154], while 64–90% removal of NH_4-N was accomplished using Ti/Pt/PbO_2 and Ti/Pt/SnO_2–Sb_2O_4 electrode [49].

Compared to other electrodes, BDD is the most attractive because of its high catalytic activity toward organic oxidation and its capability to produce reactive oxygen species (e.g., $OH^{\bullet}$, H_2O_2, O_3), peroxy species ($C_2O_6^{2-}$, $P_2O_8^{4-}$, $S_2O_8^{2-}$), and inorganic radical species (e.g., $Cl^{\bullet}$, $Cl_2^{-\bullet}$, $SO_4^{2-\bullet}$, $CO_3^{2-\bullet}$, $H_2PO_4^{\bullet}$, $PO_4^{2-\bullet}$) in the presence of coexisting ions [31, 43, 44]. Nonetheless, since $OH^{\bullet}$ was weakly adsorbed on the BDD surface, BDD could only oxidize larger molecules with a lower diffusion coefficient. Consequently, BDD shows lower total nitrogen and ammonium nitrogen removal than other electrodes [49].

Current density

Current density refers to the applied current per unit area of the electrode. On the one hand, the increase in current density could reduce the electrolysis time needed for effective COD and NH_4^+ removal and substantially enhance the production of hydroxyl radicals and chlorine species responsible for contaminants degradation [5, 27, 76]. The effect of current density on contaminant removals has been widely studied. [4] obtained 0.43, 0.53, and 0.72 g of COD removal at current densities of 50, 75, and 100 mA cm^{-2}, respectively. Meanwhile, [82] revealed more than 90% of TOC removal with the increase of current density to 83 mA cm^{-2}. Another study achieved more than 99% ammonia removal with a rise in current density from 50 to 150 mA cm^{-2} [146].

At higher current density, the direct anodic oxidation responsible for COD removal is increased with the higher production of hydroxyl radicals. This increase in contaminants removal was more pronounced at the electrode with high oxygen overpotential due to insignificant side reactions of oxygen evolution [47]. However, further current density increases were reported to significantly enhance the ammonium nitrogen removal rate rather than COD [149, 150]. An increased active chlorine production at a higher current density promotes the indirect oxidation process and eventually depresses the anodic oxidation.

On the other hand, increasing current density could also enhance the electrical charge, leading to lower efficiency [5]. For instance, the increase of current density from 30 to 70 mA cm^{-2} enhanced the specific energy consumption on the BDD anode from 82 to 110 Wh g^{-1} COD. Similar results were obtained at Ti/Pt/PbO_2 and Ti/Pt/SnO_2-Sb_2O_4 anodes,

i.e., 77–125 Wh g^{-1} COD and 88 to 143 Wh g^{-1} COD [48]. The instantaneous current efficiency (ICE) for the anodic oxidation can be calculated as follows [86].

$$ICE = FV\left(\frac{[(COD)_t - (COD)_{t+\Delta t}]}{8I\Delta t}\right) \tag{12.44}$$

where F is the faraday constant (96,487 C mol^{-1}), V is the electrolyte volume (L), $(COD)_t$ and $(COD)_{t+\Delta t}$ are the chemical oxygen demand (g L^{-1}) at times t and t + Δt, 8 is the oxygen equivalent mass(g eq^{-1}), and I is the current (A).

pH

The effect of pH on organic contaminants removal was relatively significant. As known, chlorine species vary with the specific pH value. The dominant chlorine species at pH < 3 is Cl_2, at pH 3–7.5 is HOCl, while at pH > 7.5 is OCl^- [33, 82]. At pH < 3, the oxidation process was enhanced because there was a significant reduction of CO_3^{2-} and HCO_3^- known as hydroxyl radical scavengers [35, 83]. However, the oxidation rate was not too high because more active chlorine is transformed into chlorine gas [5]. At pH 4–5, the indirect oxidation process extensively occurred due to the large formation of HOCl compounds. At pH > 6, HOCl compounds are gradually transformed into OCl^- (a less effective oxidant) and consequently reduce the oxidation process [33]. At a further higher pH level (pH > 9), more ammonium removal would increase due to the volatilization of ammonium compound into NH_3 gasses [33]. The increase of COD and ammonium removal with higher pH levels was also reported by Luu [82], i.e., the COD and ammonium nitrogen removal at pH 2 was approximately 20 and 11%, respectively. These removal percentages increased to 29.17% (COD) and 35.14% (ammonium nitrogen) when the pH rose to 6.

Chloride ions concentration

Chloride ion concentration influences the production of active chlorine involved in contaminants removal. High chloride ions concentration significantly enhances the production of active chlorine/hypochlorite and ensures the removal of most organic contaminants through indirect oxidation. Several authors confirmed the increase of organics and nitrogen removal with the rise of chloride ion concentration [13, 27, 48, 103]. Bagastyo et al. [13] revealed the increase of COD removal to 86% with 2 g L^{-1} of NaCl addition. Meanwhile, another study found that the ammonium transformation to nitrogen gas could increase to 85% by adding 20 g/L of chloride [103]. Overall, the chemical reactions between ammonium and chlorine/hypochlorite during the indirect oxidation process are as follows [103]:

$$NH_4^+ + HOCl \rightarrow NH_2Cl + H_2O + H^+ \tag{12.45}$$

$$NH_2Cl + HOCl \rightarrow NHCl_2 + H_2O \tag{12.46}$$

$$NHCl_2 + HOCl \rightarrow NCl_3 + H_2O \tag{12.47}$$

$$2NH_4^+ + 3HOCl \rightarrow N_2 + 3H_2O + 5H^+ + 3Cl^- \tag{12.48}$$

$$NH_4^+ + 4HOCl \rightarrow NO_3^- + H_2O + 6H^+ + 4Cl^- \tag{12.49}$$

Further increase in chloride ion concentration does not significantly affect COD removal. Thus, adding chloride ions has to be carefully measured to minimize the production of chlorinated by-products that are harmful to the environment. Nonetheless, an increase in chloride ion concentration could also increase the solution's electric

conductivity, leading to lower energy consumption. A previous study showed that specific energy consumption reduction could reach 25 – 31% by adding 2 g L^{-1} chloride ions [48].

Additional electrolytes concentration

Electrolytes are mainly added to increase the electrical conductivity of wastewater to be treated and reduce the energy consumption of the EO system. However, the addition of electrolytes has other advantages. Turro et al. [126] used perchloric acid ($HClO_4$) as the main electrolyte to reduce pH levels and remove a partial portion of inorganic carbon into CO_2. The inorganic carbon reduction was reduced by 70%, dramatically dropping the pH before the EO process. Conversely, no significant removal of organic contaminants was reported with the increase in electrolyte concentration. The effect of other electrolytes, such as sodium sulfate (Na_2SO_4) and sodium nitrate ($NaNO_3$), was also investigated in several studies [12, 44, 84]. The additions of those electrolytes have been known to improve the oxidation efficiency and the removal of selected contaminants.

Sulfate radicals have comparable oxidation rates with hydroxyl radicals because of their similar redox potential at acidic pH, i.e., E° = 2.5–3.1 V ($SO_4^{2-\bullet}$) and 2.7 V ($OH^\bullet$) [22, 148]. However, since $SO_4^{2-\bullet}$ is likely to react via electron transfer mechanisms, their effect on the oxidation and degradation of contaminants is more significant than $OH^\bullet$. Meanwhile, a previous study indicated that contaminants' electrooxidation was 10–15 times faster in Na_2SO_4 than in $NaNO_3$ anolytes [44]. Those faster electrooxidation rates ensure the reduction in electrolysis time and energy consumption for persistent organic contaminants removal.

12.4.3 Performance and Application of Electrochemical Oxidation for Leachate Treatment

The performance and application of electrochemical oxidation in several studies are compared in Table 12.8.

Table 12.8 The performance and application of EO for stabilized leachate treatment

Experimental scale	Feed characteristics	Operational parameters	Removal efficiency	Energy consumption	References
Single treatment (Lab-scale)	pH: 8.5 COD: 2040 mg L^{-1} NH_4^+: 535 mg L^{-1} Cl^-: 1400 mg L^{-1}	Electrodes: BDD (anode), Ag (cathode) Feed vol: 1 L Flow rate: 2 mL s^{-1} pH: 3 *j*: 75 mA cm^{-2} Molar ratio of $[Cl^-]$: $[SO_4^{2-}]$ = 18:1 t: 420 min	COD: 51% NH_4^+: 59%	48.1 Wh g^{-1} COD 175.4 Wh g^{-1} NH_4^+	[4]
Pre-treatment (Lab-scale)	pH: 8.58 COD: 2011 mg L^{-1} NH_4^+: 173.77 mg L^{-1} Cl^-: 5198 mg L^{-1}	Electrodes: BDD; Ti/IrO_2 (anode), SS (cathode) Ratio of effective volume: 1/12.5 Feed vol: 5 L Flow rate: 5 mL s^{-1} pH: 3 *j*: 30 mA cm^{-2} t: 240 min	• BDD 36% COD • Ti/IrO_2 48% COD	• BDD 0.3 Wh g^{-1} COD • Ti/IrO_2 0.7 Wh g^{-1} COD	[96]

Experimental scale	Feed characteristics	Operational parameters	Removal efficiency	Energy consumption	References
Pre-treatment (Lab-scale)	pH: 7.84 COD: 4980 mg L^{-1} NH_4^+: 206.76 mg L^{-1} TP: 59.53 mg L^{-1} Cl^-: 3710 mg L^{-1}	Electrodes: graphite (anode), SS (cathode) Feed vol: 2.5 L Flow rate: 0.7 mL s^{-1} pH: 7.5 *j*: 17 mA cm^{-2} t: 720 min NaCl addition: 2000 mg L^{-1}	COD: 86.2% NH_4^+: 85% TP: 89% Cl: 83%	1.13 Wh g^{-1} COD 2.27 Wh g^{-1} NH_4^+ 12.5 Wh g^{-1} TP 1.45 Wh g^{-1} NH_4^+	[13]
Post-treatment (Lab-scale)	pH: 7.22 COD: 2464 mg L^{-1} NH_3–N: 154.4 mg L^{-1}	Electrodes: BDD-Fe (anode); carbon felt (cathode) Feed vol: 0.25 mL pH: 11 *j*: 25 mA cm^{-2} t: 540 min	81.3% COD 99.8% NH_3–N	n.a	[123]
Post-treatment (Lab-scale)	pH: 5.54 COD: 4900 mg L^{-1} TOC: 1874 mg L^{-1} TKN: 520 mg L^{-1} NH_3-N: 21.35 mg L^{-1} Cl^-: 5700 mg L^{-1}	Electrodes: Ti/TiO_2–RuO_2–IrO_2 and Ni/BDD (anodes) SS (cathode) Feed vol: 1.5 L pH: 5.5 *j*: 33.3 mA cm^{-2} t: 480 min	• Ti/TiO_2–RuO_2–IrO_2 34.3% COD; 27.3% TOC; 93.7% TKN; 97.4% NH_3-N • Ni/BDD 86.4% COD; 77.4% TOC; 93.4% TKN; 94.4% NH_3-N	n.a	[66]
Post-treatment (Lab-scale)	pH: 4.62 COD: 1840 mg L^{-1} TOC: 698.78 mg L^{-1} NH_3-N: 636 mg L^{-1} TN: 1290 mg L^{-1}	Electrodes: Ru-Ir/Ti (anode), graphite (cathode) Feed vol: 0.25 L pH: 9 *j*: 8 mA cm^{-2} t: 180 min	COD: 59.1% TOC: 70.1% NH_3-N: 98.8% TN: 73.0%	81.86 Wh L^{-1}	[118]
Post-treatment (Lab-scale)	pH: 8.3 COD: 816 mg L^{-1} NH_4^+: 13.6 mg L^{-1} Cl^-: 999.7 mg L^{-1}	Electrodes: BDD; Ti/IrO_2; Ti/Pt (anode), SS (cathode) Feed vol: 2 L Flow rate: 5 mL s^{-1} pH: 6 *j*: 58.3; 66.7 mA cm^{-2} t: 600 min	• Ti/Pt 78% (66.7 mA cm^{-2}) • Ti/IrO_2 76% (58.3 mA cm^{-2}) • BDD 85% (58.3 mA cm^{-2})	• Ti/Pt 56.7 Wh g^{-1} • Ti/IrO_2 58.9 Wh g^{-1} • BDD 36.9 Wh g^{-1}	[11]
Post-treatment (Lab-scale)	pH: 8.72 COD: 1168 mg L^{-1} NH_4^+: 63.5 mg L^{-1} Cl^-: 1900 mg L^{-1}	Electrodes: BDD (anode), SS (cathode) Feed vol: 2000 mL Flow rate: 7 mL s^{-1} pH: 5 *j*: 37.5 mA cm^{-2} t: 480 min	COD: 93% NH_4^+: 59%	5.7 Wh g^{-1} COD 7.5 Wh g^{-1} N	[12]
Post-treatment (Pilot-scale)	pH: 7.6 – 8.1	Electrodes: Ti/BDD; Ti/RuO_2(anode), Pt	• Ti/BDD 95.17% COD	• Ti/BDD	[82]

Experimental scale	Feed characteristics	Operational parameters	Removal efficiency	Energy consumption	References
	COD: 568 – 736 mg L^{-1} TOC: 168 – 293 mg L^{-1} NH_4^+–N: 248 – 365 mg L^{-1}	(cathode) Feed vol: 50 mL Flow rate: 13.89 mL s^{-1} *j*: 83 mA cm^{-2} t: 480 min	91.89% TOC 81.18% NH_4^+ 63.54% TN • Ti/RuO_2 92.57% COD 78.76% TOC 97.85% NH_4^+ 72.12% TN	160 Wh g^{-1} COD • Ti/RuO_2 180 Wh g^{-1} COD	

12.5 Advantages and Limitations of Electrochemical Processes for Leachate Treatment

In this section, we compare the advantages and limitations of each electrochemical process (Table 12.9). Other challenges that may be involved during processes are described in Table 12.10.

Table 12.9 The advantages and limitations of electrochemical-based treatment

Process	Advantages	Limitations	References
Electrocoagulation (EC)	• Simple equipment • Fewer chemical requirements • High flexibility • Low energy requirements compared to EO • Ease of control	• Higher sludge production compared to EF and EO • The sacrificial anode needs to be replaced periodically • Passivation of the electrode reduced pollutant removal efficiency and increased power consumption • pH adjustment, equalization may be necessary in some cases • The presence of chloride may result in the formation of toxic by-products	[58, 64, 70]
Electro-Fenton (EF)	• Low quantity and cheaper catalyst source • Low-cost cathodic materials to produce H_2O_2 • In-situ production of H_2O_2 and Fe^{2+} • Able to treat different water matrices • Versatile design and construction • Easy automation and operability • Low energy requirement when coupled with solar cells • Easy post-treatment separation and reusability of effluent • Can be upgraded or combined with other processes	• Require an acidic environment (pH 2.8–3.5) to achieve optimal pollutant degradation • pH adjustment and neutralization are required for treated wastewater • pH ≥ 4.5 promotes the precipitation of $Fe(OH)_3$, reducing the catalyst, decreasing the EF performance, and adding operational cost • The formation of Fe^{3+} complexes is difficult to eliminate • The occurrence of H_2 evolution limits H_2O_2 generation • Unsuitable for the continuous process since the catalysts are difficult to reuse and recycle	[46, 94, 114]
Electrochemical oxidation (EO)	• No sludge production • Breakdown of higher molecular organic compounds	• High-cost electrodes • High energy consumption • Potentially generated toxic chlorinated by-products	[6, 47, 83]

Process	Advantages	Limitations	References
	• Complete mineralization of organics • Robust and versatile • Less or no chemical addition • Ease to automation		

Table 12.10 Other challenges in the electrochemical process

Aspects	Challenges	Approach
Electrocoagulation		
Passivation layer	Removing the initial passivating layer during reactor setup is challenging [129]	Frequent mechanical cleaning is the most effective strategy for decreasing the passivation layer. However, it may increase labor costs and termination of the treatment process. Other strategies that can be used include adding aggressive ions, implementing alternating current, applying polarity reversal, ultrasonication, chemical cleaning, and hydrodynamic scouring [64]
Electrode cleaning	Intensive mechanical cleaning leads to holes in the electrode, compromising their structural integrity [15]	Electrode deep cleaning targeting the light-colored external electrode surface. Otherwise, different types of steel can be used as the electrode material [15]
Electro-Fenton		
Electrode materials	Efficient anodic material (e.g., Pt and BDD) might be too expensive to be applied on an industrial scale [114]	Fabrication of low-cost cathodic material and catalyst with higher stability and good effectiveness could minimize the operational cost of the EF process
Catalyst	Electrogenerated H_2O_2 has lower performance and inadequate Faradaic efficiency	Enhance catalyst and reactor design is important for the commercialization of the EF process
Electrochemical oxidation		
Electrodes material	Some electrodes that have higher removal efficiency (e.g., DSA and BDD) are quite expensive and may not be commercially available for a full-scale system	Applying carbon or graphite electrodes may become a better option since it is cheaper and easy to find. Integrating the EO process with other treatments may become a better option to reduce the operational cost and increase removal efficiency
Energy	The energy used for the EO process might be quite large. Energy consumption highly depends on the electrodes used, applied current density, and the amount and characteristics of leachate to be treated	Alternative energy sources such as solar panels or energy-derived waste can be used to reduce the total energy cost

12.6 Future Perspective of the Electrochemical Process for Leachate Treatment

The electrochemical process is a promising technology for stabilized leachate treatment. High removal of pollutants can be achieved with less or no sludge addition. Considering their mechanisms, electro-Fenton and electrochemical oxidation are more suitable as post-treatment, while electro-coagulation can be used as pre-treatment or post-treatment. The selection of the most appropriate treatment has to consider the economic aspects. Electrocoagulation can be performed with the cheapest electrodes. Chemical addition is often less or unnecessary depending on the electrode used. However, it produces higher sludge compared to electro-Fenton and electrochemical oxidation.

On the other hand, the use of carbonaceous material in the electro-Fenton process can be done to reduce the electrode costs. However, the requirement for a strongly acidic environment for an effective EF process may be challenging since it requires more chemical addition for pH adjustment and neutralization. Although a heterogeneous EF

process can be applied to widen the working pH, it requires a modified cathode, which may be more expensive and difficult to synthesize or fabricate.

In contrast, the major drawbacks of the electrochemical oxidation process are expensive electrodes and high energy consumption. Even though the electrodes can be substituted with cheaper materials, such as graphite, the degradation of pollutants may not be optimum. Therefore, applying biological processes before electrochemical oxidation may be the best option to reduce the organic loads, minimize the formation of toxic by-products caused by chlorine reactions, and decrease energy consumption. Furthermore, since the pilot and full-scale study is very rare, future research is required to obtain more data prior to its implementation.

12.7 Conclusion

Stabilized leachate is a complex wastewater with a low biodegradability index and high recalcitrant organic compounds. Because biological processes are ineffective in treating this type of wastewater, electrochemical treatment can become an alternative. All electrochemical treatments mentioned in this chapter have high efficiency in pollutant removal, either by agglomerating particles into bigger flocs that can be removed or oxidizing the organic matter. The overall electrochemical treatment performance and operational cost are mostly influenced by the electrode materials, pH, current density, and conductivity. Other factors (e.g., electrode distance, temperature, etc.) could enhance the efficacy performance, but they may be insignificant. Selecting the appropriate electrode material and current density is crucial to ensure the optimum pollutant degradation. Moreover, all economic aspects related to the treatment must be considered before their full-scale implementation.

References

1. Abuabdou SMA, Bashir MJK, Aun NC et al (2021) Development of a novel polyvinylidene fluoride membrane integrated with palm oil fuel ash for stabilized landfill leachate treatment. J Clean Prod 311:127677. https://doi.org/10.1016/j.jclepro.2021.127677
2. Aftab B, Cho J, Shin HS, Hur J (2020) Using EEM-PARAFAC to probe NF membrane fouling potential of stabilized landfill leachate pretreated by various options. Waste Manag 102:260–269. https://doi.org/10.1016/j.wasman.2019.10.035
3. Agladze GR, Tsurtsumia GS, Jung B-I et al (2007) Comparative study of hydrogen peroxide electro-generation on gas-diffusion electrodes in undivided and membrane cells. J Appl Electrochem 37:375–383. https://doi.org/10.1007/s10800-006-9269-x
4. Agustina F, Bagastyo AY, Nurhayati E (2019) Electro-oxidation of landfill leachate using boron-doped diamond: Role of current density, pH and ions. Water Sci Technol 79:921–928. https://doi.org/10.2166/wst.2019.040
5. Anglada A, Ortiz D, Urtiaga AM, Ortiz I (2010) Electrochemical oxidation of landfill leachates at pilot scale: evaluation of energy needs. Water Sci Technol 61:2211–2217. https://doi.org/10.2166/wst.2010.130
6. Anglada Á, Urtiaga A, Ortiz I et al (2011) Boron-doped diamond anodic treatment of landfill leachate: evaluation of operating variables and formation of oxidation by-products. Water Res 45:828–838. https://doi.org/10.1016/j.watres.2010.09.017
7. Apaydin Ö, Özkan E (2020) Landfill leachate treatment with electrocoagulation: optimization by using Taguchi method. Desalin Water Treat 173:65–76. https://doi.org/10.5004/dwt.2020.24719
8. Atmaca E (2009) Treatment of landfill leachate by using electro-Fenton method. J Hazard Mater 163:109–114. https://doi.org/10.1016/j.jhazmat.2008.06.067
9. Aziz SQ, Aziz HA, Bashir MJK, Mojiri A (2015) Assessment of various tropical municipal landfill leachate characteristics and treatment opportunities. Glob NEST J 17:439–450
10. Badellino C, Rodrigues CA, Bertazzoli R (2007) Oxidation of herbicides by in situ synthesized hydrogen peroxide and fenton's reagent in an electrochemical flow reactor: study of the degradation of 2,4-dichlorophenoxyacetic acid. J Appl Electrochem 37:451–459. https://doi.org/10.1007/s10800-006-9275-z

11. Bagastyo AY, Hidayati AS, Herumurti W, Nurhayati E (2021) Application of BDD, Ti/IrO_2, and Ti/Pt anodes for the electrochemical oxidation of landfill leachate biologically pretreated by moving bed biofilm reactor. Water Sci Technol 83:1357–1368. https://doi.org/10.2166/wst.2021.060
12. Bagastyo AY, Novitasari D, Nurhayati E, Direstiyani LC (2020) Impact of sulfate ion addition on electrochemical oxidation of anaerobically treated landfill leachate using boron-doped diamond anode. Res Chem Intermed 46:4869–4881. https://doi.org/10.1007/s11164-020-04243-3
13. Bagastyo AY, Sari PPI, Direstiyani LC (2021) Effect of chloride ions on the simultaneous electrodialysis and electrochemical oxidation of mature landfill leachate. Environ Sci Pollut Res 28:63646–63660. https://doi.org/10.1007/s11356-020-11519-z
14. Bagastyo AY, Sidik F, Anggrainy AD et al (2022) The performance of electrocoagulation process in removing organic and nitrogenous compounds from landfill leachate in a three-compartment reactor. J Ecol Eng 23:235–245. https://doi.org/10.12911/22998993/145290
15. Bandaru SRS, Roy A, Gadgil AJ, Van GCM (2020) Long-term electrode behavior during treatment of arsenic contaminated groundwater by a pilot-scale iron electrocoagulation system. Water Res 175:115668. https://doi.org/10.1016/j.watres.2020.115668
16. Benaddi R, Ferkan Y, Bouriqi A, Ouazzani N (2022) Impact of landfill leachate on groundwater quality—a comparison between three different landfills in Morocco. J Ecol Eng 23:89–94. https://doi.org/10.12911/22998993/153006
17. Bouhezila F, Hariti M, Lounici H, Mameri N (2011) Treatment of the OUED SMAR town landfill leachate by an electrochemical reactor. Desalination 280:347–353. https://doi.org/10.1016/j.desal.2011.07.032
18. Boyer TH, Graf KC, Comstock SEH, Townsend TG (2011) Magnetic ion exchange treatment of stabilized landfill leachate. Chemosphere 83:1220–1227. https://doi.org/10.1016/j.chemosphere.2011.03.040
19. Brennan RB, Healy MG, Morrison L et al (2016) Management of landfill leachate: the legacy of European Union directives. Waste Manag 55:355–363. https://doi.org/10.1016/j.wasman.2015.10.010
20. Brillas E, Sire I, Oturan MA (2009) Electro-Fenton process and related electrochemical technologies based on Fenton's reaction chemistry. Chem Rev 109:6570–6631. https://doi.org/10.1021/cr900136g
21. Budi S, Suliasih BA, Othman MS et al (2016) Toxicity identification evaluation of landfill leachate using fish, prawn and seed plant. Waste Manag 55:231–237. https://doi.org/10.1016/j.wasman.2015.09.022
22. Buxton GV, Greenstock CL, Helman WP, Ross AB (1988) Critical review of rate constants for reactions of hydrated electrons, hydrogen atoms and hydroxyl radicals (OH/·O− in aqueous solution. J Phys Chem Ref Data 17:513–886. https://doi.org/10.1063/1.555805
23. Cañizares P, Martínez F, Lobato J, Rodrigo MA (2006) Electrochemically assisted coagulation of wastes polluted with eriochrome black T. Ind Eng Chem Res 45:3474–3480. https://doi.org/10.1021/ie051432r
24. Cao HT, Khue DN, Nguyen NTT et al (2021) Insight into removal TOC and NH_4^+ from mature landfill leachate using coupled bio-coagulation from Moringa oleifera seeds and ZVI/H_2O_2 process. J Water Process Eng 42:102112. https://doi.org/10.1016/j.jwpe.2021.102112
25. Chen X, Chen G (2005) Stable Ti/RuO_2-Sb_2O_5-SnO_2 electrodes for O_2 evolution. Electrochim Acta 50:4155–4159. https://doi.org/10.1016/j.electacta.2005.01.032
26. Cherevko S, Geiger S, Kasian O et al (2016) Oxygen and hydrogen evolution reactions on Ru, RuO_2, Ir, and IrO_2 thin film electrodes in acidic and alkaline electrolytes: a comparative study on activity and stability. Catal Today 262:170–180. https://doi.org/10.1016/j.cattod.2015.08.014
27. Chiang LC, Chang JE, Wen TC (1995) Indirect oxidation effect in electrochemical oxidation treatment of landfill leachate. Water Res 29:671–678. https://doi.org/10.1016/0043-1354(94)00146-X
28. Christensen TH, Cossu R, Stegmann R (2005) Landfilling of waste: leachate. CRC Press
29. Crispim AC, de Araújo DM, Martínez-Huitle CA et al (2022) Application of electro-Fenton and photoelectro-Fenton processes for the degradation of contaminants in landfill leachate. Environ Res 213:113552. https://doi.org/10.1016/j.envres.2022.113552
30. Das PP, Sharma M, Purkait MK (2022) Recent progress on electrocoagulation process for wastewater treatment: a review. Sep Purif Technol 292:121058. https://doi.org/10.1016/j.seppur.2022.121058
31. Davis J, Baygents JC, Farrell J (2014) Understanding persulfate production at boron doped diamond film anodes. Electrochim Acta 150:68–74. https://doi.org/10.1016/j.electacta.2014.10.104

32. De Laat J, Le Truong G, Legube B (2004) A comparative study of the effects of chloride, sulfate and nitrate ions on the rates of decomposition of H_2O_2 and organic compounds by Fe(II)/H_2O_2 and Fe(III)/H_2O_2. Chemosphere 55:715–723. https://doi.org/10.1016/j.chemosphere.2003.11.021
33. Deborde M, von Gunten U (2008) Reactions of chlorine with inorganic and organic compounds during water treatment-Kinetics and mechanisms: a critical review. Water Res 42:13–51. https://doi.org/10.1016/j.watres.2007.07.025
34. Deng Y, Englehardt JD (2006) Treatment of landfill leachate by the Fenton process. Water Res 40:3683–3694. https://doi.org/10.1016/j.watres.2006.08.009
35. Deng Y, Englehardt JD (2007) Electrochemical oxidation for landfill leachate treatment. Waste Manag 27:380–388. https://doi.org/10.1016/j.wasman.2006.02.004
36. Deng Y, Zhu X, Chen N et al (2020) Review on electrochemical system for landfill leachate treatment: performance, mechanism, application, shortcoming, and improvement scheme. Sci Total Environ 745:140768. https://doi.org/10.1016/j.scitotenv.2020.140768
37. Devlin TR, Kowalski MS, Pagaduan E et al (2019) Electrocoagulation of wastewater using aluminum, iron, and magnesium electrodes. J Hazard Mater 368:862–868. https://doi.org/10.1016/j.jhazmat.2018.10.017
38. Dia O, Drogui P, Buelna G et al (2017) Electrocoagulation of bio-filtrated landfill leachate: fractionation of organic matter and influence of anode materials. Chemosphere 168:1136–1141. https://doi.org/10.1016/j.chemosphere.2016.10.092
39. Ding W, Zeng X, Hu X et al (2018) Characterization of dissolved organic matter in mature leachate during ammonia stripping and two-stage aged-refuse bioreactor treatment. J Environ Eng 144:1–7. https://doi.org/10.1061/(asce)ee.1943-7870.0001291
40. Dura A (2013) Electrocoagulation for water treatment: the removal of pollutants using aluminium alloys, stainless steels and iron anodes. PhD Thesis, National University of Ireland Maynooth
41. El Kateb M, Trellu C, Darwich A et al (2019) Electrochemical advanced oxidation processes using novel electrode materials for mineralization and biodegradability enhancement of nanofiltration concentrate of landfill leachates. Water Res 162:446–455. https://doi.org/10.1016/j.watres.2019.07.005
42. El Mrabet I, Bellouk H, Bencheqroun Z et al (2021) Treatment of stabilized landfill leachate using coupled Fenton-like and adsorption process onto Moroccan bentonite clay. Desalin Water Treat 240:43–54. https://doi.org/10.5004/dwt.2021.27754
43. Farhat A, Keller J, Tait S, Radjenovic J (2017) Assessment of the impact of chloride on the formation of chlorinated by-products in the presence and absence of electrochemically activated sulfate. Chem Eng J 330:1265–1271. https://doi.org/10.1016/j.cej.2017.08.033
44. Farhat A, Keller J, Tait S, Radjenovic J (2015) Removal of persistent organic contaminants by electrochemically activated sulfate. Environ Sci Technol 49:14326–14333. https://doi.org/10.1021/acs.est.5b02705
45. Feki F, Aloui F, Feki M, Sayadi S (2009) Electrochemical oxidation post-treatment of landfill leachates treated with membrane bioreactor. Chemosphere 75:256–260. https://doi.org/10.1016/j.chemosphere.2008.12.013
46. Fernandes A, Labiadh L, Ciríaco L et al (2017) Electro-Fenton oxidation of reverse osmosis concentrate from sanitary land fill leachate: evaluation of operational parameters. Chemosphere 184:1223–1229. https://doi.org/10.1016/j.chemosphere.2017.06.088
47. Fernandes A, Pacheco MJ, Ciríaco L, Lopes A (2015) Review on the electrochemical processes for the treatment of sanitary landfill leachates: present and future. Appl Catal B Environ 176–177:183–200. https://doi.org/10.1016/j.apcatb.2015.03.052
48. Fernandes A, Santos D, Pacheco MJ et al (2016) Electrochemical oxidation of humic acid and sanitary landfill leachate: Influence of anode material, chloride concentration and current density. Sci Total Environ 541:282–291. https://doi.org/10.1016/j.scitotenv.2015.09.052
49. Fernandes A, Santos D, Pacheco MJ et al (2014) Nitrogen and organic load removal from sanitary landfill leachates by anodic oxidation at Ti/Pt/PbO_2, Ti/Pt/SnO_2-Sb_2O_4 and Si/BDD. Appl Catal B Environ 148–149:288–294. https://doi.org/10.1016/j.apcatb.2013.10.060
50. Foo KY, Hameed BH (2009) An overview of landfill leachate treatment via activated carbon adsorption process. J Hazard Mater 171:54–60. https://doi.org/10.1016/j.jhazmat.2009.06.038
51. Friedrich JM, Ponce-de-León C, Reade GW, Walsh FC (2004) Reticulated vitreous carbon as an electrode material. J Electroanal Chem 561:203–217. https://doi.org/10.1016/j.jelechem.2003.07.019
52. Galvão N, de Souza JB, de Sousa Vidal CM (2020) Landfill leachate treatment by electrocoagulation: effects of current density and electrolysis time. J Environ Chem Eng 8:1–8. https://doi.org/10.1016/j.jece.2020.104368

53. Ghanbari F, Moradi M (2015) A comparative study of electrocoagulation, electrochemical Fenton, electro-Fenton and peroxi-coagulation for decolorization of real textile wastewater: Electrical energy consumption and biodegradability improvement. J Environ Chem Eng 3:499–506. https://doi.org/10.1016/j.jece.2014.12.018
54. Guo Z, Zhang Y, Jia H et al (2022) Electrochemical methods for landfill leachate treatment: a review on electrocoagulation and electrooxidation. Sci Total Environ 806:150529. https://doi.org/10.1016/j.scitotenv.2021.150529
55. Guvenc SY, Dincer K, Varank G (2019) Performance of electrocoagulation and electro-Fenton processes for treatment of nanofiltration concentrate of biologically stabilized landfill leachate. J Water Process Eng 31. https://doi.org/10.1016/j.jwpe.2019.100863
56. Hakizimana JN, Gourich B, Chafi M et al (2017) Electrocoagulation process in water treatment: a review of electrocoagulation modeling approaches. Desalination 404:1–21. https://doi.org/10.1016/j.desal.2016.10.011
57. Han Z, Ma H, Shi G et al (2016) A review of groundwater contamination near municipal solid waste landfill sites in China. Sci Total Environ 569–570:1255–1264. https://doi.org/10.1016/j.scitotenv.2016.06.201
58. Hanif MHM, Kamaruddin MA, Norashiddin FA et al (2022) Landfill leachate treatment technology via electrocoagulation: a review of operating parameters, intensification, and modelling. Desalin Water Treat 260:77–101. https://doi.org/10.5004/dwt.2022.28456
59. Hao X, Dan S, Qian Z et al (2014) Preparation and characterization of PbO_2 electrodes from electro-deposition solutions with different copper concentration. RSC Adv 4:25011–25017. https://doi.org/10.1039/c4ra03235g
60. He H, Zhou Z (2017) Technology electro-Fenton process for water and wastewater treatment. Crit Rev Environ Sci Technol 0:1–32. https://doi.org/10.1080/10643389.2017.1405673
61. Hilles AH, Abu Amr SS, Hussein RA et al (2016) Optimization of leachate treatment using persulfate/H_2O_2 based advanced oxidation process: case study: Deir El-Balah Landfill Site, Gaza Strip, Palestine. Water Sci Technol 73:102–112. https://doi.org/10.2166/wst.2015.468
62. Huda N, Raman AAA, Bello MM, Ramesh S (2017) Electrocoagulation treatment of raw landfill leachate using iron-based electrodes: effects of process parameters and optimization. J Environ Manage 204:75–81. https://doi.org/10.1016/j.jenvman.2017.08.028
63. Ilhan F, Kurt U, Apaydin O, Gonullu MT (2008) Treatment of leachate by electrocoagulation using aluminum and iron electrodes. J Hazard Mater 154:381–389. https://doi.org/10.1016/j.jhazmat.2007.10.035
64. Ingelsson M, Yasri N, Roberts EPL (2020) Electrode passivation, faradaic efficiency, and performance enhancement strategies in electrocoagulation—a review. Water Res 187:116433. https://doi.org/10.1016/j.watres.2020.116433
65. Ishak AR, Hamid FS, Mohamad S, Tay KS (2018) Stabilized landfill leachate treatment by coagulation-flocculation coupled with UV-based sulfate radical oxidation process. Waste Manag 76:575–581. https://doi.org/10.1016/j.wasman.2018.02.047
66. İskurt Ç, Aliyev E, Gengec E et al (2022) Electrochemical oxidation of pretreated landfill leachate nanofiltration concentrate in terms of pollutants removal and formation of by-products. Chemosphere 307:135954. https://doi.org/10.1016/j.chemosphere.2022.135954
67. Jegadeesan C, Somanathan A, Banu R (2023) Sanitary landfill leachate treatment by aerated electrochemical Fenton process. J Environ Manage 337:117698. https://doi.org/10.1016/j.jenvman.2023.117698
68. Jiao Y, Ma L, Tian Y, Zhou M (2020) A flow-through electro-Fenton process using modified activated carbon fiber cathode for orange II removal. Chemosphere 252:126483. https://doi.org/10.1016/j.chemosphere.2020.126483
69. Jones CW (2000) Applications of hydrogen peroxide and derivatives. Org Process Res Dev 4:232–233. https://doi.org/10.1021/op000003w
70. Kabdaşlı I, Arslan-Alaton I, Ölmez-Hancı T, Tünay O (2012) Electrocoagulation applications for industrial wastewaters: a critical review. Environ Technol Rev 1:2–45. https://doi.org/10.1080/21622515.2012.715390
71. Keyikoglu R, Karatas O, Rezania H et al (2021) A review on treatment of membrane concentrates generated from landfill leachate treatment processes. Sep Purif Technol 259:118–182. https://doi.org/10.1016/j.seppur.2020.118182
72. Kim KJ, Kim YJ, Kim JH, Park MS (2011) The effects of surface modification on carbon felt electrodes for use in vanadium redox flow batteries. Mater Chem Phys 131:547–553. https://doi.org/10.1016/j.matchemphys.2011.10.022
73. Košutić K, Dolar D, Strmecky T (2015) Treatment of landfill leachate by membrane processes of nanofiltration and reverse osmosis. Desalin Water Treat 55:2680–2689. https://doi.org/10.1080/19443994.2014.939863

74. Le TS, Dang NM, Tran DT (2021) Performance of coupling electrocoagulation and biofiltration processes for the treatment of leachate from the largest landfill in Hanoi, Vietnam: impact of operating conditions. Sep Purif Technol 255:117677. https://doi.org/10.1016/j.seppur.2020.117677

75. Li J, Yang ZH, Xu HY et al (2016) Electrochemical treatment of mature landfill leachate using Ti/RuO_2-IrO_2 and Al electrode: optimization and mechanism. RSC Adv 6:47509–47519. https://doi.org/10.1039/c6ra05080h

76. Li L, Liu Y (2009) Ammonia removal in electrochemical oxidation: Mechanism and pseudo-kinetics. J Hazard Mater 161:1010–1016. https://doi.org/10.1016/j.jhazmat.2008.04.047

77. Li M, Zhou M, Qin X (2022) A feasible electro-Fenton treatment of landfill leachate diluted by electro-Fenton effluent: evaluation of operational parameters, effect of dilution ratio and assessment of treatment cost. J Water Process Eng 47:102754. https://doi.org/10.1016/j.jwpe.2022.102754

78. Li X, Yang H, Ma H et al (2023) Carbonaceous materials applied for cathode electro-Fenton technology on the emerging contaminants degradation. Process Saf Environ Prot 169:186–198. https://doi.org/10.1016/j.psep.2022.11.037

79. Li Y, Zhang M (2017) Mechanical properties of activated carbon fibers. Act Carbon Fiber Text 49:167–180. https://doi.org/10.1016/B978-0-08-100660-3.00006-7

80. Lindamulla LMLKB, Jayawardene NKRN, Wijerathne WSMSK et al (2022) Treatment of mature landfill leachate in tropical climate using membrane bioreactors with different configurations. Chemosphere 307:136013. https://doi.org/10.1016/j.chemosphere.2022.136013

81. Liu H, Zhao X, Qu J (2010) Electrocoagulation in water treatment. In: Comninellis C, Chen G (eds) Electrochemistry for the environment. Springer, New York, NY, pp 245–262

82. Le Luu T (2020) Post treatment of ICEAS-biologically landfill leachate using electrochemical oxidation with Ti/BDD and Ti/RuO_2 anodes. Environ Technol Innov 20:101099. https://doi.org/10.1016/j.eti.2020.101099

83. Mandal P, Dubey BK, Gupta AK (2017) Review on landfill leachate treatment by electrochemical oxidation: drawbacks, challenges and future scope. Waste Manag 69:250–273. https://doi.org/10.1016/j.wasman.2017.08.034

84. Mandal P, Gupta AK, Dubey BK (2020) Role of inorganic anions on the performance of landfill leachate treatment by electrochemical oxidation using graphite/PbO_2 electrode. J Water Process Eng 33:101119. https://doi.org/10.1016/j.jwpe.2019.101119

85. Martínez-Huitle CA, Brillas E (2008) Electrochemical alternatives for drinking water disinfection. Angew Chem Int Ed 47:1998–2005. https://doi.org/10.1002/anie.200703621

86. Martínez-Huitle CA, Ferro S (2006) Electrochemical oxidation of organic pollutants for the wastewater treatment: direct and indirect processes. Chem Soc Rev 35:1324–1340. https://doi.org/10.1039/b517632h

87. Martínez-Huitle CA, Panizza M (2018) Electrochemical oxidation of organic pollutants for wastewater treatment. Curr Opin Electrochem 11:62–71. https://doi.org/10.1016/j.coelec.2018.07.010

88. Martínez-Villafañe JF, Montero-Ocampo C, García-Lara AM (2009) Energy and electrode consumption analysis of electrocoagulation for the removal of arsenic from underground water. J Hazard Mater 172:1617–1622. https://doi.org/10.1016/j.jhazmat.2009.08.044

89. McBeath ST, Nouri-khorasani A, Mohseni M, Wilkinson DP (2020) In-situ determination of current density distribution and fl uid modeling of an electrocoagulation process and its effects on natural organic matter removal for drinking water treatment. Water Res 171:115404. https://doi.org/10.1016/j.watres.2019.115404

90. Muñoz-Morales M, Ramírez A, Cañizares A et al (2023) Evaluating key properties of carbon materials as cathodes for the electrogeneration of hydrogen peroxide. Carbon N Y 210:118082. https://doi.org/10.1016/j.carbon.2023.118082

91. Nair KM, Kumaravel V, Pillai SC (2021) Carbonaceous cathode materials for electro-Fenton technology: mechanism, kinetics, recent advances, opportunities and challenges. Chemosphere 269:129325. https://doi.org/10.1016/j.chemosphere.2020.129325

92. Naje AS, Ajeel MA, Ali IM et al (2019) Raw landfill leachate treatment using an electrocoagulation process with a novel rotating electrode reactor. Water Sci Technol 80:458–465. https://doi.org/10.2166/wst.2019.289

93. Nidheesh PV, Ganiyu SO, Martínez-huitle CA et al (2023) Recent advances in electro-Fenton process and its emerging applications. Crit Rev Environ Sci Technol 53:887–913. https://doi.org/10.1080/10643389.2022.2093074

94. Nidheesh PV, Oladipo AA, Yasri NG et al (2022) Emerging applications, reactor design and recent advances of electrocoagulation process. Process Saf Environ Prot 166:600–616. https://doi.org/10.1016/j.psep.2022.08.051

95. Nidheesh PV, Gandhimathi R (2012) Trends in electro-Fenton process for water and wastewater treatment: an overview. Desalination 299:1–15. https://doi.org/10.1016/j.desal.2012.05.011

96. Nurhayati E, Bagastyo AY, Hartatik DD, Direstiyani LC (2020) The enhancement of biodegradability index of mature landfill leachate by electrochemical oxidation process using boron-doped diamond and dimensionally stable anode. Res Chem Intermed 46:4811–4822. https://doi.org/10.1007/s11164-020-04242-4
97. Oturan N, Oturan MA (2018) Electro-fenton process: background, new developments, and applications. In: Electrochemical water and wastewater treatment. Elsevier Inc., pp 193–221
98. Oumar D, Patrick D, Gerardo B et al (2016) Coupling biofiltration process and electrocoagulation using magnesium-based anode for the treatment of landfill leachate. J Environ Manage 181:477–483. https://doi.org/10.1016/j.jenvman.2016.06.067
99. Panizza M, Cerisola G (2009) Direct and mediated anodic oxidation of organic pollutants. Chem Rev 109:6541–6569. https://doi.org/10.1021/cr9001319
100. Panizza M, Martinez-Huitle CA (2013) Role of electrode materials for the anodic oxidation of a real landfill leachate—comparison between Ti-Ru-Sn ternary oxide, PbO_2 and boron-doped diamond anode. Chemosphere 90:1455–1460. https://doi.org/10.1016/j.chemosphere.2012.09.006
101. Park YJ, Lee J, Park YS et al (2020) Electrodeposition of high-surface-area IrO_2 films on Ti felt as an efficient catalyst for the oxygen evolution reaction. Front Chem 8:1–9. https://doi.org/10.3389/fchem.2020.593272
102. Patel PS, Bandre N, Saraf A, Ruparelia JP (2013) Electro-catalytic materials (electrode materials) in electrochemical wastewater treatment. Procedia Eng 51:430–435. https://doi.org/10.1016/j.proeng.2013.01.060
103. Pérez G, Saiz J, Ibañez R et al (2012) Assessment of the formation of inorganic oxidation by-products during the electrocatalytic treatment of ammonium from landfill leachates. Water Res 46:2579–2590. https://doi.org/10.1016/j.watres.2012.02.015
104. Pirsaheb M, Azizi E, Almasi A et al (2016) Evaluating the efficiency of electrochemical process in removing COD and NH_4-N from landfill leachate. Desalin Water Treat 57:6644–6651. https://doi.org/10.1080/19443994.2015.1012560
105. Podlasek A, Vaverková MD, Koda E et al (2023) Characteristics and pollution potential of leachate from municipal solid waste landfills: Practical examples from Poland and the Czech Republic and a comprehensive evaluation in a global context. J Environ Manage 332. https://doi.org/10.1016/j.jenvman.2023.117328
106. Qiang Z, Chang JH, Huang CP (2002) Electrochemical generation of hydrogen peroxide from dissolved oxygen in acidic solutions. Water Res 36:85–94. https://doi.org/10.1016/S0043-1354(01)00235-4
107. Ricordel C, Djelal H (2014) Treatment of landfill leachate with high proportion of refractory materials by electrocoagulation: system performances and sludge settling characteristics. J Environ Chem Eng 2:1551–1557. https://doi.org/10.1016/j.jece.2014.06.014
108. Saleem M, Spagni A, Alibardi L et al (2018) Assessment of dynamic membrane filtration for biological treatment of old landfill leachate. J Environ Manage 213:27–35. https://doi.org/10.1016/j.jenvman.2018.02.057
109. Sangeetha A, Gandhimathi R, Nidheesh PV (2023) Treatment of stabilized landfill leachate using pyrite-activated persulfate oxidation process. Process Saf Environ Prot 171:413–422. https://doi.org/10.1016/j.psep.2023.01.012
110. Ben SM, Calmano W, Adin A (2009) Iron-oxidation processes in an electroflocculation (electrocoagulation) cell. J Hazard Mater 171:704–709. https://doi.org/10.1016/j.jhazmat.2009.06.057
111. Shah TM, Ramaswami S, Behrendt J, Otterpohl R (2017) Simultaneous removal of organics and ammonium-nitrogen from reverse osmosis concentrate of mature landfill leachate. J Water Process Eng 19:126–132. https://doi.org/10.1016/j.jwpe.2017.07.024
112. Shan Z, Lu M, Wang L et al (2016) Chloride accelerated Fenton chemistry for the ultrasensitive and selective colorimetric detection of copper. Chem Commun 52:2087–2090. https://doi.org/10.1039/c5cc07446k
113. Shao D, Li X, Xu H, Yan W (2014) An improved stable Ti/Sb-SnO_2 electrode with high performance in electrochemical oxidation processes. RSC Adv 4:21230–21237. https://doi.org/10.1039/c4ra01990c
114. Shokri A, Nasernejad B (2023) Challenges and future roadmaps in heterogeneous electro-Fenton process for wastewater treatment. Water Air Soil Pollut. https://doi.org/10.1007/s11270-023-06139-5
115. Singh UK, Kumar M, Chauhan R et al (2008) Assessment of the impact of landfill on groundwater quality: a case study of the Pirana site in western India. Environ Monit Assess 141:309–321. https://doi.org/10.1007/s10661-007-9897-6
116. Sirés I, Brillas E, Oturan MA et al (2014) Electrochemical advanced oxidation processes: today and tomorrow. A review. Environ Sci Pollut Res 21:8336–8367. https://doi.org/10.1007/s11356-014-2783-1

117. Song G, Atrens A (2003) Understanding magnesium corrosion. A framework for improved alloy performance. Adv Eng Mater 5:837–858. https://doi.org/10.1002/adem.200310405
118. Song K, Ren X, Zhang Q et al (2022) Electrochemical treatment for leachate membrane retentate: performance comparison of electrochemical oxidation and electro-coagulation technology. Chemosphere 303:134986. https://doi.org/10.1016/j.chemosphere.2022.134986
119. Sopaj F, Oturan N, Pinson J et al (2020) Effect of cathode material on electro-Fenton process efficiency for electrocatalytic mineralization of the antibiotic sulfamethazine. Chem Eng J 384:123249. https://doi.org/10.1016/j.cej.2019.123249
120. Sun X, Lv J, Yan Z, Sun Z (2022) A three-dimensional gas diffusion electrode without external aeration for producing H_2O_2 and eliminating amoxicillin using electro-Fenton process. J Environ Chem Eng 10:107301. https://doi.org/10.1016/j.jece.2022.107301
121. Tahreen A, Jami MS, Ali F (2020) Role of electrocoagulation in wastewater treatment: a developmental review. J Water Process Eng 37:101440. https://doi.org/10.1016/j.jwpe.2020.101440
122. Tałałaj IA, Bartkowska I, Biedka P (2021) Treatment of young and stabilized landfill leachate by integrated sequencing batch reactor (SBR) and reverse osmosis (RO) process. Environ Nanotechnol Monit Manag 16. https://doi.org/10.1016/j.enmm.2021.100502
123. Tang J, Yao S, Xiao F et al (2022) Electrochemical oxidation of landfill leachate after biological treatment by electro-Fenton system with corroding electrode of Iron. Int J Environ Res Public Health 19:7745
124. Thanikkal MP, Antony SP (2021) Integrated electro-fenton and membrane bioreactor system for matured landfill leachate treatment. J Hazardous Toxic Radioact Waste 25:1–7. https://doi.org/10.1061/(ASCE)HZ.2153-5515.0000556
125. Tripathy BK, Ramesh G, Debnath A, Kumar M (2019) Mature landfill leachate treatment using sonolytic-persulfate/hydrogen peroxide oxidation: optimization of process parameters. Ultrason Sonochem 54:210–219. https://doi.org/10.1016/j.ultsonch.2019.01.036
126. Turro E, Giannis A, Cossu R et al (2011) Electrochemical oxidation of stabilized landfill leachate on DSA electrodes. J Hazard Mater 190:460–465. https://doi.org/10.1016/j.jhazmat.2011.03.085
127. UNEP (2005) Closing an open dumpsite and shifting from open dumping to controlled dumping and to sanitary land filling
128. UNEP (2021) Roadmap for the progressive closure of dumpsites in Latin America and the Carribean
129. van Genuchten CM, Bandaru SRS, Surorova E et al (2016) Formation of macroscopic surface layers on Fe(0) electrocoagulation electrodes during an extended field trial of arsenic treatment. Chemosphere 153:270–279. https://doi.org/10.1016/j.chemosphere.2016.03.027
130. Varank G, Yazici S, Kaan G, Ahmet D (2020) Concentrated leachate treatment by electro-fenton and electro-persulfate processes using central composite design. Int J Environ Res.https://doi.org/10.1007/s41742-020-00269-y
131. Vasconcelos VM, Santos GOS, Eguiluz KIB et al (2022) Recent advances on modified reticulated vitreous carbon for water and wastewater treatment—a mini-review. Chemosphere 286. https://doi.org/10.1016/j.chemosphere.2021.131573
132. Verma M, Chakraborty S, Kumari S et al (2023) Co-treatment of stabilized landfill leachate and municipal wastewater in a granular activated carbon-sequencing batch reactor (GAC-SBR). Process Saf Environ Prot 174:424–432. https://doi.org/10.1016/j.psep.2023.04.015
133. Verma M, Kumar RN (2018) Coagulation and electrocoagulation for co-treatment of stabilized landfill leachate and municipal wastewater. J Water Reuse Desalin 8:234–243. https://doi.org/10.2166/wrd.2017.102
134. Vlyssides A, Karlis P, Loizidou M et al (2001) Treatment of leachate from a domestic solid waste sanitary landfill by an electrolysis system. Environ Technol (United Kingdom) 22:1467–1476. https://doi.org/10.1080/09593330.2001.11090881
135. Wang Y, Li X, Zhen L et al (2012) Electro-Fenton treatment of concentrates generated in nanofiltration of biologically pretreated landfill leachate. J Hazard Mater 229–230:115–121. https://doi.org/10.1016/j.jhazmat.2012.05.108
136. Wang Y, Liu Y, Liu T et al (2014) Dimethyl phthalate degradation at novel and efficient electro-Fenton cathode. Appl Catal B Environ 156–157:1–7. https://doi.org/10.1016/j.apcatb.2014.02.041
137. Wang Y, Zhao H, Chai S et al (2013) Electrosorption enhanced electro-Fenton process for efficient mineralization of imidacloprid based on mixed-valence iron oxide composite cathode at neutral pH. Chem Eng J 223:524–535. https://doi.org/10.1016/j.cej.2013.03.016

138. Warmadewanthi IDAA, Zulkarnain MA, Ikhlas N et al (2021) Struvite precipitation as pretreatment method of mature landfill leachate. Bioresour Technol Reports 15:100792. https://doi.org/10.1016/j.biteb.2021.100792
139. Xaypanya P, Takemura J, Chiemchaisri C et al (2018) Characterization of landfill leachates and sediments in major cities of Indochina Peninsular countries—heavy metal partitioning in municipal solid waste leachate. Environ MDPI 5:1–24. https://doi.org/10.3390/environments5060065
140. Xu Q, Siracusa G, Di Gregorio S, Yuan Q (2018) COD removal from biologically stabilized landfill leachate using advanced oxidation processes (AOPs). Process Saf Environ Prot 120:278–285. https://doi.org/10.1016/j.psep.2018.09.014
141. Yang Q, Zhong Y, Zhong H et al (2015) A novel pretreatment process of mature landfill leachate with ultrasonic activated persulfate: optimization using integrated Taguchi method and response surface methodology. Process Saf Environ Prot 98:268–275. https://doi.org/10.1016/j.psep.2015.08.009
142. Yang Z, Wu S, Sun H et al (2022) Efficient degradation of organic compounds in landfill leachate via developing bio-electro-Fenton process. J Environ Manage 319:115719. https://doi.org/10.1016/j.jenvman.2022.115719
143. Yasmin C, Lobna E, Mouna M et al (2020) New trend of Jebel Chakir landfill leachate pre-treatment by photocatalytic TiO_2/Ag nanocomposite prior to fermentation using Candida tropicalis strain. Int Biodeterior Biodegrad 146:104829. https://doi.org/10.1016/j.ibiod.2019.104829
144. Yildiz YŞ, Koparal AS, Keskinler B (2008) Effect of initial pH and supporting electrolyte on the treatment of water containing high concentration of humic substances by electrocoagulation. Chem Eng J 138:63–72. https://doi.org/10.1016/j.cej.2007.05.029
145. Zainal SFFS, Aziz HA, Omar FM, Alazaiza MYD (2021) Influence of Jatropha curcas seeds as a natural flocculant on reducing Tin (IV) tetrachloride in the treatment of concentrated stabilised landfill leachate. Chemosphere 285:131484. https://doi.org/10.1016/j.chemosphere.2021.131484
146. Zambrano J, Min B (2020) Electrochemical treatment of leachate containing highly concentrated phenol and ammonia using a Pt/Ti anode at different current densities. Environ Technol Innov 18:100632. https://doi.org/10.1016/j.eti.2020.100632
147. Zamri MFMA, Kamaruddin MA, Yusoff MS et al (2017) Semi-aerobic stabilized landfill leachate treatment by ion exchange resin: isotherm and kinetic study. Appl Water Sci 7:581–590. https://doi.org/10.1007/s13201-015-0266-2
148. Zhang BT, Zhang Y, Teng Y, Fan M (2015) Sulfate radical and its application in decontamination technologies. Crit Rev Environ Sci Technol 45:1756–1800. https://doi.org/10.1080/10643389.2014.970681
149. Zhang H, Li Y, Wu X et al (2010) Application of response surface methodology to the treatment landfill leachate in a three-dimensional electrochemical reactor. Waste Manag 30:2096–2102. https://doi.org/10.1016/j.wasman.2010.04.029
150. Zhang H, Ran X, Wu X, Zhang D (2011) Evaluation of electro-oxidation of biologically treated landfill leachate using response surface methodology. J Hazard Mater 188:261–268. https://doi.org/10.1016/j.jhazmat.2011.01.097
151. Zhang H, Zhang D, Zhou J (2006) Removal of COD from landfill leachate by electro-Fenton method. J Hazard Mater B135:106–111. https://doi.org/10.1016/j.jhazmat.2005.11.025
152. Zhang Z, Meng H, Wang Y et al (2018) Electrochimica Acta Fabrication of graphene @ graphite-based gas diffusion electrode for improving H_2O_2 generation in electro-Fenton process. Electrochim Acta 260:112–120. https://doi.org/10.1016/j.electacta.2017.11.048
153. Zhao G, Pang Y, Liu L et al (2010) Highly efficient and energy-saving sectional treatment of landfill leachate with a synergistic system of biochemical treatment and electrochemical oxidation on a boron-doped diamond electrode. J Hazard Mater 179:1078–1083. https://doi.org/10.1016/j.jhazmat.2010.03.115
154. Zhou B, Yu Z, Wei Q et al (2016) Electrochemical oxidation of biological pretreated and membrane separated landfill leachate concentrates on boron doped diamond anode. Appl Surf Sci 377:406–415. https://doi.org/10.1016/j.apsusc.2016.03.045
155. Zhu Z, Zhao Y, Zhu Y et al (2021) Efficient treatment of mature landfill leachate with a novel composite biological trickle reactor developed using refractory domestic waste and aged refuse. J Clean Prod 305. https://doi.org/10.1016/j.jclepro.2021.127194
156. Zielińska M, Kulikowska D, Stańczak M (2020) Adsorption—Membrane process for treatment of stabilized municipal landfill leachate. Waste Manag 114:174–182. https://doi.org/10.1016/j.wasman.2020.07.011

© The Author(s), under exclusive license to Springer Nature Switzerland AG 2024
S. Souabi, A. Anouzla (eds.), *Landfill Leachate Treatment Techniques*, Springer Water
https://doi.org/10.1007/978-3-031-63157-3_13

13. Mini-Review on Aged Refuse Bioreactors for Landfill Leachate Management

Ronei de Almeida[1] ✉ and André Luís de Sá Salomão[2] ✉
(1) School of Chemistry, Universidade Federal do Rio de Janeiro, Rio de Janeiro, Brazil
(2) Department of Sanitary and Environmental Engineering, Rio de Janeiro State University, Rio de Janeiro, Brazil

✉ **Ronei de Almeida (Corresponding author)**
Email: ronei@eq.ufrj.br

✉ **André Luís de Sá Salomão**
Email: andre@andresalomao.com

Abstract
Landfill leachate contains a wide range of macro and micro-pollutants of varying concentrations. Hence, its treatment remains a major socio-environmental problem in the municipal solid waste (MSW) management chain. It is essential to manage the landfill leachate to protect water resources and public health. Aged refuse bioreactors (ARB) can be a cost-effective solution to treat MSW leachate, as they use low-cost materials and adsorptive/ biological mechanisms to degrade and remove contaminants from

landfill wastewater. The bioreactors are typically constructed from concrete or plastic and contain a mixture of aged MSW, used as support media for microorganisms, which degrades the pollutants. The authors reviewed recent studies on ARB for landfill leachate treatment in this chapter. Overall, organic matter removal is above 80%, and ammonia nitrogen can be 100% eliminated after ARB treatment. ARB-treated effluent could be directed to a constructed wetland (CW). The CW is effective at further reducing contaminants through biological and physical processes. In this sense, treatment layouts are proposed, and the scope for future studies is recommended.

Keywords Biodegradation – Biotreatment – Developing countries – Municipal solid waste – Small landfills – Sorption

---

13.1 Introduction

Landfill leachate is polluting and complex wastewater and should be adequately managed before it reaches the environment [29]. Treatment techniques comprise physical, chemical, and biological processes and their combinations. Overall, the leachate treatment chain is designed with several treatment steps aiming to meet landfill wastewater disposal requirements. Despite the low biodegradability of landfill leachate, especially those generated in mature landfills (more than ten years), biological treatments are still needed for carbon and nitrogen removal or uptake [16, 17, 19 20].

The Aged refuse bioreactor (ARB) presents a cost-effective biological approach and a promising technology for effectively removing diverse pollutants from landfill wastewater. It has proven to be a successful technique for treating various types of wastewaters, including sanitary effluents, fishpond and sugar mill wastewaters, coking

wastewater, and livestock waste [2, 25, 52]. ARB method utilizes stabilized material or aged refuse derived from municipal solid waste in landfills for over eight years. The aged refuse material is a sustainable and low-cost medium within bioreactors, offering an environmentally friendly waste management solution. Also, given the challenges of constructing new landfills, repurposing and extracting aged refuse from existing landfills for leachate treatment presents a viable and pragmatic alternative [37].

ARB adheres to the principle of waste control by waste and demonstrates its viability in practical applications [9]. This technology is advantageous for smaller and medium-sized landfills due to its cost-efficiency, environmental compatibility, and straightforward operation. Previous work reviewed the utilization of ARB for landfill management [28]. However, the literature lacks an up-to-date study on this topic; since then, much attention has been given to the applicability of ARB technology.

This chapter reviews recent studies on aged refuse bioreactors for landfill leachate management. The ARB aspects and operating conditions are introduced. Next, a brief overview of ARB for leachate wastewater management is presented. Last, a treatment scheme is suggested for leachate treatment in small landfills in developing nations.

13.2 Aged Refuse Bioreactors

13.2.1 Construction and Operation

The ARB's configuration resembles a trickling filter. This system fosters the growth of bacteria attached to surfaces while enabling the downward flow of wastewater due to gravitational forces. Support materials such as crushed stones, slag, pumice, and plastics are generally used in trickling filters in sewage treatment plants. Conversely, in

the case of ARBs, aged refuse serves as the medium. In contrast to trickling filters, ARBs produce reduced amounts of sludge and eliminate the necessity for an additional sedimentation unit to manage sludge removal [28, 52].

While circular or cylindrical shapes were typically employed for laboratory-scale reactors to facilitate water distribution, rectangular or square configurations were adopted in pilot and full-scale experiments for their simplicity in construction, space-saving and operation. The height/inner diameter (H/Ø) ratio of aged refuse reactors tested on a scale range from 1.5 to 4.0 [28]. Figure 13.1 illustrates a representative ARB used in experimental evaluations [25, 27].

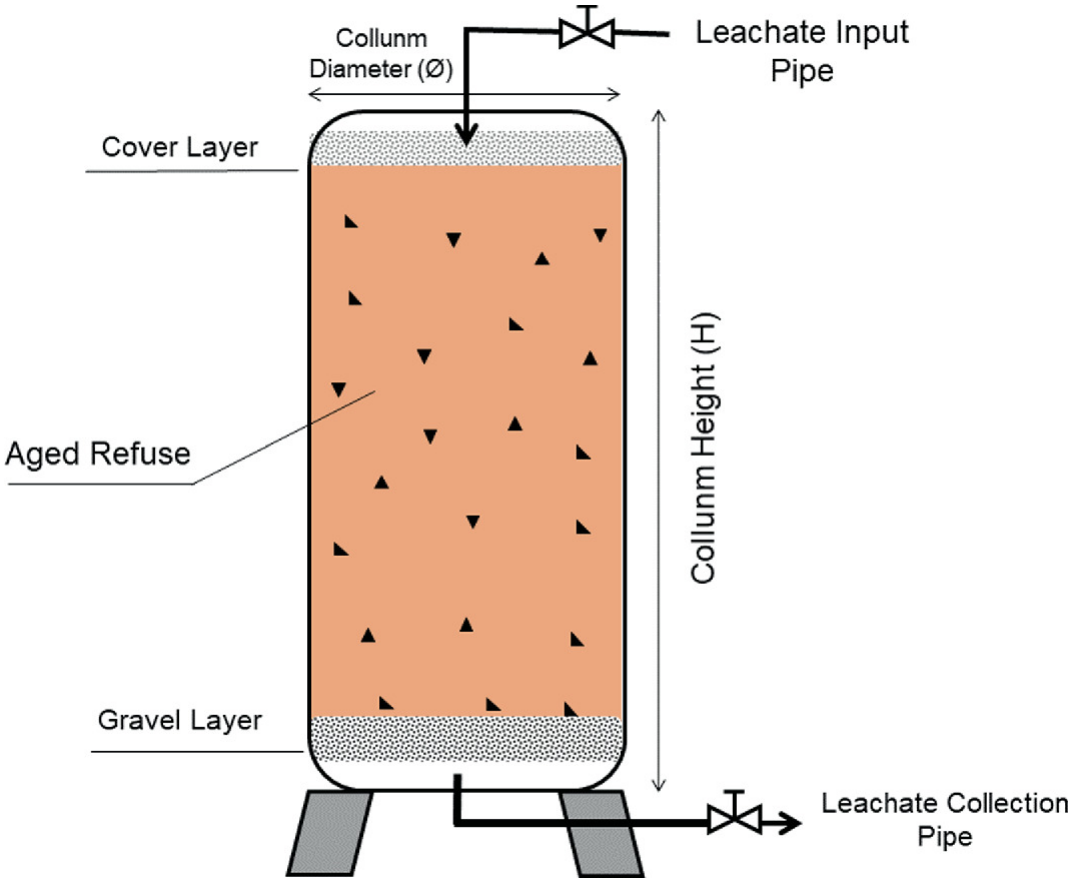


Fig. 13.1 Schematic representation of a circular aged refuse bioreactor

Within the ARB setup, leachate continually flows through the refuse layer, facilitated by the leachate collection and exhaust systems. The introduction of ambient air occurs organically via the leachate collection pipes designed with slack flow features, which results in a primarily semi-aerobic bioreactor environment [31].

It is widely acknowledged that the presence of aerobic or anaerobic conditions profoundly influences the efficiency of ARBs in removing pollutants from leachate. Recognizing that aerobic conditions could expedite refuse stabilization, diminish leachate and methane gas generation, and enhance leachate quality, the concept of a semi-aerobic landfill process emerged in the 1990s at Fukuoka University in Japan [46]. In this context, the semi-aerobic ARB concept was introduced. This has been explored in recent years [30, 38, 49]. Therefore, regarding operation mode, ARBs are categorized as anaerobic, aerobic, and semi-aerobic aged refuse bioreactors.

The treatment mechanisms involve two main processes: adsorption onto aged refuse and biodegradation. The aged refuse serves as a supporting medium for the growth and development of a specific degrading microbiota. As a result, microbial activity thrives, removing/ biodegrading pollutants from the introduced wastewater. Interestingly, adsorption rather than biodegradation was identified as the primary mechanism in the anaerobic condition [43].

Furthermore, the temperature gradient existing between the interior and exterior of the system initiates a cycle of air conduction. Within aerobic and semi-aerobic ARB might exist distinct anaerobic, anoxic, and aerobic zones, which work synergistically to expedite the conversion and breakdown of organic compounds, ammonia, and other pollutants present in the landfill leachate [27, 50].

13.2.2 Operating Parameters and Influencing Conditions

The biochemical environment of the aged refuse bioreactor (i.e., anaerobic and aerobic), temperature, and hydraulic load rate are identified as the main parameters influencing ARB performance in treating landfill wastewater.

Sun et al. [45] showed that organics and ammonia are removed more effectively when ARB aeration is adopted. The chemical oxygen demand (COD) removal improved as aeration increased, and at an optimal aeration rate of 670 L m^{-3}, total nitrogen removal approached 100%. In sum, it was observed that the aeration helps achieve simultaneous nitrification–denitrification, contributing to ammonia removal. However, it is crucial to consider that implementing an aeration system can lead to increased treatment costs, making a comprehensive cost–benefit analysis of this strategy essential [44, 45]. Exploring intermittent aeration could offer a cost-effective approach to ARB operation and merits further investigation in future research endeavors.

Temperature plays a pivotal role in treatment performance in aged refuse bioreactors. In a study conducted by [47], the effectiveness of carbon and nitrogen removal in ARB was assessed at varying temperatures of 35, 45, and 55 ºC. Notably, the highest treatment performance and the swiftest carbon mineralization were observed at the elevated temperature of 55 ºC, indicating the significance of thermophilic conditions. Also, volatilization significantly impacted ammonia nitrogen removal under aerobic conditions, resulting in more than 60% nitrogen in organic forms [47].

On the other hand, a primary obstacle faced when treating leachate using outdoor ARB pertains to the cold winter temperatures, during which microbial activity is notably subdued. A potential remedy could involve the

implementation of leachate recirculation as an alternative approach to attain the necessary bioreactor efficiency even under cold temperature conditions [26].

Another possibility, proper management of the hydraulic load rate (HLR) is essential for achieving efficient pollutant removal and ensuring the bioreactor system's long-term stability and environmental integrity [27]. A previous study showed that when the HLR of ARB increases from 8.5 L m^{-3} d^{-1} to 17 L m^{-3} d^{-1}, the removal efficiency of ammonia nitrogen and 5-days biochemical oxygen demand (BOD_5) change less, and that of COD and total nitrogen increase significantly, which indicates the HLR affects the reactor performance. In another work, the reactor achieved high removal efficiencies under shock loading [43]. Besides, an inappropriate HLR might lead to imbalances in the microbial community and increase the potential for odors and air pollution [49].

13.3 Leachate Treatment Performance of Aged Refuse Bioreactors

The pollution parameters monitored in ARB treatability tests include BOD_5, COD, Absorbance at 254 nm (UV_{254})—linked to recalcitrant organics present in leachate samples [18], color, ammonia nitrogen, and total nitrogen. COD and UV_{254}—indicators for organic matter and ammonia nitrogen are typical polluting parameters evaluated in recent studies (Table 13.1).

Table 13.1 Studies from the past five years on ARB for landfill leachate management

Treatment configuration	Operating conditions	Removal Efficiency (%)			References
		COD	UV_{254}	N-NH_3	
Ozonation + ARB	Semi-aerobic, 178 days of operation, flowrate of 2 L d^{-1}	> 80	> 70	> 90	[35]
ARB	Semi-aerobic, 30 days of operation, flowrate of 1 L h^{-1}	> 70	> 70	> 80	[49]
ARB + 3D-EF	Anaerobic, ten weeks, hydraulic loading ratio of 15 L m^{-3} d^{-1}	52	n.d	54	[34]
ARB	Semi-aerobic, 300 days of operation, flowrate of 2 L d^{-1}	85–95	37–62	> 90	[12]
ARB + Fenton	Semi-aerobic, HRT of 12 h, flowrate of 60 – 70 mL min^{-1}	85.15	78.89	n.d	[11]
Two-stage ARB	Anaerobic, > 30 weeks of operation, hydraulic load rate of 10 to 50 L m^{-3} d^{-1}	85	n.d	97.4	[36]
ARB + ozonation	Semi-aerobic, HRT of 12 h, flowrate of 60 – 70 mL min^{-1}	> 50	5–20	n.d	[13]
ARB	Anaerobic, one year of operation, hydraulic load rate of 18 L m^{-3} d^{-1},	80	n.d	n.d	[5]
ARB	Anaerobic, two treatment cycles, hydraulic load rate of 55 L m^{-3} d^{-1}	75–95	n.d	n.d	[22]
Air stripping + two-stage ARB	Anaerobic, monitoring time not identified, hydraulic load rate of	29–80	n.d	n.d	[21]

Treatment configuration	Operating conditions	Removal Efficiency (%)			References
		COD	UV_{254}	N-NH_3	
	0.10 L kg^{-1} aged refuse d^{-1}				

3D-EF = three-dimensional electrode eletro-Fenton, ARB = aged refuse bioreactor, BOD = biochemical oxygen demand, COD = chemical oxygen demand, HRT = hydraulic retention time. N-NH_3 = ammonia nitrogen. n.d = not determined.

Overall, COD removal efficiency is greater than 80% [5, 10, 35, 36], while the UV_{254} removal ranges from 5 to 70% [9, 49]. Ammonia and total nitrogen were effectively removed using ARB, especially semi-aerobic ones. Values of removal efficiency ranged from 50 to 100% [34].

Other parameters not presented in Table 13.1, such as color and heavy metals, were assessed in recent studies. Nájera-Aguilar et al. [36] reported color and BOD removals of 86.1 and 87.9%. In another work, the removal of cadmium, lead, and copper was 89, 82, and 91%, respectively [22].

The findings from the table indicate that the performance of different treatment configurations varies based on factors such as operating conditions, treatment duration, and the specific removal parameters targeted. Hydraulic load rates, anaerobic/semi-aerobic/aerobic conditions, and the combination of treatment methods all contribute to the overall effectiveness of leachate treatment using aged refuse bioreactors [44, 45, 49].

Regarding treatment combinations, it is observed that advanced oxidation processes are preferred due to the necessity of eliminating recalcitrant organics after ARB treatment. Fenton and ozonation were investigated in integrated systems [11–13, 34] reported that semi-aerobic

ARB cannot remove poly-cyclic aromatic and polyphenol substances. Analyzing combined advanced treatments, they registered that ozonation efficiently removes refractory dissolved organic matter from ARB leachate; however, the Fenton reagent is comparatively better at mineralizing the organics.

In a recent study, the ARB + three-dimensional electrode electro-Fenton (3D-EF) system was developed at a laboratory scale. The optimum operating conditions were 15 L m^{-3} d^{-1} hydraulic loading rate for ARB, Fe^{2+} concentration 1.0 mM, initial pH 3.0, and current density 30 mA cm^{-2} for 3D-EF. COD, ammonia nitrogen, and color removal ratios were 96.2%, 94.3%, and 93.6%, respectively. The ARB mechanism effectively targeted a significant portion of the organic matter. At the same time, the 3D-EF system's contribution encompassed the removal of recalcitrant substances and the refining the final effluent via post-treatment polishing. Leachate ecotoxicity was reduced after undergoing the hybrid treatment [34].

Moreover, a recent investigation has demonstrated the potential efficacy of ARB in the on-site removal of antibiotics and antibiotic-resistant genes (ARGs) from leachate. The collective removal performance of antibiotics reached approximately 76.75%, with notably high removal rates observed for sulfanilamide and macrolide compounds, exceeding 80% elimination efficiency. Notably, within the scope of targeted ARGs, crucial genes responsible for tetracycline and macrolide resistance (tetM, tetQ, and ermB) were successfully eliminated from landfill leachate [43].

13.4 Leachate Treatment Layout: Low-Cost Configurations for Small Landfill Sites in Developing Countries

Landfills in large cities from developing nations (e.g., Brazil and China) invest in advanced landfill leachate treatment technologies, such as nanofiltration and reverse osmosis, as in developed countries [8, 24]. Meanwhile, the biological process is still commonly applied since this is a simple and economical technology, especially in small municipalities [7, 14, 33].

A low-cost treatment system for a small landfill while incorporating an ARB could be a promising and sustainable approach in developing countries. ARB might be constructed using readily available materials and local labor for maintenance, supporting the community and minimizing treatment costs [2, 10]. It is important to note that monitor the treatment system's performance and adjust parameters is still necessary even in small demands. Maintenance should include periodic cleaning, ensuring the proper functioning of components, check the hydraulic loading rate, reactor temperature, and addressing any issues promptly. Moisture, aeration, and aged refuse composition should also be monitored and controlled [15, 31].

The ARB-treated effluent could be directed to a specifically designed constructed wetland (CW) to carry out the final treatment step. CW is effective in removing contaminants resulting from chemical, physical and biological interactions in the complex formed by the “substrate-microbiota-plants”, where the microbial biofilm is considered a fundamental part of the sorption, biodegradability and decontamination processes [40]. Furthermore, the symbiosis between macrophyte roots and microorganisms facilitates nutrient uptake by plants and leads to an exchange of metabolic products. The roots of aquatic macrophytes (rhizosphere) play a fundamental role, as oxygen transfer can provide a favorable environment for the microorganism’s development that act in response to biological treatment [40].

The support media create empty spaces that serve as drainage channels, facilitating the flow of effluent, providing an ideal place for the formation of microbial biofilm, in addition to providing support for plants and macrophytes [23]. The selection of support media depends on some factors (flow, drainage, types of macrophytes, among other factors) and different types can be used, the most common being gravel (different sizes), soil and sand, which can also be used in layers or zones [4, 42]. Leverage natural processes like soil filtration and wetland treatment, reducing the need for complex equipment [4, 41, 51].

Constructed wetlands can also be classified according to the type of flow into two groups, superficial and subsurface. Subsurface flow CW can be categorized according to the direction of hydraulic flow (vertical or horizontal) [39]. In surface flow, the effluent flows above the supporting medium (apparent), while in subsurface flow, the effluent flows below the surface of the substrate. In vertical hydraulic flow, the effluent is distributed on the surface of the support layer, gradually flowing through all layers of the constructed bed [32]. In horizontal hydraulic flow, the effluent drains the filling material and the roots of the macrophytes horizontally.

WCs can also be combined, being classified as hybrid systems [48]. Hybrid CWs generally have higher treatment efficiencies than non-hybrid systems, especially for total nitrogen, but typically require more space and are more expensive to build [3].

Different mechanisms are responsible for ammonia removal in constructed wetlands: absorption by plants and other living microorganisms; nitrification (under aerobic conditions); volatilization (in FWS CWs); anammox (under anaerobic conditions); and adsorption (very limited) [4]. According to the literature, organic matter removal in CWs occurs mainly due to sedimentation, filtration and biological decomposition by microorganisms under aerobic,

facultative and anaerobic conditions [4, 40, 48]. The biological removal of phosphorus is carried out by the microbiota and macrophytes (as long as they are harvested), however they are considered low. Particulate phosphorus can be removed by filtration on a support medium or by sedimentation, when in surface flow [4, 40].

It must be considered that young leachate is biodegradable and effectively treated by biological processes; however, biological systems might lose their effectiveness with leachate aging [6]. Therefore, a lime treatment could be incorporated for ammonia and refractory organic removal in the long run. This physicochemical treatment is recognized for its cost-effectiveness. Lime is often readily available, making it a practical option for leachate treatment, particularly in areas where other treatment methods might be financially challenging [1, 16].

Figure 13.2 shows the proposed leachate treatment schemes. By combining an ARB with other natural treatments and/or low-cost physicochemical processes, the suggested treatment chains can effectively handle leachate from a small landfill while being environmentally friendly and economically sustainable.

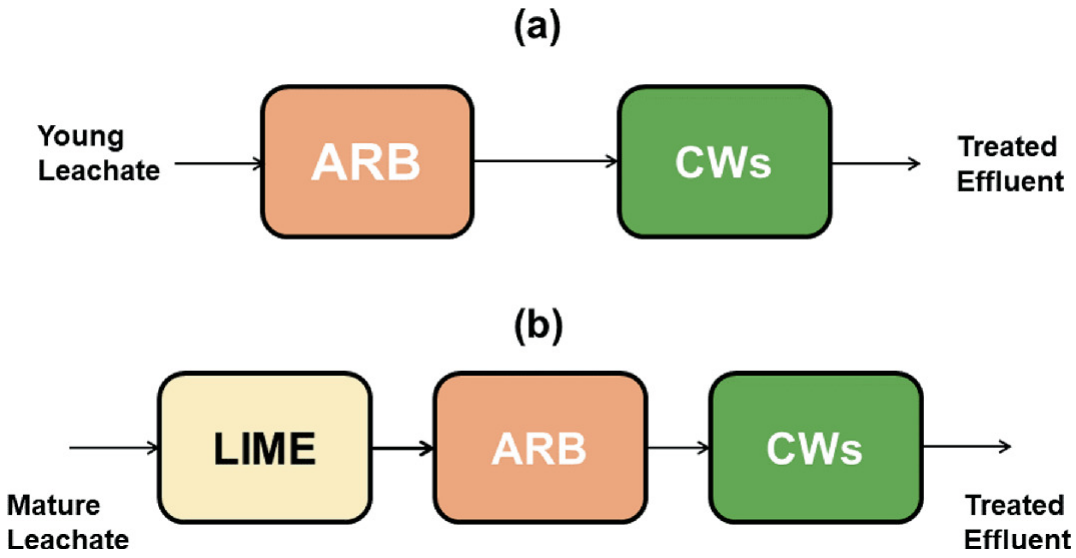


Fig. 13.2 Proposed low-cost treatment configurations for landfill leachate management. **a** Young landfill leachate management. **b** Mature landfill leachate management. ARB = aged refuse bioreactor. CWs = constructed wetlands. LIME = lime treatment

13.5 Conclusions

Effective landfill leachate management is imperative for safeguarding water resources, life in aquatic environments and public health. An exploration of recent research focusing on ARB for treating landfill leachate was undertaken in this chapter. ARB aligns with the principle of "waste control through waste," demonstrating its practicability on a full scale. Overall, organic matter removal is above 80%, and ammonia nitrogen can be wholly eliminated after ARB treatment. Integrating ARB-treated effluent into a leachate treatment sequence could involve routing it to CWs, ensuring compliance with discharge standards. The proposed approach is grounded in surveyed literature and the author's prior expertise. Nevertheless, its viability necessitates rigorous evaluation encompassing technical and economic considerations at laboratory and pilot scales before implementation. This endeavor seeks to delineate future research avenues and contribute to addressing leachate management challenges in landfills from emerging economies.

References

1. Almeida R, Campos JC, Oroski FA (2020) Techno-economic evaluation of landfill leachate treatment by hybrid lime application and nanofiltration process. Detritus, 10(10):170–181. https://doi.org/10.31025/2611-4135/2020.13897
2. Anijiofor SC, Daud NN, Idrus S, Che Man H (2018) Recycling of fishpond wastewater by adsorption of pollutants using aged refuse as an alternative low-cost adsorbent. Sustain Environ Res 28(6):315–321. https://doi.org/10.1016/j.serj.2018.05.005

3. Bakhshoodeh R, Alavi N, Paydary P (2017) Composting plant leachate treatment by a pilot-scale, three-stage, horizontal flow constructed wetland in central Iran. Environ Sci Pollut Res 24:23803–23814
4. Bakhshoodeh R, Alavi N, Oldham C, Santos RM, Babaei AA, Vymazal J, Paydary P (2020) Constructed wetlands for landfill leachate treatment: a review. Ecol Eng 146:105725. https://doi.org/10.1016/j.ecoleng.2020.105725
5. Bautista JA (2018) Biorreactor empacado con materiales estabilizados (beme), como pretratamiento para lixiviados de rellenos sanitarios. Revista Mexicana de Ingeniería Química 17(2):561–571. https://doi.org/10.24275/uam/izt/dcbi/revmexingquim/2018v17n2/Bautista
6. Bouaouda S, Souabi S, Bouyakhsass R, Taleb A, Madinzi A (2023) Techniques for treating leachate discharges: a critical review. Euro-Mediterranean J Environ Integr 8(3):573–599. https://doi.org/10.1007/s41207-023-00366-2
7. Cavalheri PS, Machado BS, da Silva TF, Rodrigues JBG, Gozzi, F, Magalhães Filho FJC, de Oliveira S, Cavalcante RP, Machulek Junior A (2023) Optimization of a combined system of vertical flow constructed wetland and solar photo-fenton for Ketoprofen removal in sewage and landfill leachate. Available at SSRN: https://ssrn.com/abstract=4524316 or https://doi.org/10.2139/ssrn.4524316
8. Chen F, Li G, Li X, Wang H, Wu H, Li J, Li C, Li W, Zhang L, Xi B (2021) The cotreatment of old landfill leachate and domestic sewage in rural areas by deep subsurface wastewater in filtration system (SWIS): performance and bacterial community**. Environ Pollut 274:115800. https://doi.org/10.1016/j.envpol.2020.115800
9. Chen W, Li Q (2020) Elimination of UV-quenching substances from MBR- and SAARB-treated mature landfill leachates in an ozonation process: a comparative study. Chemosphere 242:125256. https://doi.org/10.1016/j.chemosphere.2019.125256
10. Chen W, Gu Z, Ran G, Li Q (2021) Application of membrane separation technology in the treatment of leachate in China: a review. Waste Manage 121:127–140. https://doi.org/10.1016/j.wasman.2020.12.002
11. Chen W, He C, Gu Z, Wang F, Li Q (2020) Molecular-level insights into the transformation mechanism for refractory organics in landfill leachate when using a combined semi-aerobic aged refuse biofilter and chemical oxidation process. Sci Total Environ 741:140502. https://doi.org/10.1016/j.scitotenv.2020.140502

12. Chen W, Wang F, Gu Z, Li Q (2020) Recovery of efficient treatment performance in a semi-aerobic aged refuse biofilter when treating landfill leachate: washing action using domestic sewage. Chemosphere 245:125618. https://doi.org/10.1016/j.chemosphere.2019.125618
13. Chen W, Zhang A, Jiang G, Li Q (2019) Transformation and degradation mechanism of landfill leachates in a combined process of SAARB and ozonation. Waste Manage 85:283–294. https://doi.org/10.1016/j.wasman.2018.12.038
14. Costa AM, Alfaia RGSM & Campos JC (2019) Landfill leachate treatment in Brazil—an overview. J Environ Manage 232(October 2018):110–116. https://doi.org/10.1016/j.jenvman.2018.11.006
15. Daud NN, Mansor NM, Ahsan A, Idrus S (2015). Leachate treatment using aged refuse (AR) as a biofilter medium. Pol J Environ Stud 24(2):605–609. https://doi.org/10.15244/pjoes/25364
16. de Almeida R, Campos JC (2020) Análise tecnoeconômica do tratamento de lixiviado de aterro sanitário. Revista Ineana 8(1):6–27
17. de Almeida R, Bila DM, Quintaes BR, Campos JC (2020) Cost estimation of landfill leachate treatment by reverse osmosis in a Brazilian landfill. Waste Manage Res J Int Solid Wastes Public Cleansing Assoc ISWA 38(10):1087–1092. https://doi.org/10.1177/0734242X20928411
18. de Almeida R, Costa AM, Oroski FA, Campos JC (2019) Evaluation of coagulation–flocculation and nanofiltration processes in landfill leachate treatment. J Environ Sci Health Part A Toxic/Hazard Subst Environ Eng 54(11):1091–1098. https://doi.org/10.1080/10934529.2019.1631093
19. de Almeida R, Couto JM, Gouvea RM, Oroski FA, Bila DM, Quintaes BR, Campos JC (2020) Nanofiltration applied to the landfill leachate treatment and preliminary cost estimation. Waste Manage Res J Sustain Circular Econ 38(10):1119–1128. https://doi.org/10.1177/0734242X20933333
20. de Almeida R, Porto RF, Quintaes BR, Bila DM, Lavagnolo MC, Campos JC (2023) A review on membrane concentrate management from landfill leachate treatment plants: the relevance of resource recovery to close the leachate treatment loop. Waste Manage Res J Int Solid Wastes Public Cleansing Assoc ISWA 41(2):264–284. https://doi.org/10.1177/0734242X221116212
21. Ding W, Zeng X, Hu X, Deng Y, Hossain MN, Chen L (2018) Characterization of dissolved organic matter in mature leachate during ammonia stripping and two-stage aged-refuse bioreactor treatment. J Environ Eng 144(1):1–7. https://doi.org/10.1061/(ASCE)EE.1943-7870.0001291

22. Erabee KI, Ethaib S (2018) Treatment of contaminated landfill leachate using aged refuse biofilter medium. Orient J Chem 34(3):1441–1450. https://doi.org/10.13005/ojc/340334
23. Fadanelli LEA, Andrade AGD, Wiecheteck GK, Döll MMR (2019) Considerations on design and implementation parameters of domestic wastewater treatment by subsurface flow constructed *wetlands*. Engenharia Sanitária e Ambiental 24:809–819
24. Gripa E, Dario SDA, Almeida R, Valéria F, Campos JC (2023) Landfill leachate treatment by high-presssure membranes and advanced oxidation techniques with a focus on ecotoxicity and by-products management: a review. Process Saf Environ Prot. https://doi.org/10.1016/j.psep.2023.03.074
25. Gutiérrez-Hernández RF, Nájera-Aguilar HA, Araiza-Aguilar JA, Martínez-Salinas RI, García-Lara CM, González-Vázquez U, Cruz-Salomón A (2021) Novel treatment of sugar mill wastewater in a coupled system of aged refuse filled bioreactors (ARFB): full-scale. Processes 9(3):516. https://doi.org/10.3390/pr9030516
26. Han Z-Y, Liu D, Li Q-B (2013) A removal mechanism for organics and nitrogen in treating leachate using a semi-aerobic aged refuse biofilter. J Environ Manage 114:336–342. https://doi.org/10.1016/j.jenvman.2012.10.026
27. Han Z-Y, Liu D, Li Q-B, Li G-Z, Yin Z-Y, Chen X, Chen J-N (2011) A novel technique of semi-aerobic aged refuse biofilter for leachate treatment. Waste Manage 31(8):1827–1832. https://doi.org/10.1016/j.wasman.2011.03.016
28. Hassan M, Xie B (2014) Use of aged refuse-based bioreactor/biofilter for landfill leachate treatment. Appl Microbiol Biotechnol 98(15):6543–6553. https://doi.org/10.1007/s00253-014-5813-5
29. Hogland M, Āriņa D, Kriipsalu M, Jani Y, Kaczala F, de Sá Salomão AL, Orupõld K, Pehme KM, Rudoviča V, Denafas G, Burlakovs J, Vincēviča-Gaile Z, Hogland W (2018) Remarks on four novel landfill mining case studies in Estonia and Sweden. J Mater Cycles Waste Manage 20:1355–1363
30. Huang Y (2022) Degradation of refractory organic matter in the effluent from a semi-aerobic aged refuse biofilter-treated landfill leachate by a nano-Fe_3O_4 enhanced zonation process. Waste Manage Res J Sustain Circular Econ 40(8):1242–1255. https://doi.org/10.1177/0734242X211066229
31. Jiang G, Liu D, Chen W, Ye Z, Liu H, Li Q (2017) Impact of vent pipe diameter on characteristics of waste degradation in semi-aerobic

bioreactor landfill. Waste Manage Res J Sustain Circular Econ 35(10):1064–1071. https://doi.org/10.1177/0734242X17723979

32. Lana LCO, Moraes DC, Von Sperling M, Morato MLN, Vasconcellos GR, Paraense MO, Moreira TPA (2013) Performance of a single stage vertical flow constructed wetland system treating raw domestic sewage in Brazil. Water Sci Technol 68(7):1599–1606. https://doi.org/10.2166/wst.2013.408

33. Lino AM, Ismail KAR, Casta JA (2023) Municipal solid waste treatment in Brazil : a comprehensive review. 11(March). https://doi.org/10.1016/j.nexus.2023.100232

34. Lu M (2021) Advanced treatment of aged landfill leachate through the combination of aged-refuse bioreactor and three-dimensional electrode electro-Fenton process. Environ Technol 42(11):1669–1678. https://doi.org/10.1080/09593330.2019.1677781

35. Mu S, Chen X, Song B, Wu C, Li Q (2022) Enhanced performance and mechanism of the combined process of ozonation and a semiaerobic aged refuse biofilter for mature landfill leachate treatment. Chemosphere 308(P3):136432. https://doi.org/10.1016/j.chemosphere.2022.136432

36. Nájera-Aguilar HA, Gutiérrez-Hernández RF, Bautista-Ramírez J, Martínez-Salinas RI, Escobar-Castillejos D, Borraz-Garzón R, Rojas-Valencia MN, Giácoman-Vallejos G (2019) Treatment of low biodegradability leachates in a serial system of aged refuse-filled bioreactors. Sustainability (Switzerland) 11(11):1–16. https://doi.org/10.3390/su11113193

37. Nájera-Aguilar HA, Mayorga-Santis R, Gutiérrez-Hernández RF, Araiza-Aguilar JA, Martínez-Salinas RI, García-Lara CM, Rojas-Valencia MN (2021) Aged refuse filled bioreactor using like a biological treatment for sugar mill wastewater. Sugar Tech 23(1):201–208. https://doi.org/10.1007/s12355-020-00881-4

38. Pan X, Chen M, Wang F, Li Q (2020) Effect of biochar addition on the removal of organic and nitrogen pollutants from leachate treated with a semi-aerobic aged refuse biofilter. Waste Manage Res J Sustain Circular Economy 38(10):1176–1184. https://doi.org/10.1177/0734242X20957411

39. Pedescoll A, Rodríguez L, Sarañana AA, Hijosavalsero M, Bécares E (2016) Microfaunal community in horizontal constructed *wetlands* with different design configurations. Ecol Eng 91:16–23. https://doi.org/10.1016/j.ecoleng.2016.02.006

40. Rabello VM, Teixeira LCRS, Gonçalves APV, de Sá Salomão AL (2019) The efficiency of constructed wetlands and algae tanks for the removal of pharmaceuticals and personal care products (PPCPs): a systematic review. Water Air Soil Pollut 230(10):236

41. Ravikumar Y, Yun J, Zhang G, Zabed HM, Qi X (2022) A review on constructed wetlands-based removal of pharmaceutical contaminants derived from non-point source pollution. Environ Technol Innov 26:102504. https://doi.org/10.1016/j.eti.2022.102504
42. Sezerino PH, Bento AP, Decezaro ST, Magri ME, Philippi LS (2015) Experiências brasileiras com *wetlands* construídos aplicados ao tratamento de águas residuárias: parâmetros de projeto para sistemas horizontais. Engenharia Sanitária e Ambiental 20:151–158
43. Su Y, Wang J, Huang Z, Xie B (2017) On-site removal of antibiotics and antibiotic resistance genes from leachate by aged refuse bioreactor: Effects of microbial community and operational parameters. Chemosphere 178:486–495. https://doi.org/10.1016/j.chemosphere.2017.03.063
44. Sun X, Sun Y, Zhao Y, Wang Y-N (2014) Leachate recirculation between alternating aged refuse bioreactors and its effect on refuse decomposition. Environ Technol 35(7):799–807. https://doi.org/10.1080/09593330.2013.852625
45. Sun X, Zhang H, Cheng Z, Wang S (2017) Effect of low aeration rate on simultaneous nitrification and denitrification in an intermittent aeration aged refuse bioreactor treating leachate. Waste Manage 63:410–416. https://doi.org/10.1016/j.wasman.2016.12.042
46. Sun Y, Sun X, Zhao Y (2011) Comparison of semi-aerobic and anaerobic degradation of refuse with recirculation after leachate treatment by aged refuse bioreactor. Waste Manage 31(6):1202–1209. https://doi.org/10.1016/j.wasman.2011.01.022
47. Tong H, Yin K, Giannis A, Ge L, Wang J-Y (2015) Influence of temperature on carbon and nitrogen dynamics during in situ aeration of aged waste in simulated landfill bioreactors. Biores Technol 192:149–156. https://doi.org/10.1016/j.biortech.2015.05.049
48. Vymazal J (2013) The use of hybrid constructed *wetlands* for wastewater treatment with special attention to nitrogen removal: a review of a recent development. Water Res 47:4795e4811. https://doi.org/10.1016/j.watres.2013.05.029
49. Wen P, Huang Y, Qiu Z, Li Q (2021) Microbial response during treatment of different types of landfill leachate in a semi-aerobic aged refuse biofilter. Chemosphere 262:127822. https://doi.org/10.1016/j.chemosphere.2020.127822
50. Xiang Y, Chen Y, Luo S, Zou J Zhang A (2020) Degradation of recalcitrant organic matter in SAARB leachate by a combined process of coagulation

and catalytic ozonation. Environ Sci Pollut Res 27(32):40219–40228. https://doi.org/10.1007/s11356-020-08292-4

51. Yang C, Fu T, Wang H, Chen R, Wang B, He T, Pi Y, Zhou J, Liang T, Chen M (2021) Removal of organic pollutants by effluent recirculation constructed wetlands system treating landfill leachate. Environ Technol Innov 24:101843. https://doi.org/10.1016/j.eti.2021.101843

52. Zhao Y, Lou Z, Guo Y, Xu D (2007) Treatment of sewage using an aged-refuse-based bioreactor. J Environ Manage 82(1):32–38. https://doi.org/10.1016/j.jenvman.2005.11.015

© The Author(s), under exclusive license to Springer Nature Switzerland AG 2024
S. Souabi, A. Anouzla (eds.), *Landfill Leachate Treatment Techniques*, Springer Water
https://doi.org/10.1007/978-3-031-63157-3_14

14. Assessment of the Effectiveness of Ouled Berjal Landfill Leachate Treatment by Sequential Batch Reactor Technique: Elimination of Nitrogen Pollution, Phenolic Compounds, and Phytotoxicity

Hajar Bakraouy[1], Khalid Digua[1], Moncif Sabar[2], Roukaya Bouyakhsass[1], Abdelaziz Madinzi[1], Tonni Agustiono Kurniawan[3], Abdelkader Anouzla[1] ✉, Younes Abrouki[4], Amane Jada[2] and Salah Souabi[1]

(1) Process and Environmental Engineering Laboratory, Faculty of Science and Technics, Hassan II University, Mohammedia, Morocco
(2) Institute of Materials Science of Mulhouse, Mulhouse, France
(3) College of the Environment and Ecology, Xiamen University, Xiamen, 361102, China
(4) Faculty of Science, Mohammed V University in Rabat, Rabat, Morocco

✉ **Abdelkader Anouzla**

Email: aanouzla@gmail.com

Abstract
Solid waste management is one of the significant issues faced by countries worldwide. The waste management approach used in Morocco consists mainly of improving waste collection, sorting, and treatment conditions while limiting the nuisance caused by methane emissions and leachate production. In the present work, we have treated leachate from the Ouled Berjal landfill (Kenitra -Morocco) using the Sequential Batch Reactor (SBR) technique. Thus, the leachate was put into two reactors, which differed in the times allocated to each phase. We have assessed the treatment effectiveness by monitoring the removal of NH_4^+ ions, nitrates, nitrites, phenol, Absorbance at 254 nm, and phytotoxicity. The results showed that the first SBR is suited for leachate treatment with removal efficiencies reaching 98.5% for NH_4^+ ions, 53% for nitrates, 61.5% for nitrites, 82.5% for phenol, and 66% for Absorbance. In addition, the phytotoxicity tests showed an increase in the germination index from 19 to 47%, indicating that a significant part of the pollutants was removed from the leachate.

Keywords SBR – Landfill leachate – Phytotoxicity – Phenol – Nitrogen pollution

---

14.1 Introduction

Solid waste management remains a big challenge for countries around the world. In Morocco, the improvement in the standard of living and the change in production and consumption patterns led to an increase in the volume of solid waste, estimated at 6.9 million tones/year in 2015 (Report of the Ministry of the Environment 2015). The

quantity produced is 0.76 kg/day per habitant in urban areas and 0.3 kg/day per habitant in rural areas. Such an increase affects both the quality of the collection and the quality of treatment in the landfill. Thus, Landfills become sources of nuisance instead of being the solution to eliminate waste [5, 30]. They are sources of methane and leachate. Leachate results from the solubilization of compounds during the percolation of rainwater through the pile of waste and moisture contained in the waste [19].

The challenge is now significant because it will be necessary to ensure adequate drainage, recovery, and treatment of leachates for reuse or rejection in nature. This approach will provide an important water resource, particularly in areas experiencing water shortages and drought.

Due to their toxicity, we must treat leachates before releasing them into the natural environment [1].

Several studies on the characterization of landfill leachate revealed that it is a complex medium containing more than 200 families of organic compounds.

According to [17], the pollutants contained in leachate can be classified in four categories:

- Dissolved or suspended organic matter such as chemical oxygen demand, volatile fatty acids, humic and fulvic compounds ...
- Mineral macros include Sodium, Calcium, Magnesium, Iron, and Ammonium ions ...
- Heavy metals such as Zinc, Cadmium, Nickel, Chrome ...
- Xenobiotic organic compounds such as aromatic hydrocarbons, phenols, chlorinated aliphatic hydrocarbons, pesticides, and plasticizers.

There are several leachate treatment techniques: Leachate transfer, biological, physicochemical, and membrane processes [23].

The choice of a treatment technique despite another is strongly linked to the composition of the leachate, its age, and the purification efficiency required by the standards applied in a country.

Biological treatment is increasingly applied to wastewater, particularly leachate discharges etc., which produce high concentrations of more or less biodegradable organic matter. Biological wastewater treatment methods are more cost-effective and environmentally friendly than physicochemical techniques.

They rely on the use of microorganisms to degrade organic pollutants in effluents, using technologies such as the batch sequential reactor (SBR), the moving bed biofilm reactor (MBBR), and the anaerobic up-flow sludge bed (UASB) [10]. SBR consists of a discontinuous mixed biomass suspension culture process in which the biological reaction phases and settling processes take place in the same basin [6].

The SBR process is used mainly to achieve the nitrification–denitrification of the effluent, which leads to the elimination of ammonium ions and their conversion into nitrites and nitrates, and finally to nitrogen gas [12].

The SBR treatment has several advantages compared to conventional activated sludge treatment. Indeed, in the SBR process, there are different growth kinetics due to the presence of varying microorganism species. Such different growth kinetics overcome the problem of the appearance of filamentous bacteria [13]. In addition, the SBR improves the reaction kinetics by ensuring substrate concentrations and maximum reaction rates at the end of the filling phase.

The SBR operating cycle consists of five steps: Feeding, reaction, settling, drawing, and idle. The succession of the reaction steps (characterized by alternating Aeration and anoxic phases) and decantation lead to a reduction in nitrogen, carbon, and phosphorus pollution [21].

The SBR (Sequential Batch Reactor) reactor, characterized by integrating biological reaction phases and sedimentation processes within a single basin, involves waste treatment in the presence of an activated bacterial population [6].

SBR systems offer several advantages, including their straightforward construction, ability to accommodate nitrification and denitrification processes, and versatility in adapting to a wide range of pollutant loads [14]. The primary goal of biological treatment using the SBR (Sequential Batch Reactor) method is to achieve nitrification–denitrification of effluents. This involves the removal of ammonium ions from the effluent and their conversion into nitrites, which are further converted into nitrates and ultimately transformed into nitrogen [12].

The sequencing batch reactor (SBR) is one of the aerobic biological methods based on bioreactors used to treat various domestic and industrial wastewater. SBR technology has been the subject of much research in recent years to treat domestic and industrial wastewater.

[18] conducted the co-treatment of landfill leachate and municipal wastewater in an SBR. Elimination efficiencies were 93% for NH_3, 83% for nitrates, 70% for COD, 80% for phosphorus, and 83% for turbidity.

Sun et al. [25] studied leachate treatment from the Peking landfill in China using a UASB system combined with an SBR. Total nitrogen removal reached 95.4% at a temperature of 14.9° C.

The present work is devoted to the biological treatment of the leachate from the Ouled Berjal landfill through two SBRs processes that differ in the times allocated to each step. The effectiveness of the treatment was evaluated by monitoring the removal of ammonium ions, nitrates, nitrites, phenol, Absorbance at 254 nm (Abs 254 nm), and phytotoxicity.

14.2 Experimental Details

14.2.1 Sanitary Landfill Leachate Sample

We collected the raw leachate sample from the sanitary landfill of Ouled Berjal, located in Kenitra, Morocco. The rapid development of Kenitra city is followed by increased solid waste production, reaching about 120,000 tons in 2011 and 130,296 tons in 2015.

In the Ouled Berjal landfill, household and similar waste and green waste come from Kenitra city. This waste is composed of 70% of organic matter.

This site covers a surface of about 20 ha, made of:

- Landfill area consisting of a 4 ha bin, which is subdivided into 4 cells;
- Leachate and stormwater drainage pipe;
- Three large basins to collect leachate and a reserve basin.

Figure 14.1 illustrates the Ouled Berjal landfill.

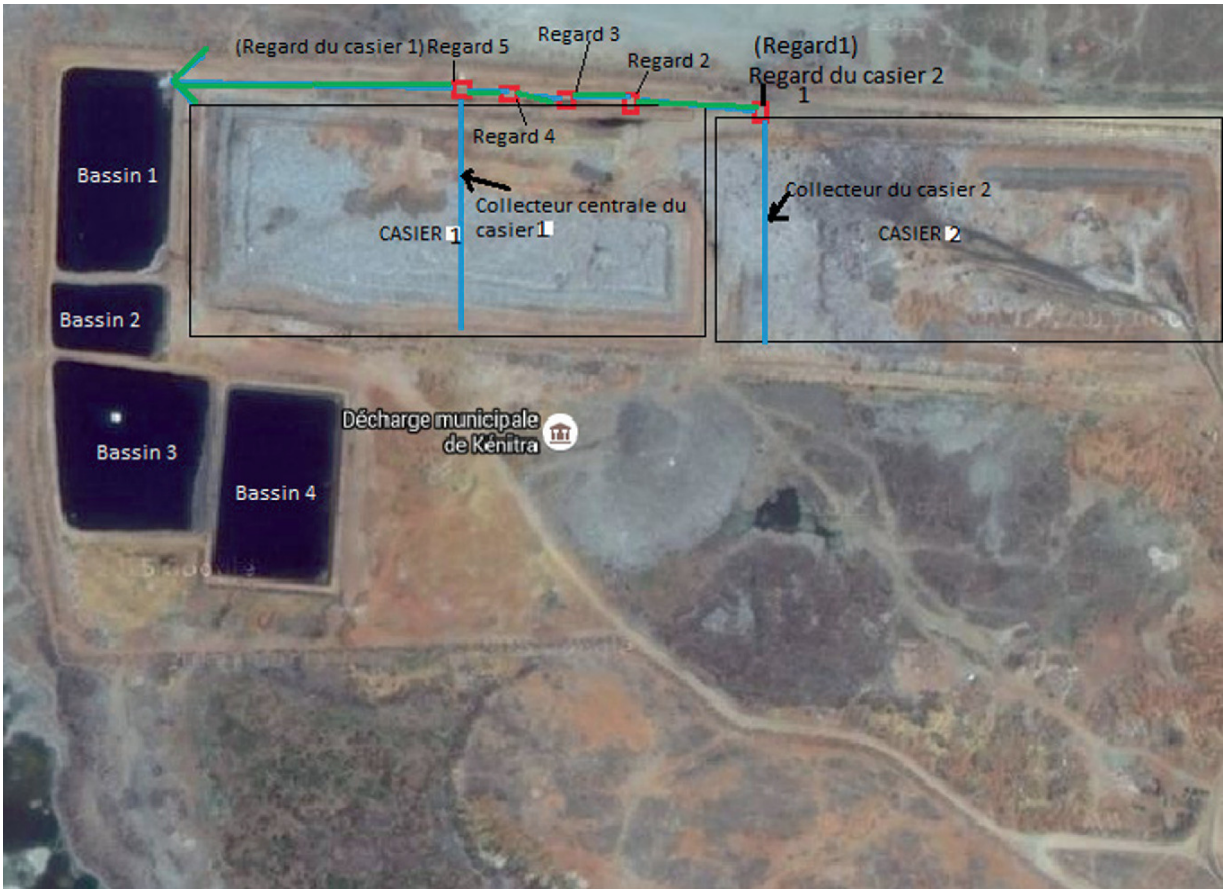


Fig. 14.1 Ouled Berjal landfill

Currently, the Ouled Berjal landfill faces several difficulties. Indeed, there is a great risk of saturation of the landfill. In addition, leachates are stored in basins and are not treated. This can lead to overflowing leachates at any time, altering the groundwater and surrounding agricultural land.

Landfill leachate samples were collected in 30-L plastic bottles, transported to the laboratory, and analyzed before starting the SBR treatment.

Leachate was analyzed for phenol, turbidity, suspended matter, COD, BOD_5, color, and Abs 254 nm by the standard APHA (APHA, 1998) and for toxicity by the Wang method [28].

We have conducted germination tests on 15 turnip seeds on Petri dishes with filter paper soaked in raw or treated leachate. We have performed three replicates for each

effluent [28]. We conducted the tests in the dark and at room temperature (25 °C) for 120 h. Further, we carried out the phytotoxicity test by monitoring the emergence of seeds and the growth of radicles.

The germination index (GI) is determined concerning emergence and root growth in leachate and distilled water according to the following equation:

$$\mathrm{GI}\,(\%) = \frac{\mathrm{NG_{Lix}} * \mathrm{LR_{Lix}}}{\mathrm{NG_{ED}} * \mathrm{LR_{ED}}} * 100$$

With:

NG_{Lix}, NG_{ED}: Number of germinated seeds in leachate and distilled water.

LR_{Lix}, LR_{ED}: Length of radicles in leachate and distilled water.

14.2.2 Experimental Procedure

In this study, leachate from basin B_1 was treated biologically in an SBR for 30 days. To do this, we have implemented two SBR reactors that differ in the duration allocated to the operating cycle phases. The succession of phases and their durations are as follows (Fig. 14.2).

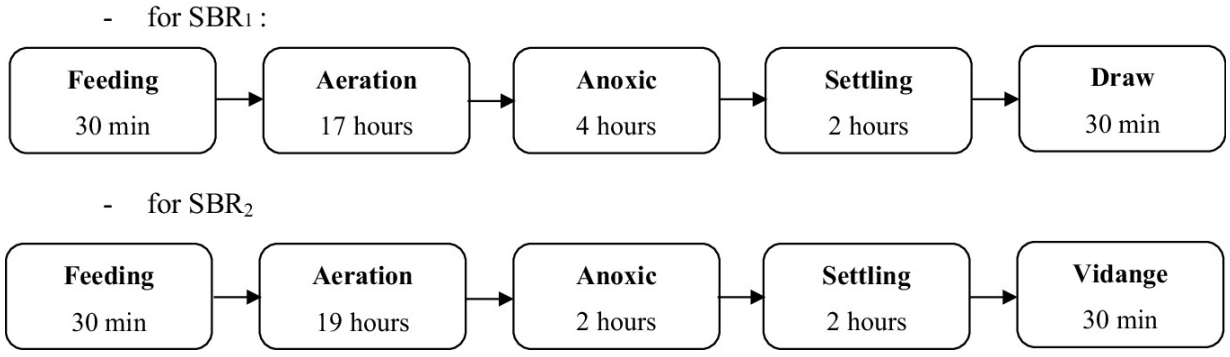


Fig. 14.2 Cycles adopted for the two SBRs during the experiment

Several parameters condition the excellent functioning of the SBR, namely the liquid's residence time, the sludge's

age, the effluent's pollution load, the dissolved oxygen content, the temperature, and the agitation speed [24].

14.2.2.1 Sludge Age

We chose an infinite sludge age, i.e. the inoculum and the sludge formed during the biodegradation process were only removed at the end of the experimentation period (30 days). The inoculum used for this study consists of sludges from the STEP of Lessieur Crystal Company (a vegetable oil refinery industry). During the experimental period, the MLSS (Mixed Solids Suspended Solids) of the two SBRs reached 13,000 and 9000 mg/L for SBR1 and SBR2, respectively.

14.2.2.2 Dissolved Oxygen Content and Carbon Source

Aeration was provided by aquarium aerators linked to timers to adjust the duration and sequence of different phases.

The dissolved oxygen content was maintained at values around 2 mg/L. No carbon sources were added to the leachate during the experimental period.

14.2.2.3 Start-Up and Stabilization of SBR_1 and SBR_2

Both reactors were fed with 3 L of water to be treated. They are kept under magnetic stirring during all phases except for settling and drawing.

These two reactors were subjected to an acclimation phase that lasted 7 days; they were fed with a mixture of 50% leachate and 50% tap water. This phase aims to adapt the biomass of the inoculum to the new substrate, which is, in our case, landfill leachate.

From the eighth day, both reactors were fed with raw leachate.

14.2.2.4 Evaluation of Treatment Efficiency

A peristaltic pump, «Perimatic GP» of Jencons brand, carried out the draw (Fig. 14.3).



Fig. 14.3 Draw the SBR through the peristaltic pump

Treated leachate was analyzed daily using the following parameters: COD, BOD_5, NH_4^+, NO_3^-, NO_2^-, turbidity, phenol, surfactant, total phosphorus, color, and Abs 254 nm.

In addition, we monitored the evolution of the phytotoxicity of treated leachates compared to raw leachate using the turnip grain germination method.

14.3 Results and Discussions

14.3.1 Characteristics of the Landfill Leachate

The physicochemical analysis of the leachate is essential before starting the treatment. Indeed, the choice of the treatment technique is strongly related to the composition. On the other hand, this analysis also makes it possible to

determine the impact of leachate on groundwater (in case of infiltration), the environment adjacent to the landfill, and human health.

Leachate composition depends on several parameters, including the landfill's age, the type of waste, and the landfill technique [3].

The physicochemical characteristics of the leachates used for this study are summarized in Table 14.1.

Table 14.1 The mean and extent of the different physicochemical parameters of the studied leachate

Parameter	Mean	Max	Min
Turbidity (NTU)	110.8	149	75.2
Phenol (mg/L)	158.6	409.6	69.8
NO_3^- (mg/L)	356.0	416.0	248.0
NH_4^+ (mg/L)	1095.3	1231.5	523.5
NO_2^- (mg/L)	716.7	1040.0	460.0
Total phosphorus (mg/L)	26.2	33.2	16.9
Surfactant (mg/L)	55.0	67.7	16.5
COD (mg O_2/L)	7840.0	11.520	3840
Color ($F_D = 200$*)	0.039	0.057	0.025
Abs 254 nm ($F_D = 200$*)	0.192	0.380	0.073

* $F_D = Dilution\ factor$

We note that the leachate from B_1 basin is loaded with organic matter, as evidenced by the high COD value of 11.5 g/L. This remains below the values reported by [4, 7], which are 54 and 72 g/L, respectively.

This leachate can be classified as young since the value of the COD exceeds 10 g/L [19].

It contains 1.1 g/L of NH_4^+, close to the value reported by [32], for leachate from Beijing.

The mean nitrate concentration reached 356 mg/L. In a research conducted by [26] on the Meknes City landfill, the leachate characterization revealed a nitrate content of 751.1 mg L, higher than the present study. In contrast, the concentration of nitrates is lower in the leachates of El Jadida city landfill: 2.4 mg/L [3], and Fez city landfill: 11.3 mg/L [4], as well as the city of Oran: 0.92 mg/L [2].

The mean concentration of total phosphorus is 26.2 mg/L, which exceeds the values reported by [22], with a value of 19.1 mg/L, and by [32], with a value of 5.2 mg/L, but remains lower than that of the Agadir city landfill, which reached 188.4 mg/L [7].

14.3.2 Elimination of Ammoniacal Nitrogen

In addition to refractory organic matter, landfill leachate contains high ammonium levels. According to [8], 50% of the nitrogen is solubilized following the anaerobic digestion of municipal solid waste. The fermentation of the amino acid leads to the formation of organic acids and ammonia. Thus, due to the hydrolysis and fermentation of these nitrogen fractions, leachate from old landfills is heavily loaded with ammonia nitrogen [9].

One of the major goals of SBR treatment is eliminating nitrogen pollution. In an SBR, the nitrification–denitrification process takes place, initially transforming ammoniacal nitrogen into nitrates and nitrites (nitrification) and gaseous nitrogen (denitrification).

We consider NH_4^+ removal as an indicator of the completion of the acclimation phase.

Indeed, Fig. 14.4 shows that during the first days, the elimination of NH_4^+ of treated leachates is low.

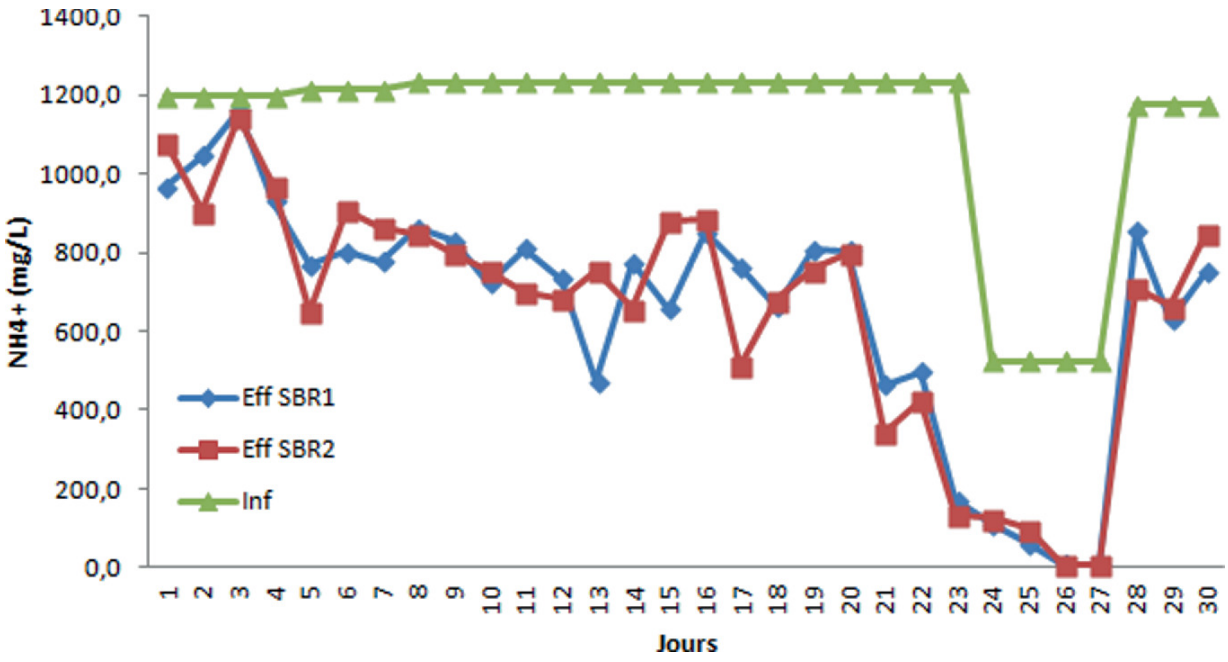


Fig. 14.4 Variation of NH_4^+ concentration during SBR treatment

This is probably due to the non-adaptation of the bacteria. Indeed, from the 5th day, we note that the removal of NH_4^+ stabilized at 35% for SBR_1 and 30% for SBR_2, which means that the acclimation phase took an end. Therefore, we can feed the two reactors with raw leachate without dilution and from the 8th day.

Moreover, NH_4^+ concentration at the outlet of the two SBRs continues to decrease progressively to reach minimum concentrations of 7.9 and 7.5 mg/L for SBR_1 and $SBR_{2,}$ respectively, achieving elimination efficiencies exceeding 98%.

These efficiencies are higher than [15]. Indeed, these authors' study focused on treating the stabilized leachate by ultrasound coupled to SBR. The cycle time of the SBR is 3 h, divided as follows: Feeding (0.15 h), reaction (3 h), decantation (0.3 h), and draw (0.08 h). As a result, 90% of NH_4^+ was eliminated.

14.3.3 Elimination of Nitrites and Nitrates

The formation of nitrites and nitrates, commonly known as nitrification, is obtained aerobically via the following reactions:

$$\mathrm{NH_4^+} + \frac{3}{2}\mathrm{O_2} \rightarrow \mathrm{NO_2^-} + \mathrm{H_2O} + 2\mathrm{H^+} \ \text{(Nitritation)}$$

$$\mathrm{NO_2^-} + \frac{1}{2}\mathrm{O_2} \rightarrow \mathrm{NO_3^-} \ \text{(Nitratation)}$$

Nitritation requires reducing the activity of oxidative nitrite bacteria (ONB), without affecting oxidative ammonia microorganisms (OAB). This can be achieved by limiting the dissolved oxygen supply and regulating the pH [9].

In addition, eliminating these elements (nitrates and nitrites) is called denitrification, carried out under anaerobic conditions. The following reaction governs it:

$$\mathrm{NO_3^-} \rightarrow \ \mathrm{NO_2^-} \rightarrow \ \mathrm{NO} \rightarrow \mathrm{N_2O} \rightarrow \mathrm{N_2} \ \text{(Denitrification)}$$

Figure 14.5 shows that both SBRs can remove up to 30% of the nitrites. However, we notice that the concentration of nitrites at the exit of SBR is higher than that at the inlet, indicating a significant elimination of NH_4^+. As a result, there is a formation of nitrites, which are less toxic compared to NH_4^+.

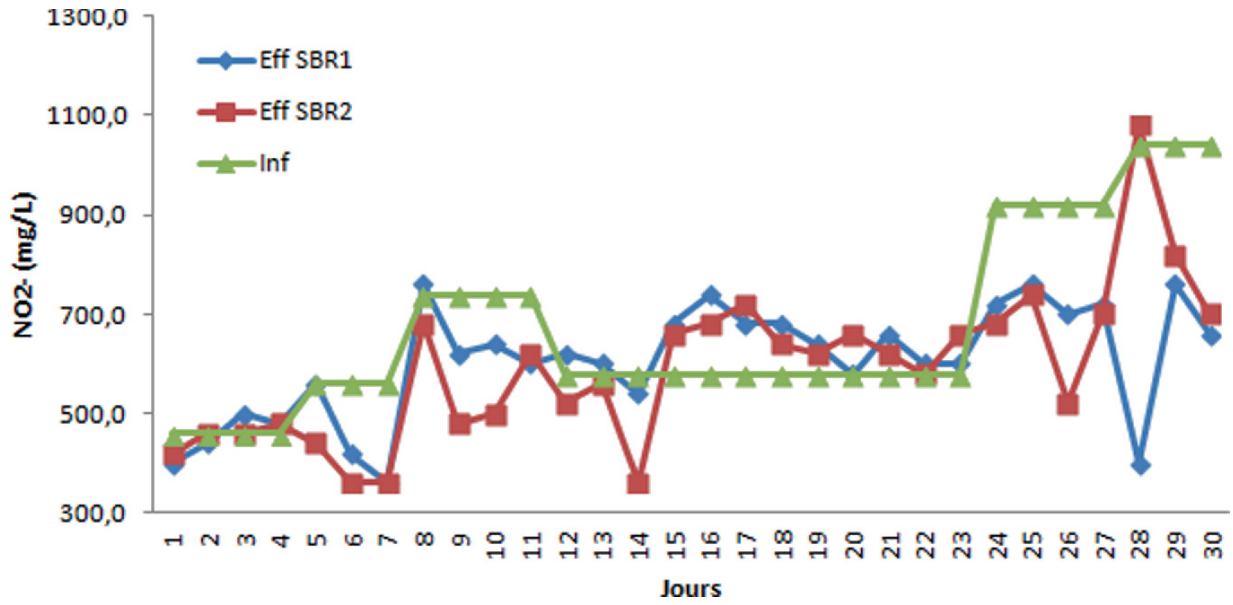


Fig. 14.5 Variation of NO_2^- concentration during SBR treatment

As a result, we were unable to achieve complete denitrification. This is mainly due to the lack of organic carbon sources since our experiments were conducted without using an external carbon source. Indeed, Tora et al. [27] indicated that the COD/N ratio required for eliminating nitrites depends on the type of organic carbon source (ethanol, glycerin, etc.). This ratio must equal 3 if ethanol is added as a carbon source. In addition, the kinetics of conversion of NH_4^+ to NO_3^- and NO_2^- is often slow.

The concentration of nitrates (Fig. 14.6) marks a trend that fluctuates a lot. Generally, it is higher at the entrance than at the exit of the two SBRs, except from the 12th to the 25th day. Which shows that denitrification is implemented.

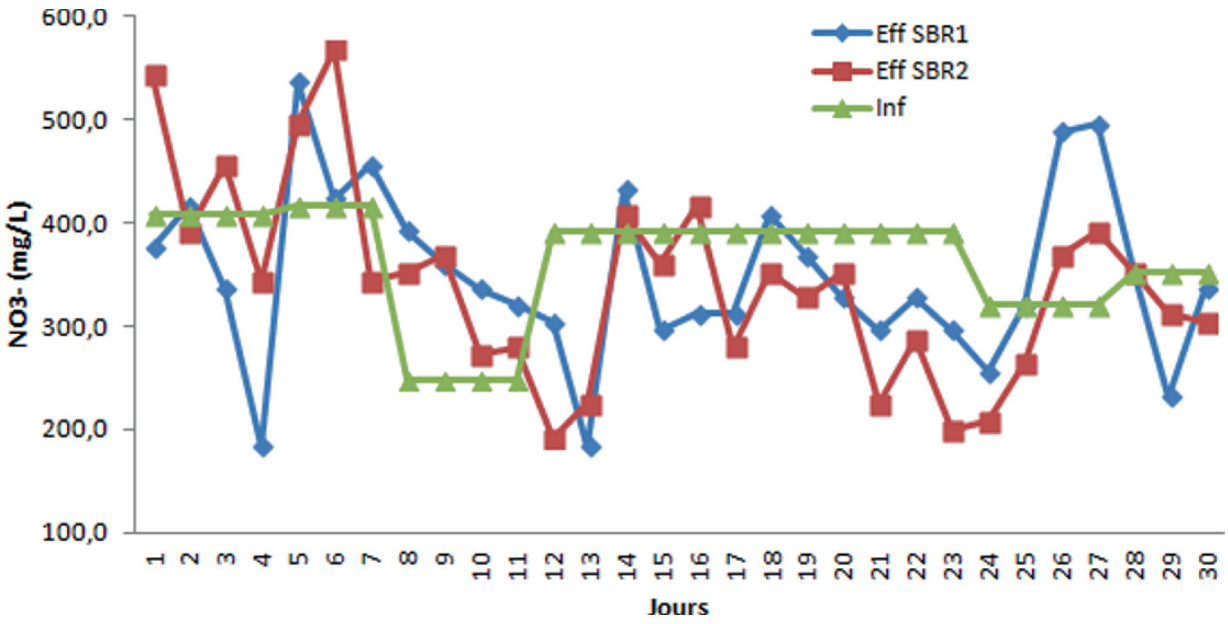


Fig. 14.6 Variation of NO_3^- concentration during SBR treatment

Nitrate removal efficiencies reached 53.1% for SBR_1 and 51% for SBR_2.

[9] investigated the effect of limited oxygen alimentation during the aeration phase of the SBR cycle on the denitrification efficiency of landfill leachate. The mixture of glycerin with Sodium acetate or Sodium acetate was used as an external carbon source. These authors showed that 99% of the ammonium ions and 61% of the nitrites were eliminated.

14.3.4 Elimination of Phenol and Abs 254nm

Figures 14.7 and 14.8 show that phenol and Abs 254 nm elimination efficiency variations follow the same trend.

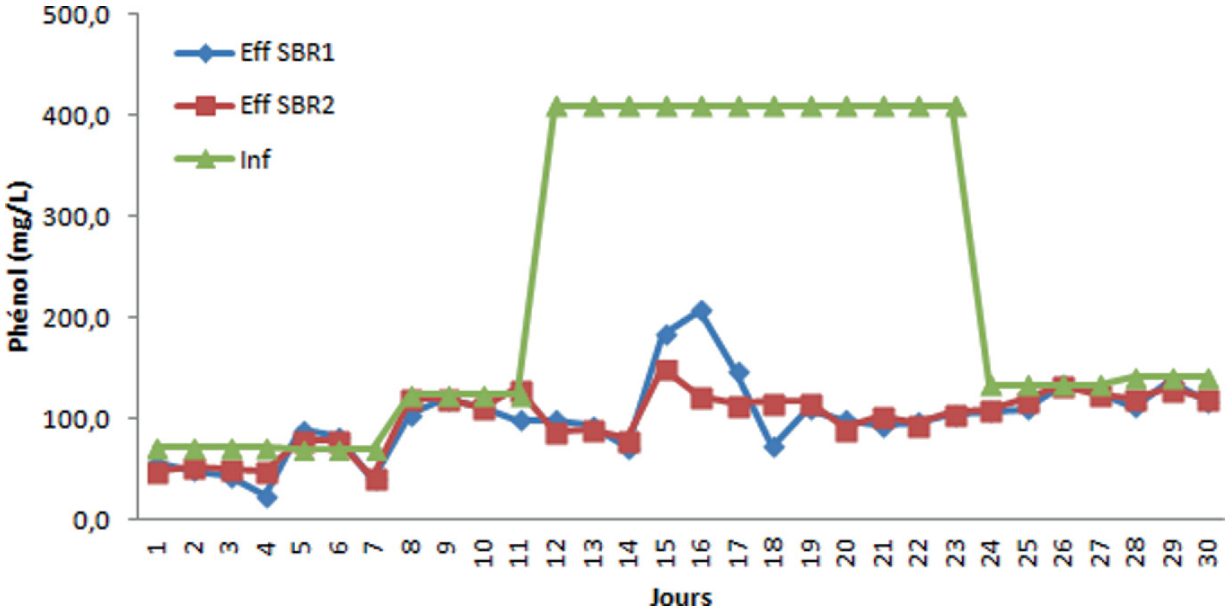


Fig. 14.7 Variation of phenol concentration during SBR treatment

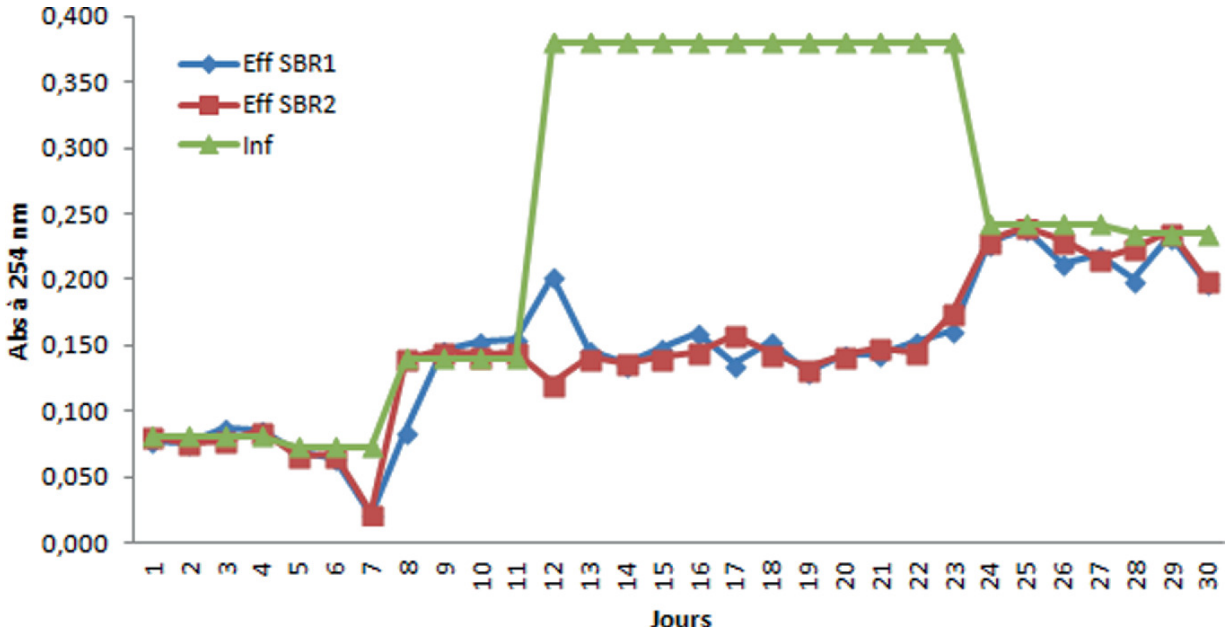


Fig. 14.8 Variation of Abs 254 nm concentration during SBR treatment

We note that the removal of phenol and Abs at 254 nm is low during the first 11 days of testing. This may be due to the formation of complex molecules from other simpler ones or to the biodegradation of non-humic compounds followed by the formation of poly-compact humic structures.

As a result, removals increase (the two curves follow the same trend) to reach:

- 82.5 and 80.5% for removal of phenol for, respectively, for SBR_1 and SBR_2;
- 65.8 and 68.2% for removal of Abs 254 nm, respectively, for SBR_1 and SBR_2.

[31] treated synthetic industrial wastewater by an SBR operating for 24 h cycle, distributed as follows: 1 h of feeding, 20 h of Aeration, 2 h of settling, and 1 h of drawing. As a result, the phenol content was reduced by 80%.

14.3.5 Reduction of Phytotoxicity

Landfill leachate is a real danger to people and the environment. This is due to the high concentrations of recalcitrant pollutants, whose toxicity is very worrying. Therefore, reducing leachate toxicity is essential in treating this effluent [22].

Phytotoxicity tests make it possible to highlight the impact of leachate on the soil, in particular on the germination of grains.

Phytotoxicity is often assessed by the study of germination or by growth tests [29]. Several authors used phytotoxicity tests to evaluate the toxicity of raw and treated leachates and, thus, the effectiveness of the treatment. Treated leachate must not contain substances that prevent seed germination and plant growth. Several seeds can be chosen for this test given their sensitivities (Turnip, Watercress, Alfalfa, Tomato, Lettuce ... etc.).

In this study, we opted for the germination method of turnip seeds, a susceptible species. Thus, phytotoxicity is evaluated by calculating the germination index and the length of the radicles.

We compared raw leachates with treated leachates during the 12th and 21st days to evaluate the efficacy of SBR_1 and SBR_2 treatment.

Referring to Fig. 14.9, we note that germination indices and root lengths for both SBRs are higher than raw leachate. This shows that the toxicity of treated leachates is lower than that of raw leachate due to eliminating toxic elements.

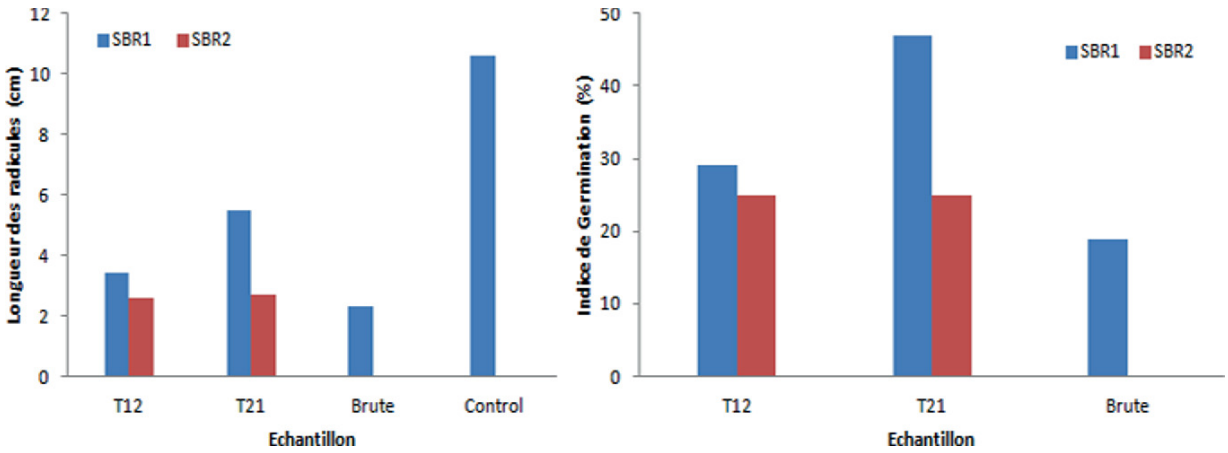


Fig. 14.9 Evaluation of phytotoxicity parameters for leachates treated with SBR

The germination index is 19% for raw leachate. For SBR_1, this index increases from 29% (for the 12th day) to 47% (for the 21st day).

Comparing the leachates treated during the 12th and the 21st day, we note an increase in the length of the radicles of the order of 2.1 cm.

For SBR_2, the germination index reached 25%, higher than the raw. However, no improvement was observed after comparing the two leachates treated during the 12th and 21st days.

Regarding phytotoxicity, SBR1 is better suited for eliminating toxic compounds in leachate from B_1 basin.

Olivero-Verbel [16] evaluated the phytotoxicity of leachate using the aquatic species *Artemia franciscana*.

The study aims to determine the relationship between the composition and toxicity of leachate. The authors showed that the toxicity decreases with the increase of ionic substances. In contrast, organic matter (COD) and heavy metals such as cadmium act as a promoter of toxicity. The authors conclude that toxicity may depend on forming organic complexes rather than free ionic compounds.

Li [11] studied the evolution of the toxicity of stabilized leachate before and after coagulation-flocculation treatment. They chose the wheat seed growth inhibition test because wheat is the main grain consumed worldwide. At the end of this study, samples treated with aluminum-based coagulants showed the highest toxicity. This has been explained by residual aluminum (in the form of Al^{3+} ions) in the treated leachate, which is toxic to plants. The authors add that the toxicity is strongly related to the organic matter contained in the leachate since the toxicity increases at high concentrations of COD.

Conclusion

This study showed the effectiveness of the Sequential Batch Reactor (SBR) technique in treating leachate from the Ouled Berjal landfill. Thus, we have compared two SBRs, which differ by the time allocated to the different phases of the operating cycle. We have obtained the best results with SBR having the following cycle phases: Feeding (30 min), Aeration (17 h), Anoxic (4 h), Settling (2 h), and draw (30 min).

The observed removal efficiencies were 98.5% of NH_4^+, 53% of nitrates, 61.5% of nitrites, 82.5% of phenol, and 66% of the Absorbance at 254 nm. We also examined the efficiency of SBR treatment on the phytotoxicity reduction. Hence, we conducted phytotoxicity tests that monitored germination indices and root lengths of turnip seeds.

According to the data, SBR_1 is more efficient in phytotoxicity.

Acknowledgements

The authors thank the Moroccan Ministry of Environment, which funded this study.

References

1. Bashir MJK, Xian TM, Shehzad A, Sethupathi S, Ng CA, Abu Amr S (2017) Sequential treatment for landfill leachate by applying coagulation-adsorption process. Geosyst Eng 20:9–20
2. Bennama T, Younsi A, Derriche Z, Debab A (2010) Caractérisation et traitement physico-chimique des lixiviats de la décharge publique d'El-Kerma (Algérie) par adsorption en discontinu sur de la sciure de bois naturelle et activée chimiquement. Water Qual Res J Can 45(1):81–90
3. Chofqi A, Younsi A, Lhadi EK, Mania J, Mudry J, Veron A (2007) Lixiviat de la décharge publique d'El Jadida (Maroc): caractérisation et étude d'impact sur la nappe phréatique. Déchets Sciences et Techniques 46:4–10
4. Ezzoubi Y, Merzouki M, Bennani L, El Ouali Lalami A, Benlemlih M (2010) Procédé pour la réduction de la charge polluante du lixiviat de la décharge contrôlée de la ville de Fès. Déchets, sciences et techniques—Revue francophone d'écologie industrielle 58:22–29
5. Gupta A, Paulraj R (2017) Leachate composition and toxicity assessment: an integrated approach correlating physicochemical parameters and toxicity of leachatesfrom MSW landfill in Delhi. Environ Technol 38(13–14):1599–1605
6. Ismail NF, Jami MS, Amosa MK, Muyibi S (2014) Modelling and simulation of energy-saving potential of sequential batch reactor (SBR) in the abatement of ammoniacal-nitrogen and organics. J Pure Appl Microbiol 8:809–814
7. Jirou Y, Harrouni MC, Belattar M, Fatmi M, Daoud S (2014) Traitement des lixiviats de la décharge contrôlée du Grand Agadir par aération intensive. Rev Mar Sci Agron Vét 2(2):59–69
8. Jokela JPY, Rintala JA (2003) Anaerobic solubilisation of nitrogen from municipal solid waste (MSW). Rev Environ Sci Biotechnol 2:67–77

9. Kulikowska D, Bernat K (2013) Nitritation–denitritation in landfill leachate with glycerine as a carbon Source. Biores Technol 142:297–303
10. Li J et al (2019) Adaptation of semi-continuous anaerobic sludge digestion to humic acids. Water Res 161:329–334
11. Li W, Hua T, Zhou QX, Zhang SG, Li FX (2010) Treatment of stabilized landfill leachate by the combined process of coagulation/flocculation and powder activated carbon adsorption. Desalination 264:56–62
12. Liu Z, Wu WH, Shi P, Guo J, Cheng J (2015) Characterization of dissolved organic matter in landfill leachate during the combined treatment process of air stripping, Fenton, SBR and coagulation. Waste Manage 4:111–118
13. McQuarrie JP, Boltz JP (2011) Moving bed biofilm reactor technology: process applications, design, and performance. Water Environ Res 83:560–575
14. Mojiri A, Aziz HA, Aziz SQ, Zaman NQ (2012) Review on municipal landfill leachate and sequencing batch reactor (SBR) technique. Arch Des Sci 65:22–31
15. Neczaj E, Okoniewska E, Kacprzak M (2005) Treatment of landfill leachate by sequencing batch reactor. Desalination 185:357–362
16. Olivero-Verbel J, Padilla-Bottet C, De la Rosa O (2008) Relationships between physicochemical parameters and the toxicity of leachates from a municipal solid waste landfill. Ecotoxicol Environ Saf 70:294–299
17. Peng Y (2013) Perspectives on technology for landfill leachate treatment. Arab J Chem 10:S2567–S2574
18. Ranjan K, Chakraborty S, Verma M, Iqbal J, Naresh Kumar R (2016) Co-treatment of old landfill leachate and municipal wastewater in sequencing batch reactor (SBR): effect of landfill leachate concentration. Water Qual Res J Can 51(4):377–387
19. Renou S, Givaudan JG, Poulain S, Dirassouyan F, Moulin P (2008) Landfill leachate treatment: review and opportunity. J Hazard Mater 150:468–493
20. Report on the State of the Environment of Morocco (2015) Ministry delegate to the minister of energy of mines, water and environment, in charge of the environment
21. Sathian S, Rajasimman M, Radha G, Shanmugapriya V, Karthikeyan C (2014) Performance of SBR for the treatment of textile dye wastewater: optimization and kinetic studies. Alex Eng J 53(2):417–426

22. Silva AC, Dezotti M, Jr. Sant'Anna GL (2004) Treatment and detoxification of a sanitary landfill leachate. Chemosphere 55:207–214
23. Souabi S, Touzare K, Digua K, Chtioui H, Khalil F, Tahiri M (2011) Sorting and valuation solid waste in the garbage dump of the city of Mohammedia. Thesis in Chemistry, vol 6. Faculty of Science and Technical Chemistry Department Fes, No 25
24. Sri Shalini S, Joseph K (2012) Nitrogen management in landfill leachate, application of SHARON, ANAMMOX and combined SHARON–ANAMMOX process. Waste Manage 32:2385–2400
25. Sun H, Yang Q, Peng Y, Shi X, Wang S, Zhang S (2010) Advanced landfill leachate treatment using a two-stage UASB-SBR system at low temperature. J Environ Sci 22(4):481–485
26. Tahiri AA, Laziri F, Yachaoui Y, El Jaafari S, Tahiri AH (2014) Etude des polluants contenus dans les lixiviats issus de la décharge publique de la ville de Meknès (MAROC). Eur Sci J 10(35):170–186
27. Tora JA, Baeza JA, Carrera J, Oleszkiewicz JA (2011) Denitritation of a high strength nitrite wastewater in a sequencing batch reactor using different organic carbon sources. Chem Eng J 172:994–998
28. Wang W (1989) Toxicity assessment of the aquatic environment using phytoassay methods. By Water Quality Section Illinois State Water Survey
29. Wang X, Sun C, Gao S, Wang L, Shuokui H (2001) Validation of germination rate and root elongation as indicator to assess phytotoxicity with *Cucumis sativus*. Chemosphere 44(8):1711–1721
30. Yong ZJ, Bashir MJK, Ng CA, Sethupathi S, Lim J-W (2018) A sequential treatment of intermediate tropical landfill leachate using a sequencing batch reactor (SBR) and coagulation. J Environ Manage 205:244–252
31. Yusoff N, Ong S, Ho L-N, Wong Y-S, Mohd Saad FN, Khalik W, Lee S-L (2016) Evaluation of biodegradation process: comparative study between suspended and hybrid microorganism growth system in sequencing batch reactor (SBR) for removal of phenol. Biochem Eng J 115(2016):14–22
32. Zhu R, Wang S, Li J, Wang K, Miao L, Ma B, Peng Y (2013) Biological nitrogen removal from landfill leachate using anaerobic–aerobic process: denitritation via organics in raw leachate and intracellular storage polymers of microorganisms. Biores Technol 128:401–408

© The Author(s), under exclusive license to Springer Nature Switzerland AG 2024
S. Souabi, A. Anouzla (eds.), *Landfill Leachate Treatment Techniques*, Springer Water
https://doi.org/10.1007/978-3-031-63157-3_15

15. Filtration Technique for Leachate Using Recycled Coarse Aggregate

J. Logeshwari[1] ✉ and T. Anstey Vathani[1] ✉
(1) Vel Tech Rangarajan Dr. Sagunthala R&D Institute of Science and Technology, Avadi, Chennai, 6000062, India

✉ **J. Logeshwari (Corresponding author)**
Email: drjlogeshwari@veltech.edu.in

✉ **T. Anstey Vathani**
Email: vtd985@veltech.edu.in

Abstract
Landfill is a place where wastes such as degradable and non-degradable, hazardous and non-hazardous wastes are disposed. One of the important factors that should be considered in a landfill is leachate segregation and the covers that are used to protect the ground from contamination. Recycled Coarse Aggregate (RCA) which is a byproduct of the construction and demolition waste (C&D) was studied to generate an adsorbent characteristic with the help of adsorbents with different pilot studies. A pilot study was carried out with the RCA sample having a size of 6 mm, paper pulp, adsorbents and the unfiltered

leachate. The performance of RCA was observed for a period of over 2 weeks. The research work shows the performance of various adsorbents at different proportions for turbidity and color and its capacity to change from acidic nature to basic. The study mainly focusses on treating the leachate with RCA and adsorbent and focus on using it as a liner in landfill. Further studies are to be conducted to determine the changes in other parameters of the leachate that is collected.

Keywords Activated carbon – Zeolite – Sodium chloride – Adsorbents – Recycled coarse aggregate (RCA)

15.1 Introduction

The waste is disposed of and buried in a landfill by first being dumped in a designated location and then having the area covered with soil. It is now widely utilized as a standard approach to waste management in a great number of municipalities located all around the world. Landfills are designed to limit the amount of pollution that is caused to the environment surrounding them by concentrating and isolating rubbish in one location. A number of measures are often required to be followed in order to successfully manage a landfill. The location is analyzed, taken into account, and ultimately engineered so as to have the least amount of a deleterious impact on the ecosystem of the surrounding area. In order to achieve this goal, there are a number of factors that need to be taken into account, such as the quality of the soil, how close it is located to different water sources, and the potential consequences for the population in the area. After a landfill has been constructed, waste is taken there and dumped there; once it has been delivered there, it is placed in designated cells.

The garbage is scattered throughout a larger area and then packed into a smaller volume in order to make the most efficient use of the space that is available. Landfills make it possible to dispose of a diverse array of waste, such as municipal solid waste, building debris, and industrial rubbish; nevertheless, regulations and limits frequently stipulate what may and cannot be accepted at landfills. The majority of today's landfills employ a wide variety of engineering solutions to lessen the negative effects they have on the surrounding community. A substance known as leachate is generated when waste is broken down into smaller and smaller pieces. To prevent the leachate from leaking out of the landfill and into the groundwater and soil around it, these safety measures involve sealing the base and sidewalls of the landfill with materials such as clay or synthetic liners. The practice of installing landfill gas collection systems to manage the gases produced as a result of the decomposition of organic waste is widespread. This is done for the purpose of reducing greenhouse gas emissions. It is not inconceivable that landfill gas, which is predominately composed of the potent greenhouse gas methane and is produced as rubbish decomposes in landfills, might be produced over an extended period of time. In order to collect and control this gas, several landfills have constructed intricate networks of wells and pipes that extract it. This allows the gas to be managed. Burning the gathered gas or converting it into electricity can both be done to provide usable forms of energy. There are positive aspects to either of these options.

It's possible that landfills will result in a mixed bag of positive and negative impacts. To name only a few advantages, a landfill provides a centralized site for the disposal of trash, aids in the prevention of pollution and the spread of illness, and generates electricity through the utilization of landfill gas. Other problems associated with landfills include the consumption of land, the emission of

odors, the contamination of groundwater, and the potential for adverse effects on the ecosystem over the course of many years. In an effort to diminish humanity's dependency on landfills, numerous programs are being undertaken to lessen the amount of trash that is generated, to promote recycling and composting, and to investigate alternative ways of waste management. Some of these initiatives include the creation of technologies that can convert garbage into electricity as well as campaigns to reduce garbage production to the greatest extent that is practically possible.

Leachate, which is formed in landfills when rubbish decomposes, needs to be efficiently contained in order to safeguard the integrity of the ecosystem as well as human health. This abstract discusses the construction, the many different types of landfill liners, as well as their value to the ecosystem. To prevent leachate from seeping into the soil and groundwater below landfills, man-made barriers known as landfill liners are installed at the dump's base in strategic spots. The primary objective is to lessen the chances that hazardous materials will spread and that water will get contaminated. This abstract examines the many components of landfill liners, including clay liners, geosynthetic materials, and composite liners, all of which are designed to discourage the percolation of leachate and to promote its controlled collection.

Because of their superior resistance to chemical deterioration and punctures, high-density polyethylene (HDPE) or polypropylene-based geosynthetic liners are often used for geosynthetic lining applications. Clay liners, on the other hand, are effective at preventing the passage of leachate because of the inherent impermeability of clay minerals. Composite liners blend clay and geosynthetic layers in order to boost their effectiveness and provide an additional layer of protection. It is essential to have a solid understanding of the environmental significance of landfill

liners in order to achieve effective waste management and long-term sustainability. The use of efficient lining techniques drastically reduces the likelihood of groundwater pollution, hence lessening the danger of damage to ecosystems and sources of drinking water. In addition, efficient liner design and installation contribute to compliance with regulatory and environmental protection criteria. In conclusion, it is vital to make use of the appropriate landfill liner systems in order to reduce the damage that leachate does to the ecosystem. This abstract focuses on the importance of landfill liners in waste management processes and describes the many different types of liners that can be used for effective containment. Also discussed is the variety of liners that may be used. It serves as a jumping off point for further research into this significant aspect of environmental protection and landfill management.

To guarantee that they meet the standards set by the federal government, landfills must be properly located, developed, managed, and inspected. In addition to this, they are made to shield the natural world from any harmful chemicals that may be carried by the waste stream. Due of the need to find landfills using on-site environmental monitoring technology, landfills cannot be constructed in areas that are considered to be environmentally sensitive. In addition to providing an additional layer of protection, these monitoring devices also look for any indications of contamination in the landfill gas and groundwater. It is difficult for plants to become established in the cover soil of a landfill because of unfavorable pedoclimatic circumstances, such as the presence of gas problems, shallow soil, a lack of nutrients, higher temperatures, and contaminated leachate. When it comes to newly reconstructed landfills, early successional plants need to be chosen and planted on a regular basis. The establishment of an ecosystem in a closed landfill frequently occurs

concurrently with both natural processes and human activity.

Solid waste landfills pollute groundwater. Landfill leachate is very contaminated. This is likely. Even small landfills may last decades or millennia. This shows that a trash dump without leachate barriers can pollute groundwater long after dumping stops. Leachate leaks from poorly designed, constructed, and maintained landfills with liners and leachate collection systems may also contaminate groundwater. These procedures may leak leachate from landfills. The Building Material Promotion Council reports 150 million tonnes of building and demolition detritus annually. The study's goals are to generate adsorbent properties for the RCA and use it as a landfill cover. After being treated to remove the deposits on the aggregate, the C&D wastes are crushed to the necessary size to create RCA. Activated carbon, sodium chloride, and zeolite are used as adsorbents in an experimental inquiry to test the RCA's adsorbing properties under various modifications. The 20-day trial came to the conclusion that the chemicals performed well on RCA, but that further research on color and turbidity is needed.

15.2 Background Study

Based on the background search and previous study the adsorbents are shortlisted for the pilot study. The background study is done based on the liners in landfill adsorbents that are suitable for RCA, properties of RCA and water absorption characteristics of RCA.

Ergene et al. [16] had investigated on Heterogeneous statistics were used to examine landfill leachate data from 220 landfills in 46 nations. Advanced statistical analysis like cluster and PCA could be performed on the raw data matrix after addressing anomalies, adding in missing data, and normalizing data. Leachate characteristics were

obtained using regression modeling. The initial components with the strongest correlations were dominated by inorganic parameters, which explained the majority of the data variation in the PCA. These closely related measures in landfill leachate may be crucial for analyzing their biodegradation and transit processes as well as for determining if they should be excluded from landfill monitoring sampling and analysis. High quantities of salts, inorganics, and organics in the leachate changed how PCA components developed. This provides important details regarding leachate samples and their landfills. Chen et al. [12] had studied on Landfills leachate per- and polyfluoroalkyl substances (PFAS) from consumer and commercial items, industrial wastes, and municipal wastewater treatment plant dewatered sludge. On-site leachate treatment lowers PFAS. PFAS concentration fluctuations have been assessed at few large landfills with leachate treatment systems. This study tracked PFAS trends in fully operational on-site garbage treatment plants. Leachate from 15 sites before and after treatment included 26 PFAS, including 11 PFCAs, 7 PFSAs, and 8 PFAA-precursors. TOP assays measured precursor transformation. Only RO-equipped on-site treatment systems reduced 26PFAS in permeate 98–99%. The study validates RO systems and identifies other predecessors. Chavan et al. [11] had investigated on Physico-chemical parameters of the garbage, CO concentrations, and landfill surface temperature (LST) were used to estimate the risk of spontaneous ignition of landfilled refuse at the landfill surface. The leachate pollution index was created to assess the condition of leachate throughout the monsoon, pre-monsoon, and post-monsoon seasons. The thermal properties of municipal solid waste were studied using regression analysis. New refuse is more difficult to ignite compared to older garbage.. Early smoldering and ignition are also most likely caused by the reduced MC of old waste

samples, which ranges between 3.4 and 18.2%. Landfill fire pollution will be decreased by CO concentrations, LST, and the Physico-chemical properties of landfilled trash.

Ates and Argun [6] had studied Supercritical water oxidation (SCWO) may produce phthalate esters (PAEs), however this study hypothesizes that SCWO can reduce landfill leachate's complex organic load in one step. The experiments employed 250–500 °C, 10–35 MPa, 2–18 min, and 0.2–2.3 DOD. SCWO reduced leachate COD 97% from 34,400 to 1120 mg/L. The intricate SCWO process chemistry has bonded and made these byproducts heavier. Non-target PAEs such bis(2-ethylhexyl) isophthalate and bis (1,2-benzenedicarboxylic acid bis(2-methylpropyl) ester generate DEHP. Three pathways oxidation, binding, and substitution explain how PAE breaks down during leachate SCWO. Jin et al. [21] had worked on 5 laboratory landfill reactors with different rates of ventilation. Proper ventilation, stabilized landfills and accelerated organic matter breakdown and garbage dehydration. The water mass balance showed that 94% of input water was landfilled waste water, which partially turned into leachate and evaporated water. The waste also became rougher and contained more hydrophilic substances like humic acid. The waste's bound water content reduced significantly ($p < 0.05$) as organic matter content degraded. Firmicutes and Actinobacteria broke down organic matter to increase bio-heat and water evaporation under proper ventilation. Waste bacteria used carbs and amino acids for energy. This research advances leachate minimization in microaerobic landfills. Das et al. [14] had estimated the amount of methane that is generated by landfills is an urgent necessity for nations that are rapidly urbanizing. The rapid growth of both the human population and the industrial sector has a direct and deleterious effect on the natural world. In order to take steps toward lowering emissions of

greenhouse gases, it is essential to measure the amount of methane released by municipal solid waste. It was decided to select these cities due to the unique approaches that each one takes to the management of waste.

Cui et al. [13] had examined food waste VOCs and their link to bacterial communities at different temperatures. Ethanol and esters dominated 110 VOCs in seven categories. Microbial activity and volatilization increased VOCs with temperature. From 10 to 35 °C, esters increased while alcohols declined from 94.5 to 56.7%. After 24 h, esters dominated fermentation at 35 °C and alcohols at 10 °C. Temperature affects food waste bacterial community structure. Early food waste degradation required fermentation temperature for VOC release and bacterial growth. Galvín et al. [18] had examined Three leaching methods are used on 7 RA and two natural materials to characterize the release of heavy metals and anions, which are regulated by the EU Landfill Directive. The study compares legal leaching tests to sophisticated leaching analysis methods. Percolation, compliance, and availability tests influenced data integration. Except asphalt and concrete, all recycled aggregates were safe. Finally, statistical analysis showed heavy metal levels within the permitted limits. The high quantities of Ni, Cr, Sb, Zn, and Cu necessitate ecological protection. Bolonga et al. [9] had reviewed new contaminants in our water systems are a continuing health and safety problem. Current water treatment systems cannot handle these unnamed chemicals. These threaten water delivery infrastructure. Because there are no strict guidelines for these new contaminants, the wastewater treatment system cannot remove them. Discharging undesired compounds into the aquatic environment, whether intentionally or unintentionally, harms aquatic life. The research examines how these unique contaminants may harm water consumers and cleanup methods.

Bae et al. [7] had investigated the impact of ex situ biological wastewater treatment with leachate on landfill stability. An LC system's nitrogen content and waste composition were compared to those of a nitrified leachate circulation (NLC) system. The NLC system uses an ex-situ sequencing batch reactor (SBR) to convert ammonium into nitrite or nitrate in order to denitrify landfills into nitrogen gas. NLC leachate has lower COD than LC leachate. Because NLC accelerated landfill denitrification and converted nitrite/nitrate into nitrogen gas, the gas contained more nitrogen than LC. Because landfill material acted as the carbon source for denitrification, LC combined with an SBR may reduce leachate nitrogen and accelerate landfill stabilization. Sani et al. [32] had studied reusing rubble from demolition projects reduces the amount of waste that is sent to landfills, and this practice is gaining popularity. The manufacturing of concrete from recycled construction or demolition debris (also known as C&D waste) aggregate is the focus of this particular research endeavor. Because of their low cost and high level of performance, building materials that are based on cement are utilized in the vast majority of applications; nonetheless, it is essential to keep an eye on the environmental impact that these materials have. In hydrated cementitious materials, the pH of the interstitial pore fluid is buffered by alkaline ions. Portlandite and hydrated calcium silicate are also involved in this process. The structural qualities of concrete as well as the alkali release in aqueous solution were examined through the use of a controlled experiment. The leaching of calcium was reduced by recycled aggregate, whereas the leachability of non-reactive ions (such as sodium, potassium, and chlorine) was increased. Waste concrete may be more environmentally friendly, despite the fact that it has a 40% lower mechanical resistance. Roussat et al. [31] had studied three lysimeters to conduct leaching tests after

identifying building materials that might contaminate a landfill. Only inert demolition waste was present in one lysimeter, while hazardous and inert demolition waste was present in the other two. Heavy metals and other properties were all examined in these lysimeter leachates. They compared leachate concentrations and overall emissions to the maximum landfill admission of inert wastes allowed by European law. Several things were beyond the limit. To pinpoint exactly how much demolition hazardous wastes contributed to the discovered leaks, they conducted a percolation column test using only these materials. Chang et al. [10] had monitored 96 emissions from a municipal solid waste transfer facility monthly for a year and tested over 100 odorous compounds. Ethanol, with the most oxygen, was discharged into the environment. Ethanol, dimethyl disulfide, and methyl mercaptan caused the stench. Release rate was associated with a few aromatic compounds and saturated hydrocarbons. 19 compounds more than half had an ethylbenzene correlation coefficient greater than 0.50. Representative compounds cannot estimate drug transfer release rates. Ahmadifar et al. [2] had studied for 160 days on three identical laboratory bioreactors. Bioreactors were filled with municipal solid wastes with an 80% organic content and 550 kg/m3 density. 28 L of fresh leachate were cycled daily by garbage collection vehicles. Waste is aerated in aerobic bioreactors at 0.15–0.24 L/min/kg. Leachate's 91–93% reduction in chemical oxygen demand was comparable to treatment. Between aerobic and anaerobic bioreactors were semi aerobic ones. 90 L/kg of biogas, containing 57–63% methane, was produced by anaerobic bioreactors. Less than 5% of the methane in the bioreactor was semi-aerated.

Adelopo et al. [1] had studied on FTIR, SEM/EDX, and proximal studies to compare an active dump and a decommissioned landfill. Active and closed landfill samples

had 96.5 and 98.4% primary elemental components, respectively. Al and Ca levels increased in the active landfill ($p = 0.024$ and 0.023, respectively). Cluster analysis shows that composite deterioration affects similarities more than depth or seasonality. FTIR absorption peaks at both locations supported the rubbish mineralization and stabilization concept. Active landfills had O–H vibration peaks of clayey minerals (kaolin) at 3690 and 3619 cm^{-1}, while closed landfills had a carbonates peak at 873 cm^{-1}. Both locations had garbage with over 51% ash. Al-Tabbaa and Aravinthan [4] had studied on tires with angular diameters from 1–4 to 4–8 mm and weight percentages from 6 to 15%. Despite the combination's 40% better unconfined compressive strength than clay, the compacted dry density declined because the tire was lighter. Clay-tire composite strain range at failure was higher than pure clay. Paraffin was fifty times less permeable than water. NRA and TCLP leachability testing found varied copper and nickel amounts, which must be matched to requirements. Paraffin caused clay-tire combinations to expand by 600 kPa. When analyzing test results, consider landfill liner design specifications. Maddah [25] had studied the adsorption cycles, initial salt level, and applied electric potential on 1000–3600 mg/L synthetic brackish water. The Freundlich isotherm model suited the data linearly. The results indicated using ACCs in front of membrane units in water and industrial treatment plants to improve treatment and prevent membrane fouling. Puthussery et al. [29] had examined recycling concrete into construction aggregate. Metal leaching from recycled concrete aggregates (RCA) obtained from the demolition of a 50-year-old structure was assessed using three leaching tests: compliance, availability, and Toxic Characteristic Leaching Procedure (TCLP). Metal leaching, excluding Hg, was tolerable. Fine and coarse recycled concrete aggregates were tested for

physical and mechanical qualities. RA Portland pozzolanic cement and recycled fine and coarse aggregate concrete mix was also examined. Natural and recycled concrete mixtures have similar compressive strengths. Only the 25% recycled fine aggregate mix had control concrete-like tensile strength. Other combinations dramatically reduced concrete tensile strength.

Anstey Vathani and Logeshwari [37] had studied on Demolition waste. Their study looks at the optimum technique to use Recycled Coarse Aggregate (RCA) in landfill layers. Because the RCA lacked adsorbent capacity, chemicals were applied to induce it. The adsorption of RCA was evaluated using pH, clarity, and color. In this study, the ability of different adsorbents was studied. The pilot configuration includes different layers to filter the raw leachate. RCA was used to monitor adsorbent performance. Anstey Vathani and Logeshwari [36] had studied on C&DW, the most widely collected waste, which was recycled into aggregates. The regenerated aggregate is flaky, elongated, and water-absorbing. To solve raw material shortages, this study compares recycled concrete aggregate to natural aggregate in concrete constructions. Life Cycle Assessment was evaluated on recycled and natural aggregates. LCA compares and improves product life cycle environmental impacts. Based on LCA product enhances development and good quality in RCA. Wang and Peng [38] had reviewed the use of natural zeolites as adsorbents in the treatment of water and wastewater has made considerable strides, which we analyze in this study. They discussed the features and alterations of natural zeolite. Various natural zeolites have shown varying ion-exchange capabilities for cations like ammonium and heavy metal ions all around the world. A number of techniques, including as acid treatment, ion exchange, and surfactant functionalization, are used to boost the adsorption capacity of modified zeolites for organics and anions.

Yin et al. [39] had studied both in the lab and outdoors, Recycled Aggregate Concrete (RAC) with fly ash and slag mineral admixtures was put to the test. Fly ash significantly lowered the appearance of RAC's density while increasing its workability. The ratio of RAC's splitting tensile strength to compressive strength and tension strength to compression strength both rose, enhancing its frangibility, crack resistance, and volume stability. The RAC can withstand abrasion now. The compression strength of concrete and the flexural strength, both of which met the specifications for heavy and special heavy traffic roads and could be opened to traffic in 3 days, according to the project application of RAC. Zhao et al. [40] had worked on controlled lab impregnation procedure to change the surface structure of recycled aggregate in different pastes. Pre-coated recycled aggregate did not function as well as surface-coated recycled aggregate, which fulfilled standards and had a paste thickness of 0.035 mm. With a higher cement to water ratio in the coating paste, concrete's compressive strength rose. Compressive strength was 10% greater than the control concrete after 56 days. In tests for sulfate attack resistance, recycled fine aggregate amendment significantly reduced mortar strength loss. In a study conducted by Akbarnezhad et al. [3], the researchers investigated the combined impacts of parent concrete strength, natural aggregate size, and crushing stages on various properties of coarse recycled concrete aggregates. These properties included mortar content, water absorption, bulk density, Los Angeles abrasion loss, and compressive strength. It is worth noting that the study exclusively focused on granitic coarse aggregates, as Singapore employs them exclusively for concrete production. Al-Waked et al. [5] had studied the construction and demolition rubble that can replace natural aggregate in concrete. Recycled aggregate concrete is only suitable for non-structural uses due to its low RA. The AIV

and WA of recycled coarse aggregate were examined before and after treatment to assess their effectiveness. Cyclic limewater increased RA, AIV, and WA beyond carbonation. 0.1 mm cement-silica fume slurry on RA enhanced AIV and WA by 12 and 54%, respectively. Treatment regimens may encourage RA construction. Bazienė and Vaiškūnaitė [8] had studied tire shreds that was tested in landfill bottom drainage layers to reduce congestion. Four 500-mm-long, 200-mm-diameter trial columns were fabricated. Tire shreds and fillings varied. Due to diminishing porosity and pores, landfill drainage layers clog with time. Year-long trial was carried out. Calcium, iron, and total suspended particles reduce conductivity. Because each column's monthly concentration was declining, the column research showed how these substances altered. Rubber waste (tire shreds) as column fillers enhanced layer porosity. Landfill drainage layers benefit from trash and tire shreds.

Duan et al. [15] had discussed on RA WSSD determination methods. These include wiping, slumping, centrifugation, infrared, evaporation, airflow drying, conductivity, pycnometer, hydrostatic balance, and extrapolation. Technical foundations and real-world applications are highlighted. Question all test outcomes. Open-ended pores that run opposite the centrifugal field could store significant volumes of water. The breeze's drying hindered temperature change. The original reading was calculated after RA absorbed water and established hydrostatic equilibrium. This suggests improving AC methods with liquid hydrostatic balance, micro mixers in centrifuge tubes, evaporation, and airflow drying. This evaluation improves RA WSSD estimation. Ferreira and Calisto [17] had investigated on intensive aquaculture environments, tricaine methane sulfonate (MS-222), benzocaine, and 2-phenoxyethanol (2-PE) reduce stress experienced during handling and confinement. In order to

remove water anesthetics, two waste-based adsorbents were produced by pyrolyzing paper mill sludge with commercial activated carbon. At 631, 435, and 289 mg g^{1}, commercial activated carbon adsorbs MS-222, benzocaine, and 2-PE. The results obtained from fitting the Langmuir–Freundlich model were 4–8 times greater than those obtained from the other adsorbents. García-González et al. [19] had determined the aggregates' capacity to take in water in order to calculate the amount of time they would need to spend soaking before attaining the appropriate degree of humidity for incorporation into the mixture. In addition, a number of different concrete mixes were produced by varying the amount of pre-saturation time as well as the percentage of natural aggregate that was substituted. Ismail and Ramli [20] had studied on low-concentration acid treatment for RCA to make high-quality structural concrete. The study concluded that using varying acid molarities to remove loose mortar particles from RCA surfaces improves its physical and mechanical qualities. Reducing loose mortar on RCA particles improves surface contact between fresh cement paste and aggregate. Concrete mechanical strength had increased significantly. Katz [22] had studied the 28 MPa 28-day compressive strength concrete which was crushed at 1, 3, and 28 days to imitate precast concrete facilities. Recycled aggregate was compared to fresh concrete with around 100% less aggregate.Although regenerated aggregates differed by particle size, crushed material age had minimal effect. New aggregate concrete outperformed recycled. When a strong cement matrix was used, concrete made with 3-day-old aggregates fared better than concrete made with other aggregates. Weaker cement matrix reversed it. Crushed recycled aggregates can cement. Kazmi et al. [23] had studied improvement of RA in terms of performance and evaluates RAC. The research employs carbonation,

mechanical rubbing, immersion in acetic acid, immersion in lime, immersion in acetic acid with carbonation, and acetic acid immersion with lime immersion. How RA therapy modalities alter the RAC's mechanical characteristics in both treated and untreated RA is determined through mechanical testing. Treatment improves split tensile/flexural strength and stress–strain behavior. By imitating the stress–strain behavior of natural aggregate concrete, RA immersion in acetic acid with mechanical rubbing and immersion in lime with carbonation enhance the performance of reinforced aggregate concrete (RAC).

Khatib [24] had examined concrete with FRA. Crushed concrete (CC) or crushed brick (CB) under 5 mm was referred to as recycled aggregate. The ratio of free water to cement was preserved in every combination. Concrete was replaced with CC or CB to varying degrees (0, 25, 50, and 100%). CC-concrete weakens by 15–30%. The strength of concrete with up to 50% CB, however, is comparable over time. CB can replace fine aggregate while only losing 10% of its strength. After 28 days, concrete with CC or CB builds strength more quickly than control concrete does because fine recycled aggregate improves cementing action. Concrete made of CC/CB shrinks and enlarges more. Mefteh and Kebaïli [26] had studied replaced natural aggregates with RCA in varied ratios. RCA's moisture conditioning affects hardened concrete slump and other properties. After mixing, 15, 30, 45, and 60 min, slump values were measured. Tests showed that coarse recycled aggregates need moisture treatment to maintain workability. In the initial few minutes, recycled aggregates absorb water quickly. After 30 min, slump loss decreases, indicating workability. Recycled coarse aggregate concrete compressive and splitting tensile strengths at 7 and 28 days depend on mix proportions and RCA moisture conditions. SSD recycled aggregates tend to weaken concrete most. Dry recycled aggregates had the best

strength. Nafai et al. [27] had studied two metal cations with differing valences, Ni(II) and La(III) in order to assess the adsorption of metal cations from aqueous solutions on calcium sulfate. Numerous studies looked into how the initial metal concentration, contact time, and adsorbent mass affected adsorption capacity. Ni (II) ion adsorption on calcium sulfate is best explained by the Langmuir model, which has a correlation coefficient of 0,976, while La (III) ions are best explained by the Freundlich model, which has a correlation coefficient of 0,875. Adsorption is controlled by the pseudo-second-order kinetic model for both metal cations. Metal cations (Ni (II) and La (III)) are adsorbed by calcium sulfate produced during the wet process of phosphoric acid manufacture. Norashiddin and Kamaruddin [28] had studied solid waste disposal at landfills was most cost-effective and desirable. Landfilling produces filthy leachate. Leachate management can contaminate groundwater and surface water, causing health issues. Chemical, biological, and combination therapies remediate landfill wastewater. Recent researches show that activated carbon adsorption can purify landfill leachate. The best approach physically and chemically sorbs refractory chemicals from real and liquid effluent. Natural, synthetic, and agricultural waste precursors have also been studied. Landfill leachate treatment uses activated carbon and agricultural debris. The primary development of activated carbon adsorption, focusing on inexpensive precursors, its limitations, and its future prospects will also be covered.

Reddy et al. [30] had reported the use of waste tire shreds as drainage material in abandoned landfill cover systems. Extensive laboratory testing and outdoor demonstrations in a vacant landfill near Carlinville, Illinois are part of the investigation. With various cover materials, the clogging, long-term transmissivity, and interface shear strengths of tire fragments were evaluated. The slope stability of the completed coverings with tire drainage

layers was evaluated. In a field test, tire fragments were compared to sand as a final cover drainage medium for landfills. Crushed tires serve as an effective drainage material. Ground tires can be used to cover abandoned landfills and discharge used tires for a low price. Sátiro et al. [33] had investigated the best bacterial and microalgae inoculation ratio for system initiation and removal of COD, and other configurations in six 1000-mL photobioreactors. Scenedesmus sp. microalgae and pulp mill biomass aerobic (BA) ratios were utilized in reactors for 14 days: 0:1 (PBR1), 1:1 (PBR2), 1:1, 3:1, 5:1, and 1:3 (PBR6). PBR4, PBR5, and PBR6 eliminated COD in two days, unlike reactors with lower inoculum. PBR5 and PBR6 eliminated $NH4+$-N with 86.81 and 77.11% bacterial nitrification and microalgal absorption, respectively. Bacteria and phytoplankton combined removed 86% PO43-P. The highest BA PBRs, PBR4 and PBR5, flocculated 90% on the second day, increasing biomass output and sediment ability. PBR4 and PBR5 showed increased protein synthesis and BA, 49.90 and 49.05 mg/L. All reactors except PBR1 (just MA) had cyanobacteria and Chlorophyceae, indicating microalgae-bacteria consortiums. SEM showed bacteria and algae in filamentous microalgae. Singh et al. [34] had examined 28-day-old concrete density and water absorption for 10 blocks of GGBFS and FA at 60 and 100% wastes from 30-year-old buildings which are replaced by RCA. Normal weight concrete passed a physical test. The experiment showed that 40% NCA/RCA replacement with 20% GGBFS yielded good results. Their study analyzes how GGBFS and FA cement substitutes affect RCA physical properties. Tong et al. [35] had examined $CaCl_2$, P-DcE, and CPAM disperse ultraheavy oil from effluent. With the optimal P-DcE to $CaCl_2$ to CPAM ratio of 20:600:1.2 (m/m), a new Reverse Emulsion Breaker (REB) removes 98.04% mineral oil and 94.48% COD. In terms of COD removal,

cost, and environmental sustainability, the REB outperforms P-DcE and concentrated sulfuric acid. The GC–MS analysis demonstrates that the REB removes more organic compounds than mineral lubricants, leaving predominantly oil field chemicals such as corrosion inhibitors, scaling inhibitors, biocides, and demulsifies. REB is safe, efficient, and cost-effective for SHOW anaerobic digestion and synergistic $CaCl_2$ demulsification.

15.3 Research Gap/Objective

Based on extensive reviews and articles, it has been observed that Recycled Aggregate (RA) has the potential to absorb and can be utilized as a landfill liner when endowed with adsorption capabilities. To construct an effective adsorption layer on the RCA, various adsorbents available in the industry were carefully selected and shortlisted. The research was conducted in four stages: (1) Evaluating the interaction between the adsorbents and Recycled Aggregate (RA) to ensure compatibility. (2) Identifying the most suitable layer adsorbents and RA. (3) Identifying the most suitable adsorbents. (4) Experimenting with different proportions of the selected adsorbents to optimize their performance. Further research is needed to assess the success and efficiency of the adsorbents in treating unfiltered leachate in RCA. The primary objective of this study is to confer adsorbing properties to RCA and explore its potential as a landfill liner. In this investigation, broken aggregate composed of crushed waste concrete obtained from disposal sites was used without removing the cementitious particles. The study involved incorporating chemicals such as clay, sawdust, and paper pulp to create a coating on RA and assess the effectiveness of these chemicals as adsorbents at varying proportions. The aim was to develop a filtration system using adsorbents to treat

leachate collected from waste disposed of in landfills. The results and optimal applications of these substances were further explored in this research.

15.4 Methodology

The study's methodology, as illustrated in Fig. 15.1, involves a step-by-step process. The initial phase includes the selection of Construction and Demolition (C&D) wastes, which are then crushed to obtain the desired aggregate size, specifically those retained on a 6.7 mm sieve. Essential tests for Recycled Concrete Aggregate (RCA) are conducted, and unfiltered leachates are collected using a setup alongside municipal solid wastes obtained from the surrounding area. Following the collection of unfiltered leachates, a specialized setup, depicted in the figure, is designed to accommodate layers of RCA, Paper Pulp, and adsorbents. The experiment spans over two weeks, during which observations are made to assess further characteristics. The collected leachate is passed through the setup to undergo filtration. In summary, the methodology involves selecting C&D wastes, crushing them for specific aggregate sizes, conducting essential RCA tests, collecting unfiltered leachate with a setup using municipal solid wastes, designing a setup for layering RCA, Paper Pulp, and adsorbents, and then observing the experiment's outcomes over two weeks with filtered leachate passing through the setup.

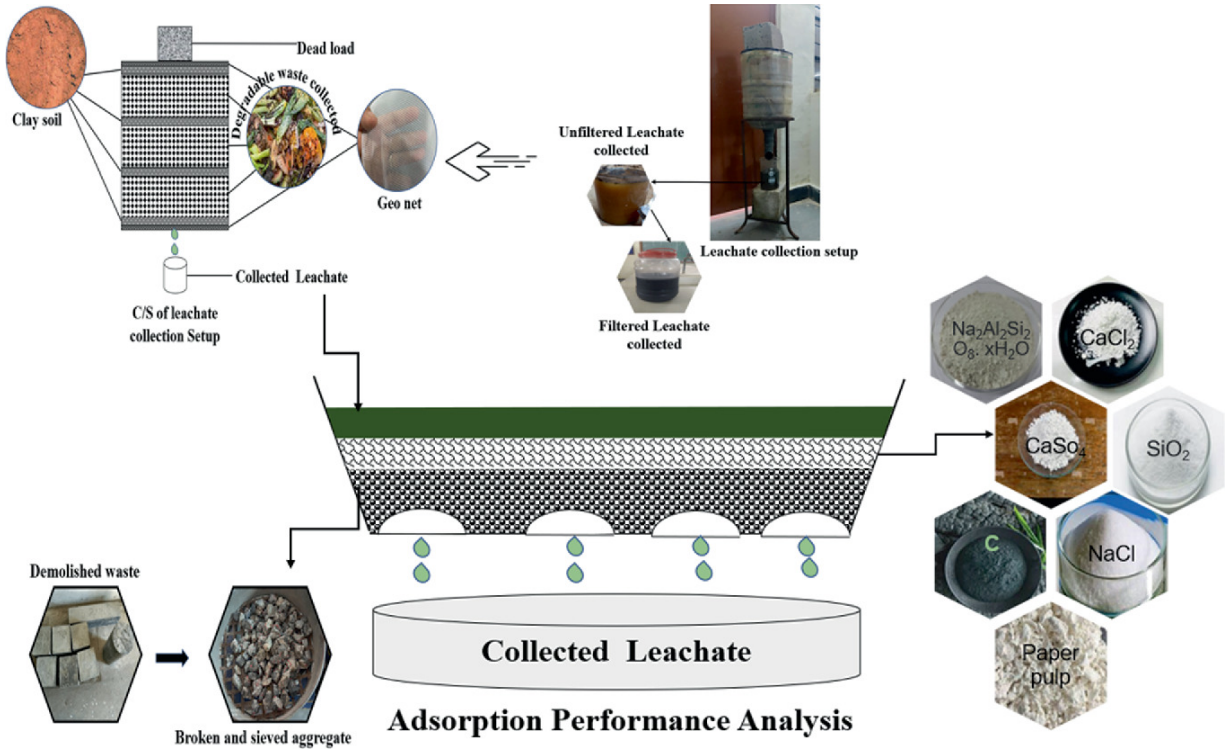


Fig. 15.1 Methodology for feasibility study

15.5 Materials

Figure 15.2 illustrates the materials utilized in the study and the various stages of the pilot study. It also includes a list of chemicals and their proportions used in each stage to assess the capacity and quality of the adsorbents.

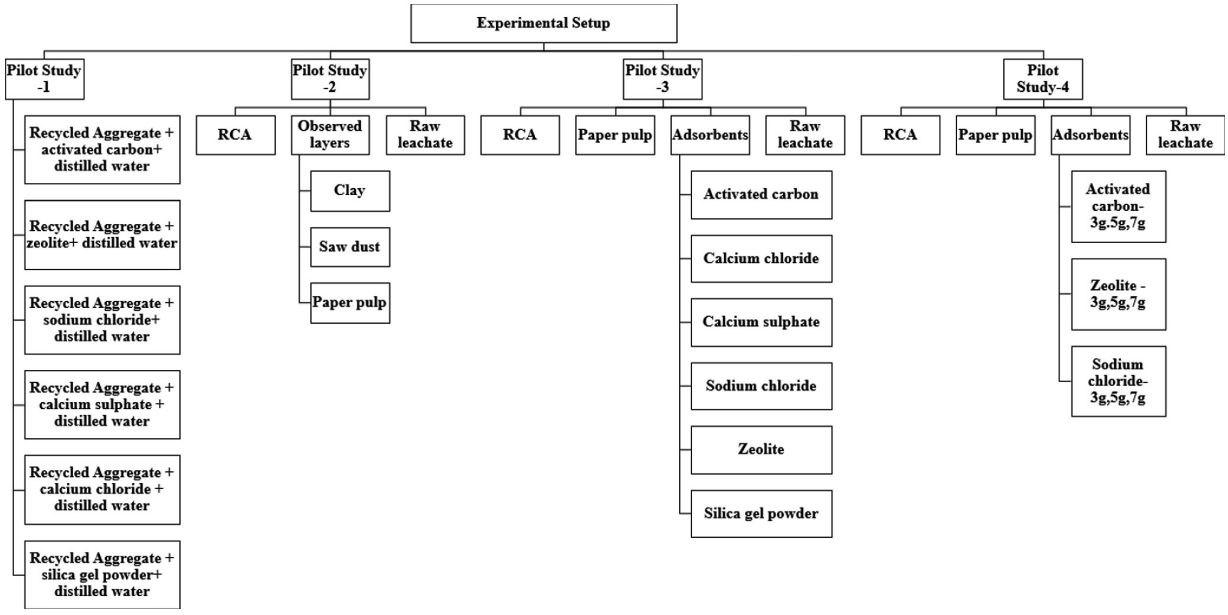


Fig. 15.2 Materials used in the feasibility study

The flowchart displays the following key components: 1. Material Selection: Various materials used in the study are identified and chosen. 2. Pilot Study Stages: The study is divided into different stages to systematically evaluate the adsorbents. 3. Chemicals and Adsorbents: A list of chemicals is presented, along with their respective proportions used in each stage.4. Evaluation: The capacity and quality of the adsorbents are assessed throughout the study. The flowchart serves as a visual representation of how the study progresses through different stages, testing various adsorbents and their effectiveness in treating the leachate.

15.6 Experimental Investigation

The experimental investigation is carried out in four different pilot studies briefed below

1. Evaluating the interaction between the adsorbents and Recycled Aggregate (RA) to ensure compatibility.

The study was carried out on RA to check if there was any reaction on the surface of the RA which is having cementious particle. The Sample is placed in paper cup along with various proportions of adsorbents namely activated carbon, zeolite, calcium sulphate, calcium chloride, sodium chloride and silica gel powder which was mixed with 500 ml of distilled water and stirred uniformly using Magnetic stirrer. Figure 15.3 shows the samples placed in paper cup which was monitored for 24–48 h. The observed result indicates that the chemicals do not directly stick to the aggregate as it forms a layer on top of the surface but becomes loose when they are shaken. Further study on finding a coating for RA was needed.



Fig. 15.3 Setup carried out to find the interaction between RA and adsorbent

2. Identifying the most suitable layer adsorbents and RA.

Based on the pilot study-1 it was concluded that RA needs a coating material in order to allow the adsorbent characteristics to boost and filter the raw leachate. Further literatures were studied and the materials such as saw dust, paper pulp and clay were shortlisted as a binder material. A small study was carried out to check the capacity of these material as shown in Fig. 15.4. Based on

the study it was concluded that saw dust showed poor performance it absorbed most of the raw leachate, Clay showed a promising result but with a disadvantage of hardening and clogging. Paper pulp in the other hand showed a good performance in terms of filtering the raw leachate as well as acting as a binder in RA.



Fig. 15.4 Trail carried out to identify the binder material (from left saw dust, clay, paper pulp)

3. Identifying the most suitable adsorbents.

The third stage of pilot study was to identify the most suitable adsorbents among Activated Carbon (AC), Zeolite (Z), Sodium Chloride (SC), Calcium Chloride (CC), Calcium Sulphate (CS) and Silica Gel Powder (SGP) (The adsorbents are henceforth mentioned as AC, Z, SC, CC, CS AND SGP). These adsorbents were added to the setup containing Recycled Coarse Aggregate (RCA), paper pulp and raw leachate collected from municipal solid waste. The setup was observed for over 2 weeks to check the performance of adsorbents. The study concluded that the performance of CC, CS and SGP was rather poor when compared to AC, Z and SC. It was also noticed that the cause of failure for poor performing adsorbents where clogging. The raw leachate was observed to be changed from acidic nature to basic nature. Figure 15.5 indicated the study carried out in identifying the suitable adsorbents.



Fig. 15.5 Setup used to observe the adsorbents

4. Experimenting with different proportions of the selected adsorbents to optimize their performance

Figure 15.6 shows the experimental setup designed for the final pilot study. Based on the third pilot study and carrying out the setup with the same materials along with an increased proportion of adsorbents at 3, 5 and 7 g for the shortlisted three adsorbents namely AC, Z and SC. The setup for absorbed for over 1 month i.e., approximately 45 days. The observed results states that RCA indeed can be induced with adsorbent characteristics with the chemical. Moreover, the results on the collected leachate that is passing through the leachate shows that the nature of the sample changes from acidic nature to basic nature but the color of the sample and the turbidity of the sample remains the same. During the inspection the filtered leachate passing through the setup was collected and tested for pH for the 45 days of observation. It was concluded that the chemicals AC, SC and Z with various proportions shows gradual increase in the nature of the sample. Whereas the color and turbidity of the sample remains the same throughout the observation. Further research should be carried out in the future to check the change in the color of filtration as the day's increases. The results of the pH observed is shown in Fig. 15.8. This also

indicates that with further study it can be used as a liner in landfill.



Fig. 15.6 Leachate adsorption setup

15.7 Results and Discussions

Following is a discussion of the findings of the best suitable adsorbents for filtering the leachate that was gathered from the municipal solid waste and identifying the coating layer for RA. The results on pilot study-3 and 4 are discussed below.

They were examined for the following parameters

1. Leachate Collection

2. pH Observation

3. Lucidity

4. Color /Luminosity.

15.8 Pilot Study-3

1. Collection of Leachates

The filtered leachate that was collected using the setup as shown in Fig. 15.7 is depicted below. The raw leachate was refilled for every third day. It was observed that as the days goes the collection was lesser in CC.CS and SGP.

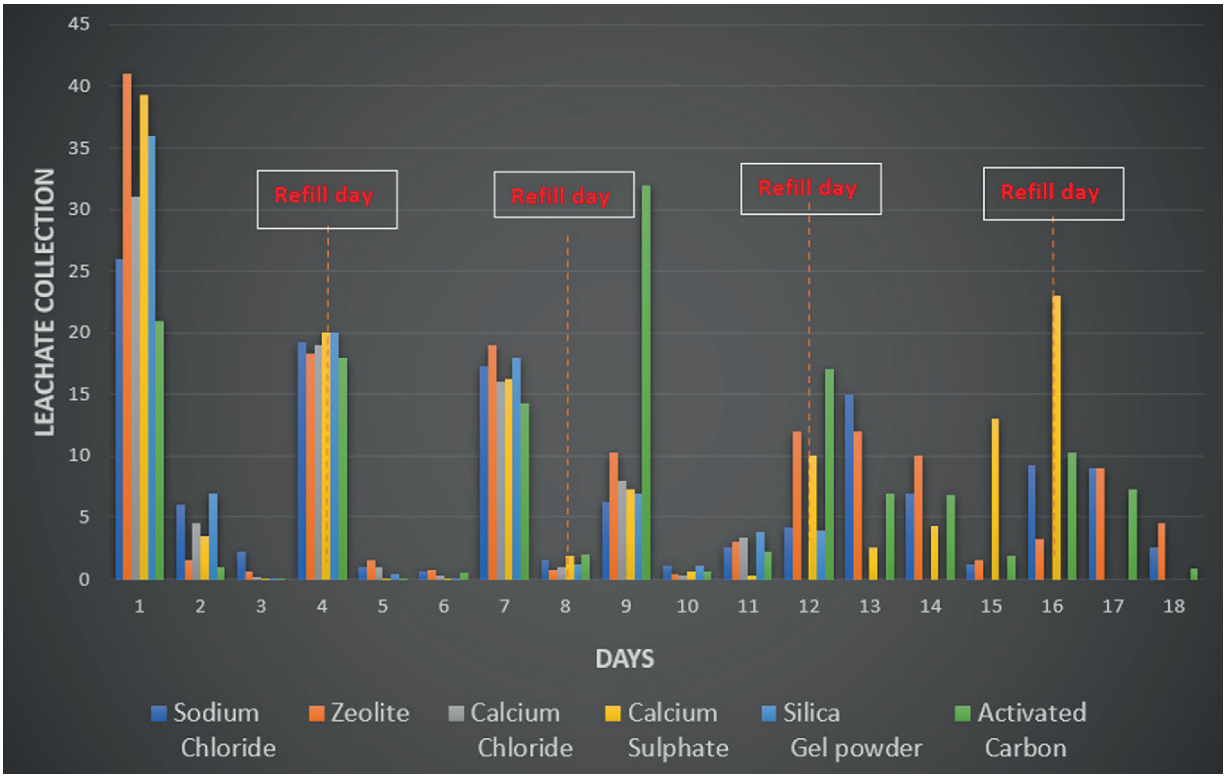


Fig. 15.7 Leachate collected for the adsorbents

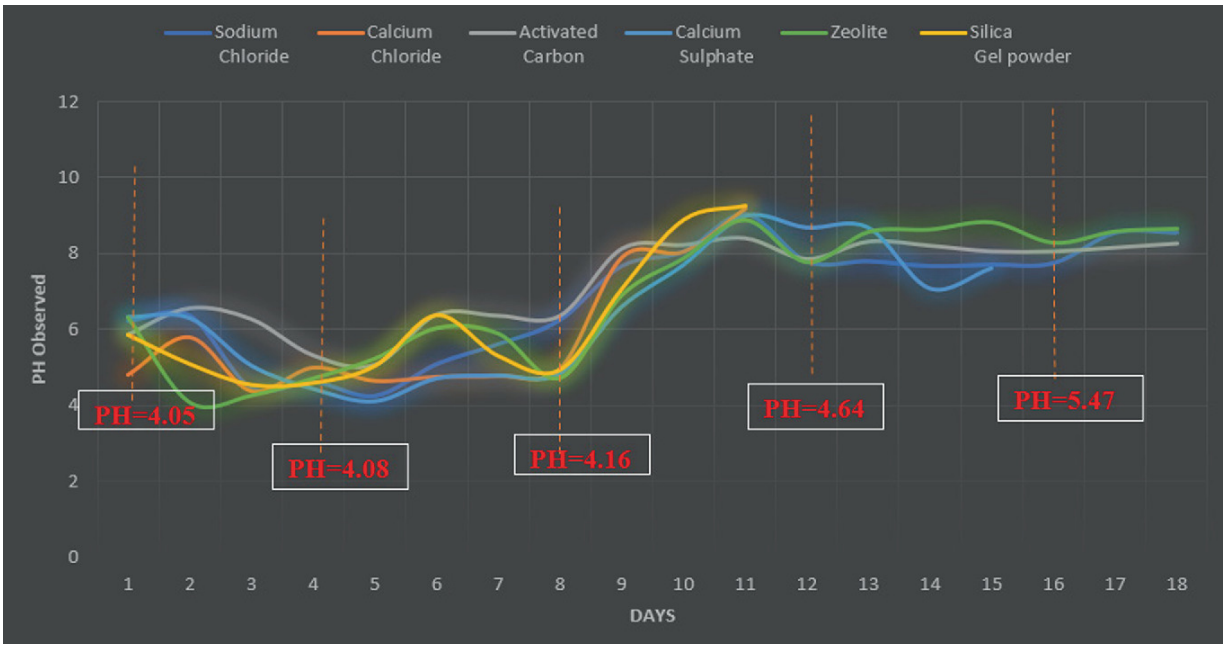


Fig. 15.8 pH observed for the adsorbents

2. **PH Observation**

The pH observed for the six adsorbents are depicted as shown in Fig. 15.8. The performance of AC, Z and SC shows good performance than CC.CS and SGP.

3. **Color**

The variation in the of the filtered leachate sample is noted. Figure 15.9 shows the representation of color change of collected leachate for AC, CC, CS, Z, SC and SGP. It was noted that the filtered leachate from activated carbon, zeolite and NaCl shows greater filtration than calcium sulphate, calcium chloride and silica Gel powder.

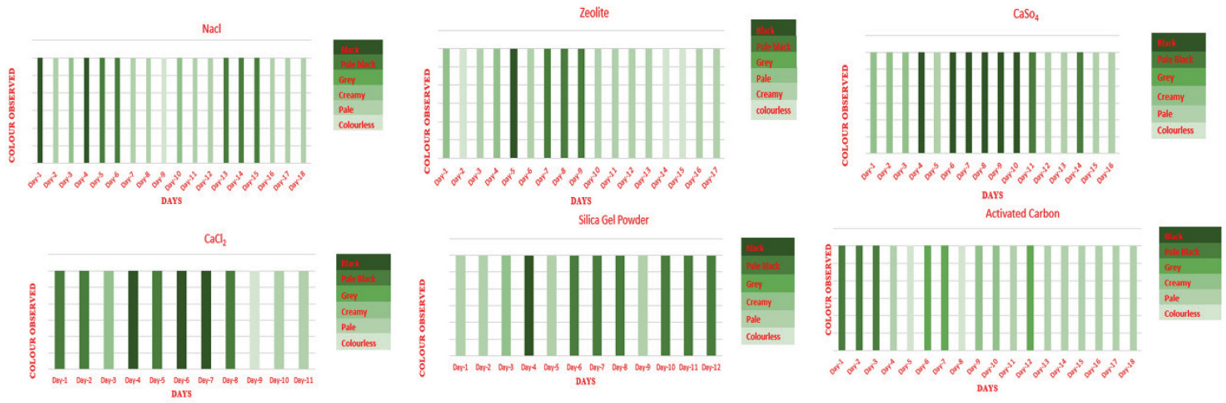


Fig. 15.9 The color chart for the observed adsorbents

4. **Lucidity**

As the day progresses, it has been noticed that the sample transitions from acid to base. The lucidity of the samples as observed from the setup was depicted in Fig. 15.10.

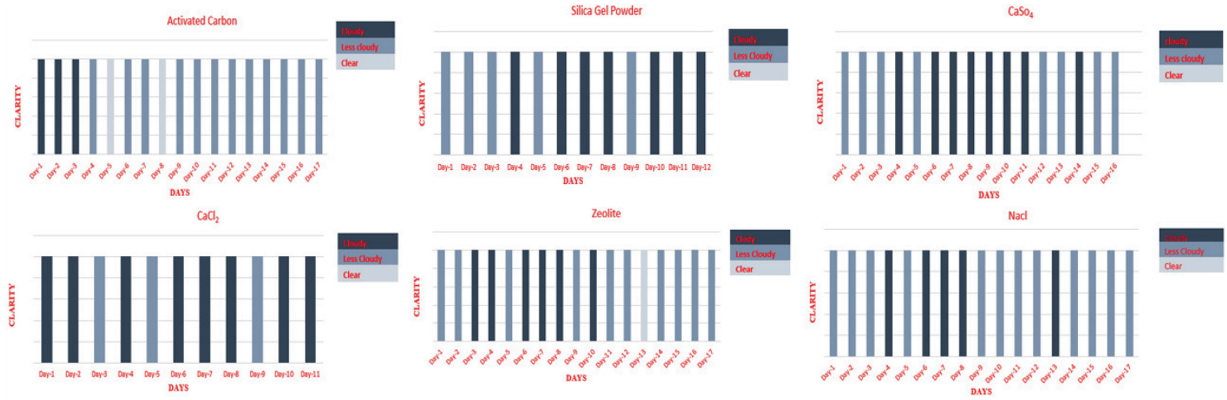


Fig. 15.10 Lucidity chart for the observed adsorbents

15.9 Pilot Study-4

1. **Collection of Leachates**

Leachate from all of the adsorbents is collected using the set-up depicted in Fig. 15.6. The amount of leachate collected through the setup varies, as illustrated by the curve in Fig. 15.11, for activated carbon, sodium chloride

and zeolite. When the leachate is filtered via RCA, it has been found that activated carbon, sodium chloride, and zeolite function well.

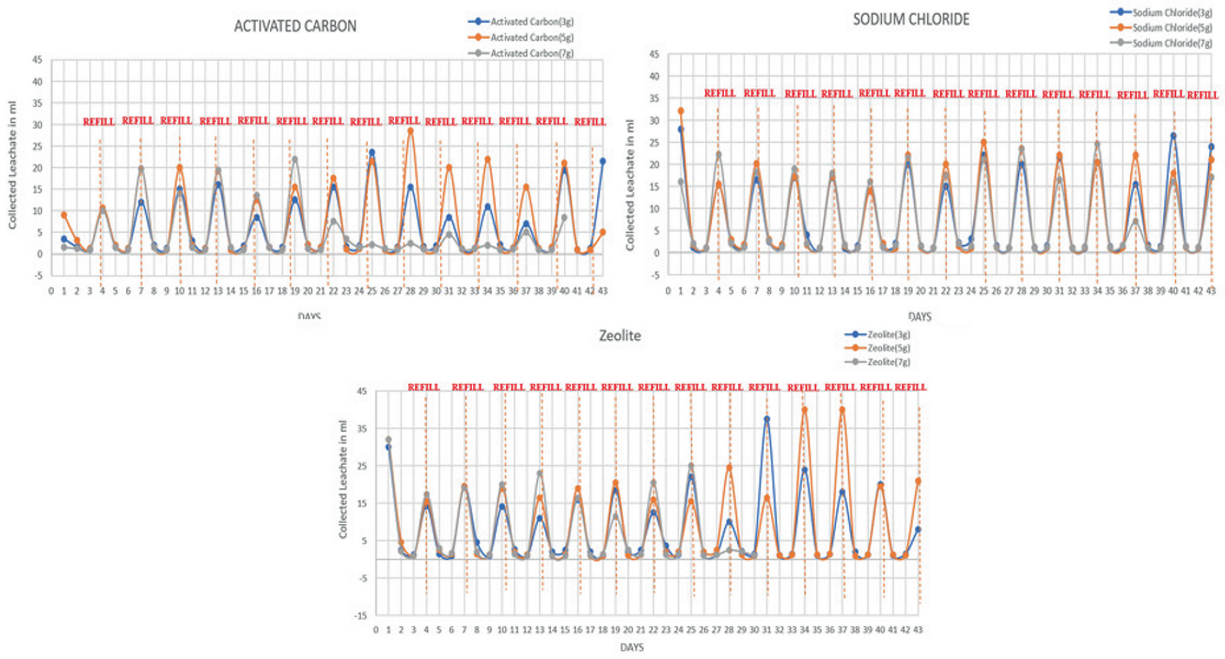


Fig. 15.11 Leachate collection observed for the adsorbents

2. **pH Observation**

It was observed that the sample changes from acid to base as the day's increases. The graph shown in Fig. 15.12 indicated the pH value of the samples collected from activated carbon, calcium chloride and zeolite induced RCA setup. The amount of leachate varies based on the adsorbents added on RCA. It was observed that activated carbon, zeolite and sodium chloride shows great change from acid to base.

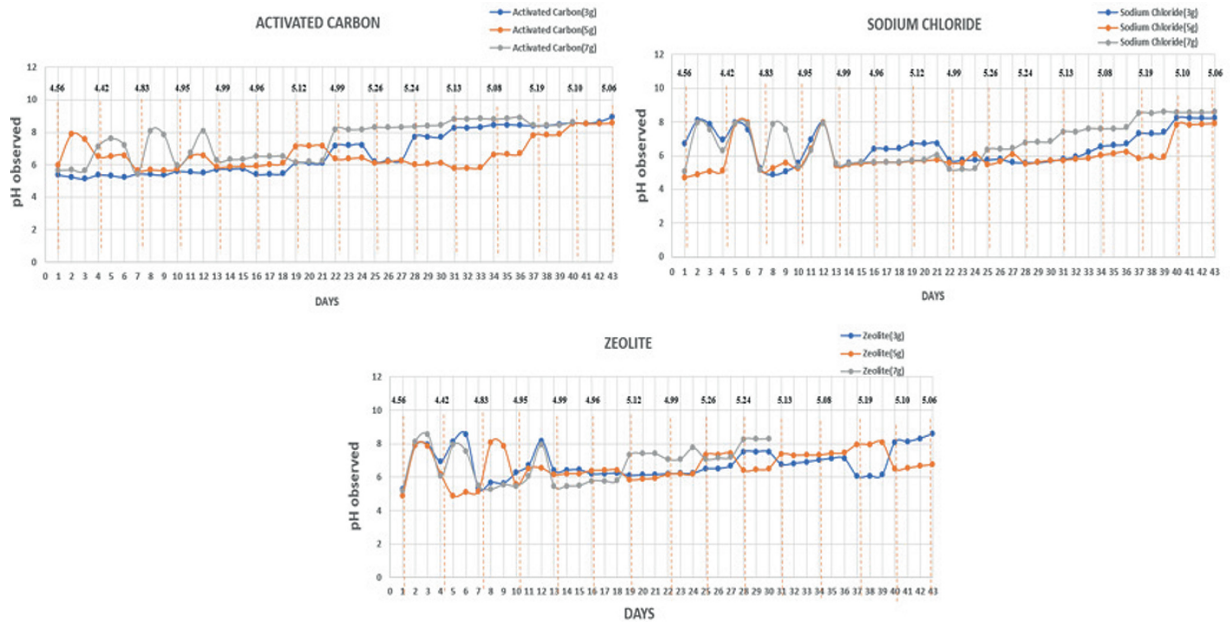


Fig. 15.12 pH observed for the adsorbents

3. **Lucidity**

As the day progresses, it has been noticed that the sample transitions from acid to base. The lucidity of the samples as observed from the setup was depicted on the graph in Fig. 15.13. It was noted that the lucidity of the filtered leachate collected was rather slow and further investigation is needed.

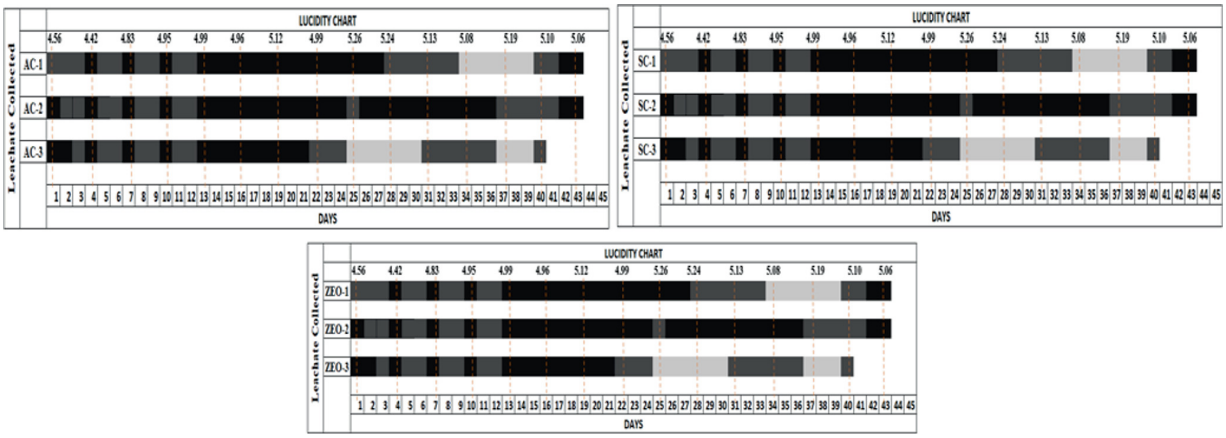


Fig. 15.13 Lucidity chart observed for the adsorbents

4. **Color**

The variation in the filtered leachate sample is noted. Figure 15.14 shows the representation of color change of collected leachate for activated carbon, NaCl and zeolite. It was noted that the filtered leachate from activated carbon, zeolite and NaCl shows slow Pigmentation.

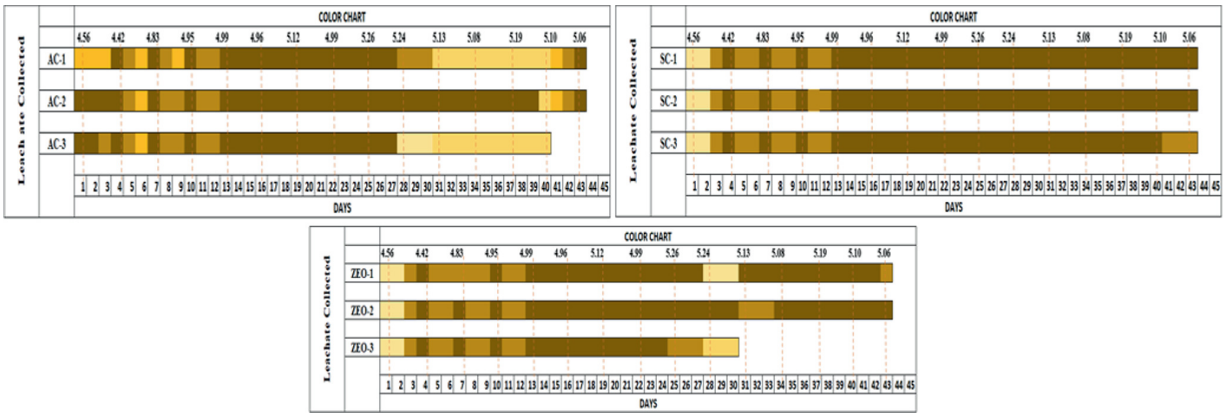


Fig. 15.14 Color chart observed for the adsorbents

15.10 Conclusion

The studies that are carried out to identify the suitable adsorbent and to identify the suitable binder for RA and adsorbent are discussed below. Various literatures are studied and the adsorbents are shortlisted based on the property and availability. The shortlisted adsorbents are activated carbon, Zeolite, Calcium Chloride, Calcium Sulphate, Sodium Chloride and Silica Gel Powder. The initial Study was carried out to check if the adsorbents and RA are binding together and to check for any reactions on the concrete deposited on RA. The study concluded that the adsorbents do not stick directly on the aggregate and further investigation on the binding material is needed. The second stage of study was to find a suitable material to act as a binder for the RA and the selected adsorbents. The

study was carried out using paper pulp, saw dust, clay and raw leachate. The study concluded that clay and saw dust shows the failure in filtering the raw leachate as the clay started hardening at the end of observation and Saw dust absorbed most of the leachate injected for filtration. On the other hand, the paper pulp showed a good property of filtration and acted as a good binder for RA and adsorbents without causing any further problems.

The third stage of study included a setup that consists of RCA, paper pulp, adsorbents (AC, Z, SC, CC, CS and SGP), Raw leachate. The study was observed for over two weeks and concluded that AC, Z and SC showed good performance in filtration as well as changing the acidic nature of raw leachate to basic nature. whereas, CC, CS and SGP showed poor characteristics as it started clogging and showed poor change in nature. Further investigations should be carried out using the shortlisted chemicals from this study by increasing the proportion of the adsorbents. The final stage of study was carried with the similar setup as that of pilot study -3 but with AC, Z and SC at 3, 5 and 7 g. The observation was carried out for forty-five days in total. Throughout the study it was observed that the collection of filtered leachates was smooth even after 40 days but the color of the filtered leachate was changing rather slow but the nature of the filtered leachate changed from acidic nature to basic nature. To examine the quality of filtration, or the transformation of the color from dark black to colorless, more research should be done.

Acknowledgement

Authors are thankful to Vel Tech Rangarajan Dr. Sagunthala R & D Institute of Science and Technology for providing the facility to carry out research work and also the financial support in the form of SEED FUND (Sanctioned Order : VTU SEED(FY22-23)-11).

References

1. Adelopo AO, Haris PI, Alo B, Huddersman K, Jenkins RO (2017) Comparative characterisation of closed and active landfill composites using EDX, FTIR and proximate techniques. Waste Biomass Valorization 8:1313–1323
2. Ahmadifar M, Sartaj M, Abdallah M (2016) Investigating the performance of aerobic, semi-aerobic, and anaerobic bioreactor landfills for MSW management in developing countries. J Mater Cycles Waste Manage 18:703–714
3. Akbarnezhad A, Ong KCG, Tam CT, Zhang MH (2013) Effects of the parent concrete properties and crushing procedure on the properties of coarse recycled concrete aggregates. J Mater Civ Eng 25(12):1795–1802. https://doi.org/10.1061/(ASCE)MT.1943-5533.0000789
4. Al-Tabbaa A, Aravinthan T (1998) Natural clay-shredded tire mixtures as landfill barrier materials. Waste Manage 18(1):9–16. https://doi.org/10.1016/S0956-053X(98)00002-6
5. Al-Waked Q, Bai J, Kinuthia J, Davies P (2022) Enhancing the aggregate impact value and water absorption of demolition waste coarse aggregates with various treatment methods. Case Stud Constr Mater 17:e01267. https://doi.org/10.1016/j.cscm.2022.e01267
6. Ateş H, Argun ME (2023) Fate of phthalate esters in landfill leachate under subcritical and supercritical conditions and determination of transformation products. Waste Manage 155:292–301
7. Bae W, Kim S, Lee J, Chung J (2019) Effect of leachate circulation with ex situ nitrification on waste decomposition and nitrogen removal for early stabilization of fresh refuse landfill. J Hazard Mater 371:721–727
8. Bazienė K, Vaiškūnaitė R (2016) Research of sustainable use of tire shreds in landfill. Sustainability 8(8):767. https://doi.org/10.3390/su8080767
9. Bolong N, Ismail AF, Salim MR, Matsuura T (2009) A review of the effects of emerging contaminants in wastewater and options for their removal. Desalination 239(1–3):229–246
10. Chang H, Tan H, Zhao Y, Wang Y, Wang X, Li Y, Lu W, Wang H (2019) Statistical correlations on the emissions of volatile odorous compounds from the transfer stage of municipal solid waste. Waste Manage 87:701–708
11. Chavan D, Lakshmikanthan P, Manjunatha GS, Singh D, Khati S, Arya S, Tardio J, Eshtiaghi N, Mandal P, Kumar S, Kumar R (2022) Determination

of risk of spontaneous waste ignition and leachate quality for open municipal solid waste dumpsite. Waste Manage 154:272–282

12. Chen Y, Zhang H, Liu Y, Bowden JA, Tolaymat TM, Townsend TG, Solo-Gabriele HM (2022) Concentrations of perfluoroalkyl and polyfluoroalkyl substances before and after full-scale landfill leachate treatment. Waste Manage 153:110–120

13. Cui Y, Zhang H, Zhang J, Lv B, Xie B (2022) The emission of volatile organic compounds during the initial decomposition stage of food waste and its relationship with the bacterial community. Environ Technol Innov 27:102443

14. Das D, Majhi BK, Pal S, Jash T (2016) Estimation of land-fill gas generation from municipal solid waste in Indian Cities. Energy Procedia 90:50–56

15. Duan Z, Zhao W, Ye T, Zhang Y, Zhang C (2022) Measurement of water absorption of recycled aggregate. Materials 15(15):5141. https://doi.org/10.3390/ma15155141

16. Ergene D, Aksoy A, Sanin FD (2022) Comprehensive analysis and modeling of landfill leachate. Waste Manage 145:48–59

17. Ferreira CI, Calisto V, Otero M, Nadais H, Esteves VI (2016) Comparative adsorption evaluation of biochars from paper mill sludge with commercial activated carbon for the removal of fish anaesthetics from water in recirculating aquaculture systems. Aquacult Eng 74:76–83. https://doi.org/10.1016/j.aquaeng.2016.06.003

18. Galvín AP, Ayuso J, Agrela F, Barbudo A, Jiménez JR (2013) Analysis of leaching procedures for environmental risk assessment of recycled aggregate use in unpaved roads. Constr Build Mater 40:1207–1214

19. García-González J, Rodríguez-Robles D, Juan-Valdés A, Morán-del Pozo JM, Guerra-Romero MI (2014) Pre-saturation technique of the recycled aggregates: solution to the water absorption drawback in the recycled concrete manufacture. Materials 7(9):6224–6236. https://doi.org/10.3390/ma7096224

20. Ismail S, Ramli M (2013) Engineering properties of treated recycled concrete aggregate (RCA) for structural applications. Constr Build Mater 44:464–476. https://doi.org/10.1016/j.conbuildmat.2013.03.014

21. Jin P, Bian S, Yu W, Guo S, Lai C, Wu L, Zhao H, Xiao K, Liang S, Yuan S, Huang L (2023) Insights into leachate reduction in landfill with different ventilation rates: balance of water, waste physicochemical properties, and microbial community. Waste Manage 156:118–129

22. Katz A (2003) Properties of concrete made with recycled aggregate from partially hydrated old concrete. Cem Concr Res 33(5):703–711. https://doi.org/10.1016/S0008-8846(02)01033-5
23. Kazmi SMS, Munir MJ, Wu YF, Patnaikuni I, Zhou Y, Xing F (2019) Influence of different treatment methods on the mechanical behavior of recycled aggregate concrete: a comparative study. Cement Concr Compos 104:103398. https://doi.org/10.1016/j.cemconcomp.2019.103398
24. Khatib JM (2005) Properties of concrete incorporating fine recycled aggregate. Cem Concr Res 35(4):763–769. https://doi.org/10.1016/j.cemconres.2004.06.017
25. Maddah HA (2020) Adsorption isotherm of NaCl from aqueous solutions onto activated carbon cloth to enhance membrane filtration. J Appl Sci Eng 23(1):69–78
26. Mefteh H, Kebaïli O, Oucief H, Berredjem L, Arabi N (2013) Influence of moisture conditioning of recycled aggregates on the properties of fresh and hardened concrete. J Clean Prod 54:282–288. https://doi.org/10.1016/j.jclepro.2013.05.009
27. Nafai H, Lamzougui G, Chafik D, Bouhaouss A, Bchitou R (2017) Performance evaluation of calcium sulfate in adsorption process of metal ions from aqueous solutions. JMES 8:982–989. http://www.jmaterenvironsci.com
28. Norashiddin FA, Kamaruddin MA, Emmanuel IM, Pakir MF (2019) Activated carbon for landfill leachate treatment: a review. Int J Dev Sustain 8(1):19–29. www.isdsnet.com/ijds
29. Puthussery JV, Kumar R, Garg A (2017) Evaluation of recycled concrete aggregates for their suitability in construction activities: an experimental study. Waste Manage 60:270–276
30. Reddy KR, Stark TD, Marella A (2010) Beneficial use of shredded tires as drainage material in cover systems for abandoned landfills. Pract Period Hazard Toxic Radioact Waste Manage 14(1):47–60. https://doi.org/10.1061/(ASCE)1090-025X(2010)14:1(47)
31. Roussat N, Méhu J, Abdelghafour M, Brula P (2008) Leaching behaviour of hazardous demolition waste. Waste Manage 28(11):2032–2040
32. Sani D, Moriconi G, Fava G, Corinaldesi V (2005) Leaching and mechanical behaviour of concrete manufactured with recycled aggregates. Waste Manage 25(2):177–182
33. Sátiro J, Cunha A, Gomes AP, Simões R, Albuquerque A (2022) Optimization of microalgae-bacteria consortium in the treatment of paper

pulp wastewater. Appl Sci 12(12):5799. https://doi.org/10.3390/app12125799

34. Singh OP, Kumar R, Singar MK (2022) Experimental studied of density and water absorption of recycled coarse aggregate concrete incorporating GGBFS and FA. In: IOP conference series: materials science and engineering. IOP Publishing, vol 1224, no 1, p 012023. https://doi.org/10.1088/1757-899X/1224/1/012023

35. Tong K, Zhang Y, Chu PK (2013) Evaluation of calcium chloride for synergistic demulsification of super heavy oil wastewater. Colloids Surf A 419:46–52. https://doi.org/10.1016/j.colsurfa.2012.11.047

36. Vathani A, Logeshwari L (2023) Sustainability in using recycled aggregate. In: AIP conference proceedings. AIP Publishing, vol 2766, no 1

37. Vathani TA, Logeshwari J (2023) A novel approach to utilize recycled concrete aggregates as landfill liner. Waste Manage Bullet 1(1):39–48

38. Wang S, Peng Y (2010) Natural zeolites as effective adsorbents in water and wastewater treatment. Chem Eng J 156(1):11–24

39. Yin J, Chi Y, Gong S, Zou W (2010) Research and application of recycled aggregate concrete. In: Paving materials and pavement analysis, pp 162–168

40. Zhao Z, Wang S, Lu L, Gong C (2013) Evaluation of pre-coated recycled aggregate for concrete and mortar. Constr Build Mater 43:191–196

© The Author(s), under exclusive license to Springer Nature Switzerland AG 2024
S. Souabi, A. Anouzla (eds.), *Landfill Leachate Treatment Techniques*, Springer Water
https://doi.org/10.1007/978-3-031-63157-3_16

16. Performance Comparison of Conventional Method and Ultrafiltration in the Pretreatment of Landfill Leachate

Mohamed Zait[1] and Mohamed Taky[1] ✉
(1) Laboratory of Advanced Materials and Process Engineering, Faculty of Sciences, Ibn Tofail University, P.O. Box: 1246, Kenitra, Morocco

✉ **Mohamed Taky**
Email: mohamed.taky@uit.ac.ma

Abstract
After biological and/or physico-chemical treatment of landfill leachates (LFLs), chemical oxygen demand (COD) and total dissolved solids (TDS) content are often larger than reject requirements. In this case, reverse osmosis (RO) can effectively treat these residual COD and TDS. However, RO feasibility is limited by membrane fouling. LFL treatment of Oum Azza treatment is performed in two stages: a biological pretreatment, which consists of an aeration basin and an anoxic basin followed by a pocket filter, and in the second stage, the pretreated LFL is sent to

RO unit. The monitoring of the performances of the plant shows a frequent stops for cleaning the fouling membranes due to the poor effluent quality after pretreatment. The pretreatment is the key of this operation, in terms of permeate fluxes and membranes lifetime. In this study, two pretreatment methods will be studied and compared: coagulation-flocculation by ferric chloride and aluminium sulfate and ultrafiltration (UF) membranes with different pore sizes (0.02, 0.05 and 0.1 µm). The performances of two pretreatment processes will be analyzed in terms of reduction of the major pollution indicators suspended solids (SS), COD and Biochemical oxyden demand (BOD_5).

Keywords Landfill leachate – Pollutant load – Ultrafiltration – Ceramic membrane – Coagulation-flocculation

Abbreviations

LFL Landfill leachate

COD Oxygen demand

TOC Total organic carbon

SS Suspended solids

OLR Organic loading rate

TDS Total dissolved solids

TN Total nitrogen

TP Total phophorus

EC Electric conductivity

RO Reverse osmosis

UF Ultrafiltration

MF Microfiltration

NF Nanofiltration

Coag-Floc Coagulaton-Flocculation

TMP Transmembrane pressure

CV Circulation velocity

BOD_5 Bichemical oxyden demand

$PNDM$ National Household Waste Program

VFA Volatile fatty acid

AC Activated carbon

PAC Powdered activated carbon

AOP Advanced oxidation processes

AD Anaerobic digestion

16.1 Introduction

Fast urbanization, population growth and changing production and consumption patterns generate significant volumes of waste, both in liquid and solid. With a population of 34 million and an urbanization rate of over 60%, Morocco has not escaped the inexorable growth in the quantity of waste produced which has reached 5.5 million tons of urban household waste and which will reach 9.3 million tons in 2030. At the national level, the total yearly production of waste reaches more than 7 million tons according to the National Household Waste Program (PNDM) [1], with a predominance of organic matter (70% causing 18% of the country's total greenhouse gas emissions). The industry generates more than 1.5 million tons annually, of which 256,000 tons are hazardous waste. In addition, medical waste represents 6000 t/y [2]. This waste has increased in volume, variety and toxicity, endangering public health, and the environment.

The PNMD was launched in 2008 with the main objective to meet the challenges of sustainable management of household and similar waste and to be part of the modernization process of this sector. It aims to

ensure the collection and cleaning of household waste to reach a collection rate 100% in 2030 [1]. Morocco has moved from a policy of burial, closure, and rehabilitation of wild landfills to the development of sorting, recycling and recovery sectors [1].

A landfill is a source of several kinds of environmental nuisance such as: emission of odor, noise, dust, production of biogas and especially landfill leachate (LFL). LFL is defined as the water that percolates through the waste. During this percolation, the liquid is loaded bacteriologically and chemically by mineral and organic substances, this is what is commonly called “landfill juice”. The composition of buried waste, degree of decomposition, humidity rate and age of the discharge are the main parameters influencing the composition of LFL [3–5]. Massively producing this “landfill juice” generates pollution risks of soils, rivers and groundwater. It is, therefore, necessary to collect and treat it before discharge into the natural environment [6–8].

To overcome this problem, several processes drawn from wastewater treatment technology have been applied to treat LFL: aerobic and anaerobic biological degradation, chemical oxidation, chemical precipitation, coagulation-flocculation, activated carbon adsorption and membrane-based processes [9, 10]. Membrane-based technologies have become popular for water treatment [11, 12]. They are increasingly used to treat groundwater, surface water, or wastewater. Thus, many researches currently deal with integrating these membrane processes in the LFL treatment. These processes include microfiltration (MF), ultrafiltration (UF), nanofiltration (NF) and reverse osmosis (RO).

The objective of this work is to study of the performance of coagulation-flocculation (Coag-Floc) and UF, used as pretreatment in the reduction of organic load in the Oum Azza LFL to improve the quality of the pretreated effluent,

and of reduce the frequency of membrane cleaning to improve membrane lifetime and reduce the process cost.

16.2 Part 1. Literature Review

16.2.1 Physicochemical and Biochemical Characteristics of LFL

In light of the information gathered from the literature, LFLs can be categorized into three types: young, intermediate, and old. Generally, the first one is the young LFL. It is less than 5 years old, characterized by a relatively biodegradable organic (biodegradability threshold $BOD_5/COD > 0.3$), and consisting mainly of volatile fatty acids (VFAs). This LFL can be loaded with metals (up to 2 g/L) due to it relatively low pH (<6.5). The second type is the intermediate LFL. These leachates, are between 5 and 10 years old, and generally correspond to the stable methanogenic phase. As the LFL ages and the waste stabilize, the organic load decreases and the VFAs become rare (20–30% of the LFL load). The emergence of these compounds tends to reduce the biodegradability of LFL (BOD_5/COD biodegradability threshold between 0.1 and 0.5). The last one is the stabilized LFL. This LFL is more than 10 years old, characterized by a low organic load, and consisting mainly of humic substances (fulvic and humic acids) of high molecular weight that are refractory to biodegradation ($BOD_5/COD < 0.1$). These stabilized LFLs correspond to the maturation phase of the landfill.

16.2.2 LFL Treatment Processes

16.2.2.1 Physicochemical Processes Overview

Coag-floc is one of the physical–chemical water treatments with a long history and widely used for LFL treatment. Adsorption processes are also used for the treatment and it works in a way that LFL attaches to the surface of the absorption material and is removed from the liquid. Dissolved air flotation (DAF) is another treatment

technique but that is not much used in LFL treatment. Chemical oxidation consists of adding a strong oxidizing agent, such as hydrogen peroxide or ozone that breaks the organic contaminants into carbon dioxide and water.

- **Coag-floc**

Coag-floc is a relatively simple and controllable technique that is widely used in the pretreatment of old and stabilized LFLs before a biological or other physico-chemical process. Several reports on the use of coag-floc for treating LFL are available [13]. A study carried out by Chaouki et al. compared different coagulants (lime ($Ca(OH)_2$, ferric chloride ($FeCl_3$) and alum $Al_2(SO_4)_3$) in terms of efficiency in decolorizing LFL from the city of Casablanca (Morocco). Process efficiency was measured in terms of reduction of COD, trace metals, color and turbidity contents. Lime treatment significantly decreased COD, color, and turbidity (66.25, 98, and 80%, respectively). $FeCl_3$ treatment resulted in reductions in COD (62.5%), color (82%) and turbidity (92.5%). On the other hand, the application of $Al_2(SO_4)_3$ only removed about 11% of the COD and 6% of the turbidity. Regarding the analysis of trace metals, the lime treatment induces a notable reduction in chromium (86%). $FeCl_3$ treatment showed high affinities for Cd, Cu, Ni, Cr and Zn removal [13]. Djeffal et al. [14] reported the effectiveness of three coagulants (ferric chloride, aluminum sulphate and ordinary alum) as LFL treatments from the city of Souk-Ahras in Algeria. A significant reduction in turbidity (99.4%) was obtained with a coagulant dosage of 15%, a stirring speed of 250 rpm and a reaction time of approximately 15 min for ferric chloride. These results agree with the conclusions of Chaouki et al. [15], who used ferric chloride to remove pollutants from LFL. Experimental results demonstrated that the coagulation process reduced color by 80%, turbidity by 90%, COD by

50% and BOD_5 by 99% with the optimal dose of 12 g/L of Fe^{3+}.

- **Activated carbon (AC)**

Adsorption is the most widely applied technology to remove recalcitrant organic compounds from LFL. AC is one of the most influential and convenient adsorbents for removing various types of organic and inorganic contaminants such as ammonia nitrogen and trace metals [16]. Erabée et al. [17] worked on the adsorption of organic compounds in LFL using powdered activated carbon (PAC). These authors explored three methods to modify PAC to improve its adsorption capacity, they activated it with nitric acid (HNO_3) or potassium permanganate ($KMnO_4$) and heating up to 600 °C. The removal rates of SS, NH_3–N, Zn, Cu, and S^{2-} reached 91, 99, 86, 100, and 57%, respectively, after 120 min of contact time when AC-$KMnO_4$ was been applied. Research by Rohers et al. [18] showed the performance of an AC column in the treatment of LFL. Reductions up to 74% of COD, 47% of BOD_5, 93% of color and 90% of ammonia have been achieved. Additionally, the AC adsorption process resulted in the removal of up to 60% of the heavy metal content.

- **Advanced oxidation processes (AOPs)**

Destructive processes are commonly referred to AOPs. The main mechanism of AOP is the production of hydroxyl radicals ($^{\bullet}OH$). These radicals are able to oxidize many complex organic compound. They react effectively with carbon–carbon double bonds and attack aromatic kernels, which are common characteristics of refractory organic compounds [19]. Among POA, the Fenton process is used for pretreatment, post-treatment or complete LFL treatment due to its ability to oxidize refractory organic molecules and its moderate cost, and because it is simple to

operate [19]. This process uses a mixture of ferrous ions (Fe^{2+}) and hydrogen peroxide (H_2O_2). Smaoui et al. tested the use of Fenton oxidation for LFL treatment. Experiments involving H_2O_2 concentrations between 0 and 4 g/L were carried out at pH 3 with 1 g/L of Fe^0. The overall COD and BOD_5 removal efficiencies were 48 and 30%, respectively. The authors demonstrated that increasing the H_2O_2 concentration to 3 g/L H_2O_2 had a positive effect on the elimination of COD, but that increasing the H_2O_2 concentration did not improve the efficiency of the COD removal [20]. In a study realized by Roudi et al. using the optimal operating conditions of 781.25 mg/L Fe^{2+}, pH 3 and a reaction time of 28.03 min, they removed 100% of COD [21]. Makhatova et al. tested the applicability of the Fenton process for the removal of toxic compounds from LFL. They proved that the most favorable concentrations of H_2O_2 and ferric ions for carbon removal were 6660 and 400 mg/L, respectively. As a result, the removal rates of total organic carbon (TOC), total inorganic carbon, total nitrogen (TN), and color were 88.7, 100, 96.5, and 98.2%, respectively [22]. Mahtab et al. studied the effectiveness of the Fenton process in the treatment of LFL. This process was optimized using Response Surface Methodology (RSM) coupled with Core Composite Design (CCD). Several variables were optimized to study the COD removal response. The higher COD removal rate 61%, was obtained at a pH of 3.1, a reaction time of 36 min, a Fe^{2+} dose of 0.04 mol/L and a H_2O_2 dose of 0.075 mol/L [23].

Ozonation is another AOP based on the infusion of ozone into water. Ozone (O_3) is one of the strongest oxidants [24]. Ozonation is a suitable method for the treatment of LFL because it involves the formation of highly reactive oxygen species that can degrade a wide range of organic and inorganic pollutants and all microorganisms [25].

Leszczyński and Jolanta Walery used ozone to remove organic compounds from LFL. COD has been reduced by 46%. The best conditions for the H_2O_2/O_3 process were a H_2O_2/O_3 ratio of 0.8 and an ozone dose of 0.6 g O_3/dm^3 [26]. Kwarciak-Kozłowska showed that applying ozone as a pretreatment of LFL was more effective at alkaline pH 8.5. The TOC removal efficiency was found to be 37% (346 mgO_3/dm^3) after 60 min of ozonation [27].

According to the literature, electro-oxidation (EO) has been shown to be very effective when applied to reduce several pollutants of LFL. However, the performances of the electro-oxidation depend on the experimental conditions used and on the nature of the electrode materials used. Yan et al. [28] studied the electro-oxidation process for LFL treatment. Under optimal conditions (a reaction time of 3 h, a current density of 32.89 mA/cm^2, a liquid circulation speed of 0.46 cm/s and a specific electrode surface of 65.1 m^2/m^3), COD and TOC removal efficiencies respectively achieved are 68 and 40.6%.

Another method that can be used to treat LFL is supercritical water oxidation (ScWO). This AOP takes advantage of the fascinating properties of supercritical water as a reaction medium at temperatures and pressures above those corresponding to the critical point of water (374 °C and 22.1 MPa). The main oxidants used in (ScWO) are pure oxygen and hydrogen peroxide. In 2018, Scandelai et al. [29] tested combined ozonation and ScWO processes for LFL degradation. The combination of ozonation for 30 min and supercritical water oxidation (O3-30°/ScWO) proved to be the most effective in degrading LFL. Under these conditions, high removal efficiencies of color (97%), COD (92%), TOC (79%), nitrite (78%), nitrate (84%), and solids dissolved (96%) and in suspension (94%) were achieved. In addition, this technology also leads to a significant decrease (68%) in electrical conductivity (EC).

16.2.2.2 Biological Treatments

Due to its reliability, simplicity and high cost-effectiveness, biological treatment is commonly used for the removal of the bulk of LFLs containing high concentrations of BOD. Biodegradation is carried out by microorganisms, which can degrade organic compounds into carbon dioxide and sludge under aerobic conditions and into biogas (a mixture mainly comprising CO_2 and CH_4) under anaerobic conditions [30, 31]. Sequential Batch Reactor (SBR) technology is the predominant process applied to LFL. This configuration has simple structure, flexibility and high capacity to remove nutrients from wastewater. Chakraborty et al. [32] used an SBR to treat LFL with a hydraulic retention time (HRT) of 2.5 days and an organic loading rate (OLR) of 1.7 kg COD/m^3.day.The authors obtained 29–67% reduction rates and 54–85% for phosphate and COD, respectively. In addition, the ammonia removal rate ranged between 13 and 35%, while the sulfate removal rate was between 7 and 66%. In another study Hasemi et al. [33] tested the performance of an RDS in LFL treatment. The average removal efficiencies for COD, TN, phosphorus and BOD_5 were 92.45, 73.6, 66.5 and 96% respectively. In general, many studies have proven that algae and fungi are more effective than bacteria in degrading organic matter and different pollutants in LFL [34]. Spina et al. [35], compared different autochthonous fungi based on their ability to decolorize LFL. Five autochthonous fungi, Penicillium brevicompactum, Pseudallescheria boydii, Phanerochaete sanguinea and Flammulina velutipes, were selected for treating LFL. Regarding color degradation, they found that the most effective species were P. brevicompactum and P. boydii, which removed 60% of the color. Moreover, the biotreatability of organic compounds using microorganisms, namely algae, has received much attention due to their effectiveness when applied in

wastewater treatment [36]. Paskuliakova et al. [37], demonstrated the application of microalgae in the treatment of LFL. Their work indicated that Chlamydomonas sp was able to grow on 10% leachate. They reported that 93% of ammonia nitrogen and 54% of nitrate were removed after 70 days of treatment.

Another biological treatment that has been applied for the treatment of LFL is anaerobic digestion (AD), which is carried out by anaerobic bacteria, AD has proven to be an effective treatment of LFL. It eliminates organic compounds due to the low energy requirements of these bacteria, their low sludge production and their generation of high-energy methane. However, these bacteria can be inhibited by toxic pollutants such as total ammoniacal nitrogen, sulfides and heavy metals [38, 39]. In 2021 Cirik et al. [40], demonstrated the performance of an anaerobic baffled reactor (ABR) in the treatment of LFL. In their work, the effect of the level of dilution (5, 10 and 20%) on the removal of pollutants from LFL was investigated. Maximum elimination was noted when the dilution was 20%. The results indicated that the removal efficiencies of dissolved organic carbon (DOC), color, COD, TN, nitrate (NO_3^-) and ammonium (NH_4^+) respectively were approximately 61, 17, 81, 15, 1 and 5%. Jinghuan et al. [41], worked on the anaerobic methanogenese of fresh municipal solid waste leachate, their experiments resulted in a COD reduction rate between 60 and 96% with an optimal production of methane between 0.3 and 0.4 m^3/kg $COD_{eliminated}$. A study carried by Kheradmand et al. [42] focused on the treatment of LFL from municipal landfills using a hybrid system that combines AD and activated sludge, this system achieved a rate 94% COD reduction, the methane production rate was between 0.02–0.04, 0.04–0.07 L/g $COD_{elimination}$ for the first digester, 0.02–0.04 L/g $COD_{elimination}$ for the second digester and the combination

of two digesters respectively gives a methane content that varies between 60 and 63%. Zait et al. [43], in 2018 tested the feasibility of the anaerobic production of biogas from the RO concentrate of LFL very rich in salts. The performance results obtained during this study indicate that the concentrates are not biomethanizable, due to the very high values of EC due to the high value of TDS.

16.2.2.3 Membrane Processes

Over the past two decades, membranes have become increasingly popular in industrial technologies where reliable and reproducible purification or concentration is required. Membrane-base processes are suitable for eliminating and reducing various pollutants such as organic and inorganic matter, microorganisms and traces of metals. According to the literature, there are four types of membrane methods: microfiltration (MF), ultrafiltration (UF) nanofiltration (NF) and reverse osmosis (RO) [24, 44].

- **Reverse osmosis (RO)**

Unlike conventional treatments (lagoon, biological, physico-chemical, etc.), RO offers an ultra-fine filtration system that only lets water molecules through, thus effectively purifying the treated effluent. Indeed, several studies carried out both at the laboratory and industrial scales have shown this process's effectiveness in eliminating the contaminants present in the LFLs. The first LFL treatment units by RO, which date from the early 1980s, comprised two stages, the first stage was equipped with tubular membranes and the second with spiral membranes. The concentrate from the first stage is recirculated on the waste locker while the permeate is sent to the second stage. The focus from the second stage is returned to the top of the treatment and permeate the natural environment. The LFL is prefiltered and then acidified before the RO operations, but there is no other

pretreatment. The transmembrane pressures (TMPs) used are less than 60 bar and the recovery rates are generally between 50 and 70%. The retention of pollutants is very high. The COD and N-NH_4^+ removal efficiencies with these tubular and spiral membranes are over 99%. Fatima et al. [45] tested the performance of RO in LFL treatment, their results showed that RO removes BOD, COD and TDS, with rates of 99.7, 98 and 94% respectively. Additionally, the reduction rates of various trace metals ranged between 76 and 99.99%. A study by Ushikoshi et al. [46] in 2002 focused on treating LFL by RO using the disc-tube type membrane called DT-Module, which made it possible to obtain a removal rate of 99,9% of dioxin. In a similar study, Sabah et al. [47] found 98 and 97% removal rates for COD and BOD_5.

- **Nanofiltration (NF)**

Due to its unique properties between UF and RO, NF has proven itself in removing recalcitrant organic compounds and heavy metals from LFLs [48, 49]. This process can remove particles with a molecular weight greater than 300 Da and inorganic substances through electrostatic interactions between ions and membranes. A study by Istirokhatuna et al. [50] reports that COD, SS and TDS were reduced by 96, 100 and 62%, respectively, using an NF process. Along the same lines, Silva et al. demonstrated that NF technology can be considered as an effective polishing step when applying the Bioreactor Membrane (MBR) process in LFL treatment. Trebouet et al. [51] tested different NF membranes in LFL treatment and showed that the MPT-31 membrane allowed retention rates of 80 and 98% respectively for BOD_5 and COD.

- **Ultrafiltration (UF)**

UF is widely used as a pretreatment before a final stage involving RO in MBR or as a pretreatment at a final stage of RO [24, 52]. Additionally, UF is applied to remove particulates, microorganisms and some dissolved organic matter. As it is a simple and easy to use process, UF is increasingly used in water treatment and purification of different proteins. Abuayyash et al. [53] investigated the feasibility of UF as an additional enhanced treatment step during biological treatment. They showed that UF is very effective in removing SS, nitrates and phosphates, Al and Zn (for which the removal rates were 100, 98, 95, 100 and 82% respectively). However, the porosity of UF limits the passage of SS and large dissolved compounds through the membrane. A study by Pi et al. [54] focused on the treatment of LFL by UF using a membrane with a cut-off threshold of 1 kDa which made it possible to obtain a removal rate of 86% of COD. Ozkaya et al. [55] In (2015), by coupling two MP005 + ZW-UF membranes, obtained a COD reduction rate of 33.54%.

- **Microfiltration (MF)**

MF removes high molecular weight compounds, colloids and SS (0.02–10 µm). For LFL treatment, MF can, for example, be used as a pretreatment for another membrane process (UF, NF or RO) or coupled with chemical treatments. But, it cannot be used alone [56]. The main advantages of the MF process are its low operating costs, reduced number of operations, high separation efficiency and better effluent quality. Pertile et al. [57] evaluated the application of MF as a tertiary LFL treatment from the Rincão landfill (Brazil) and reported that this process removed 43% of organic matter and 63% of BOD_5. Another study, in Poland, carried out by Piatkiewicz et al. [58], report the use of MF as a prefiltration step. No significant

retention rate (COD reduction between 25 and 35%) was achieved.

16.2.2.4 Conclusion

Over the years and around the world, various LFL treatment technologies have been implemented, including physico-chemical methods, membrane technologies, and biological processes.

Conventional treatment is not sufficient to reduce the high concentrations of pollutants present in LFLs. There are many disadvantages, including:

- Difficult denitrification, significant congestion;
- Energy consumption for oxygen supply;
- Significant production of sludge (20% of the treated volume);
- Production of metal hydroxide sludge;
- Sensitivity to load variations;
- Partial dissipation to the atmosphere. Depends on the quality of the biogas.

Membrane processes play an essential role in leachate treatment. Here are some important conclusions about their role:

- Contaminant Removal: Membranes are used to effectively remove contaminants present in leachate, such as heavy metals, organic compounds, nutrients, salts and suspended solids. The membranes allow selective separation, helping to purify the liquid.
- Reduction of environmental impact: Membrane processes help reduce the environmental impact of leachate by minimizing the release of harmful substances into groundwater and waterways. This helps prevent environmental pollution.
- Increased efficiency: Membranes offer an effective solution for concentrating leachate contaminants, which

reduces the amount of waste to be further treated. This can reduce overall treatment costs.
- Improved resource management: Membrane processes can also contribute to the recovery of useful resources, such as clean water and recoverable materials, from treated leachate. This recovery can be economically and environmentally beneficial.
- Operational flexibility: Membrane systems are adaptable and can be integrated into larger treatment configurations to fit the specific needs of a leachate treatment facility.
- Maintenance requirements: It is important to note that membrane processes require regular maintenance to maintain their effectiveness, as membranes can become clogged over time. However, many technological advances have been made to minimize this problem.

16.3 Part II. Pretreatment of LFLs by Coag-Floc and UF

16.3.1 Study Zone (Landfill of Oum Azza)

The project of the Oum Azza center for the treatment and recovery of waste was born following the will of his majesty King Mohammed VI, to eradicate uncontrolled dumping which constitutes a threat to the environment and public health. The Oum Azza waste recovery center, inaugurated by His Majesty, meets the objectives of the National Household Waste Program (PNDM) to improve waste treatment conditions in the country. This new center has eliminated many wild dumps (Akrach, Oualaja, Aïn Atiq, etc.), The Oum Azza landfill is a big step towards a greener Morocco. Although it receives more than 850,000 tons of rubbish per year, it uses environmentally friendly technology and has an area of 110 hectares. One third of which is devoted to sorting and two thirds to landfilling. It started in 2007, even if waste sorting only started in 2011. The Oum Azza landfill is therefore a revolution in Morocco as a sorting center. Functional and respectful of the environment, it is also a center for the recovery and optimization of treatment conditions for the waste produced. The Oum Azza controlled landfill site is located in the town of Oum Azza, 20 km from Rabat. The landfill is supplied by 3 Transfer Centers in the prefectures of Rabat, Salé and Temara. The daily quantity of LFL produced in the landfill is 660 m^3/d. Figure 16.1 shows the location of the landfill.

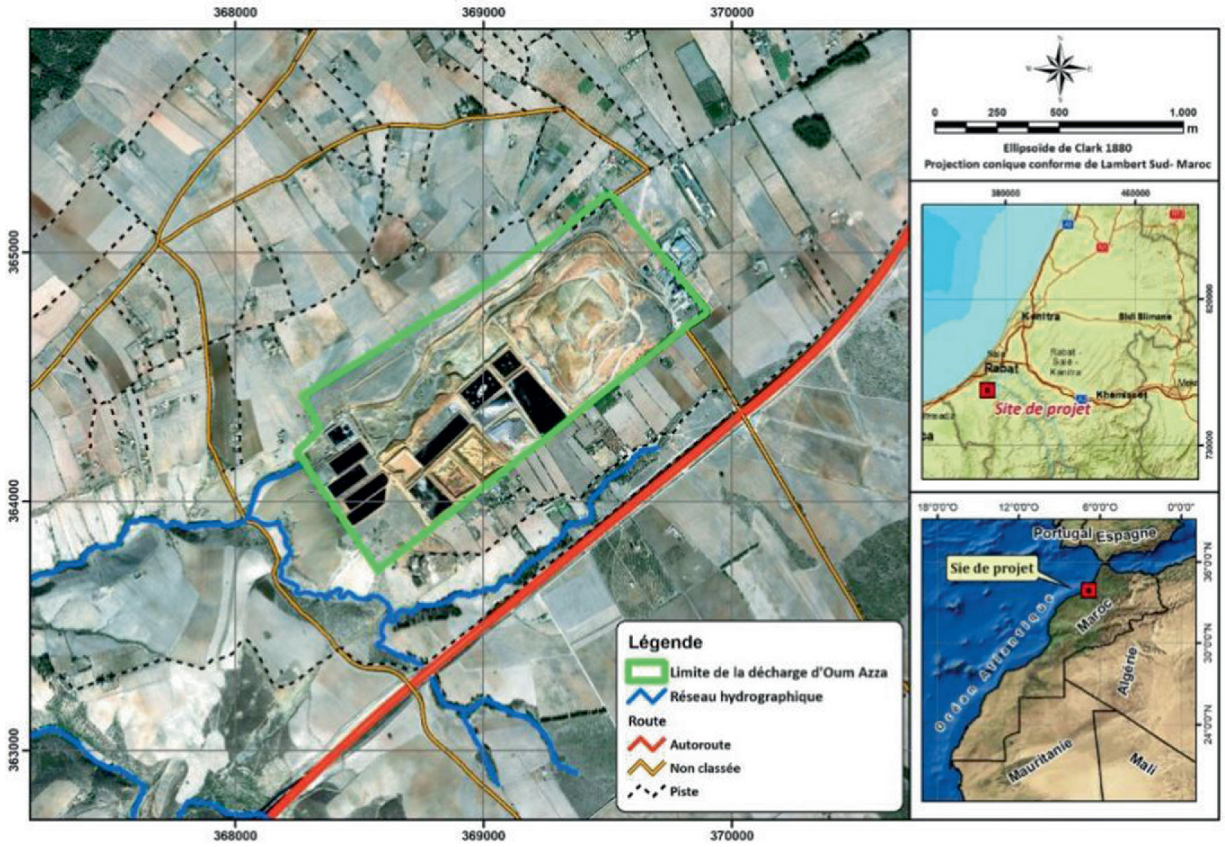


Fig. 16.1 Satellite view of the location of the Oum Azza landfill

Burying this waste releases harmful elements: biogas, produced by its fermentation, and LFL, a liquid rich in organic and inorganic matter that cannot be immediately released into nature. The daily quantity of waste received by the burying center Technical of Oum Azza goes through several stages according to the diagram in Fig. 16.2.

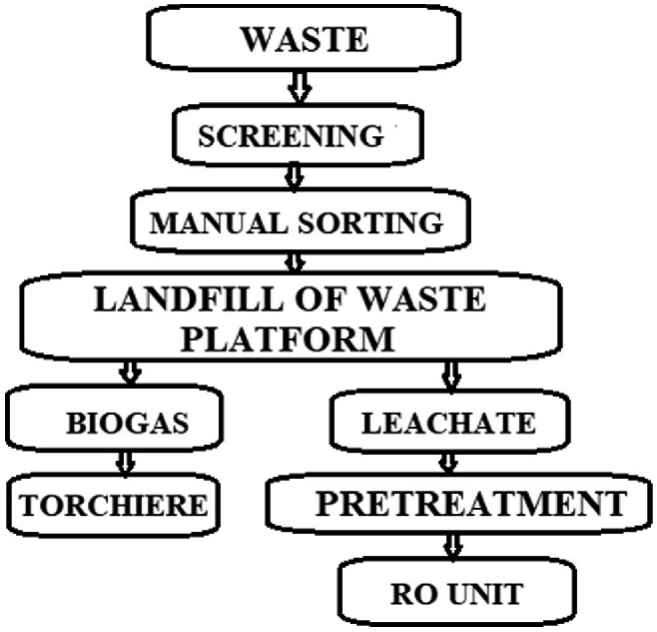


Fig. 16.2 Operating stages of the Oum AZZA landfill plant

The Oum Azza LFL treatment is carried done in two stages: a biological pretreatment which consists of an aeration basin and an anoxic basin followed by a pocket filter, and RO unit in the second stage. Before being treated, the LFL is stored in nine basins with a surface area of one hectare. When the LFL is stored in the basins, it is aerated by sprinklers to accelerate its evaporation. While a quantity of it is treated according to the following process (Fig. 16.3).

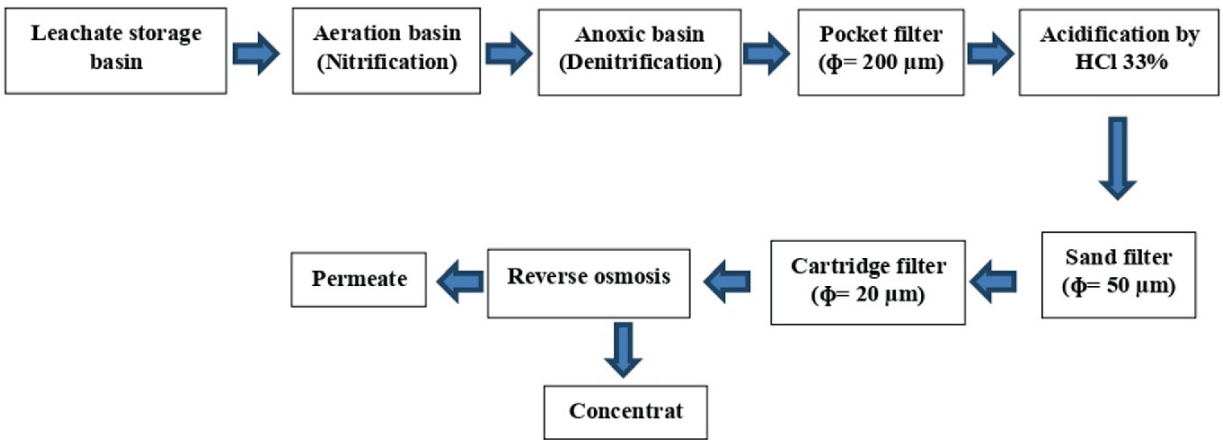


Fig. 16.3 LFL treatment plant process in Oum Azza

The Oum Azza site is a Controlled Technical Landfill Center (TLC) which receives tonnages large of household waste daily and which generates a large quantity of LFL. The LFL produced has a very high EC and very high values of SS, COD and BOD_5 pollution indicators. LFL treatment performance monitoring at the Oum Azza station, which includes biological pretreatment followed by by RO, shows a recovery rate varying between 40 and 50%, and frequent shutdowns of the installation due to membrane fouling. It is to remedy this problem that this work was launched, with the aim of improving the pretreatment before RO by proposing two methods (i) coag-floc and (ii) UF.

16.3.2 Materials and Methods

16.3.2.1 Coagulation-Flocculation

The Coag-Floc experiments were carried out using a propeller flocculator. The test equipment comprises six stirrers (stainless steel blades) whose rotation speed can vary between 0 and 200 rpm. This device allows for simultaneous agitation of the liquid contained in a series of beakers, each filled with 500 mL of LFL. Different concentrations of the chosen coagulant were added to the LFL. The mixture is agitated rapidly at 150 rpm for 3 min, which is the coagulation phase. The speed is then reduced to 45 rpm for 20 min, which is the flocculation phase. After 120 min of settling, the supernatant is collected for analyzing parameters such as pH, EC, COD, BOD_5, and SS. The optimal coagulant concentration was determined based on the reduction results of SS and COD.

Two coagulants were compared: ferric chloride with concentrations ranging from 0.5 to 8.5 g/L, and aluminum sulfate with concentrations ranging from 1 to 14 g/L. The flocculant used in our study is an anionic polymer (anionic polyelectrolyte based on anionic polyacrylamide) used as a coagulant aid, which promotes agglomeration and the

formation of flocs, with concentrations ranging from 0 to 0.2 g/L.

16.3.2.2 UF

Experiments are performed on a UF laboratory pilot supplied by the French company TIA (Technologies Industrielles Appliquées). It consists of a feeding tray with a capacity of 50 L and two pumps: one for circulation and the other for filtration (Fig. 16.4). The tangential speed of recirculation is in the range 0.5–6 m/s. The TMP varies from 0 to 10 bar.

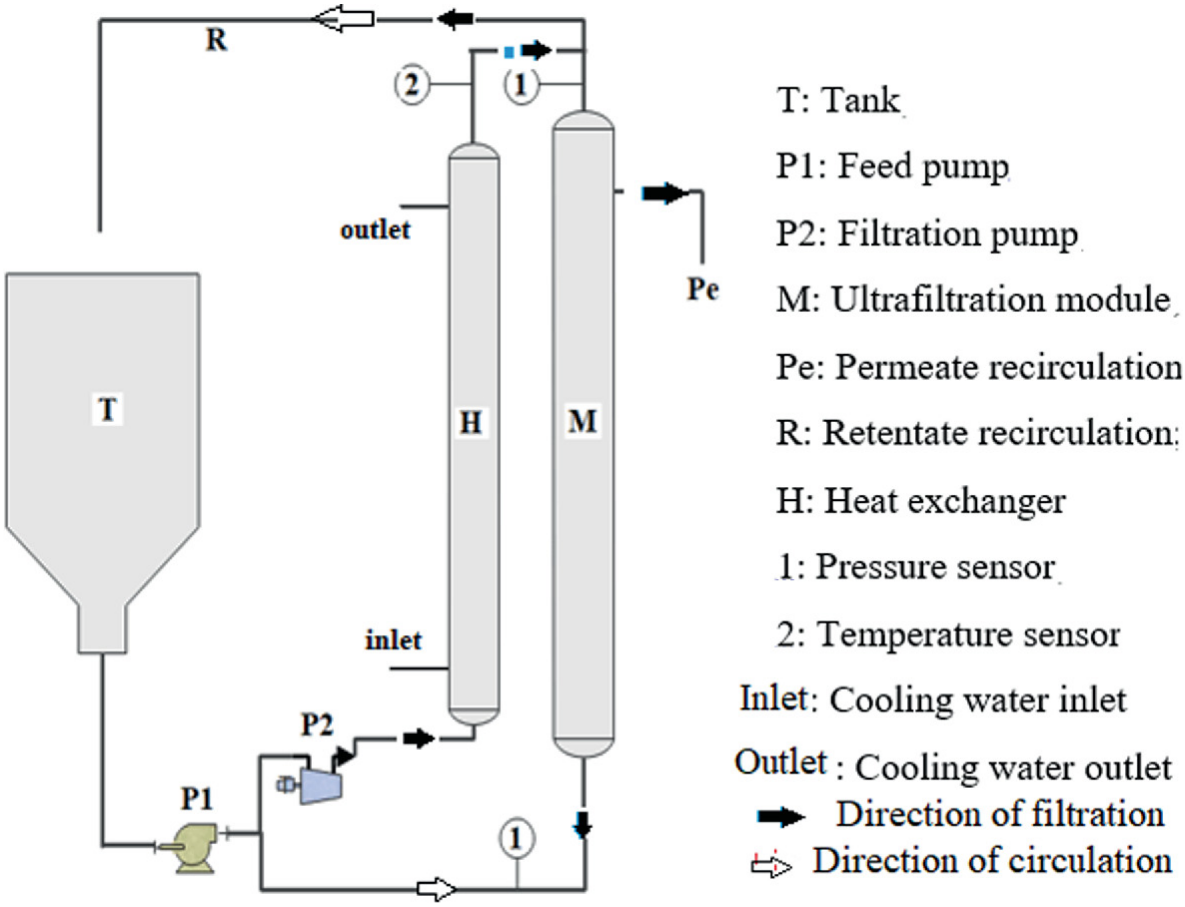


Fig. 16.4 Diagram of the UF pilot plant

Table 16.1 gives the main characteristics of the membranes used. After UF tests, the membranes are

cleaned with alkaline and acidic cleaning solutions according to the manufacturer's recommendations.

Table 16.1 Characteristics of the membranes

Characteristics	UF 1	UF 2	UF 3
Pore size (µm)	0.02	0.05	0.1
Nature	Ceramic	Ceramic	Ceramic
Surface (m^2)	0.35	0.35	0.24
Maximum pressure (bar)	10	10	10
Maximum temperature (°C)	100	100	100
pH range	3–11	3–11	3–11

16.3.2.3 Analytical Techniques Implemented

The physico-chemical characterization of the LFLs obtained after each test is based on the analysis of the physico-chemical and organic parameters measured in the laboratory, such as: T, pH, EC, COD, BOD_5 and SS.

COD (NF T90-101) and BOD_5 (NF T90-103) measurements are made according to AFNOR standards (1994). Determination of SS (NF T90-105-2) is carried out according to the AFNOR standards (1997).

The temperature is measured using a graduated thermometer from 0 to 100 °C equipped with a WTW Multifine F/SET-3 probe. The EC of the analyzed samples is measured using a WTW Multifine F/SET-3 probe and is expressed in µS/cm.

16.3.3 Results and Discussion

16.3.3.1 Characterization of the Raw LFL

The main LFL analysis are presented in Table 16.2. The sampling campaign was achieved during January 2019. As shown in Table 16.2, all the pollution indicator contents exceed the discharge limit values [59].

Table 16.2 Physico-chemical and organic characterization of LFL and Moroccan discharge limit values

Settings	Value	Domestic discharge limit values [59]
pH	8.26 ± 0.2	5.5–9.5
Temperature (°C)	18.6	30
EC (mS/cm)	26 ± 3	2.7
COD (mg d'O_2/L)	8212 ± 2000	500
BOD_5 (mg d'O_2/L)	4500 ± 1000	100
SS (mg/L)	3840 ± 500	100

These analyses show a strong organic pollution which results in high load of BOD_5, COD, SS and a high EC. The average temperature of 18.6 °C is fairly characteristic of the month during which the samples were taken and this recorded value is much lower than the value recommended for the (general limit value of discharge). Oum Azza LFL potential hydrogen (pH) is 8.5, which is in the range of Moroccan release limits. This slight basic character has been emphasized in several works on Moroccan landfills [60–63]. One of the most features of this LFL is its high EC which is around 26 ± 3 mS/cm. This value exceeds the standards of treated wastewater. Similar values were mentioned by different researchers working on LFL of Zalaghi et al. [61] and Khalil et al. [62] in Morocco. Due to this high EC of Ouma Azza LFL, decision-makers and stakeholders have opted for RO treatment. Indeed, it is the only technology capable of providing a very significant reduction in TDS; this is why RO is currently the main seawater desalination process worldwide. However, the efficiency of RO in the treatment of LFL is largely dependent on the performance of the pretreatment. If the pretreatment is not very refined, RO membranes will be quickly fouled, and flows will drop, which leads to the implementation of RO membrane washing sequence with high frequency. For COD, the value obtained, $8212 \pm$

2000 mg O_2/L, exceeds largely the limit value of discharges (500 mg O_2/L). This high value indicates a very high organic load and shows that these effluents are in the reducing conditions (oxygen drop). Similar values of COD are reported in different studies on Moroccan LFL. [61, 63, 64]. For BOD_5, the content measured is 4500 ± 1000 mg d'O_2/L. This value is very high compared to the limit value of discharges (100 mg d'O_2/L). Finally, analysis of SS reveals a content of 3840 ± 500 mg/L. This concentration is slightly higher than the limit value fixed by Moroccan Liquid Discharges Standards (MLDS, 2018).

16.3.3.2 Pretreatment of LFL by Coag-Floc

Although the coagulant dose is the most influential parameter on coagulation efficiency, it is also responsible for the presence of residual aluminum or iron in the treated water. For this reason, it is necessary to determine the limit concentration to avoid the saturation of the water in Al or Iron (III).

Indeed, the coagulant overdose can cause the opposite effect, as positive charges can be deposited on the surface of the particles, which causes the inversion of the Zeta potential, floc re-stabilization and particle redispersion [65]. During the over-injection of the coagulant, the pH drops and leads to the formation of more soluble species than the $Al(OH)_3$ or $Fe(OH)_3$ precipitate.

Effect on COD and SS

Figures 16.5 and 16.6 give, for the two coagulants tested, the variation in the content and reduction rate of SS and COD as a function of the initial concentration of the coagulant.

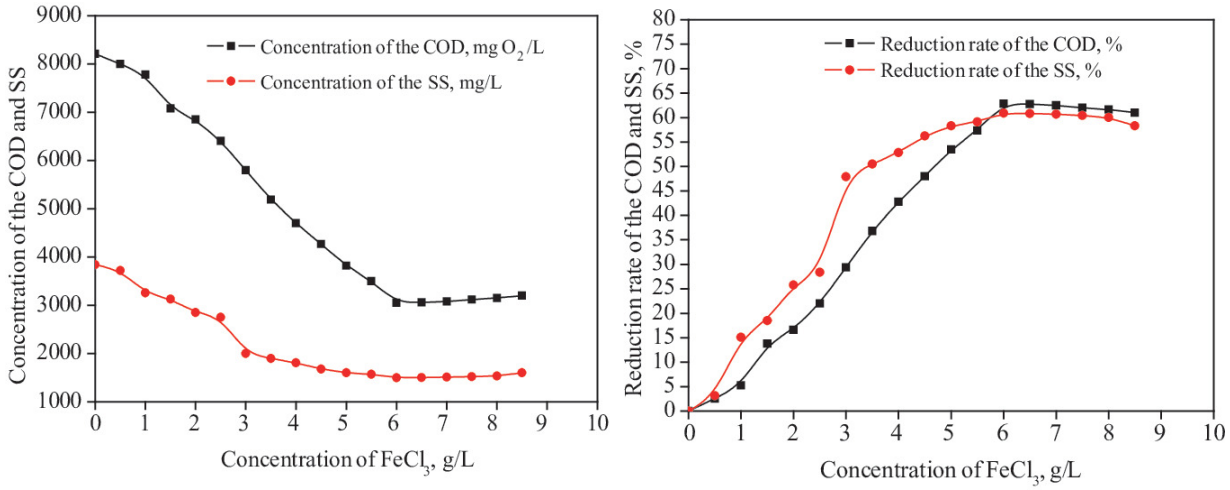


Fig. 16.5 Evolution of the content and reduction rate of SS and COD as a function of the dose of $FeCl_3$

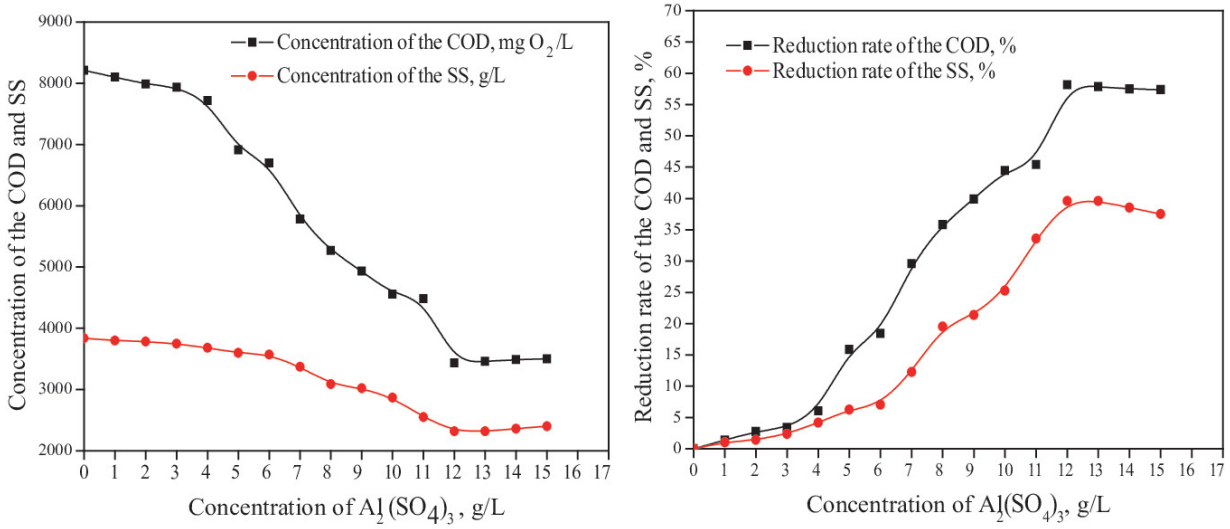


Fig. 16.6 Evolution of the content and reduction rate of SS and COD as a function of the dose of $Al_2(SO_4)_3$

From Figs. 16.5 and 16.6:

- For the two coagulants, the value of COD and SS in the treated LFL decreases with the increase in the concentration of the coagulant, until reaching a minimum value. Then a slight increase was observed.
- The reduction rates obtained by ferric chloride are better than those obtained with aluminum sulphate and with doses which are low.

– For ferric chloride and for an initial concentration of 6 g/L, the minimum COD value reached is 3052 mg of O_2/L with an elimination of 62.83% and the minimum SS value reached is 1500 mg/L with a reduction rate of 60.94%.
– For aluminum sulphate and for an initial coagulant content of 12 g/L, the COD value decreases with the addition of $Al_2(SO_4)_3$, down to a minimum value of 3434 mg of O_2/L, which corresponds to a removal efficiency of 58.18%. The SS decreases with the increase in the concentration of the coagulant until it reaches a minimum value of 2320 mg/L which corresponds to a reduction rate of 39.58%.

According to Fletcher et al. [66], COD presents almost everything that is likely to consume oxygen in water, for example mineral salts and organic compounds. The introduction of coagulant particles which trap these materials, consequently reduces the COD, for the optimal doses of 6 g/L of $FeCl_3$ and 12 g/L of $Al_2(SO_4)_3$, almost all of the colloids are trapped, which explains the low COD value. For coagulant concentrations greater than 6 g/L of ferric chloride or 12 g/L of aluminum sulphate, it can be seen that the COD value increases with the increase in coagulant concentration. This is due to the fact that the overdose of the particles leads to an oxygen demand which remains low compared to that of the initial materials. This also confirms that the optimal dose of the coagulant is indeed around 6 g/L for ferric chloride and 12 g/L for aluminum sulphate.

The elimination of SS can be explained by the phenomenon of coagulation which acts in such a way as to neutralize the negative charges of SS, in order to promote their agglomeration and allow their decantation. The increase in SS is explained by the coagulant overdose which causes the re-stabilization of the colloidal particles

by reducing the availability of their sites which prevents the formation of inter-particle bridges [67].

Effect on pH and EC

Figure 16.7 gives, for the two coagulants, the variation in pH and EC as a function of the initial concentration of the coagulant.

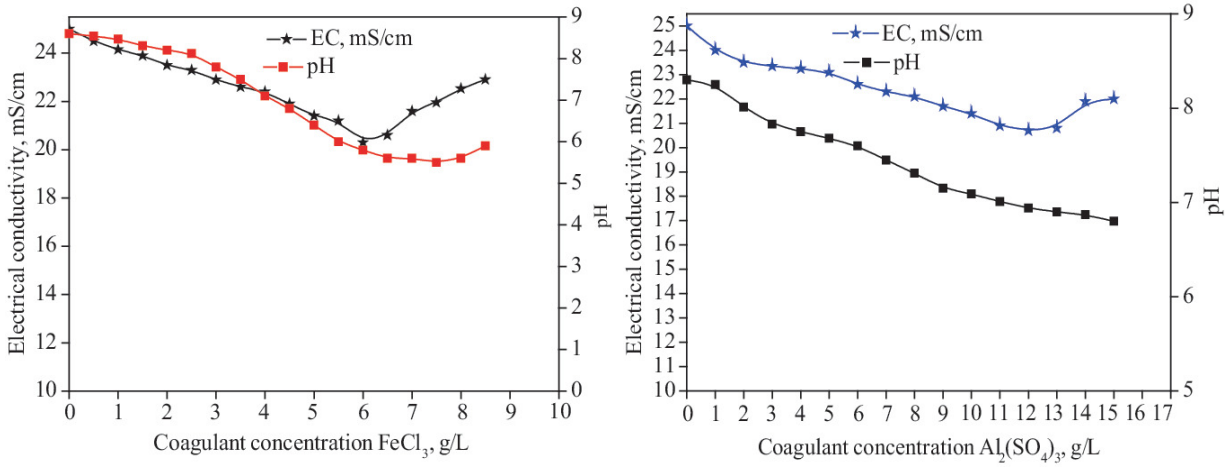


Fig. 16.7 Variation of EC and pH as a function of $FeCl_3$ and $Al_2(SO_4)_3$ dose

The pH of the collected LFL is alkaline due to the negative charges of the colloids. After the treatment, the pH values gradually decrease with increasing coagulant concentration added, reaching a value of 5.8 at the concentration of 6 g/L of ferric chloride and 6.94 at the concentration of 12 g/L aluminum sulfate. The results obtained confirm the bibliographic data [68–70] concerning the addition of the coagulant which leads to the formation of metal hydroxide with the release of a certain acidity (hydrolysis) which explains the decrease in the pH value. The pH value continues to decrease with increasing coagulant concentration. The particles of the coagulant also bring positive charges which contribute to the drop in pH. Arriving at a certain concentration of coagulant, the pH drops slowly: 5.8 at a concentration of 6 g/L of ferric chloride and 6.94 at a concentration of 12 g/L of aluminum

sulphate are considered to be the best values of pH. These values are close to the neutralization pH, which shows that at this concentration, the water has a better quality and also confirms the value of the optimal dose of the coagulant. For the EC of LFL treated, it decreases slightly with the increase in the concentration of the coagulant. At 12 g/L, the EC varies around 20.7 mS/cm with a reduction of 17.2% with aluminum sulphate and an EC close to 20.3 mS/cm with an elimination efficiency of 18.8% for ferric chloride. These results are explained by the precipitation of ionized dissolved minerals following the addition of the coagulant [71]. The increase in EC observed after this dose is explained by the presence of aluminum ions (Al^{3+}) or iron (Fe^{3+}) dissolved in LFL in excess resulting from the coagulant overdose.

Effect of Flocculant Concentration

Starting from the optimal concentration of the coagulant previously determined, a series of tests with different concentrations of flocculant were carried out. The flocculant used in our study is an anionic polymer (anionic polyelectrolyte based on anionic polyacrylamide) used as a coagulant aid, which promotes agglomeration and the formation of flocs. This series of tests was carried out at a temperature of 24 °C to approach the optimum concentration of the flocculant with a concentration fixed at 6 g/L for ferric chloride and 12 g/L for aluminum sulphate. Figures 16.8 and 16.9 represent the variations in the concentration and reduction rate of COD and SS of LFL coagulated with $FeCl_3$ and $Al_2(SO_4)_3$ as a function of the dose of the flocculant.

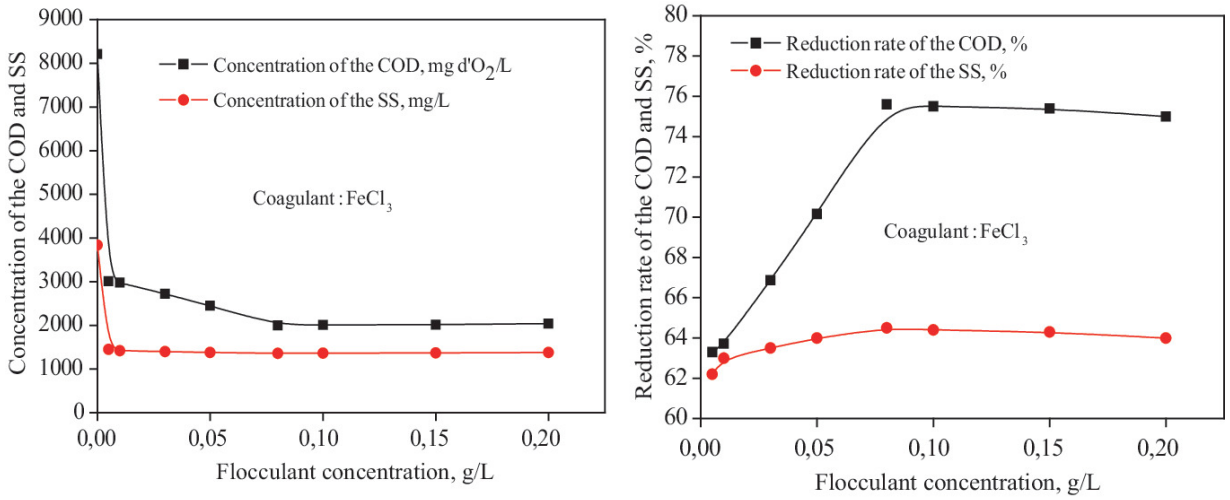


Fig. 16.8 Variation in the concentration and reduction rate of COD and SS as a function of the dose of flocculant

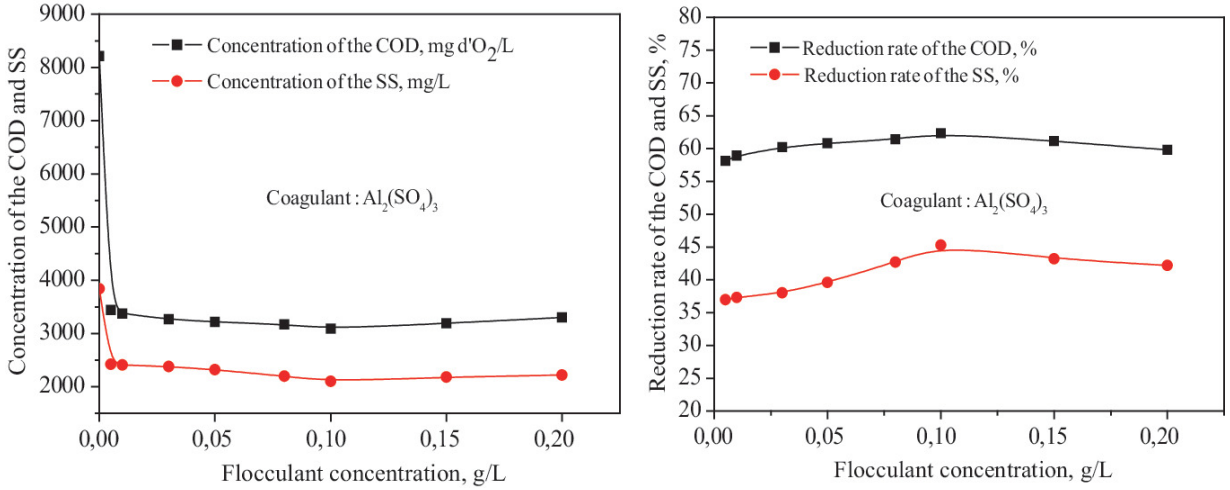


Fig. 16.9 Variation in the concentration and reduction rate of COD and SS as a function of the dose of flocculant

For ferric chloride as a coagulant, the SS value decreases with increasing concentration of added flocculant, reaching a minimum value of 1360 mg/L, and an elimination of 64.5% for a concentration of 0.08 g/L in flocculant.

For aluminum sulphate as a the coagulant, the SS value decreases with increasing flocculant concentration, reaching a minimum value of 2100 mg/L, and an elimination efficiency of 45.3%, for a concentration of 0.1 flocculant. The optimal dose of the flocculant is 0.08 g/L

with $FeCl_3$ and 0.1 g/L with $Al_2(SO_4)_3$. Indeed, the micro-flocs formed by agglomeration of the particles previously discharged by the effect of the added coagulant are more reinforced by the macromolecules of the added flocculant. In this so-called flocculation stage, these micro-flocs aggregate forming flakes which therefore hold more particles in suspension and consequently become larger, which facilitates their decantation [72, 73]. Beyond the optimal flocculant concentration, the SS value increases with increasing flocculant dose for both coagulants.

With regard to the COD, the results show that its value decreases with increasing flocculant dose until reaching a minimum value of 2000 mg d'O_2/L and a removal efficiency of 75.6% for a concentration of 0.08 g/L of ferric chloride, and a value of 3092 mg d'O_2/L with a removal efficiency of 62.34% for a concentration of 12 g/L of aluminum sulphate. However, beyond the optimum flocculant concentration, the COD value increases with increasing flocculant dose.

COD represents all substances that consume oxygen in water. The introduction of the floculant particles promotes the agglomeration of micro-flocs formed by the aggregation of the particles previously neutralized by the added coagulant. these micro-flocs aggregate forming flakes which therefore hold more consumable particles from the oxygen and consequently become more voluminous, which facilitates their decantation [73]. And consequently, a reduction in the COD. For concentrations higher than that of the optimal concentration, the increase in COD is due to the overdose of the flocculant in the leaching water.

With regard to the effect of the flocculant on the pH and the EC, according to Fig. 16.10 we notice that the addition of anionic flocculant has no significant influence on these two parameters.

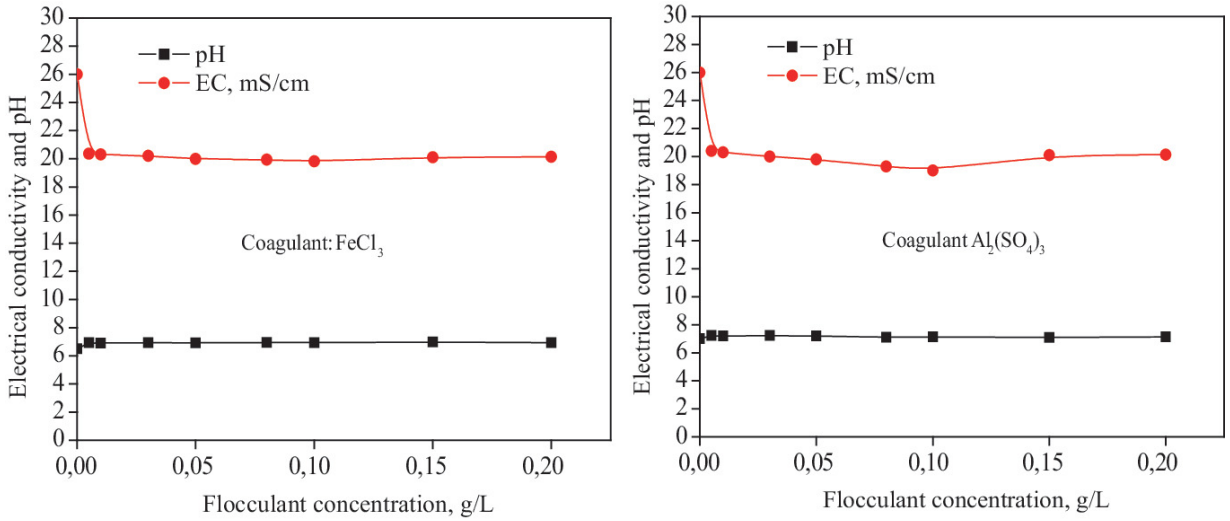


Fig. 16.10 Variation of pH and EC of coagulated LFL as a function of flocculant dose

Decanting Time Effect

The settling time of the coagulated particles depends on the type of particles, the diameter of the particles and their specific surface, it is therefore impossible for certain particles, in particular colloids and particles with a very high specific surface, to settle naturally. To obtain the decantation of these particles, it would be necessary to assemble a very large number of colloids into aggregates of at least 10–100 µm. To determine the optimal settling time, all the parameters were kept constant (concentration of coagulant and flocculant).

The effect of settling time on COD and SS removal for both coagulants is shown in Figs. 16.11 and 16.12.

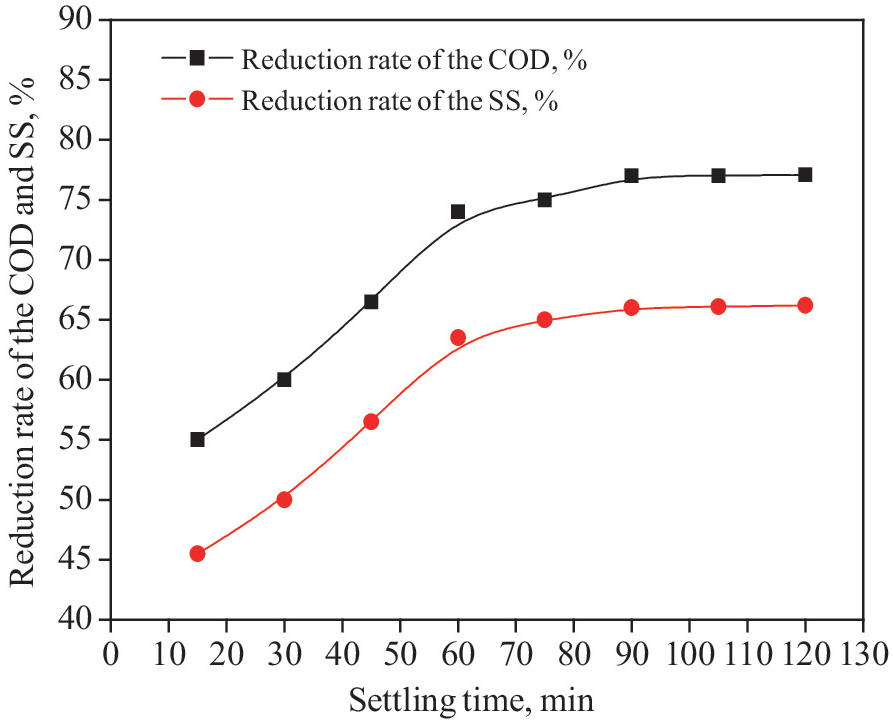


Fig. 16.11 Variation in COD and SS removal rate of LFL coagulated by $FeCl_3$ as a function of settling time

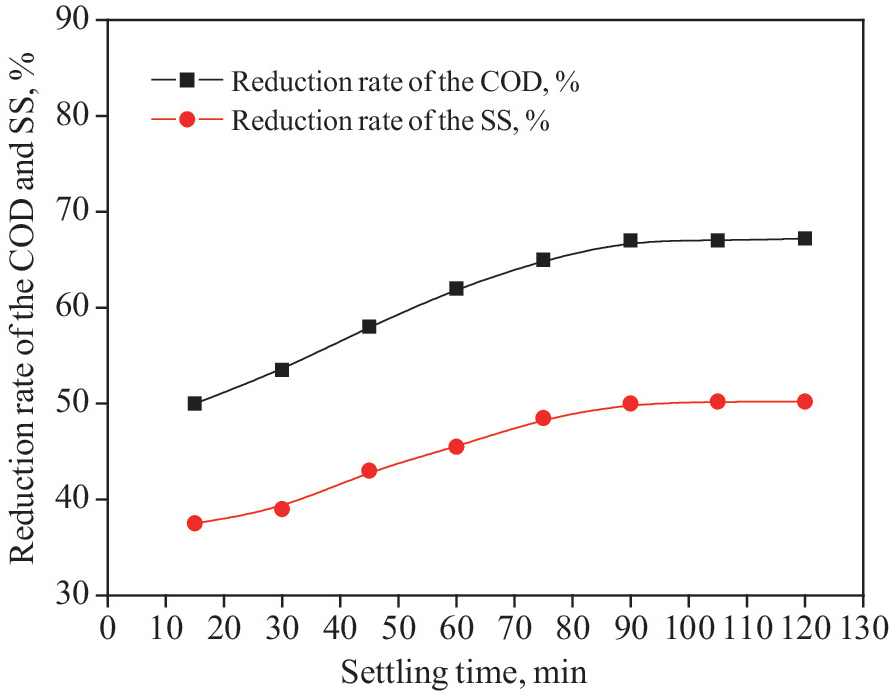


Fig. 16.12 Variation in COD and SS removal rate of LFL coagulated by $Al_2(SO_4)_3$ as a function of settling time

The results obtained prove that the settling time plays an important role in the elimination of the polluting load of LFL. An improvement in treatment efficiency was observed with increasing settling time, as shown in Figs. 16.11 and 16.12.

The reduction rate increases with the settling time applied up to a value equal to 90 min, where the process seems to reach its best efficiency; the abatement efficiency then reaches the following values: 77 and 66% for COD and SS respectively for $FeCl_3$ and 67%, 50% respectively for COD and SS for $Al_2(SO_4)_3$.

If the pollution removal rates are high, this is due to the production of an optimal amount of bubbles, which offers a high probability of encounter between the bubbles and the aggregates formed during flocculation. The flocculated particles will then be carried away with the bubbles in their upward movement. Beyond the optimum time, the solid–liquid separation process is slowed down and stabilized. This can be explained by the fact that the suspended particles likely to be attached to the bubbles are of different sizes; the larger ones are easily eliminated, while the small-sized particles remain in suspension, since they have difficulty adhering to the bubbles. The process reaches its limits under these conditions [74].

Conclusion

The coag-floc process has revealed various mechanisms, often complex and highly dependent on the pH of the LFL to be treated. In addition, the optimization of this step is required for better reduction of pollutants, especially organic ones.

The optimization and adjustment of physico-chemical parameters can lead to better elimination of suspended matter, colloidal matter and macromolecules. Table 16.3 summarizes, for the two coagulants tested, all of the

optimal conditions and results of the coag-floc conducted on Oum Azza LFL.

Table 16.3 COD and SS reduction rate according to the optimal values of the parameters studied

Parameters studied	Optimal values		COD reduction rate (%)		SS reduction rate (%)	
	$FeCl_3$	$Al_2(SO_4)_3$	$FeCl_3$ (%)	$Al_2(SO_4)_3$ (%)	$FeCl_3$ (%)	$Al_2(SO_4)_3$ (%)
Coagulant concentration (g/L)	6	12	62.83	58.18	60.94	39.58
Flocculant concentration (g/L)	0.08	0.1	75.6	62.34	64.5	45.3
Settling time	90	90	77	67	66	50

According to the experimental results and the analysis of the specialized articles, the treatment by coag-floc with ferric chloride is more effective than that with aluminum sulphate.

16.3.3.3 Pretreatment of LFL by UF

Membrane Permeability

Figure 16.13 gives the variation of pure water flux as a function of transmembrane pressure (TMP) for the three membranes. The circulation velocity (CV) and temperature are respectively fixed at 0.5 m/s and 25 °C. As shown in Fig. 16.13, the permeate flow varies linearly with the TMP in accordance with Darcy's law for the three membranes. On the other hand, this flux increases linearly with the pore size of the membranes within the range of pressure studied. The obtained permeabilities of the three membranes are presented in Table 16.4.

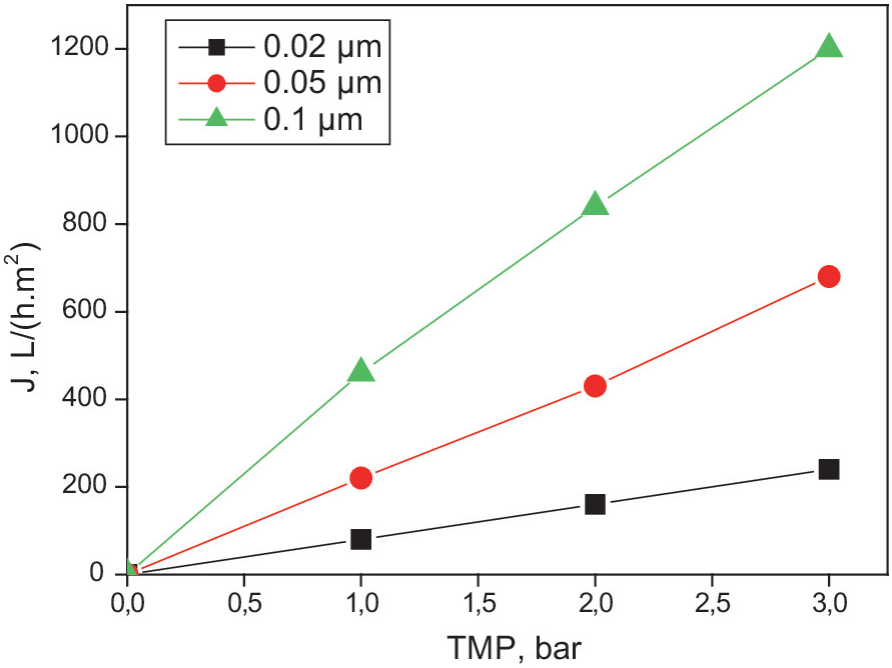


Fig. 16.13 Evolution of the water permeates flux as a function of TMP at 25 °C

Table 16.4 Permeabilities and R-squared of the three UF membranes

Membranes (µm)	Permeability (L/h m^2)	R squared
UF 0.02	80	1
UF 0.05	225	0.99
UF 0.1	398	0.99

Influence of Pore Size

Figure 16.14 illustrates the evolution of flux and EC of the permeate and retention of COD and SS for as a function of pore size of the membranes. The CV of 0.5 m/s is considered as the minimum value that can be issued by the laboratory pilot. The TMP is 5 bar, which is the medium pressure, and a temperature is maintained at 26 °C. Results show that the permeate flux increases with the membrane pore size. The highest flux is obtained with the membrane that has the greatest pore size (100 L/h m^2, 0.1 µm), the lowest flux is obtained with the membrane that has the lowest pore size (46.79 L/h m^2, 0.02 µm) and

for the third membrane, with intermediate pore size (0.05 µm), the obtained flux is 27 L/h m^2. These fluxes are lower than those obtained with pure water (Fig. 16.13) as result of concentration polarization by the retained compounds and membrane fouling. In general, membranes with small pore sizes are fouled quickly due to the particles having molecular weights between 500 and 1000 Da, which are widely present in LFL [75]. On the other hand, pore size of UF membrane has no influence on the EC of the ultrafiltrate, only a TDS retention of 6% is exhibited by the membrane with the lowest pore size (0.02 µm). Indeed, TDS is distributed equally between permeate and retentate of UF, which means that EC will be practically the same in permeate and retentate.

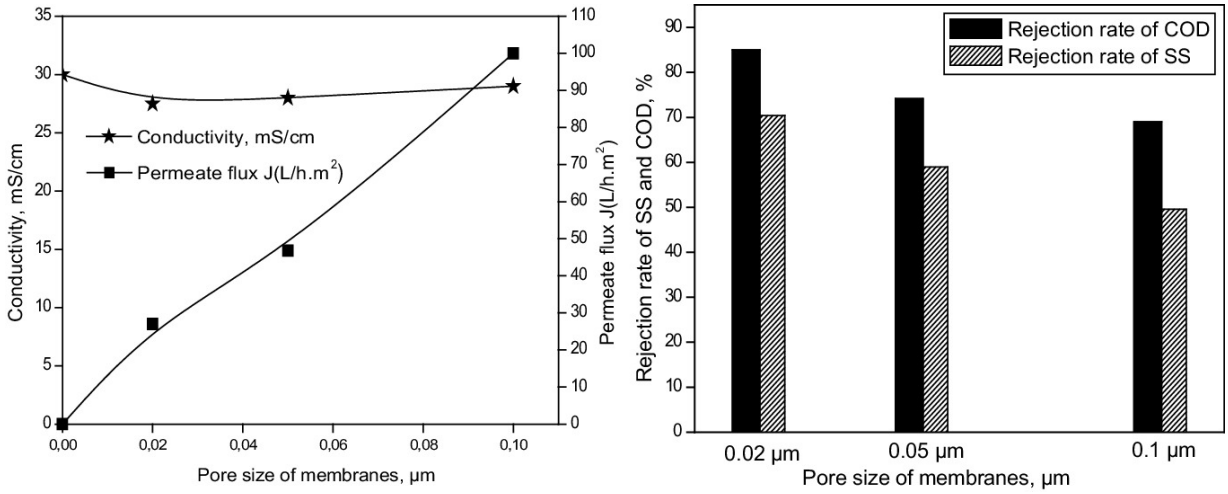


Fig. 16.14 Variation of flux, EC and retention of SS and COD as a function of pore size

Retention of COD and SS decreases with increasing pore size of membranes. The highest retentions of COD (85%) and SS (70.4%) are obtained with membrane having the lowest pore size (0.02 µm), intermediate retention values of COD (74.14%) and SS (58.96%) are exhibited by the membrane with intermediate pore size (0.05 µm) and the lowest retentions of COD (69%) and SS (49.53%) are obtained with the membrane of the highest pore size

(0.1 µm). Particle size influences the removal efficiency as membranes with smaller pore sizes show better retentions.

Influence of TMP and CV

Influence of TMP

For the three membranes and for a CV of 0.5 m/s, Fig. 16.15 gives, the variations of permeate flux, retention of COD and SS, EC and pH as a function of TMP. The effect on flux is studied in order to find the optimum TMP and control the membrane fouling.

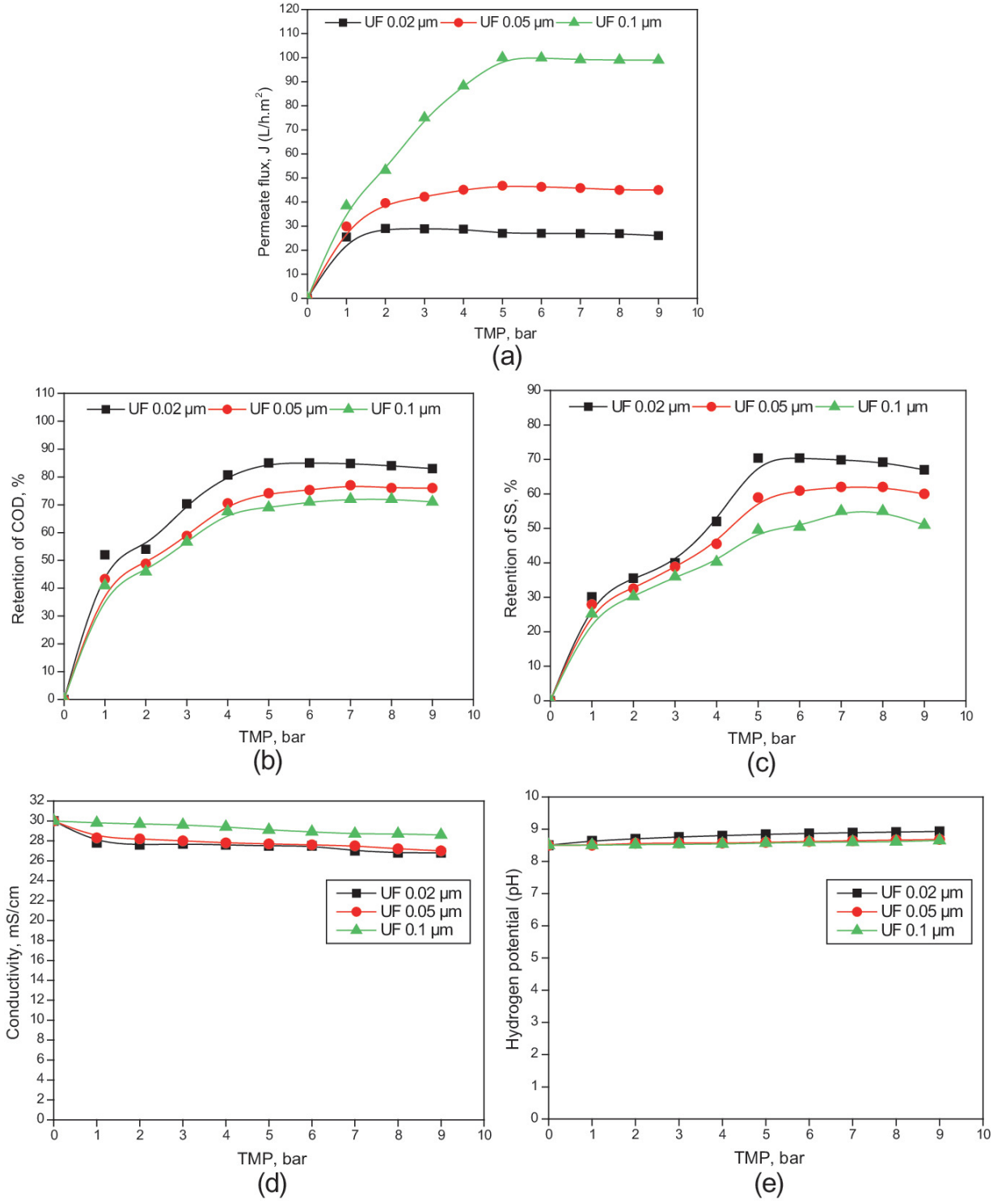


Fig. 16.15 Variations of permeate flux (**a**), COD (**b**) and SS (**c**) retention, EC (**d**) and pH (**e**) as a function of pressure (CV = 0.5 m/s)

Following to the results obtained in Fig. 16.15a, the permeate flux of the three membranes increases with TPM

and tends towards a plateau which indicates the limiting flux. This limiting flux value becomes independent of TMP when the fouling layer is formed.

For the two membranes of UF 0.05 and 0.1 µm, the limiting flux is observed at an applied TMP of 5 bar. On the other hand, with the 0.02 µm UF membrane, the limiting flux, which is the lower, one (29 L/h m^2) is observed at a TMP of 2 bar. For the other membranes 0.05 and 0.1 µm, the limiting fluxes reach respectively 46.79 and 100 L/h m^2.

As the TMP increases beyond a certain threshold value, the flux becomes constant and independent of pressure. The optimum TMP is preferably 80% of the limiting pressure: $P_{opt} = 1.2$ bar for the UF membrane (0.02 µm), $P_{opt} = 1.6$ bar for (0.05 µm) and $P_{opt} = 2.5$ bar for (0.1 µm).

Concerning the influence of TMP on retention of COD and SS (Fig. 16.15b, c), it appears that initially retention increases with TMP and then tends towards a plateau, that means the limit retention.

As the TMP increases, more pollutants accumulate on the membrane surface, forming a gel layer and clogging the pores, which increase filtration resistance due to higher compression of pollutants. Depending on the polarization model of the gel, the existence of limiting flux is related to concentration polarization that occurs when the feed solution containing suspended and soluble solids (colloids) passes through the membrane. The formation of a viscous, gelatinous layer is responsible for additional resistance to the permeate flux in addition to that of the membrane [76] which explain the increase in retention. The highest retentions of COD and SS are obtained with the membrane having the smallest porosity, since with these membranes the concentration polarization conditions are quickly reached. This result is in accordance with the convective

transport mechanism that governs transfer through UF membranes.

As shown in Fig. 16.15d, TMP has no impact on EC of the permeate since the UF membranes used are porous and allow the mineral salts to pass through the membranes. EC is almost the same and have the same value in the feed, permeate and retentate.

Figure 16.15e shows that the pH of permeate remains almost stable with the increase of TMP [77, 78].

Influence of CV on the Permeate Flux

In general, the increase in tangential velocity decreases the thickness of the concentration polarization layers on the surface of membrane. It also makes it possible to reduce the deposition of large particles in suspension which are more easily trained. Finally, it generates shear forces in the vicinity of the membrane, leading to an increase in ultrafiltrate flow. This is why the study of the influence of CV is crucial for the three membranes, in order to determine the best operating conditions.

As shown previously, membrane with the higher pore size exhibits the higher flux. The evolution of the permeation flux as a function of TMP for different CV is illustrates in Fig. 16.16. At equal TMP, the increase of CV improves the permeation flux across membranes.

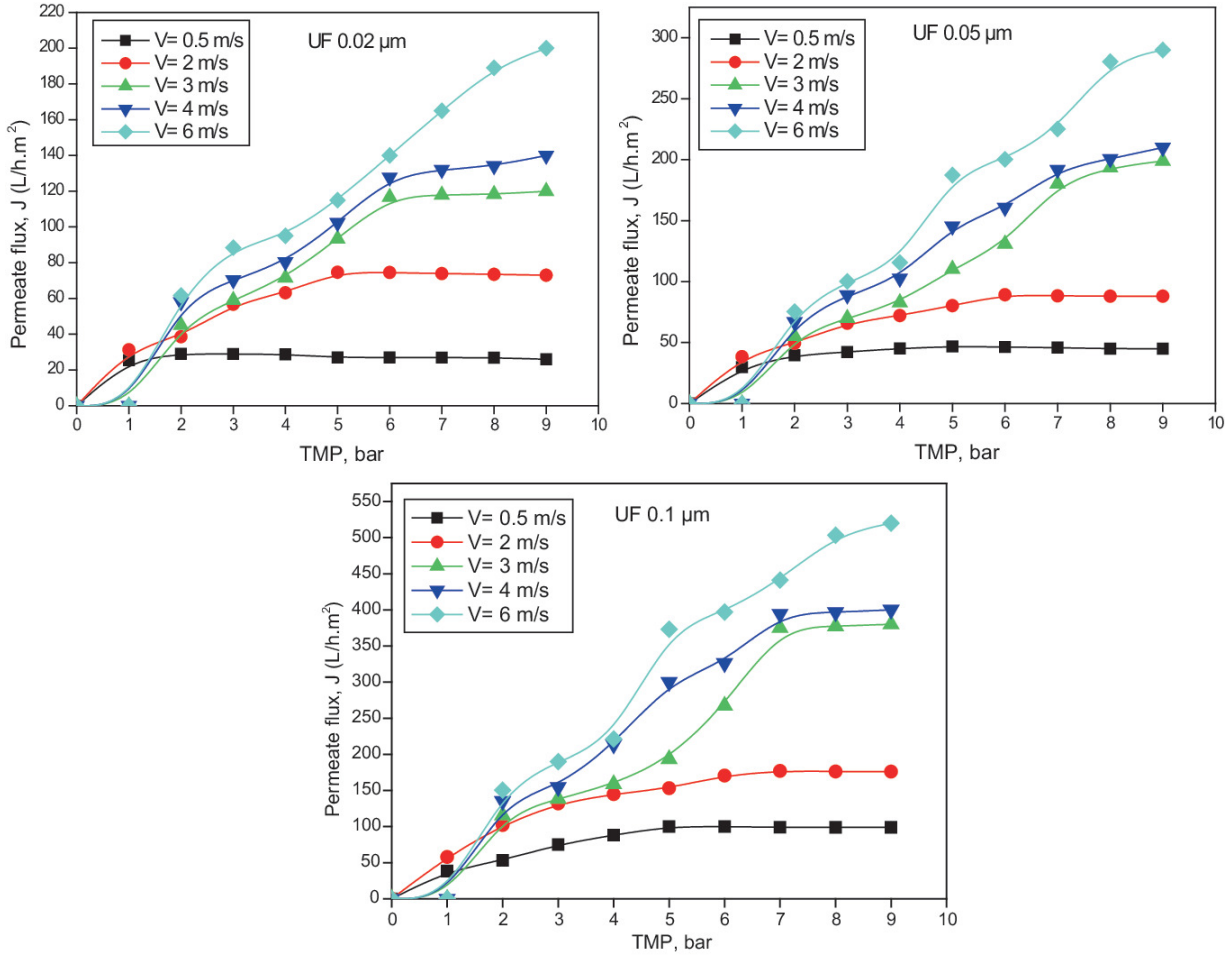


Fig. 16.16 Evolution of the permeation flux as a function of TMP at different CV for the three membranes studied (26 °C)

At equal CV, the increase of TMP enhances the permeation flux until it reaches a limiting flux, a plateau. This limiting flux disappears as soon as the CV reaches 6 m/s.

An increase in CV leads to an increase in permeate limiting flux across UF membranes. Depending on the film model, an increase in CV improves the hydrodynamic conditions by reducing the concentration polarization layer thus avoiding the formation of the fouling layer and enhancing the mass transfer coefficient resulting in an increase of the permeation flux [79]. At the higher CV (6 m/s), the permeate limiting flux is not reached meaning that in these conditions, ongoing hydrodynamic conditions reduce concentration polarization and allow continuous

LFL renewal at the solution-membrane interface for the three studied membranes. This of course has an energy cost since circulation requires energy. For this reason, a compromise must be found between the CV and the permeation flow across the membranes.

Influence of CV on Permeate Quality

Figure 16.17 gives the evolution of the COD and SS retention as a function of CV for different TMP for the three membranes. Results show that for each applied TMP, retention of COD and SS exhibited by membranes remain stable independently of the change in CV. CV has no influence on the retention of COD and SS and consequently on permeates quality. Thus, for the UF membrane with porosity of 0.02 µm and for a TMP of 5 bar, the change in CV from 0.5 to 6 m/s does not affect retention values of COD and SS which remain invariant, 85 and 70.4% respectively. The same phenomenon is observed with the other two membranes.

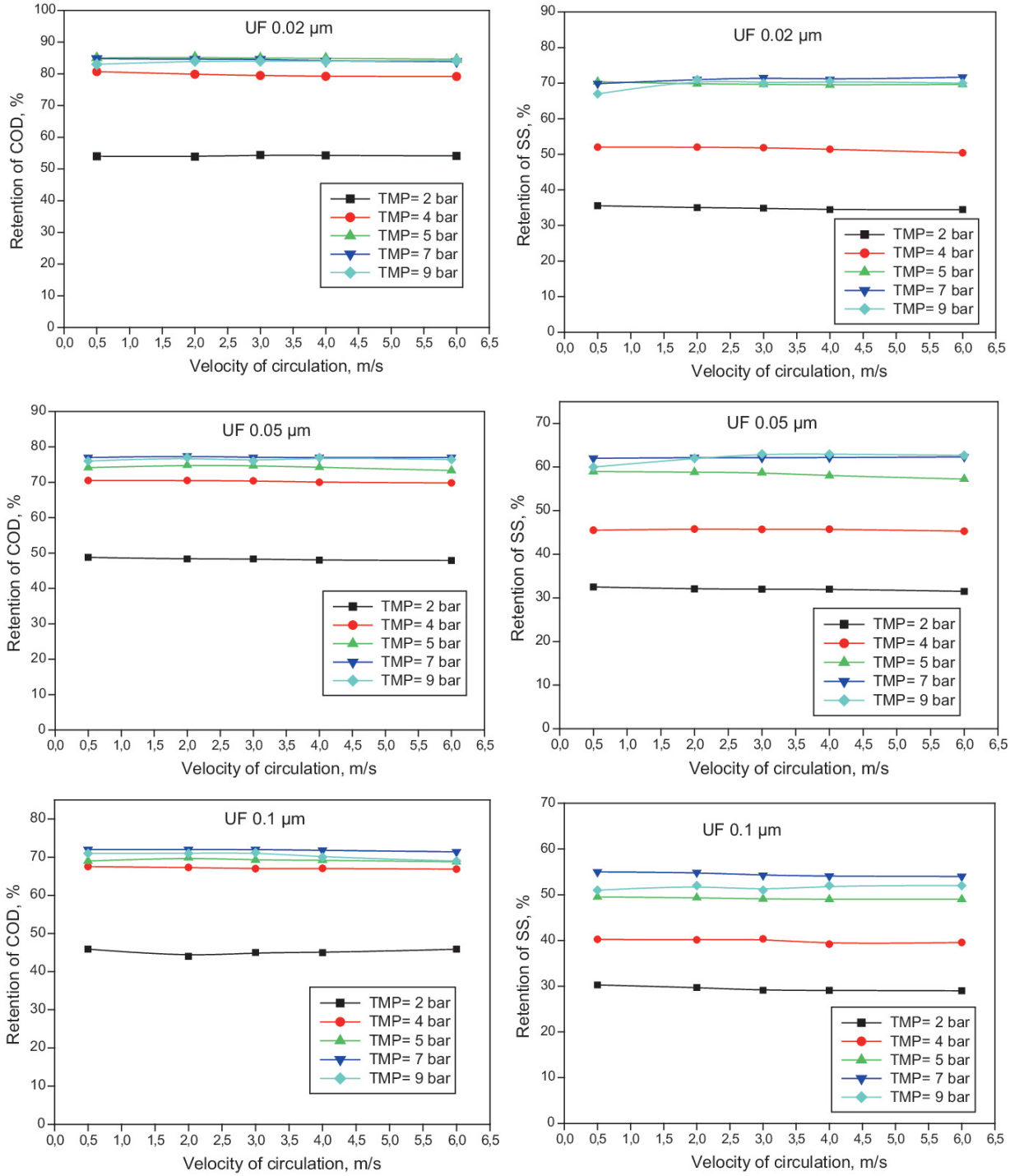


Fig. 16.17 Evolution of the COD and SS retentions as a function of CV for different TMP

According to these results, as confirmed previously, CV greatly affects the permeation flux, disrupting the establishment of concentration polarization. But it does not affect COD and SS retentions for the three membranes

which are related to the inherent structural properties of UF membrane (pore size) and the size of the particles retained.

Regarding the obtained quality of permeate, average retention of 70% of the COD and 60% of the SS have been achieved.

So, the choice of the working CV must be a subject of depth reflection in the process of optimizing LFL pretreatment by UF. A low CV seems to be penalizing in terms of filtration quantity performance and a strong propensity to concentration polarization and fouling. Inversely, a high CV (between 4 and 6 m/s) will provide better permeation flow rate with stable retention as mentioned previously. But this flow improvement will be thanks to additive energetic consumption. The choice of operating TMP and CV must be the subject of a compromise only capable of sustaining working conditions and guaranteeing stability in terms of filtration performance.

The objective of this work is to study and compare performances of three UF ceramic membranes in the pretreatment of the fresh Oum Azza LFL, raw effluent, which has not undergone any pretreatment, in order to improve the quality of the effluent at entrance of RO plant and reduce the membrane fouling and washing frequencies. The pretreatment consists to reduce the pollutant load of LFL, in order to improve the physico-chemical quality of the effluent upstream of RO. The results obtained with a single stage show a clear improvement in the quality of the effluent compared to the existing pretreatment provided by the operators of Oum Azza center. These improvements will reduce the RO membrane fouling and washing frequencies. However, the abatements of the major pollutants are far from guaranteeing protection of membranes according to RO manufacturer recommendations.

Conclusion

Treating the LFL by UF reduces a large part of organic pollution, however the TDS of these LFLs still very high. So, to meet the Moroccan discharge standards in terms of TDS, RO seems to be the unique process capable to responding to this constraint. This technology is known to be cost expensive and RO membranes are sensitive to fouling.

The aim of this work was to study the feasibility of UF membranes in the reduction of pollution indicators of Oum Azza LFL with the ultimate goal of replacing the current pretreatment in work at Oum Azza by UF separation. This study showed that, increasing the applied TMP for the three membranes, causes a rise in permeate flux until a limiting flux and improve the retention of SS and COD, but it has no influence on TDS.

Moreover, the CV has no influence on the retention rate of SS and COD. On the other hand, the permeate fluxes and the limiting fluxes have been improved with the increase in the CV.

A clear improvement in effluent quality compared to the conventional pretreatment used in the LFL treatment plant of Oum Azza but pollutants indicators contents still high and the effluent cannot be directed to the RO unit. Finally, among the three membranes tested, the one that best meets the requirements of RO is the 20 nm membrane.

16.3.3.4 Comparison Between Coag-Floc and UF

Coag-Floc involves the introduction of coagulants to neutralize charges on suspended particles and colloids, promoting their aggregation into larger particles. These flocs can be easily separated from the liquid phase by sedimentation or flotation, resulting in improved clarity and reduced pollutant concentrations. In contrast, UF is a

membrane-based separation process that relies on semi-permeable membranes to selectively retain particles, colloids, and macromolecules while allowing treated water to pass through. This technology offers a more advanced level of filtration, capable of achieving finer particle removal compared to conventional Coag-Floc.

This comparative study examines the advantages, limitations and applicability of Coag-Floc and UF as LFL pretreatment techniques. Factors such as efficiency, cost-effectiveness, scalability and environmental sustainability will be assessed to provide a comprehensive understanding of their respective roles in addressing the challenges posed by LFL treatment. By examining these techniques together, we can gain clearer insight into the optimal approach for LFL pretreatment, thereby contributing to the development of more efficient and environmentally friendly waste management strategies.

Table 16.5 gives the reoval rate of the polluting load obtained by Coag-Floc with ferric chloride and by the UF20 nm membrane.

Table 16.5 Removal rate of COD, SS and EC by Coag-Floc and UF

Settings	**Removal rate (%)**	
	Coag-floc with ferric chloride	**UF20 nm**
EC	18.8	6
COD	77	85
SS	66	70.4

From the results presented in Table 16.5, it is clear that the UF technique is more effective than Coag-Floc in terms of reducing the pollutant load.

16.3.4 General Conclusion

The comparison between Coag-Floc and UF in LFL pretreatment highlights the strengths and challenges

specific to each technique. Coag-Floc, thanks to its operational simplicity and its efficiency in the aggregation of particles, proves to be an economical and practical option to reduce the concentrations of contaminants in LFLs. However, its ability to treat fine particles remains limited.

On the other hand, UF stands out for its ability to eliminate particles of smaller size, thus offering a better quality of treatment. However, this method requires more complex equipment and higher operational costs, which may restrict its implementation, especially for smaller scale installations.

The choice between these two techniques must take into account various factors such as the specific composition of the LFL, the financial resources available, the scale of the installation and the treatment objectives. Depending on these considerations, a judicious combination of the two techniques might even prove to be optimal for achieving maximum preprocessing performance.

Ultimately, this comparison sheds light on the way to more suitable and efficient pretreatment solutions for LFL, thus contributing to sustainable waste management and environmental preservation.

In this study, the results of the Coag-Floc and UF comparison in the pretreatment of Oum Azza LFL show that:

Ferric chloride is more effective than aluminum sulphate for the removal rate of SS (66%) and the COD (77%).

The use of UF membrane technology in the pretreatment gives a very important removal rate of COD and SS especially for of 20 nm membrane (85 and 70.4% respectively). The removal rate increases with TMP and the CV has no significant effect on the rate of removal of COD and SS.

References

1. Minister of Energy (2019) Mines and sustainable development, national strategy for waste reduction and recovery, summary report, Morocco
2. Communication Strategy and Plan (2014) National program the household and assimilated waste, summary note, state secretariat for water and the environment, Rabat
3. Mejbri R, Matejka G, Lafrance P, Mazet M (1995) Fractionation and characterization of the organic matter in sanitary landfill leachate. J Water Sci 3:217–236
4. Matejka G, Rinke M, Mejbri R, Bril H (1994) Pollution generated by household waste disposal: water balance and characterization. Environ Technol 15:313–322
5. Khattabi H (2002) Interests of the study of hydrogeological and hydrobiological parameters for the understanding of the operation of the leachate treatment station of the household refuse dump of Etueffont (Belfort, France), Thesis, University of Franche-Comte, France
6. Baun A, Jensen SD, Bjerg PL, Christensen TH, Nyholm N (2000) Toxicity of organic chemical pollution in groundwater downgradient of a landfill (Grindsted, Denmark). Environ Technol 34:647–1652
7. Chofqi A (2004) Demonstration of the mechanisms of groundwater contamination by leachate from an uncontrolled landfill (El Jadida, Morocco): geology, hydrogeology, geoelectrics, geochemistry and epidemiology, Thesis, University of El Jadida, Morocco
8. Khattabi H, Lotfi A, Mania J (2002) Assessment of the impact of leachate from a household refuse dump on the physicochemical and bacteriological quality of the water in a Franche-Comte stream. Waste Sci Technol 24:1–4
9. Enzminger JD, Robertson D, Ahlert RC, Kosson DS (1987) Treatment of landfill leachates. J Hazard Mater 14:83–101
10. Lisk DJ (1991) Environmental effects of landfill. Sci Total Environ 100:415–468
11. Renou S, Givaudan JG, Poulain S, Dirassouyan F, Moulin P (2008) Landfill leachate treatment: review and opportunity. J Hazard Mater 150:468–493
12. Wiszniowski J, Robert D, Surmacz-Gorska J, Miksch K, Webe JV (2006) Landfill leachate treatment methods: a review. Environ Chem Lett 4:51–61

13. Chaouki Z, El Mrabet I, Khalil F, Ijjaali M, Rafqah S, Anouar S, Zaitan H (2017) Use of coagulation-flocculation process for the treatment of the landfill leachates of Casablanca city (Morocco). J Mater Environ Sci 8:2781–2791
14. Djeffal K, Bouranene S, Fievet P, Déon S, Gheid A (2021) Treatment of controlled discharge leachate by coagulation-flocculation: influence of operational conditions. Sep Sci Technol 56(1):168–183
15. Chaouki Z, Hadri M, Nawdali M, Benzina M, Zaitan H (2021) Treatment of a landfill leachate from Casablanca city by a coagulation flocculation and adsorption process using a palm bark powder (PBP). Sci Afr 12:e00721
16. Shehzad A, Bashir MJ, Sethupathi S, Lim JW (2015) An overview of heavily polluted landfill leachate treatment using food waste as an alternative and renewable source of activated carbon. Process Saf Environ Prot 98:309–318
17. Erabee IK, Ahsan A, Jose B, Aziz MMA, Ng AWM, Idrus S, Daud NNN (2018) Adsorptive treatment of landfill leachate using activated carbon modified with three different methods. KSCE J Civ Eng 22:1083–1095
18. Rohers F, Dalsasso RL, Nadaleti WC, Matias MS, de Castilhos Júnior AB (2021) Physical–chemical pre-treatment of sanitary landfill raw leachate by direct ascending filtration. Chemosphere 285:131362
19. Jain B, Singh AK, Kim H, Lichtfouse E, Sharma VK (2018) Treatment of organic pollutants by homogeneous and heterogeneous Fenton reaction processes. Environ Chem Lett 16:947–967
20. Smaoui Y, Mlaik N, Bouzid J, Sayadi S (2018) Improvement of anaerobic digestion of landfill leachate by using coagulation flocculation, Fenton’s oxidation and air stripping pretreatments. Environ Prog Sustain Energy 37:1041–1049
21. Maslahati Roudi A, Chelliapan S, Wan Mohtar W, Kamyab H (2018) Prediction and optimization of the Fenton process for the treatment of landfill leachate using an artificial neural network. Water 10:595
22. Makhatova A, Mazhit B, Sarbassov Y, Meiramkulova K, Inglezakis VJ, Poulopoulos SG (2020) Effective photochemical treatment of a municipal solid waste landfill leachate. PLoS ONE 15(9):e0239433
23. Mahtab MS, Islam DT, Farooqi IH (2021) Optimization of the process variables for landfill leachate treatment using Fenton based advanced oxidation technique. Eng Sci Technol 24(2):428–435
24. Rathnayake WAPP, Herath GBB (2018) A review of leachate treatment techniques. In: 9th International conference on sustainable built

environment, Kandy, Sri Lanka, pp 13–14
25. Ramdhani MY, Sururi MR, Ainun S (2018) Leachate treatment from Sarimukti landfill using ozone with sludge from water treatment plant as a catalyst. MATEC Web Conf 147:04006
26. Leszczyński J, Maria JW (2018) The removal of organic compounds from landfill leachate using ozone-based advanced oxidation processes, E3S Web Conf 45:00046
27. Kwarciak-Kozłowska A (2018) Pretreatment of stabilized landfill leachate using ozone. J Ecol Eng 19:186–193
28. Yan C, Cheng Z, Quan X, Feng C, Cheng G (2021) Electrochemical pretreatment of landfill leachate RO concentrate with multichannelmesh electrodes. Environ Chem 2:603–613
29. Scandelai APJ, Cardozo Filho L, Martins DCC, de Souza Freitas TKF, Garcia JC, Tavares CRG (2018) Combined processes of ozonation and supercritical water oxidation for landfill leachate degradation. Waste Manag 77:466–476
30. Kargi F, Pamukoglu MY (2003) Pamukoglu, Aerobic biological treatment of pre-treated landfill leachate by fed-batch operation. Enzyme Microb Technol 33:588–595
31. Vilar VJP, Rocha EMR, Mota FS, Fonseca A, Saraiva I, Boaventura RAR (2011) Treatment of a sanitary landfill leachate using combined solar photo-Fenton and biological immobilized biomass reactor at a pilot scale. Water Res 45:2647–2658
32. Chakraborty S, Ranjan K, Verma M, Iqbal J, Naresh R (2015) Assessing the feasibility of co-treatment of landfill leachate and municipal wastewater in sequencing batch reactor (SBR). In: Proceedings of International Conference on Sustainable Energy and Built Environment, pp 574–578
33. Hashemi H, Zad TJ, Derakhshan Z, Ebrahimi AA (2017) Determination of sequencing batch reactor (SBR) performance in treatmentof composting plant leachate. Health Scope. 6:13356
34. Reis BG, Silveira AL, Teixeira LPT, Okuma AA, Lange LC, Amaral MCS (2017) Organic compounds removal and toxicity reduction of landfill leachate by commercial bakers’ yeast and conventional bacteria based membrane bioreactor integrated with nanofiltration. Waste Manag 70:170–180
35. Spina F, Tigini V, Romagnolo A, Varese G (2018) Bioremediation of landfill leachate with fungi: autochthonous vs allochthonous strains. Life 8:27
36. Goncalves AL, Pires JC, Simoes M (2017) A review on the use of microalgal consortia for wastewater treatment. Algal Res 24:403–415

37. Paskuliakova A, McGowan T, Tonry S, Touzet N (2018) Microalgal bioremediation of nitrogenous compounds in landfill leachate the importance of micronutrient balance in the treatment of leachates of variable composition. Algal Res 32:162–171
38. Pirsaheb M, Hossini H, Secula MS, Parvaneh M, Ashraf GM (2017) Application of high rate integrated anaerobic-aerobic/biogranular activated carbon sequencing batch reactor (IAnA-BioGACSBR) for treating strong municipal landfill leachate. Sci Rep 7:3109
39. Begum S, Anupoju GR, Sridhar S, Bhargava SK, Jegatheesan V, Eshtiaghi N (2018) Evaluation of single and two stage anaerobic digestion of landfill leachate: effect of pH and initial organic loading rate on volatile fatty acid (VFA) and biogas production. Biores Technol 251:364–373
40. Cirik K, Goçer S, Duyar A, Kozak M (2021) Treatment of landfill leachate by anaerobic baffled reactor (ABR). Environ Res Technol 4(2):134–139
41. Luo J, Qian G, Liu J, Ping Xu Z (2015) Anaerobic methanogenesis of fresh leachate from municipal solid waste: a Brief review on current progress. Renew Sustain Energy Rev 49:21–28
42. Kheradmand S, Karimi-Jashni A, Sartaj M (2010) Treatment of municipal landfill leachate using a combined anaerobic digester and activated sludge system. Waste Manage 30:1025–1031
43. Zait M, Lmbakini L, Kitanou S, El Filali N, Bakraoui M, El Bari H, El Midaoui A, Taky M (2018) Feasibility Study of biogas anaerobic production of concentrate of leachates treated by reverse osmosis. In: International workshop ecological sustainable waste management-energetic utilization of organic waste “BIOWASTE4E”, 26–28 Nov 2018, Kenitra, ISBN 978-3-86387-935-8
44. Anand A, Singh S (2014) Membrane technique for leachate treatment: a literature review. Int J Environ Res Dev 4:33–36
45. Fatima S, Jehangir A, Bhat G (2017) Treatment of Landfill leachate using reverse Osmosis and its potential for reuse. ID 214704296
46. Ushikoshi K, Kobayashi T, Uematsu K, Toji A, Kojima D, Matsumoto K (2002) Leachate treatment by the reverse osmosis system. Desalination 150:121–129
47. Sabah F, Arshid J, Bhat GA (2017) Treatment of landfill leachate using reverse osmosis and its potential for reuse. IJIRSET 6:15827
48. Linde K, Jönsson AS (1995) Nanofiltration of salt solution and landfil leachate. Desalination 103:223–232

49. Chaudhari LB, Murthy ZVP (2010) Treatment of landfill leachates by nanofiltration. J Environ Manage 91:1209–1217
50. Istirokhatun T, Amalia DA, Oktiawan W, Rezagama A, Budihardjo MA, Nofiana N, Susanto H (2018) Removal of refractory compounds from landfill leachate by using nanofiltration. Jurnal Teknologi. 80:2–3
51. Trebouet D, Schlumpf JP, Jaouen P, Quemeneur F (2000) Stabilized landfill leachate treatment by combined physicochemical–nanofiltration Processes. Wat Res 35:2938
52. Elfilali N, Essafi N, Zait M, Tahaikt M, Elazhar F, Elmidaoui A, Taky M (2021) Effectiveness of membrane bioreactor/reverse osmosis hybrid process for advanced purification of landfill leachate. Desalin Water Treat 240:24–32
53. Abuayyash A, Sayara T, Kanan A, Qurie M (2018) Characterization and treatment of Al-Menya landfill leachate using biological and physical methods. Environ Sci Indian J 14(2):161
54. Pi KW, Gaoa LX, Fana MX, Gongb WQ, Wana DJ (2009) Two-stage biodegradation coupled with ultrafiltration for treatment of municipal landfill leachate. Process Saf Environ Protect 338
55. Özyaka VŞ, Çakmakc M, Busra Yaman F, Özkaya B, Karadağ D (2015) treatment of compost leachate by membrane processes. Environ Eng Manage J 2239
56. Ameen ES, Muyibi SA, Abdulkarim MI (2011) Microfiltration of pretreated sanitary landfill leachate. Environmentalist 31(3):208–215
57. Pertile C, Zanini M, Baldasso C, Andrade MZ, Tessaro IC (2018) Evaluation of membrane microfiltration fouling in landfill leachate treatment. Matéria (Rio de Janeiro) 23(1)
58. Piatkiewicz W, Biemacka E, Suchecka T (2001) A polish study: treating landfill leachate with membranes. Filtr Sep 38:22–26
59. Preservation of the Quality of Water Resources and Fight Against Pollution (2014) Ministry delegated to the minister of energy, mines, water and the environment, loaded with water, direction of the research and planning the water. Water Quality Division, Water Pollution Control Service, Morocco
60. Saadi S, Sbaa M, El khormouz M (2013) Physico-chemical characterization of leachate from the technical landfill center in the city of Oujda Morocco. Science Lib Editions Mersenne 5:1–12
61. Zalaghi A, Lamchouri F, Toufik H, Merzouki M (2014) Valorization of natural porous materials in the treatment of leachate from the landfill

uncontrolled city of Taza. J Mater Environ Sci 5:1643–1652
62. Khalil F, Bouaouine O, Chtioui H, Souabi S, Aboulhassan MA, Ouammou A (2015) Treatment of landfill leachate by coagulationflocculation. J Mater Environ Sci 6:1337–1342
63. Mejraoua Z, Zine N (2017) Physico-chemical characterization of the leachate from the wild landfill of Meknes. Eur Sci J 13:1857–7431
64. Jirou Y, Harrouni MC, Belattar M, Fatmi M, Daoud S (2014) Treatment of leachate from the controlled landfill of Grand Agadir by intensive ventilation. Rev Mar Sci Agron Vét 2:59–69
65. Bechac JP, Boutin P, Nuer P (1984) Traitement des eaux usées. Ed. Eyrolls, Paris
66. Fletcher I, Adgar A, Cox CS, Boehme (2001) Neural network applications in the water industry, London
67. Cardot C (1999) Les traitements de l'eau. Procédés physico-chimiques et biologiques. Ellipses Edition Marketing S.A.
68. Dhimni S, Qlihaa A, Melrhaka F, Chebabe D, Dermaj A, Hajjaji N (2015) Characterization of liquid waste of packaging industry. J Mater Environ Sci 6(11):3197–3206
69. Lamrini B, Lann MV, Benhammou A, Lakhal K (2005) Detection of functional states by the "LAMDA" classification technique: application to a coagulation process in drinking water treatment. Elsevier, C.R. Physique, pp 1161–1168
70. Lind C (1994) Coagulation control and optimization: Part 1. Public Works
71. Amokrane A (1994) Epuration des lixiviats de décharges. Prétraitement par coagulation-floculation. Traitement par osmose inverse. Post-traitement par incinération. Thèse de Doctorat, INSA de Lyon, France, p 286
72. Maulding SJ, Harris HR (1968) Effect of ionic environment and temperature on the coagulation of color-causing organic compounds with ferric sulfate. J Am Water Works Assoc 60:460–476
73. Bayramoglu M, Kobya M (2007) Treatment of wastewater by electrocoagulation economical evaluation. Chem Eng J 198:155–161
74. Hosny AY (1996) Separating oil from oil-water emulsions by electroflotation technique. Separ Technol 6:9–17
75. Trebouet D, Berland A, Schlumpf JP, Jaouen P, Quemeneur F (1998) Treatment of stabilized landfill leachates by nanofiltration membranes, Revue des sciences de l'eau/J. Water Sci 11:365–382

76. Qaid S, Zait M, EL Kacemi K, EL Midaoui A, EL Hajji H, Taky M (2017) Ultrafiltration for clarification of Valencia orange juice: comparison of two flat sheet membranes on quality of juice production. J Mater Environ Sci 8:1186–1194
77. Renou S, Poulain S, Givaudan JG, Moulin P (2009) Amelioration of ultrafiltration process by lime treatment: case of landfill leachate. Desalination 249:72–82
78. Yu-Dong X, Dong-Bei Y, Zhu Y, Yong-Feng N (2006) Fractionation of dissolved organic matter in mature landfill leachate and its recycling by ultrafiltration and evaporation combined processes. Chemosphere 64:903–911
79. Nilsson SL (1990) Protein fouling of UF membrane: causes and consequence. J Membr Sci 52:121–142