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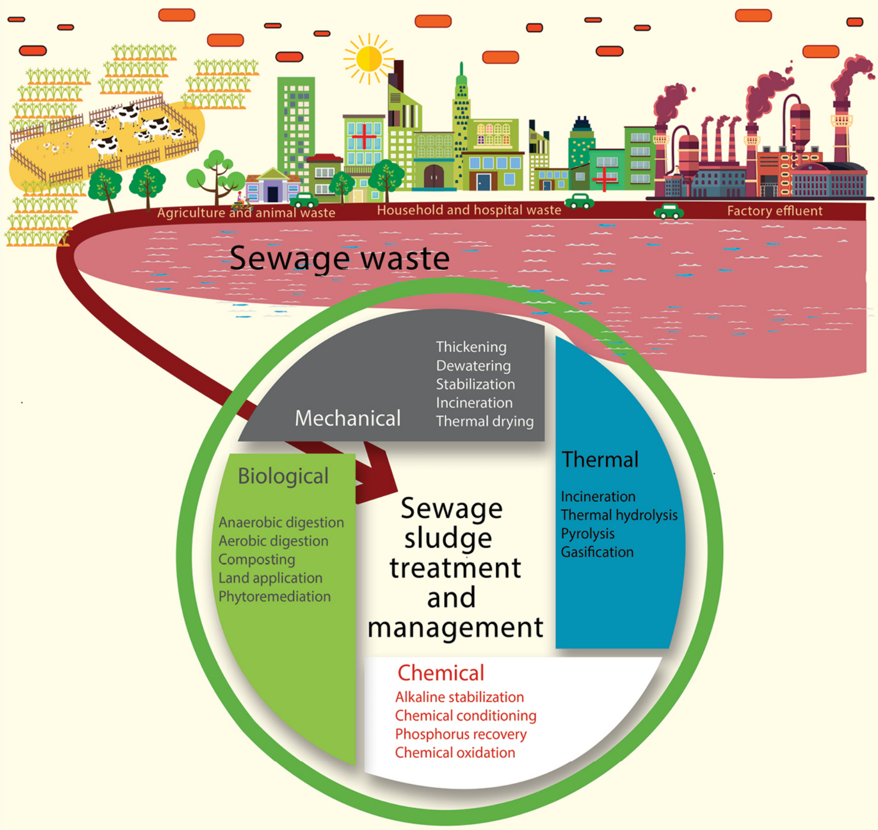
Sustainable Treatment and Management of Sewage Sludge



Edited by
Mukesh Kumar Awasthi, Zengqiang Zhang and Ashok Pandey

The Biotech Research Society, India

Sustainable Treatment and Management of Sewage Sludge

This reference book provides updated information about the technological advancement in sustainable thermochemical bioprocessing of sewage sludge disposal and resource recovery. It discusses the innovative strategies of resource recovery for the formulation of feedstock, clean compost production and safe application. This book traces the main chemical and biological properties of sewage sludge and covers biostabilization, detoxification, the role of microorganisms in sewage sludge management and the sustainable use of sewage sludge from a circular economy perspective.

Key Features

- Discusses organic waste disposal and recycling
- Covers knowledge transfer from waste bioprocessing to commercially important end products
- Includes industrial application of biological and thermochemical sewage sludge treatment toward emerging nutrient recovery technologies
- Reviews the function and applications of microorganisms in sewage sludge treatment
- Describes the application of sewage sludge as fertilizers in agriculture

This book is meant for researchers and industry experts in environmental sciences, biochemical engineering and biotechnology.

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Contents

Preface

There has been tremendous technological advancement in sustainable thermochemical and bioprocessing of sewage sludge (SS) disposal and innovative strategies for resource recovery, both for the formulation of feedstock and clean compost production as well as safe application. Though there are many books available on this subject, there is still a need for a book which consists of comprehensive information on the advancement in sustainable thermochemical and bioprocessing of SS and its applications. With this aim, this book intends to provide extensive updated and state-of-art information about various aspects and issues on the technological development of SS disposal and bioprocess technologies for the valorization of SS, including biorefinery approaches with circular economy perspective in a very simple, understandable and effective illustrated manner. This book is devoted to SS, its sustainable management and its use and bioprocessing. It traces the main chemical and biological properties of SS and covers topics such as SS biostabilization and detoxification, biological and thermochemical treatment technologies, emerging nutrient recovery technologies, the role of microorganisms in SS management and the sustainable use of SS as fertilizer in agriculture.

This book is comprised of 17 chapters. Chapter 1 introduces the SS treatment and management, highlighting the perspective of sludge recycling, combined with the concepts of SS reclamation, reduction and harmless treatment. The chapter also introduces SS disposal and management methods in line with the sustainable development strategy. Chapter 2 discusses the global scenario of SS management, introducing various methods for its management and sustainable interventions, required for the SS recycling. Chapter 3 presents an overview of the ecological and health risk assessment associated with the direct exposer of SS in the environment and prospects for the development trend as well as proposes innovative ideals for the management. Chapter 4 provides a holistic overview of municipal wastewater sludge as a sustainable bioresource in the developed countries and also discusses the viable practices of SS management and resource recovery for sustainable development. Chapter 5 presents the application of SS as an agricultural soil amendment and predicts the problems associated with the direct exposer of SS and their control. Chapter 6 focuses on the sustainable treatment and resource recovery from the SS and elaborates resource recovery processes such as enzymes, bioplastics, biopesticides, biofertilizers, proteins and nutrients from the SS. In addition, this chapter gives details on the understanding of recovering the resources and energy from the SS by advanced technologies. Chapter 7 described important principles of thermochemical processing of SS, presenting the fundamentals and challenges. It discusses the main reasons why thermochemical methods seem to offer the optimal solution for the heavy metals, pathogenic and specific microflora, and persistent organic pollutants present in SS. Chapter 8 deals with the recent update on the pyrolysis of SS and provides a basic understanding of research, development and practice of efficient and safe utilization technology of SS.

Chapter 9 discusses the modern technological development for the combustion of SS. In addition, it introduces the generation and basic composition of sludge, factors and mechanisms affecting SS combustion, equipment used for incineration process, and the pollution control measures used during the combustion process. Chapter 10 introduces the challenges and opportunities associated with hydrothermal carbonization and liquefaction of SS. This chapter also discusses the significance of hydrothermal processing due to high moisture and sludge heterogeneity. Chapter 11 covers SS properties and composition, gasification mechanism, chemistry, operating conditions, technologies and practical implications, specifically in the light of life cycle assessment and circular economy. Commercial applications of gasification process and co-conversion of waste blends is already discussed in this chapter, which seems to be a practical approach in the implementation of the process. Chapter 12 presents typical SS treatment technologies, involving microbial mechanisms.

Chapter 13 provides the details on the role of emerging resource recovery technologies in SS management, discussing best practices regarding the aforesaid technologies. Chapter 14 summarizes the biostabilization of SS by various resource recovery technologies and provides corresponding solutions; it also discusses the aerobic and anaerobic digestions, their characteristics, technology, advantages and disadvantages during SS stabilization. Chapter 15 provides details on the antibiotic-resistant genes (ARGs) types, concentrations and factors in wastewater treatment plants. Chapter 16 provides the application of bioleaching to improve sludge dewatering performance, including its principle, influencing factors and dewatering mechanism. Engineering applications of this technology can be promoted by coupling with physicochemical pretreatment technology and enhancing the stability of microbial systems in large-scale applications. Finally, the last chapter, i.e., Chapter 17 presents a proper evaluation and understanding of cocktail effects of several pollutants present in SS and the application of ecotoxicological analysis of SS prior to disposal. Effective awareness and orientation in household refurbishment and waste disposal from domestic, industrial and agricultural sources are primary and efficient steps toward SS release, control and toxicity minimization.

We believe this book will provide extensive updated information about the latest research trends technological developments related to the chemical and biological properties of SS, In particular, this book will be a valuable resource for the professionals, teachers, researchers, policy makers and graduate students interested in the SS treatment and management and its recycling by various approaches of biotechnology, biochemical and environmental engineering technology.

The editors are grateful to the reviewers for their time and support in evaluating the chapters' manuscripts and providing useful suggestions for their improvements. MKA and ZQZ acknowledge the support provided by the College of Natural Resources and Environment, Northwest A&F University, Yangling, Shaanxi, China. Finally, the editor and authors would like to thank Taylor & Francis – CRC Press and the Biotech Research Society, India for providing the opportunity to publish this book.

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Section I

General

1 Sewage Sludge Treatment and Management

An Introduction

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1.1 INTRODUCTION

The increasing industrialization and urbanization lead to the fact that the resulting waste of human life threatens the degradation (and in some cases led to the degradation) of biological resources. Among the variety of waste, special attention is paid to the problem of disposal of excess sludge and sewage sludge (SS), which has not yet been solved. Nevertheless, in our opinion, the present attempts to optimize the processes of wastewater sludge management should be synergistic and based primarily on the synthesis of (1) modern technologies for processing wastewater sludge and (2) intelligent computer systems. Let's highlight a few of the most priority areas. For example, thermal hydrolysis technologies allow to obtain more biogas compared to conventional fermentation technologies. Plasma gasification of SS in comparison with conventional gasification makes it possible to obtain gas with more favorable physico-chemical parameters.

Another important area is the intelligent technology of the Internet of Things (IoT) used in wastewater treatment technology, which allows online monitoring of changes in controlled parameters of wastewater quality, helping to carry out better wastewater treatment. For example, the idea of using a microbial fuel cell (generating electricity due to the vital activity of microorganisms) as an optimal solution in the management of wastewater and SS is very attractive today (Ana et al., 2022). Nevertheless, wastewater sludge management has a wide range of technological directions. In this chapter, we consider the technologies of anaerobic digestion, composting and vermicomposting, in most cases used today, which is economically justified, but which, with deeper implementation of intelligent technologies in these processes, can very quickly become optimal solutions in the treatment of SS in the future.

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1.2 THE GLOBAL PRODUCTION OF SEWAGE SLUDGE AND THE MAIN DIRECTIONS OF ITS MANAGEMENT

To date, the volume of wastewater in the world reaches more than 300 cubic kilometers per year (Macedo et al., 2022), while the formation of wastewater precipitation in the world can reach hundreds of millions of tons per year (Di Giacomo and Romano, 2022). However, the management of SS is not a one-sided problem as it may seem at first glance. The multidimensional nature of this problem closely intersects with the implementation and achievement of the sustainable development goals (Figure 1.1).

In order to reduce the amount of biodegradable waste, including SS, 20 years ago, each EU Member State was required to develop a corresponding waste management policy, given the apparent trend of increasing wastewater and SS. Among these management measures, it is economical and efficient to heat the waste, and then the ash after heat treatment is disposed of in landfill. However, for the sake of natural ecological health, many countries have implemented stricter waste management standards. For example, the disposal of SS is prohibited in Sweden: SS is used as fertilizers, building soil, coating material, for energy production by combustion or biogas production (Burgman, 2022). In Russia, problems with SS are also a serious problem both because of national environmental standards (Dregulo and Kudryavcev, 2018; Dregulo et al., 2022) and because of outdated sludge treatment systems (Dregulo and Bobylev, 2021a, 2021b). In the countries of Southeast Asia, wastewater precipitation amounts reach values of 24–40 million tons per year (Lwin et al., 2015; Jing et al., 2022). These volumes are comparable to the production of SS in China ~ 40 million tons per year, where about a quarter is buried, another quarter is incinerated, most of it is disposed of on earth (Wei et al., 2020). In countries with underdeveloped or developing economies, the problem of sludge disposal causes serious environmental problems, primarily associated with the risk of diseases of the population and land degradation.

1.2.1 Sewage Sludge as a Substrate – Characteristics

The policy regarding the management of SS is based primarily on the fact that these are organic fast-decaying substrates capable of having a multifactorial negative effect. However, this problem, not in everything, is largely solved by using the nutrient potential of SS as a substrate. For example, in Germany, each federal state must annually report on the quantity and quality of SS used in agriculture (Sichler et al., 2022). This is very important both from the point of view of restoring the fertility of the earth and ensuring environmental safety. Table 1.1 shows the main characteristics of biogenic substances contained in SS. From the data presented in Table 1.1, it can be seen that the variability of nutrients in wastewater sediments is significant. Such a difference in concentrations of nutrients, for example: for N 1–6 times, P 1.5 times, K 10 times and Mg 2 times, is probably caused by a difference in the life cycle of wastewater, which is dominated

FIGURE 1.1 Sewage sludge management in the implementation of the sustainable development goals.

TABLE 1.1
Composition of Biogenic Substances Contained in Sewage Sludge

Source of Sewage Sludge	N	P	K	Ca	Mg	C	Total Carbon	Organic Matter	References
Unit of measurement	[a]g kg^{-1} | [b]g kg						%		
Urban wastewater treatment plants	[a]18.3	[a]7.6	[a]1.3	[a]1.9	[a]26.1	–	–	33.7	Alonso et al. (2022)
Mixed sewage sludge	[a]15–62	[a]15432	[a]1.9–6.5	[a]10–38	[a]4–26	[a]360–412	–	–	Colón et al. (2017)
Anaerobically digested sludge	[a]39–59	[a]34	[a]2.3	[a]19–50	[a]0.3–19.2	[a]340–412	–	–	
Composted sludge	[a]22–39	[a]13–28	[a]2.8–5	–	–	[a]181	–	–	
Nine urban wastewater treatment plants with various wastewater treatment technologies	[b]18.4–70.8	[b]17.5–23.9	[b]1.6–5.1	[b]9.9–235.5	[b]2.5–9.0	[b]316–428	–	–	Sichler et al. (2022)
Six urban wastewater treatment plants with various wastewater treatment technologies	[b]18.6–26.2	[b]13.8–23.6	–	–	–	[b]228–373	–	45–63	Marzougui et al. (2022)

a|*b* The data are given in accordance with the original units of measurement.

by household effluents or industrial and other factors. This suggests that not all SS has a positive economic and agronomic value for using them as a substrate.

Nevertheless, for many countries, cities or industrial facilities, as well as agriculture, the use of SS as a substrate is firstly the most affordable technology for wastewater sludge treatment and secondly meets the principles of a "green" economy, the success in achieving which largely depends on comprehensively thought-out actions and improvements (development of new) technologies.

1.2.2 Sewage Sludge Final Treatment and Management

Globally, the amount of SS produced in the world is huge. If it is not properly treated and managed, it will have a huge impact on the natural ecological environment and even threaten human health (Yang et al., 2017). Therefore, it is necessary to conduct reasonable final treatment and management of SS. How to treat and manage this SS is shown in Figure 1.2.

1.2.2.1 Anaerobic Digestion

Anaerobic digestion is one of the most widely used SS treatment methods (Khawer et al., 2022). Traditional SS disposal methods, such as sanitary landfill, are only used to treat SS without considering the recycling and resource utilization of SS, which wastes the value of SS in vain. Anaerobic digestion breaks this status quo. Anaerobic digestion can be operated on a large scale, saving a lot of manpower, material resources, financial resources at the same time, can be an effective treatment of SS. More importantly, the sludge can be recycled and reused. On the one hand, in the process of anaerobic digestion, anaerobic bacteria can digest and decompose organic matter in SS, so that the volume and quality of SS is reduced and converted into nutrient-rich biogas residue and digestive liquid, which is eventually applied to agricultural production (Zhang et al., 2019).

On the other hand, the anaerobic digestion of SS is accompanied by the production of a large amount of biogas, which can be converted into natural gas after proper processing, thus easing the energy pressure. In addition, in order to obtain better benefits through anaerobic digestion, SS can be used for anaerobic co-digestion with other substrates. For example, SS and food waste (1:1) anaerobic co-digestion can promote the decomposition of organic matter and increase biogas

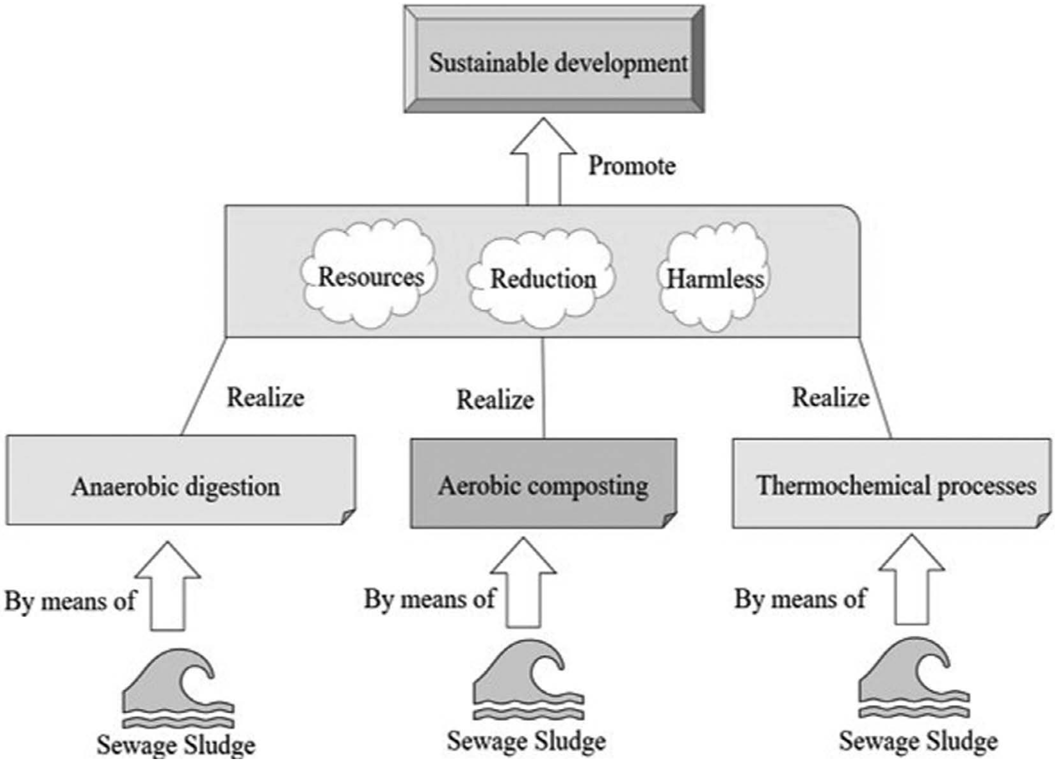

FIGURE 1.2 Analysis of sewage sludge disposal and management.

production (Arelli et al., 2021). At the same time, some special additives can be selectively added to promote the operation of the reaction in the anaerobic digestion system. For example, under the condition of adding nano-zero-valent iron (nZVI-BC) supported by biochar, anaerobic co-digestion of SS and food waste can improve the degradation efficiency of organic matter and methane production (Zhang and Wang, 2021).

However, from another point of view, anaerobic digestion also has some disadvantages. On the one hand, sludge anaerobic is carried out by microorganisms, and the survival of microorganisms requires a certain temperature. In other words, the efficiency of SS anaerobic digestion is different under different temperatures. Anaerobic digestion itself takes a long time, if the temperature conditions cannot meet the requirements, anaerobic digestion time may be longer. In winter, due to the low temperature, SS treatment is very ineffective or cannot be treated. On the other hand, since the anaerobic digestion device is sealed, and a large amount of methane and other gases are produced, it is necessary to control the gas collection of the anaerobic digestion device, otherwise it is prone to accidents. But with the development of science and technology, these problems can be solved well. In short, anaerobic digestion is an effective way to enhance the stability of sludge and implement the utilization of sludge resources recovery, so that SS waste into treasure, not only reduces the harm of SS, but also increases the commercial value of SS, realizes the sludge resources, reduction, harmless and promotes sustainable development.

1.2.2.2 Composting and Vermicomposting

Aerobic composting is one of the final disposal management methods of SS. Through aerobic composting treatment, the contents of toxic elements and various pollutants in SS are reduced, and nutrients in SS are accumulated, so that SS can be transformed into high-quality and stable organic fertilizer (Chen et al., 2022). For earthworm composting, earthworm composting is a process that degrades organic matter by the interaction of earthworms and microorganisms. The effect of SS treatment is similar or even better than that of aerobic compost described earlier. In particular, in the process of earthworm composting, there are many earthworms living in SS, and the earthworms will use the composting environment to realize population growth and produce a large amount of excrement, which is mixed in SS, so that the fertility of SS is enhanced (Doan et al., 2013). In addition, at the end of worm composting, earthworms in SS can be recycled to realize the recycling of biological resources (Zhou et al., 2022). However, SS often contains toxic elements and various pollutants, which will pose a great threat to the survival and reproduction of earthworms. Therefore, in the process of earthworm composting, in order to ensure the activity of earthworm, certain materials need to be added (Mali productive, Malińska et al., 2017).

On the other hand, aerobic composting has its limitations. First of all, there is a huge amount of SS to be treated, and aerobic composting takes a long time, so it cannot be quickly and efficiently treated SS, resulting in a large amount of SS accumulation. Second, temperature and C/N are important influencing factors of aerobic compost. Taking temperature as an example, aerobic composting is similar to anaerobic digestion and needs to be carried out at an appropriate temperature. In winter, due to the decrease in temperature, aerobic composting and earthworm composting cannot be carried out normally, thus losing the SS treatment capacity, which requires artificial application of temperature. In addition, in the composting process, it is necessary to turn the heap to improve the dredging of oxygen in the composting matrix, the workload is huge, generally need to carry out large-scale mechanized operation, which requires the occupation of a certain amount of land for the construction of the plant, and spend a certain amount of money to purchase the corresponding equipment, so the early stage of SS composting treatment investment is larger. At the same time, composting can be done safely and cost less money if it is fully equipped and with fewer people involved. Finally, sometimes SS composting treatment is not effective, and corresponding treatment needs to be carried out, such as anaerobic co-digestion with other substances, adding additives into the compost matrix, which needs to be verified by scientific experiments and can only be implemented if it is suitable for large-scale production. But with scientific progress, these questions will be solved one by one. In short, compost and earthworm compost are the treatment methods of SS recycling, reduction and harmlessness. Compost and vermicompost are the best options for the treatment and management of SS at a low consumption level. They can greatly reduce the pressure of SS treatment and realize the resource utilization of sludge, thus promoting sustainable development.

1.2.2.3 Thermochemical Processes

The thermochemical process is an effective means to promote sustainable development of continuously produced SS. As a chemical treatment method, the thermochemical process can maximize SS treatment and utilization, reduce the negative impact of SS and improve the utilization value of SS (Liu et al., 2018). Through thermochemical treatment, the volume of SS can be reduced to a large extent, and disease-causing microorganisms and organic pollutants in SS can be substantially eliminated, which greatly reduces the negative impact of SS on the environment (Chan and Wang, 2016). In addition, SS can finally be converted into hydrogen, petroleum and other energy sources for recycling, becoming a substitute for non-renewable resources, alleviating energy pressure and improving economic benefits of SS (Shatir et al., 2017). It is important to note that the thermochemical process is not as time-consuming and seasonal as methods such as aerobic composting and anaerobic digestion. Thermochemical processes can be carried out continuously and efficiently (Hu et al., 2022; Gururani et al., 2022). Finally, despite the complexity of the process and equipment involved in thermochemical treatment, the treatment effect of SS and the economic value of the product under thermochemical treatment are superior to other methods (Gururani et al., 2022). Therefore, thermochemical treatment is regarded as one of the promising sludge management methods, which can carry out the recycling, reduction and harmless management of SS, so as to promote sustainable development.

1.3 SEWAGE SLUDGE AS SOURCES AND DRIVE PATHWAYS FOR CONTAMINANTS

Wastewater treatment is a well-established process used to convert wastewater into water that cannot harm the environment and generate sludge as the end product. Sludge is wastewater residue that can be solid or semi-solid. Municipal wastewater treatment plants (WWTP) treated sewages of various sources such as houses, industries, hospitals and restaurants. Globally, SS volume has rapidly increased and is used widely as a soil conditioner and fertilizer in agriculture. It is a source of energy and operated as a cropland fertilizer and soil fertility regulator, and it also can be used to extract compounds such as phosphorous. SS dries out by a dryer or sunlight up to 95% and then dumped in the agricultural land as fertilizers. Moreover, a sludge-based fertilizer is accepted as an inexpensive fertilizer strategy and plays an essential role in the agriculture sector to enhance crop production (Chojnacka et al., 2019).

In recent years, a primary concern has arisen about sludge-associated contaminates in many developing countries such as Australia, Europe and North America (Cucina et al., 2021). Therefore, many countries started to identify and treat sludge-associated contaminants or organic pollutants (such as metals, pathogens and organic contaminants) to regulate quality and safety during the application as a fertilizer. The pathway of chemical contaminants mainly depends on the source, chemical nature and treatment procedures. Pathogen and chemical contaminants may be degraded or absorbed during sewage treatment, or they will sorb on sludge, and in this condition, sludge needs to pretreat through anaerobic digestion or lime amendments. These processes significantly impact the pathogen and contaminants concentration; however, many developing countries standardize the quality criteria for the pollutants and contaminants that are dangerous to humans (Figure 1.3). Sludge-associated contaminants are described in Table 1.2.

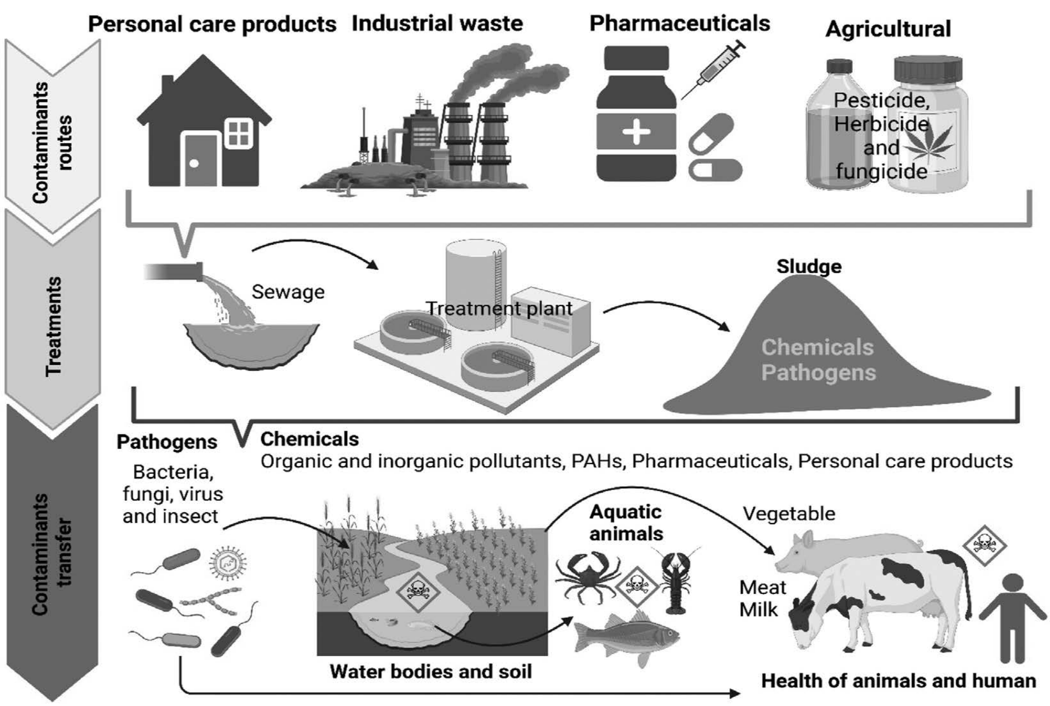

FIGURE 1.3 An schematic overview of sewage sludge contaminants and the associated bodies. PAHs: polyaromatic hydrocarbons.

TABLE 1.2
List of Examples of Sewage Sludge-Associated Contaminants

Country	Contaminates	Major Finding	Reference
Italy	Nonylphenol ethoxylates – NPnEOs (sum of NP, NP1EO and NP2EO)	5% of sludge samples showed higher values than 50 mg kg^{-1} dm (EU limit).	Lamastra et al. (2018)
	Di (2-ethylhexyl) phthalate (DEHP)	2.5% of sludge samples showed higher values than 100 mg kg^{-1} dm (EU limit).	Lamastra et al. (2018)
China	Phthalate esters (PAEs)	6 PAEs (range 10–114 mg kg^{-1} dry weight) were found in sewage sludge from Beijing.	Cai et al. (2007)
	Polycyclic aromatic hydrocarbons (PAHs)	64% of sludge samples exceeded the maximum permissible concentration (6.0 mg kg^{-1} d.w.).	Cai et al. (2007)
	Semi-volatile organic compounds (SVOCs)	The concentrations of DEHP in 91% sludge samples met the limit (100 mg kg^{-1} d.w.) proposed by the Europe Union for land application.	Cai et al. (2007)
Ireland	Metals	All metals were within EU regulatory limits except two potentially hazardous metals, antimony (Sb) and tin (Sn), concentrations in soils (17–20 mg Sb kg^{-1} and 23–55 mg Sn kg^{-1}).	Healy et al. (2016)
China	Metals	Results showed that the immobilization of heavy metals in sewage sludge significantly affected that are Zn (2.42 mg kg^{-1}), Cd (1.65 mg kg^{-1}), Cu (3323 mg kg^{-1}), Ni (422 mg kg^{-1}), Pb (69.7 mg kg^{-1}) and Cr (1983 mg kg^{-1}), and hydrothermal treatment combined with pyrolysis reduced the contaminants.	Wang et al. (2016)
China	Metals	Mercury was detected in all sewage sludge collected, and the total mercury concentration ranged from 0.3 to 7.7 mg kg^{-1}.	
Spain	Antibiotics	The antibiotics most frequently detected and in higher quantities in these sludge materials were ciprofloxacin and levofloxacin.	
India	Polychlorinated biphenyls (PCBs)	The concentration of ΣPCBs in the sludge, sewage and agriculture wastes ranged from 497 to 800 µg kg^{-1} with the mean value of 634 ± 146 µg kg^{-1}.	Patel et al. (2015)
South Africa	Organochlorine pesticides	Six predominant congeners found higher than EU limits are in this order dichlorodiphenyl trichloroethane (DDT), dichlorodiphenyl dichloroethane (DDD), α-BHC, γ-BHC, aldrin and endosulfate 1, and with values of 1512, 1330, 1095, 998, 994 and 547 ng g^{-1}, respectively.	Ademoyegun et al. (2020)
Canada	Pharmaceuticals and personal care products (PPCPs)	Carbamazepine, ibuprofen, acetaminophen, triclosan, triclocarban, venlafaxine and citalopram were detected in range (5–74 ng L^{-1}).	Gottschall et al. (2012)
China	Me-PAHs and PAHs	The concentrations of ∑PAHs, ∑Me-PAHs ranged from 567 to 5040 and 48.1 to 479 ng. g^{-1} dw, which is higher than the safety limit for sludge in agriculture in China.	Mohammed et al. (2021)

1.3.1 Chemical Contaminants in Sludge

Recent reports have reported on the types of chemical contaminants in SS, such as heavy metals, polycyclic aromatic hydrocarbons (PAHs), hydrocarbons, polychlorinated biphenyl (PCB), perfluorinated surfactants (PFCs), personal care products (PCPs), pharmaceuticals (PhCs) and benzotriazole (Fijałkowski et al., 2017; Lamastra et al., 2018; Wluka et al., 2021). Heavy metals are metals that have higher gravity (more than 4.5 g cm^{-3}), and it primarily associated with industrial effluents and surface runoff. As a nondegradable contaminant, it is transmitted to the soil, water and plant through SS and impacts the health of water bodies, animals and humans. The major metals considered contaminants in higher concentrations are Cr, Mn, Fe, Co, Ni, Cu, Zn, Hg, Cd, Pb, Sn, Mo and V, metalloids such as As, Se and light metals such as Al. The metal contaminants range from 0.2 to 2% of dry sludge. Wang et al. (2016) studied the number of metals in SS and arranged the number of metals from high to low as follows: (1) Zn, Cu, Cr, Ni, Pb, Cd and (2) Zn, Cr, Pb, Cu, Ni, Cd. SS application accumulates heavy metals in soil (Kowalik et al., 2021). Higher values of heavy metals (higher than US EPA limits) were identified in Kenya's agricultural soil by applying SS (Mungai et al., 2016). Industrial effluent releases many contaminants such as PAHs and organic contaminants that impact the soil after the sludge application.

Recently, 16 PAHs have been recognized as major pollutants, and among them, 7 are registered as carcinogens by the Environmental Protection Agency (US EPA) (Fijałkowski et al., 2017; Lamastra et al., 2018; Wluka et al., 2021) (Table 1.2). In the case of organic contaminants, different chemicals associated with pharmaceuticals, PCP, pesticides and metabolites are found in WWTPs (Bueno et al., 2012). Industrial and domestic discharge, urban and agricultural runoff, is the main route of organic contaminants that directly or indirectly impact the soil and other associated creatures. Lipophilic (fat-soluble) and hydrophobic contaminants settle into SS solids during wastewater processing. Organic contaminants associated with SS incorporate with the top layer of soil and then enter the plant and transport to the animal and humans (Figure 1.3). However, Brändli et al. (2007) reported that low molecular mass PAHs could be degraded in the soil during composting. However, contaminants such as PCBs and polychlorinated dibenzodioxins and furans (PCDDs/Fs) could be sustained in the soil for a long time (Umlauf et al., 2011).

Next, a harmful substance associated with the SS is grouped as pharmaceuticals that contain antibiotics (antimicrobials, antivirals and fungicidal), disinfectants and sanitizer, steroids, hormones, nutraceuticals and different medicinal products. The direct route of these compounds is animal and human waste, and hospital wastewater transfers to sewage water. Some compounds degrade during the process, and many are deposited and detected in the SS or in treated soil (Samal et al., 2022). Despite pharmaceuticals, daily use of PCPs such as body soap, shampoos, cosmetics and lotions discharges into the sewage through domestic water. The SS that has the substance of these PCPPs can enter the food chain of humans and animals (Latare et al., 2014). Many recent reports clarify the association of pharmaceuticals with domestic wastewater, including antibiotics, antiepileptics, anticoagulants, analgesics and anti-inflammatories, lipid regulators, steroidal compounds, cosmetics and psycho-stimulants (Luo et al., 2014). Another contaminant that impacts the environment through SS is microplastics and fibers. SS biosolid is the major receptor of microplastics and fiber during sewage treatment (Carr et al., 2016; Mahon et al., 2017), and land application of sludge pollutes the ecosystem. Moreover, nanoparticle-based products also lead the attention of policymakers to regulate their side effects or impacts on animals and humans (Godwin et al., 2015; Jain et al., 2018). Recent reports identify the nanoparticles such as Cu, TiO_2, Ag or CeO_2 in the biosolid of sludge (Fijałkowski et al., 2017).

1.3.2 Pathogenic Organisms

SS contains different pathogens that are most prevalent for humans and animals. Biosolid application to soil enhances the public health risk through pathogen association in the food chain and resistance development (Sorinolu et al., 2020). These pathogens are primarily present in human and animal waste discharged into the sewage. In the soil, only some pathogens can survive for a long time, and a few can also sustain in the dormant stage till they get suitable conditions (i.e., temperature, soil type and moisture content) to grow in the soil. Pathogen activity varies according to many factors. Among the bacterial pathogen, *Escherichia coli*, a fecal indicator organism (FIO), has been used to check SS quality. Zhang et al. (2015) demonstrated the information of *Truepera* and other bacterial groups in SS through DNA sequencing. At the same time, the information of *Truepera* and other bacterial groups in biofilm was also displayed. Godzieba et al. (2022) identified that nitrifiers (e.g., *Nitrosomonas* and *Nitrospira*) and phosphorus-accumulating organisms (*Candidatus Accumulibacter*) are dominantly growing in the biofilms and low in activated sludge. A Q fever-causing bacteria *Coxiella burnetii* DNA was detected in activated sludge samples (Schets et al., 2013).

In the case of viral pathogens, recent sequencing tools identify the new viruses (e.g., *Coronavirus* HKU1, *Klassevirus* and *Cosavirus*) that can infect humans and are dominantly associated with SS. A systemic review on viruses in SS reported that two enteric viruses (adenovirus and norovirus) are abundant in SS, and rotavirus, hepatitis A virus and enterovirus are frequently found in SS samples, and untreated biosolid application to land may improve the risk of virus infection (Gholipour et al., 2022). Next, health risks associated with helminth contamination of land applied SS have not been well documented. Only a few studies reported their presence and effect on human health through SS application. Mahapatra et al. (2022) reported that helminth eggs detection and treatment are an efficient technology for reducing public health risks. Amoah et al. (2018) discovered ten helminth eggs, such as Taeniid eggs, from SS samples of Tunisia.

1.4 CONCLUSIONS AND PERSPECTIVES

The continued production of large quantities of SS poses a significant ecological threat, but it is worth noting that SS is not only harmful. Through proper SS disposal and management methods, SS can show different utilization values. In other words, appropriate SS disposal and management methods can remove harmful substances in sewage sludge, turn SS into treasure, improve the value of SS, and realize the resource utilization of SS, which is conducive to the harmonious coexistence of human and nature. In addition, the amount of SS produced is huge and the production of SS is sustainable. Therefore, the management of SS by manual is not enough. From this perspective, aerobic composting and anaerobic digestion have been implemented on a large scale, which makes them more advantageous than sludge management methods such as thermochemical processes. In fact, there are many ways to dispose and manage sewage sludge in line with sustainable development strategies, and many treatments are better than aerobic composting and anaerobic digestion. However, most of these sludge treatment and management are in the laboratory stage, and financial and technical conditions limit the large-scale promotion of these treatment methods. Therefore, in the future, it is necessary to optimize the management policy of SS according to the available data, and select high-quality SS disposal and management methods, carry out large-scale mechanization operation, and realize the sustainable utilization of SS resources. Most importantly, it is necessary to conduct hazard investigation for SS under different treatment conditions, and then develop appropriate SS management standards on a global scale, because the protection of the global village needs to rely on the joint efforts of all countries and regions.

DECLARATION OF COMPETING INTEREST

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this chapter.

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2 Global Scenario of Sewage Sludge Management

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2.1 INTRODUCTION

Recently due to the increase in population and industrialization, the amount of wastewater produced increased significantly (Figure 2.1), resulting in the production of a huge amount of sewage sludge after the treatment of this wastewater. The residual, semi-solid, or slurry material, a by-product during wastewater treatment (WWT) of various sectors such as municipal or industrial, is called sewage sludge. The amount of sludge from WWT can broadly vary between 35 and 85 g of dry solids/population equivalent/day (Foladori et al. 2010). Sewage sludge generation varies between different countries depending on the population and development stages. For instance, the United States produces more than 210 million tons of sewage sludge annually (around 7 million dry tons), whereas Japan produces 50 million tons (Alam et al. 2007). The annual sewage sludge production in South Korea is much lesser, showing 5.4 million tons by 2025, corresponding to its less territory area (Supaporn and Yeom 2022).

Typical sewage management strategies can be as follows: (i) direct disposal without any recycling or valorization (e.g., landfill, disposal to sea, and dumping); (ii) recycling or reuse by drying and applying in agriculture; and (iii) transformation to energy and other important chemicals by compositing, anaerobic fermentation, and incineration (Shaddel et al. 2019). Disposing of sewage sludge in the ocean was an old practice of disposal, which significantly impaired the ecology of this aqueous environment. For instance, 61% of the sewage sludge generated in Korea was mainly disposed of in the ocean (Korea Ministry of Environment 2013). However, by acknowledging the ocean contamination and disadvantages toward the soil and water inhabitants, such practice was prohibited by the 1996 London Protocol (Choi et al. 2014). Although several advanced treatment technologies have been considered for sludge minimization during the WWT process (WWTP), such as dewatering and stabilization (reducing the fraction of volatile solids), yet the amount of sludge produced is economically questionable.

Recently, due to the changing interpretation of people regarding waste, few advances are happening socially to accept that waste is wealth. Therefore, many companies address sewage as a veritable "black gold." Supaporn and Yeom (2022) stated recycling the sludges for cement and

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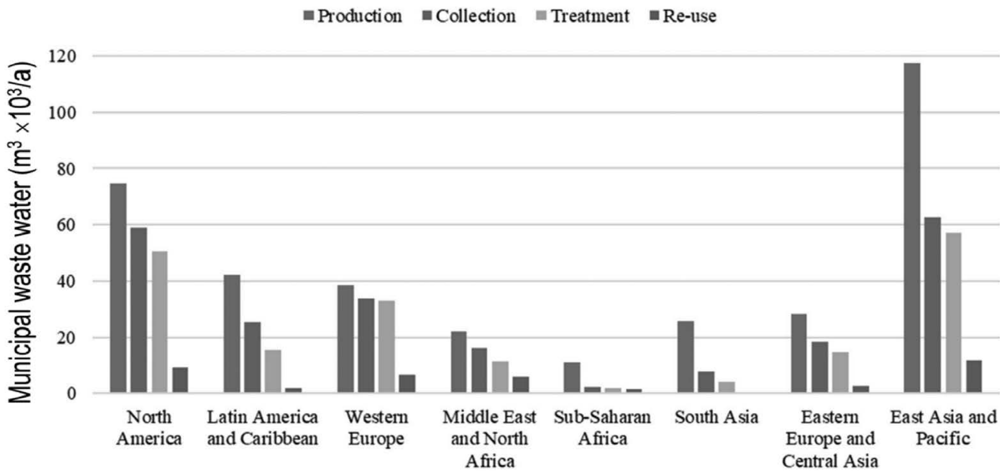

FIGURE 2.1 Global annual municipal wastewater generation and management scenario. (Reproduced from Di Giacomo and Romano 2022.)

construction application and incineration as the most current practices. However, to utilize the valuable energy potential of the organic sewage sludge, the alternative way is through bioconversion using biological treatment processes. In general, the composition and characteristics of the sewage sludge, albite, depend on the source and WWT plant steps (primary and secondary WWT steps). Sewage sludge mainly consists of various macro- and micronutrients, for example, carbon, nitrogen, phosphorus, and potassium. Sludge has been known as a source of energy and nutrients which can be considered an alternative to answering the legislation requirements and the circular economy standards (Gherghel et al. 2019). Typically, the sewage sludge is directed into the anaerobic digestion facilities to produce methane. During anaerobic digestion presence of different microbial diversity, including bacteria and fungi, accelerates the biodegradation process to produce value-added products. Hydrogen is one of the intermediate products during the anaerobic process, which is found to be an important component (Alam et al. 2003). Therefore, many industries are focusing on producing hydrogen as a "clean energy" output from an anaerobic fermentation system. Based on the above facts, this chapter discusses global perspectives of sewage management through traditional techniques and recent valorization options from the perspective of the global scenario.

2.2 TYPICAL SEWAGE SLUDGE MANAGEMENT STRATEGIES

A typical WWT follows primary and secondary processes. Each step follows different process sections to produce the final added value outcome. Figure 2.2 shows a conventional WWTP from the initial stage as screening to several secondary stages, such as dewatering, thickening, and secondary sludge treatment, toward the final product usage. Sewage sludge can be produced in several WWTP and could be classified as treated and untreated sludge. However, regardless of the different types of sewage sludge, they share common composition characteristics. Usually, 95% of the sludges contain water, and the rest composition consists of different proportions of organic and inorganic substances. According to Rulkens (2008), sewage sludge may contain toxic compounds, pathogens, or other harmful microorganisms. Therefore, proper treatment technologies are needed to eliminate such harmful substances before disposing of them to land or other similar applications. Sludge management cost has been estimated to be almost 50% of the total cost of WWT plant operation (Campo et al. 2021). Therefore, scholars define and investigate technologies or propose different approaches to minimizing the production of sludges and treating them that could eventually fall into the concept of circular economy (Collivignarelli et al. 2019).

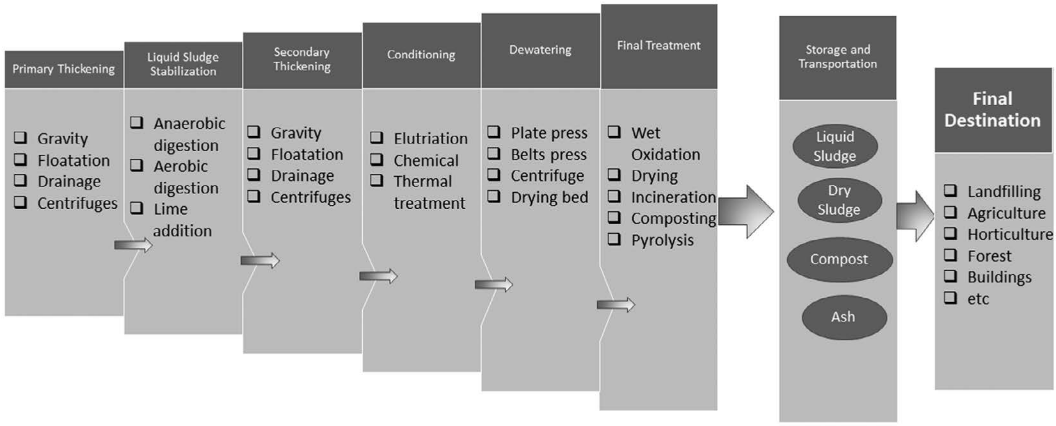

FIGURE 2.2 Standard treatment techniques are used for the treatment of sewage sludge as a way to manage this waste. (Redrawn from Liu et al. 2019.)

The treatment processes used to manage sewage sludge can vary with countries due to the indigenous characteristics of sludge and the feasibility of using a treatment method for that particular country. In 2011, Korea disposed of 61.6% of sludge through ocean dumping, which was later banned under the related London Protocol 1996 (Choi et al. 2014). It presently manages the sludge by both recycling and destructive treatment, which include 24.5% recycling for cement and construction, 24.4% drying for energy generation, 22.3% incineration, and 19% landfill (Supaporn and Yeom 2022). Although these approaches are comparatively more acceptable to ocean dumping, they are still questionable for land usage and environmental pollution. On the other hand, Japan disposed of its sewage sludge through incineration or landfill approaches, while a small fraction of the sludge is used as fertilizer for agriculture (Alam et al. 2007). In the scenario of Europe, data analysis for 2005 and 2011 shows that sewage sludges are mainly managed in these countries through landfill, agricultural use, incineration, composting, and some unknown ways (Figure 2.3). Among these approaches, agricultural usage was predominant in most countries of EU 15 and EU 17, accounting for 40.3–50.5%, while the most widely used approach in EU 12 countries was unknown. Incineration of sludge was the second-most used approach in managing sewage sludge in most European countries. Making Europe's sewage sludge treatment more efficient and circular to meet zero-pollution targets, most of the WWTPs advanced the strategy of cleaning the water and returning it back to the environment. Recent data from 2019 shows that in most EU countries (e.g., Denmark, Spain, Portugal, etc..), more than 80% of the produced sewage sludges are used for agricultural purposes (Gillman 2019). However, the authors emphasize that the

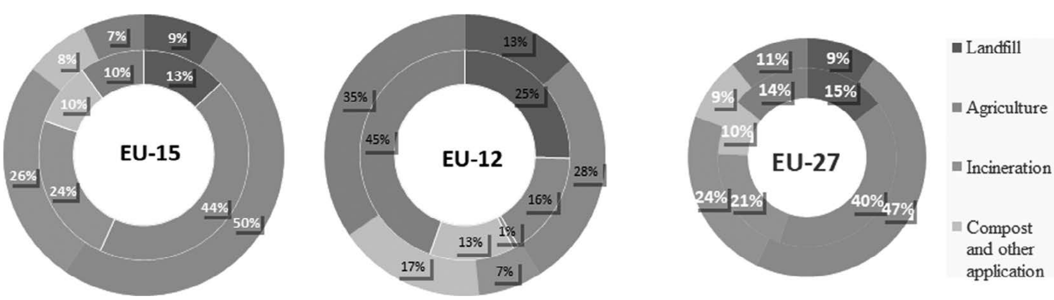

FIGURE 2.3 Common sewage sludge management approaches in Europe (EU 15, EU 12, and EU 27) for 2005 (inner layer) and 2011 (outer layer). (Redrawn based on the data provided in Bianchini et al. 2016.)

percentage situation varies based on local conditions and local policy authorities. Around 55% of the sewage sludge in the United States is used in agriculture, while the remaining 45% is managed by landfill and incineration (Shaddel et al. 2019). Sewage management in China mainly exposes the land to avoid disposal costs. In contrast to the developed countries, non-developed countries, particularly the African and South American countries, are concentrating less on sewage management (Spinosa 2011).

2.3 SEWAGE SLUDGE MANAGEMENT TOWARD THE CIRCULAR ECONOMY

2.3.1 Bioenergy

2.3.1.1 Biogas and Biomethane from Sewage Sludge

Considering waste as wealth with developing and implementing novel approaches to convert it into bioenergy can provide new aspects of renewable energy sources, resulting in minimized greenhouse gas effects. Valorization of sewage sludge in this regard will play an essential role in prospective bioenergy production and meeting the circular bioeconomy concept. In fact, sewage sludge has already been investigated for the production of almost all major bioenergy sources, including biogas/biomethane, biohydrogen, biodiesel, and bio-alcohol (Demirbas et al. 2016; Kargbo 2010; Melero, Sánchez-Vázquez and Vasiliadou 2015; Salameh et al. 2020; Siddiquee and Rohani 2011). In this regard, biogas or biomethane production from sewage sludge would be a promising technology (Kiselev et al. 2019; Mills et al. 2014). Biogas is a mixture of two gaseous components (CH_4 and CO_2), which is used to generate heat and electricity, and as car fuel and fed gas (Zabed et al. 2020). However, the production of biomethane or upgrading biogas to biomethane is considered to be a more favorable alternative to achieve density and a calorific value of this bioenergy comparable to those of natural gas (Paolini et al. 2018; Yang et al. 2019).

Anaerobic digestion is a common and traditional approach for producing biogas or biomethane. For example, batch anaerobic digestion of sewage sludge at 37°C produced biogas with an electric potential of 8.81 kWh (Selamawit and Agizew 2022). However, due to the complex structure (solid-liquid) of sewage sludge with high organic matter content, multi-state treatment is required for the bioconversion of such waste to bioenergy by-products. Moreover, the anaerobic digestion process of sewage sludge is affected by slow biodegradability and limited hydrolysis rate. This, therefore, resulted in various research studies into advancing the technology and improving the biodegradation of sewage sludge. For example, acetogenesis and methanogenesis were improved during anaerobic digestion. Certain anaerobic bacteria such as Geobacteria can use ethanol to provide sufficient electrons for the methanogens to enhance biogas production. In addition to using as the primary substrate, sewage sludge is also a promising substrate to co-digest other wastes (Gundoshmian and Ahmadi-Pirlou; Nghiem et al. 2014). For example, co-digestion of olive pomace and sewage sludge was reported to be promising for improving buffer capacity, stability, and organic loading rate (OLR), with a 39% increase in biogas yield, 40% increase in specific methane yield, and an opportunity to save energy at a rate of 20,328.6 kWh/year (Fragoso et al. 2022). Likewise, co-digestion of organic waste and sewage sludge was reported to significantly improve methane yield with an increase in the chemical oxygen demand (COD) removal of up to 41% (Wickham et al. 2016).

2.3.1.2 Biohydrogen from Sewage Sludge

Hydrogen is considered a clean energy option as its burning only produces water and no other gaseous products responsible for global warming (Hosseini and Wahid 2016). With the growing increase in global sewage sludge generation, there has been a rising interest in using this waste resource as a substrate for biohydrogen production. Waste resources, including sewage sludge, can be converted into hydrogen using various techniques, which are broadly categorized into thermochemical (pyrolysis and gasification), biochemical (dark and photo fermentation, and microbial electrocatalysis), and combined approaches (Wijayasekera et al. 2022). Among these methods, anaerobic fermentation (dark and photo fermentation) of sewage sludge for hydrogen generation is another emerging technology with techno-economic and environmental viability (Yao et al. 2018). Nevertheless, sewage sludge contains organics (carbohydrates, protein, and lipids) that are not directly consumable by the microorganisms, requiring hydrolysis of those polymeric organics into monomeric components. In addition, sewage sludge often contains unwanted compounds, metal elements, and microorganisms, which necessitates a suitable pretreatment method before using it as a substrate for hydrogen production. In this context, several pretreatment methods have been investigated to obtain the fermentable form of sewage sludge under physical, chemical, and microbial contexts. Pretreatment of sludge can expedite digestion of sludge by breaking down and releasing intracellular components of sludge microbes, and selectively killing the non-hydrogen forming microbes as some hydrogen-producing microbes can form spores under the pretreatment conditions (Yao et al. 2018).

In addition to the sludge characteristics-related issues, several factors associated with anaerobic fermentation of sludge have also been reported to affect the process efficiency and hydrogen production. Among these factors, the most pronounced factors are the C/N ratio, pH, temperature, and the presence of metal ions. Considering these facts, anaerobic fermentation of combined alkali and ultrasonication pretreated sewage sludge was carried out under various pH conditions (pH 5–11), which provided a seven-fold increase in hydrogen yield than what was obtained from the untreated sludge (El-Qelish et al. 2020). Another study reported that biohydrogen yield enhanced from 9.1 mL/g DS (dry solids) with untreated sludge to 16.6 mL/g DS with alkali-pretreated sludge (Cai et al. 2004). Because of the inefficient hydrolysis of sludge and sub-optimal C/N ratio as two major factors, sludge was pretreated using various physical, chemical, and biological methods, followed by subjecting the pretreated sludge to dark fermentation under various process conditions (Yang and Wang 2017). The authors reported that hydrogen yield was affected not only by the pretreatment methods and process conditions, but also by the accumulation of some unwanted metabolites (acetic, propionic and butyric acids, and ethanol). Therefore, a second-step conversion of these metabolites to improve hydrogen yield was suggested. In another approach, ethanol was considered the target product to improve overall sustainability through the integrated production of hydrogen and ethanol on the same platform (Han et al. 2011).

2.3.1.3 Bioethanol from Sewage Sludge

In the search for renewable liquid fuels as an alternative to gasoline, bioethanol has been recognized as the most promising alternative (Akter et al. 2020). In fact, the consumption of bioethanol as a transport fuel is now practiced in real applications and has been widely explored in various countries, particularly the United States, Brazil, and European countries. Bioethanol is mainly produced by the fermentation of soluble sugars obtained from sugar juices (e.g., sugarcane and sorghum), starch hydrolysates (e.g., corn, wheat, and potatoes), lignocellulosic hydrolysates, and carbohydrate-rich microalgae (Zabed et al. 2017). However, these sugar sources, particularly lignocellulosic ones, are techno-economically not attractive due to the polymeric nature of carbohydrates that microorganisms cannot consume and remain confined to a complex matrix (Zabed et al. 2019). Although sugars can be obtained relatively easily from sugary juices or starchy feedstocks, they raise further concern about the "food versus fuel" debate. In this context, sewage sludge could be an attractive alternative substrate for bioethanol as it can avoid both of the above issues or limitations.

In one recent study, lipid-extracted sewage sludge was used as the carbohydrate-rich substrate for bioethanol production to maximize the valorization of sewage (Supaporn and Yeom 2022). The lipid-extracted sludge was subjected to dilute acid treatment under optimum conditions (120°C, 5.9%v/v H_2SO_4, and 85 min), which provided a sugar solution containing 5 g/L sugars (75.5% yield). The yeast fermentation of obtained sugar solution produced 1.8 g/L of ethanol. In another recent study, sewage sludge hydrolysate was prepared by subjecting the sludge to *Bacillus flexus* treatment, followed by fermentation of the hydrolysate with yeast under various process conditions, which produced up to 40 mL/L of ethanol (Manyuchi et al. 2018). Another study reported that yeast fermentation of sewage sludge can simultaneously produce ethanol (9.8% in 48 h) and remove COD, Cu, and Cr by 62, 68, and 45 percentage, respectively, in the treated sludge after 72 h of fermentation (Alam et al. 2007). This strategy can open a new gateway for sewage sludge's cost-effective management and valorization.

2.3.2 Resource Recovery as Value-Added Products

Sewage sludge consists of 61% protein content. Since a significantly large portion of the dry weight of bacterial cells consists of protein, recovery of such material could be used as an animal feed application. Sustainable food protein production can be useful for geographical areas where agricultural food is scarce (Linder 2019). According to Pikaar, Matassa, and Bodirsky (2018), protein food is expected to be replaced by crop food by up to 19% by 2050. In order to successfully extract the vital microorganisms from the sludge, understanding the synergistic interaction between different microorganisms and their development in various stages of WWT is vital. Figure 2.4 shows the potential of growing different types of microorganisms on a typical activated sludge derived from sewage WWT plant. The foremost important factor in developing different types of microorganisms is the initial waste source entering the WWTP. Basically, the wastewaters with high nitrogen contents tend to enrich more microbial communities and hence accumulate larger protein concentrations (Webster and Lim 2002).

In other words, the initially available COD and TKN ratio of 5:1 is considered a satisfying fraction to meet efficient bacterial growth. For example, poultry waste is a good example of high nitrogen content and a potential resource for protein recovery. Several other factors that could affect the efficient microbial growth in the system include the nutrient bioavailability, biodegradability, and potential toxicity of the component during the WWTP (Hülsen et al. 2018).

Different treatment methods, such as biological technologies (anaerobic digestion and fermentation), accelerate the solubilization and release of protein compounds from the sludge. Chen et al. (2022) suggested that high alkaline pretreatment (pH 12) is the most effective method to achieve maximum protein solubilization as compared to other physical-chemical processes (ultrasonic, thermal treatment, etc.). The authors reported that fermentation of chicken manure having extreme

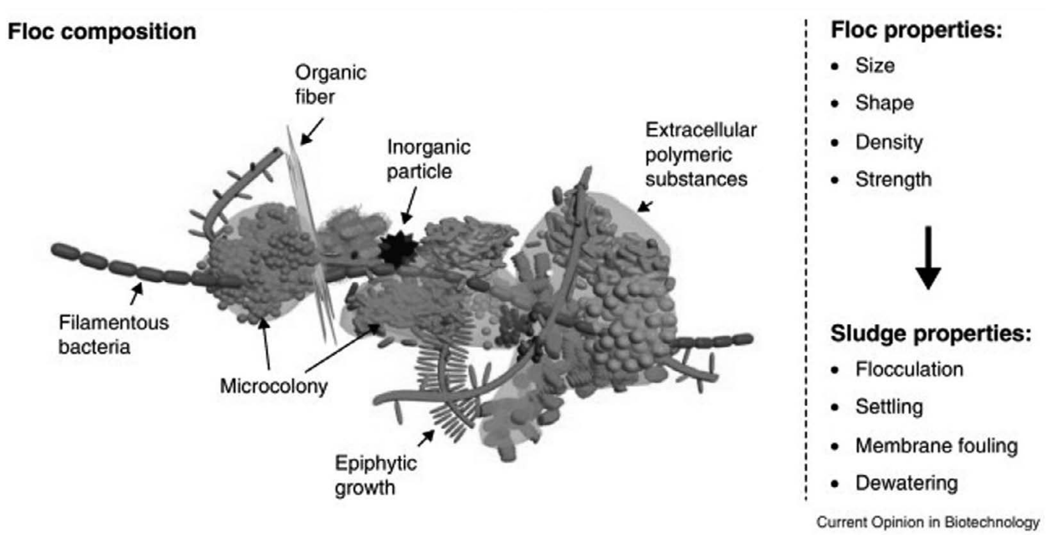

FIGURE 2.4 Schematic picture of microbial growth on the activated sludge. (Adapted from Nielsen et al. 2012.)

alkaline pH is a promising technology for obtaining highly valued by-products. Eventually, the authors suggested that high pH biowaste can inhibit the methanogenic activity of microorganisms, producing high-valued volatile fatty acid chains in the system. Different protein by-products derived from the fermentation process include various volatile fatty acid chains, namely acetic acid, butyric acid, and propionic acids According to the global petrol price (Valev 2022), VFA components share more than 2000 €/ton in the EU market, which is higher than the value of methane production (1.8 €/m^3) from the waste. It is then regarded that the production of VFA through anaerobic fermentation is a promising roadmap to the circular economy.

Different scholars have extensively studied and compiled data based on VFA production from different waste sources. However, investigation on deriving VFA from sludge effectively is still under investigation. The challenge lies in the low hydrolysis rate and biodegradability of waste sludges (Lee et al. 2014). In line with the previously discussed method, the recent idea adopted by Chen et al. (2022) introduced a novel idea of changing the redox-balance strategy during the fermentation process to achieve higher butyric acid production as a front-line research idea to overcome this challenge.

2.3.3 Resource Recycling as Fertilizer

Moreover, sludge is enriched with different micro- and macronutrients that can be applied to the fields as a fertilizer. The appropriate land application will increase the stability of agricultural production. Additionally, the application of sludge for the growth and development of bioenergy crops will be in line with the renewable energy directive policies, which emphasize the significant growth of such crops, which could produce a minimum of 20% of total energy (Urbaniak et al. 2017).

However, the direct application's main drawback is the possibility of sludge contamination. Depending on the source, the sludge might contain toxic elements, heavy metals, and pathogens harmful to crop production. Mainly it will let to soil and groundwater contamination, deteriorating the vegetation. Therefore, composting technology is the best solution answer to this. During the composting process, sludges are subjected to a high-temperature degradation process (>55°C) that enriches the anaerobic microorganisms. Moreover, it will disinfect the sludge from any possible contamination. It is important to note that, even though composting can be used as a safe fertilizer, it still does not fulfill the EU's safe disposal regulation due to the inert characteristics of several pollutants present in the sludge, such as polychlorinated biphenyls (PCBs), polynuclear aromatic hydrocarbons (PAHs), and pharmaceutical compounds, (Seleiman et al. 2020).

Literature studies show that vermicomposting (interaction between earthworm and the soil microbes) can effectively eliminate antibiotic resistance and pathogenic human bacteria from the sludges. A study reported by Huang, Xia, and Zhang (2020) shows that earthworms are capable of reducing the abundant human pathogenic bacteria by promoting the growth of proteobacteria and actinobacteria during the vermicomposting process. Moreover, Kui et al. (2020) published exciting results based on biochar application during the vermicomposting process. The authors concluded that adding biochar can significantly decrease the high content of antibiotics and the corresponding resistance genes (ARGs) from the sludge. A recent study by Domínguez et al. (2021) supports that earthworms are able to eliminate most of the microbial communities present in the sewage sludge during the vermicomposting process. The authors confirm that converting sewage sludge into safe vermicompost is considered a sustainable agricultural practice.

2.4 GLOBAL SCENARIO OF SEWAGE MANAGEMENT THROUGH THE CIRCULAR ECONOMY CONCEPT

Sludge as a source of energy considers a good alternative for energy resources. Hence, it fits the legislation consideration toward circular economy approaches. The circular economy action plan (CEAP) was adopted in 2015 by European Commission to cover the whole cycle from production, consumption, and waste production and close the loop by redesigning the process and achieving economic and environmental profit. The information on the total sludge production is incomplete. Eurostat web has updated the total production estimation until 2020, showing the Netherlands, Poland, and Austria as countries with higher annual sludge production (Eurostat 2022).

Although all the policymakers, stakeholders, and industry advisors agree that sludge could be a potential source of applying the circular economy, conflicts in the implementation and proposing coherent strategies have often been controversial. For example, the Swedish government first applied a total ban on sludge land application in 2018, with the approach of phosphorus recovery. However, in 2020, the law again favored the reapplication of sludge on agricultural lands (Ekman Burgman 2022). The conflicts generated as the level of heavy metal and micropollutants detected in the sludge could fluctuate. Therefore, strategic plans to reduce the micropollutant concentration directly affect the rules and legislation on implementing proper acts.

Most importantly, "re-thinking" how to use the resources with safe disposal characteristics should be the core of the decision-makers to create a winning sustainable circular economy (Smol et al. 2020). Kacprzak and Kupich (2023) comprehensively discussed the necessity of proper technology selection and implementation of a circular economy in Poland as a case study. The authors discussed decentralization as an effective technique to achieve positive environmental, economic, and social determination. On the other hand, the transformation of biowaste production into energy and biorefineries is equally essential and falls into the EU's circular economy plan's direct action plan. Germany (8 Mtoe), the United Kingdom (2.3 Mtoe), and Italy (1.9 Mtoe) are among the countries with the highest biogas production out of waste generated to meet both the aspects of GHG reduction and circular economy (EurObserv'ER 2016).

In the end, although many organizations tend to tackle the increasing waste generation and global warming problems by providing different green policies, an important question is "Is the green deal a global strategy?" (Smol 2022). The author of this impactful study (Smol) systematically discussed the present scenario of global progress toward achieving the green deal approach. However, a logical fact has been discussed: the world's uncertainties in facing unexpected crises such as COVID-19. Such elements could directly affect the policymakers and industries to revise their action plans and national strategies to meet the circular economy of that time. Taking advantage of the COVID-19 crisis forced the legislation to re-think and rebuild tactics to more resilient plans. The European Green Deal and the CEAP organization agree with the concept of resource recovery and recycling. However, public awareness and education highlighting social and socio-political aspects are vital to reaching a successful ground plan.

2.5 CHALLENGES AND RESEARCH NEEDS

Activated sludge is a widely accepted process for the treatment of sewage and effluents from various industries (Håland 2019). It can provide advantages such as low cost, easy operation, and high efficiency for treating biodegradable organic compounds (Kim et al. 2020). However, there are drawbacks to the application of these methods such as the generation of relatively high amounts of sludge waste that needs further treatment (Kamali et al. 2019). Some techniques, such as incineration, have been widely used for energy extraction from such waste materials. However, such a process results in air pollution and is highly avoided in European countries (Caprai et al. 2020). Land application is also another approach adopted to deal with the sludge produced in activated sludge processes to improve the properties of the soil such as total organic content and water holding capacity required for specific applications such as agriculture (Piao et al. 2016).

However, novel technologies have been developed in recent years for the further valorization of the produced sludge such as extracting energy using technologies such as anaerobic digestion (to yield biogas) or converting the generated sludge to valuable compounds (such as carbonaceous materials) for a wide range of applications such as (waste)water treatment and energy storage (Li, Champagne and Anderson 2013; Yu et al. 2004). Nevertheless, there are issues with the application of such novel technologies. For instance, promoting the anaerobic digestion processes requires a relatively high C/N ratio leading to the optimization of the anaerobic digestion processes. However, in most sewage sludges, there is a relatively low C/N ratio which makes the anaerobic digestion process difficult to succeed (Yue et al. 2009; Zeshan and Visvanathan 2012). To overcome this issue, approaches such as co-digestion of the sewage sludge with those from other sources have been considered. There are examples of such a strategy such as co-digestion with agricultural wastes (Solé-Bundó et al. 2017). In this regard, a possible strategy is to supplement the anaerobic digesters with the wastes from neighboring industries which can potentially reduce the overall treatment costs by minimizing the transport costs. In this regard, there is a need for studies to have a conclusion on the feasibility of mixing different types of waste materials to achieve the optimum performance of the system in terms of methane production and treatment efficiency of the system. There is also a need for comprehensive studies to identify the C/N rations and other potential inhibitory elements present in various sewage sludge compositions. Other parameters can promote such technologies for real applications. The quantity of the sludge produced in sewage sludge systems needs to be compatible with the waste materials from other sources to provide the required C/N ratio. The quantity of sewage sludge in different countries also depends on the specific technologies implemented and the pollution load in the composition of the effluents to be treated using such technologies.

There are also other approaches for the valorization of the sewage sludge toward producing sustainable energies, such as bioelectricity, using relatively novel technologies, for instance, microbial fuel cells (MFCs). The current trend in the scientific community is to develop efficient MFC components such as anode, cathode, and proton exchange membranes to enhance the generation of bioelectricity. Optimizing the microbial communities can also enhance the total yield of such processes and make this approach more viable for real applications in the treatment of sewage sludge (Liu et al. 2011; Miran et al. 2018; Yuan et al. 2013).

Other viable approaches can also be adopted in order to valorize the sewage sludge into valuable products. There are recent studies on the conversion of sewage sludge with relatively high carbon content to carbonaceous materials such as biochar through the pyrolysis process (Racek et al. 2020; Yuan et al. 2013). Such a process can aid in achieving the goals of the circular economy by providing valuable materials as the starting point of the supply chain of other processes. Such low-cost and sustainable materials can replace conventional materials used for various applications. For instance, they can be used for the adsorption of pollutants from polluted (waste)waters or soil content instead of relatively expensive materials such as clay-based compounds (e.g., bentonite) (Liu et al. 2021; Paranavithana et al. 2016). State-of-the-art technologies offer the preparation of catalytic biochar materials which can be used for the degradation of pollutants instead of removing them using technologies such as adsorption. There are also trends in the literature for the application of the prepared carbonaceous materials from sewage sludge for the fabrication of electrode materials that can be used in electrochemical-based techniques. Such electrodes can also be used in MFCs for the generation of electricity from sewage sludge and to satisfy the circular economy considerations.

Recent studies have also demonstrated that such materials can be used efficiently for the improvement of soil properties for crop production. The addition of such materials has been demonstrated with the potential to improve soil properties such as pH and water holding capacity with the potential of adding the chemicals needed for the growth of the plant species such as nutrients and organic and inorganic carbon compounds. This strategy can also be used to deal with the current global environmental issues such as global warming and climate change. This is achieved by sequestration of carbon in the content of the sludge-derived materials and preventing them from being released into the atmosphere by using conventional ways of dealing with the sludge wastes such as incineration (Abbas et al. 2018; Gonçalves et al. 2018; Xiang et al. 2020).

There are also approaches for the conversion of such materials to other compounds such as activated carbon, with a wide range of potential applications. The pyrolysis process is a well-known approach for converting such feedstock materials to biochar materials which still need a high input of energy. In this regard, sustainable technologies need to be developed to provide the energy needed for such energy-intensive processes using renewable energy resources such as solar or wind energy.

2.6 CONCLUSIONS AND PERSPECTIVES

Wastewater generation is one of the growing management crises worldwide, with the amount of wastewater generated in a country being directly related to its population. The WWT is a common approach to recycling or reusing this large water body that generates huge amounts of sewage sludge. With increasing urbanization and industrialization, the volume of sewage has also increased dramatically worldwide. Different countries, especially developed countries, have taken different measures to manage the sludges, including direct disposal, recycling or reuse, nutrient recovery, and transformation to energy and other important chemicals. Among these approaches, the use of sludge components as feedstock for the production of advanced materials and energy is becoming increasingly attractive. Such approaches can either stabilize carbon prevent global warming and climate change and decrease the reliance on fossil-based energies by generating green and sustainable energy resources for the circular economy.

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3 Ecological and Health Risk Assessment in Heavy Metal Contaminated Soils Irrigated by Sewage

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3.1 INTRODUCTION

Due to the low sewage treatment capacity, high sewage treatment costs and the weak environmental awareness, much untreated or lack of necessary pretreatment of sewage is used for the irrigation of farmland, which the water quality seriously exceeded the standard (Ashraf et al., 2018; Meng et al., 2016). Generally, the sewage contains the organic and inorganic pollutants. The organic pollutants in sewage can be divided into two categories: one is carbon organic matter, such as cellulose, hemicellulose and xylogen. And the other is the nitrogenous organic matter, such as protein and amino acid. The inorganic pollutants in sewage mainly include the metal elements, such as lead, zinc, copper and nickel, cadmium and mercury (Hui et al., 2022a; Yan et al., 2020). Especially for cadmium and mercury, they are easy to enter the food chain, which is not easy to be noticed and is worthy of attention. Non-metallic elements include boron, selenium, arsenic, sulfur and their compounds, oxides and salts (Hui et al., 2022b). Hence, the blind wastewater irrigation has led to the accumulation of toxic metals and harmful substances in soil in some areas, and the watered soil has been polluted to different degrees (Fu et al., 2021; Hu et al., 2022). The texture of soil is deteriorating, and the soil is even abandoned, with a serious threat to human health (Sayo et al., 2020; Yang et al., 2021). The main reason for this phenomenon is that people see its considerable “the water and fertilizer effect” but despise the environment pollution problem caused by the sewage irrigation. At the same time, pollutants can also enter the human body through the food chain, resulting in the pathological reactions and thus affecting the health of populations (Oubane et al., 2021; Song et al., 2022).

Kiani et al.’s (2022) study investigated the rate of accumulation, human health risk assessment and nitrate-related transfer factor in vegetables irrigated with different sources, including treated wastewater effluent from Kermanshah wastewater treatment plant, Gharasoo River water of Kermanshah, and well water with chemical fertilizer. Meanwhile, Jalil et al. (2022) explored the impact of wastewater irrigation on the severity of heavy metal pollution and health risk potential in different vegetables under wastewater irrigation versus fresh water irrigation. Meanwhile, many earlier studies reported the long-term effect of the sewage irrigation on the heavy metal contents in soils, plants and groundwater and hazard quotient (HQ) to assess the risk of heavy metals on human health through the consumption of green leafy vegetables grown on the sewage-irrigated soil (Rattan et al., 2005; Singh and Ramteke 2012).

Although the research on the risk analysis of sewage irrigation started late, there are already many methods of the risk assessment through the continuous deepening of research. However, many unilateral studies are conducted in one specific field, and there is little research on the

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comprehensive environmental risks caused by sewage irrigation. In the current research on the risk of sewage irrigation, researchers have only carried out short-term purposeful artificial experiments to draw their own conclusions but have not carried out systematic risk identification, that is, there are few profound understandings of various risks caused by sewage irrigation. There are few studies on the severity of all kinds of risks caused by sewage irrigation, and only the available years of soil and groundwater are studied. There is no profound quantitative analysis of the environmental risk degree after sewage irrigation. Thus, this chapter summarizes the ecological risk assessment methods caused by sewage irrigation to provide reference and the basis for the correct guidance of farmland sewage irrigation and expects to obtain the maximum environmental security with the largest trend of sewage irrigation.

3.2 THE BASIC DEVELOPMENT PROCESS OF HEAVY METAL RISK ASSESSMENT IN SEWAGE IRRIGATION SITES

3.2.1 Development History of Health Risk Assessment of Heavy Metals

Health risk assessment was first applied to medicine. Later, people gradually found that pollutants in the environment would also cause harm to human health. They gradually combined the exposure of pollutants in the environment with the harm to human health. In the 1950s, the safety factor method was used to calculate the maximum allowable intake of pollutants. In the 1960s, the safety dose was used to evaluate the risk of pollutants. Until 1976, the environmental protection agency of the United States first proposed the evaluation method of carcinogenic risk caused by carcinogenic pollutants. In 1983, the four steps of health risk assessment published by the American Academy of Sciences were widely recognized in the world and should be used in health risk assessment in other countries. In the 1990s, scientists studied the exposure parameters of pollutants in the environment and combined human health risks with the environment to conduct regional health risk assessments. It was also at this time that the heavy metals in the environment began to be linked with human health (VanBuren et al., 2022). In China, the research on soil health risk assessment was carried out relatively late. It was not until 2014 that the Ministry of environmental protection issued the first health risk assessment standard combined with the environmental situation of China – Technical Guidelines for risk assessment of contaminated sites (hereinafter referred to as the “guidelines”). The “guidelines” specify in detail the assessment models that should be selected when conducting health risk assessment in various polluted areas, and at the same time, according to the actual environmental conditions in China, the values of parameters of some new pollutants under different exposure paths are proposed. Due to the late appearance of the “guidelines”, the health risk assessment of pollutants in China mainly uses the assessment models proposed by foreign countries (MEEC, 2014).

3.2.2 Development History of Ecological Risk Assessment

The risk assessment of heavy metal soil under sewage irrigation is included in the ecological risk assessment. It is a new environmental risk assessment method developed in the 1980s. It is a model that applies quantitative methods to assess the possible risks of various environmental pollutants (including physical, chemical and biological pollutants) to biological systems other than human beings and to assess the acceptability of the risks, mainly including the raising of problems, exposure assessment, Ecological effect evaluation and risk characterization. So far, it has mainly experienced the embryonic stage of taking environmental risk as the evaluation content. The risk sources at this stage are mainly based on the possibility analysis of unexpected events. There are no clear risk receptors, let alone clear exposure assessment and risk characterization methods. The whole evaluation process is usually based on simple qualitative analyses. Subsequently, ecological risk assessment developed into a development stage with toxicological assessment and health assessment as the main assessment contents. The risk assessment at this stage is mainly aimed at the environmental risk assessment of chemical pollution. Most of the risk receptors are human health, and the assessment of human health is mainly focused on the carcinogenic risk. The third stage is a major development stage based on the establishment of national ecological risk assessment frameworks and guidelines. This stage has developed from the first attempt to adapt the human health risk assessment framework into the ecological risk assessment framework to the formal release of the ecological risk assessment framework by USEPA, providing a complete process of ecological risk assessment. A relatively complete ecological risk assessment framework has been formed. Since the end of the 1990s, the ecological risk assessment has been in a large development stage, and it mainly conducts large-scale comprehensive ecological risk assessment research.

3.2.2.1 USEPA Ecological Risk Assessment

The concept of ecological risk assessment was first proposed by USEPA in 1990, that is, after the ecosystem is affected by one or more stress factors, the possibility of forming adverse ecological effects is assessed. Ecological risk assessment in the United States is developed on the basis of human health risk assessment. In 1990, USEPA formally put forward the definition of ecological risk assessment. After eight years of discussion, revision and improvement, in 1998, USEPA formally promulgated the Guidelines for Ecological Risk Assessment.

3.2.2.2 Ecological Risk Assessment of Other Countries

Canada promulgated the ecological risk assessment framework in 1996. The European Union issued the technical guidance document on risk assessment in 2003. The Dutch Ministry of housing, natural planning and environment (NMHPPE) also proposed the Dutch risk management framework in 1989. The core content of this framework is to apply thresholds to determine whether the risk level can be accepted. Subsequently (1995), the environment department of the United Kingdom also proposed for the first time that all environmental risk assessment and risk management must comply with the national sustainable development strategy. It is intended to emphasize that if there are serious environmental risks, even if the current scientific evidence is insufficient, it is necessary to take preventive measures to mitigate the potential risks. In 1999, the National Environmental Protection Commission of Australia also established a relatively complete set of guidelines for soil ecological risk assessment. Part B5 of the guidelines is the topic of ecological risk assessment. The commonly used method in foreign ecological risk assessment is the probabilistic risk analysis method. The risk assessment uses the comparison of exposure distribution/effect distribution in the probabilistic risk assessment (PRA) method to evaluate the research site. The exposure distribution adopts the measured environmental concentrations (MEC) distribution of pollutants in the soil, and the point estimation of the effect adopts the repair guide value of the corresponding substance, and the risk is characterized by the probability that the measured concentration distribution exceeds the repair guide value.

In 2011, China issued the first official guidance document for ecological risk assessment, the guidelines for risk assessment of chemical substances (Draft for comments). The work flow of risk assessment of contaminated sites is divided into five parts: hazard identification, exposure assessment, toxicity assessment, risk characterization and control value calculation.

3.2.3 Basic Process of Risk Assessment of Heavy Metal Soil under Sewage Irrigation

The ecological risk assessment of heavy metal soil under Sewage Irrigation generally adopts the Guidelines for Ecological Risk Assessment issued by USEPA in 1998 – The "three steps" of ecological risk assessment, namely problem formation, problem analysis and risk characterization, are clearly proposed in. Later, it was revised as the assessment guide, which divided the risk assessment into three stages: (i) Problem description, determination of assessment scope and formulation of plan; (ii) problem analysis is carried out from two aspects: heavy metal exposure characterization and ecological effect characterization; (iii) risk characterization, possibility analysis and uncertainty analysis of ecological hazards (Yuan et al., 2005), as shown in Figure 3.1.

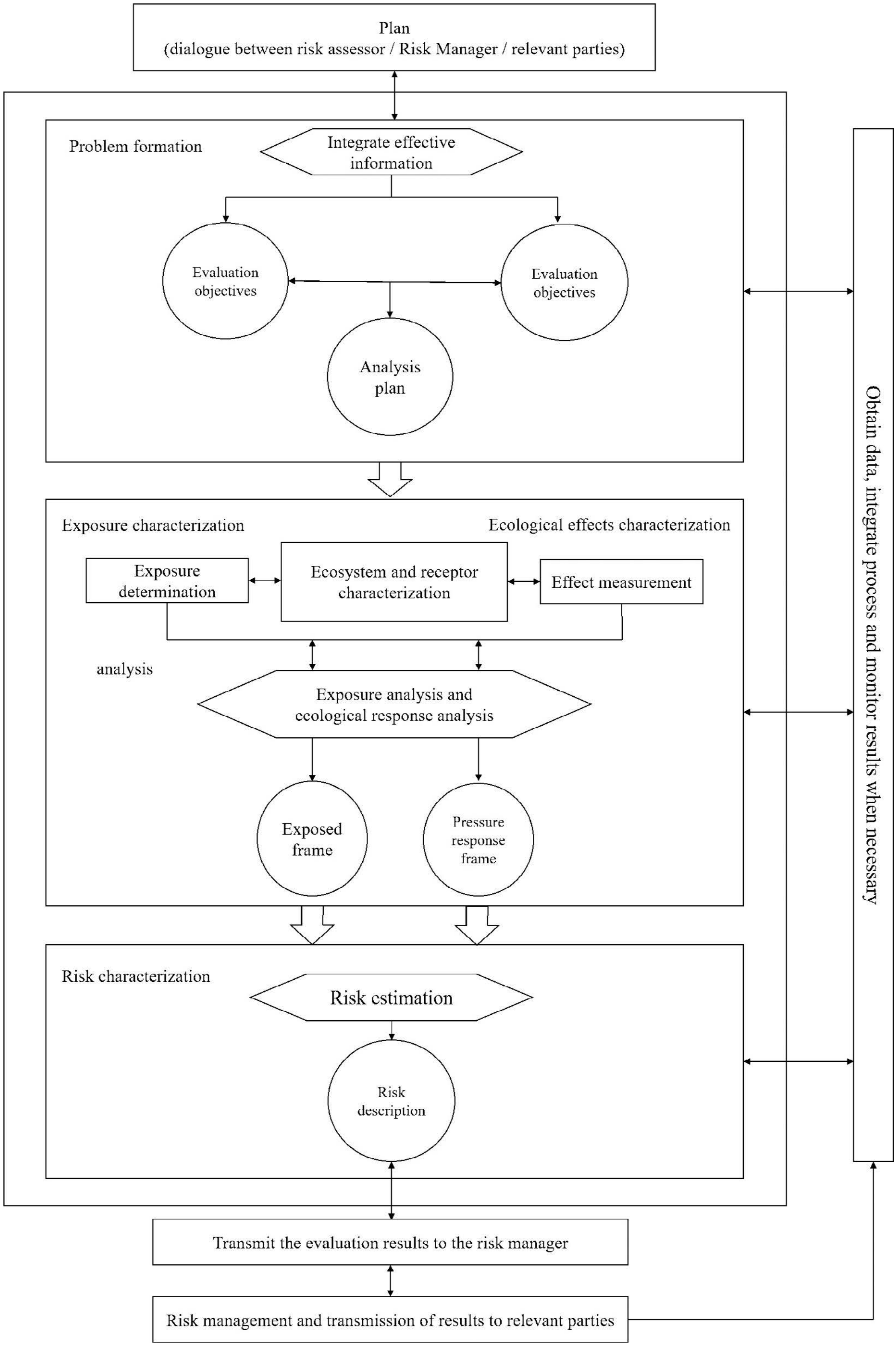

FIGURE 3.1 The technical portal of risk assessment of heavy metal soil under sewage irrigation.

3.3 ECOLOGICAL RISK ASSESSMENT

3.3.1 Methods

Risk classification method is a qualitative characterization method for ecological risk assessment of toxic and harmful substances proposed by the European Community. The potential ecological risks of pollutants are compared intuitively by formulating grading standards. The comparative risk evaluation method is proposed by USEPA, which compares the relative risks of a series of environmental problems through expert judgment and gives the final ranking conclusion. Quantitative characterization generally calculates the degree or probability of adverse effects through models or other methods. The main methods are probabilistic ecological risk assessment and HQ.

In probabilistic ecological risk assessment, each exposure concentration and toxicology data is regarded as an independent observation quantity, and its probability significance is considered on this basis. Exposure evaluation and effect evaluation are two important evaluation contents. The characterization result is not a specific value but is given in the form of risk probability.

Multi-level risk assessment method is a qualitative and quantitative characterization method. In the process of risk characterization, it integrates the probabilistic ecological risk assessment and HQ and makes full use of various methods and means to carry out risk evaluation from simple to complex. The multi-level evaluation process starts with a conservative assumption and gradually transits to a more realistic estimation. It generally includes four levels: preliminary screening of risks, further confirmation of risks, accurate estimation of risks and further effectiveness study of risks. Based on this theory, Weeks and Comber (2005) proposed a "cascade" evaluation framework for ecological risks of soil pollutants, and Critto et al. (2007) developed a decision support expert system for ecological risk assessment of environmental pollution based on the cascade ecological risk assessment framework.

3.3.1.1 Ecological Risk Assessment in Polluted Areas

Ecological Simulation

Microcosm and mesocosm ecological simulation are techniques that apply small- or medium-sized ecosystems or laboratory-simulated ecosystems for experiments based on multi-species testing. In this technique, ecological risk assessment at a regional ecosystem level is achieved by defining an acceptable effect-level endpoint (HC_5 or EC_{20}). It can simulate the biodiversity of the ecosystem and the whole life cycle of the representative species and can characterize the indirect effects of competition and food chain interaction between species under the action of stress factors. In addition, it can also predict the migration, transformation, destination and overall effect of chemical pollutants on the ecosystem (Chu et al., 2022). The disadvantages are that it is expensive to run and the selected test species may not be representative of the entire ecological environment.

Exposure Assessment

The exposure assessment of regional ecological assessment is relatively difficult to carry out, because risk sources and receptors have the characteristics of spatial differentiation, different types and levels of impact will compound superposition, which makes the relationship between risk sources and receptors more complex.

Hazard Assessment

Hazard assessment is the core of ecological risk assessment. Its purpose is to determine the damage degree of risk source to risk receptor and regional ecosystem. For organisms, hazard assessment is toxicity assessment, which studies the relationship between risk pressure, such as the concentration of hazardous substances and the response of receptors, and the degree of harm to receptors under what concentration and for how long. When the receptors are extended to high levels, such as communities and ecosystems, the impact of risk pressure is evaluated according to the actual situation.

Risk Characterization

Risk characterization is a comprehensive stage of ecological risk assessment, which refers to the potential adverse effects of risk pressure on the ecosystem or its components, such as individual organisms, populations and communities or the judgment and expression of the magnitude of such potential adverse effects. The ecological risk value is the statistical analysis of the regional ecological risk loss (Zhang et al., 2022).

Ecological risk value is the statistical analysis of regional ecological risk loss, which includes the intensity and frequency of risk sources, the characteristics of risk receptors, the harm of risk sources to receptors and other information, while risk value is the synthesis of these information indicators. The risk value is used to measure the risk intensity of the risk source and the loss degree of the receptor:

$$R = P \times D + Q$$

where R is the risk value of the receptor; P is the risk degree of physical pressure; D is the potential ecological loss index of habitat system receptors; Q is the quotient of hazardous substances. In the study of regional ecological risk assessment, each patch is subjected to the superposition of different types and levels of risk sources.

The acceptor is the risk taker. In ecological risk assessment, it refers to the component of the ecosystem that has been or may be adversely affected by a certain pollutant or other stress factors. Physical pressure in a region mainly causes ecological loss by destroying and destroying the habitat of target organisms. Therefore, the risk caused by regional physical pressure can be evaluated by taking the habitat ecosystem as the acceptor. Ecological index, ecological fragility index and potential ecological loss index can be used to calculate the ecological risk value of different types of patches.

i. Potential ecological loss degree index (D)

The potential ecological loss degree index refers to the degree of difficulty and possible ecological loss of the recipient to be damaged by risk pressure. The index of potential ecological loss refers to the synthesis of ecological index and ecological fragility index in each patch. The formula for calculating potential ecological loss index is as follows: potential ecological loss refers to the degree of difficulty and possible ecological loss of the receptor under risk pressure. Potential ecological loss index refers to the combination of ecological index and ecological vulnerability index in each patch (Yuan et al., 2005). The calculation formula of potential ecological loss index is

$$D_i = E_i \times F_i$$

where D_i is the index of potential ecological loss in patch i; E_i is ecological index of patch i; F_i is the ecological fragility index of patch i.

ii. Ecological index (E)

The ecological index reflects the ecological integrity, ecological importance and naturalness of each patch. In the assessment of regional ecological risk, there are three indexes to measure ecological index: species origin index, biodiversity index and naturalness index. Species protogenicity index is expressed as the percentage of the number of native species in a patch to the total number of species in the patch:

$$O_i = C_i / C$$

where O_i is the species protogenicity index of patch i; C_i is the number of native species in patch i; C is the total number of species in the patch. The biodiversity index is expressed by the proportion of species in a patch to species in the whole area:

$$V_i = N_i / N$$

where V_i is the biodiversity index of patch i; N_i is the number of species in patch i; N is the number of species in the whole region.

The degree of naturalness is negatively correlated with the disturbance intensity, which represents the interference effect of human beings and can be expressed by the length of corridors (highways, ditches etc.) in a patch per unit area:

$$D_i = L_i \,/\, S_i$$

where D_i is the interference intensity; L_i is the total length of corridors (roads, railways, ditches) in the patch i; S_i is the total area of patch i. Then Z_i=1/D_i represents the naturalness of patch i. Three indexes, O_i, V_i and Z_i, can be calculated according to the earlier formula and then normalized and weighted to synthesize the ecological index of each patch:

$$E_i = aO_i + bV_i + cZ_i$$

where E_i is the ecological index of patch i; a, b and c are the weights of each indicator, $a + b + c = 1$.

iii. Ecological fragility index (F)

The fragility of the landscape or ecosystem is the result of the interaction and influence of various environmental factors. Under certain climatic conditions, the fragility of the ecological system of each patch in the region is mainly manifested in the aspects of topography, vegetation degradation, reduction of biological productivity, water and soil loss and reduction of soil quality. Generally, the ecological fragility can be reflected by investigating the vegetation status and soil properties. The calculation formula of ecological fragility is

$$1 \,/\, F_i = \frac{F(A)a_1 + F(B)a_2 + F(C)a_3 + F(D)a_4 + F(E)a_5}{\sum_{i=1}^{5} a_i}$$

where F_i is the ecological fragility index of patch i, and the greater the F_i value, the greater the ecological fragility of the patch i; $F(A) \sim F(E)$ is the dimensionless value of items $A \sim E$, and $F = 1$ is the maximum value; $a_1 \sim a_5$ is the weight of items $A \sim E$. The risk degree can be calculated from the occurrence probability/rate of physical event risk (landslide, drought, flood, human development interference etc.) and the event risk intensity. The risk value can be obtained by multiplying the risk degree with the potential ecological loss degree.

Assessment Endpoint

The relationship between the assessment endpoint and the ecological risk assessment depends on the degree to which they reflect the sensitive ecological integrity. It is the special typical harm or potential harm of environmental stress factors to a receptor. According to the characteristics of the evaluation environment, appropriate indicators should be selected as the evaluation endpoint, which can reflect the value of environment protection. Therefore, the indicators with social value, biological value, sensitivity to risk factors, operability and easy to predict and measure should be selected as the assessment endpoint.

In regional ecological risk assessment, receptors are affected by multiple risk pressures, and risk degree index can be used to measure the characteristics of risk sources:

$$P = \sum \beta_j \times P_j + \sum \zeta_i \times P_i$$

where P is the risk degree; P_j is the occurrence probability/rate of type j risk (such as landslides, subsidence, drought, flood and human development disturbance); β_j is the weight of type j risk; P_i is type i risk intensity (such as pesticides, heavy metals and other toxic and harmful substances), and the risk degree of harmful substances is determined through toxicological test, which is the ratio of the concentration of harmful substances to the concentration of safety threshold; ζ_i is the weight of type i risk.

3.3.1.2 Assessment Factor

When few toxicity data are available, the assessment factor is usually used to assess the predicted no effect concentration (PNEC), which is the acute toxicity data or chronic toxicity data of a species (usually obtained by dividing the acute toxicity data and the ratio of acute to chronic toxicity, ACR) by a factor to obtain PNEC. The determination of its factors mainly depends on the quantity and quality of toxicity data available for the most sensitive organisms, such as the number of species, test endpoint and test time. AF usually ranges from 10 to 1000. The assessment factor is relatively simple, but there is great uncertainty in the selection of factors.

3.3.1.3 Hazard Quotient

Due to the simple application of HQ, most of the current quantitative or semi-quantitative ecological risk assessment is based on it. The HQ method is suitable for assessing the toxicological effect of a single compound and to calculate the HQ by comparing the actual monitored or model-estimated environmental exposure concentration (EEC or PEC) with the toxicity data representing the hazard degree of the substance (predicted no effect concentration, PNEC). A ratio greater than 1 indicates risk, and the higher the ratio, the greater the risk. A ratio of less than 1 is considered safe, and reference doses and baseline toxicological values for various chemicals are widely used. The HQ method is usually conservative in the determination of exposure amount and the selection of toxicity reference value. It is only a rough estimate of risk, and there are many uncertainties in its calculation (Kulikowska and Gusiatin, 2015).

3.3.1.4 Species Sensitivity Distribution

When more toxicity data are available, species sensitivity distribution (SSD) can be used to calculate PNEC. It is assumed that the acceptable effect level of different species in the ecosystem follows a probability function called the population sensitivity distribution and it is assumed that a limited number of species are randomly sampled from the whole ecosystem, so the assessment of the acceptable effect level of a limited number of species can be considered appropriate for the whole ecosystem. The slope and confidence interval of the SSD curve reveals the certainty of the risk estimate. It is generally used as the maximum environmental permissible concentration threshold (HC_X, usually HC_5). HC_5 represents the concentration at which the affected species do not exceed 5% of the total species, or the concentration at which 95% of species are protected. While the choice of protection level is arbitrary, it reflects a compromise between statistical considerations (the HC_X is too small to make risk predictions reliable) and environmental protection needs (the HC_X should be as small as possible).

3.3.1.5 Bioavailability Assessment

There are three dynamic steps of bioavailability: the first step, environmental bioavailability or bioavailability, describes the potential availability of environmental pollutants and refers to the exchange behavior between bound and free states of pollutants. The second step, environmental bioavailability, refers to the process by which pollutants cross a biofilm for uptake by organisms. The third step, toxic bioavailability, refers to the distribution, metabolism and discharge of pollutants in living organisms and also includes the adverse effects and enrichment of pollutants caused by their target action points in living organisms. It can be evaluated by toxic unit, bioassay, environmental risk index, biological vulnerability index, genotoxicity index and other indexes.

The bioavailability assessment methods of pollutants in soil include model organisms, chemical extraction and passive sampling. Model organisms mainly through biological pollutants concentration in the body, or biological changes before and after exposure to pollutants in environmental medium concentration. It mainly measures the concentration of pollutants in the organism or the change of pollutant concentration in the environment medium before and after the organism is exposed. The model organisms include soil animals, soil microorganisms and plants. For example, the ratio of the concentration of pollutants in earthworms to the concentration in soil can characterize the level of soil pollution and warn the potential risk of pollution. Chemical extraction method includes intense chemical extraction method and mild chemical extraction method. Mild chemical extracts include methanol, acetonitrile, water, n-butanol, ethanol, n-hexane, toluene and methylene chloride or a mixture of them. Accelerated solvent extraction (ASE), supercritical fluid extraction (SFE) and sequential ex-traction are also used for extraction of contaminants. Semi-permeable membrane devices (SPMDs) are common methods of passive sampling.

3.3.1.6 Assessment of Soil Ecosystem Service

The assessment of soil ecosystem service is an important aspect of ecological risk assessment. The richer the biodiversity of an ecosystem, the more stable the ecological balance. The stronger the ecological service function, the stronger the anti-interference ability. The evaluation indicator system of soil ecosystem service is mainly referred to in Table 3.1. Meanwhile, the soil ecosystem service evaluation indicator system is shown in Table 3.2.

3.3.1.7 Expert Judgment

The expert judgment method is a commonly used qualitative method. The specific method is to invite experts from related industries and different levels to analyze the issues discussed from different angles and determine whether unacceptable risks exist and whether the risk level is high, medium or low. Then the final conclusion is made by synthesizing the opinions of all the experts.

TABLE 3.1
Indicator System for Quantifying Soil Function

Soil Function	Quantitative Indicator
The water cycle	Bulk density, available water capacity, water conductivity, soil texture, effective soil layer thickness, gravel content, water pressure property, plant rooting depth
Nutrient cycling	Bulk density, CEC, soil texture, effective soil layer thickness, humus content, pH value and plant rooting depth
Filter and buffer heavy metals	Bulk density, soil texture, effective soil layer thickness, humus content, organic matter decomposition degree, pH value, gravel content
Filter and buffer organic compounds	Bulk density, CEC, available water capacity, soil texture, thickness of effective soil layer, humus content, decomposition degree of organic matter, pH value, water solubility, root depth of plants
Buffering acids and contaminants	Bulk density, CEC, alkali saturation, soil texture, effective soil layer thickness, pH value, gravel content, rooting depth
Carbon storage	Bulk density, humus content, effective soil layer thickness, gravel content, rooting depth
Regulate nutrient loss	Bulk density, soil texture, gravel content, effective soil thickness, humus content, water pressure property, rooting depth
Habitat of natural plant populations	Bulk density, available water capacity, CEC, SOIL texture, effective soil thickness, Humus content, metamorphic nature, gravel content, rooting depth
Habitat for microorganisms	Bulk density, microbial biomass, soil texture, pH value, gravel content, effective soil layer thickness, water solubility, plant rooting depth
Suitability of agricultural production	Bulk density, soil texture, effective soil thickness, humus content, pH value, gravel content, soil structure, water pressure properties

TABLE 3.2
Soil Ecosystem Service Evaluation Indicator System

Ecosystem Service	Characteristic	Quality Dimension	Indicator
Supply service	Production characteristic	Fertility quality	Profile configuration, soil bulk density, soil nutrient elements, barrier layer type and depth from the ground surface, gravel content, soil beneficial trace elements, organic matter content, soil texture of plough layer, soil pH, soil cation exchange capacity, effective soil layer thickness, total nitrogen, plough layer thickness, electrical conductivity
Support service	Sustainability	Environmental quality	Annual deposition flux of atmospheric dry and wet sediment, environmental quality of irrigation water, soil heavy metal pollution, pesticide residue, white pollution
Regulating service		Elastic quality	Organic matter content, surface soil texture, soil pH, soil cation exchange capacity, effective soil layer thickness, coefficient of variation of annual grain yield, soil microbial carbon, soil earthworms

3.3.1.8 Application Scope of Methods

Based on the previous analysis, Table 3.3 summarizes the applicable scope of several ecological risk assessment methods.

3.3.2 Models

With the emergence of ecological risk assessment model, ecological risk assessment has been transformed and combined with toxicology and model simulation from simply relying on experimental tools of ecotoxicology. The main difference between different evaluation methods lies in the different models used in the process of toxicity assessment and risk characterization, such as assessment

TABLE 3.3
Application Scope of Assessment Methods

Assessment Method	Application Scope
Ecological risk assessment in polluted areas	This method can systematically assess the ecological risk in a certain area.
Assessment factor	Suitable for compounds with less toxicological data.
Hazard quotient	It is suitable for assessing the toxicological effect of a single compound.
Species sensitivity distribution	The amount of toxicity data collected is 10 or more.
Bioavailability assessment	Used to assess the bioavailability of heavy metal elements.
Assessment of soil ecosystem service	It is used to analyze and screen the key factors affecting soil health, and to evaluate the synergistic and countervailing effects of different indicators under environmental change and human disturbance.
Expert judgment	In the absence of basic data, this method can estimate and judge the overall situation

TABLE 3.4
Comparison of Ecological Risk Models

Model	Risk Characterization Method	Advantages	Disadvantage
HQ	Assessment factor or hazard quotient	The requirement of toxicology data is less, and the degree of operation difficulty is low.	High uncertainty of results
SSD	Hazard quotient	High confidence	The results are uncertain.
PERA	Probabilities eco-logical risk assessment	The evaluation results are given in the form of probability, which is more objective.	It has high requirements for data and professional technology and is difficult to operate.

factor or HQ, SSD and probabilities eco-logical risk assessment (PERA). The comparison of ecological risk models is shown in Table 3.4.

3.4 HEALTH RISK ASSESSMENT

3.4.1 Exposure Estimation Methods

A stratification strategy should be used to select a suitable method to quantitatively estimate the exposure of the population to the target environmental factors via different pathways and routes according to the purpose and type of assessment.

The exposure estimation should be based on different pathways of population exposure to the target environmental factors, combined with exposure concentrations and exposure parameters, and quantitative estimation of exposure levels of different exposure pathways should be performed separately using exposure models.

Intake through the mouth

$$CDI_m = \frac{C_s \times IR \times CF \times EF \times ED}{BW \times AT}$$

CDI_m is intake through the mouth, mg/(kg·d); C_s is the amount of contaminants in the soil, mg/kg; IR is the amount of soil ingested per hour, mg/d; CF is the conversion factor, 10^{-6} kg/mg; EF is the exposure frequency, d/a; ED is the exposure period, a; BW is the average weight of the population during the exposure period, kg; AT is the average action time, d.

Intake through breathing

$$CDI_b = \frac{C_s \times \left(\frac{1}{PEF}\right) \times IR \times EF \times ED}{BW \times AT}$$

CDI_b is intake through breathing, mg/(kg·d); PEF is the soil dust dispersion factor; IR is the respiration rate, L/min.

Intake through skin contact

$$CDI_s = \frac{C_s \times CF \times SA \times AF \times ABS \times EF \times ED}{BW \times AT}$$

CDI_s is intake through skin contact, mg/(kg·d); SA is the area of skin that may come in contact with soil, cm^2; AF is the adsorption coefficient of skin on soil, mg/cm^2; ABS is the skin absorption coefficient.

Intake through diet

$$CDI_w = \frac{C_w \times IR \times EF \times ED}{BW \times AT}$$

$$CDI_f = \frac{C_f \times IR \times FI \times EF \times ED}{BW \times AT}$$

CDI_w is intake through drinking water, mg/(kg·d); C_w is the amount of pollutants in water, mg/kg; C_f C is the amount of contaminants in the food, mg/kg; *FI* is the proportion of contaminated food to total food.

3.4.2 Determination of Exposure Concentration

The methods for determining exposure concentrations include: (i) directly monitoring the concentration of target environmental factors in ambient air, indoor air, indoor dust, soil, food, drinking water and other media, and monitoring the concentration of target environmental factors in different media according to HJ 839 and other relevant standards or technical specifications; (ii) based on the information on the source, use, release, transformation and convergence of the target environmental factors, choose a suitable environmental convergence model to predict the concentration of the target environmental factors in the environmental media and their spatial and temporal distribution. Suitable environmental fate models should be selected based on factors, such as assessment objectives, technical capability of the model, access and difficulty of use.

3.4.3 Risk Characterization

3.4.3.1 Assessment Steps

Based on the toxic effects of the target environmental factors, the risk of carcinogenic effects and the risk of non-carcinogenic effects were calculated: (i) Carcinogenic effect risk. For the same environmental factor, the corresponding carcinogenic slope coefficient or unit risk factor should be selected according to different exposure routes for risk estimation. When multiple exposure routes or multiple target environmental factors produce similar carcinogenic effects on the same target organ, the excess carcinogenic risks of different exposure routes or different target environmental factors can be summed up to calculate the total excess carcinogenic risks. (ii) Risk of non-carcinogenic effects. Generally, the HQ is used for characterization. For the same target environmental factor, the corresponding reference concentration or reference dose should be selected according to different exposure routes for risk estimation. When multiple exposure routes or target environmental factors produce similar non-carcinogenic effects on the same target organ, the HQs of different exposure routes or target environmental factors can be added up to calculate the total HQ (Figure 3.2). The risk of carcinogenic or non-carcinogenic effects can be estimated using point estimates or probability estimates.

3.4.3.2 Exposure Risk Contribution Ratio Analysis

The contribution of carcinogenic and non-carcinogenic risks of a single pollutant by different exposure routes are calculated using the following equations:

$$PCR_i = \frac{CR_i}{CR_n} \times 100\%$$

$$PHQ_i = \frac{HQ_i}{HI_n} \times 100\%$$

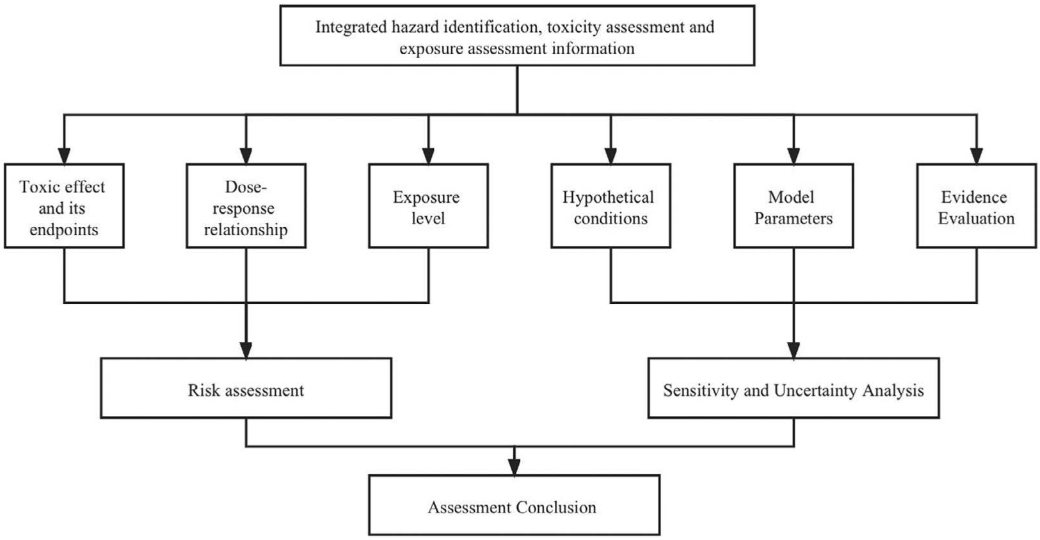

FIGURE 3.2 Risk characterization assessment steps.

CR_i is the carcinogenic risk of a single pollutant via the ith exposure route, dimensionless: PCR_i is the contribution rate of carcinogenic risk of a single pollutant via the ith exposure route, dimensionless; HQ_i is the HQ of a single pollutant via the ith exposure route, dimensionless; PHQ_i is the contribution rate of non-carcinogenic risk of a single pollutant via the ith exposure route, dimensionless.

The larger the percentage obtained from the previous formula, the higher the contribution of the specific exposure pathway to the total risk.

3.4.3.3 Calculation of Pollutant Remediation Targets

Focus on the calculation of pollutant remediation targets and the initial determination of soil remediation targets. Contaminants with both carcinogenic and non-carcinogenic risks should be calculated separately for remediation targets under carcinogenic and non-carcinogenic risks. In addition, soil remediation limits based on the protection of groundwater should be calculated at the same time, and the minimum value should be taken as the final remediation target. The specific calculation method is as follows.

Calculation of Non-Carcinogenic Risk-Based Soil Remediation Limits for Contaminants

i. Calculation of soil remediation limits for non-carcinogenic risks via mouth intake soil pathway:

$$HSRL_m = \frac{RfD_m \times AHQ}{MER_{nc}}$$

where $HSRL_m$ is the soil remediation limits based on non-carcinogenic risks via mouth intake soil pathway, mg/kg; AHQ is the acceptable HQ, dimensionless, and takes the value of 1; MER_{nc} is the mouth intake soil exposure (non-carcinogenic effect, per unit body weight), kg/(kg·d); RfD_m is the mouth intake reference dose (per unit body weight contaminant), mg/(kg·d).

ii. Soil remediation limits based on non-carcinogenic risks from the skin contact soil pathway:

$$HSRL_S = \frac{RfD_s \times AHQ}{SER_{nc}}$$

where $HSRL_s$ is the soil remediation limit based on non-carcinogenic risk from skin exposure, mg/kg; SER_{nc} is the soil exposure from skin exposure (non-carcinogenic effect in unit body weight), kg/(kg·d); RfD_d is the reference dose from skin exposure (in unit body weight), mg/(kg·d); and AHQ is the acceptable HQ, dimensionless and takes the value of 1.

iii. Soil remediation limits based on non-carcinogenic risks from the inhalation soil particle pathway:

$$HS\mathrm{RL}_{iP} = \frac{RfD_i \times AHQ}{IPER_{nc}}$$

where $HSRL_{ip}$ is the soil remediation limit based on the non-carcinogenic risk of inhaled particulate matter, mg/kg; $IPER_{nc}$ is the soil exposure to inhaled soil particulate matter (non-carcinogenic effect, per unit body weight), kg/(kg·d); RfD_i is the respiratory inhalation reference dose (per unit body weight), mg/(kg·d).

iv. Soil remediation limits based on non-carcinogenic risks from inhalation of contaminant vapor pathways in outdoor air:

$$HSRL_{ov} = \frac{RfD_i \times AHQ}{oVER_{nc1} + oVER_{nc2}}$$

where $HSRL_{ov}$ is the soil remediation limit based on the non-carcinogenic risk of inhaling outdoor pollutant vapor, mg/kg; $oVER_{nc1}$ is the soil exposure (non-carcinogenic effect) corresponding to the pollutant vapor from the surface soil in inhaled outdoor air, kg/(kg·d); $oVER_{nc2}$ is the soil exposure (non-carcinogenic effect) corresponding to the pollutant vapor from the lower soil in inhaled outdoor air, kg/(kg·d).

v. Soil remediation limits based on non-carcinogenic risks from inhalation of indoor air contaminant vapor pathways:

$$HSRL_{iv} = \frac{RfD_i \times AHQ}{iVER_{nc}}$$

where $HSRL_{iv}$ is the soil remediation limit value based on the non-carcinogenic risk of inhaling indoor pollutant vapors, mg/kg; $iVER_{nc}$ is the soil exposure corresponding to the inhalation of pollutant vapors from the underlying soil in indoor air (non-carcinogenic effect), kg/(kg·d).

Calculation of Soil Remediation Limits for Contaminants Based on Carcinogenic Risk

i. Calculation of Soil remediation limits for carcinogenic risks via mouth intake soil pathway:

$$RSRL_m = \frac{ACR}{MER_{ce} \times SF_m}$$

where $HSRL_m$ is the soil remediation limits based on carcinogenic risks via mouth intake soil pathway, mg/kg; ACR is the acceptable carcinogenic risk, dimensionless and takes the value of 10^{-6}; MER_{ce} is the mouth intake soil exposure (carcinogenic effect), kg/(kg·d); SF_m is the mouth intake carcinogenic slope factor, $[\mathrm{mg/(kg \cdot d)}]^{-1}$.

ii. Soil remediation limits based on carcinogenic risks from the skin contact soil pathway:

$$RSRL_s = \frac{ACR}{SER_{ce} \times SF_s}$$

where $RSRL_s$ is the soil remediation limit based on carcinogenic risk from skin exposure, mg/kg; SER_{ce} is the soil exposure from skin exposure (carcinogenic effect), kg/(kg·d); SF_s is the skin contact carcinogenic slope factor, $[\text{mg}/(\text{kg·d})]^{-1}$.

iii. Soil remediation limits based on carcinogenic risks from the inhalation soil particle pathway:

$$RSRL_{isp} = \frac{ACR}{ISPER_{ce} \times SF_i}$$

where $RSRL_{isp}$ is the soil remediation limit based on the carcinogenic risk of inhaled particulate matter, mg/kg; $ISPER_{ce}$ is the soil exposure to inhaled soil particulate matter (carcinogenic effect), kg/(kg·d); SF_i is the respiratory inhalation carcinogenic slope factor, $[\text{mg}/(\text{kg·d})]^{-1}$.

iv. Soil remediation limits based on carcinogenic risks from inhalation of contaminant vapor pathways in outdoor air:

$$RSRL_{ivo} = \frac{ACR}{\left(IVOR_{ce1} + IVOR_{ce2}\right) \times SF_i}$$

where $RSRL_{ivo}$ is the soil remediation limit based on the carcinogenic risk of inhaling outdoor pollutant vapor, mg/kg; $IVOR_{ce1}$ is the soil exposure (carcinogenic effect) corresponding to the pollutant vapor from the surface soil in inhaled outdoor air, kg/(kg·d); $IVOR_{ce2}$ is the soil exposure (carcinogenic effect) corresponding to the pollutant vapor from the lower soil in inhaled outdoor air, kg/(kg·d).

v. Soil remediation limits based on carcinogenic risks from inhalation of indoor air contaminant vapor pathways:

$$RSRL_{ivi} = \frac{ACR}{IVIER_{ce} \times SF_i}$$

where $RSRL_{ivi}$ is the soil remediation limit value based on the carcinogenic risk of inhaling indoor pollutant vapors, mg/kg; $IVIER_{ce}$ is the soil exposure corresponding to the inhalation of pollutant vapors from the underlying soil in indoor air (carcinogenic effect), kg/(kg·d).

3.4.4 Models

3.4.4.1 Exposure Model

Exposure modeling specifically refers to the use of conceptual models and mathematical simulations to describe the process of human exposure to contaminants and to predict and estimate exposure. There are several basic types of exposure models: single-media models, multi-media models, probabilistic exposure models, generalized dose models and pharmacokinetic models.

i. Single-media model
Pollutants enter the human body through only one medium, and the medium can be air, water, soil, food etc. According to the different mediums of human exposure to pollutants, the calculation methods of exposure in single-media models are different and can be divided into air inhalation, intake through water bodies, food and drugs and calculation through cosmetics, water, and air contact with human skin.

ii. Multi-media model
The multi-media model refers to a single pollutant or multiple pollutants entering the human body through multiple media. The exposure of multiple media into the human body is summed up by calculating each media separately after exposure and is accumulated on the basis of the single-media model.

iii. Probabilistic exposure model
A probabilistic exposure model is a method for assessing the exposure of a population in a given area based on the probability distribution of the occurrence of possible contamination events. This assessment method can consider all possible outcomes of pollutant exposure and the likelihood of each outcome occurring, and the assessment results are generally expressed as a probability distribution of all possible exposure levels. Compared with other assessment methods, probabilistic exposure models can provide more accurate and detailed information on the exposure of the population to a specific pollution event.

iv. Generalized dose model
The development of the Generalized Linear Model (GLM) and the Generalized Additive Model (GAM) has been an important advance in the field of statistics over the past 30 years. The GAM is a non-parametric extension of the GLM with a broader scope of application; GAM can fit non-parametric regressions and is suitable for dealing with overly complex non-linear relationships between the response variable and a large number of explanatory variables. GAM applies the underlying assumption that the functions are additive. It allows each covariate to be fitted as an unconstrained smoothing function, rather than just as a dull parametric function, by applying a smoothing function to some or all of the explanatory variables to build the model.

v. Pharmacokinetic model
Pharmacokinetic models were originally established to quantitatively study the rate law of in vivo processes of drugs and simulation of mathematical models, which include the absorption, distribution, biotransformation, excretion and other dynamic processes of drugs in the body, can also be applied to pollutants, in order to quantitatively study the changes of pollutants through the previous processes, mathematical methods to simulate the metabolic processes of pollutants in the body. The empirical formula model is a good description of the general contaminant concentration versus time and can be used to derive the main contaminant metabolic kinetic parameters such as clearance and half-life. The traditional pharmacokinetic models are atrial pharmacokinetic model, physiological pharmacokinetic model and circulating pharmacokinetic model.

3.4.4.2 Health Risk Evaluation Model: RBCA

There are many health risk evaluation models, but from the comparison of pollution sources, exposure pathways, pollutant transport and transformation models, the RBCA model is more comprehensive in considering pollution sources and fully considers the multi-media environment of water, soil and gas in contaminated sites in terms of exposure pathways and analyzes the attenuation effect of pollutants themselves due to volatilization, dilution or leakage, biodegradation and other effects in the evaluation process. Most of the contaminated sites in China have both soil and groundwater contamination, and the RBCA model is more suitable for the health risk evaluation of contaminated sites in China.

There are three main types of parameters that need to be input to RBCA: site characteristics, toxicological parameters of contaminants and sensitive receptors and exposure parameters.

i. Site characteristics parameters. Site characteristics parameters include physical parameters of soil, groundwater and atmosphere that are required to calculate the health risk caused by contaminant transport in groundwater, soil and atmosphere, including average annual temperature, average annual wind speed, precipitation, the average depth of groundwater, pH, water content, and soil type, in the site area. These parameters are generally obtained through the second stage of the site sampling survey.
ii. Toxicological parameters of pollutants. Toxicological parameters include reference doses and slope factors of various pollutants exposed by various exposure pathways etc.
iii. Sensitive receptors and exposure parameters. The exposed population is chosen to live on the contaminated site. Sensitive receptors for residential sites include children and adults, and only

adults are considered for commercial or industrial sites. Exposure parameters include exposure frequency, exposure period, soil and groundwater intake and human-related parameters. Human body-related parameters, such as body weight, life expectancy, air respiration and daily water intake, were mainly determined based on relevant domestic statistics. Soil intake by mouth, exposure frequency, exposure period and other relevant parameters are often obtained based on the ASTM recommendations in the United States and modified with the domestic reality.

3.5 UNCERTAINTY ANALYSIS OF HEAVY METAL SOIL RISK IN SEWAGE IRRIGATION

3.5.1 Uncertainty Analysis Mode

3.5.1.1 Monte Carlo Analysis (MCA)

Monte Carlo method is to simulate various random phenomena that may occur in the real system by using random numbers that follow a certain distribution pattern. Specifically, the uncertainty of parameters is expressed by probability method to make the risk characterization and exposure evaluation more objective. The MCA method provides the probability method to spread the uncertainty of parameters and better represent the risk and exposure evaluation. The analysis steps include: (i) defining the statistical distribution of input parameters; (ii) random sampling from these distributions; (iii) repeated model simulation using randomly selected parameter series; (iv) analyze the output value and get a reasonable result. At present, most of the risk assessments are based on the baseline risk assessment (BRA) under the rational maximum exposure (RME). The assessment method is relatively conservative, with great uncertainty, the degree of conservatism is difficult to measure and the information provided to decision makers is limited. When the risk value obtained by the BRA method is 10-5, the MCA method can obtain a reasonable probability distribution interval and provide more information for decision makers. However, the disadvantages of MCA are as follows: (i) the evaluation process is complex and requires a lot of calculations; (ii) It is difficult to determine the degree of superiority or inferiority of MCA itself. USEPA tends to apply the probability technology of MCA to study the consequences of accidents under different probabilities, so as to provide more extensive reference for environmental risk managers. The parameters in PRA are in the form of probability distribution, and random values are taken from known distribution characteristics for Monte Carlo simulation. The output results are also in the form of probability distribution.

3.5.1.2 Taylor's Simplified Method

Because the function relationship between the input value and the output value in the risk model is too complex, the probability distribution of the output value cannot be obtained from the probability distribution of the input value. Taylor extension sequence is used to simplify and approximate the input risk model, and the relationship between the input value and the output value is expressed in the form of deviation. With this simplification, we can express the mean value, deviation of the evaluation model and other relations between the input value and the output value.

3.5.1.3 Probability Tree Method

The probability tree method comes from the fault tree analysis in risk assessment. The probability tree can represent three or more uncertain results, and the probability of their occurrence can be quantitatively expressed by discrete probability distribution. If the uncertainty is continuous, the probability tree method can still be applied when the continuous distribution can be approximated by the discrete distribution.

3.5.1.4 Expert Judgment Method

The expert judgment method is based on the Bayesian theory, which considers that any unknown data can be regarded as a random variable. The analyst can express the unknown data in the form of a probability distribution and set the unknown parameters as a specific probability distribution. The confidence interval can be obtained from the probability distribution. Subjective risk assessment based on the probability given by experts. Bayesian theory holds that individuals have rich professional knowledge, become familiar with the situation after research and have information for risk assessment. The information not only comes from the traditional statistical model but also includes some empirical data. Therefore, the information provided by the experts is scientific and technical in accordance with the judgment of logic and rules. The first step in applying this method is to organize experts in the professional field to hold seminars. Although there is great uncertainty in the health risk assessment, the use of technical means can minimize the uncertainty and provide useful help to environmental managers.

3.5.2 Sensitivity Analysis of Model Parameters

3.5.2.1 Principles for Determining Sensitive Parameters

The parameters (P) selected for sensitivity analysis shall be those that have a great impact on the risk calculation results, such as population-related parameters (body weight, exposure period, exposure frequency etc.), and parameters related to exposure route (daily intake of soil, soil adhesion coefficient on skin surface, daily intake of air volume, indoor space volume and steam infiltration area ratio etc.). When the risk contribution rate of a single exposure pathway exceeds 20%, sensitivity analysis of population-related parameters and parameters related to this pathway should be conducted.

The current site risk assessment method is the point assessment method. The certainty of model parameters or the point evaluation method is simple, but there are: (i) 95% quantile of the parameter is selected to solve the maximum value. In fact, it is a scenario in which the calculation rarely occurs, which often leads to the result being too conservative; (ii) since the risk is mainly calculated by multiplication, division, addition and subtraction, the final calculated risk is not necessarily the risk value of 95% quantile; (iii) it is not significant to make sensitivity analysis around the 95% quantile (solving the maximum value). The introduction of PRA provides an effective method to solve the uncertainty of parameters in risk assessment. The hierarchical risk assessment framework is shown in Figure 3.3.

3.5.2.2 Sensitivity Analysis Method

The sensitivity of model parameters can be expressed as the sensitivity ratio, that is, the change of model parameter values (from P_1 to P_2). The ratio to the change in the carcinogenic risk or HQ (change from X_1 to X_2). The recommended model for calculating the sensitivity ratio is shown in Equation D (D.3) in Appendix (HJ25.3-2014). The sensitivity ratio of model parameter (P) can be calculated by the following formula:

$$SR = \frac{\dfrac{X_2 - X_1}{X_1}}{\dfrac{P_2 - P_1}{P_1}} \times 100\%$$

where SR is the sensitivity ratio of model parameters, dimensionless; P_1 is the value before the model parameter P changes; P_2 is the value after the model parameter P changes; X_1 is the carcinogenic risk or HQ calculated according to P_1, dimensionless; X_2 is the carcinogenic risk or HQ calculated according to P_2, dimensionless.

The greater the sensitivity ratio, the greater the impact of the parameter on the risk. For sensitivity analysis of model parameters, the actual value range of parameters shall be comprehensively considered to determine the variation range of parameter values. With more and more data acquisition, the uncertainty factors from 1984 to 2004 gradually decreased, and the uncertainty factors in 2004 were all lower than 3000.

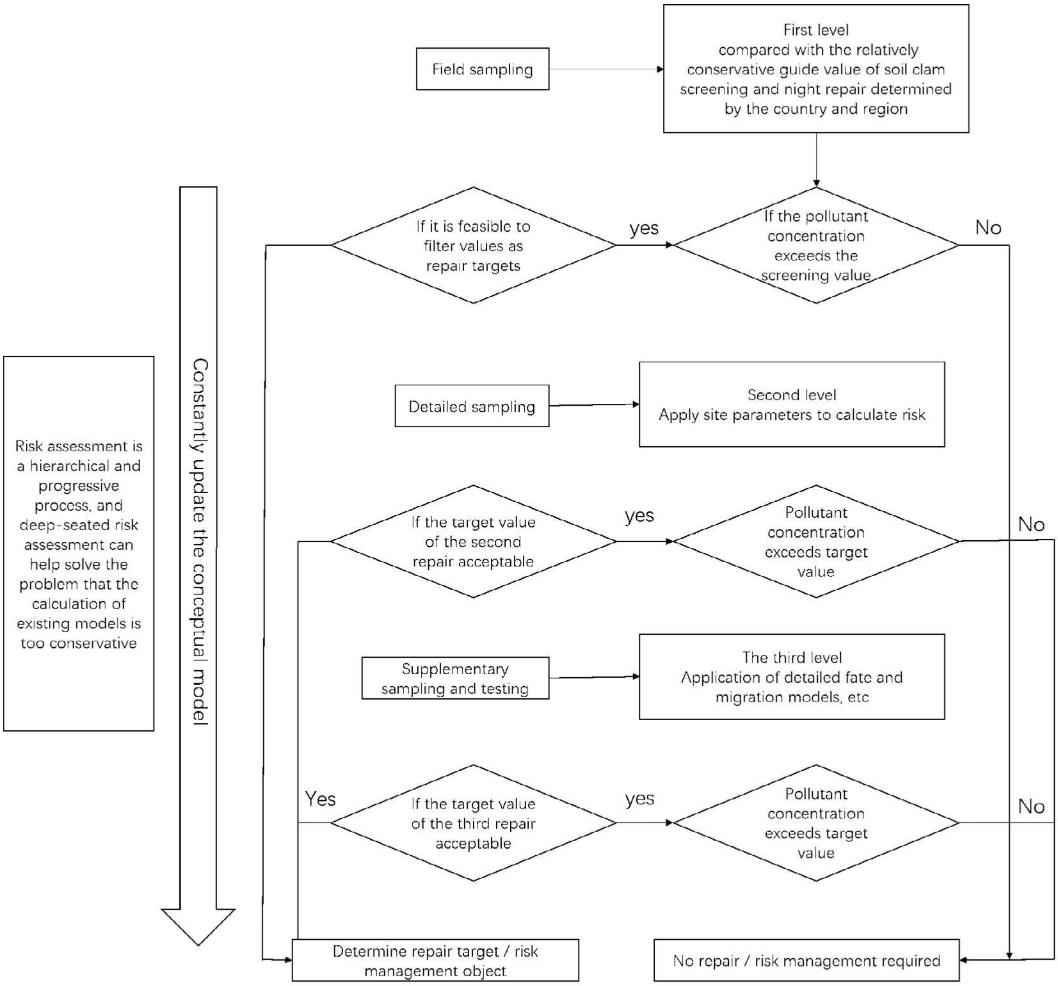

FIGURE 3.3 The hierarchical risk assessment framework.

3.6 CONCLUSIONS AND PERSPECTIVES

It is very important to assess the potential risk of heavy metal pollution in soil. The potential risk assessment can be said to be an objective and comprehensive risk assessment, and the potential risk assessment of heavy metal pollution in soil is to use the comprehensive pollution index to analyze. This chapter comprehensively introduced the risk assessment procedures, methods and models of heavy metal-contaminated soil caused by sewage irrigation through ecological and health risk assessment. It includes multi-element index analysis, soil toxicity level analysis, soil pollution degree analysis and other important contents. Through the analysis of the comprehensive pollution index, the potential risks in the soil contaminated by heavy metals can be effectively understood, and the degree of soil pollution can be more intuitively understood, which is also conducive to solving the problem of heavy metal-contaminated soils irrigated by sewage. We look forward to providing theoretical support and help for practical risk assessment work. Meanwhile, the perspectives are present as follows:

i. For the ecological and health risk assessment of heavy metal-contaminated soil caused by sewage irrigation, there is currently no systematic research method and no detailed evaluation indicators. Also, some basic indicators are not necessarily in line with the risk assessment under the specific conditions of sewage irrigation, which may lead to uncertainty in the assessment results. Meanwhile, the degree of pollution is closely related to pollution sources, sewage water quality, irrigation years, soil physical, regional climate characteristics and chemical properties (pH, EC, CEC, content of organic matter, nitrogen, phosphorus and potassium, types and speciation states of heavy metals etc.), which will inevitably cause deviations in assessment results. Therefore, it is hoped that the perfect standards and systems for ecological and health risk assessment in sewage irrigation areas can be established in the future.

ii. When studying heavy metal-contaminated soil in sewage irrigation areas, the scope of the study area should be expanded, especially the analysis of the forms and types of heavy metals, growing plants and climate conditions (wind direction, wet deposition and temperature etc.), so as to precisely analyze the impact of sewage irrigation on soil and human health.

iii. Sewage irrigation is the main cause of heavy metal pollution in soil, and water pollution is the root cause of sewage irrigation. Therefore, pursuing its source and preventing water pollution is the core work of solving heavy metal pollution in farmland. Relevant departments and environmental protection departments should pay high attention to the prevention of water pollution as the core work, vigorously rectify the enterprises and units that produce water pollution and pay attention to the work of water protection. Hope in the face of environmental protection and economic development, we should pay more attention to the importance of environmental protection work, environmental protection is to protect ourselves, protect our soil from harm, is responsible for our own safety. And it is also necessary to carry out the relevant risk assessment on the polluted water body. While the relevant research in this area is extremely lacking.

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4 Municipal Wastewater Sludge as a Sustainable Bioresource in Developed Countries

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4.1 INTRODUCTION

Sewage sludge (SS) is defined as the solid, semi-solid, or liquid residue generated during wastewater treatment released from various sources, including domestic, industries, street runoff, businesses, and industrial establishments (Zuloaga et al., 2012). Industrialization and urbanization have resulted in a substantial increase in the volume of wastewater and SS generated from wastewater treatment plants (WWTP). The annual average dry SS production in the US, European Union, Japan, Germany, and Australia is 17.8, 9.0–9.5, 2.2, 2.0, and 0.36 million metric tons, respectively (Campo et al., 2021). This huge production of SS is the outcome of various factors such as the level of WWTP and its efficiency (primary, secondary, and tertiary), SS stabilization method (anaerobic or aerobic digestion), treatment technology (membrane bio-reactor, nutrient removal, P-precipitation), and operation conditions (Ivanová et al., 2018). In addition, wastewater contains a substantial amount of other organic and inorganic contaminants that are not efficiently biodegraded or volatilized in WWTPs (Petroody et al., 2021). Further, it also includes human pathogens, antibiotic resistance genes, mobile genetic elements, and antibiotic-resistant bacteria (Chen et al., 2020). All these chemical and biological contaminants were found to be concentrated in SS, making its disposal one of the most challenging concerns in modern society (Vinay et al., 2023). Therefore, raw SS is treated before its application with different sludge treatment processes such as lime stabilization, anaerobic/aerobic digestion, and composting to reduce sludge volume, weight, and potential health risk (Poornima et al., 2021); treated SS is known as biosolids. SS management challenges are anticipated to emerge with an increase in population, WWTP coverage, stringent regulations, and efficiency of treatments.

The common SS management strategies are (1) no recycling (dumping, landfilling, and storage), (2) reuse (land application), (3) substance conversion (AD, composting, wet air oxidation [WAO], and incineration), and (4) nutrient (phosphorous and nitrogen)/energy (anaerobic digestion (AD), gasification, incineration) recovery. Landfilling was the common SS disposal strategy globally. However, increasingly stringent environmental regulations, generation of secondary contaminants from leachate, substantial greenhouse gases (GHG) emissions from organic degradation, and utilization of valuable landfill space are the drivers of phasing out of landfilling (Poinen and Bokhoree, 2022). For instance, European Union Community Landfill Directive has set targets to reduce organic waste disposal, inferring that landfilling must not be considered a sustainable method to dispose of SS (Kacprzak et al., 2017). Therefore, the production of a large volume of SS, regulatory control over prior disposal methods such as a ban or phasing out of landfilling and ocean dumping, and recognizing SS as a bioresource are the other drivers for the beneficial utilization of SS in developed countries. The US, EU, Japan, and Australia are the leaders in beneficial SS utilization. Table 4.1 highlights the status of SS utilization in different developed countries.

SS contains substantial organic matter and beneficial plant nutrients (Benassi et al., 2019), promoting its land utilization instead of landfilling and incineration, particularly from an environmental and

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TABLE 4.1
SS Use in Different Developed Countries (Poornima et al., 2021; Sharma et al., 2017)

Countries	Rate of SS Use (%)	SS Production (Million Tons Dry Solids Annually)	Main SS Applications
US	55	17.8	Land applications
Australia	80	0.36	Land application
UK	85	1.05	Land application, energy recovery
EU	40	9	Land application
Japan	74	2.2	Energy recovery and construction products

economic perspective. Various factors which influence the selection of SS management strategies are population density, economic conditions, available land area, social acceptability, level of urbanization, local needs, and livestock density index (Shaddel et al., 2019). For instance, land application is generally practiced in an area with abundant land availability and low population density, substantially reducing the SS disposal cost (US). In contrast, alternative nutrient and energy recovery routes are preferred in regions with high population density and limited land availability (Japan) (Christodoulou and Stamatelatou, 2016). In addition, nutrient recovery technologies have experienced rapid advancement in past years due to increased environmental awareness, operational benefits, stringent discharge limits on these nutrients, and supply security (Shaddel et al., 2019). However, energy recovery and biofuel production from SS is one of the modern practices in developed nations through different processes such as AD, pyrolysis, co-incineration in coal power plants, wet oxidation, gasification, and energy generation through a microbial fuel cell (Raheem et al., 2018). Incineration and AD are very popular in most countries. Incineration is practiced when land is limited (Japan incinerates >70% of SS [Singh et al., 2020]), and local perception is against the land application (Netherlands and Switzerland) (Kirchmann et al., 2017). Moreover, AD is commonly practiced in various countries at small, medium, and large scales to produce biogas that produces power and heat. Moreover, SS is a viable wet feedstock for biofuels productions. Diverting SS from further treatment to produce biofuels has various benefits such as reduction of non-beneficial sludge disposal at landfilling and incinerators, reduction of residual management costs, reduction of GHG emissions, and saving energy use (Seiple et al., 2017). Moreover, disposal of SS through incineration and reusing as a raw material for manufacturing construction material (Portland cement and blended) had been reported on a wider scale (Pavlík et al., 2016). Therefore, this chapter extensively discusses the existing SS management legislations in developed countries and the approach to reuse SS in engineering applications, along with recovering energy and nutrients as much as possible to minimize the adverse impact of SS on the environment. This chapter also provides a techno-economic comparison of current sludge management strategies and possible innovative methods, which opens prospects for developing nations.

4.2 MUNICIPAL SEWAGE SLUDGE CHARACTERISTICS AND ITS MATERIAL AND ENERGY RECOVERY POTENTIAL

Table 4.2 provides characteristics of primary, secondary, and mixed sludge. SS contains a high amount of organic matter in labile (40% total organic matter on a dry weight basis) and a recalcitrant fraction (such as polysaccharides, proteins, amino acids, and fatty acids) in dissolved or suspended states. SS also contains micronutrients (e.g., Fe, Zn, B, Mo, Mn) and macronutrients (e.g., N, P, K). N and P are the most abundant elements in SS, present in both organic and inorganic forms. Organically bonded N consists of protein-N, pyridinic-N, amine-N, and pyrrolic-N, while inorganic N includes ammonia-nitrogen (NH_3-N), nitrate-nitrogen (NO_3-N), and nitrite-nitrogen (NO_2-N). However, unlike inorganic N, inorganic P tends to be concentrated in organic forms or deposited in the inorganic solid phase due to its sparingly soluble nature. Organic P consists of different organic compounds, including lipids

TABLE 4.2
Characteristics of Different Types of SS; DM: Dry Matter (Collivignarelli et al., 2019; Poornima et al., 2021)

Parameter	Primary Sludge	Secondary Sludge	Mixed Sludge
Total dry solids (Total solids, TS) (%)	2–9	0.8–3.3	N.R.
Organic solids/volatile solids (% TS)	60–80	30–88	72–75
pH	5–8	6.5–8	6.5–8.2
Phosphorus (% TS)	0.2–2.8	0.5–11	1.2–3
Nitrogen (%TS)	1.5–4	2.4–5	2.8–4.9
Hydrogen (% TS)	4.6	4.0–5.2	4–4.6
Carbon (% TS)	33.5	35.2–40.8	N.R.
Oxygen (% TS)	23.1	22.1–25.4	18.5–21.9
Density (kg/m³)	1003–1010	1000–1020	N.R.
Organic carbon (% TS)	N.R.	N.R.	20.5–40.3
Silica (SiO_2, % TS)	15–20	N.R.	N.R.
Cellulose (% TS)	8–15	7.0–9.7	N.R.
Organic acids (mg/L, as acetate)	200–2000	1100–1700	N.R.
Alkalinity (mg/L as $CaCO_3$)	500–1500	580–1100	N.R.
Grease and Fats (% TS)	7.0–65	2–12	N.R.
Energy content (kJ/kg TS)	2900–23000	19000–23000	N.R.
Higher heating value (MJ/kg)	23–29	19–23	11.3–20
Fe (mg/kg_{DM})	2000–4000	2000	24000–38000
Mn (mg/kg_{DM})	N.R.	N.R.	100–200
Cd (mg/kg_{DM})	N.R.	N.R.	<3
Pb (mg/kg_{DM})	N.R.	N.R.	30–300
Cu (mg/kg_{DM})	N.R.	N.R.	100–200
Ni (mg/kg_{DM})	N.R.	N.R.	17–50
Zn (mg/kg_{DM})	N.R.	N.R.	300–3600
Cr (mg/kg_{DM})	N.R.	N.R.	500–900

Note: N.R: Not reported

and nucleic acid, while phosphate PO_4^{3-} is dominant among inorganic P forms (Hoang et al., 2022). Further, micronutrients are present in sufficient concentration in SS to meet the nutritional requirements of plants. The potential to recover energy from SS depends on the volatile solids content, which is further divided into two categories: (1) readily degradable (50% in primary SS and 25% in secondary SS) and (2) not readily degradable (30% in primary SS and 55% in secondary SS). Further, it is estimated that 1 lb of dry biosolid has the energy of (6–9) $\times 10^3$ British thermal units (Tyagi and Lo, 2016). Moreover, biological SS can be regarded as a fossil fuel substitute as dry biological SS produced from water resource recovery facilities have a calorific value of 12.0–20.0 MJ/kg, approximately similar to brown coal, 14.6–26.7 MJ/kg (Collivignarelli et al., 2019).

4.3 CURRENT MANAGEMENT PRACTICES FOR GENERATED MUNICIPAL WASTEWATER SLUDGE IN DIFFERENT DEVELOPED COUNTRIES AND THEIR LEGISLATION

The developed countries have implemented various practices of using biosolids for various purposes such as soil amendment in agriculture, mine site rehabilitation, energy recovery through various treatment processes, phosphorus recovery, and in construction sector (Figure 4.1). The US,

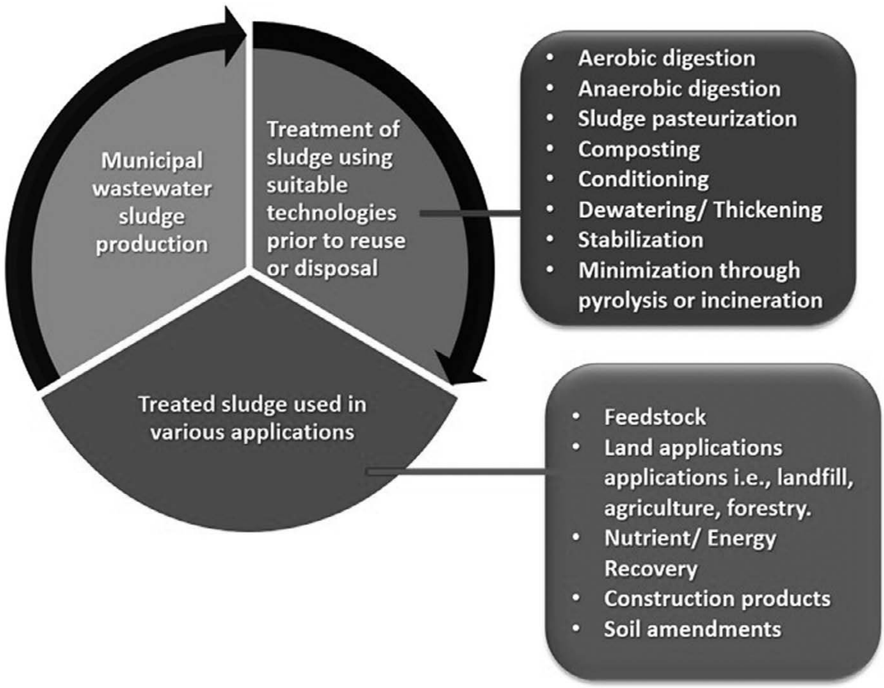

FIGURE 4.1 Municipal wastewater sludge technologies and applications in developed countries.

UK, Canada, Ireland, Japan, Netherlands, Australia, New Zealand, and Switzerland are among the nations in a transition of making the best out of the municipal SS sustainably.

4.3.1 Status of Beneficial Utilization of Biosolids in Developed Countries and Their Legislations

4.3.1.1 The US

In the US, nearly 55% of the SS produced is used for land applications such as agronomic and land restoration due to the low population density in the region (Peccia and Westerhoff, 2015). The remaining biosolids are disposed of in incinerators and landfills. Since the early 1970s, US waste legislation policies have been developed to encourage the potential advantageous use of biosolids on land. Congress passed the Ocean Dumping Ban Act in 1988, mandating land-based SS (Mulchandani and Westerhoff, 2016). United States Environmental Protection Agency (USEPA) has performed four SS surveys so far to identify contaminants in SS for their regulation, and the information from the 1988 to 1989 survey was used to establish Title 40 of CFR Part 503, that is, "The standards for the use or disposal of sewage sludge" (Venkatesan et al., 2015). This rule identified different parameters for pollutants present in SS, its treatment and use, and pathogen reduction. Under this rule, biosolids were no longer specified as a waste but were viewed as a potential nutrient source. The rules provided greater flexibility to manage SS via agronomic application, land reclamation, landscaping, forestry, and incineration. Further, biosolids were also marketed under the basic regulatory scheme in the US. However, land application is followed by strong odors, and biosolids encompass hazardous pollutants, pathogens, and antibiotic-resistant bacteria. Owing to this, 37 of 50 US states either ban land applications or enforce more stringent restrictions than federal regulations through their local ordinances (Peccia and Westerhoff, 2015). Further, the energy potential of SS has been contemplated in some large US municipalities where they started to operate modern low-emission sludge incinerators.

4.3.1.2 European Union and the UK

EU uses different approaches for SS disposal and reuse, such as incineration (37%), agriculture (35%), landfilling (6%), and other applications (12%) (Hušek et al., 2022). Implementing the Urban Wastewater Treatment Directive 91/271/EEC banned the sea disposal of SS (Tyagi and Lo, 2011). The SS Directive 86/278/EEC is the basic EU legislation about SS management which aims to encourage the agricultural application of SS along with establishing the provisions and general requirements to prevent its potential harm to the terrestrial ecosystem and laying down limit values for potentially toxic elements (Milojevic and Cydzik-kwiatkowska, 2021). However, continuous revisions in the SS Directive, based on the investigation impacts of land application of SS on environmental, social, economic, and health indicators, led to more stringent limits on heavy metals and further restrictions on emerging pollutants. Further, SS recycling is governed under the Waste Framework Directive (European Parliament and Council Directive 2008/98/EC on sewage). The priority is to avoid waste, reuse, recycling, or other types of recovery, and ultimately landfill disposal (Balkrishna et al., 2022). The use of SS differs from one European country to another. However, P recovery is being considered for larger WWTPs. Norway is promoting nutrient recovery from SS and developing technologies to improve sustainability (Shaddel et al., 2019). In Sweden, approximately one-third of generated SS is used in agricultural applications. However, in 2016, the Sweden government set up an inquiry to explore the implementation of a ban on agricultural use of SS to reduce environmental risk and to set up new guidelines for P-recovery from SS. The inquiry recommended either a complete ban on agricultural use or limited ban/reuse, allowing land spreading if meeting stringent standards (but not landfill capping or landscaping) (Ekane et al., 2021). The use of SS for agricultural applications is banned in Western members of the EU (mainly France, Germany, and the Netherlands) and prefers the incineration of SS (Coskun et al., 2020). Similarly, Switzerland banned the agricultural use of SS in 2006. Since then, SS has been incinerated exclusively, along with safe storage of ash and flue gas treatment residues in managed landfill sites (Suess et al., 2020). Other countries, such as Denmark, Belgium, Spain, Ireland, and the UK, directly use over 50% of the biosolids on agricultural land. The UK is considering energy recovery opportunities. Furthermore, EU countries such as Germany, Denmark, and the Netherlands promote anaerobic co-digestion of raw SS with organic wastes.

4.3.1.3 Japan

More than 70% of SS is incinerated in Japan, owing to the New Sewerage Law enforced in the 1970s dealing with the mandatory obligation of reducing sludge production and abandoning landfilling due to land unavailability, as Japan is one of the most densely populated regions. Over the last decades, thermal methods such as gasification, incineration, drying, melting, and carbonization have flourished (Wang and Nakakubo, 2021). The residues of the previously mentioned thermal treatments are widely used in construction, phosphorous recovery, and soil amending products or compost in Japan. The objective of these preferred management methods in Japan is to secure its energy sector and achieve a high SS recycling rate along with noticeable minimization (Christodoulou and Stamatelatou, 2016). However, direct land application of SS for agricultural or non-agricultural uses (land reclamation, forestry, landscaping) is not preferred. The government of Japan also has launched innovative projects which involved multiple stakeholders in the implementation and development of new technologies for SS recycling and reuse, such as "SPIRIT21 – Combined Sewer Overflow (CSO) Control Technologies" and "LOTUS, that is, Lead to Outstanding Technology for Utilization of Sludge" (Fukao, 2010).

4.3.1.4 Germany

Agricultural use of SS is widespread in Germany; 14 out of 16 federal states practice agricultural use of SS (Sichler et al., 2022a). However, a continuous reduction has been observed in the share of SS use in agriculture in the last years due to the increased risk of micropollutants and emerging contaminants present in the SS (Milojevic and Cydzik-kwiatkowska, 2021). Therefore, SS disposal is evolving in Germany from agriculture to thermal treatments, especially incineration. Agricultural application of SS in Germany is regulated by the fertilizer law, which defines that organic materials such as SS can be used to produce fertilizers. In addition, in 1992, Germany established the SS ordinance, which compiles the national legislative framework for SS use in agriculture (86/278/EEC Directive). However, overfertilization, particularly intensive livestock farming, contributed to high nitrogen surpluses in Germany and broke the European Nitrates Directive (91/676/EEC). Due to this, Germany was sentenced and sued before the European Court of Justice. As a result, Germany tightened its fertilization laws in 2017 and 2020, which increased competition for land applications between SS and other organics (Sichler et al., 2022a). Furthermore, the amended SS ordinance comprises stringent thresholds and a ban on agricultural SS use for middle-size and large WWTP. Given that, Germany prohibited the agricultural use of SS for large WWTPs from 2029 (>100,000 PE) and middle-sized WWTPs from 2032 (>50,000 PE) (AbfKlär, 2017). Nutrient and energy recovery from SS is considered the best alternative to land application in Germany. Further, due to high phosphorous imports and contaminated reserves, phosphorous recovery from SS is politically intended in Germany (Sichler et al., 2022b). Germany is the first EU to introduce legal amendments for obligatory P-recovery from all SS, which contain more than 2% phosphorous content by 2029 (AbfKlär, 2017).

4.4 ENERGY RECOVERY FROM MUNICIPAL SEWAGE SLUDGE

Energy recovery and biofuel production from SS is one of the modern practices in developed nations through different processes such as AD, pyrolysis, co-incineration in coal power plants, gasification, wet oxidation, energy generation through a microbial fuel cell, and high-temperature hydrolysis. However, incineration and AD are very popular in most countries. Gasification, pyrolysis, and other advanced methods are still nascent. This section comprises a discussion of different energy recovery methods from SS.

4.4.1 Incineration

Incineration is one of the most popular thermochemical oxidative methods in developed nations through which renewable energy can be recovered from organic waste streams, including SS. Electric infrared, multiple hearths, and fluidized bed incinerators are the main incinerators used for SS incineration (Schnell et al., 2020). Among them, the most popular incinerator used for incinerating SS is the fluidized bed reactor in which dewatered sludge is supplied to a bubbling fluidized bed (BFB) combustor. These combustors are powered by supplementary fuels, including crude oil or natural gas, where drying, devolatilization, and combustion occur. Flue gases are exhausted into the atmosphere through stacks after passing through a smoke prevention preheater, an air preheater, a bag filter, a gas cooler, and a wet scrubber. Incinerators can reduce the volume and weight of SS, along with safely destroying pathogens and micropollutants (Vinay et al., 2023, Zhang et al., 2020). Fluidized bed incinerators and multiple hearth furnaces are widely used globally, along with other technologies such as rotary kilns for small applications. Circulating fluidized bed (CFB) and BFB-based technologies are used to incinerate SS. BFB is favored for mono-combustion of SS, whereas CFB is for co-combusting SS with coal, biomass, or other fuel (Kwapinski et al., 2021). The fluidized bed incinerators consist of hot sand mass suspended by air and used to incinerate SS (van Caneghem et al., 2012).

4.4.2 Pyrolysis

Pyrolysis is another thermochemical route to recover energy, in which organic compounds in SS are converted into solid carbonaceous residue in a limited or complete supply of oxygen between

300 and 900°C temperature (reduction and endothermic process) (Li et al., 2022). Generally, fluidized bed reactors are used for the pyrolysis of SS due to their simple construction, easy operation, ease of scale-up, and user-friendliness. However, other pyrolytic reactor units used worldwide for SS pyrolysis include a rotating cylinder and conical spouted bed. For commercial applications, the EnerSludge™ plant was installed in Western Australia, and approximately 45% of energy present in SS was converted into bio-oil; however, the plant was discontinued after 16 months of operation due to expensive production (Goh et al., 2018; Tyagi and Lo, 2013). However, there is no full-scale installation in operation. Various factors affect the pyrolysis process, including temperature, SS composition, heating rate, residence time in the reactor, pressure, and turbulence (Djandja et al., 2020). The pyrolytic products derived from SS pyrolysis include biochar (solid by-product that can be used as soil fertilizer, soil conditioner, and effective sorbent for environmental remediation), bio-oil (condensable vapor used for energy production), and pyrolytic gases (or syngas which is non-condensable gas mixture includes CO, CH_4, CO_2, and H_2). The bio-oil obtained from the process has an HHV of 29–38 MJ/kg and high viscosity. The typical elemental composition is C – 76%, H –11%, O – 6.5%, N – 4%, and S – 0.5%. Moreover, the bio-oil resembles heavy crude oil in terms of C – content (70–80%) (Werther and Ogada, 1999).

4.4.3 Anaerobic Digestion (AD)

AD is another promising route to treat SS, known as bio-methanation, specifically SS with water content >90%, without pre-treatment. This biochemical process, mediated by the collective actions of diverse microbial groups, converts organics present in SS to biogas in an anaerobic condition with a mixture of gases, including methane and carbon dioxide (Khawer et al., 2022). The AD process involves four stages: hydrolysis, acidogenesis (fermentation), acetogenesis, and methanogenesis (Kumar et al., 2021). In hydrolysis, complex organic matter (carbohydrates, lipids, and proteins) is converted to soluble sugars, fatty acids, and amino acids by hydrolytic-fermentative bacteria. The most common phyla of bacteria involved in the hydrolytic fermentation of organic compounds are Proteobacteria, Bacteroidetes, Actinobacteria, Firmicutes, and Chloroflexi. These soluble organics further break down in the acidogenesis process by acetogenic bacteria; *Clostridium, Ruminococcus*, and *Paenibacillus* are the active genera in acidogenesis. In the third step, acetogenic bacteria convert acidogenic products into acetic acid, CO_2, and H_2; the acetate-degrading microbes (syntrophic acetate-oxidizing bacteria [SAOB]) oxidize acetate into CO_2 and H_2. Lastly, CH_4 and CO_2 are produced during methanogenesis by hydrogenotrophic methanogens and acetoclastic methanogens using volatile fatty acids as a food source (Yadav et al., 2022). The commercial mesophilic AD technology is Bio-terminator[24/85] developed by Total Solid Solutions; the process can destroy 85% of total solids in 24 hours at a reactor retention time of 24 hours or less (Tyagi and Lo, 2016). Another commercial AD technology is "Columbus Advanced Biosolids Flow-through Thermophilic Treatment (CBFT-3)," a modified thermophilic AD using a plug flow reactor. This CBFT-3 has an overall energy efficiency of 68–83%, and the process integrates advanced reciprocating engines to generate electricity that supplies 40–50% of the electricity requirement of a plant (Dalke et al., 2021). However, various factors that affect the efficiency and rate of the AD process include pH, temperature, C/N ratio, hydraulic retention time, organic loading rate, process configuration, and inhibitors (Zamri et al., 2021). Further, hydrolysis is considered a rate-limiting step; SS is pretreated to disintegrate the recalcitrant particles along with it to increase the abundance of hydrolytic enzymes and competent microorganisms through mechanically (i.e., microwave irradiation, ultrasonication), (hydro) thermally (temp. range: 50–240°C), chemically (e.g., ozonation), biologically (e.g., hydrolytic enzymes), or any combination of these. Moreover, the inherent low C/N ratio in SS (typically between 6 and 10) can cause a nutritional imbalance in microbes; however, anaerobic co-digestion is increasingly used in which SS is mixed with other organic waste homogenously to achieve optimal C/N ratio (range: 20–30) and improve stable conditions of AD. In conclusion, pre-treatment methods and co-digestion can significantly modify the dynamics and transformation of both contaminants and nutrients in SS.

4.4.4 Other Methods

Gasification is another major route that involves incomplete oxidation of degradable materials in an oxidant-restricted atmosphere in gasifiers. The gasification reactor can be entrained bed, fixed bed, and fluidized bed (Werle and Sobek, 2019). The process produces flammable gases, such as CO, H_2, and CH_4, which can be used to generate electrical or mechanical power. In addition, gasification inhibits emissions of SO_2 and NO_x, polychlorinated dibenzofurans and dibenzodioxins, and heavy metals because N, S, F, and Cl may be released in the form of NH_3, H_2S, HF, and HCl, respectively. Smaller and less expensive gas-cleaning installations are needed when compared to combustion. EBARA-fluidized bed gasification (Japan) co-treats SS with other wastes, such as municipal solid waste, medical waste, fly ash, and plastic waste, for recovering energy-rich syngas. Another commercial method is KOPF gasification technology, which includes (a) a solar drying unit to dry the wet digested SS; (b) a fluidized bed gasification unit operates at 800–850°C; (c) a gas engine unit to generate electricity; and (d) post-combustion chamber for disposing of a surplus gas that cannot be used (Schnell et al., 2020).

WAO is an effective hydrothermal oxidation-based technology in which organic and inorganic contents present in an aqueous solution or suspension undergo oxidation in air or oxygen at elevated temperatures temperature (between 150 and 325°C) and pressure (10–200 atm) at a residence time of 0.25–2 h, either in presence or absence of catalysts (Bertanza et al., 2015). The process produces CO_2, hetero-atom dissolved ions, water, and smaller molecules such as short chains of carboxylic acids. After the wet oxidation process, the chemical oxygen demand is typically reduced by 75% to 90%, along with a solids volume reduction of 90 to 95%. The WAO process is ideal for treating matrices that contain a high concentration of organic matter, such as sludge, waste liquors, and slurries. The benefits of WAO processes are (i) no generation of toxic emissions such as N_2O, SO_2, HCl, dioxins, furans, and fly ash, and (ii) 99% conversion of toxic organics to harmless end products (Hii et al., 2014). The first commercial WAO process was the Zimpro™ process; other commercial WAO processes are Loprox™, Athos™, and VerTech™. The US and Europe had many units of Zimpro™ (Roy et al., 2010), and the Athos™ by Veolia Water is currently commercially provided main WAO (Hu et al., 2020).

4.5 BIOSOLIDS REUSE FOR SOIL AMENDMENT, RESTORATION OF DEGRADED LAND, AND MINE SITE REHABILITATION

The land application of biosolids consists of spreading, injection, spraying, or incorporating biosolids/biosolid-compost/pelletized biosolids onto or below the land surface. Raw SS must be treated biologically (anaerobic or aerobic), chemically (e.g., lime stabilization), and thermally to remove pathogens and contaminants. Notably, high pathogen reduction and stabilization are required for the land application of biosolids (Buta et al., 2021). In countries such as Australia (75%) and Europe (50%), the most used biosolid disposal method is soil amendment/fertilizer in arable crops (Mohajerani and Karabatak, 2020). Land application of biosolids improves site productivity due to increased soil organic matter and fertility. The other advantages of using biosolids include (i) decreasing bulk density, (ii) improving soil structure, (iii) increasing soil moisture retention, (iv) increasing soil porosity, and (v) improving hydraulic conductivity (Urra et al., 2019).

Restoration of contaminated soils (e.g., heavy metals) and mine abandoned land are the other opportunities for biosolids reuse. For instance, Mora et al. (2005) performed in situ remediations of heavy metal-contaminated soil in Spain. They found an increased soil pH with reduced heavy metal availability (As, Cd, Cu, Zn, Pb). Further, biosolids can also support the phytoremediation of heavy metal-contaminated soils, resulting in increased plant biomass and microbial communities in soil due to the breakdown and leaching of organic matter present in SS. Further use of bioenergy crops has been considered a sustainable option with economic viability for the phytoremediation of contaminated soils (Nunes et al., 2021).

Mine spoils are generally characterized by extreme pH, compacted and high bulk density, extremely low water holding capacity, toxic metal contamination, low organic carbon and nutrient contents, and low cation exchange capacity (CEC) (Ghosh and Maiti, 2020). SS comprises high concentrations of organic matter and nutrient content, including N and P, making SS a potentially inexpensive organic ameliorant for mine spoil rehabilitation (Petrova et al., 2022). Incorporating biosolids enhances many physical properties of soil, including high porosity and aggregation, increased infiltration rate, increased water holding capacity, and hydraulic conductivity, maintaining soil texture, reduced erosion, and lowered bulk density and temperature (Wijesekara et al., 2016). In addition to physical characteristics, biosolid application also improves the chemical characteristics of soil, such as pH, organic matter, electrical conductivity (EC), nutrient contents, and CEC. Lower pH of acidic tailing mine spoils experiences increases in the pH with biosolids application (Mingorance et al., 2014), whereas high pH of alkaline tailing mine spoils encounters decreases in pH. Furthermore, biosolids can increase the organic matter in degraded land, accelerating the rehabilitation of the mine site (Carabassa et al., 2018; Mingorance et al., 2014). Biosolids application is also observed to enhance the EC of soil due to an increase in soluble salts; however, it also can reduce EC due to the immobilization of metal ions and the leaching of soluble salts. Moreover, biosolids also contain mineral particles, clay, and organic colloids, increasing the CEC of degraded lands (Gardner et al., 2010). Biosolids can increase the nutrient content in the soil, such as N, P, and Ca, ultimately increasing the soil fertility of degraded land. Combining biosolids with chemical fertilizer can accelerate the mine site rehabilitation process.

Biosolids are reported to create an energy-rich soil environment favorable for the soil microbial community. In Canada, denitrifies, iron reducers, total anaerobic heterotrophs, total aerobic microorganisms, and sulfate reducers were increased in the mine tailing site rehabilitated with anaerobically digested biosolids (Gardner et al., 2010). However, the combined use of biosolids and chemical fertilizer shows synergistic impacts on soil biological properties. For instance, Li et al. (2013) reported an increase in total microorganism population, total nitrogen, organic matter, soil biological fertility, and available N, P, and K; when open-cast mining areas were amended with SS and nitrogen fertilizer. Furthermore, stabilized biosolids increase microbial enzyme concentrations such as dehydrogenase, alkyl phosphatase, arylsulfatase, and b-glucosidase (Mingorance et al., 2014). In northwest Spain, copper mine soils in Touro are physio-chemically degraded and contaminated by copper and chromium. SS was use to rehabilitate the soil, resulting in increased fungal and bacterial abundance along with microbial functions (urease, glucosidase, arylsulfatase, glucosaminidase, and xylosidase content). In addition, amending with SS decreased the activities of specific enzymes. It increased the ratio C_{mic}:N_{mic}, ultimately enhancing the efficiency of the soil microbial community in metabolizing C and N (Asensio et al., 2013).

4.6 NUTRIENT RECOVERY FROM SEWAGE SLUDGE

SS contains a substantial concentration of plant nutrients, especially phosphorous (P). These macronutrients are available in the form of proteinaceous material. The breakdown and solubilization of SS biomass and its subsequent conversion to phosphates and ammonia could be used in producing plant fertilizers such as magnesium ammonium phosphate (MAP). As MAPs have a very slow nutrient release rate, it is also considered an effective fertilizer; nevertheless, ammonia and phosphorus release rate in the soil depends on the size of their crystals (Min et al., 2019).

Different methods have been used for P-recovery from SS and SS ash, such as wet chemical, precipitation, thermochemical, and metallurgical. These convert insoluble phosphate compounds into plant-available forms or leach phosphorous to produce phosphoric acid. The wet chemical technology involves the dissolution of SS with acid or base (under temperature and pressure, if required) in which most of the metals are re-dissolved. Phosphorous can be recovered from phosphorous-rich water after removing the insoluble components through various processes such as precipitation, reactive liquid-liquid extraction, ion exchange, or nanofiltration. This process can also be used to recover phosphorous from SS ash. However, the process has some challenges: (a) large amount of chemicals is required for leaching, (b) separation of other metals leached by acid, (c) precipitation of phosphorous in desired bioavailable form, and (d) amount of SS ash that was treated is lesser than the amount of wet solid residues for disposal (Herzel et al., 2016). In thermochemical treatment, SS ashes are treated with chlorine compounds such as magnesium chloride or potassium chloride and exposed to high temperatures (>1000°C) (Jeon and Kim, 2018). Under high temperatures, heavy metals vaporize as metal chlorides which are further captured at flue gas treatment and transform the phosphate-bearing mineral phase into a plant-available form. Carbonization or metallurgical processes are another route to recover phosphorous from SS through the high-temperature metallurgical process. The process either reduces phosphorous to its elemental form that is separated in an inductively heated shaft furnace through the gas phase or transfers phosphorous into slag by reductive smelting above 1450°C in a shaft furnace (Aragón-Briceño et al., 2021). The challenges involved in the commercial application of this process are technical challenges due to the distribution of phosphorous over several output mass flows (slag, gas, and metal), and the controversial fertilizing effect of carbonizates (Sichler et al., 2022a). Another route is the precipitation/crystallization process which is widely used; it involves precipitation/crystallization of phosphoric minerals present in SS, in the form of struvites, hydroxyapatites, or calcium phosphates, suitable for direct application in agriculture (Kim et al., 2015). The commercial techniques for P-recovery from SS include Aqua-Reci™, BioCon™, and AshDec™. The Aqua-Reci™ was developed for phosphorous and energy recovery in Sweden and used combined supercritical water oxidation. This process reports 100% phosphorus recovery when SS is treated with H_2SO_4 or HCl at 90°C for 2 hour (Stendahl and Jäfverström, 2004). Another commercial technology is the OSTARA™, operational in Edmonton, Canada, which recovers struvite from SS using magnesium chloride with an 80–85% efficiency. BioCon™ technology was developed and studied on a pilot scale in Denmark, which involves phosphorous recovery in the form of phosphoric acid through thermal treatment of SS ash at 850°C. AshDec™ is a thermochemical method in which phosphates are recovered from SS ash in a rotary kiln. In Japan and the Netherlands, crystallization is widely used for phosphorous recovery (Cornel and Schaum, 2009).

N-recovery has received less attention than P-recovery due to lower economic motivation and operational needs. Moreover, it is cost-effective only when the product ammonia has instant use. The ammonia recovery process is a commercial method for recovering nitrogen in which an ion-exchange unit concentrates ammonia in the influent from about 1000 to 15,000 ppm, followed by vaporization of the concentrated ammonia stream. After vaporizing, the ammonia gas comes into contact with sulfuric acid, forming ammonium sulfate (Nazari et al., 2021). However, the N-concentration in wastewater depends on the type of SS and the extent of wastewater treatment. Approximately 30% of the N in a side stream, which accounts for 4% of wastewater, can be recovered (Winkler and Straka, 2019). This concentration is significantly less than the need for inorganic fertilizers in agriculture; thus, N-recovery from wastewater poses a more significant sustainability challenge. The Haber-Bosch process synthesizes ammonia from N_2 gas (from the atmosphere) and H_2 (from natural gas), making it energy-intensive. Furthermore, it is technically possible to bypass the nitrogen cycle; however, it is not promising from an energy point of view. In addition, a large amount of chemicals is required, making the present and future technologies non-competitive from the cost perspective. N-recovery may, however, be economical and sustainable in some scenarios when residual chemicals or waste heat are available. Therefore, it is relevant to consider more sustainable and energy-efficient pathways for N-recovery and to synthesize N-fertilizers, which comprises of interventions in the present (anthropogenic) N-cycle. In addition, these interventions should address and satisfy GHG regulations and goals. Further, innovative approaches are required with lower usage of energy, heat, and/or chemicals to make N-recovery more competitive and feasible from side streams in the future.

4.7 OTHER BIOSOLID APPLICATIONS-CONSTRUCTION SECTOR

Through thermal treatments, the organic and inorganic complexes present in the SS can be transformed into valuable end products such as brick, cement, slags, and artificial lightweight aggregates (ALWA). Considerable work has been carried out, especially in Japan, to manufacture valuable products. Biosolids ash can be used to produce brick, upon mixing with clay or on its own, with similar physical properties and appearance to standard building bricks. The first full-scale SS brick plant with a production capacity of 5500 bricks per day using 15,000 kg of incinerated SS ash was commenced in Tokyo in 1991 (Tyagi and Lo, 2016). "Bitublocks" made up of mixing waste such as SS, incinerator ash, metal slag, and recycled glass with binder bitumen were considered to be almost six times stronger when compared with traditional concrete blocks (Forth et al., 2010; Tyagi and Lo, 2016). Another approach to use valuable components of SS is the production of Portland, in which SS can be used in three different forms, that is, dried SS, dewatered SS, and incinerated ash (Rulkens, 2008). Out of the three forms, the use of dewatered SS into Portland cement kilns is the most preferred in Europe as it does not need new incinerators (Chatziaras et al., 2016). Heavy metals become immobilized in cement at high temperatures, and toxic organic pollutants will be completely oxidized (Rulkens, 2008). Therefore, SS can be used as a raw material in Portland cement manufacturing, reducing the burden of limestone and clay.

ALWA are manufactured in a centrifugal pelletizer by blending ash, binder material (such as alcohol distillation waste), and water. The produced pellets are then dried at 270°C for 7–10 minutes, followed by heating in a fluidized bed kiln at 1050°C for a few minutes to obtain the final aggregate material (Gherghel et al., 2019). ALWA produced from SS can be used in various applications such as flowerpot additives, planter soils, a replacement for water-infiltrating pavement, heat-proofing panels, fillers for clearance between kerosene storage tanks and room walls, substitute of anthracite media in walkways pavement as well as in rapid sand filters (Lee et al., 2021). Slag is a marble-like mineral with a semi-crystalline structure. Two types of slags can be produced from SS, that is, air-cooled and water-cooled slag, to reduce waste volume and immobilize heavy metals from SS. These produced slags can be used as an alternative to natural coarse aggregates such as roadbed material, interlocking tiles, ready-mixed concrete aggregates, back-filling materials, and other secondary concrete products. GlassPack™ is a vitrification process developed in the US to produce an inert glass aggregate product from inorganic (ash) fraction and uses the organic fraction of SS as a fuel source (renewable). In this process, wet sludge is pre-dried to reduce moisture by less than 15%; these dried solids are subjected to a high temperature between 1330 and 1500°C, at which inorganic fraction (ash) melts into the molten glass (Tyagi and Lo, 2013).

4.8 TECHNO-ECONOMIC COMPARISON OF DIFFERENT SLUDGE TREATMENT TECHNOLOGIES

Methods such as biogas production to generate electricity, conversion of biosolids to biofertilizer for land application, incineration, nutrient recovery, and substitution or supplement biosolids in engineering applications as raw materials were found to be promising to manage SS sustainably (Poinen and Bokhoree, 2022). Incineration of sludge requires an advanced emission control system to comply with stringent flue gas emissions requirements. However, co-firing SS with biomass/coal appears to be a favorable option. Pyrolysis of SS is a complex technology compared with incineration, but it entitles more potential benefits such as lower cost, zero waste method, and less environmental impact. These features endow SS pyrolysis with adequate capacities to be stepped up in the near future for commercial applications. SS conversion technologies such as pyrolysis and gasification are still nascent. Claims such as economic viability and pollution free are based on pilot scale demonstrations; however, when scaled up to real-world applications, plants fail to meet stringent air quality and emission standards.

Land application of biosolids is one of the cost-effective methods, but biosolids may contain toxic pollutants which are not degraded or volatilized in the sludge treatment method. Although the concentration of these pollutants is low but can harm the environment; therefore, care must be taken when using biosolids frequently or at high rates. Furthermore, the reuse of biosolids in various engineering applications would be more sustainable in terms of energy and environment, reduce emissions, and immobilize heavy metals. Different studies have investigated using biosolids in various engineering applications as a raw material substitute. The results highlighted that good qualities of lightweight aggregates, bricks, and cement were achieved with biosolids addition. However, lack of social acceptance and imposing more stringent limit values on waste pose significant obstacles to using biosolids in engineering applications. In addition, nutrient recovery technologies have experienced rapid advancement in past years due to increased environmental awareness, operational benefits, stringent discharge limits on these nutrients, and supply security. Despite having huge potential for better nutrition management, nutrient recovery confronts various business challenges, including legislative challenges, marketing of recovered material, and public awareness. In summary, reuse alternatives of biosolids are significant, lessening the high energy associated with incineration. However, lack of social acceptance and business challenges hinder the promotion of biosolid reusing options. SS management options given by developed nations can serve as references for developing nations. Table 4.3 provides a techno-economic comparison of different methods for beneficial use of SS for various applications, including land applications, energy recovery, nutrient recovery, and in the construction sector, along with their status in developed and developing countries.

4.9 CONCLUSIONS AND PERSPECTIVES

The production of a large volume of SS, more stringent criteria for sludge disposal, and the banning conventional disposal methods have compelled developed nations to realize the resource recovery potential of municipal SS. The US, EU, and Japan are the leaders in beneficial SS use. Land application is the most preferred SS management method; however, the presence of micropollutants and biological pollutants, along with possible toxic effects, have obligated developed nations to move toward energy and nutrient recovery. AD is the most preferred technology for energy recovery due to its low capital cost and drying requirements but suffers from long processing time and poor efficiency. In contrast, thermal conversion processes (incineration, pyrolysis, gasification) have improved efficiencies and are faster but are energy-intensive; further, they require ash disposal strategies, expensive emission control (flue gas treatment in incineration), and downstream gas treatment (syngas treatment in gasification). Therefore, more innovations are required in the existing pathways to deal with the limitations. Nutrient recovery technologies have experienced rapid advancement, but nutrient recovery is confronting various challenges, such as legislative barriers, marketing of recovered material, and public awareness. In addition, reusing biosolids in engineering applications is considered a worthwhile approach in developed nations that convert waste into valuable material that addresses the sludge disposal issues by reducing emissions, saving energy, and immobilizing heavy metals. Despite the development in the feasibility of technical conversion processes, most techniques are not cost-efficient. Further, a lack of social acceptance also poses a significant challenge in reusing biosolids in engineering applications. In conclusion, advanced SS management must transform toward promising resource recovery opportunities from just a treatment liability, while protecting the environment and public health.

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TABLE 4.3
Techno-Economic Comparison of Different Biosolids Reuse Options and Their Status in Developed and Developing Nations (Collivignarelli et al., 2019; Raheem et al., 2018)

Reuse Options	Advantages	Disadvantages	Status
Energy Recovery Options			
Anaerobic digestion	• Reduction of VSS by 60%, production of methane for bioenergy • No need of large energy requirement; biological process • Accepts organic feedstock with high moisture content • Environmentally friendly and economical	• Solids retention time up to 30 days; long reaction • Large land footprint of digesters • High capital and maintenance cost • Production of odor if influent is high on sulfur • Doesn't remove ammonia nitrogen • Maintenance of temperature in target zone year-round	• Widely used in developing countries • Well established in developed nations
Incineration	• Reduction in volume of waste; elimination of organic pollutants and pathogens • Recovery of 18% of heat input as energy • Potential of co-feeding with biomass • Energy self-sustaining • Economies of scale and automation favor large-scale operations	• Emission of toxic gases and presence of heavy metals in ash/slag require further treatment. • Dilution of metal concentration in final product when SS co-incinerated with coal/food and yard waste • Dewatering and drying are essential • High investment and operational cost	• Fastest growing practice in developed nations • Not practiced in developing nations due to more capital costs
Pyrolysis	• Reduction in waste volume • Conversion of raw and digested sludge into useful products and bioenergy • Neutralizes antibiotic resistance genes and eliminated bioactive compounds • Economies of scale and automation favor large-scale operations	• Requirement of pre-dried SS • Biochar consists of concentrated metal and nutrient • Complex compared to incineration • Dried sludge requirement limits its application; limited technological acceptance	• Under research stage in developed nations • Not used in developing countries
Gasification	• Syngas is marketable product • High energy efficiency; potential of co-feeding with biomass, self-sustaining energy	• Sludge dewatering and drying needed • Gas cleaning for syngas application • Organic pollutants in exhaust stream • High investment and operational cost • Economies favor large-scale operation	• Under research stage in developing nations • Exploited in developed nations

(Continued)

TABLE 4.3 ***(Continued)***

Techno-Economic Comparison of Different Biosolids Reuse Options and Their Status in Developed and Developing Nations (Collivignarelli et al., 2019; Raheem et al., 2018)

Reuse Options	Advantages	Disadvantages	Status
Land Application			
Agricultural and mine spoil sites	• Decrease bulk density; improve soil physical structure, increase soil porosity, moisture retention capacity, and hydraulic conductivity • Increase crop yield, Inexpensive • Potential as a fertilizer and substitute for inorganic fertilizer	• Release of odor • Increased level of contaminants and probable risk from spreading of human pathogen • Reduction in biodiversity in slow and long term • GHG emissions	• Less exploited in developing nations due to reluctance • Less used in developed countries and discouragement of landfilling in most countries
Nutrient Recovery			
Phosphorous recovery	• Improving the quality of compost • Direct application of biosolids after phosphorous precipitation in agriculture • Reduce high energy associated with ashing of biosolids	• Complexity in some processes • Requirement of pre-treatment	• Extensively used in developed nations • Nascent stages in developing nations
Other Applications			
Construction sector (bricks and ceramics, cement production, and lightweight aggregates)	• Brick and ceramics production: low cost, reduction in quarry activities' impact, less impact on mechanical requirements with a low percentage of sludge • Cement: characteristics similar to cement; saving of energy; rapid implementation with low investment cost; sustainable • Lightweight aggregates: meeting strength requirements; saving non-renewable materials	• Brick and ceramic production: high percentage of SS is not recommended in the mixture; increase the degree of shrinkage • Cement: possible emissions of heavy metals, dioxins, and PAHs with change in properties of end product • Lightweight aggregates: requirement of high sintering temperatures	• At research stage in developing nations • Extensively used in developed countries

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5 Application of Sewage Sludge as an Agricultural Soil Amendment

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5.1 INTRODUCTION

Sewage sludge is an indirect product of wastewater treatment plants (WWTPs), and it exists in massive quantities; for example, the Yangtze River Economic Belt (YREB) sewage sludge production is close to 20 million t/a by 2017 (Wang et al., 2020), while Flamme et al. (2018) reported that about 1.8 million Mg dry matter (DW) of sewage sludge is annually produced in Germany from over 9600 WWTPs. In this scenario, the efficiency in treating wastewater and applied technological systems determine the quality and quantity of sewage sludge. It was reported that the amount of sewage sludge can be as much as 3% of wastewater in WWTPs (Buta et al., 2021). The world's unregulated population, urban and industrial development determine the quantity and fate of raw wastewater in WWTP in many parts of the world (Eid et al., 2019), while Zhang et al. (2017) reported that around 45 million megagrams of sewage sludge existing in dry form are annually produced. Its potential value continues to rise as a result of urbanization and pollution, but only as long as sludge's economic usefulness is put to good purpose (Wang et al., 2017a).

Unwise, unscientific and improper disposal of solid wastes may cause serious environmental damage to and degradation of land and groundwater contamination and health of people flora and fauna. This is attributed to the amount of sewage sludge that includes poisonous organic pollutants, biological macromolecules, bacterial cells, diverse transition metals, such as Co, Fe and Ni, inorganic substances, such as Al_2O_3, SiO_2, MgO and CaO, and more than 80% of water (Becker et al., 2019; Fijalkowski et al., 2017; Haiba et al., 2018). Exposure of sewage sludge to wind and sun may result in air pollution since it may emit a peculiar and noxious smell (Xiao et al., 2022). The treatment of sewage sludge in WWTPs accounts for around 50% of the annual operating costs and 40%

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of the greenhouse gas (GHG) emissions because it is a highly energy-consuming process (Gherghel et al., 2019; Kacprzak et al., 2017).

Since essential plant nutrients are involved in sewage sludge, its use in agriculture could be a sustainable approach for waste management compared to conventional disposal methods, including landfilling, agricultural use and composting and thermal processing (Lamastra et al., 2018), which also makes the recycling of such nutrients possible. Moreover, sewage sludge as a biosolid could work as an effective soil amendment for stimulating soil respiration, soil enzyme activity and soil microbial activity by degrading organics (Andriamananjara et al., 2016). This may be attributed to its agronomic properties with a lot of nutrients (macro/micro-nutrients and essential elements), which enables its usage as conditioning materials (Seleiman et al., 2020). After the addition of sewage sludge to the soil, macromolecules, such as lipids, proteins and polysaccharides, could be eventually acetate and CO_2/H_2 under anoxic conditions through a multi-step methanogenic digestion process (i.e., hydrolysis, acidogenesis, acetogenesis and methanogenesis) (Appels et al., 2008). The generated acetate and hydrogen could be used as carbon sources and electron donors to support cell growth and respiration of microorganisms in soil (Marcet et al., 2018). Sierra et al. (2012) also reported that an increase in microbial biomass and activity would be achieved after the introduction of sewage sludge to the soil. Sewage sludge is no longer regarded as a waste product or something to be disposed of and never used again (Rorat et al., 2019). For conducting a zero waste strategy, sewage sludge is increasingly reused or recycled rather than landfilled, which can minimize the huge amounts of waste and thus promote the bioeconomy (Duan et al., 2020).

Apart from this, the use of sewage sludge in agriculture could reduce reliance on chemical fertilizers, which provides intelligent waste management solutions that deliver tangible environmental benefits (Sharma et al., 2017). However, heavy metals may also be contained in sewage sludge, which poses a potential risk to the environment and human health and thus hinders the commercial uptake of sewage sludge in agriculture. The portion and quantity of sewage sludge determine the improvement in nutrient and physical properties of soil. Thus, the application of sewage sludge in croplands could positively contribute to good waste management, reduce the bioavailability of heavy metals and increase crop production (Sharma et al., 2017).

This chapter offers an overview and discussion of the applications of sewage sludge as an agricultural soil amendment, including their comparison regarding technological, economic and social-environmental perspectives. The physico-chemical properties of sewage sludge are also comprehensively summarized while the effects of sewage sludge on soil and plants are evaluated. Environmental issues caused by the application of sewage sludge in agriculture are identified. Technological improvements are a key theme of this chapter.

5.2 PHYSICO-CHEMICAL PROPERTIES OF SEWAGE SLUDGE

The physico-chemical characteristics of sewage sludge are greatly shaped by its treatment technologies and compounds, including moisture, organics and minerals. In this scenario, sewage sludge is mostly processed and neutralized by compaction, stabilization, conditioning and dewatering before its being implemented as soil amendment in agriculture. Digestion can avoid the decomposition of sewage sludge, eliminate odor and reduce the amounts of pathogenic microorganisms, in which organic compounds in sewage sludge can be mineralized while the sludge is stabilized. For this reason, anaerobic and aerobic digestions are mostly used to process sewage sludge. Meanwhile, thermal treatment, including incineration, is also employed to remove microorganisms at a pressure of 0.5–2 MPa and temperature of 120–150°C for reusing sewage sludge as an alternative source of energy. The composting process mainly converts sewage sludge to materials that remediate agricultural soils (Haiba et al., 2017).

Rorat et al. (2019) found that cellulose, organic acids, mineral compounds, fat, protein and volatile solids are mainly contained in untreated sewage sludge, while dehydrated sewage sludge contains around 20–45% of dry matter (DM). Factors, including the origin and quantity of raw wastewater and applied processing technology, influence the composition of raw sewage sludge. The large amount of organic matter (up to 70%) determines the suitability of sewage sludge for agriculture as well as biogenic elements such as phosphorus (approx. 2.5%) and nitrogen (approx. 4%) (Cheng et al., 2014). Such elements and nutrients are essential substrates for the healthy growth and development of soil microbiota and plants, for which they can increase soil fertility (Tyrrell et al., 2019). Moreover, appropriately 45–55% of organic compounds are involved in stabilizing sewage sludge (Rorat et al., 2019). Potentially toxic biological and chemical compounds, including pathogens, persistent organic pollutants and heavy metals that are contained in sewage sludge, may be pathogenic for plants, animals and humans, thus endangering the usage of sewage sludge in agriculture (Iglesias et al., 2018). Microbial loads in sewage sludge can be minimized or completely removed by the common methods (e.g., disinfection and membrane technique) that are used in WWTPs, but the effective elimination of heavy metals in sewage sludge is difficult to achieve (Kang et al., 2018).

Chemical compounds such as metals and antibiotics can temporarily bond on the surface of sewage sludge, in which significant quantities of micropollutants are accumulated in sewage sludge. Moreover, sewage sludge contains DM, which is abundant in organic matter and offers a supportive environment for the adhesion and growth of microorganisms (Andreoli, 2007). Sewage sludge can serve as a biosolid for soil remediation, thus improving the soil properties and crop growth after being processed by biological stabilization, digestion or composting. The use of sewage sludge for agriculture is subject to the presence of organic pollutant compounds and heavy metals. Even though the toxins generated by industrial and municipal accumulation can be eradicated at WWTPs, there is still a wide range of organic pollutants existing in sewage sludge (Morgano et al., 2018). Such organic pollutants include surfactants, pesticides, nanoparticles, perfluoroalkyl substances, benzotriazoles, pharmaceuticals, personal care products, perfluorinated surfactants, polychlorinated biphenyls, polyaromatic hydrocarbon and chlorinated paraffins (Brandsma et al., 2017; Fang et al., 2018; Muñoz et al., 2018). It should be noted here that industrial activities, including the production of leather, textile, fire-fighting foams, paints and coatings and metal plating, contribute to the concentration of perfluoroalkyl substances. This poses a serious danger to ecological life and people's health (Zacs & Bartkevics, 2016).

5.3 EFFECTS OF SEWAGE SLUDGE ON SOIL PROPERTIES AND PLANT GROWTH

5.3.1 Soil Properties

Soil amendment functions to improve soil fertilizer and water retention capacity and reduce the risks related to the transfer of pollutants to recipient organisms or surface and groundwater, which is a long-standing process (Rehman et al., 2018). The application of biochar in soil can: firstly, increase its nutrient retention capacity; and secondly, reduce the migration potential of inorganic and organic pollutants in soil (Callegari & Capodaglio, 2018). The application of sewage sludge as a soil amendment is favored because it contains a lot of organic matter and micro/macronutrients, which can enhance soil microbial activity, serve as a storage place for nutrients, stabilize soil temperature fluctuation, increase water infiltration and soil water-holding capacity (Seleiman et al., 2020). The uptake of elements from soil is influenced by elemental interactions, temperature, water content, cation exchange capacity, aeration, organic matter and pH, in which the pH value could determine the solubility of trace elements. Furthermore, soil organic matter can reduce or increase the amount of cationic trace elements (e.g., Zn and Ni) available for plants to uptake according to the environmental conditions.

Many studies also reported that biochar converted by sewage sludge could be employed to remediate soil, which will certainly diminish the amount of sewage sludge (Karimi et al., 2020; Liu et al., 2021). In particular, the biochar produced by sewage sludge costs less ($US60) than that derived from agricultural feedstock ($US382) and wood-based source materials ($US501) (Gopinath et al., 2021). Chu et al. (2020) explored the retention capacity of fertile soil while applying sewage sludge-derived biochar (SSBC), and these authors reported that the rice N-fertilizer use and soil N retention could be enhanced since ammonia volatilization is inhibited by SSBC. Moreover, increasing the pyrolysis temperature could result in lower biochar yield due to the development of the aromatic structure, loss of H_2O, CO and CO_2, destruction of organic matter and loss of chemically bound H_2O content (Hossain et al., 2011). Co-pyrolysis of sewage sludge with other materials such as wood waste and willow may result in the decreased yield of SSBC (Chen et al., 2019; Kończak et al., 2019). A few years earlier, Frišták et al. (2018) argued that adding SSBC to soil can increase the P content two- or threefold. Moreover, SSBC can convert Cd from an unstable to a stable form, thus enhancing Cd immobilization in soil (Ren et al., 2017). The removal of organic pollutants by SSBC in soil is another concern; for example, Ding et al. (2019) found that SSBC has high affinity for carbendazim with the maximum adsorption capacity of 144.05 ± 0.32 μg/g while pyrolyzing the SSBC at 700°C. However, the addition of SSBC to the soil does not always positively help soil properties, given that some researchers have reported the negative or negligible impacts of SSBC on soil (Xiao et al., 2022).

5.3.2 Plant Growth

Sewage sludge has proven to be useful for plant growth because of the high organic matter content and nutrients (Figure 5.1). To explore the feasibility of sewage sludge as fertilizers, Phung et al. (2020) conducted a pot experiment to compare the fertility of composted sewage sludge to conventional mineral fertilizer. In their study, they discovered that the dose of sewage sludge is double that of mineral fertilizer, in which a similar rice protein content (7.5%) could be obtained. However, the accumulation of As (0.34 mg/kg) in rice grains was observed at the 2.6 g N/pot of sewage sludge. Interestingly, the application of 1.3 g N/pot of sewage sludge did not result in the accumulation of toxic metals, in which the amount of rice protein rose by 25% while the yield was 27% higher. Koutroubas et al. (2020) reported that additional sewage sludge could positively contribute to the growth of plants, which is attributed to the soil water retention enhanced by sewage sludge. Tóth and Moloi (2019) used lime sewage sludge to cultivate corn and sunflower, where it was found that the application of lime sewage sludge is feasible for effective plant growth. More specifically, the roots of sunflower and corn increased by 30 and 22% compared to the controls.

5.4 ENVIRONMENTAL ISSUES RELATED TO AGRICULTURAL USE OF SEWAGE SLUDGE

5.4.1 Antibiotics and ARGs

Due to the last few decades' rapid industrial and agricultural development, large amounts of antibiotics have ended up in treated wastewater and soil environments (Buta et al., 2021). It was reported that some 70% of non-degraded antibiotics are detected in sewage sludge, including sulfonamide antibiotics, macrolide, quinolone and tetracycline (Sun et al., 2019). For example, sewage sludge derived from hospital wastewater contains at least five antibiotics, in which moxifloxacin had the highest concentration (219 μg/kg) (Ashfaq et al., 2016). Moreover, fluoroquinolone antibiotics can persist in soil for up to several years since these compounds are not completely degraded when exposed to light in soil and sewage sludge (Jechalke et al., 2014). At present, WWTPs have the ability to remove antibiotics as long as the treatment technologies are effective. In this scenario, additional antimicrobials can contribute to the rapid emergence of antibiotic resistance (Giebułtowicz et al., 2018) due to the presence of significant amounts of highly biodiverse microorganisms in sewage sludge and wastewater (Karkman et al., 2018). The increasing spread and severity of antibiotic resistance may be attributed to the undefined requirement of antibiotic concentrations in the treated

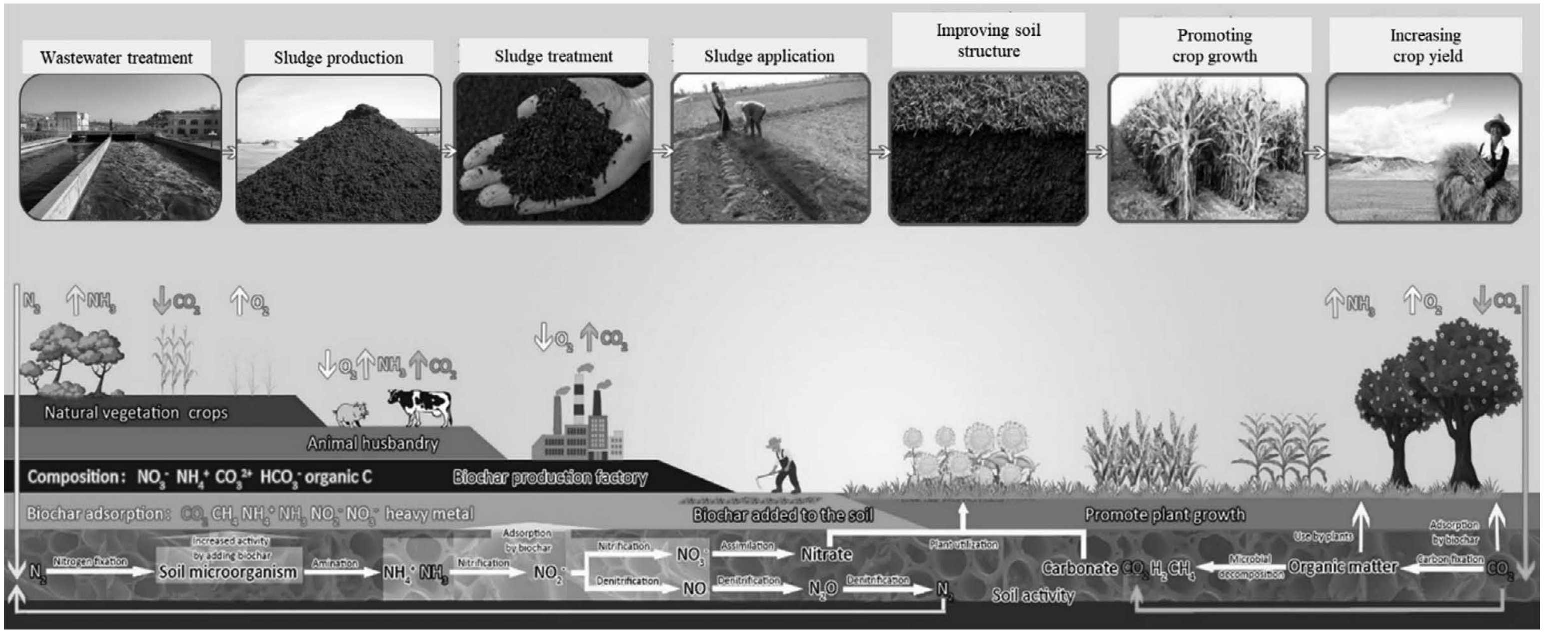

FIGURE 5.1 Effects of biochar added in soil on geochemical cycle of C and N. (Adopted from Zhang et al. 2021.)

wastewater and sewage sludge (Korzeniewska & Harnisz, 2018). The antibiotic-resistant genes (ARGs) in treated wastewater and sewage sludge include *van*A (encoding resistance to vancomycin), tem1 (encoding resistance to penicillins), $bla_{NDM\text{-}1}$ (encoding resistance to many beta-lactams and carbapenem antibiotics regarded as a last-resort treatment for bacterial infections), *erm*B (encoding resistance to erythromycin) and *amp*C (encoding resistance to most beta-lactams) (Guo et al., 2017; Munck et al., 2015).

The antibiotics are released into the soil with sewage sludge, in which the weather conditions, soil microbiome, biochemical properties of soil and soil structure may significantly affect the behavior of antibiotics, such as fixation, degradation and inactivation (Verlicchi & Zambello, 2015). Removing antibiotics may greatly depend on biodegradation and the related mechanisms, while the breakdowns of antibiotics are determined by physico-chemical properties of pharmaceuticals, origin soil bacteria and allochthonous sewage sludge microorganisms (Pan & Chu, 2016). Local conditions, such as seasonal variations, temperature insolation, pH and oxygen concentration, have significant impacts on the persistence of antibiotics in the environment (Barra Caracciolo et al., 2015). In one study, under aerobic conditions, the degradation rates of five antibiotics in soil were in the following order: tetracycline < sulfamethazine <. In one study, < erythromycin < norfloxacin (Pan & Chu, 2016). Miller et al. (2016) reported that some antibiotics are transported via phloem tissue and xylem to seeds, leaves or fruits while some of them end up in roots. Some plants have a high capacity to adsorb antibiotics; for instance, carrot and lettuce have a high affinity for amoxicillin and tetracycline (Pan et al., 2014).

The addition of antibiotics to the soil with sewage sludge may trigger a strong response from living organisms (Guo et al., 2017) because soil microorganisms have to adapt to the environment that may regularly receive antibiotics found in sewage sludge (Cycon et al., 2019). As a result of this, frequent genetic mutations and chromosomal rearrangements as well as the exchange of adaptive genes between microorganisms with different levels of phylogenetic affinity (Felden & Cattoir, 2018). More importantly, the presence of antibiotics and ARGs in sewage sludge may result in the pollution of soil, groundwater and surface water (Tyrrell et al., 2019; Wu et al., 2020). Pruden et al. (2012) reported that infiltration of micropollutants by deep soil layers could be enhanced by rainfall. Furthermore, plants can effectively adsorb antibiotics from agricultural soil through the roots and xylem, which are responsible for the uptake of water and nutrients (Tyrrell et al., 2019). Rahube et al. (2016) reported that the diversity and concentrations of ARGs in crops could be increased through the use of sewage sludge in agriculture. The impact of pollutants contained in sewage sludge used in agriculture is summarized in Table 5.1.

5.4.2 Heavy Metals

Increases in concentrations of heavy metals are triggered by industrial development and waste disposal practices, whereby heavy metals may transfer to sewage sludge during wastewater treatment (Cesare et al., 2015). In this scenario, both soil surface runoffs and industrial wastewater contribute to the large amounts of heavy metals in sewage sludge (Fijalkowski et al., 2017). The major heavy metals that are contained in it include Se, As, Pb, Cd, Hg, Zn, Cu, Ni, Co, Fe, Mn and Cr (Table 5.2). Even though the concentration of heavy metals in sewage sludge rarely exceeds the legal limits, the long-term accumulation of heavy metals in soil may still pose serious environmental risks (Iglesias et al., 2018), including changes in plant morphology and reduction in soil bacteria.

It was reported that the most heavy metals that occur naturally in soil have small concentrations (Song et al., 2017). Agricultural operations may improve soil properties and result in the accumulation of toxic metals, including fertilizer application, soil liming and tillage (Liu et al., 2017). Similarly, the application of sewage sludge to croplands could lead to an increase in the concentration of heavy metals in soil (Marguí et al., 2016). In this case, the heavy metals may transfer to crops and groundwater, while factors including the chemical structure of heavy metals, soil pH, root secretions, cation exchange capacity and soil structure could determine the persistence of heavy

TABLE 5.1
Description of the Impact of Pollutants Contained in Sewage Sludge Used in Agriculture

Pollutants	Results	References
Antibiotics and ARGs	• The abundance of ARGs could be increased by the application of sewage sludge, which may result in the dissemination of antibiotic resistance. • There is correlation between the abundance of ARGs and soil Cu and Zn.	Urra et al. (2019)
Antibiotics and ARGs	• The suggested amount of sewage sludge added to the soil may not increase the number of resistant bacteria or resistance determinants in the fertilized soil and the dosage of sewage sludge has negligible impacts on soil fertility. • Sewage sludge is the source of ARGs	Markowicz et al. (2021)
Antibiotics and ARGs	• The soil physico-chemical properties such as carbon content that are modified by additional sewage sludge highly influence the persistence of antibiotics in soil. • There is a positively correlation between the amount of sewage sludge added to soil and residual levels of antibiotics in soil.	Dong et al. (2021)
Antibiotics and ARGs	• The impacts of additional sewage sludge on the antibiotic resistance development in soil are insignificant, which may be attributed to the low antibiotic concentrations and resistance load of the sludge in Sweden.	Rutgersson et al. (2020)
Antibiotics and ARGs	• The addition of sewage sludge to soil had greater effect on the prevalence of ARGs in phyllosphere than in soil.	Han et al. (2022)
Ag_2O nanoparticles	• The presence of Ag_2O nanoparticles has negligible effects on the plant morphology. • The accumulation of Ag in spinach leaves was observed.	Singh and Kumar (2020)
Heavy metals	• The amount of sewage sludge added to the soil could increase the concentration of heavy metals accumulated in soil. • The bioconcentration factors of the heavy metals in wheat and maize grains are in the order: Zn > Cu > Cd > Hg > Cr = Ni > Pb > As.	Yang et al. (2018)
Heavy metals	• The concentrations of heavy metals in carrot are influenced by soil exchangeable K, rganic matter contents and soil pH, in which the increasing soil pH may inhibit the plant heavy metal concentrations while the soil organic matter content has a positive correlation with the values.	Nahar and Shahadat Hossen (2021)
Heavy metals	• The prolonged applications of sewage sludge as the soil amendment could increase the bioavailable concentrations of heavy metals, which is independently of the type of additional sewage sludge.	Rossi and Beni (2018)
Heavy metals	• The application of SSBC could reduce the bioavailable levels of Cd, Pb and Zn.	Penido et al. (2019)
Microplastics	• The application of sewage sludge containing microplastics may result in the increase the microplastic content in soil with direct soil pollution and subsequent contamination of the environment outside the field.	Corradini et al. (2019)
Microplastics	• Microplastic can transmit organic pollutants and heavy metals.	Li et al. (2018)
Microplastics	• It is essential to understand the fate of plastic in the overall environment through quantifying transport of microplastic within and outside fields. • The concentration of microplastics has no threshold value.	van den Berg et al. (2020)
Microplastics	• The predominant shape of microplastics in the sewage sludge and soil is fibers. • Sources and seasons are the two main factors affecting the abundance of microplastics in sewage sludge.	Yang et al. (2021)
Microplastics	• There is no direct evidence that the microplastics added to the soil with sewage sludge cause direct toxicity to soil.	Edo et al. (2020)

TABLE 5.2
Concentration of Chosen Heavy Metals in Various Sewage Sludge Samples (μg/kg·DM)

	Cu	Zn	Pb	Cd	Ni	Cr
China (domestic sewage sludge)	5.57×10^4–1.1×10^6	3.43×10^5–3×10^6	1.1×10^4–1.03×10^5	0.71×10^3–7.82×10^3	3.86×10^3–1.24×10^5	2.29×10^4–7.37×10^5
(industrial sewage sludge)	6.9×10^4–1.2×10^7	4.02×10^5–3.38×10^6	3.24×10^4–1.09×10^5	0.62×10^3–5.38×10^3	1.19×10^4–2.73×10^5	8.55×10^4–1.44×10^5
Slovenia	1.85×10^5–6.95×10^5	1.03×10^6–2.65×10^6	8.8×10^4–1.75×10^5	1.5×10^3–4.22×10^3	3.72×10^5–9.95×10^5	4.08×10^5–1.235×10^6
France	1.49×10^5–3.4×10^5	5.48×10^5–1×10^6	1.97×10^4–6.2×10^4	0.6×10^3–2.2×10^3	2.64×10^4–4.4×10^4	2.76×10^4–1.2×10^5
Spain	1.31×10^5–4.06×10^5	5.19×10^5–2.47×10^6	4.72×10^4–2.23×10^5	1.68×10^3–9.2×10^3	9.8×10^3–3.66×10^4	2.32×10^4–4.39×10^5
China	1.69×10^5–2.05×10^6	1.69×10^5–6.72×10^6	1.36×10^4–9.37×10^4	0.9×10^3–1.12×10^5	1.58×10^4–2.33×10^5	6.16×10^4–1.844×10^6
Egypt	1.84×10^5–1.38×10^6	3.5×10^5–3.54×10^6	NA	2.3×10^3–3.9×10^3	3.9×10^4–2.71×10^5	1.07×10^5–1.12×10^6

Source: Adopted from Cheng et al. (2014).

metals in plants and soil (Wang et al., 2017b). Plants and crops may absorb heavy metals through their roots, which results in the accumulation of heavy metals in edible plant parts (Jolly et al., 2013). It was reported that various parts of crop plants (wheat) have different adsorption capacities for heavy metals from soil (Shi et al., 2016).

5.4.3 Microplastics

Microplastics may migrate into soil after the application of sewage sludge, which may spread throughout the ecosystem and greatly influence soil properties and plant growth (Figure 5.2). However, the effect of microplastics on soil is not clear in spite of their content ranging from 1000 to 54,000 MP/kg (Li et al., 2018; Zhang & Chen, 2020) Various substances such as flame retardants and chlorinated paraffins may exist in microplastics and these toxic materials may harm soil organisms if they are released to the soil (Nizzetto et al., 2016). For this reason, the presence of microplastics may destroy microbial diversity and soil properties, which results in concerns about food quality and safety (He et al., 2019). Besides, the presence of microplastics may change the relative distribution of aerobic and anaerobic bacteria, which is attributed to changes in the flow of oxygen in soil caused by soil moisture and soil porosity (Rubol et al., 2013). It was reported in one study that the substrate-induced respiration (SIR) rates fell and the microbial community structure was interfered with in the presence of microplastics (Judy et al., 2019). This indicates that adding microplastics could alter the soil microbial function.

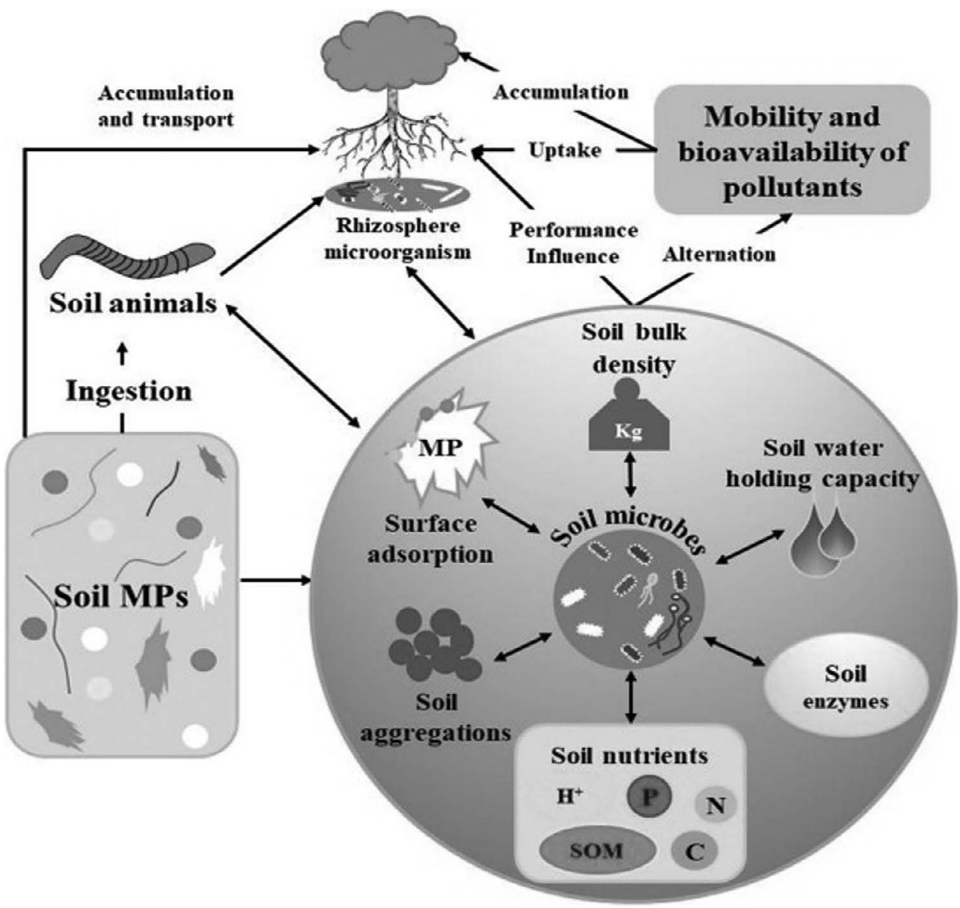

FIGURE 5.2 Impact of microplastics on soil. (Adapted from Guo et al. 2020.)

The emergence of microplastics may result in complex changes in the environmental behavior of other soil pollutants through altering soil properties (Alimi et al., 2018; Wang et al., 2018). Moreover, microplastics could be used as a carrier to adsorb, transport and release various pollutants, such as antibiotics, ARGs, heavy metals and toxic organic chemicals (Gao et al., 2020). In this scenario, such pollutants may diffuse in the soil and thus pose serious ecological and health risks (Li et al., 2018). Hüffer et al. (2019) noted that the adsorption capacity of soil for organic pollutants may decline because of additional microplastics, in which the mobility of organic contaminants in the soil would be improved. This could be attributed to molecular interactions (Hüffer & Hofmann, 2016). Simultaneously, the properties of soil and microplastics could also in turn affect the adsorption characteristics of microplastics (Yang et al., 2019).

Furthermore, Jiang et al. (2019) observed the negative effects of polystyrene microplastics on the hydroponic *Vicia faba*, including the oxidative damage, genotoxic and growth inhibition. In their study, root tips accumulate a large amount of 100 nm polystyrene microplastics. The accumulation of microplastics could seriously compromise the uptake and transport of nutrients by plants since they may trigger a blockade effect on cell wall pores or cell connections (Ma et al., 2010). In contrast, the presence of microplastics has insignificant impacts on the biomass and seedling emergence of wheat, according to one study (Judy et al., 2019). Thus, it is of great interest to conduct more research on filling in the gap in our knowledge on the influences of microplastics on plants.

5.5 CONCLUSIONS AND PERSPECTIVES

In this chapter, an overview of sewage sludge is introduced, mainly focusing on its properties, application in agriculture and possible environmental issues as a consequence of such applications. Since sewage sludge contains large amounts of organic matter and plant nutrients, its land application could positively improve soil characteristics and plant growth. Nevertheless, the agricultural application of sewage sludge may result in the accumulation of toxic substances such as heavy metals. To facilitate the safe application of sewage sludge, future efforts should focus on the following issues:

1. The emergence of bacterial resistance to antibiotics and heavy metals is triggered by mutual interactions between these pollutants. However, what is required is efficient and effective technology that can remove ARGs from sewage sludge. This could avoid the transfer of genes encoding resistance to nearly all antibiotic groups. Moreover, the accumulation of heavy metals in soil may reduce the quality of agricultural soils and result in the inactivation or death of soil microbes in spite of even only small amounts of heavy metals. Therefore, the presence of pollutants such as heavy metals and antibiotics has long-term effects on living organisms and great potential to transfer to the soil environment, posing risks to consumers through the food chain. For this reason, further research should consider the long-term presence of pollutants in treated sewage sludge.
2. Different activating agents can be used to modify sewage sludge to improve its physicochemical properties. For example, the content of fixed carbon in sewage sludge could be increased through pretreatment methods, while the modification of sewage sludge by metal ions or metal oxide could enhance adsorption and catalytic capacity. Apart from this, the cost of stream and gas modification is higher than that of alkaline modification. Thus, an analysis of the modification of sewage sludge should comprehensively evaluate the technological and economic feasibility aspects.
3. The various purposes and origins of microplastics should be included to explore the environmental effects of microplastics in sewage sludge, especially given the fact that microplastics with different product uses, shapes, sizes and types have distinct impacts on soil. Moreover, qualitative or quantitative analyses of soil microplastics may not comprehensively include the toxicity of microplastics in sewage sludge since the dosage and exposure time of microplastics greatly determine the dangers posed by microplastics. For this

reason, more field and laboratory experiments must be conducted to identify the minimum exposure time and concentration of dangerous microplastics and explore the effects of microplastics on soil physical properties and other soil pollutants. Apart from this, investigations on the uptake, translocation and accumulation of microplastics in plants and crops are still in their early stages, so future research should examine the stress responses induced by microplastics.

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6 Sustainable Treatment and Resource Recovery from Sewage Sludge

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6.1 INTRODUCTION

Sewage sludge is the residual, semi-solid material generated during the wastewater treatment process. Recently, the sewage sludge production has increased manifolds due to the increasing urbanization and industrialization leading to better living standards. About 32–52 million tonnes of sewage sludge (80 wt% water content) is produced in China every year which is expected to increase by approximately 10% per year (Hu et al., 2021). The high production with increasing stringent sludge disposal regulations is compelling the treatment facilities to re-examine their sludge management/disposal approaches as they encounter challenges regarding processing, reuse, management and disposal of sludge. The treatments in wastewater treatment plants (WWTPs) are categorized into primary, secondary and tertiary processes for removing the pollutants, facilitating the management of produced by-products and meeting the legislative quality standards (Tyagi & Lo, 2013).

The structural composition and chemical properties of activated sludge are the functions of stabilization techniques and engineering parameters in WWTPs. Sludge consists of various compounds ranging from organic matter, inorganic compounds, micropollutants and microbes. Bacterial constituents such as proteins, lipids and carbohydrates along with inorganic matter construct the chemical structure of waste activated sludge. The traditional methods of sludge disposal include incineration, landfilling, ocean disposal and agricultural soil application. However, these conventional methods each impose a variety of disadvantages, thus shifting the interest towards using the sludge as a renewable resource (Raheem et al., 2018). The growing attention towards renewable energy is fostered from reducing primary energy sources, climate change issues, public awareness and advancements in renewable energy technologies.

The waste activated sludge is rich in organic carbon and nutrients (primarily nitrogen [N] and phosphorus [P]). Thus, the researchers' interest has shifted towards the sustainable sludge management strategies by recovering the energy and resources. Some of the significant routes for sludge are anaerobic digestion (AD) with biogas recovery, incineration, pyrolysis, gasification, wet air oxidation (WAO), hydrothermal treatment (HTL), supercritical water oxidation (SCWO) and use of bioelectrical systems such as microbial fuel cells (MFCs) for the generation of biofuels (hydrogen, syngas, biooil), electricity, heavy metals, nutrients, proteins, enzymes, construction materials and biopolymers. Therefore, this chapter aims at providing a comprehensive overview and discusses

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strategies for sustainable sludge treatment via biochemical, thermo-chemical and bio-electrochemical technologies and recovering energy-rich products.

6.2 WASTEWATER TREATMENT AND SLUDGE GENERATION

6.2.1 Characterization of Sewage Sludge

In 2020, the volume of municipal wastewater that was produced worldwide was estimated to be 360–380 km^3 which was predicted to increase by 24% by 2030 and 51% by 2050 (Di Giacomo & Romano, 2022). In the EU, the quantity of sludge produced is estimated to be more than 10 million tons of dry mass of sewage sludge every year (Grobelak & Jaskulak, 2019). The sewage sludge around the world exhibits wide variations in their properties depending on characteristics of feedwater, amount of ageing, type of processing/unit operations etc. Therefore, the characteristics of sludge vary from one place to another (Mahmoud et al., 2003).

The wastewater undergoes several physical, chemical and biological processes where the solids get separated as sewage sludge. The different processes in WWTPs consist of sedimentation, flocculation, membrane filtration, adsorption, oxidation, biodegradation and electrochemical treatment. Advanced techniques such as electro-coagulation and electro-flocculation have been applied to remove suspended solids as well as emerging contaminants such as micropollutants from wastewaters (Raveendravarrier et al., 2020).

Sewage sludge is a complex heterogeneous mixture of microbes, organic matter, inorganic material and moisture. The organic materials in the sludge consist of a wide range of compounds such as proteins, peptides, polysaccharides, lipids, plant molecules with phenols (lignin, tannins) or aliphatic structure, and micropollutants such as polycyclic aromatic hydrocarbons, polychlorinated biphenyls, adsorbable organohalogens, surfactants, pharmaceuticals and hormones (Kacprzak et al., 2017). The mechanical wastewater treatment process such as screening, grit removal and sedimentation generates primary sludge which usually contains 93–99.5% water, high suspended and dissolved organic solids. The biological treatment of wastewater generates the secondary (or activated) sludge which comprises high concentrations of microbial cells that are complex polymeric organic materials. The total solids concentration of waste activated sludge is between 0.8 and 1.2% w/w depending on the type of biological process used. The waste activated sludge consists of 59–88% (w/v) of organic matter composed of 50–53% carbon, 25–30% oxygen, 10–15% nitrogen, 6–10% hydrogen, 1–3% phosphorus and 0.5–1.5% sulphur. In addition, metal ions such as Ca, Cd, Cr, Cu, Fe, Hg, K, Mg, Ni, Pd and Zn are also found (Kacprzak et al., 2017; Tyagi & Lo, 2013; Yan et al., 2009). The typical chemical composition of primary and secondary sludge is presented in Table 6.1.

6.2.2 Sewage Sludge and Environmental Sustainability

Based on the physico-chemical processes in wastewater treatment, various studies have reported high concentrations of heavy metals, toxic organics, micropollutants, pathogenic microbes, parasites and recalcitrant compounds in sewage sludge (Rizzardini & Goi, 2014), which can cause pollution to environment and affect human health, if mishandled. The nitrogen in activated sludge can interact with soil colloids by various interactions (physical, chemical and biological) and form NO_2 which can be nitrified to form NO_3^- that can ultimately cause photochemical smog, acid rains and deplete ozone layer (Hu et al., 2021). Nitrogen and phosphorus can leach out in the water bodies and can cause eutrophication. The presence of heavy metals such as Cd, Ce, Cu, Fe, Mn, Ni, Pb and Zn in the sludge can result in serious health problems in humans if they bioaccumulate and reach the food chain cycle. Moreover, disease-causing pathogens and antibiotic resistance microbes from sewage sludge can spread and affect various ecosystems (Bondarczuk et al., 2016). Heteroaggregation of microplastics with other suspended solids in wastewater leads to the accumulation of microplastics in sewage sludge. Applications of sludge to soil can further change the functions and structure of soil, influence microbial diversity and soil fertility, and are a threat to human life (El Hayany et al.,

TABLE 6.1
Characterization of Municipal Sewage Sludge (Primary and Activated) along with Pathogens of Concern in Sewage Sludge

Parameter	Primary Sludge	Secondary Sludge
pH	5–8	6.5–8
Total solids (TS) (%)	5–9	0.8–1.2
Volatile solids (% TS)	60–80	59–88
Nitrogen (%TS)	1.5–4	2.4–5
Phosphorus (%TS)	0.8–2.8	2.8–11
Protein (%TS)	20–30	32–41
Grease and fats (%TS)	7–35	5–12
Cellulose (%TS)	8–15	7–9.7
Pathogens	Bacteria: *Campylobacter jejuni, Escherichia coli, Salmonella, Shigella, Vibrio cholerae* Virus: *Astrovirus, Calicivirus, Enterovirus, Hepatitis A, Norwalk, Reovirus, Rotavirus* Protozoa: *Balantidium coli*, *Cryptosporidium*, *Entamoeba histolytica*, *Giardia lamblia*, *Toxoplasma gondii* Helminth worms: *Ascaris lumbricoides*, *Ascaris suum*, *Hymenolepis nana*, *Necator americanus*, *Taenia saginata*, *Taenia solium*, *Toxocara canis*, *Trichuris trichiura*	

2022; Yadav et al., 2023). Therefore, it is important to effectively manage the sewage sludge for its safe treatment and disposal.

The conventional ways of sewage sludge disposal or utilization are landfills, incineration and land farming. Landfilling of sewage sludge has been widely performed in China; however, disadvantages such as limited landfill sites and associated environmental and public health risks are ubiquitous. Incineration can decrease the sludge solids by 70%; however, the process results in release of harmful gases (Hu et al., 2021). The process becomes costly due to cleaning of exhaust gases and dewatering/drying the sludge prior to be incinerated. Utilization of sludge for land farming can further depreciate the soil quality and cause potential risks because of the presence of heavy metals, pathogens and various emerging contaminants. It has been reported that in the U.S., 61% of the generated sewage sludge is applied on agricultural lands, 22% is used for incineration and 17% for landfilling. However, in Europe, 57% of the produced sludge is used for agricultural applications, 20% for incineration, 13% for landfilling and 10% is disposed in seas (Raveendravarrier et al., 2020). As described, all these methods are not completely safe and ecofriendly for sludge disposal. Apart from these methods, the sewage sludge can be used for land reclamation, forestry, industrial processes, resource recovery and energy recovery. Recovering energy and resources through biochemical and/or thermal processes are promising in terms of economic and environmental sustainability and can open avenues for harnessing renewable energy.

6.3 CONVENTIONAL AND ADVANCED METHODS FOR SUSTAINABLE SEWAGE SLUDGE TREATMENT AND ENERGY/RESOURCE RECOVERY

6.3.1 Biochemical Conversion

6.3.1.1 Anaerobic Digestion

AD converts sludge organics to biogas in the absence of oxygen via the reaction given below:

$$C_cH_hO_oN_nS_s + yH_2O = xCH_4 + nNH_3 + xH_2S + (c - x)CO_2$$
$$\text{where, } x = 1/8(4c + h - 2o - 3n - 2s) \text{ and, } \qquad \text{Eq. (6.1)}$$
$$y = 1/4(4c + h - 2o + 3n + 3s)$$

The AD process basically involves four stages, i.e., hydrolysis, acidogenesis, acetogenesis and methanogenesis. In hydrolysis, organic compounds (polysaccharides, proteins, fats) are broken down into simpler compounds such as amino acids, fatty acids and simple sugars. The microbes involved in this stage constitute *Bacteroides*, *Cellulomonas*, *Clostridium*, *Erwinia*, *Firmicutes*, *Microbispora*, *Prevotella* and *Ruminococcus*. In acidogenesis, hydrolysis products are further converted to shorter volatile fatty acids (VFAs), ammonia, H_2S and CO_2 by acidogenic bacteria (*Clostridium*, *Lactobacillus*, *Geobacter*, *Bacteroides*, *Eubacterium*, *Desulfovibrio*, *Desulfobacter*, *Rhodopseudomonas*, *Sarcina* sp.). The third step, i.e., acetogenesis consists of breakdown of short-chain organic acids and alcohols (generated during acidogenesis) into acetic acid, CO_2 and H_2. The bacterial species involved in the step are *Syntrophobacter*, *Syntrophus*, *Syntrophomonas*, *Syntrophothermus* and *Pelotomaculum*. Lastly, in methanogenesis, biogas is generated from hydrogen, formate and acetate via acetoclastic and hydrogenotrophic methanogenesis. The acetoclastic methanogenesis is conducted by *Methanosarcina* and *Methanosaeta* sp. by converting acetate and water to methane. In hydrogenotrophic methanogenesis, *Methanobacterium* and *Methanoculleus* sp. form methane by reaction of CO_2 and H_2 (Liew et al., 2022).

The biogas is used as an energy source for production of heat and electricity. It comprises 60–70% CH_4, 30–40% CO_2 and trace amounts of other gases such as H_2, N_2 and H_2S. Biogas has a calorific value of 13–21 MJ/kg (lower than coal and equivalent to lignite) and a relative density of 0.85. The energy acquired as biogas from sewage sludge can cover up to 50% of the total operation costs of WWTP. Digestate is also generated as final product after AD which contains high amounts of nutrients (N and P) that can be used as fertilizer or compost (Cecconet & Capodaglio, 2022; Raheem et al., 2018).

Batch and continuous-flow reactors have been used for AD of sewage sludge. The cost of operating batch anaerobic digester is high despite its low initial investment cost (Deublein & Steinhauser, 2011; Esposito et al., 2011). Based on the amount of water in the pre-processed sludge, the AD process can be wet or dry. In a single-phase digester that has operated in mechanical horizontal mixing or vertical plug flow configuration, dry AD typically takes 3–4 weeks (Devlin et al., 2011). Because of the presence of lignocellulosic material, a longer hydraulic retention time is needed during the digestion process for agro-waste to achieve higher degradation and biogas yield. The main elements that need to be looked into to improve the reactor efficiency are the temperature, digestion time and other control strategies (Traversi et al., 2015). Based on the types of anaerobic microbes used, AD process can be carried out at temperatures between 12 and 60°C (Cao & Pawłowski, 2012; Yadav et al., 2022).

The organic materials in waste activated sludge are immobilized and therefore, organic matter disintegration is required for the AD process. Various pre-treatment procedures including mechanical, thermal and chemical processes have been used to enhance the conversion of biomaterials in sludge to soluble fractions that in turn accelerates the AD process. Appels et al. (2013) reported an increase in soluble chemical oxygen demand (sCOD) by 214% that enhanced the biogas production by 50% during mesophilic AD when microwave pre-treatment was applied to waste activated sludge (336 kJ/kg sludge). Another study investigated the effects of alkaline and acidic chemical pre-treatment of waste activated sludge in AD (Tulun & Bilgin, 2019). Alkaline pre-treatment resulted in higher sCOD as compared to acidic treatment which could be because of the formation of refractory compounds during acidic treatment leading to reduced sCOD as these compounds don't degrade easily. The highest biochemical methane potential was obtained at pH 10 with methane yields of 43.61%. Table 6.2 summarizes the main findings of these pre-treatment studies.

Compared to pre-treatment methods, inter-stage treatments have been reported to produce higher methane yield. One such study compared several pre-treatments, for example, thermal and thermo-chemical conditions before AD with inter-stage treatments under similar conditions (Nielsen et al., 2011). The results indicated that inter-stage treatment was more efficient for AD of sewage sludge as compared to pre-treatment. The inter-stage treatment increased the methane production by 29

TABLE 6.2
Summary of Studies on AD of Waste Activated Sludge

Reactor Configuration	Results	Remarks	References
Continuous stirred tank reactor with volume of 2.5 L	Reactor instability was seen to increase the quantity of volatile fatty acids, reduce the concentration of partial alkalinity and alter pH.	Improved biogas generation and a positive energy balance were the results of a short sludge retention time (SRT) and high VS destruction efficiency.	Nges and Liu (2010)
Continuous stirred tank reactor with Volume of 0.9 L	Bacteroidetes increased from 12.5 to 20%, indicating a significant shift in the bacterial community between 20 and 4 days.	The lower SRTs are proactive signs for defining rate limitation in AD process.	Lee et al. (2011)
Batch reactor, Reactor volume: 1 L	There were higher intakes of both protein (167 g COD/kg total solids) and carbohydrates (666 g COD/kg total solids).	The optimum conditions for CH_4 production were pH 12 and C:N ratio 17:1.	Dai et al. (2016)
Batch reactor, Reactor volume: 280–300 mL	At 80°C, pre-treatment had no impact on CH_4 output, whereas post-treatment resulted in a 20% increase. Additionally, inter-stage treatment produced improvements of 9% at 130°C, 29% at 170°C and 28% at 170°C/pH 10.	When used as an inter-stage treatment rather than a preliminary treatment, the thermal treatment appears to be more effective.	Nielsen et al. (2011)

and 28% when thermal treatment (170°C) and thermo-chemical treatment (170°C, pH 10) were applied, respectively. Only 9 and 2% methane yield increased when thermal and thermo-chemical pre-treatment was used, respectively.

Digestion of sewage sludge in combination with another organic waste in AD (co-digestion) has also been reported (Liew et al., 2022). For AD, the optimum C/N ratio ranges between 20:1 and 30:1. The sewage sludge with lower (C/N) ratio will cause formation of ammonia, thus increasing pH above 8.5 which is not favourable for methanogenesis. By using a co-digestion, the carbon can be counterbalanced, thus resulting in better AD. Co-digestion of sewage sludge has benefits such as higher volatile solids reduction, increased biogas yield and higher heavy metal stabilization. Co-digestion of sludge and organic fractions of municipal solid waste led to an increase in biogas production up to 27% by adding 20% solid waste in system (Liew et al., 2022). AD process requires low capital costs than thermal treatment processes; however, the reaction time is higher in AD compared to other non-biological methods.

6.3.2 Thermo-chemical Conversion

6.3.2.1 Incineration

This process involves complete oxidation of organic compounds at temperatures >750°C, resulting in flue gas, inert material (ash) and heat. The sludge solids are burned in the presence of oxygen in a combustion chamber, thus reducing 90% sludge volume (Raheem et al., 2018). The process involves release of toxic exhaust gases in the environment which can be controlled by installing gas scrubbers. In the process, the metals are concentrated in the produced sludge ash. Based on the temperature and reaction time, the metals in the ash can range between 50 and 97% (Mulchandani & Westerhoff, 2016). The produced ash can be disposed in landfills or used to produce building materials in cement industries.

The energy recovery (heat or electricity) from the sludge determines the efficiency of sludge incineration process. Around 18% of the total heat input in the process is recovered from incineration of sludge at 20% solids which can be used in the plant for sludge drying before incineration or to produce electricity. The metropolitan WWTP in the U.S. (St. Paul, Minnesota) has implemented power generation with incineration technology of 3.5-MW electricity generation capacity. Other incineration facilities have also been reported in Ohio and Connecticut, U.S., with generation capacities of 2 and 0.8 MW, thus providing 20 and 40% plants' energy needs, respectively (Tyagi & Lo, 2013). The fluidized bed has been reported efficient for activated sludge incineration in the dry or wet phase (35–59 wt% water) with high combustion efficiency (<0.3% of organic material is ash) and low generation of SO_x and NO_x. Murakami et al. (2009) proposed the combined use of pressurized fluidized bed combustor and turbocharger driven by flue gas. Compared to conventional plants, >50% of pollutants (CO, NO_x and N_2O) were reduced in flue gas. The CO_2 emissions and costs for fuel and electricity decreased by 40% and 0.2 million dollars, respectively. As compared to conventional plant, 50% energy savings were obtained at incineration capacity of 100 tonnes/day.

It has been reported that enhancing the dewatering and drying of sludge before incineration and using low-caloric surplus heat from exhaust gases can increase the energy recovery from sludge (Hao et al., 2020). Co-incineration of sludge with other substrates such as coal and solid wastes can be performed to generate energy from multiple sources in a cost-effective manner and reduce GHG emissions; however, the process dilutes the metal concentration in sludge which can be extracted (Liang et al., 2021; Mulchandani & Westerhoff, 2016). Therefore, to develop a sustainable and cost-effective incineration process with resource recovery, future research on co-incineration of sludge with other feedstocks, designing energy efficient incinerators and management of produced ashes, is required.

6.3.2.2 Pyrolysis

In pyrolysis, activated sludge is thermally treated at 350–900°C under pressure without oxygen. The process yields char, ash, pyrolysis oil, combustible gases and water vapour. Solid and/or gaseous products can be incinerated and be used as a heating source.

Pyrolysis is characterized by heating rate, temperature and gas residence time and based on these operating conditions, the process can be used to obtain char, liquid or gas. The pyrolysis process conducted at low heating rate (0.1–1°C/s), low temperature (300–400°C) and high gas residence time (5–30 min) is known as slow pyrolysis, whereas fast pyrolysis occurs at higher heating rate (10–200°C/s), higher temperature (450–600°C) and shorter gas residence time (0.1–0.3 s). The major product obtained during fast pyrolysis is biooil/pyrolysis oil, which can be used as fuel. It has been reported that lignocellulosic pyrolysis yields around 60–75 wt% liquid, 15–25% char and 10–20% gas. The produced char and gas can be reused in the process as fuels (Fonts et al., 2012). Oil-from-sludge-based pyrolysis increases generation of biooil at 450°C for 30 min under atmospheric pressure to produce straight-chain hydrocarbons which are condensed into oil. Another technique called Siemens Schwell-Brenna Technology is used to generate biooil via pyrolysis of waste activated sludge with co-substrates (crushed wastes) at 450°C in rotatory kiln (Raheem et al., 2018).

Generally, fluidized beds are used in pyrolysis because of easy operation and readily scale-up. SlurryCarb™ is a commercially running pyrolysis installation in California, U.S. operating at a high temperature of 450°C. Thermally treating the biosolids in pyrolysis releases CO_2 and reduces mass of solids by 40%. These carbonized solids are made into slurry and thermally dried and pelletized into solid fuel which can be combusted in boilers or used as fuel. The plant produces a net energy of 2100 kWh/ton dry solids (Tyagi & Lo, 2013).

The process parameters, treatment temperature, reaction time, pressure and sludge characteristics affect the pyrolysis products. It was reported that the biochar after slow pyrolysis had around 40% energy content, whereas the biooil generated after fast pyrolysis had 60% energy content (Naqvi et al., 2021). Researchers have studied the influence of reaction temperature on sewage sludge pyrolysis (Trinh et al., 2013). The highest oil yield from sludge was obtained at 575°C temperature with a higher heating value (HHV) of 25.5 MJ/kg resulting in the reduction of sludge volume to 52% on a bulk volume basis after pyrolysis. The char obtained contained ash content of 81 wt% and HHV of 6.1 MJ/kg dry basis. Around 95% of the sludge phosphorus was also recovered in char. Another study reported that the biooil from sewage sludge had properties similar to heavy fuel and had two times HHV as compared to biooil from lignocellulosic biomass (Arazo et al., 2017). Furthermore, biooil from sludge has neutral pH that could minimize the corrosion-related problems in reactors and pipes.

Recently, microwave with catalytic pyrolysis has invoked interest for achieving higher yields of the products and energy. In microwave-assisted pyrolysis, the heat transfer direction is from inside to outside which is dispersed throughout the material volumetrically as compared to conventional pyrolysis where the inside region of particles remains cool leading to lower products' yield (Naqvi et al., 2021). Lin et al. (2017) showed microwave heating process for wet-sludge pyrolysis to be promising with maximum oil yield of 33 wt% with respect to dried sludge (oil heat values were 8700–9200 kcal/kg at 400–800°C) and highest energy recovery in oil of 54 wt% at 600°C. This approach combined dehydration and carbonation processes. At high temperature, yield of oil and char were reduced and gaseous products were enhanced.

Various studies have been done on heavy metals distribution in different products after pyrolysis of waste activated sludge. One such study focussed on heavy metals concentration in char and biooil produced from sewage sludge in screw-feeding reactor. The researchers reported that most of the heavy metals concentrated in the char and very low heavy metal concentrations were obtained in biooil (Gao et al., 2017). Another study reported that the accumulated heavy metals in char at 600°C were in oxidable and residual forms resulting in decline in their bioavailability, thus leading to low toxicity to the environment when applied to soil (Jin et al., 2016). Pyrolysis produces liquid biooil that can be easily stored, transported and used as fuel, which is an advantage over other thermal treatment methods. Pyrolysis flue gas requires less clean-up as compared to incineration to satisfy the emission limits. Pyrolysis oil from lignocellulosic biomass has been tested to be used as direct fuel for engine, turbines and boilers (Fonts et al., 2012). However, the major disadvantages of this process include complex processing, limited operating data and no cost analysis of process. Sewage sludge pyrolysis technology is a potential technology for resource recovery; however, it is still in early stages of application. Therefore, methodologies for pre-/post-treatment, development of new sorbents, optimization of operation parameters and minimization of heavy metals in products must be introduced.

6.3.2.3 Gasification

Gasification converts dried sewage sludge into ash and combustible gases at high temperatures of 700–1000°C in the presence of reduced oxygen (partial oxidation). Heat and syngas (composed of H_2, CO, CO_2 and CH_4 with calorific value of 4–6 MJ/Nm^3) are the products of the process. The gasification of sludge involves four phases: drying, pyrolysis/devolatilization, combustion and char reduction (Raheem et al., 2018). The reactions involved in these steps are given in the following equations:

$$C + O_2 = CO_2\,(-4.6\ \text{kJ/mol}) \qquad \text{Eq. (6.2)}$$

$$2C + O_2 = 2CO\ (-123\ \text{kJ/mol}) \qquad \text{Eq. (6.3)}$$

$$C + CO_2 = 2CO\,(162\ \text{kJ/mol}) \qquad \text{Eq. (6.4)}$$

$$C + H_2O = CO + H_2\,(119\ \text{kJ/mol}) \qquad \text{Eq. (6.5)}$$

$$C + 2H_2 = CH_4\,(-87\ \text{kJ/mol}) \qquad \text{Eq. (6.6)}$$

$$CO + H_2O = CO_2 + H_2\,(-42\ \text{kJ/mol}) \qquad \text{Eq. (6.7)}$$

Equations (6.2) and (6.3) are oxidation zone reactions, whereas Reactions (6.4)–(6.7) are reduction zone reactions. Then, a reforming reaction takes place between hydrocarbon and water vapour resulting in H_2 and CO mixture in gaseous products (Equation (6.8)).

$$CH_xO_y + (1-y)H_2O = CO + (1-y+x/2)H_2 \qquad \text{Eq. (6.8)}$$

The products' composition and energy content vary with process parameters such as gasification agent (e.g., air, oxygen or steam), temperature, pressure, feed characteristics (type, moisture, solids content, thermal conductivity) and feeding ratio (Raheem et al., 2018).

Ayol et al. (2019) gasified the municipal sludge in downdraft fixed bed gasifier at pilot-scale. About 1-kWh electrical power was generated for 1.2-kg sludge gasified. High-temperature steam gasification is also an innovative technology for obtaining high percentages of H_2 in syngas. Nipattummakul et al. (2010) studied steam gasification of sludge at various temperatures. They observed increase in hydrogen generation due to increase in reactor temperature. At 1000°C, the hydrogen yield obtained was 0.076 g gas/g sample. When compared to air gasification, the hydrogen yield increased three times when steam was used (Nipattummakul et al., 2010). Sewage sludge gasification using supercritical water for H_2 production was also investigated using a fluidized bed reactor (Chen et al., 2013). It was observed that the gasification was enhanced by increase in temperature and decrease in feedstock concentration. Also, the addition of catalyst increased the hydrogen formation with the highest catalytic activity of KOH.

There are several studies reported in literature for sludge gasification, though, application of gasification process for treatment of activated sludge at large scale has several challenges. For example, the high water content (80 wt%) and low heating value cause lower gasification efficiency. The dewaterability step before gasification is energy intensive step (1 kg of H_2O removal requires around 2260 kJ of energy), thus making the overall sludge conversion cost higher. Also, higher tar production in the process can cause blockage of tubing or fouling the apparatus; therefore, it needs to be removed and/or treated. This can be done either by removing the tar inside the gasifier or removing it from gaseous product by installing another equipment after gasifier. Tar removal inside the gasifier has been reported to be more economical as it eliminates high costs of additional installations in the production process. Researchers have studied the effect of parameters such as throughput, gasifying agent and dolomite catalyst on tar production and gas composition (Roche et al., 2014). The results indicated that with an increase in throughput, hydrogen production decreased, and tar production increased. In the presence of dolomite catalyst, hydrogen production increased by 20–30% with tar removal efficiency of around 71%. Various studies have also been conducted by using novel or modified gasifier for tar reduction inside gasifiers. For example, one such study investigated sludge gasification using a three-stage gasifier containing auger reactor, fluidized bed reactor and tar-cracking reactor in sequence to obtain high-quality producer gas with low impurities of tar, NH_3 and H_2S (Choi et al., 2017). They reported minimum tar quantities of 27 mg Nm^3 and NH_3, H_2S concentration of 443 ppmv (parts per million by volume) and 470 ppmv in producer gas, respectively.

Sewage sludge co-gasification with other carbonaceous materials such as woody biomass has also been extensively studied. The high volatile matter, low ash and low moisture content in woody biomass are advantageous features to be used for co-gasification with sludge. Ong et al. (2015) investigated co-gasification of woody biomass and sewage sludge. A maximum of 20 wt% dried sludge was successfully gasified to generate producer gas with lower heating value (LHV) of 4.5 MJ/Nm^3. Increasing the sludge content to 33 wt% resulted in blockage of gasifier because of ash agglomeration.

6.3.2.4 Hydrothermal Liquefaction (HTL)

HTL is a new emerging technique that includes sludge heating in water phase at 150–450°C temperatures. HTL operates at 5–30% solids and avoids dewatering and drying of sludge which considerably decrease the costs of HTL as compared to pyrolysis and other thermal processes (Raheem et al., 2018). The liquid biomass is hydrolysed at high temperature (250–350°C) and pressure (10–15 MPa) that causes cell breakage and hydrolysis of proteins, lipids and carbohydrates to form reactive molecules in solvent. The products formed after HTL consists of biocrude oil (target product), solid residue (biochar), aqueous phase containing soluble compounds and carbon dioxide. HTL comprises three major steps: (1) depolymerization of biomolecules present in sludge into monomers or oligomers; (2) breakdown of these monomeric or oligomeric units by cleavage, dehydration, decarboxylation and deamination to form unstable and active molecules; and (3) rearrangement of fragments by condensation, cyclization and polymerization forming biooil, aqueous phase and biochar (Hu et al., 2021).

Use of oxidants in HTL has a positive effect on formation of VFAs which can be used as a source of carbon to produce biogas and biopolymers (such as PHA). Several studies have been conducted to liquefy the sewage sludge to produce oil by using a catalyst (for example, NaOH) and/or solvent (acetone, ethanol) to enhance the reaction. For example, Vardon et al. (2011) reported biooil yield of only 9.4% from anaerobic sludge without a catalyst and solvent, whereas Leng et al. (2014) reported 45 and 40% biooil yield from dewatered sludge using acetone and ethanol, respectively. Xu et al. (2018) studied the yield and composition of various HTL products when temperature was varied with residence time of 10 min. The results indicated that on increasing the temperature, the biocrude oil quality and gas yield increased, whereas the water-soluble components yield, solid yield and total organic content in aqueous phase decreased. In aqueous phase, the biocrude oil yield and ammonia nitrogen content first increased and then decreased with their maximum values at temperature 340°C. Another study investigated the HTL of sewage sludge in a continuous-flow reactor (pilot-scale) at temperatures 300°, 325° and 350°C (Thomsen et al., 2020). Maximum biocrude yield of 41% was obtained at 325°C. The authors also showed destruction of micropollutants in sewage sludge using HTL with over 98% removal of pharmaceuticals and biocides. HTL is considered to be more efficient as compared to AD with regard to loading rate, sludge volume reduction, production of energy (biooil) and concentration of metals in biochar which can be recovered by further processing.

6.3.2.5 Wet Air Oxidation (WAO)

In WAO, chemical oxidation of sludge occurs at high temperatures and high pressure of 150–330°C and 6–20 MPa, respectively. Oxidation leads to formation of hydroxyl and peroxide radicals that convert the sludge organic matter to low molecular weight carbon compounds, concurrently destroying contaminants of emerging concern and pathogens. WAO can be applied to wet biomass that gives this process and additional advantage over incineration, thus making it ideal for waste activated sludge. The type of application basically determines the range of temperatures to be used in WAO. For example, low temperature (100–200°C) is used for conditioning of municipal and paper industry sludge. Medium temperature range (200–260°C) is used for the treatment of ethylene spent-caustics and regeneration of powdered activated carbon used in WWTP. Higher temperature oxidation (260–320°C) is used for industrial wastewater treatment such as pharmaceutical industry wastewater and activated sludge (Hii et al., 2014). The degree of oxidation can vary with process parameters such as temperature, oxygen partial pressure, residence time and oxidizable organic compounds in the sludge (Chauzy et al., 2010).

In Netherlands, ZIMPRO-process was developed in 1960s which is the oldest WAO process. But the process faced disadvantages such as high-energy costs, corrosion and odour issues (Hao & Phull, 2018). Therefore, the interest currently shifted towards catalyst-based WAO because of lower pressure and temperature requirements. Homogenous catalysts have been reported to be appropriate for WAO of activated sludge and copper-based homogenous catalysts, for example, copper sulphate is most extensively used in industrial applications. The Athos process is one of the major WAO sludge treatment processes currently being used commercially. It operates at 250–300°C and uses O_2 as oxidant for degrading organic materials. The process produces clean gas, organic liquid stream and mineral-based inert solids (Hii et al., 2014).

Some advantages of WAO process consist of low-level air pollution (no generation of NO_x, SO_2, HCl, furans, ash), reduction of greenhouse gases, small footprint, chemical oxygen demand reduction by 70% and volatile suspended solids reduction by 90%. However, a few disadvantages of the process include high capital costs, high maintenance, excessive ammonia production and increased rates of corrosion (Table 6.3).

6.3.2.6 Supercritical Water Oxidation (SCWO)

SCWO is an effective and innovative treatment technique for wastewaters and waste activated sludge. Normally, water exists in three states, i.e., steam, liquid or ice. When the water is heated to high temperature and compressed at high pressure (i.e., above critical point of water), an additional state called supercritical state of water is formed (Gidner & Stenmark, 2001). Water at temperatures >374°C and pressures >221 bar exists in supercritical form which is a state that is neither gas nor liquid. At the critical point, both densities (liquid and gas states) become identical and distinction between them no longer exists. The organic compounds and gases are completely miscible in this state, which creates a homogenous reaction medium, thus making the supercritical water suitable for oxidation of organic compounds releasing CO_2, water and other simple molecules. Hence, SCWO is a process of homogenous oxidation of organic compounds in aqueous medium using O_2 or H_2O_2 (as oxidizing agent) at temperature and pressures above critical point of water (Bermejo & Cocero, 2006). SCWO achieves complete oxidation of all organic wastes containing any combination of elements. The reaction time varies between 30 and 90 sec which depends on the reaction temperature (Gidner & Stenmark, 2001). The most frequently used supercritical fluids are CO_2 and H_2O. Partial oxidation products (NO_x, dioxins, CO) are not released, and the effluent can be disposed of easily without any treatment/cleaning. It has been reported that the environmental, economic and social metrics of SCWO process were more favourable than WAO or incineration (Bermejo & Cocero, 2006).

The first commercial SWCO sludge plant was installed at Harlington, Texas that can process 9.8 dry tons of sludge per day using HydroProcessing, L.L.C.'s *Hydrosolids*® (Griffith & Raymond, 2002). The *Hydrosolids* cost lower than other alternative processes (thermophilic aerobic digestion and traditional digestion processes) requiring dewatering and disposal. Cabeza et al. (2013) studied oxidation of sludge in SWCO process via hydrothermal flame regime. The waste was totally mineralized without producing any harmful products as in the case of incineration. Conversion of more than 99.99% of total organic carbon and 99.9% ammonia were obtained. However, various disadvantages still exist in this process such as corrosion-related issues, safety issues for handling pure O_2 and H_2O_2, high capital and maintenance cost, thickened sludge feed (5–10%), use of homogenous sludge with no grits and lack of pilot-scale studies for optimized parameters, i.e., reaction time, temperatures and pressure (Tyagi & Lo, 2013).

6.3.2.7 Bio-electrochemical Systems

MFC is an efficient and innovative technology to treat the organic matter in sludge and produce electricity. In MFC, the bacteria consume the organic matter in anaerobic conditions and generate CO_2, protons and electrons by cellular respiration. The produced electrons transfer to the anode through a mediator or by direct transfer from bacteria which leads to generation of power. Many bacterial species such as *Clostridium, Escherichia coli, Desulfovibrio* and *Shewanella* have been reported to oxidize organic matter and reduce ions. The electricity generation efficiency and performance of MFC depends on electrodes, especially, the material of the electrode (Mian et al., 2019). The electrode reactions in MFCs using acetate as substrate are given in the following reactions:

$$\text{Anodic reaction}: CH_3COO^- + 2H_2O + \text{microbes} = 2CO_2 + 7H^+ + 8e^- \qquad \text{Eq. (6.9)}$$

$$\text{Cathodic reaction}: O_2 + 4e^- + 4H^+ = 2H_2O \qquad \text{Eq. (6.10)}$$

TABLE 6.3
Comparison of All Techniques Used for Thermo-Chemical Conversion of Sewage Sludge

Technology	Operating Parameters	Products and by-Products	Advantages	Disadvantages
Incineration	• Temperature >750°C • In the presence of oxygen/air • Suitable for dry-activated sludge	• Heat, power • Inert material/ash	• 90% sludge volume can be reduced • High elimination of organics and pathogens	• Toxic exhaust gases • Dewatering required • Low energy efficiency • High cost (gas cleaning, ash disposal) • Corrosion of reactors
Pyrolysis	• Treated at 350–900°C under pressure • No oxygen • Suitable for dry sewage sludge	• Pyrolysis oil • Gas, char, ash	• Sludge volume reduction • Pyrolysis gases require less clean-up as compared to incineration • Deactivation of antibiotic-resistant genes • Biochar and syngas production	• Complex processing • Limited operating data and no cost analysis of process • Dewatering required • High capital costs
Gasification	• Temperatures of 700–1000°C • Partial oxidation	• Syngas • Tar, char	• Syngas, biochar potential value-added by-products • High thermal efficiency and carbon balance • Low toxic gases emissions	• Dewatering >50 wt% solids content • Syngas cleaning • Pollutant present in tar • High investment and operation costs
Hydrothermal liquefaction	• High temperature of 250–350°C and pressure of 10–15 MPa • N_2 gas atmosphere • Suitable for wet sludge	• Biocrude oil • Solid residue (biochar) • Aqueous phase containing water-soluble compounds • Carbon dioxide	• Prevents sludge dewatering costs • Lower operating temperatures • Deactivation of antibiotic-resistant genes • Removal of bioactive compounds	• Studies mostly at lab scale • Requires special process reactors • Recovery of catalysts is difficult • Loss of organics in aqueous phase
Wet air oxidation	• Temperature of 150–330°C and pressure of 6–20 MPa • Suitable for wet sludge	• Mineral-based inert solids • Aqueous phase containing organic compounds	• Destroys emerging contaminants and pathogens • No generation of toxins (NO_x, SO_2, furans) and ash • COD reduction of 70% and volatile suspended solids reduction of 90% • Small footprint	• High capital costs and high maintenance • High ammonia production (odour issues) • High corrosion problems
Supercritical water oxidation	• Temperature >374°C and pressures >22.1 MPa • O_2/H_2O_2 as oxidizing agent • Suitable for wet sludge	• CO_2, H_2O, heat, minerals	• Rapid and highly effective • Destruction of persistent and toxic chemicals • NO_x, dioxins, CO are not released (clean process)	• Significant amount of energy required (high capital and maintenance cost) • Corrosion-related issues • Safety systems for handling pure O_2 and H_2O_2 • Sludge should be homogenous and grit free

The first direct sludge-to-electricity conversion experiment using MFC was performed by Dentel et al. (2004). They used anaerobically digested sludge and attained a maximum voltage of 517 mV. Since then, research focus has been channelized to improve energy production from activated sludge. Within a residence time of 8 days, Ghadge et al. (2015) demonstrated 81% sludge degradation and energy production of 8.7 W m^3 in MFC using NaClO as the catholyte. The input voltage in MFC can be categorized into low-voltage (mV) and high-voltage (kV) processes. The low input voltage is used for enhancing the sludge dewaterability and degradability, killing pathogens and removing heavy metals, whereas the high input voltage is used to increase the biogas production in AD process. The first low-voltage sludge pre-treatment using an electrochemical system was conducted at 12 V for 15 h which increased the methane generation to 318 mL/g volatile solids (31% increase) (Zeng et al., 2021). Although higher amounts of methane were produced, one disadvantage was the long duration of pre-treatment. However, by increasing the input voltage, treatment duration and energy consumption can be decreased. So, when the input voltage was increased to 20V for 40 min, the biogas yield enhanced up to 63%. The improved methane production has also been reported by adding oxidants such as H_2O_2 and NaClO during the electrochemical treatment (Zeng et al., 2021). One study reported that combining electrochemical treatment (200 V for 45 min) and NaClO dosing enhanced the methane yields by approximately two times in pilot AD plant (Yuan et al., 2016). Jiang et al. (2011) studied ultrasonic and alkaline pre-treatments. Both techniques enhanced the disintegration of organic matter, resulting in a remarkable rate of COD elimination. The thermal pre-treatments were investigated by Yuan et al. (2012) as efficient alternatives to speed up the hydrolysis and consequently increase power output. However, these treatments can destroy the original microbial community indicating the use of inoculum in the system. Moreover, these pre-treatments will further increase the overall cost of MFC process. Therefore, economical approaches to be integrated into MFCs need further research for electricity generation from sewage sludge.

The waste activated sludge composition is complex when compared to other fermentable compounds such as glucose and acetic acid which leads to its slow biodegradation and difficulty in efficiently degrading the persistent organics (Raheem et al., 2018). The low electron transfer efficiency, power density, electrode scaling, expensive electrodes, high operational costs, low power output and high capital costs mark the drawbacks of MFC systems which needed to be solved before scaling up the MFCs for large-scale applications. Advantages of MFCs include low/no chemical use, small footprint and easy installation and operation. MFCs considerably lessen the negative impact on the environment caused due to traditional sludge treatment (e.g., incineration).

6.4 ENERGY AND RESOURCE RECOVERY

6.4.1 Nutrients Recovery

Significant amounts of nutrients are present in activated sludge, such as phosphorus (around 5.9–13.4%), total solids (0.5–0.7%) and nitrogen (2.4–5.0%), these components are mostly found as proteinaceous matter (Tyagi & Lo, 2013). After the fragmentation and dissolution of sewage sludge, it is transformed into phosphates and NH_3, which is subsequently converted to magnesium ammonium phosphate (struvite), a superior plant fertilizer that may be sprayed directly on soil (Ribarova et al., 2017). Commercial fertilizer costs roughly US $1.3 and US $2.6 for 1 kg N and 1 kg P, respectively, whereas the P-recovery from activated sludge has been noted to cost between 1.6 and 3.0. Moreover, P is a finite, non-renewable resource, so, it needs to be treated as a valuable product and effectively extracted from activated sludge (Ashokkumar et al., 2022; Weigand et al., 2013).

Depending on the phosphoric minerals precipitation in the form of struvite, hydroxyapatite, or calcium phosphate, various attempts have recently been undertaken towards P-recovery from sludge (Shih et al., 2017). These P-recovery methods were primarily established and employed in Japan and the Netherlands (Yuan et al., 2012). According to some investigations, struvite could be obtained from a biological nutrient removal process prior to wastewater treatment, recovering 80–85% of

P (Cieślik & Konieczka, 2017). Currently, research is also being conducted on P-recovery from activated sludge incineration ashes which have a substantial amount of P (Li et al., 2017). The P-extraction from ashes could be five to ten fold higher than from waste activated sludge directly, according to Cieślik and Konieczka (2017) but sadly, the feasibility of such technology is only demonstrated in industrial-scale treatment plants due to the high capital costs involved in constructing a facility that complies with all pollution regulations for sludge incineration. In large-scale circulating fluidized bed kiln, Li et al. (2017) showed how P resources from activated sludge ash may be reused. The samples were combined with 2–10 wt% of calcium oxide (CaO). The findings indicated that an elevated temperature and a larger dose of CaO were beneficial for the transformation of non-apatite inorganic phosphorus into apatite P.

By using an electro-dialysis methodology from activated sludge, Guedes et al. (2014) presented an effective way for P pre-recovery. The technique produced a total yield of 30–85%, including some gypsum, needing additional study to see whether these residues could be used as dopants in building materials. Large-scale P-recovery has been made possible by technological advancements, including Aqua Reci (AR), Ostara, Susan and Krepo, which use physico-chemical and thermal treatment to dissolve P and then recover it through precipitation.

To extract both P and energy using integrated SCWO technology, the AR process is an industrial technology, which has been established in Sweden. With 2 h of reaction time and at 90°C, the method may effectively recover up to 100% of P. In Edmonton (Canada), a different procedure known as Ostara has been used to extract P from activated sludge stream using magnesium chloride. The procedure was started in 2007, and it is anticipated to generate 200–250 Mt of struvite per year (Stendahl & Jäfverström, 2003). The Crystalactor® technology was also made commercially available in the Netherlands. But due to the high cost of P-recovery, this technology was deemed to be unprofitable. Furthermore, it is thought that the biggest barrier to scaling up the investigated systems is the significant expenditure of P-recovery from activated sludge. It is therefore expected that a new method will be put out that might enable both the extraction of valuable phosphoric material and the comprehensive management of all wastes generated during sewage sludge thermal treatment (Roeleveld et al., 2004).

6.4.2 Heavy Metals Recovery

Because of the abundance of heavy metals, including Cr, Cu, Zn, Pb, Cd, Ni, As, Hg, and Se, sewage sludge has been monitored for the effective recovery of heavy metals. Table 6.4 compares the amounts of heavy metals in various sources as well as a collection of toxic inorganic substances

TABLE 6.4
Heavy Metal Concentration in Various Biomass Materials

	Heavy Metals (mg/kg dry basis)								
Feedstock	**Cd**	**Cr**	**Cu**	**Hg**	**Ni**	**Pb**	**Zn**	**As**	**References**
Paper sludge	<0.4	110	310	1000	–	160	470	8	Nzihou and Stanmore (2013), Werle and Dudziak (2014)
Wheat straw	1.0	25	0.06	6	–	–	–	0.18	
Beech wood	1.0	2.5	43	0.12	–	33	15	3.5	
Recovered fuel	24	1020	2800	–	209	1100	–	37	
Settling pond sludge	–	60.3±1.81	–	–	132±2.0	86.0±2.58	–	–	Moodley et al. (2007)
Sewage sludge	<1	494	292		251	24	881	<1	Karaca et al. (2015)

that were tested in waste activated sludge. The extraction of biological components, value-added compounds, and energy recovery from activated sludge can all be impeded by the substantial quantities of such metallic component. Trace elements (TEs) are commonly found in the gaseous state, and charcoal residues when dealing with sludge heat conversion using pyrolysis and gasification processes (Tyagi et al., 1993).

According to a report, TEs' volatilization is related to their boiling point. Due to the increased risk of volatile pollutants generation because of greater gasification temperatures than pyrolysis, evaluating the distribution of heavy metals during gasification is becoming more important. Numerous gasification investigations have shown potential paths for heavy metals to reach different process products (Saveyn et al., 2011). The pyrolysis was carried out by He et al. (2010) in an electric furnace using a consecutive extraction process, with the major emphasis on fractionating heavy metals in activated sludge and the leftovers that were created following pyrolysis at temperatures between 250 and 700°C. At a temperature of 700°C, Cd was volatilized in the off-gas and reduced in the leftovers.

In the study by Hwang et al. (2007), distribution of metals such as Cr, Cd, Zn, Pb, Cu and K, was investigated. Except for Cd, Pb and Zn, the majority of the heavy metals in the waste activated sludge were stationary in the residue without being volatilized. Even though the residues included excess organic matter than ash, under an aerobic environment, their carbon output into the leachate was equivalent to that of the ash. Consequently, it was determined that, in contrast to earlier pyrolysis investigations, pyrolysis of polluted sludge restricted to 500°C temperature lowered the fumes of heavy metals from produced char in the landfill. When considering the effects of potential hazardous heavy metals during gasification, it should be noted that ash dust is a significant source of heavy metals (especially Cd and Pb) in turbulent waste gasification routes; as a result, particulate ash pollutants are almost eradicated if the gasification takes place in non-turbulent circumstances. The source of raw sludge, which mostly enters products at high operating temperatures used for thermo-chemical transformation processes, determines the type, speciation and quantity of heavy metals in general (Lu et al., 2016).

A promising strategy, for connecting sewage sludge and biofuel production, can be presented. Nevertheless, given the type and concentration of heavy metals in the activated sludge, the reactor architecture, and the existence of potential pre- and post-treatment methods, determines the deployment of suitable thermo-chemical process and its ideal operational circumstances. In order to prevent or reduce the discharge of heavy metals during the thermo-chemical conversion of sludge, it is envisaged that the application of the policies, such as running the bioreactor at optimal process parameters and outfitting it with post-treatment techniques, should be feasible. There haven't been enough efforts done to prevent the transfer of heavy metals to the goods, thus effective pre-treatment techniques using possible sorbents, hydrothermal and leaching pre-treatment might be a wise strategy (Saveyn et al., 2011).

6.4.3 Biofuel Production

In comparison to traditional fossil fuels, sustainable alternative fuels have the following fundamental qualities: good biocompatibility, cost-effectiveness, efficiency and reduced environmental impact. Various biofuels such as biodiesel, bioethanol, biooil, biogas and biomethane (Yadav et al., 2021) have been generated in recent decades, with biodiesel and ethanol being the most popular. The fundamental reason for their acceptability is that they don't require major alterations to the gasoline or engine components now in use. Scientists are currently investigating several non-edible oil resources, such as algae, jatropha, karanja and neem as possible substitutes for biodiesel feedstocks in order to reduce the process cost; however, their convenience of growth necessitates vast amounts of space (Nigam & Singh, 2011).

According to the assessment of the existing literature by numerous researchers worldwide, sewage sludge's lipid content is a viable substrate for the generation of biodiesel, which combined with the financial benefits of the management of waste can also be a viable solution (Kumar et al., 2021).

The various lipid extraction methods and their transformation into biodiesel were evaluated by Siddiquee and Rohani (2011). A recently published review by Gaeta-Bernardi and Parente (2016) is focussed on the financial viability of employing sewage sludge as the primary source for biodiesel production. The four processes of direct usage, micro-emulsion, pyrolysis and transesterification are used to generate biodiesel using lipid resources. Even though the direct utilization, blending and micro-emulsion techniques are straightforward, they have a number of drawbacks, including leakage of lubricating oil and the development of carbon deposits in engines because of the increased viscosity and reduced reactivity of unsaturated hydrocarbon chains. Because there is more carbon available in secondary and pulp and paper sludge than in primary and mixed sludge, microorganisms can proliferate there more easily (Zhang et al., 2021). According to a study, adding glucose to the sludge (C/N ratio 100) caused the growth of *Pichia amethionina* sp. SLY, whereas adding glycerol caused *Galactomyces* sp. SOF and *Trichosporon oleaginosus* to accumulate more lipids. In order to grow oleaginous *Lipomyces starkeyi* MTCC-1400 as a model organism for the generation of biomass and lipid with maximum yield, researchers looked into the impact of thermo-chemo-sonic treatment on pre-digestion of municipal activated sludge. In the pre-digested sample of NaOH, the highest 17.52 g/L biomass and 64.3 wt% lipids were achieved (Selvakumar & Sivashanmugam, 2017). To generate high-quality biofuels, it is also necessary to optimise the many production-related factors. The improvement must be the main priority. Emulsification and micro-emulsification techniques can boost the fuel properties of biodiesel or its mixing with biooil to improve enhanced emission qualities due to the combustion quality.

6.4.4 Biopolymers (Bioplastics, Biopesticides and Bioadsorbents)

Bioplastics can be accumulated by a number of bacteria in waste activated sludge (0.30–22.7 mg of polymer/g of sludge). Nevertheless, their widespread application has been constrained by their high cost of production (US \$4–6 kg^{-1} as opposed to US \$0.6–0.9 kg^{-1} for traditional plastic materials) (Bluemink et al., 2016; Yadav et al., 2020b). The present improvements in microbial fermentation and the utilization of sludge are probably going to increase the productivity and reduce the production expenditure of bioplastics. For the efficient culture of PHA-producing microbes, the feast and famine settings have been reported to be viable (Borea et al., 2017).

Bacillus thuringiensis (Bt) is now the most efficient biopesticide with the capacity to create delta endotoxins (δ endotoxin, also known as cry and cyt poisons) and is commonly used in agriculture, forestry and the general public sector (Sanchis & Bourguet, 2008). In addition, biopesticides are significantly more environmentally friendly than synthetic pesticides because they are highly target-specific substances with no hazardous residue. Yet, the expensive price of the raw materials has significantly limited the use of Bt for commercial purposes. About 40–60% of the expense of producing Bt-based biopesticides is borne by the traditional fermenter broth (Zhuang et al., 2011). Hence, it is now crucial to investigate emerging raw materials while considering a number of factors, including their potential to be renewable, cost-effective and year-round supply for the synthesis of Bt. Therefore, sewage sludge can be used as a rich resource of nutrients and economical substrate for the synthesis of Bt (Brar et al., 2009), which will significantly reduce the expense of Bt synthesis and contribute to the long-term use and management of sludge. Three phases are involved in the production of sludge-based biopesticides: fermentation of activated sludge, extracting and/or product recovery and product formulation. Various investigations have shown that fermenter parameters, including pH, C/N ratio, dissolved oxygen concentration, solids density and type of inoculum sludge, have a significant impact on the generation of Bt-based biopesticides. According to a study, replacing commercially used medium for the production of Bt with waste activated sludge can result in net benefits of more than 50% (\$0.25–0.34 L^{-1} sludge medium versus \$0.75 L^{-1} industrial media) (Tirado Montiel et al., 2003). Higher entomotoxicity (T_x) levels could result in additional reductions in the cost of producing Bt.

Therefore, the manufacture of Bt using sludge and application to forestry and, specifically, agro crops for pest control looks to be completely in line with present sludge disposal techniques.

Unfortunately, the complexity of sludge and low Bt production are the main drawbacks of the available techniques. To increase Bt synthesis in the worldwide pesticide market, additional research concentrating on production and extraction processes (i.e., parameter optimization, process monitoring and product purification) is necessary. Establishing the final Bt formulation will also encourage the sustainable management and exploitation of sludge (Vidyarthi et al., 2002).

Similar to biopesticides, activated sludge is a great source for the synthesis of bioadsorbents (Sellamuthu et al., 2021). Kemmer et al. (1972) were the first to discover the viability of sewage sludge as a substrate for generating activated carbon due to its carbonaceous nature. They obtained a patent for a method of chemically activating dehydrated sewage sludge to produce adsorbents. Moreover, Beeckmans and Ng (1971) published a pioneering research on the carbonization of sewage sludge to make adsorbents in the same year. However, a more comprehensive evaluation and assessment of the financial viability of manufacturing bioadsorbent on the pilot-scale within the context of sustainability measures are required.

6.4.5 Other Resources (Proteins, Enzymes)

Notably, activated sludge has a significant potential for usage as a protein source because of the increased fraction of proteins (around 50% of the dry mass of microbes). Protein serves as a vital ingredient in animal feed, providing both energy and N_2. As a result, protein recovery through sewage sludge has several benefits over protein recovery from traditional sources (Jimenez et al., 2013; Su et al., 2015). For instance, protein recovery from sludge demands less space and prevents certain organic matter, increasing dewaterability as a result of the elimination of bacteria. Numerous important pieces of research on protein extraction and evaluation from activated sludge have been mentioned (Su et al., 2015; Yadav et al., 2020a). Under the operational settings described in Hwang et al. (2008), fragmented activated sludge (5330 mg/L) for the extraction of cellular proteins was helped by a chemical treatment combined with ultrasonication. At an ideal pH of 3.3, they found a protein extraction rate of up to 80%, with nutrient content similar to that of conventional protein feeds, making it appropriate to be used as animal feed supplementation. Likewise, Jimenez et al. (2013) examined the effectiveness of various colorimetric techniques to assess the organic matter (e.g., protein, carbs, lipids) compositions of activated sludge, including Lowry, modified Lowry and Bicinchoninic acid assay. It was discovered that the proteins made up 50% of the biochemical fraction.

About 80% of the organic matter of the activated sludge is made up of the three primary molecular families (complex carbs, proteins and fats) (Tyagi & Lo, 2013). Due to their great value, it is imperative to recover different enzymes, including lipases, dehydrogenases, glycosidase and peroxidases. Nabarlatz et al. (2010) investigated various strategies for extracting enzymes. The achieved results demonstrated the efficacy of ultrasonication alone or in combination with enzyme additive under the optimum operational parameters. Additionally, they claimed that enzymes aid in increasing the biodigestibility of organic matters for greater biogas generation during AD and other processes. Extraction of various enzymes from sludge was also demonstrated by Guanghui et al. (2009) who used ultrasonication accompanied by ethylenediaminetetraacetic acid (EDTA). The technique showed successful hydrolytic enzyme recovery. Further later, the hydrolytic enzymes were extracted by Ni et al. (2017). Experimental observations demonstrated that the generated enzymes seem to be suitable compound enzymes for feed.

Enzymes are typically used for a diverse range of large-scale applications, including pharmaceutical, culinary, diagnostic, skin care, detergents, and precisely chemical plants. Given their broad commercial relevance, securing a consistent and affordable availability of these enzymes is essential because, in the aforementioned industries, the preparation of growth media accounts for between 30 and 40% of production expenses. As a result, sludge-based enzymes may reduce the time and expense of various industrial processes. To maximize enzyme recovery, a comprehensive study and a techno-economical evaluation of the techniques are therefore necessary. Table 6.5 represents a comprehensive overview of various studies conducted on resource recovery from sewage sludge.

TABLE 6.5
Studies Reported on Resource Recovery from Sewage Sludge

Product Obtained	Process Used	Recovered Results	Aspects	References
P (struvite)	AirPrex®	80–90% of P-recovery can be used as commercial fertilizer	Enhanced sludge dewaterability, low investment cost	Ye et al. (2017)
	PHOSPAQ®	70–95% of P-recovery	Enhanced sludge dewaterability but high operating cost	
P from sewage sludge ash	AshDec	P-recovery efficiency 98%	Purified phosphate was recovered but process consumed high amount of energy	Soares et al. (2017)
Heavy metals	Microwave treatment + H_2SO_4	100% recovery of Cd, Cr, Cu, Ni, Pb and Zn	Inactivation of pathogens and reduced reaction time for heavy metals extraction process	Wang et al. (2017)
Sludge-based adsorbents	Microwave treatment	Produced sludge-based adsorbents had higher metal ion adsorption capabilities	Faster heating rates and energy savings, decreased waste and equipment size	Lin et al. (2017)
	Pyrolysis	The maximum adsorption rate was 277 mg/g	Reduction of sludge volume, production of solid leftovers for additional recycling but required more processing time	
Biogas	Anaerobic digestion	Out of sludge's initial 67% of energy potential, 52% was converted into biogas.	Decreased final solids, eliminated harmful bacteria in the sludge, and lowered the cost of WWTP operations	Shen et al. (2009)
	Co-digestion of organic residues other than sewage sludge	With the incorporation of fruit waste, specific methane yield increased from 0.310 Nm^3 CH_4/kg volatile solids to 0.393 Nm^3 CH_4/kg volatile solids	High methane production; decrease in sludge volume; enhancement of the C/N ratio of the substrate, utilization of the unused AD capacity of the WWTP	Nghiem et al. (2017)
Biooils, biochar, syngas	Pyrolysis	51.0% char, 33.6% oil and 10.4% syngas were recovered at 550°C	Biochar can be utilized as a soil amendment or as a resource for chemicals and biooil as fuel.	Tyagi and Lo (2013)
Heat self-consumed in the process, Electricity	Supercritical water processing	The method degraded organic matter to a higher 99.9% degree producing syngas and useful chemicals.	Eliminating all microbes, 60–80% decrease of total solids, complete elimination of odours and GHGs, doesn't produce any damaging by-products (NO_x, SO_x, CO)	
Protein	Ultrasonication	Protein recovery of 80%	The restored protein's nutritional profile was similar to that of commercially available protein.	
	Hydrothermal treatments	Protein recovery of 87%	Demand substantial pressures and temperatures, additional research is needed to completely purify proteins	García et al. (2017)

(Continued)

TABLE 6.5 *(Continued)*
Studies Reported on Resource Recovery from Sewage Sludge

Product Obtained	Process Used	Recovered Results	Aspects	References
Enzymes	Ultrasonication	53 protease units/g volatile suspended solids	Good end-product quality	Anbazhagan and Palani (2018)
	Ultrasonic combined with stirring	Enzymatic product had protease, lipase, amylase and cellulase activity in amounts of 2534, 1, 1150 and 2340 units/g, respectively.	The produced enzyme has a considerable market potential.	
Biopesticides	Fermentation: stirred tank reactor	Maximum T_x: 16,552 SBU μL^{-1}	The proteolytic activity was directly correlated with T_x.	Tyagi and Lo (2011)
	Solid state fermentation Sludge (SL), sludge + wheat (SW), sludge + straw powder (SS) and commercial culture (CM)	The combination of SW produced most viable cells, spore counts and toxins.	Under solid-state fermentation, the solid culture media (SL, SW, SS and CM) enabled Bt production.	
	Fenton oxidation (FO)	74% biodegradability record was attained	Sludge-based FO technique was demonstrated.	Brar et al. (2009), Tyagi et al. (2009)
Bioplastics	Aerobic industrial scale	73% maximum PHA concentration	The synthesis of PHA was impacted by the availability of N and P, proving that their elimination is not required to produce PHA using waste activated sludge.	de Lourdes et al. (2001), Tirado Montiel et al. (2003)
	Feast-famine: Industrial scale	Maximum PHA – 25 g/TSS	PHA production using activated sludge was feasible. To improve the accumulation of PHA, more research on parameter design is necessary.	

6.5 CONCLUSIONS AND PERSPECTIVES

The increase in population and better living standards has resulted in increase in sludge production all over the world. Due to the limited landfill sites and strict disposal legislations, a shift has been observed from conventional sludge disposal methods to the use of advance technologies for sludge valorization. Numerous studies are being employed by researchers to investigate the potential of energy and resource recovery from sludge. However, the complex, heterogenous nature of sludge containing moisture, microbes, heavy metals and other pollutants reduces the efficiency of processes and increases the overall costs.

Biochemical conversion processes such as AD are widely reported to be used for sludge valorization; however, it suffers poor efficiency and long reaction times. Thermo-chemical processes discussed in this chapter have higher efficiencies with faster processing time. However, they are energy intensive and may require gas clean up prior to usage and emission. Economically low product yields, lack of substantial research, high operating and maintenance costs are some areas that need attention in future studies.

An integrated biorefinery system could be designed and implemented focussing on reusage to generate power and recover possible resources. Therefore, application of advance technologies for sewage sludge management can convert sludge into promising opportunity by recovering energy-rich biogas (methane, hydrogen, syngas), liquid biofuels (biodiesel, biooils), construction materials, bioplastics, biofertilizers, biopesticides, electricity using MFCs, metals and nutrients (nitrogen and phosphorus) while protecting the environment and public health. Moreover, the success of these recovering technologies will mostly depend on techno-economic feasibility, environmental sustainability, market requirements and public acceptance.

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Section II

Thermochemical Processing of Sewage Sludge

7 Thermochemical Processing of Sewage Sludge

Fundamentals and Challenges

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7.1 INTRODUCTION

To date, there is a large amount of literature devoted to the problems of the use of sludge and sewage sludge as fertilizers. In general, the literature on this topic displays data on the high fertilizing effect of silts and sewage sludge during the cultivation of agricultural crops. However, it should be borne in mind that with different technological cycles of wastewater treatment, precipitation with different physico-chemical, biological compositions can form, which in terms of their fertilizing qualities can differ sharply from each other. On the other hand, several methods of processing and/or depositing sludge and ash from their combustion can be implemented at sanitary facilities adapted to the objects of sewage sludge burial, and different operating periods. The use of waste-free technology in obtaining a secondary product from sewage sludge is very difficult (with the exception of biogas). The main problem is caused by heavy metals and the bioalkylation processes associated with them, the presence of pathogenic and specific microflora, capable of negative effects even after prolonged stabilization of waste on silt sites, landfills, in geotubes (Dregulo & Bobylev, 2021a).

Therefore, their use as a biosubstrate for agriculture (as the most rational stage of the targeted utilization of precipitation) is impossible for sewage sludge of the general sewage system. The latter determines the ways of their disposal by the specified criteria of generally accepted practice-disposal at landfills, the life cycle of operation of which becomes a factor of accumulated environmental damage. As a promising scheme for the elimination of objects of accumulated environmental damage in the wastewater disposal system, it is proposed to use co-incineration with solid waste to produce ash and slag used as an additive in the production of building materials. The latter becomes a strategic resource in the implementation of the program for the elimination of environmental damage caused by the influence of sewage sludge.

7.2 THE REASONS FOR THE USE OF THERMOCHEMICAL METHODS FOR THE DISPOSAL OF SEWAGE SLUDGE

Sewage sludge includes dissolved and undissolved impurities retained by primary and secondary settling tanks, flotation and other structures after mechanical, biological and physico-chemical treatment.

7.2.1 The Content of Dangerous Microflora

A significant negative factor, from the point of view of precipitation utilization, is the presence of pathogenic and conditionally pathogenic microflora in them. Pathogenic microorganisms can enter wastewater with urine, pus, saliva and other human secretions, especially in cases of direct

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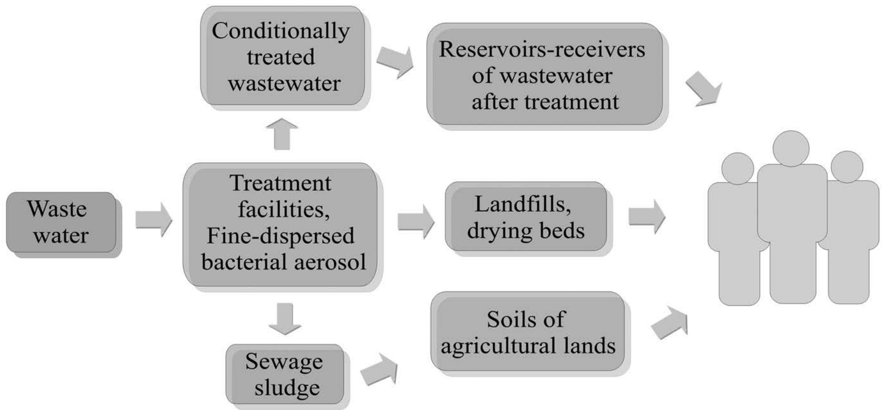


FIGURE 7.1 Migration routes of dysbiosis of environmental components associated with sewage sludge.

ingress (without pretreatment of wastewater), for example, from infectious medical and preventive institutions and then accumulate in sewage sludge. Due to the high content of protein-fat fractions, precipitation quickly rot, thereby increasing the risk of bacterial contamination of the environment (Dregulo et al., 2022). The ways of environmental pollution by pathogenic microflora from waste-water disposal systems can be schematically represented as shown in Figure 7.1.

Spores of pathogenic microflora persist in the soil and water for a long time, and under favorable conditions, they can multiply again, which negatively affects not only the deposited surface of the landfill soils, but also groundwater used as drinking water sources. Soil and water contaminated in this way can carry other parasitic infections and some helminthic infections (Sabbahi et al., 2022).

7.2.2 The Content of Heavy Metals

Another important problem in the disposal of sewage sludge is the presence of heavy metals in them (Dregulo & Vitkovskaya, 2020). To date, there is a large amount of literature devoted to the problems of the use of sludge and sewage sludge as fertilizers. In general, the literature on this topic displays data on the high fertilizing effect of sludge and sewage sludge during the cultivation of agricultural crops. However, it should be borne in mind that with different technological cycles of wastewater treatment, precipitation with different physico-chemical, biological compositions can form, which in terms of their fertilizing qualities can differ sharply from each other. Heavy metals, as the most common pollutants, can be identified as the leading limiting factor determining the direction and nature of biodiversity development (Tovar-Sánchez et al., 2018). Soil is a biological environment in which heavy metals and other pollutants accumulate, the main source of which is anthropogenic activity. Massive pollution of the environment by them can lead to toxicosis of plants, animals and people and therefore is diagnosed relatively easily and quickly. It is more difficult to assess the toxic effect of relatively small concentrations of heavy metals, which outwardly slowly and imperceptibly affect the environment. Meanwhile, acting for a long time, heavy metals are able to cause shifts in the existing biological equilibrium, inhibiting biological processes. The problem in understanding the negative impact of sewage sludge stored at landfills (Dregulo & Bobylev, 2021b) is complicated by the difference in soils, which cannot inactivate leaching heavy metals to varying degrees.

7.2.3 The Content of Persistent Organic Compounds

When disposing of sewage sludge, the problem of assessing the impact of persistent organic pollutants (POPs) present in sewage sludge, which are cumulative poisons, remains practically unresolved.

Atmospheric precipitation may be one of the sources of accumulation of POPs by sewage sludge. It can be assumed that the entry of POPs into the treatment facilities occurs by capturing hydrophobic particles of POPs by drizzle, snow and meltwater and their introduction into the soil. Often, data on the environmental impact of POPs present in wastewater sediments do not show their possible anthropogenic impact on natural ecosystems. For example, Directive 86/278/EEC on environmental protection when using sewage sludge in agriculture does not define the maximum permissible concentration (MPC) for POPs, which is quite understandable, given the limited (at the time of its introduction) and still ambiguous information regarding their assimilation and environmental impact (Council Directive 86/278/EEC). In some European countries (Holland, Germany, Austria, France) and the USA, norms of permissible and boundary concentrations of PCBs in sewage sludge used as fertilizers have been introduced. However, sewage sludge can be very dangerous due to the presence of nonylphenol, bisphenol-A, dibutyl phthalate and diethylhexyl phthalate in them, which can destroy the endocrine system. Directive 91/271/EEC on urban wastewater treatment does not mention the need for wastewater treatment from POPs at all (Council Directive [91/271/EEC]). The revision of the relevance of Directive 86/278 EC necessitated consideration of the presence of organic compounds in organic waste (including sewage sludge) in order to ensure that their use in agriculture will not cause chemical pollution. This is due to the fact that the restrictions related to pops in the regulations of the Member States of Directive 86/278 EC focused only on the bioaccumulative function of PAH, PCBs and PCDD/F compounds for animals and, consequently, dangerous to humans. Sewage sludge containing polybromodiphenyl ethers and introduced into the soil of agricultural lands for four years showed a significant exaggeration of them in the soil with a high cumulative effect (Eljarrat et al., 2008). Therefore, for such countries where there is a high percentage of the use of sewage sludge on farmland, mechanisms for financing treatment facilities and managing the movement of sewage sludge should be debugged (Ćetković et al., 2022), including

- providing solutions that are technically and economically adapted to the economic realities of these countries (reduced investment and operating costs);
- full legal support, including the ability to adapt to future restrictions that may be reflected in the disposal of treated wastewater and sediments in agriculture;
- diversification and final removal of sludge through the introduction of advanced sludge treatment systems;
- optimization of the use of weather conditions for sludge treatment.

Nevertheless, the Joint Research Council of the EU and researchers from a number of countries, including China, have concluded that the impact of dioxin-like compounds in sewage sludge is insignificant and its use as a fertilizer does not pose serious problems for human health when the sludge is used for agricultural purposes. Therefore, for each case of the use of sewage sludge, it is necessary to study the specific composition of precipitation in relation to their industrial and industrial field of education.

7.2.4 Greenhouse Gas Emissions Associated with Sewage Sludge

The use of waste-free technology in obtaining a secondary product from sewage sludge is very difficult (with the exception of biogas). It is not possible to estimate the total share of greenhouse gas emissions (carbon footprint) from wastewater disposal systems in the world. Comprehensive research on this problem is currently unknown or unavailable. It is important to consider not only the amount of greenhouse gases released, but also the geographical and ecological features of the region. For the northern regions of the earth, the most acceptable methods of waste and wastewater disposal in this region (Pippo et al., 2018). Of the safer methods (the most common in developed countries), anaerobic purification is considered due to the closed cycle of recovery of waste gases in methane tanks. Wastewater treatment in methane tanks is practically not used in urban wastewater treatment plants. The exception is local production systems. There is no doubt that different types of wastewater treatment will produce sewage sludge with different physico-chemical parameters. This will influence the decision on which method of thermochemical treatment is the most profitable and environmentally friendly.

7.3 FUNDAMENTALS OF THERMOCHEMICAL PROCESSES OF SEWAGE SLUDGE TREATMENT

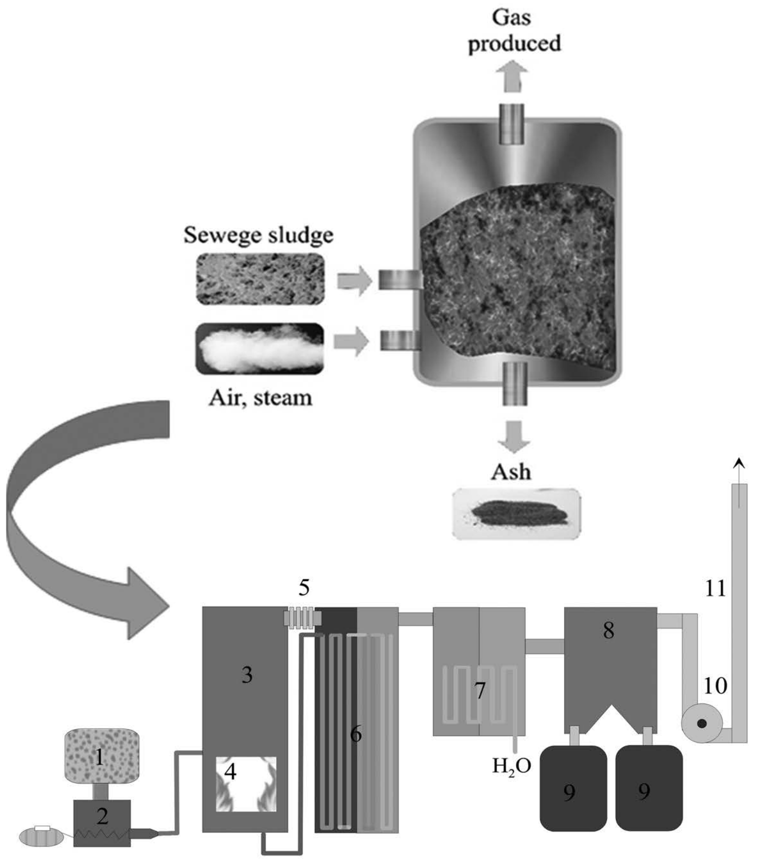
The use of thermochemical processes for the disposal of sewage sludge, as shown earlier, has a number of environmental aspects. At the same time, another fundamentally important aspect should be pointed out – this is the production of electricity and value-added products. The latter determines the choice of the technological scheme of thermochemical treatment, which we will try to consider later.

7.3.1 Incineration of Sewage Sludge

Incineration is the most well-known method used to reduce the amount of sediment formed, but this is not enough for megacities, because a large amount of ash is also formed. It is important that this method prevents the spread of pathogenic organisms existing in the sediment and eliminates unpleasant odors. Therefore, the question of its secondary use is very relevant, at least for every major city. For incineration, the sediment must be pre-cleaned from the sand and water content to increase the heat of its combustion, due to an increase in the amount of volatile compounds. Cleaning includes the removal of sand and water, thickening, and the establishment of the necessary composition. This technology is characterized by low costs for the operation and maintenance of equipment, as well as the possibility of remote monitoring of the process can be attributed to its great advantage. The process of burning precipitation consists of the following stages (Figure 7.2): heating, drying, distillation of volatile substances, burning of the organic part and calcination for the burning of carbon residues.

The ignition of the sediment occurs at a temperature of 200–500°C. Calcination of the ash part of the sediment is completed by its cooling. The temperature in the furnace should be within 700–1000°C. In each specific technological scheme of thermal processing of sewage sludge (and other waste), special attention is paid to incineration furnaces (reactors). Table 7.1 shows the comparative characteristics of waste heat treatment technologies and devices: a vertical cylinder with a furnace space divided into seven to nine horizontal hearths. Moreover, there are holes in different parts of each hearth: there is an alternation of a hearth with a hole in the center and a hearth with a hole in the periphery. In the center, there is a shaft with paddling mechanisms attached to it. Sewage sludge is fed from the top of the furnace, and then with the help of strokes, it moves down, passing through each under. This type of furnaces has a simple and reliable mechanism, which is a big plus, but at the same time it is necessary to pay attention to the paddling devices, since they can be damaged most often. Drum rotary kilns are usually used to burn a mixture of sewage sludge and urban garbage. The essence of the work is that the pre-dehydrated sludge enters a rotating drum, which is installed at an angle to the furnace. As a result, the sediment is fed into the drum, in which drying first takes place, and then combustion. Such furnaces require a lot of space in production, and the technology of operation is also more complicated; for these reasons, drum rotary kilns are less common.

Sometimes incineration processes are combined with the use of two methods: mono-incineration and co-incineration. In the first one, only sewage sludge is used with the addition of an additional type of fuel as an auxiliary. And when combined, additional fuels are used to maintain the entire process. When designing the technology of wastewater sludge incineration, it is important to pay attention to the discharge of waste gases. Since they contain toxic substances, it is necessary to think of an effective cleaning method: reducing the concentrations of pollutants to the maximum



1 – Storage of sewage sludge; 2 – Loading of sewage sludge; 3 – furnace (reactor);
4 – Burners; 5 – Smoke channel; 6 – Air–to-air energy recovery;
7 – Water–air heat exchanger; 8 – Dry filter;
9 – Ash storage; 10 – Centrifugal fan; 11 – Air compressor.

FIGURE 7.2 Schematic diagram of sewage sludge incineration.

permissible. The second important component as a result of combustion is the resulting ash. It is a finely dispersed brown powder, having the ability to dust and containing toxic compounds, homogeneous in composition. The composition is dominated by silicon oxide, then calcium and iron phosphates, metal silicates can be isolated. We also note another important aspect when solving the disposal of sewage sludge by thermochemical methods. The chemicals most common in sewage sludge are polychlorinated dioxins/furans, polychlorinated biphenyls, polybromodiphenyl and polybromodiphenyl ethers, which are formed, including, during the combustion of sewage sludge, municipal solid waste, etc. Schematically, this process is shown in Figure 7.3.

This causes additional risks of POP entering the air of the urban environment and can negatively affect humans and animals when inhaled.

7.3.2 Pyrolysis

In contrast to incineration, pyrolysis is aimed to a greater extent at obtaining solid and liquid products of thermochemical reactions resulting from the transformation of biomass and, in fact, is a process of "melting" (Mong et al., 2022).

TABLE 7.1
Comparative Characteristics of the Operation of Furnaces during the Heat Treatment of Sewage Sludge, According to Dregulo and Pitulko (2019)

Type Reactors (Furnace)	Exhaust Gas Temperature, °C	Specific Load of the Working Volume for the Substance Destroyed up to the Maximum Permissible Concentration, kg/(m³ * h)	Excess Air Ratio	Coefficient of Unevenness of Stay in the Combustion Zone	The Coefficient of Uneven Stay in
Multipod	310–520	200–400	1.08–1.2	1	Pollution of gases by organic products from fresh portions of the OSV in the upper part of the furnace; low specific heat loads; rotating elements in the high-temperature zone; the use of expensive materials for hollow shaft and scraper agitators; high capital and operating costs
Chamber	650–900	250			Low-weight loads; bulkiness; high metal consumption; increased requirements for corrosion resistance of the grate material and mechanization of furnace devices; large capital costs
Drum	650–1000	10–80	1.1–1.6		The lowest specific heat and weight loads of the furnace volume; destruction of the lining; rapid failure of the furnace due to a sharp change in temperature during the rotation of the furnace and erosion; high capital and operating costs
Spray	650–850	80–100	1.1–1.8		Low productivity; complexity in operation; high capital costs
Cyclonic	1200	600–850	1.04–1.6		The need to install powerful dust-collecting devices and additional equipment for unloading
Fluidized bed	600–850	300–800	1.04–1.3	When applying to layer 1	Uneven distribution and residence time in the layer of solid phase particles; the need for dust collection

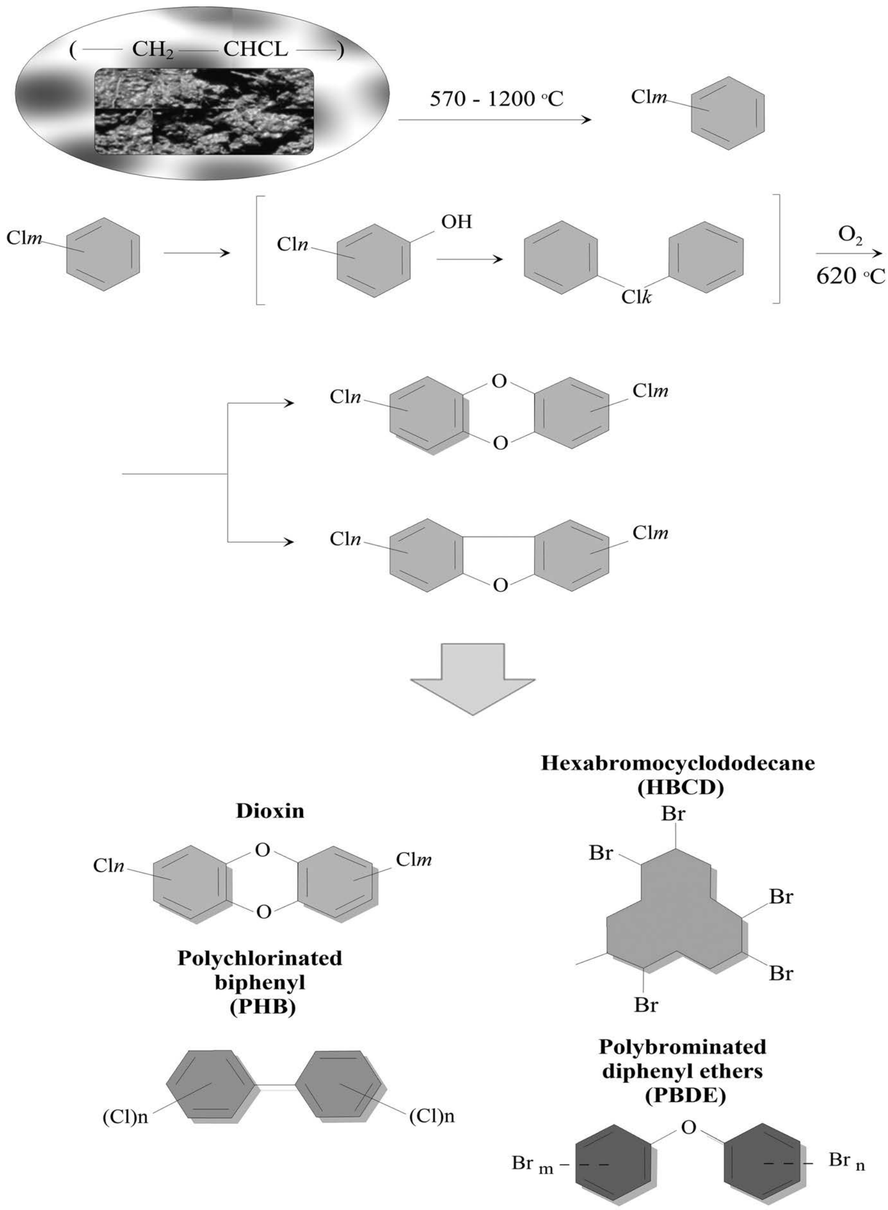
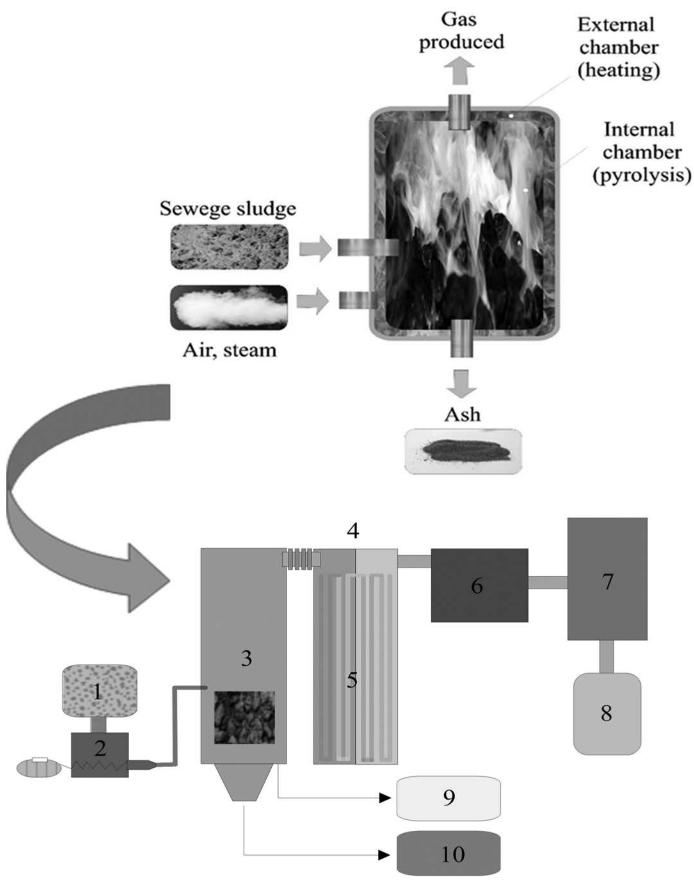


FIGURE 7.3 Thermochemical transformations of organic substances contained in sewage sludge into persistent organic compounds.

The pyrolysis process involves many complex chemical reactions. The combination of these reactions strongly affects the production of the target product, which, equally, also depends on the type of feedstock. In the case of sewage sludge, this is the mass carbon content in the conditionally solid phase of precipitation. Schematically, the pyrolysis process is shown in Figure 7.4. As we can see, technologically, the combustion process and the pyrolysis process differ to a greater extent by the heating process, that is, by a stepwise cycle of temperature exposure. Thermal decomposition of organic components in the waste stream begins at 350–550°C and reaches 700–800°C in the absence of air/oxygen., with rapid pyrolysis, the process temperature reaches 1000°C (Goldan et al.,



1 – Storage of sewage sludge; 2 – Loading of sewage sludge; 3 – Pyrolisis reactor;
4 – Pyrolytic products; 5 – Condensation;
6 – Gas refoming; 7 – Hydro-treatment;
8 – Purification ; 9 – Energy recovery; 10 – Ash dispodal to landfill.

FIGURE 7.4 Schematic diagram of pyrolysis of sewage sludge.

2022). Commonly used pyrolysis reactors are rotary kilns, rotary hearth furnaces and fluidized bed furnaces. The process requires an external heat source to maintain the required high temperature. The main products obtained during pyrolysis of household waste are high-calorie gas (synthesis gas or synthesis gas), biofuels (bio-oil or pyrolysis oil) and solid residue (coke) (Varma et al., 2018). Depending on the final temperature, pyrolysis will produce mainly solid residues at low temperatures, less than 450°C, when the heating rate is low enough, and mainly gases at high temperatures, more than 800°C, at high heating rates (Zaman et al., 2017). The potential of pyrolysis in terms of economic feasibility and environmental sustainability of pyrolysis is not enough. Despite the potential benefits to the environment and the extraction of valuable products, there are a number of challenges that need to be addressed to ensure the sustainability and commercialization of pyrolysis technologies. For this, low-temperature pyrolysis is often used. A feature of low-temperature pyrolysis is that the processed sediment passes into hydrocarbons first in a gaseous state, and after their condensation, "crude oil" is formed (Xiao et al., 2022). Unlike simple combustion, there is less pollution of the atmosphere, and many gaseous products of the process are positively involved in processing. Pyrolysis successfully uses some components of precipitation, such as silicate products and copper salts. These substances act as catalysts in the distillation process, so their additional application during pyrolysis is not required.

7.3.3 Hydrothermal Carbonization

Hydrothermal carbonation of sewage sludge is a promising technology for energy production and the use of final ash and slag products (gyrocarls) in agriculture. The process of hydrothermal carbonation is based on the influence of temperature during anaerobic fermentation of sewage sludge, where as a result of carbonation, organic compounds are released from the solid matrix of precipitation to form CO_2. This process is carried out at temperatures significantly lower than combustion, pyrolysis, gasification, in the range from 150 to 180°C at which the decomposition of carbon (carboxyl and carbonyl groups) occurs up to 250–300°C upon receipt of the final product (Zaccariello et al., 2022). The schematic diagram of hydrothermal carbonization of sewage sludge is shown in Figure 7.5.

The process begins with the preparation of biomass: mechanical impurities are removed from it and then crushed and wetted. Next, the biomass is sent to HTC reactor, which is supplied with steam 180–220°C, resulting in a pressure of 10–25 bar. The reaction produces hydroxonium (H_3O^+) hydroxonium, oxonium, hydronium. Part of the carbon in the form of carbon dioxide is released into the atmosphere. In fact, the process of hydrothermal carbonation simulates or reproduces the natural process of coal formation, only in industrial and in a shorter time. In a simplified form, the equation of the hydrothermal carbonization reaction can be represented in (7.1):

$$C_6H_2O_6 \rightarrow C_6H_6O_3 + 3H_2O \rightarrow C_6H_2O + 5H_2O \tag{7.1}$$

It is important to consider that the carbon composition in the hydro-coal obtained after the process may differ depending on the stage at which the sewage sludge was collected, for example, from primary or secondary sedimentation tanks (Merzari et al., 2020). When using this method, it is very important to find a balance between the process temperature and the final characteristics of hydrogels. For example, hydrogels obtained at a temperature of 220°C had a greater adsorption capacity

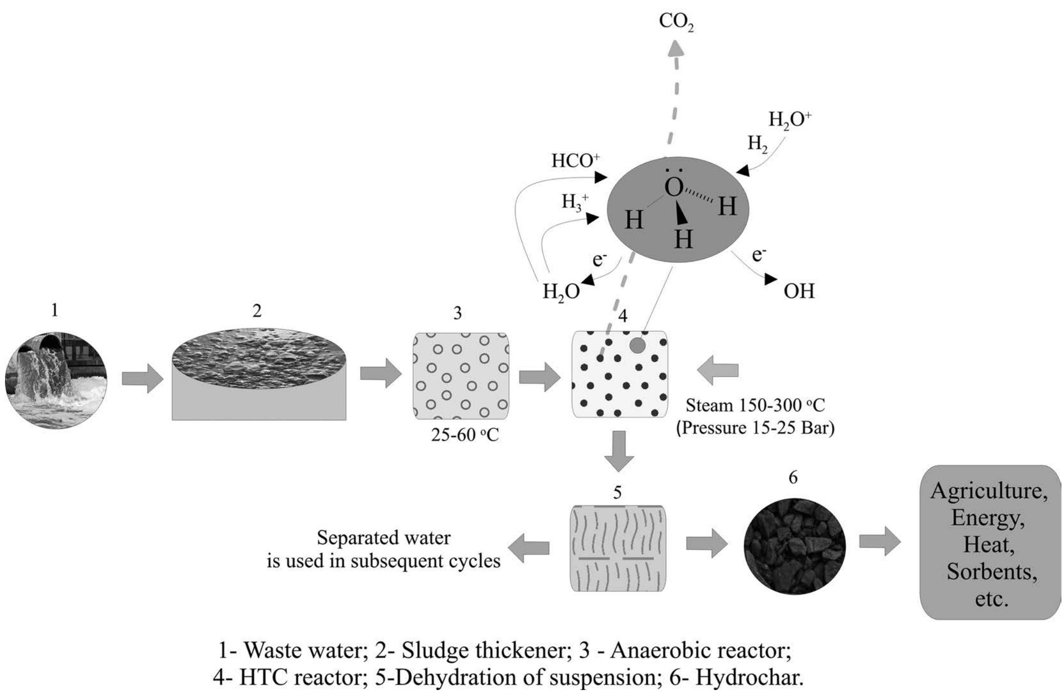


FIGURE 7.5 Schematic diagram of hydrothermal carbonization of sewage sludge.

than hydrogels obtained at a temperature of 260°C (Niinipuu et al., 2020). It is believed that the process of hydrothermal carbonation makes it possible to obtain carbon-neutral fuel and therefore this type of thermochemical processing of sewage sludge is promising from the point of view of the development and implementation of green technologies by Jellali et al. (2022).

7.3.4 Gasification

The process of gasification of waste (biomass) includes several stages that take place inside the reactor: drying, pyrolysis, combustion recovery/gasification, in fact, one process (Figure 7.6).

The process of gasification of sewage sludge takes place at a temperature of 650–1000°C. During the gasification process, useful products are released, including CO, a small amount of CH_4, H_2, as well as undesirable N_2, CO_2 and another hydrocarbons (Zhang et al., 2019).

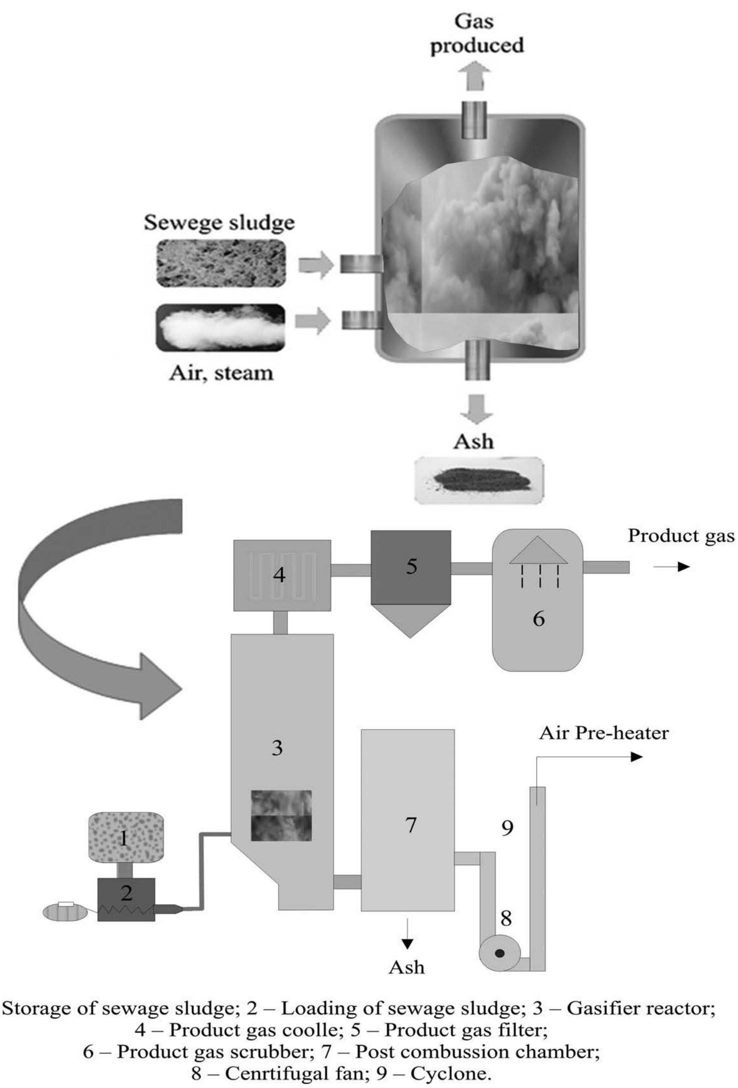


FIGURE 7.6 Schematic diagram of sewage sludge gasification.

The gasifying agent enters (Werle & Sobek, 2019) into an endothermic reaction with carbon (7.2), resulting in the formation of CO:

$$CO + H_2O \rightarrow CO + 2H \tag{7.2}$$

and then reacts exothermically with CO to form mainly CO_2 and hydrogen (7.3):

$$CO + H_2O \leftrightarrow CO_2 + 2H \tag{7.3}$$

The by-product of gasification, which has the greatest negative impact, is resin. The higher the gasification temperature and the equivalence coefficient, the lower the resin production, whereas with an increase in the value, the calorific value of the synthesis gas and the gasification efficiency decrease (Molino et al., 2018). Therefore, in order to ensure good gasification characteristics and low resin formation, it is necessary to observe a high-temperature level and low equivalence coefficients. And for this process, as for other thermochemical processes, it is important to choose the right reactor. The most common are gasification reactors consisting of a fluidizing column made of stainless steel. The main technological characteristics of this reactor: a distribution plate consisting of a series of stainless-steel gratings separates the gas inlet chamber or pressure chamber, which also acts as a gas heater, from the fluidization column. There is also a filter (ceramic) located behind the reactor, which allows you to collect waste ash. Reactors are usually equipped with a biomass supply system (a combination of mechanical and pneumatic transport devices is used). The scheme also uses nitrogen as a transport gas through the supply system, the rest of the gasifying agent is air entering the reactor through an aerodynamic chamber. This makes it possible to create conditions for a fluidized bed consisting of approximately ~3% oxygen and a fluidization rate of 0.30 m/s at a gasification temperature of 850°C (Migliaccio et al., 2021). As in other processes, gas purification plays an important role. In this case, there are two of them: (1) purification of the obtained (commercial) gas from excess impurities (as a rule, wet gas purification technology is used for this); and (2) purification of exhaust gases from soot, dust and other hazardous substances. Thus, considering the technologies of thermochemical treatment used, it is necessary to focus on the problem of ash disposal, since the ways of commercialization of secondary products (mainly gas) are in principle clear. Many publications are devoted to this. Is it possible to get a secondary product from the ash? We will try to answer this question further.

7.4 PROSPECTS FOR THE USE OF PRODUCTS OF THERMOCHEMICAL TREATMENT OF SEWAGE SLUDGE

7.4.1 Commercialization of Ash and Slag Waste

Waste incineration processes have been sufficiently studied, and technologies based on them have been around for 40 years. The use of ash from the combustion of sewage sludge in the manufacturing technologies of binders is an advantageous economic solution for production, since it has properties close to binders, but at the same time its chemical activity is higher. Comparative analysis of the component composition of ash and slag waste with prospects for use as an additive to building materials is shown in Table 7.2.

Depending on the type of binder and the concentration of ash in it, it is possible to change such properties of the resulting material as plasticity, dispersion and ability to harden. Binders can be either organic or mineral. Organic binders include bitumen and other products formed as a result of oil distillation. In general terms, binders can be described as materials in the form of powders, which, after adding water to them and mixing, are plastic dough that is conveniently processed, which over time solidifies into a stone-like body. Mineral binders are widely used in many

TABLE 7.2
Comparative Analysis of the Component Composition of Ash and Slag Waste for Use as an Additive to Building Materials, According to Dregulo and Pitulko (2019)

	Combustor Sewage Sludge Ash	MSW Combustor Ash	Combustor Burnt Shale Ash
Substance/Metal		**Concentration, mg/kg**	
Ni[a]	51	95–240	13–22
Co[a]	8.3	23–69	4.4–5.5
Cu[a]	640	860–1400	6.5–12.0
Mn[a]	1400	0.8–1.7	320–600
Cr[a]	78	140–530	15–17
Pb[a]	52	7400–19,000	31–39
Ca[a]	11	250–450	0.3–0.37
As[a]	–	3195	–
Zn[a]	850	19,000–41,000	12–14
Hg	0.05	0.8–7	61–99
Al	36,000	–	0.089–0.1
Fe	35,000	–	27,200–37,300
Mg	7100	–	13,900–32,600
K	6300	–	12,500–32,500
Na	1900	–	37,500–38,000
Ca	36,000	–	700–800
Benz[*a*]pyrene	–	–	180,000–355,000
S^- ion	3500	–	0.0054–0.72
Cl^-	–	–	4500–6400
SiO_2	740,000		19,200–27,200

[a] Gross content.

industries. The main one is construction. In this regard, there are a lot of types of such substances in our time. All of them can be divided by chemical composition into four main groups:

- Hydraulic binders – they differ in that after interacting with water they begin to harden in the air, and then, having hardened and being under water, they continue to gain strength. These include various types of cements, weakly and strongly hydraulic lime substances.
- Air binders include materials that, after setting with water in the air and solidifying, harden and gain strength exclusively while in the air. With their further use, the impact of water is unacceptable, since the formed stone-like body will collapse. Gypsum and magnesia substances, lime dough, slaked and quicklime belong to this type of binders.
- Acid-resistant binders are distinguished by the fact that after hardening and gaining strength, they are not exposed to mineral acids.

These substances are used when they can be exposed to an acidic environment, the most common example is acid-resistant cement. Autoclave hardening binders (these include calcareous-siliceous and calcareous-nepheline) are able to harden only during hydrothermal treatment under pressure (Pacewska & Wilińska, 2020). Since ash has physicomechanical properties similar to quartz sand and crushed stone, it was concluded that it can be added to building mixes as a replacement for the former, which will be beneficial both environmentally and economically. However, in addition to the good properties obtained, there may be fewer positive results. Since the ash particles have a different, irregular shape, large amounts of water are required when making samples from cement and ash from sewage sludge. With an increase in the size of the ash grains, the workability improves, and the setting time increases. Higher water absorption is associated with a larger free surface of the particles. To reduce the amount of water used in the manufacture of the mixture, chemical additives can be added: plasticizers and superplasticizers (Papayianni et al., 2005). Since the amount of ash increases and the amount of water increases, the workability decreases sharply, and the strength characteristics also decrease. Plasticizers can solve this problem. The combustion temperature of sewage sludge also affects the composition of the resulting ash. The higher the temperature, the lesser the amount of potassium and magnesium is observed in the composition; however, the calcium content increases at the same time. The latter is explained by the process of decomposition of calcite $CaCO_3$ and the formation of CaO. High concentrations of silicon oxide and aluminum oxide indicate that ash can be used in the preparation of cement mixtures. However, the presence of a significant amount of phosphates in the composition may affect the strength characteristics when ash is added to cements: cement sets more slowly, as a result of which the initial strength of the samples may be lower. The size of the ash particles also strongly affects the physical and mechanical properties of the resulting materials. It is known that with an increase in dispersion, the setting time increases, while water particles grasp better, since the specific surface area of ash particles is larger and further leads to higher strength characteristics even in the production of polyethylene (Mort et al., 2022). Since the ash particles of sewage sludge have high porosity, when used with cements, water costs may increase, which may further lead to a decrease in strength. In addition, an increase in the ash concentration in the cement mixture can negatively affect the final result – there is a decrease in density. This suggests that when using ash, it is necessary to carefully select the ratio of the source material and composite. Along with the positive practice of using ash and slag, many researchers note both the competitive advantages of products based on them, and acceptable environmental and toxicological indicators when using them. Concrete obtained with the addition of 10% biomass ash to the total composition had higher compressive strength values for the reference composition, and the levels of chemical leaching in marine and freshwater reservoirs were similar to the reference samples (Barbosa et al., 2013). Considering the difference in the component compositions of ash and slag waste, the most promising from the point of view of environmental safety are studies on the production of aggregate concrete, in the production of which heavy metal compounds are embedded in the silicate and aluminosilicate structure of the starting materials (clay and sludge) at a temperature of 1110°C (Franus et al., 2016). Therefore, one of the priority tasks of this direction is to determine the most effective technology of thermal processing. Ash and slag wastes contain a large number of silicates, a similar elemental composition is available in the materials of the silicate industry, which means that there is a possibility of using ash in the production of glass materials. It becomes possible to use a new type of glass materials as a thermal insulator, since when ash is added, the thermal conductivity coefficient turns out to be relatively low (Ozkutlu et al., 2018). The matrix structure of the material using sewage sludge is shown in Figure 7.7.

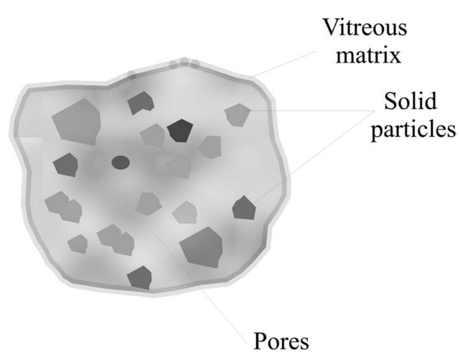


FIGURE 7.7 Matrix aluminosilicate structure of ash of sewage sludge.

But the main difficulty of this method lies in the small scope of application of such glass-crystal materials, since it is almost impossible to carry out deep cleaning from ash. The composition of ash and slag, rich in valuable chemical elements, allows them to be extracted for further use. Secondary coal can be obtained from them, which means additional fuel, iron-containing magnetic concentrate used in the metallurgical industry, as well as many other valuable elements that are most expediently used in the form of oxides: titanium dioxide in the paint industry, in the production of rubber and glass materials, manganese dioxide acts as a component of ceramic products, oxide cobalt for use as a blue dye in various industries and strontium oxide is a component of oxide cathodes of vacuum electronic devices, pyrotechnic compositions.

7.4.2 Problems and Prospects for the Formation of Clusters for the Processing of Waste into Value-Added Products

As emphasized earlier, the main task of cities is to reduce the amount of accumulated waste and turn it into a safe product. Therefore, heat treatment of waste is a promising technology, both from an environmental and economic point of view. Cluster affiliation of certain types of waste may have positive prospects over time (availability of infrastructure), since clusters have two forms of ranking: (1) on a regional basis; (2) based on production. Methods of realization of secondary products in the disposal of sewage sludge by thermochemical methods are shown in Figure 7.8.

The most likely in the conditions of the realities of obtaining a target product that is, if not competitive, then at least has the potential for further market conversion, is the co-incineration of sewage sludge and solid waste to obtain used as an additive in building materials. However, serious problems arise when different types of waste with different properties (moisture content, heat of combustion, etc.) are disposed of in a single process. Gorenje due to the fact that solid waste is multicomponent, the economic burden on the preparation of waste as an alternative fuel, which includes a number of technological processes, increases. The most problematic points of using

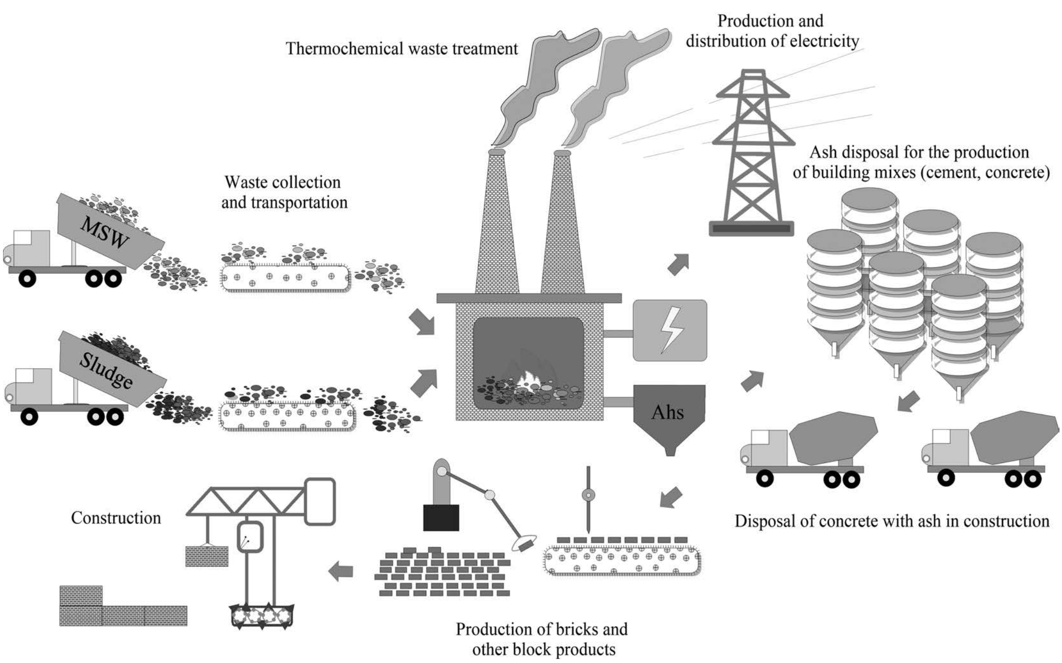


FIGURE 7.8 Ways of realization of secondary products in the use of sewage sludge by thermochemical methods as a basis for the formation of a production cluster.

thermochemical methods of wastewater sludge treatment are (1) long-term operation of devices exposed to high temperatures and the abrasive appearance of the final products affecting strength and characteristics (2) lack of comprehensive information on the transformation of pollutants at high temperatures and their subsequent impact on the environment and humans (Winchell et al., 2022). Several hundred different ways of using ash and slag waste are known, but since some of them carry large energy and monetary costs for changing technological chains in production, not all methods can be economically profitable. Nevertheless, the use of ash for construction is most widespread, due to its availability, in our and other countries.

The choice of recycling technology will have to be made sooner or later. It will be either relatively cheap soil deposition and all the environmental risks associated with this method, or economically capacious, but possibly more promising from the point of view of socio-ecological climate thermal disposal methods. Of course, choosing the second option, you should carefully analyze the most vulnerable points of this solution. In the absence of an economic component (financial benefit) for all participants of such a business project, primarily the producer and consumer, the environmental factor will inevitably affect the administrative policy of urban management. The question in this case will always be solved not how to dispose of, but where, because it will always have to be solved in the plane of territorial planning. But given the pace of urbanization of suburban areas, the appearance of places for hazardous waste disposal on them, the development of this territorial and economic unit will stagnate for a long time. That is why the state structure should unite and support the formation of a long-term perspective and guarantee of investments between the cluster operators: a secondary resource from waste (ash from incineration used as raw materials for building materials) and a manufacturer of building materials. In this case, the state structure should perform the function of a guarantor of these relationships, the condition of which should be an agreement on the use of a secondary resource in the manufacturer's technology. The scheme of such interaction can be represented as shown in Figure 7.9.

Economic instruments when creating such a cluster or as an integral part of a single waste cluster have three main goals:

- sustainable development of the state environmental strategy;
- covering the costs of waste disposal;
- making a profit.

The first two positions are quite clear, the question remains for making a profit. Infrastructure costs are rarely covered by local authorities. The investment costs of operation and maintenance

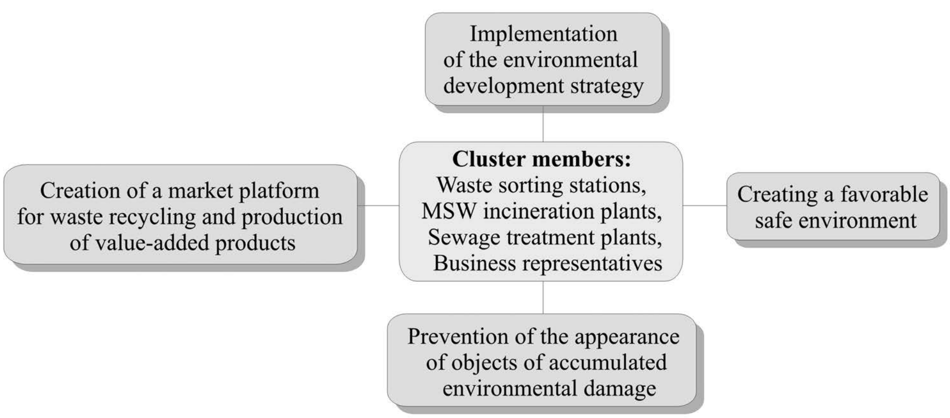


FIGURE 7.9 Potential effects of the formation of a cluster of thermochemical treatment of sewage sludge to obtain secondary products with added value.

will thus prove burdensome for waste operators. In general, for the waste industry, it will never lose its relevance, it is relatively easy to find investors for infrastructure, but it is often impossible to find those willing to participate in covering the costs of operation and maintenance. It is here that the involvement of state structures will play a decisive role in the formation of this cluster using economic support tools. Economic instruments can be the development of a national subsidy program, following the example of India, introduced in 2009 and implemented on the basis of a benchmarking system, economic instruments in solid waste management (2009). It also means tax benefits, subsidizing or corporatization of cluster enterprises, consolidation of cluster policy (e.g., the legally fixed obligation to use only BAT in the field of waste management at water utility companies and MSW operators) use at the regional level (regional support for municipal services of separate collection of MSW, etc.). However, along with the presence of a different list of promising technologies with a high potential to conquer the market of products from secondary raw materials, all actions aimed at consolidating efforts to form a cluster will not have the desired effect without the participation of state subsidies and/or legislative support.

7.5 CONCLUSIONS AND PERSPECTIVES

Thermochemical treatment of sewage sludge (incineration) is one of the most effective methods of their disposal, as their volumes are significantly reduced as a result. For large cities, this problem is similar to the problem of solid waste disposal. Using the methods of combustion, pyrolysis, gasification, hydrothermal carbonation, it is possible to obtain secondary products with added value. Nevertheless, the difference in the use of these methods for the disposal of sewage sludge plays a big role. This is due to the economic efficiency of the process and environmental aspects, which often require optimization from the point of view of sustainable development. As a result of thermochemical processes, ash or sludge remains, which also need to be disposed of correctly. The most optimal way to dispose of ash and sludge is to use them in the production of building materials (binders, etc.). This makes it possible to solve the problem of environmental pollution without compromising the economy of the enterprise, since their addition to building mixes allows you to save on expensive materials, while improving the characteristics of the products obtained. Therefore, it is advisable to form production clusters that include thermochemical treatment of waste to produce energy and ash for the production of building materials. The determining factor here will be the degree of interest of state bodies in promoting the policy of waste-free production or, for example, the popularization of the philosophy of "zero waste", which is based on the moral component in its implementation as more significant but at the same time does not reject the difficult path of its economic side.

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8 Pyrolysis of Sewage Sludge

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8.1 INTRODUCTION

A large amount of sewage or wastewater is usually produced in the daily production and living process of people. Most of this sewage or wastewater is treated by the activated sludge process. In the process of purifying sewage or wastewater, a terrific amount of sewage sludge is produced in the form of by-products (Gholipour et al., 2022). Sewage sludge generally includes two types: primary sedimentation sludge (mainly from the primary sedimentation of solid substances in sewage or wastewater) and secondary sedimentation sludge (mainly from the biological treatment of sewage or wastewater) (Figure 8.1) (Liew et al., 2022). Sewage sludge has an extreme composition, and it is a heterogeneous mixture, mainly including six parts: water (content even exceeds 95%), nutrient components such as nitrogen and phosphorus, pathogenic microorganisms, non-toxic organic substances (proteins, carbohydrates, lipids, etc.), toxic organic and inorganic substances (heavy metals, polycyclic aromatic hydrocarbons [PAHs], polychlorinated biphenyls, dioxins, etc.), and non-toxic inorganic compounds (compounds containing silicon, aluminum, calcium, and manganese) (Rulkens, 2008).

With the surge of sewage or wastewater and the improvement of environmental standards, the amount of sewage sludge to be treated is also increasing. For example, the quantity of sewage sludge (dry solids) produced by China is 7.46 million tons in 2015, which reach 11.63 million tons in 2020 (Figure 8.2). The total amount of sewage sludge produced by the EU-27 countries in 2003–2006 is 10 million tons (dry solids), which is expected to exceed 13 million tons by 2020 (Tarpani et al., 2020). By 2050, the quantity of sewage sludge produced in ASEAN countries is estimated to be 24 million to 40 million tons per year (Quan et al., 2022). The amount of sewage sludge produced in the United States in 2018 is about 14 million tons (dry sludge) (Thomsen et al., 2022). Considering that sewage sludge contains both recyclable and polluting components, it is agreed that the treatment or disposal of sewage sludge should meet three requirements, resource reuse, pollutant control, and low-cost input.

So far, people have developed a variety of sewage sludge treatment processes, which can be roughly divided into three categories, biological treatment, chemical treatment, and thermal treatment (Figure 8.3) (Liew et al., 2022). Chemical treatment is to inhibit and kill microorganisms and eliminate the possible harm of sewage sludge to the environment (odor and infectious diseases) by adding chemicals to sewage sludge. Thermal treatment is to convert sewage sludge into energy, chemical raw materials, and functional carbon materials through thermal degradation under certain temperature and pressure conditions. Common thermal treatment technologies include liquefaction, hydrothermal carbonization, pyrolysis, incineration, and gasification. The biological treatment uses aerobic or anaerobic microorganisms to accelerate the degradation of organic components in sewage sludge. Compared with chemical and biological treatment technology, more attention has been paid to thermal treatment technology because of its shorter treatment cycle and higher pollution control effect (Li et al., 2022a).

Pyrolysis of sewage sludge is usually performed at moderate to high temperatures in the absence of oxygen. After the disposal of sewage sludge through pyrolysis, the liquid-phase (bio-oil), the solid-phase (biochar), and the gas-phase (syngas) products can be separated. As shown in Figure 8.4, sewage sludge pyrolysis processes include slow pyrolysis, fast pyrolysis, vacuum pyrolysis, and

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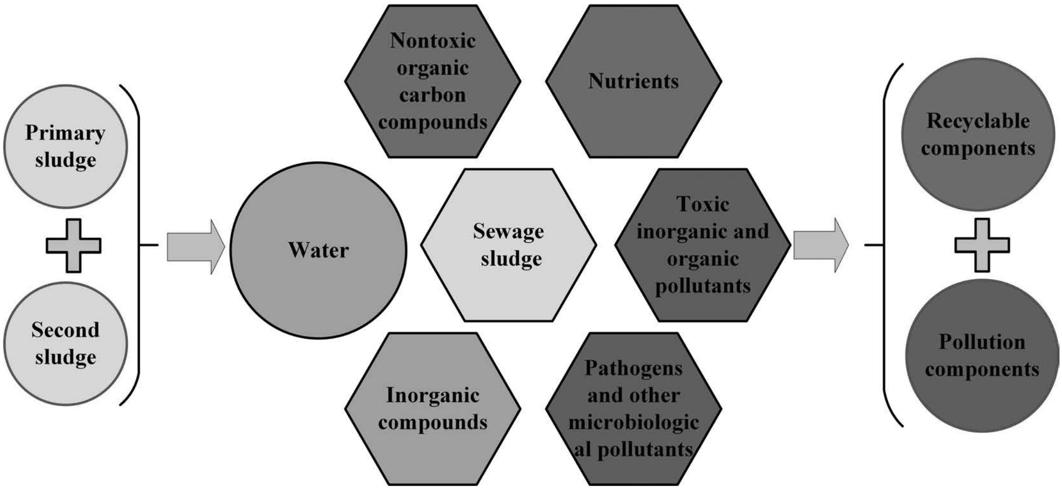

FIGURE 8.1 Basic composition of sewage sludge.

hydro pyrolysis (Haghighat et al., 2020; Syed-Hassan et al., 2017). Flash/fast pyrolysis processes are designed to obtain the maximum production of liquid bio-oil or syngas, while slow pyrolysis is usually used for producing more solid products (biochar). Thus, compared with slow pyrolysis, flash pyrolysis and fast pyrolysis are carried out at a higher heating rate with a shorter reaction time. After proper upgrading, bio-oil could be applied as liquid fuel and industrial materials, while solid-phase biochar can be further developed into functional carbon materials (Ihsanullah et al., 2022; Rangabhashiyam et al., 2022).

This chapter first clarifies the reaction mechanism of the treatment of sewage sludge by pyrolysis. Then, the influence of key factors on the treatment of sewage sludge by pyrolysis is discussed.

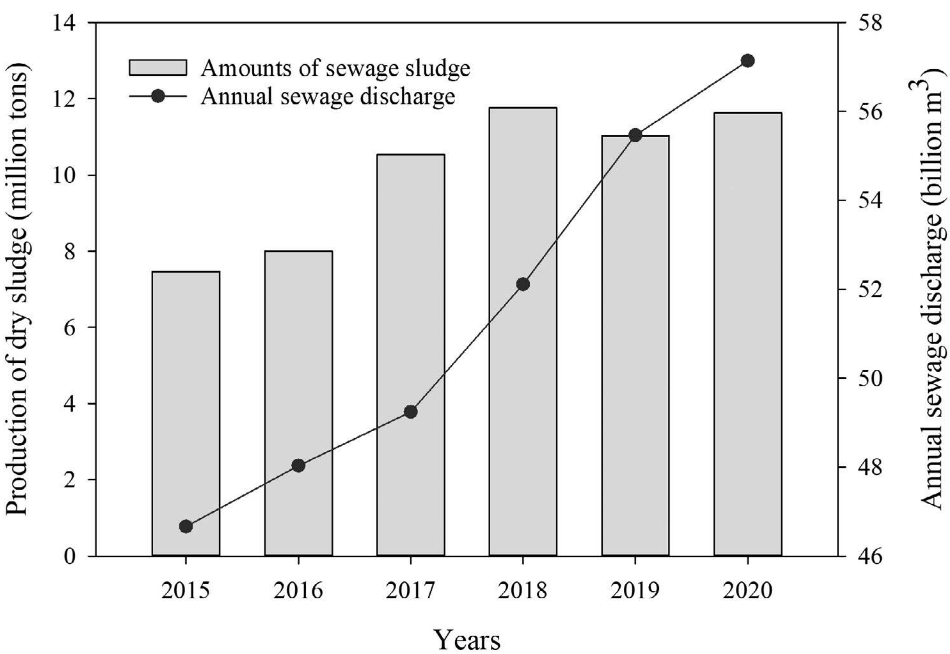

FIGURE 8.2 Production of sewage sludge versus annual sewage discharge in China.

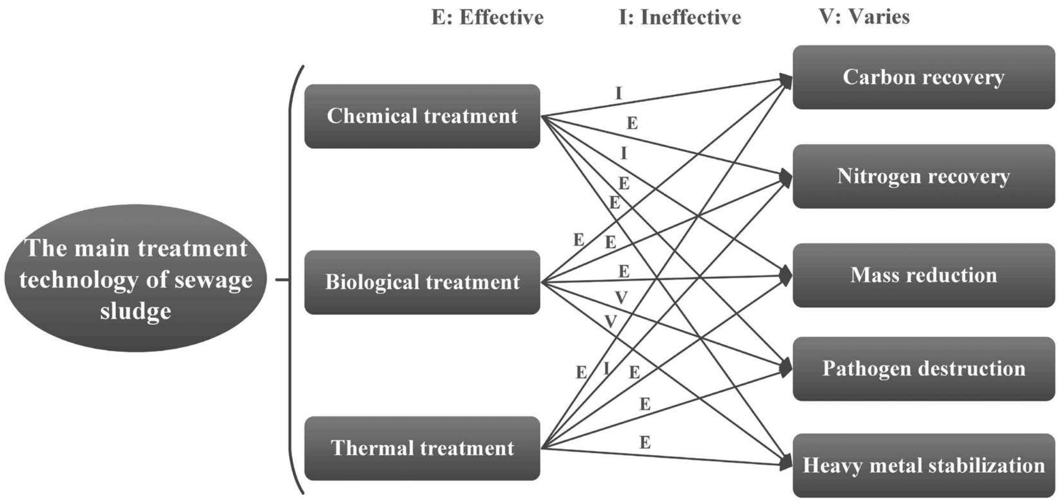

FIGURE 8.3 Comparisons of different treatment methods of sewage sludge.

Next, the co-treatment of sewage sludge with other biomass/waste by pyrolysis is summarized. In addition, the control effect of the pyrolysis process on the pollutants in sewage sludge is introduced. Finally, the development prospect of the treatment of sewage sludge by pyrolysis is discussed.

8.2 PYROLYSIS MECHANISM OF SEWAGE SLUDGE

The main organic matters in sewage sludge produced in conventional municipal sewage treatment plants include protein, carbohydrates, and lipids. The content of protein is ranging from 24% to 42%, while that of carbohydrates ranks in the range of 7–18%. The content of lipids can reach 1–14% (Li et al., 2015). The pyrolysis behaviors of the previous organic matter in sewage sludge have been explored by the thermogravimetric technique. The pyrolysis of carbohydrates was found at around 255°C, while that of lipids was observed at around 300°C. The decomposition of protein was reported between 360 and 525°C (Alvarez et al., 2015a; Alvarez et al., 2015b). However, at a

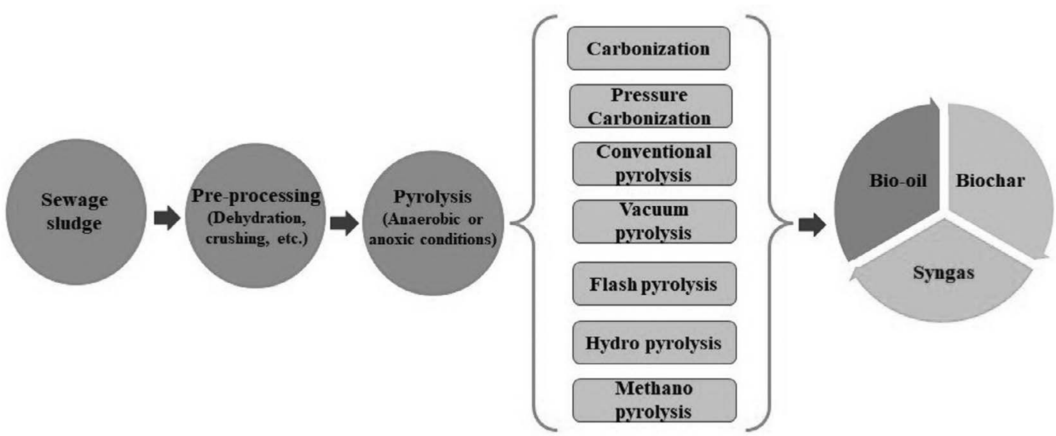

FIGURE 8.4 Typical pyrolysis processes for sewage sludge.

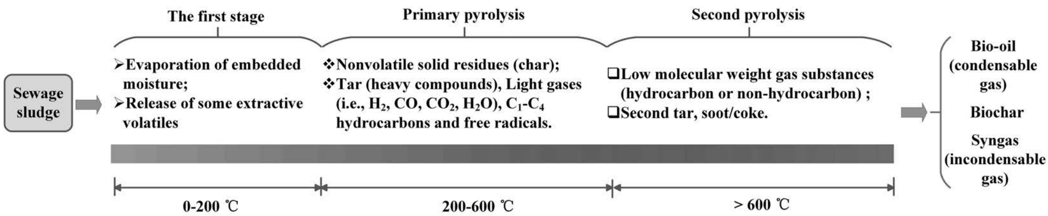

FIGURE 8.5 Pyrolysis mechanism of sewage sludge pyrolysis.

sufficiently high heating rate, the difference between the decomposition stages becomes indistinct, and the different organic components in the sewage sludge decompose almost at the same time.

Figure 8.5 depicts the basic decomposition routes for the disposal of sewage sludge by pyrolysis. In short, the decomposition of sewage sludge during the pyrolysis process includes three phases. The first phase is the evaporation of embedded moisture and the release of some extractive volatiles (100–200°C). The second phase is called primary pyrolysis, during which the organic substances in sewage sludge begin to transform at about 200°C, and volatile products and nonvolatile solid residues (char) are formed through various bond-breaking and formation reactions. The volatile products derived from primary pyrolysis mainly include tar (heavy compounds), gases, C_1–C_4 hydrocarbons, and free radicals. CH_3COOH, CH_3OH, HCHO, CH_3CHO, and cyano-compounds also appear in primary volatiles (Caballero et al., 1997; Nowicki and Ledakowicz, 2014). The compounds in primary volatiles are unstable and will be further decomposed when the reaction temperature reaches around 600°C, which is seen as secondary pyrolysis. At this time, a large number of low molecular weight hydrocarbon or non-hydrocarbon gas substances will be formed, and at the same time, some more stable aromatic compounds will be formed (de Andrés et al., 2016). Among them, some tar components form soot or coke through a recombination/polymerization reaction. After these reactions, the condensable gas is cooled to form a liquid-phase product, that is, bio-oil.

8.3 INFLUENCE OF PYROLYSIS FACTORS

The pyrolysis process of sewage sludge is very complex, involving three-phase complex chemical reactions and there are many influencing factors (Figure 8.6) (Haghighat et al., 2020). On the one hand, the physicochemical characteristics of sewage sludge itself, such as chemical composition and particle size, will directly determine the yield and physicochemical characteristics of pyrolysis products to a certain extent. For example, the content and composition of organic matter in sewage sludge will largely determine the yield and chemical composition of liquid-phase products. As well known, the ash in sewage sludge usually contains many metal oxides. Part of these metal oxides has a catalytic effect in the disposal of sewage sludge through pyrolysis. Therefore, the ash composition in sewage sludge is also an important factor affecting the disposal of sewage sludge through pyrolysis. The particular size of sewage sludge will directly affect the quality and heat exchange rate of the pyrolysis process (Fonts et al., 2012).

In addition, the treatment of sewage sludge by pyrolysis is largely affected by various process parameters, including reaction temperature, reaction time, reaction atmosphere, catalyst, and feed rate. For the fluidized-bed reactor, the reaction temperature of 400–550°C is appropriate. In this temperature range, the output of liquid products (bio-oil) can reach the maximum (Fonts et al., 2008). The increase in reaction temperature can ensure that there is enough energy required for the transformation of organic components in sewage sludge. Therefore, in the initial stage of the increase of pyrolysis temperature, the yield of liquid-phase products is gradually increased until the optimal reaction temperature. If the pyrolysis temperature continues to rise, it will cause the secondary reaction of liquid products, leading to the decline of liquid product yield. However, if the treatment of sewage sludge is performed in fixed-bed reactors, the increase in reaction temperature

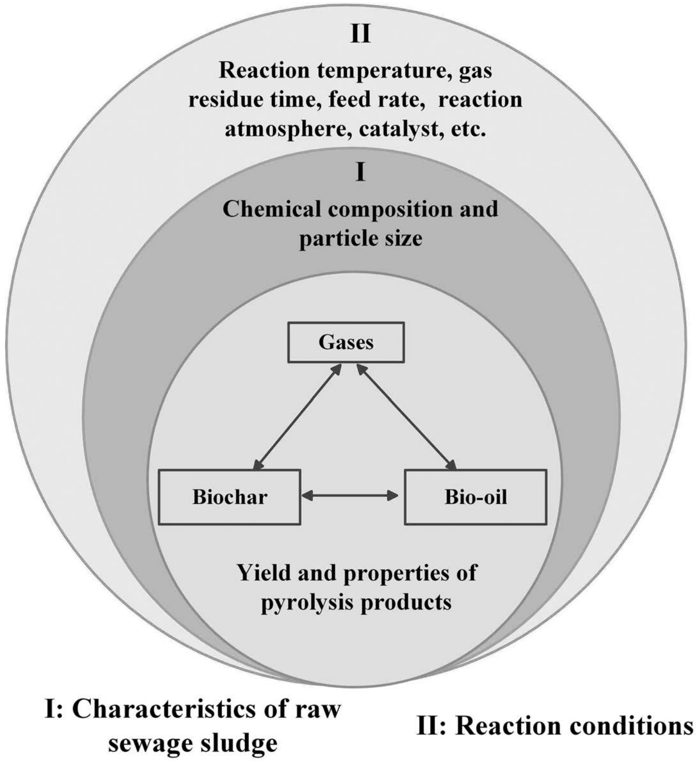

FIGURE 8.6 Key influencing factors of sewage sludge pyrolysis.

will promote the yield of liquid products (Jindarom et al., 2007). The reaction time (gas residence time) is mainly controlled by the flow rate of inert gas and usually has an optimal value. It is necessary to ensure a certain reaction time so that the solid particles can react completely on the reaction bed, but excessive reaction time will lead to the secondary degradation of liquid products (Fonts et al., 2012). As shown in Table 8.1, the operating parameters required by different pyrolysis processes often vary greatly.

TABLE 8.1
Typical Operation Factors for the Different Pyrolysis Processes

Pyrolysis Process	Residence Time	Heating Rate (°C/s)	Temperature (°C)	Major Products
Carbonization	Hours-days	Very low (<5)	300–500	Biochar
Pressure carbonization	15 min to 2 h	Medium (50)	450	Biochar
Conventional pyrolysis	Hours	Low (10)	400–600	Biochar, bio-oil, syngas
	5–30 min	Medium (50)	700–900	Biochar, syngas
Vacuum pyrolysis	2–30 s	Medium (50)	350–450	Bio-oil
Flash pyrolysis	0.1–2 s	High (100)	400–650	Bio-oil
	<1 s	High (100)	650–900	Bio-oil, syngas
	<1 s	Very high (>500)	1000–3000	Syngas
Hydro pyrolysis	<10 s	High (100)	<500	Bio-oil
Methane pyrolysis	<10 s	High (100)	>700	Chemicals

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Methane, hydrogen, carbon dioxide, and nitrogen are usually used as reaction atmospheres for sewage sludge pyrolysis. Carbon dioxide seems to be more effective than nitrogen in enhancing the formation of liquid products (Jindarom et al., 2007). At higher reaction temperatures, methane and hydrogen facilitate the formation of gas products, while the formation of liquid products is enhanced by hydrogen at the low reaction temperature. It has also been proved that directly using gas products from sewage sludge pyrolysis as a reaction atmosphere can improve the yield of liquid products (Park et al., 2010). The change in feed rate will involve the alteration of material residence time in the reaction center and gas residence time and even influence the catalytic ability of ash in raw sewage sludge feedstock. Therefore, its influence on the treatment of sewage sludge by pyrolysis often depends on the comprehensive results of many factors.

Some researchers also found that the yield or property of the target product can be improved by adding a catalyst during sewage sludge pyrolysis, for example, metal oxides, zeolites, and sewage sludge-derived char. Fe_2O_3 and ZnO can increase the production of solid products by restraining the decomposition of organic components in sewage sludge. The yield of liquid products was improved by the addition of aluminum trioxide, calcium oxide, or titanium dioxide, which will enhance the decomposition of organic components in sewage sludge (Gao et al., 2020). HZSM-5 zeolites possess deoxygenation, denitrogenation, and selectivity properties and have also been found to be an effective catalyst for the treatment of sewage sludge by pyrolysis (Liu et al., 2016; Tian et al., 2014; Wang et al., 2017). In addition, some active metals or metal oxides can be loaded on the HZSM-5 zeolite catalyst to improve the selectivity of the catalyst. Different from traditional biomass (lignocellulose and algae), sewage sludge usually contains a high proportion of ash (including many metal components), which is the basis of the catalytic activity of sewage sludge biochar. Moreover, the solid product (biochar) produced from the treatment of sewage sludge by pyrolysis at high temperatures has a high specific surface area and void volume. Therefore, sewage sludge-derived biochar itself or loaded with a certain amount of active metals has been proven to have high catalytic activity during the disposal of sewage sludge through pyrolysis (Daorattanachai et al., 2018; Tu et al., 2012; Yu et al., 2016).

8.4 CO-PYROLYSIS OF SEWAGE SLUDGE WITH OTHER BIOMASS/WASTE

Sewage sludge usually contains a high concentration of moisture as well as ash, which leads to a low calorific value of sewage sludge itself. In addition, the sewage sludge also contains a certain amount of heavy metals and sulfur-, nitrogen-, and chlorine compounds. These characteristics are not conducive to the pyrolysis process of sewage sludge. Some researchers have tried to introduce relatively clean biomass or organic wastes in the disposal of sewage sludge through pyrolysis, which unquestionably results in the dilution of undesirable components in sewage sludge. More importantly, through the mixing of different biomass/waste, a synergistic effect is usually generated. Finally, the yield or quality of the target product can be improved (Manara and Zabaniotou, 2012). So far, the joint processing of sewage sludge and other biomass/wastes by pyrolysis has received much attention. The involved biomass or organic wastes include waste plastics (Li et al., 2022b; Wang et al., 2021), lignocellulose (Dai et al., 2022; Kwapinska et al., 2021; Liu et al., 2022), algae (Chakraborty et al., 2021; Saleh Khodaparasti et al., 2022; Wang et al., 2022a), animal droppings (Ruiz-Gómez et al., 2017), and other wastes (Li et al., 2021; Wang et al., 2022b) (Figure 8.7).

The possible roles of the addition of other biomass/waste include three aspects: (i) the promotion of the production of hydrogen, carbon monoxide, and other gas; (ii) the improvement of the structure of biochar; and (iii) the decrease of undesirable components in bio-oil (Gao et al., 2020). The addition of corn straw (5 wt.%) during the disposal of sewage sludge through pyrolysis can inhibit the formation of O-containing compounds in bio-oil. Meanwhile, solid biochar products possessed a lower ecological risk of heavy metals (Zhang et al., 2022). Compared with bamboo sawdust, the introduction of rice husk during the disposal of sewage sludge by pyrolysis showed a better effect on the stabilization of heavy metals, while bamboo sawdust facilitated the formation of biochar with higher stability (Zhang et al., 2020). When the blending ratio of microalgae (*Chlorella Vulgaris*)

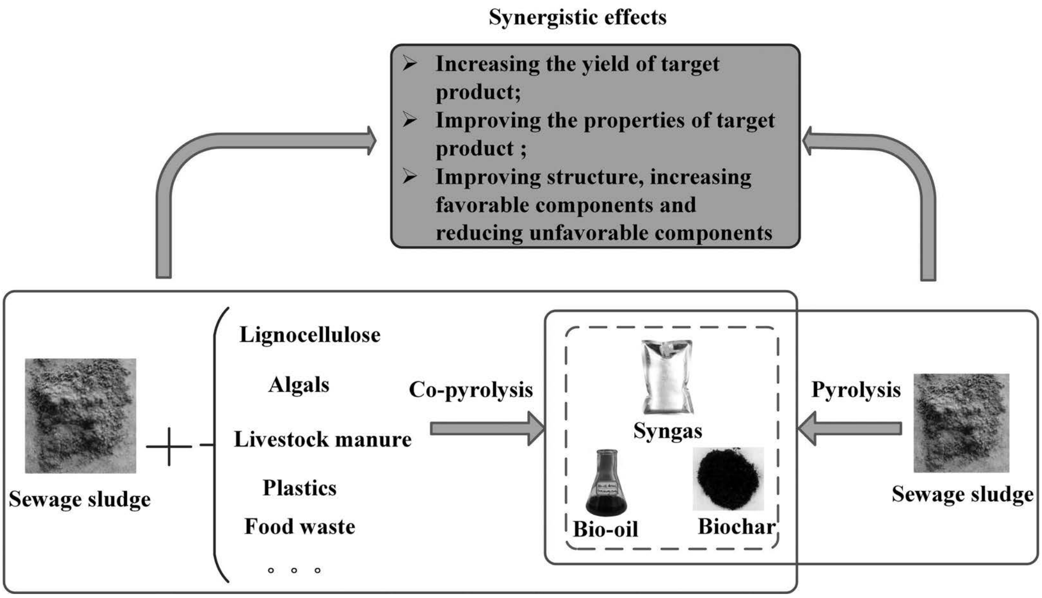

FIGURE 8.7 Co-pyrolysis of sewage sludge and other biomass/waste.

and sewage sludge is 0.82, the co-pyrolysis process yielded the maximum bio-oil at 520°C under the argon atmosphere (Saleh Khodaparasti et al., 2022). Digested food waste was also found to be a desirable additive in the treatment of sewage sludge by pyrolysis, which resulted in the improvement of the properties of biochar. Meanwhile, the pollution hazards of heavy metals were also mitigated (Wang et al., 2022b).

Pyrolysis of sewage sludge is facing some problems, such as raw material drying treatment and pollution component control, and in fact, it is not an ideal raw material for pyrolysis technology. It is a simple and feasible way to solve the previously mentioned problems by co-pyrolysis of sewage sludge with other ideal biomass materials. Through co-pyrolysis, in addition to achieving the yield/quality of pyrolysis products, it is more important to achieve the co-treatment of sewage sludge and other biomass/wastes, thus providing the economic, environmental, and social benefits of the whole process. Industrializing/commercializing co-pyrolysis of sewage sludge and other biomass or wastes still has some important issues to be solved, such as the lack of an in-depth and clear co-pyrolysis mechanism, the lack of unified kinetic analysis method, and the aggravation of the heterogeneity of raw material composition (Ma et al., 2022).

8.5 CONTROL EFFECTS OF HEAVY METALS

As mentioned previously, most heavy metals in sewage or wastewater would migrate to sewage sludge through adsorption or precipitation during sewage or wastewater treatment. Therefore, in the disposal of sewage sludge through pyrolysis, how the heavy metals in sewage sludge migrate and transform has become an important scientific issue. Because, whether the pollution effect of heavy metals can be effectively reduced during sewage sludge pyrolysis directly determines the cleanness and sustainability of the sewage sludge pyrolysis and even affects the environmental acceptability of pyrolysis products (Chanaka Udayanga et al., 2018). Considering that the temperature involved in sewage sludge pyrolysis is lower than the boiling point of most heavy metals, heavy metals are hard to volatilize, and the migration of heavy metals into the gas-phase products is less. Heavy metals are mainly concentrated in the solid-phase product (biochar). Meanwhile, the liquid product (bio-oil) still contains a little part of heavy metals. However, it is worth noting that although heavy metals

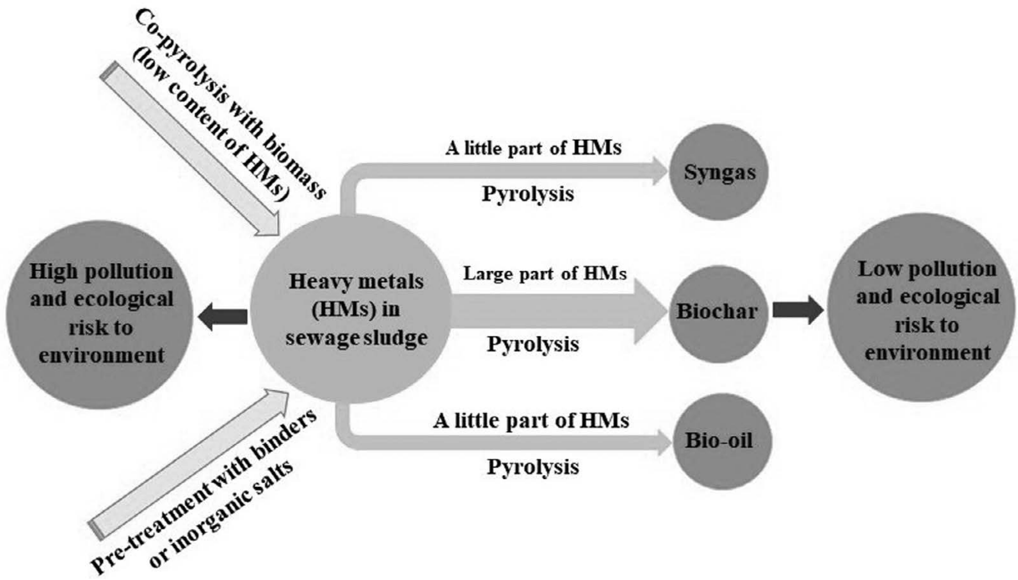

FIGURE 8.8 Control effects of heavy metals during the pyrolysis of sewage sludge.

are enriched in biochar, their leaching capacity is greatly reduced compared with that in raw sewage sludge (Figure 8.8) (Devi and Saroha, 2014; He et al., 2010; Hossain et al., 2009).

Changes in the characteristics of heavy metals in sewage sludge caused by pyrolysis treatment are determined by various factors, including reactor type/configuration, operation parameters, and additives (Galey et al., 2022; Li et al., 2022a). Hg with low thermal stability hardly remains in biochar. Cd volatilizes at temperatures above 650°C, while the volatilization of Pb and Cr occurs above 850°C and 700°C, respectively. As seems to easily volatilize at temperatures lower than 500°C. On the whole, the volatilization of heavy metals will be enhanced with the rise in pyrolysis temperature to some extent (Han et al., 2017; Hossain et al., 2011). It is generally believed that the decrease of heavy metal leachability in biochar products is attributed to three aspects: (i) the increase of pH value (acid neutralization capacity) of biochar; (ii) biochar has a more developed pore structure; and (iii) active/unstable heavy metals are transformed into stable heavy metals (He et al., 2010).

Among many operation parameters, the influence of reaction temperature is perhaps the greatest, and it has received the most attention. However, no linear correction between the changes in heavy metal chemical speciation and reaction temperature is observed. The changes in heavy metals' characteristics are unique for each metal and sewage sludge characteristic, indicating the complexity of heavy metal reactions during the treatment of sewage sludge by pyrolysis (Devi and Saroha, 2014; Hossain et al., 2011; Shao et al., 2015). At present, the influences of heating rate, reaction time, and the properties of sewage sludge (moisture content and particle size) on the changes in the heavy metals' characteristics in the disposal of sewage sludge by pyrolysis need to be paid more attention to. Existing reports confirm that the particle size of sewage sludge will influence the activity of heavy metals in biochar products, and a longer reaction time and lower heating rate may also enhance the loss of heavy metals by evaporation (Han et al., 2017; Jin et al., 2014).

Some researchers reduced the total amount and activity of heavy metals in pyrolysis products by adding other biomass during sewage sludge pyrolysis, which contains low amounts of heavy metals (Jin et al., 2017; Zhao et al., 2017). Besides the dilution effect, the co-pyrolysis process further improves the structure of biochar, resulting in better adsorption and fixation of heavy metals, which cannot be ignored (Chen et al., 2015; Wu et al., 2010). Pre-treatment of sewage sludge with different binders is also a feasible method to reduce the availability of heavy metals in pyrolysis products

(Tian and Liu, 2020). $FeSO_4$, cement, lime, hydroxyapatite, and ladle slag were applied for the pre-treatment of sewage sludge, and it was found that the percentages of Cr, Pb, Ni, Zn, and Cu present in stable chemical speciation in pyrolysis product were significantly improved (Oh et al., 2013). Some inorganic salts were applied in the pre-treatment of sewage sludge before pyrolysis; it was found that the presence of excessive chlorine reduced the retention of Cu, Mn, Zn, and Pb in bio-char. The volatilization of heavy metals was moderately promoted by sulfate. The use of phosphate promoted the residual fraction of heavy metals (Tian and Liu, 2020).

8.6 FORMATION AND TRANSFORMATION OF PAHS

Part of the PAHs contained in sewage or wastewater will be transferred to sewage sludge. The total content of eight EU-priority PAHs (Table 8.1) in the three kinds of sewage sludge in Turkey reached 690–17722 ng/g (Nas et al., 2020). In an investigation of organic pollutants in sewage treatment plant sludge in EU countries, it was found that the content of eight EU-priority PAHs reached 102.7–8383.7 ng/g (Gawlik et al., 2012). The content of 16 China/US-priority PAHs (Table 8.2) in Tunisian sewage sludge was between 96 and 7718 ng/g (Khadhar et al., 2010). In the light of the investigation of the sewage sludge of ten textile dyeing wastewater treatment plants in Guangzhou (China), it was found that the content of 16 China/US-priority PAHs was as high as 1643–16714 ng/g (Ning et al., 2014). PAHs may be produced during the processing and transformation of any organic matter, and the understanding of their formation and emission control is crucial (Purcaro et al., 2013). In the disposal of sewage sludge, especially for the thermochemical conversion process, the decomposition of organic matter in sewage sludge will also form PAHs. Therefore, the formation and pollution control of PAHs has also been deeply studied in the field of sewage sludge pyrolysis.

Sludges from food processing wastewater treatment were pyrolyzed and the total content of 21 target PAHs (including cyclopenta[c,d]pyrene, benz[e]pyrene, perylene, benzo[b]chrysene, coronene, and 16 China/US-priority PAHs) in the liquid products (bio-oils) ranged from 298 to 336 mg/L (Tsai et al., 2009). In another research, the content of 16 China/US-priority PAHs in bio-oil from the treatment of seven sewage sludges by pyrolysis was distributed in the range of 13.72–48.9 mg/kg (Hu et al., 2014). After pyrolysis, PAHs are mainly distributed in liquid-phase products (bio-oil), followed by biochar, and a few in gas-phase products (Dai et al., 2014b; Hu et al., 2020). For the disposal of sewage sludge by pyrolysis, the formation of PAHs was mainly affected by the reaction temperature, followed by gas residence time, carrier gas flow rate, and sample loading (Dai et al., 2014a; Ko et al., 2018). In general, the formation of PAHs will be enhanced with the rise of reaction temperature. The total amount of 16 China/US-priority PAHs in biochar increased from 0.21 to 1.0 μg/g

TABLE 8.2
A List of Priority PAHs Commonly Adopted in the World

Countries	PAH Compounds				
European Union (EU)	**Naphthalene**	**Anthracene**	**Fluoranthene**	**Benzo[b] fluoranthene**	**Benzo[k] fluoranthene**
	Benzo[a]pyrene	Benzo[g,h,i] perylene	Indeno[1,2,3-cd] pyrene	-	-
China, United States (US)	Naphthalene	Acenaphthylene	Acenaphthene	Fluorene	Phenanthrene
	Anthracene	Fluoranthene	Pyrene	Benzo[a]anthracene	Chrysene
	Benzo[b] fluoranthene	Benzo[k] fluoranthene	Benzo[a]pyrene	Indeno[1,2,3-cd] pyrene	Dibenz[a,h] anthracene
	Benzo[ghi] perylene	-	-	-	-

with the rise of the reaction temperature in the range of 400–800°C, while the content of 16 China/US-priority PAHs in bio-oil was improved from 1.6 to 19 μg/mL (Chiang et al., 2014).

Some recent studies on the total amount, free dissolved content, and persistence of PAHs in soils improved by sewage sludge or its derived biochar from pyrolysis indicate that compared with sewage sludge, the application of sewage sludge-derived biochar will decline the concentration and persistence of free dissolved PAHs in soil. In other words, the application of sewage sludge-derived biochar for soil improvement has lower environmental risk (Stefaniuk et al., 2018; Tomczyk et al., 2020a; Tomczyk et al., 2020b). After the soil was improved with sewage sludge-derived biochar, the content of accumulated PAHs in cucumber fruit planted in the soil was less than that of cucumber fruit planted in the soil improved with sewage sludge feedstock (Waqas et al., 2014). When the disposal of sewage sludge through pyrolysis was performed at 850°C, KCl was certified as an effective catalyst, which repressed the production of PAHs in bio-oil, reducing from 15.25 μg/g (no catalyst) to 4.23 μg/g (20% loading of KCl) (Hu et al., 2019). Calcium carbonate also showed a similar catalytic effect, which not only increased the concentration of carbon monoxide in the gaseous product but also declined the amounts of PAHs in pyrolysis products of sewage sludge (Kwon et al., 2018).

8.7 CONCLUSIONS AND PERSPECTIVES

On the one hand, pyrolysis can achieve effective treatment of sewage sludge; on the other hand, it can achieve the purpose of resource utilization. Pyrolysis bio-oil can be applied as liquid fuel after further treatment and also be used in the chemical synthesis field. Pyrolysis biochar has been successfully developed into functional carbon materials, which have been used in many fields. In addition, the pyrolysis process can effectively control the pollution of heavy metals and PAHs. Of course, in the future, more attention should be paid to the pilot scale and application promotion research, as well as the formation and control of other pollutants during sewage sludge pyrolysis, especially organic pollutants. The pyrolysis process of sewage sludge should be further optimized or improved from the perspective of circular economy and carbon neutralization. In the pyrolysis process of sewage sludge, introducing other biomass or organic wastes for co-treatment may be the most feasible direction for sewage sludge treatment/disposal at present, to achieve efficient resource recovery and effective control of pollution components, and to maintain acceptable treatment costs.

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9 Combustion of Sewage Sludge

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9.1 INTRODUCTION

Sludge refers to the mixed byproducts of solid and liquid produced by municipal or industrial wastewater treatment (Morello et al., 2022). The amount of sludge produced by treating 10,000 tons of sewage is generally about 10–20 tons (based on 98% moisture content). According to previous estimates, the volume of sewage generated annually worldwide was 360–380 km^3 with dry solid sludge contributing close to 45 million tons (MT) in 2020 (Giacomo and Romano, 2022). Among them, a significant portion of the world's sewage sludge was produced in East Asia, Europe, and North America (Shaddel et al., 2019). In the case of China, the annual production of wet sludge is expected to reach 60 MT in 2020, including over 13 MT of dry sludge (Havukainen et al., 2017). In the United States, annual dry sludge production has reached almost 8 MT (Li et al., 2016), while the amount of dry sludge produced in the European Union per year was more than 11 MT (Eurostat 2019). In general, organic matter makes up about 50–70% of solid waste sewage sludge, which contains 18–52% of volatile substances such as protein, lipids, and carbohydrates (Kacprzak et al., 2017). Due to the huge volumes produced and the unique chemical composition, sewage sludge has been connected to a variety of global environmental problems (Ghorbani et al., 2022).

Sludge is easy to decay and smell. Its particles are fine, with a small specific gravity (1.02–1.006). The moisture content of sludge is high (≥98%) and it is hard to dehydrate (Ren et al., 2015). According to its production source, the sludge can be divided into water supply plant sludge, domestic sewage sludge, industrial sludge, and hydrophobic sludge from an urban water body. Hydrophobic sludge is produced from the dredging process of natural or artificial water bodies such as rivers, lakes, and ponds. Depending upon the different treatment processes, the sludge includes the primary sludge from the primary tank, the active sludge from the secondary tank treated by aerobic microorganism, the humus sludge from the secondary tank treated by a biofilm membrane, and the chemical sludge, which is derived from the primary or tertiary treatment by chemical processes such as coagulation and chemical precipitation.

9.2 SLUDGE COMPOSITION

Sludge is mainly composed of various microorganisms and organic and inorganic particles, with some heavy metals, organic pollutants, pathogenic microorganisms, and parasitic eggs (He et al., 2009). The specific composition and content of commercial sludge are shown in Table 9.1 and include the following: moisture: water content of about 90% or higher (Mowla et al., 2013); volatile substances and ash: the former is organic impurities, the latter is inorganic impurities (Li et al., 2022); pathogens: such as bacteria, viruses, and parasitic eggs. A large number of pathogens exist in living sewage, hospital sewage, food industry wastewater, and tannery industry wastewater sludge; and toxic substances, including cyanide, mercury, chromium, or other toxic organic compounds which are difficult to decompose.

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TABLE 9.1
Typical Compositions of Sludge

Compositions	Content
Dry matter (DM) (g/L)	7–30
Volatile substances (%, DM)	50–77
pH	6–7
C (%, DM)	49–53
H (%, DM)	6–7.7
O (%, DM)	33–25.5
N (%, DM)	4.5–7.5
S (%, DM)	1–2.1
P (mg/kg, DM)	13,000–20,966
K (mg/kg, DM)	3200–5104
Na (mg/kg, DM)	1100–2937
Al (mg/kg, DM)	18,571–23,655
Ca (mg/kg, DM)	22,800–32,656
Fe (mg/kg, DM)	19,346–20,200
Mg (mg/kg, DM)	4300–6041
Fat (%, DM)	8–18
Protein (%, DM)	18–36
Fibers (%, DM)	10–17
Calorific value (kWh/t DM)	3000–4800

Source: Manara and Zabaniotou (2012) and Syed-Hassan et al. (2017).

9.3 MECHANISM OF SLUDGE INCINERATION

9.3.1 Disposal Methods of Sludge

Sludge will cause secondary pollution to groundwater and soil if discharged or piled up without effective disposal treatment because it frequently contains heavy metals and many toxic chemicals. The main disposal methods of sewage sludge include landfill, dumping into seas, incineration, and application to soil (Yang et al., 2015). Generally, the selection of a sludge disposal method in each region is determined according to the geographical environment, economic level, technical capabilities, transportation, and other factors, including the improvement of public awareness. Basic processes of sludge treatment include thickening, conditioning, dewatering, stabilization, and drying. In China, landfill, land application, and incineration were three most frequently used methods (see Table 9.2).

9.3.1.1 The Landfill

There are some problems in landfill disposal, such as the location of a suitable site and the high cost of sludge transport and landfill site construction (Zhan et al., 2014). The leakage of harmful components and waste gas emissions may cause secondary pollution to groundwater and air. In developed countries, where this method used to be more common, fewer sites are currently available for landfill.

9.3.1.2 Land Application

Land application is a promising disposal method and the sludge is applied to farmland, woodland, grassland, municipal green areas, and severely disturbed sites undergoing land rehabilitation and restoration. Sludge contains a large amount of organic matter and nutrients needed by plants, which

TABLE 9.2
Different Disposal Methods of Sludge

Disposal Methods	Advantage	Disadvantage	Reference
Landfill	Easy to implement, low cost, no dewatering required	Produces toxic landfill leachate and landfill gas, pollutes the environment, and covers a large area	Fang et al. (2016)
Land application	Low input, low energy consumption, resource recycling	Incomplete, with potential environmental risks	Singh and Agrawal (2010)
Incineration	The most thorough method, all organic matter is oxidized, killing all pathogenic bacteria and minimizing the volume	Despite high investment in treatment facilities and equipment maintenance costs, incineration process produces dioxins and other carcinogenic gases	Cammarota et al. (2019)
Building materials	Waste utilization and resource saving	Needs to be rendered harmless before use, increasing costs	Wang et al. (2005)

is a valuable biological resource. The recovery of nutrients including phosphorus and nitrogen from sludge demonstrates the potential for land application of sludge, but the high content of heavy metals in sewage sludge and strict standards may limit this use (Fang et al., 2016).

Sludge can also be used to rehabilitate severely disturbed land. Soil borrows pits for construction, intensively managed forest sites, and landfills generally diminished soil quality, which results in planting failures. The application of sludge can increase soil nutrients, improve soil physical structure, and promote the growth of surface plants. It is also important to note that the toxic organic matter contained in the sludge could be a potential threat to the groundwater (Singh & Agrawal, 2010).

9.3.1.3 Incineration

As far as disposal costs and the environmental risks are concerned, incineration is the safest disposal method for pollutants such as sludge. Dong et al. (2014) examined four different disposal methods, including composting, co-combustion in the power plant, thermal drying-incineration, and cement manufacturing with life cycle assessments, and found that incineration was the most safe and efficient disposal method. In addition, there are several other disposal methods of sludge, such as refining and inclusion into building materials which are not widely used.

9.3.2 Mechanism of Sludge Incineration

Dry based sludge is a material of great energy value because organic matter is the main chemical composition of sludge. That means sludge can be disposed of with thermochemical treatment methods, including pyrolysis, gasification, and incineration (Li et al., 2023). Sludge incineration refers to the use of an incinerator to heat and dry dehydrated sludge, and then using high temperatures to oxidize organic matter in the sludge, so that the sludge is converted to ash. The fundamental mechanisms of sewage sludge transformation in an incinerator are depicted in Figure 9.1. The whole process of sludge combustion can be usually divided into five stages: (1) dewatering, (2) release of organic matter, (3) fixed carbon combustion, (4) mineral decomposition, and (5) burning to ash.

Depending on the configuration of the incinerator, the reaction conditions, and the characteristics of sludge, devolatilization happens either parallel or sequential to the drying phase during sludge combustion even at a very low temperature (150°C). As the water vapor and CO_2 are released from inside of the sludge matrix, they may react with the dehydrated/devolatilized outside layers of the sludge (char layers) before oxygen in the gas phase diffuses to the same hot surface layers. With the increases of temperature and the heating rate, the proportion of gaseous substances increases, and

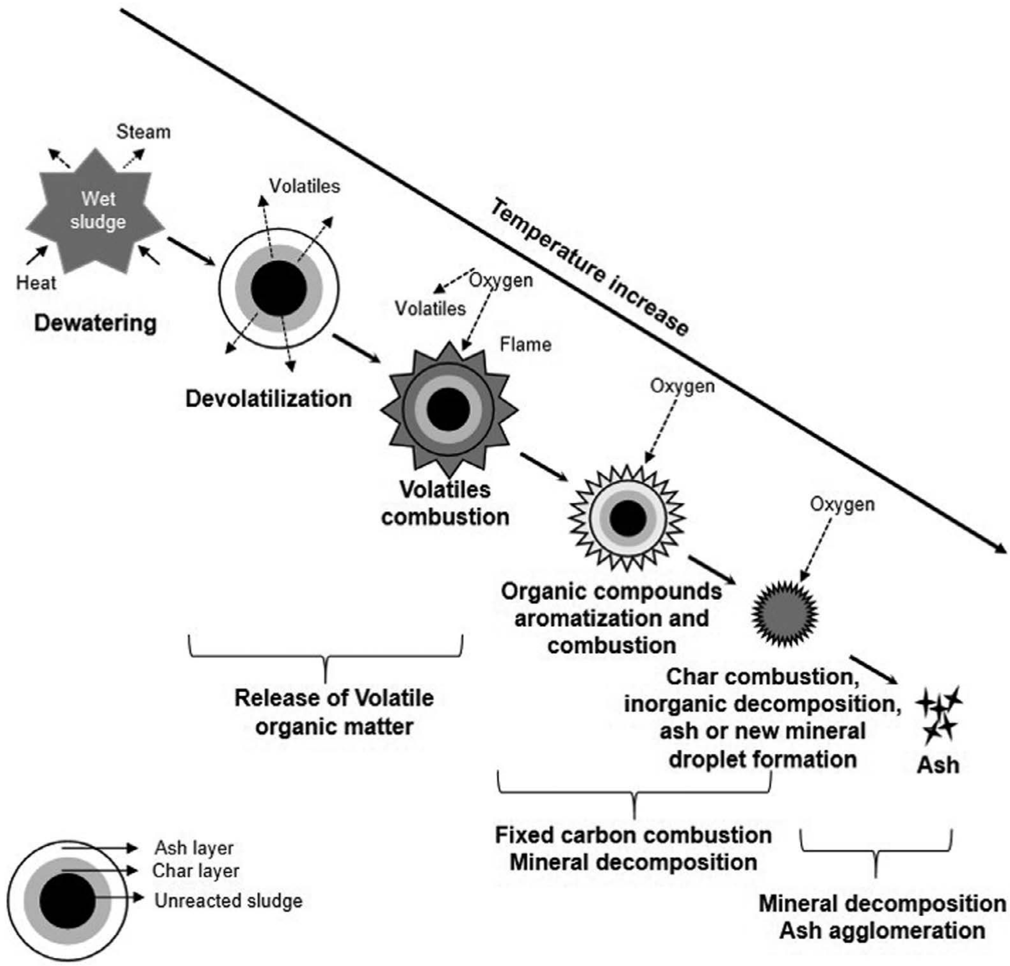

FIGURE 9.1 Fundamental mechanisms of sewage sludge combustion.

the proportions of char and condensable liquid decrease, resulting in the increased role of volatiles in combustion. In a typical sludge combustion process, oxygen is in excess, and the gases formed from the pyrolysis of sludge are rapidly ignited to form other gases, including CO, CO_2, H_2O, NO, NO_2, SO_2, and SO_3 in the combustion chamber. The combustion of volatiles can be characterized by the rapid consumption of oxygen. The burning of volatiles and light gases around the sludge surface increases the surface temperature rapidly, and when the temperature is high enough, it causes melting of the ash layers and disintegration of some inorganic substances. Some small molten droplets are formed at around 900–1300°C and cover the particle surface. The remaining sludge char then proceeds with reactions with oxygen in the air until it burns out (Syed-Hassan et al., 2017).

9.3.3 Impact Factors of Sludge Incineration

As shown in Table 9.3, during the process of heating, many factors will affect the result of incineration.

9.3.3.1 The Sludge Moisture

Sludge is a complex substance with high moisture content. General sludge contains 70–75% free water, 20–25% flocculent water, and 1% capillary water and binding water. The high moisture sludge directly enters the incinerator, which will cause some adverse effects, such as the decrease

TABLE 9.3
Main Factors Affect Sludge Incineration

Factors	Advantage	Disadvantage	Optimal Parameter
Moisture	Low moisture content of sludge is important to control the efficiency and the cost of sludge incineration equipment and treatment.	High moisture sludge entering the incinerator will cause some adverse effects, such as the decrease of incineration temperature, the delay of the ignition process, and temperature fluctuations in the furnace.	The ratio of moisture content to the volatile content should be reduced to less than 3.5 to allow spontaneous combustion and to save fuel.
Temperature	High temperature conditions favor the loss of moisture and volatiles and the dechlorination and cyanide decomposition (800–950°C). Thus, increasing the incineration temperature is beneficial to the decomposition and destruction of organic toxins in sludge and will also inhibit the generation of black smoke.	High temperature conditions will cause the sludge to break down in the initial stage of incineration and, thus, increase the loss of fly ash. In addition, the high incineration temperature not only increases fuel consumption but also increases the amount of nitrogen oxide and heavy metals in emissions, which will cause secondary pollution.	The sludge incineration temperature is generally recommended to be controlled at 800–1000°C.
Residence time	The longer the residence time, the better the treatment effect.	The longer the residence time, the higher fuel consumption.	The characteristics of residence time vary with different sludge modes.
Turbulence	The greater the turbulence, the better the mixing of combustible with air, and the more complete combustion of sludge.	The greater the turbulence, the higher energy consumption and oxygen demands.	The characteristics of Turbulence vary with different sludge modes.
Air volume	Insufficient ventilation will lead to incomplete combustion of organic matter, which will affect efficiency of sludge combustion.	Excessive ventilation supply will reduce the incineration temperature and increase the energy consumption.	Generally, 50–100% excess air is appropriate.

of incineration temperature, the delay of the ignition process, and temperature fluctuations in the furnace. Hence, reducing the moisture content of sludge is important to control the cost of sludge incineration equipment and treatment. The ratio of moisture content to the volatile content should be reduced to less than 3.5 to allow spontaneous combustion and to save fuel.

9.3.3.2 The Burning Temperature

Increasing the incineration temperature is beneficial to the decomposition and destruction of organic toxins in sludge and will also inhibit the generation of black smoke. However, under high temperature conditions, the loss of moisture and volatiles is fast due to the rapid heating, and this will cause the sludge to break down in the initial stage of incineration and thus increase the loss of fly ash. In addition, the high incineration temperature not only increases fuel consumption but also increases the amount of nitrogen oxide and heavy metals in emissions, which will cause secondary pollution. Therefore, the appropriate temperature for incineration should be determined by test at a certain residence time.

9.3.3.3 Residence Time

The residence time is proportional to the solid particle size, and the heating time is approximately proportional to the square of the particle size. Therefore, it is important to consider the solid particle size when determining the residence time of waste in the combustion chamber. The characteristics

TABLE 9.4
Summary of the Advantages and Disadvantages of the Three Most Common Types of Sludge Incineration

Instrument Types	Advantages	Disadvantages
Multiple hearth incinerator	Small occupation space Low fly ash production	Bad odors Gaseous emission High energy consumption
Fluidized bed incinerator	High combustion efficiency Good applicability Intermittent operation available Low maintenance cost	Auxiliary fuel needed for high water content of sludge Gaseous emission
Rotary kiln incinerator	Easy manipulation and maintenance	Incomplete combustion Difficult to control temperature High demand for heat value

of residence time vary with different sludge modes. In the case of intermittent sludging, the period and the amount of sludging are two important factors that determine the residence time of sludge in the furnace. The longer the residence time, the better the treatment effect. When continuous sludge input is adopted, the longer the residence time, and the incineration is less effective.

9.3.3.4 The Air Volume

Sludge incineration must be supported by oxygen, which is usually supplied by air. Generally, 50–100% excess air is appropriate.

The factors affecting sludge incineration also include volatile content and the mud-gas mixing ratio. The higher the content of volatile matter in sludge, the lower the water content, and the easier it is to maintain spontaneous combustion.

9.4 INSTRUMENTS OF SLUDGE INCINERATION

Multi-chamber incinerator (MIF) and fluidized bed (FBC) are the most widely used furnaces, although other furnaces, such as the rotary furnace, cyclone furnace, and various forms of melting furnaces, are also used. The main advantages and disadvantages of three methods are listed in Table 9.4.

9.4.1 Multiple Hearth Incinerator

The multiple hearth incinerator, also known as the vertical multi-section incinerator, is a vertical cylindrical fire-resistant piece of equipment that is steel lined. There are many horizontal refractory furnaces formed in the interior and a series of horizontal adiabatic furnaces arranged by layer from top to bottom. A multi-chamber incinerator may contain 4–14 chambers with a rotating central shaft from the bottom to the top, as shown in Figure 9.2. Each layer of the furnace has coaxial rotary tooth harrows. Generally, the upper and lower layers of the furnace have four tooth harrows, and the middle layer of the furnace has two tooth harrows. The dehydrated mud cake enters the furnace from the outside of the top furnace and moves to the center and enters the lower layer through the hole in the center, while the sludge enters the lower layer and moves to the outside and enters the next layer through the hole on the exterior. During this process, the sludge moves from top to bottom in a spiral route. The cast iron shaft is equipped with a sleeve, and the air is pumped into the outer sleeve from the lower portion of the shaft. The shaft is cooled, while the air is preheated. Part or all the preheated air returns from the upper part to the inner sleeve and enters the lower furnace and then serves as the combustion air as it moves upward to incinerate the sludge.

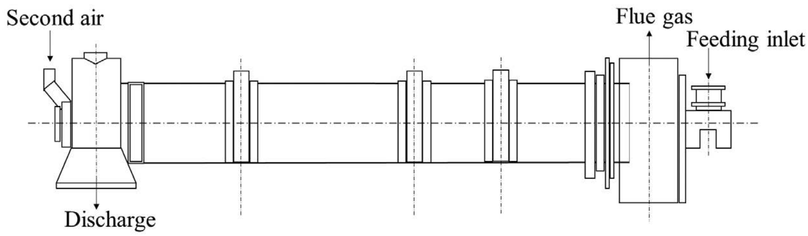

FIGURE 9.2 Diagram of rotary kiln.

For the whole sludge incineration process, a multi-bore furnace can be divided into three parts. First, the top layer is the sludge drying zone, and the temperature in this layer is about 425–760°C, which can reduce the moisture content of sludge to less than 40%. Second, the central layer is the sludge incineration zone, where the temperature reaches 760–925°C. The upper part of this layer is for the volatile gases and part of the solid combustion area, the lower part is the fixed carbon combustion area. Third, the bottom layers of the multi-chamber furnace are the slow cooling zone, which mainly plays the role of cooling and preheating air, and the temperature in this layer is 260–350°C.

The multi-chamber incinerator has many characteristics. The diameter of the furnace body can reach 7 m, which provides a large heating surface for heat exchange. In this type of incinerator, the number of layers can range from 4 to 14. In continuous operation, fuel consumption is less except for the first 1–2 days. Some issues with the multi-chamber incinerator are that there are more mechanical parts which sometimes need more repair and maintenance, the energy consumption is relatively high, and thermal efficiency is low. To reduce the flue gas pollution from combustion, it is necessary to add secondary combustion equipment.

9.4.2 Fluidized Bed Incinerator

Fixed fluidized bed technology is widely used worldwide (Yang et al., 2015). The fluidized bed incinerator is lined with refractory material, and the combustion chamber is formed by the cloth plate. The combustion chamber is divided into two zones, the upper dilute phase zone (suspension section) and the lower dense phase zone. With this technology, there is a large amount of inert bed material (such as coal ash or sand) in the dense phase bed of the fluidized bed, which has a large heat capacity. This creates conditions for the evaporation of sludge moisture and the pyrolysis and combustion of volatile matter.

The air is sent to the dense phase area by the air distribution device, which causes the bed to be in a good fluidization state. The heat transfer condition in the bed is stable and the temperature in the bed is uniform (800–900°C), which is conducive to the decomposition and burning of organic matter. The flue gas generated after incineration enters the dilute phase area of the fluidized bed with a small quantity of solid particles and unburned organic matter. The high-speed air flow sent by the secondary air forms a rotary cut circle in the center of the furnace, which makes the disturbance strong and the mixture sufficient, thus the unburned components can continue to be combusted.

Depending on the fluidizing wind speed and material movement status in the furnace, fluidized bed incinerators can be divided into boiling fluidized beds and circulating fluidized rings. High pressure air (20–30 kPa) is sprayed in the refractory grid from the blast outlet at the bottom of the furnace, so that the silicon sand layer on the refractory grid is mixed and suspended with the added sludge. The dry and broken sludge is added to the furnace from the bottom of the furnace and is violently mixed with hot silica sand and burned. The temperature of the fluidized bed is controlled between 725°C and 950°C.

The residence time of sludge in the circulating fluidized bed and boiling fluidized bed incinerator is seconds and tens of seconds, respectively. Incineration ash and gas are discharged from the top of the furnace together. After the separation of gas and solid by the cyclone separator, hot gas is used to preheat air and incineration ash is used to preheat the dry sludge to recover heat. Very little of the silica sand in the fluidized bed will be lost along with the gas. About 5% of the silica sand in the fluidized bed should be replenished every 300 h to ensure that there is enough silica sand in the fluidized bed.

Sludge incineration in fluidized bed incinerators is completed in two zones. The first zone is the silica sand fluidization zone, where the evaporation of water and the decomposition of organic matter in the sludge occur almost simultaneously. The second zone is the free space above the silica sand layer, which acts as a post-combustion chamber, where carbon and combustible gases from the sludge continue to burn.

The incineration temperature of the sludge fluidized bed incinerator is generally 660–830°C, in which the sludge odor can be effectively eliminated. The temperature can be adjusted and controlled by the auxiliary burner and hot air located in the furnace bed.

Compared with multi-chamber incinerators, fluidized bed incinerators have many advantages. First, the fluidized bed incinerator has higher incineration efficiency. In this type of incinerator, the furnace gas and solid mix strongly, and sludge evaporation and combustion can be completed in an instant due to stable combustion and the uniform temperature field in the furnace coupled with the use of secondary air to increase the disturbance in the furnace. In addition, the incomplete combustion of material in the suspension section continues, eventually resulting in full combustion. This method works well with different kinds of sludge because there is a large amount of high temperature inert bed material in the fluidized bed, the heat capacity of the bed is large, and thus, it can provide a large amount of heat needed for evaporation, pyrolysis, and combustion.

The fluidized bed incinerator has a good environmental performance, and it integrates drying and incineration, which can deodorize the sludge. Also, if low temperature combustion and graded combustion are adopted, the amount of NOx produced in the incineration process is very small. At the same time, appropriate additives can be added to the bed material to eliminate and reduce the emission of harmful incineration products. For example, adding limestone to the bed material can neutralize the SOx and HCl produced in the incineration process, so that it can meet the requirements of environmental regulations. Moreover, the emission of heavy metals is less since the incineration temperature of the fluidized bed incinerator is lower than that of the multi-chamber incinerator.

Finally, the fluidized bed incinerator has a high combustion intensity and large waste disposal capacity. There is a strong heat transfer in the furnace and the integration of a waste heat recovery device and incinerator makes the whole system a compact structure with a small footprint.

However, there are some problems when using the fluidized bed incinerator to treat salt-contained sludge. It is easy to form low melting point co-crystals (melting point between 635 and 815°C) in the bed when the sludge contains alkali metal salt or alkaline earth metal salt. If the melting salt accumulates in the bed, it will lead to coking, slagging, and even fluidization failure. If the molten salt is carried out by the flue gas, it will adhere to the furnace wall and solidify into fine particles that cannot be easily removed by scrubbers. The solution to this problem is to add suitable additives to the bed, which can encase the alkali metal salts, forming a high melting point substance between 1065°C and 1290°C.

9.4.3 Rotary Kiln Incinerator

The rotary-kiln incinerator is used to deal with solid, liquid, and gas combustible waste and waste of complex components, including asphalt, organic residue distillation kettle, tar slag, sludge, waste solvent, waste rubber, halogenated aromatics, polymers, and especially waste containing PCB (printed circuit board) (Jiang et al., 2019). Most hazardous waste disposal plants in the United States use this type of furnace. The rotary-kiln incinerator can treat a wide range of waste simultaneously and operate in a stable manner. But its complex management and high maintenance costs (general refractory lining replaced every 2 years) limit its use to some degree.

The rotary kiln adopts a horizontal cylindrical shape, and the shell is generally made of a steel sheet (Schacht, 2004). Lined with refractory materials (brick structure or high temperature refractory concrete precast), the inner wall of the kiln is smooth and there is also an arranged internal component structure. One end of the kiln body is fed by a spiral feeder or other means, and the other end exhausts the burned ash out of the furnace. The sludge in the rotary kiln can be in reverse contact with the high temperature air flow and can also flow in the same direction with the air flow.

In the reverse flow, the high temperature air stream can preheat the incoming sludge, and the heat is fully used, and the transfer efficiency is high. Exhaust often carries toxic and harmful gases volatilized from sludge, which must be treated by secondary incineration. For rotary kilns that flow in tandem, a burner is generally set at the rear of the kiln for secondary incineration.

The furnace lining of the sludge rotary kiln is concrete and brick, and the concrete part is set with an internal component structure. The combustion chamber configured in the rotary kiln is made of a structure with a roller, which can be moved and easily maintained. The temperature of the rotary kiln incinerator ranges from 810°C to 1650°C. The temperature is regulated by the burner fuel at the end of the kiln. Liquid fuel or gas fuel is usually used, and pulverized coal or waste itself can be used as fuel.

The outer layer of the furnace is a metal cylinder, and the inner layer is generally lined with refractory materials. The rotary kiln incinerator is usually placed at a slight angle and equipped with a rear burner. Generally, the length-to-diameter ratio of the furnace is 2–10 and the rotational speed is 1–5R/min, and the installation inclination is 1–3°. The rotation of the rotary kiln mixes the waste with gas, which is burned and volatilized. The sludge is in a gaseous and residual state, and the transformed gas is completely burned through the high temperature of the rear burner (1100–1370°C). The average residence time of gas in the rear burner is 1.0–3.0 s, and the air excess coefficient is 1.2–2.0. The average heat capacity of the rotary kiln incinerator is about 63×106 kJ/h. The incineration temperature in the furnace (650–1260°C) depends on two aspects: the nature of the sludge and the slagging method (wet or dry). For the sludge containing halogenated organic matter, the incineration temperature should be above 850°C, and for the sludge containing cyanide, the incineration temperature should be above 900°C.

The incineration temperature in the rotary kiln incinerator is controlled by auxiliary fuel burners. The harmful gases produced by incineration, such as dioxins and furans, cannot be effectively removed in the furnace chamber of rotary kiln. To ensure the complete combustion of harmful substances in the flue gas, a burnout chamber is usually set. When the residence time of flue gas in the burnout chamber is greater than 2 s and the temperature is higher than 1100°C, the toxic substances can be eliminated. The flue gas from the burnout chamber is recycled into the waste heat boiler to generate steam or electricity.

9.4.4 Grate Incinerator

Grate incinerators can be divided into ladder reciprocating, a chain type, a grid moving type, a multi-section rolling type, and a fan grate with different grate structures. The step reciprocating grate incinerator is usually used in sludge incineration. The structure of the ladder reciprocating grate incinerator is shown in Figure 9.3. Generally, the incinerator grate is composed of 9–13 pieces, and the fixed and movable grates are placed alternately. The first few sections include the dry preheating grates, after are the burning grates, and the bottom pieces are the slag grates. The reciprocating motion of the movable grate is driven by a hydraulic cylinder or by mechanical means. The reciprocating frequency can be adjusted to a wide range according to the production capacity.

The sewage sludge is incinerated by the grate furnace, and the fixed section and the movable section are interactively configured. The hydraulic device makes the movable section move back and forth, stirring the sludge layer while transporting the sludge. The drying zone is longer than the

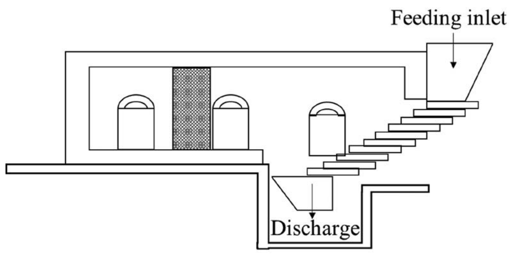

FIGURE 9.3 Diagram of ladder reciprocating grate incinerator.

burning zone. Sludge with moisture content below 50% can spontaneously ignite at high temperature. The upper part of the furnace is equipped with a waste heat boiler, and the recovered steam can be used for sludge drying. Dehydrated sludge cake (moisture content, 75–80%) is transported to the incineration grate furnace after drying (moisture content, 40%~50%) and, finally, becomes incineration ash.

9.4.5 Electric Heating Infrared Incinerator

The electric heating infrared incinerator is shown in Figure 9.4. Its body is a horizontal adiabatic furnace, and the sludge conveyor belt is arranged along the length of the furnace. The electric heating infrared incinerator is generally composed of a series of prefabricated parts, which can meet the requirements for different incineration lengths. The dehydrated sludge is sent to the incinerator through one end of the conveyor belt, and a rolling mechanism is arranged at the inlet end to make the sludge cover the conveyor belt with a thickness of nearly 12.5 mm. In the incinerator, the sludge is dried and then incinerated in the infrared heating section. The incineration ash is discharged into the ash hopper at the other end, and the air enters the incinerator from the back end after preheating the incineration ash layer above the ash hopper. The sludge goes in the opposite direction. Exhaust gas is discharged from the feed end of the sludge. The air excess coefficient of electric heating infrared incinerator is 20–70%. The purification treatment of the exhaust gas from the electric heating infrared incinerator can be carried out by wet purifiers such as Venturi scrubbers and/or absorption towers.

The electric heating infrared incinerator requires low investment and is suitable for small sludge incineration systems. The disadvantage is its high energy consumption, and the life of the metal conveyor belt is short and has to be replaced every 3–5 years.

9.4.6 Melting Incinerator

The operating temperature of many furnace types is lower than the melting point of ash in sludge, which contains high concentrations of heavy metals that pollute the environment. Thus, the objective of the sludge melting treatment is to control the discharge of harmful heavy metals contained in

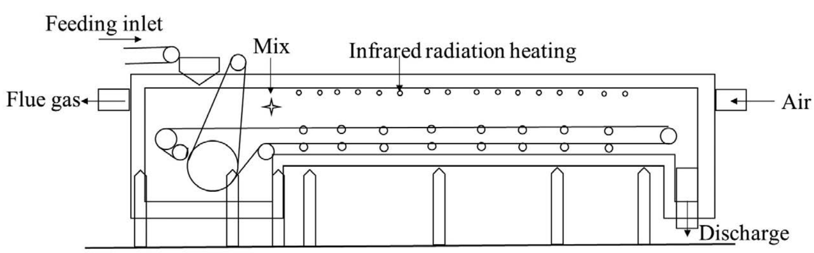

FIGURE 9.4 Diagram of electric heating infrared incinerator.

sewage sludge. If the dry sludge is exposed to temperatures beyond the ash melting point (generally 1300–1500°C), the sludge ash becomes vitreous or crystalline substances. Heavy metals will then exist in a stable state in the SiO_2, which is vitreous and will not dissolve and damage the environment. Adding lime and silica to the sludge can reduce the melting temperature, making the operation more efficient and reducing furnace loss.

The sewage sludge melting equipment system consists of four processes. First, the drying process reduces the water content of the dehydrated sludge cake from 70–80% to 10–20%. Second, in the adjustment process, granulation, crushing, thermal decomposition, and carbonization are carried out according to the applicable mode of each furnace. Third, the combustion and melting of organic matter lead to the formation of ash, which is melted into slag. Finally, the water cooling and slag granulation are used to obtain slowly cooled slag. The slag of the crystallization furnace is granulated to reach resource utilization.

9.4.7 Cyclone Incinerator

The cyclone incinerator is a single movable furnace, where the tooth rake is fixed. Air is brought into the burner. The incinerator is a domed cylindrical structure of linearly arranged refractory materials that heats the air with immediate fuel replenishment, creating a powerful vortex that provides good mixing of sludge and air. Air and smoke rise vertically in a spiral stream following the exhaust gas from the center of the dome. The sludge is fed by a spiral feeder, deposited at the periphery of the rotary furnace, and raked out toward the center of the furnace as fly ash. A screw pump is used to supply sludge. The temperature in the incinerator is 815–870°C. These incinerators are relatively small and can be started up within 1 h at operating temperatures.

The horizontal incinerator is a variant of the cyclone incinerator. Sludge is pumped into the incinerator along the tangential direction from the furnace wall, and air is brought into the tangential part of the burner to form a cyclone effect. Fly ash is discharged through flue gas. This kind of incinerator has no furnace, only a furnace shell and refractory material. The residence time of sludge in this furnace does not exceed 10 s. Combustion products are discharged from the vortex at 815°C to ensure complete combustion.

9.5 CO-COMBUSTION OF SLUDGE WITH OTHER BIOMASS/WASTE

In order to improve the alternative fuel's characteristics and to promote the treatment efficiency, the application of sludge blends with biomass residues or other organic waste as a feedstock in the thermochemical processes of sludge combustion is an interesting prospect. For example, sewage sludge often has some characteristics such as high moisture content, high ash content, low density, low viscosity, and a low heating value. Compared with sewage sludge, crop biomass contains less ash, a high carbohydrate content, and a low ash fusion temperature in general. Mixing the sewage sludge with crop residue in an appropriate proportion may compensate for both residues' weaknesses, and the new blend-material may have improved mechanical and physicochemical characteristics for combustion such as the reduction of total moisture content, an increase of calorific value, and the dilution of the sludge's undesirable species content. Currently, some research has attempted co-combustion with sludge-coal (Hong et al., 2013; Li et al., 2015; Stelmach & Wasielewski, 2008), municipal solid waste-sludge (Lin & Ma, 2012), and lignocellulosic biomass-sludge (Kijo-Kleczkowska et al., 2016; Lee & Bae, 2009).

Pinto et al. (2008) studied the thermochemical treatment of straw pellets and sewage sludge mixture, and they pointed out that the presence of straw pellets had a positive effect in sludge energy use. Other researchers tested the co-combustion of sludge with coal (García et al., 2013), wood pellets (Roy et al., 2011), pine sawdust (Zhu et al., 2015), and plastics (Akkache et al., 2016), and they concluded that co-combustion of sludge with other combustibles had better performance in terms of limiting the emission of NH_3 and H_2S, maximizing the heating value, and lowering the ash problems compared to the mono-combustion of sludge (Ninomiya et al., 2004; Roy et al., 2011). These findings were supported by the co-utilization study of Cartmell et al. (2006) in terms of energy input and policy analysis of sludge co-incineration. However, a life cycle assessment study indicated that sludge co-incineration brings a greater environmental burden to the coal-based energy production technology although it presents higher economic benefits. The life cycle assessment suggested that sludge co-combustion was unsuitable for mitigating sewage sludge pollution, and the reduction in the overall negative environmental effects of sludge co-combustion necessitates a decrease of sludge water content as well as an enhancement in net coal consumption efficiency, ash reuse rate, and dust removal efficiency (Hong et al., 2013). Based on these considerations, optimizing incineration conditions and the feedstock mixing ratio during the co-combustion of sludge with other biomass/waste ash has exhibited greater environmental benefits (Gu et al., 2022; Häggström et al., 2023; Ma et al., 2022; Xia et al., 2023).

9.6 POLLUTANTS CONTROL

Sewage sludge often carries pathogenic microorganisms and undesirable toxic pollutants, including heavy metals (Zn, Pb, Cu, Cr, Ni, Cd, Hg, and As with levels varying from <1 mg/kg to > 1g/kg), synthetic organic compounds (polychlorinated biphenyls [PCBs], polycyclic aromatic hydrocarbons [PAHs], dioxins, pesticides, linear-alkyl-sulfonates, nonylphenols, polybrominated fire retardants, etc.) (Manara & Zabaniotou, 2012; Jaroslav, 2018). Incineration can effectively eliminate pathogenic microorganisms and remove most synthetic organic compounds from sludge (Syed-Hassan et al., 2017). In addition, during the sludge incineration, most heavy metals such as Cr, Cu, and Ni were retained in the ash (Manara & Zabaniotou, 2012), but the volatilization of some heavy metals (Hg, As, Cd, Zn, and Pb) in gases phase is still very common due to volatility difference of heavy metals (Table 9.5). Therefore, there have been several environmental problems with sludge incineration during the operating process, including heavy metal emission, dioxin emission, acid gas emission, and ash discharge.

9.6.1 Heavy Metal Emission and Control

1. Heavy metal emission. Heavy metals come from sewage discharge, and these include metal oxides, metal hydroxides, and salty substances (Saqib & Bäckström, 2016). Some heavy metals in the combustion process are discharged as solids. However, some substances are discharged in the form of gas during the process of combustion, and the gaseous heavy metals enter the atmosphere, causing harm to the environment and human health.

TABLE 9.5
Volatilities of Different Heavy Metals and Related Compounds

Elements	Boiling Point (°C)	Vapor Pressure (Pa) 760°C	980°C	Characteristic
Hg	357	_	_	Volatile
As	615	159,960	2.399×10^{7}	Volatile
Cd	767	94,643	733,150	Volatile
Zn	907	18,662	213,280	Volatile
$PbCl_2$	954	9997.5	106,640	Volatile
Pb	1620	4.67	173.29	Nonvolatile
Cr	2200	0.7998	5.87×10^{-3}	Nonvolatile
Cu	2300	1.1997	7.20×10^{-3}	Nonvolatile
Ni	2900	7.46×10^{-8}	1.47×10^{-4}	Nonvolatile

2. Control of heavy metals. Adding sorbents in the burning process such as limestone and kaolin enhance heavy metal adsorption and make heavy metal enrichment and condensation occur rapidly on the adsorbent. Heavy metals will become ash after precipitation, which reduces the volatilization of heavy metals into the atmosphere and avoid the harm to human health. Kuo et al. (2011) examined the factors influencing the adsorption of heavy metals (Pb, Cr, and Cd) by aluminum- and calcium-based adsorbents during fluidized bed incineration and the effect of sodium addition on the agglomeration of adsorbent particles. The addition of Na increases the tendency to agglomerate, leading to the enrichment and entrapment of heavy metals in large particles, reducing their uncontrolled emission through the flue gas. Wang et al. (2018) used wet grinding in a grate incinerator and fluidized bed incinerator to improve the stability of toxic heavy metals in incinerated fly ash. The relative leaching rates of all studied heavy metals (Cr, Cu, Zn, Cd, and Pb) were reduced after 24 h of wet milling.

9.6.2 Dioxin Emission and Control

1. Emission of dioxins. A large number of PAHs are inevitably produced during sludge incineration. The list of common priority PAHs in different countries are summarized in Table 9.6. Dioxins are the most common and harmful pollutants in sludge incineration process. The formation mechanism of dioxins is quite complex. There are three possible ways to generate dioxins during sludge incineration. First, there can be an incomplete combustion

TABLE 9.6
List of Common Priority PAHs in Different Countries

	European Union	German	America	New Zealand	China
1	Benzo(a)pyrene	Naphthalene	Naphthalene	Acenaphthylene	Naphthalene
2	Benzo(e)pyrene	Acenaphthylene	Acenaphylene	Phenanthrene	Acenaphthylene
3	Benzo(a)anthracene	Acenaphthene	Acenaphthene	Fluoranthene	Acenaphthene
4	Chrysene	Fluorene	Fluorene	Pyrene	Fluorene
5	Benzo(b) fluoranthene	Phenanthrene	Phenanthrene	Benzo(g,h,i) perylene	Phenanthrene
6	Benzo(j) fluoranthene	Anthracene	Anthracene		Anthracene
7	Benzo(k) fluoranthene	Fluoranthene	Fluoranthene		Fluoranthene
8	Dibenzo(a,h) anthracene	Pyrene	Pyrene		Pyrene
9		Benzo(a)anthracene	Benzoanthracene		Benz[a]anthracene
10		Chrysene	Chrysene		Chrysene
11		Benzo(b)fluoranthene	Benzo(b)fluoranthene		Benzo[b] fluoranthene
12		Benzo(k)fluoranthene	Benzo(k)fluoranthene		Benzo[k] fluoranthene
13		Benzo(a)pyrene	Benzo(a)pyrene		Benzo[a]pyrene
14		Indeno(1,2,3-cd) pyrene	Dibenzo[a,h]anthracene		Dibenzo[a, h] anthracene
15		Dibenzo(a,h) anthracene	Benzoperylene		Indeno[1,2,3-cd] pyrene
16		Benzo(g,h,i)perylene	Indenopyrene		Benzo[g, h, i] perylene

of compounds containing polychlorinated dibenzo-p-dioxins (PCDDs)/polychlorinated dibenzofurans (PCDFs) in the combustion chamber. Second, chlorine-containing compounds (such as chlorophenol and chlorobenzene) will be pyrolyzed and rearranged at temperatures ranging from 500°C to 800°C, rapidly producing a large amount of dioxins, which is the "high-temperature in-phase synthesis mechanism". At high temperatures, the decomposition rate of dioxins is much higher than the rate of dioxins synthesized from precursors. The third theory is the reaction synthesis of inorganic chlorides and organic compounds with the participation of catalysts, including a de novo reaction and heterogeneous precursor formation mechanism. The metal compounds existing in fly ash catalyze the formation of dioxins in a lower temperature range. Dioxins are discharged in the form of gaseous or dust particles during sludge incineration.

2. Control of dioxins. The formation of dioxins is affected by incineration temperature, residence time, oxygen content, and sulfur/chlorine content. If the production conditions and process parameters are strictly controlled, the formation of dioxins can be effectively controlled. When the combustion temperature is higher than 850°C and the residence time is longer than 2 s, the decomposition rate of dioxin in flue gas is greater than 98%. Therefore, the generation of dioxins can be effectively controlled by controlling the incineration temperature and residence time. The peak temperature range of dioxin resynthesis is 250–500°C. The temperature of the flue gas from incineration can rapidly drop below 200°C to reduce the formation of dioxin. Dioxin production decreases with the decrease of oxygen content, and a 50% reduction of oxygen can reduce the reformation of dioxins by 30%. Therefore, it is recommended to control the oxygen content below 8%. The chlorine of dioxins mainly exists in the form of chlorine or hydrogen chloride. The chlorine content and sulfur chloride ratio participate in the formation of incomplete combustion. With the increase of chlorine content in sludge, the emission of PCDD/PCDF in flue gas increases. Therefore, calcium oxide and limestone can be added to control the formation of hydrogen chloride, the precursor of dioxins. The formation of chlorine gas is mainly generated through a Deacon reaction, and sulfur dioxide can inhibit this reaction. With the increase of sulfur chloride ratio in sludge, the formation concentration of dioxins and furans decreases, which inhibits the formation of dioxins.

In recent research, scholars have developed more treatment methods for the removal of toxic gases generated by sludge incineration. Xu et al. (2018) used spent anion exchange resin to inhibit PCDD/PCDFs generated during the incineration of hot-rolled sludge. Their results indicated that the spent anion exchange resin inhibited the formation of PCDD/Fs with an efficiency of 97.8%.

9.6.3 Acid Gas Emission and Control

1. Acid gas emission. Nitrogen oxides and sulfur oxides are produced from sludge incineration and cause atmospheric pollution (Liu et al., 2021; Lu et al., 2013).
2. Acid gas control. It is found that the generation of nitrogen oxides can be reduced by controlling the incineration temperature or by adding an alkaline sorbent. Therefore, the emission of nitrogen oxides into the atmosphere can be reduced by studying the selective catalytic reaction of flue gas. The formation of sulfur oxides in the incineration process is mainly due to the combination of sulfur elements in the sludge and oxygen in the incineration process. The desulfurization in the combustion process is fixed by adding a sulfur reinforcement agent, and the flue gas purification and sulfur removal are carried out through the desulfurization device. Shin et al. (2009) prepared a high specific surface area calcium hydroxide adsorbent for the removal of SO_2 from incineration flue gas. At present, many studies have shown that sulfur elements are related to the control of heavy metals

and dioxins in sludge incineration. Han et al. (2015) found that SO_2, fine particulate matter, and heavy metals from plant sewage sludge incineration could be removed simultaneously using porous calcium-based bead adsorbents.

9.6.4 Ash Discharge and Control

1. Ash discharge. The soot produced by sludge incineration includes three components: black smoke, fly ash, and ash slag. Heavy metals in sludge are deposited on the incinerated ash slag (including bottom slag and fly ash) after incineration, which makes sludge incineration ash more toxic (Saqib & Bäckström, 2016; Thomé-Kozmiensky, 1998).
2. Ash control. Pollutants can be inhibited by controlling the incineration temperature and environment, and using adsorbents in the process of sludge incineration. Currently, widely used applications in China include electrostatic precipitators, baghouse precipitators, and ionized wet scrubbers. Fly ash from municipal domestic waste incineration can be disposed of as ordinary solid waste in landfills after special treatment (such as vitrification, melting, and chemical stabilization) to meet the requirements of the Pollution Standards including the domestic waste landfill control (standard number: GB 16889-2008). We also note that sludge contains large amounts of nitrogen and phosphorus. Thant Zin and Kim (2021) recovered both phosphorus and nitrogen in fly ash from incinerated sludge by using magnesium-modified biochar in the form of guano stone, making the fly ash a usable resource for phosphorus and nitrogen.

9.7 CONCLUSIONS AND PERSPECTIVES

Sludge incineration is a promising method for sludge treatment and disposal. In the incineration process, the incineration method and equipment are particularly important. The most widely used incineration equipment includes the multi-furnace, fluidized bed, and rotary kiln. The flue gas and fly ash from the incineration process should be treated properly because they contain a large concentration of heavy metals and toxic substances. The current search for effective treatment methods for hazardous substances generated during sludge incineration should be continued to maximize resource utilization.

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10 Hydrothermal Liquefaction and Carbonization for Sustainable Treatment of Sewage Sludge

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10.1 INTRODUCTION

Waste generation is inevitable when natural resources are transformed into usable forms and are marketized to large populations. The demand for natural resources has risen at an alarming rate and thus has influenced allied waste generation. Waste as a byproduct of consumption activities is returned to the environment in solid, liquid, and gaseous forms. The changing consumption pattern and the advancement in technology-based lifestyles are followed by the generation of larger and typically challenging quality wastes. Among the wastes generated, wastewater, not only from industrial but municipal sources, has been an area of investigation to measure the environmental performance of a region. The population and resource demand have influenced a load of organic matter and nutrients in wastewater streams worldwide. From a sustainability aspect, addressing the needs together with the minimization of waste is thought to be a prudent approach. But it is well understood that complexities in achieving the real targets differ from what can be inferred from the documentation. Environmental performance does not depend on the quality of resources alone; instead, it depends on the impacts of the changing quantity and long-term effects. Wastewater quality has been changing rapidly with the change in lifestyle and composition of edible consumption. The consumption rate of antibiotic drugs and polymer-based products has recently changed with the advancement of public goods technologies (Aydın et al., 2022). It has affected the composition of wastewater to a considerable extent.

Sewerage has been an essential element of civilization and an integral part of the public management system. The increasing population has increased the load on the sewerage systems, and treatment facilities have started operating over the optimum load capacities. Since the wastewater has experienced a positive shift, the quantity of sewage sludge from the treatment process has also changed. Not only the quantity but the quality of sewage sludge has been variable to a greater extent in recent times than what has been observed in the last decade. To manage the wastewater, treatment of varying scales and modes is applied. This is modern-day wastewater treatment (WWT) technology. WWT reduces the pollutant and toxic load from the urban streams by employing physical, biological, and advanced integrated processes to generate acceptable water quality and sludge. Sewage sludge is the semi-solid residual stream originating from wastewater and/or

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liquid processing systems where the influent stream is characterized by a high total suspended solids (TSS) load (Dichtl et al., 2007). Depending on the stages of WWT, the sludge generated can be categorized as primary, secondary, or tertiary sludge (Banerjee et al., 2020). While depending on the process adopted for WWT, the sludge can be broadly classified as activated sewage sludge, up-flow anaerobic sludge, aerobic-oxic, anoxic, and raw sludge (Gong et al., 2014). Sewage sludge quantity and quality are both serious concerns in recent times. Organic pollutants characterize it significantly, including priority pollutants, such as polychlorinated biphenyls (PCBs), heavy metals (HMs), polyaromatic hydrocarbons (PAHs), and pathogenic microbes. These substances also pose a question about the direct application of sludge for land and other purposes, which may expose the compound to the consumers in the vicinity of disposal or treatment sites (Amir et al., 2005; Banu et al., 2011). WWT is an established process, and different combinations of treatment processes are employed globally depending on the quality and load of wastewater to be handled by the treatment plant. However, it is still observed that treatment and the fate of sewage sludge are not standardized and is a concern in most areas struggling with waste disposal and applications. Being rich in organic carbon, organic/inorganic nitrogen, phosphorous and inorganic aluminates, and silicate materials, the sludge from the secondary and tertiary facilities is an asset for resource recovery (Dentel, 2004; Fytili and Zabaniotou, 2008). The challenge remains with the selection of the conversion pathway and its sustainability aspect. Another aspect of the consideration of sewage sludge processing and conversion is the economics of the WWT industry. The cost incurred on the treatment facilities may reduce significantly if sewage sludge is converted to utilizable compounds, as about 50% of the cost of WWTP is contributed by sludge handling and management practices (Liu et al., 2013). Hence, recovering resources and conversion of sewage sludge for secondary and value compound synthesis may aid in the sustainability of the WWT process as well as may contribute to compensating resource depletion.

10.1.1 The Necessity for Sewage Sludge Hydrothermal Conversion

Currently, the treatment capacity for wastewater in developed nations stands at 70%, whereas it is about 8% in the developing world (Shan, 2018). It is unclear, due to the unavailability of data that in reality what proportion of sludge is treated and what could be the trend in its generation in coming times (Cieślik et al., 2015). With a focus on resource recovery and sustainable development goals, several pathways have been defined to convert sewage sludge (Banerjee et al., 2020). Earlier, sludge composting and disposal was thought to be a good option for the management of sewage sludge, but investigations on the fate of the HM and the pathogenic microbial consortia have raised concern about its direct settlement with land applications (Lu et al., 2021). Although composting may recover the NPK in sludge, HMs tend to mobilize with the nutrients, which may pose a threat to the vegetation and may trigger the risk of biomagnification with time (Yang et al., 2015; Xiong et al., 2022).

Apart from traditional composting, incineration has been applied for volume reduction and conversion of sewage sludge to ash. Sewage sludge-based ash from incineration is rich in phosphorous. Several suggestive methods have been discussed for P recovery with other nutrients, but the concern in this conversion process is energy efficiency (Murakami et al., 2009; Zhu et al., 2021). Sewage sludge is variable in its characteristics, and the moisture and organic load are significant factors for incineration. The moisture content of more than 80% and lower organic load would significantly reduce the incineration process' energy efficiency by lowering the feedstock's high heating value (HHV). Also, if the moisture content of sewage is managed by prior dewatering and sludge drying, the process cost and overall energy balance would be less sustainable (Liu et al., 2022). For these reasons and associated pollution risk to air quality, incineration or direct combustion process also get limited in mass scale and technological applications.

The thermochemical pathways, such as pyrolysis, combustion, gasification, and hydrothermal treatment (HT), have gained attention for their short run time and higher yield. The pathogen and the organic load in sewage sludge are decomposed to H_2, CO, CO_2, CH_4, bio-oil, and bio-char during pyrolysis (Liu et al., 2018; Seo et al., 2022). The conversion through the pyrolysis route largely depends on the moisture in the feedstock and ash content, which in the case of sewage sludge is as high as 80–95%. For this reason, drying and/or dewatering of sludge is essential before directing it to pyrolysis. In gasification, the major product is syngas, which is evident in the energy recovery potential from sewage sludge. But the large quantity of tar poses a challenge in downstream processing, raising the cost and complexity in handling the plant operations (Choi et al., 2018). The effective removal of moisture, either by dewatering through physical or mechanical means, somehow becomes necessary for these processes to be effective in resource and energy recovery (Syed-Hassan et al., 2017). While thermochemical processes may be integrated with biochemical or physical processing to avoid the lacuna of moisture and ash, HT and conversion could address this issue successfully.

Recently, advanced and novel conversion methods, such as thermal plasma, wet oxidation, radiation, ultrasound treatment, microwave treatment, and hydrothermal processing, have also been investigated for sewage sludge conversion (Bertanza et al., 2015; Ali et al., 2016; Jákói et al., 2021). Among them, hydrothermal processing is much more efficient for resource and energy recovery as well as for scalability. Other methods may convert sludge into acceptable quality, but their scale of operation is still limited, and due to larger energy consumption, these processes are not applied on industrial scales (Yin and Wang, 2019). HT is considered suitable for sewage sludge because its process compatibility with wet feedstocks is its major advantage. The concept of hydrothermal processing of biomass and the behaviour of water in the subcritical and supercritical states is well studied (Venkatachalam et al., 2020). The solvent action of the water can degrade organic matter content in sewage sludge at the subcritical and supercritical conditions for effective conversion of organics and nutrients to biocrude, bio-char, and gases. HT can degrade about 90% of organic matter while reducing the volume of sludge significantly without the emission of noxious gases and toxicants out of the system (Quitain et al., 2002). Compared to other methods, HT has been recognized to work efficiently with sewage sludge. It can be integrated with biochemical processes such as anaerobic digestion and bioconversion in aerobic conditions to yield further value compounds (Villamil et al., 2018). Such approaches make HT promising for treating sewage sludge, and its advantages concerning scale and prospects in biorefinery design are suitable to fit in circular economy plans (Gherghel et al., 2019; Zohar et al., 2021; Cao et al., 2021).

10.1.2 Characteristics of Sewage Sludge Favourable for Hydrothermal Process

The compositional profile of sewage sludge is formed of organic matter, volatiles, ash, and moisture. The volatile content of sewage sludge consists of carbohydrates, proteins, and lipid compounds which are majorly from the biological processing of wastewater, while lipid may arise from both wastewater processing and wastewater source. Sewage sludge can be considered a multifarious amalgam consisting of microbial biomass, inorganics, and undigested organics with a relatively minor content of cellulose and lignin (Manara and Zabaniotou, 2012; Vickers, 2017). An illustration of major components based on their origin in sewage sludge is presented in Figure 10.1.

Alike lignocellulosic biomass, the cellulose, hemicellulose, and lignin content in sewage sludge are limited. Carbohydrates and proteins are abundant, while a small fraction of lignin can be observed in sewage sludge (He et al., 2013). The cellulose content in sludge is sourced majorly from the use of toilet paper, and thus, the cellulose loading in the raw wastewater is found to be higher than in the treated effluent. WWT concentrates the cellulosic fraction into sludge which may vary depending on the WWT stage and process. In a comparative study of sludge, it was reported that cellulose concentration varied between 3,556 and 67 mg/L for primary and secondary sludge, while variation was between 11,905 and 150 mg/L for primary and secondary sludge for North American and European scenarios, respectively (Ahmed et al., 2019). Also, the cellulose content may vary from 7% to 1% of TSS for primary and secondary sludge, respectively. It must be noted that when thermochemical conversion is considered, the major consideration is that secondary sludge, which

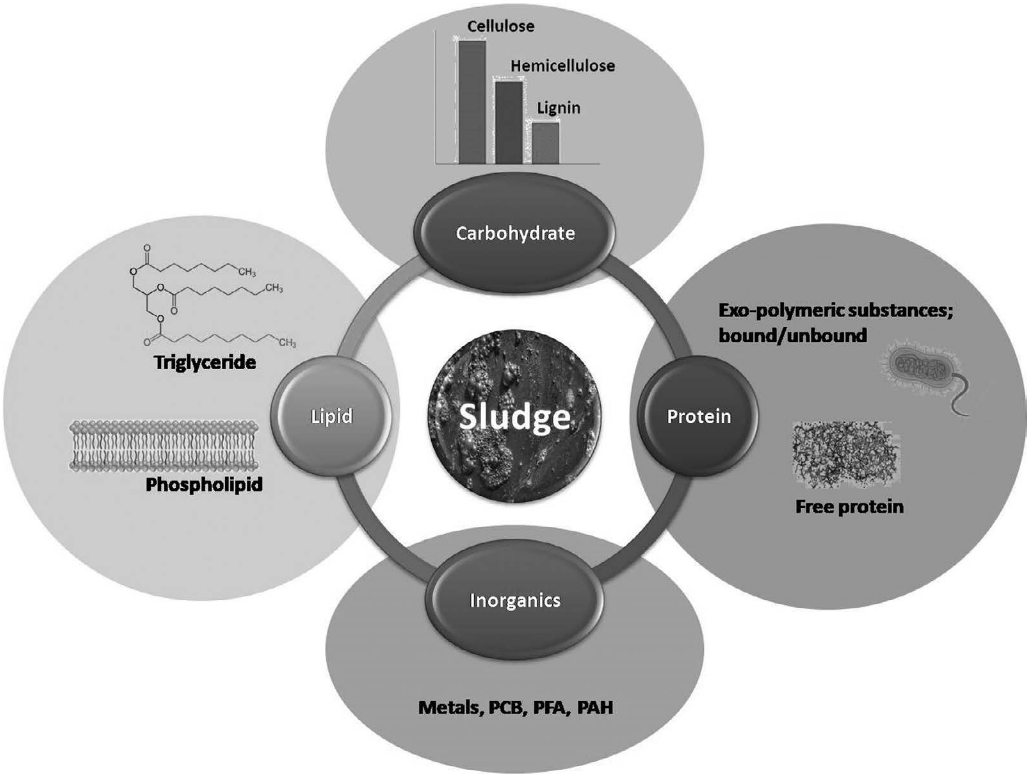

FIGURE 10.1 Major components in sewage sludge based on its origin.

is rich in carbon and other mineral content. Hence, the composition of sludge is necessary for consideration before HT. Apart from the carbon content, phosphorus and nitrogen make up a significant proportion of sludge biomass. The proteins and extracellular polymeric substances (EPSs) formed during biological or secondary treatment are a major source of nitrogen as well as for holding the metal and inorganic species in agglomeration (Samolada and Zabaniotou, 2014; Wang et al., 2018).

Sewage sludge is also characterized by non-biodegradable constituents, which, if present in high quantities, may render it unsuitable for biotransformation effectively. Substances of concern, such as polychlorinated biphenyls (PCBs), per- and poly-fluoroalkyl (PFAs), PAHs, benzotriazoles, and surfactants, should not be ignored when any conversion process for sewage sludge is under consideration (Fijalkowski et al., 2017; Ozaki et al., 2017; Wołejko et al., 2018). Among the alkyl moieties, the PFAs in the sewage sludge are also characterized by perfluorooctanoic acid (PFOA) and perfluorooctanesulfonic acid (PFOS). The biotoxicity and bioavailability of these organic acids and derivatives can lead to the oxidation load on the conversion processes and may, thus, require the intensity of temperature and pressure (Yang et al., 2020). HMs are significantly present in sewage sludge among which copper, nickel, zinc, iron, and cadmium are frequently reported (Fijalkowski et al., 2017). The presence of HMs may sometimes pose an additional challenge in sewage sludge conversion as mere temperature-based treatment can immobilize it with the carbon matrix in the remaining solid (Xiong et al., 2018). For this reason, co-conversion of sludge is also proposed or, conversion of sewage sludge can be undertaken at moderate temperature and hydrothermal profile to stabilize the metals and reduce its mobility after application of the product from the conversion process (Meng et al., 2018; Oleszczuk and Hollert, 2011).

The available organic content with the presence of hydrocarbons and volatiles, and its complexity with the metal species, make sewage sludge highly heterogeneous. With all these constituents, the presence of moisture is an integral quality of sludge. In all these factors for conversion, HT stands suitable concerning yield and quality improvement according to sludge volume reduction and reducing toxicity while granting products such as biocrude and hydrochar. Thus, the HT for sewage sludge and its potential for attaining conversion sustainability along with environmental implications for resource and energy recovery have been discussed in the coming sections.

10.2 HYDROTHERMAL TREATMENT OF SEWAGE SLUDGE

Hydrothermal processing is described in terms of temperature, pressure, and reaction time parameter range. Water is necessary for HT; thus, it also serves as a solvent in most conversion processes. Bio-oil, hydrochar, and gases are obtained from all HTs but are variable in proportion. Hydrothermal liquefaction (HTL) and hydrothermal gasification (HTG) are applied when the targeted product is bio-oil and gases, while for hydrochar, hydrothermal carbonization (HTC) is preferred (Figure 10.2). All three modes of HT have their merits and limitations depending on the feedstock and required quality of the product, but they are efficient for sewage sludge handling. The reactivity of water is the influencing factor that differentiates HT from other thermal processes. The mechanism of cracking, decarboxylation, dehydration, and hydrolysis is initiated by altering the reactivity of water at subcritical and supercritical conditions (Djandja et al., 2021).

HTL process is identified with high temperature and pressure profiles ranging from 260°C to 450°C and 7 to 25 MPa, respectively (Li et al., 2010; Zhang and Chen, 2018; Hu et al., 2020). Conversion of sewage sludge, such as other organic-rich feedstock, is believed to follow the actions of free radicals and ionic moieties with temperature as a major governing factor for process yield (Hu et al., 2020). HTL may be carried out in both subcritical and supercritical conditions of the water. At subcritical conditions, the conversion is less due to the partial diffusive action of water concerning sewage sludge. But when temperature and pressure reach the critical point, the reaction kinetics change, and the conversion increases beyond 374°C and 22 MPa (Chen, 2017). However, the reality of this process is not this simple to infer that subcritical and supercritical conditions can decide the quality and quantity of yield. The feedstock characteristic is a decisive factor and must not be ignored when understanding the conversion of a highly heterogeneous feedstock as sewage sludge. Concerning water availability in sewage sludge and water as a solvent in the HTL process,

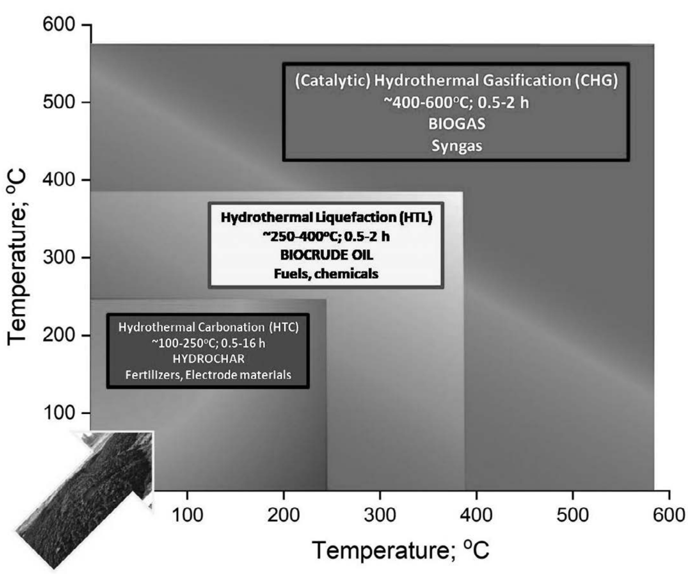

FIGURE 10.2 Hydrothermal conversion processes for sewage sludge.

the possibility of reactions favoured by H^+ ions occurs better at subcritical conditions. In contrast, ionic reactions occur better at supercritical conditions, implying the significant role of temperature and pressure at subcritical and supercritical conditions, respectively (Kruse et al., 2013). Both feedstock and conversion processes are thus complex in operation and mechanism. The discussions in the following sections are useful to understand the phenomenon of HT in application to convert sewage sludge into value products.

10.2.1 Hydrothermal Liquefaction of Sewage Sludge

The primary target for HTL of sewage sludge is the production of bio-oil or sometimes chemicals. Majorly performed in an oxygen-free atmosphere, the temperature and pressure conditions are maintained at 200°C–300°C and 5–15 MPa, respectively. The proportion of yield is more towards bio-oil than towards hydrochar or gas. The process condition and related trends are most important in understanding the HTL process and product formation from the feedstock and sewage sludge in the present case.

10.2.1.1 Influence of Parameters in HTL of Sewage Sludge

Temperature

Temperature is the major influencing parameter in the HTL process. A temperature range of 250°C–350°C is typical for the degradation of carbohydrates, fatty acids, and aliphatic molecules, while a range of 400°C–550°C is required for the degradation of complex aromatics (Prestigiacomo et al., 2019). The yield of bio-oil increases with increasing temperature until it reaches a critical point beyond which the yield is inhibited. The subcritical condition is found suitable for sewage sludge HTL for two reasons: First, it does not raise the operational cost, and second, it restricts the proportion of yield towards gaseous products. At a higher temperature range, the acidic compounds decompose, which increases the aqueous phase pH, ultimately promoting a more gaseous product to bio-oil (Akhtar and Amin, 2011).

Water

Water, being the major solvent in the HTL process, influences the mechanism. The percentage of moisture and solvent in the feedstock synergistically influences the product yield and quality. The higher moisture in the feedstock favours hydrolysis of polysaccharides, peptides, and lipids, while more water in the HTL reaction means a denser solvent medium. This dissolute the sewage sludge components into aqueous products and bio-oil. The consideration for pH of water is essential as in alkaline conditions, the formed bio-oil may transform to aldehydes and acids, leading to gaseous products, while in acidic conditions, carbohydrate compounds such as cellulose may be transformed to 5-hydroxymethylfurfural, which would get distributed to the liquid and solid phases.

Reaction Time

Reaction time influences the energy recovery, HHV, and bio-oil yield positively. However, it may be considered that reaction time under isothermal conditions may also have a critical point where the HHV may rise, but the energy recovery may fall. A longer reaction time enhances intermediate matter condensation and repolymerization. Pre-asphaltenes and asphaltenes can be broken down into lighter compounds that can increase light oil and gas yields. Reaction time independently is not a significant influence on the product and energy. The effect of temperature and reaction time can be explained as, for a higher temperature and short period, the bio-oil yield decreases, while the yield increases for mild temperatures and longer residence time.

Pressure

An optimized pressure condition is necessary to hold the reaction in a thermodynamically favourable state. With an increase in the pressure during HTL, the density of solvent increases, which can increase the components' decomposition and yield. However, while applying a catalyst, it should also be considered that the denser solvent medium may inhibit the catalysis effect by blocking the active sites. When the reaction medium is water, it is important to consider the state of water depending on the temperature regime during the reaction, as a subcritical and supercritical state of water is directly related to pressure and product yield. At subcritical conditions, the pressure has less influence on enhancing the product yield at HTL. Water does not change its density even at high pressure when in subcritical conditions, and in such cases, the pressure may have negligible impact on the reaction yield (Yokoyama et al., 1987). The yield of bio-oil may increase to a certain extent under supercritical conditions. Qian et al. (2017) described that the yield of bio-oil increased from 21.7% to 25% while increasing the pressure from 20 to 22.1 MPa. However, the yield was not significantly increased beyond 25 MPa, which is the critical pressure for water. Another important aspect to note is that in the absence of an external gas, the pressure during HTL reaction is the saturated pressure of water, which ranges nearby 9 MPa at 300°C. This can be used for sewage sludge conversion and, hence, would be more reliable and cost-effective.

Reaction Atmosphere

HTL atmosphere is typically maintained with inert and/or reducing gases. The inert gases used are nitrogen and argon, while syngas and steam are applied to facilitate reducing environment. Stabilization of the product is achieved by facilitating multiple reactions, such as condensation, cyclization, and inhibition of free radical repolymerization (Jindal and Jha, 2015). It is reported that the bio-oil yield for the HTL process is favoured under CO and H_2 atmospheres than at N_2. The suitability of CO for HTL of organic-rich feeds is for the reason that it can react to carbonate, yielding hydrogen free radicals, which would prevent char formation by stabilizing intermediates and thus enhancing bio-oil yield. Also, CO and H_2 favour the Fischer-Tropsch synthesis, hydrogenolysis, and/or hydrogenation, which leads to lower polarity in the formed compounds rendering the oil quality from the process suitable for applications (Yin et al., 2010). To focus on the sustainability aspect of CO, the application is limited as it is a pollutant and has prohibitions on its use. Hydrogen is suitable for reducing gas, but the cost limits its application on a larger scale. In this regard, the application of H_2 donating catalysts has gained attention for their applicability and cost-effectiveness in the HTL process.

Catalyst

The catalysts used in HTL reactions are both homogenous and heterogeneous. Catalyst application for HTL of sewage sludge is more oriented towards homogenous compounds. Na_2CO_3, $FeSO_4$, MoS_2, Raney-nickel, and alkaline catalysts have been tested for sewage sludge HTL. It has been reported that a bio-oil yield of 45.58% can be obtained with a total conversion of 68.21% using $FeSO_4$. However, sulphur-based catalyst led to a high S content on the bio-oil fraction. On the other hand, the Na_2CO_3 and Raney-nickel reduced the N content in bio-oil up to 16%. Thus, the selection of catalyst is also important for the fate of metal species on the product side (Malins et al., 2015). The conditioning of the process with the catalyst may further improve the solid-to-liquid ratio in the HTL process and may open possibilities for HTC as well. But to note, investigations are required as the catalyst performance is dependent on the pressure and solvent state in the HTL process, as discussed in Section 10.2.1.1.4.

10.2.2 Hydrothermal Carbonization of Sewage Sludge

Sewage sludge can be converted to hydrochar with various applications via HTC. HTC occurs as a result of multiple reactions and is understood to be exothermic for pure compounds (Libra et al., 2011). This may favour the energy efficiency potential of sludge HTC compared to the usual carbonization process with minimized moisture content. Sludge HTC is carried out at a temperature range of 180–250°C under a pressure range of 2–10 MPa (Hitzl et al., 2015; Afolabi et al., 2020). HTC reactions are complex, and their precise construction is not understood well. The process reactions are simplified and presented by considering the solid yield without much focus on the liquid and gaseous yields (Funke and Ziegler, 2010). The major reactions involved in the process of hydrochar formation are evaluated to be dehydration, hydrolysis, carboxylation, polymerization, and aromatization (Berge et al., 2011; Saqib et al., 2019).

The influence of sewage sludge components on the properties of hydrochar is significant. Sewage sludge with carbohydrate content may produce the respective monosaccharides after hydrolysis, which again would form 5-hydroxymethylfurfural and furfural after the dehydration reaction. The HMF produced may lead to coke formation, initiating polymerization and aggregation, while devolatilization may increase the porosity of hydrochar. The nitrogen content from the proteins in sludge may trigger an alternate pathway of conversion. The release of the ammonia group from protein hydrolysis may produce an alkaline condition during HTC, which may induce the formation of aliphatic compounds and acids (Usman et al., 2019).

10.2.2.1 Influence of Parameters in HTC on Sewage Sludge

Temperature

The temperature influences the carbon, inorganics, and texture of hydrochar. The application of temperature defines the heat transfer to the system for macromolecule disintegration and fragmentation. Selecting a suitable temperature for the initiation of a process and complete conversion of sewage sludge is essential. At lower temperatures, depolymerization is dominant, which yields fragments of organic polymers, followed by polymerization reactions, which takes place at higher temperature forming hydrochar. Increasing the temperature to the retention time may significantly yield hydrochar with high fixed carbon, but the mineral content of the char may be compromised in such cases (Danso-Boateng et al., 2015). Wang et al. (2020) evaluated the HTC of sewage sludge with distilled water as solvent at temperatures of 180, 200, 220, and 240°C for a residence time of 1 hour and obtained a maximum hydrochar yield of 74.96% at 180°C. It was also reported that with an increase in the reaction temperature and time, the carbon, hydrogen, and nitrogen content in hydrochar reduced while these accumulated in increasing ash content (Wang et al., 2020). Under such conditions, the recovery of P and S can also be achieved if the reaction conditions are optimized with proper mapping of their mobility between char and ash with temperature. An increase in the temperature from 180 to 300°C was reported to increase the yield of hydrochar from 60 to 80% with a minor influence of residence time on yield. The nitrogen content was observed to mobilize towards the liquid phase at elevated temperature and residence time with 20% of retention in hydrochar (Zhuang et al., 2017).

Water

Like the process of liquefaction, HTC is dependent on water maintained typically at subcritical conditions. The contribution of moisture in the reaction medium is an influencing factor for product distribution in HTC. The rise in temperature decreases the density of water with an increase in the ionic product (dissociation constant), allowing reactions favoured by ionic pathways. This favours the reaction of non-polar organic compounds of sewage sludge during carbonization (Kritzer, 2004).

Sewage sludge has a high moisture content which may pose a challenge in its conversion to hydrochar. In such a case, the solid loading must be considered before the HTC process. Sewage sludge with a high moisture content ranging beyond 80–85% would have low solid loading in the reactor, and the organic matter present would undergo hydrolysis initially followed by decomposition and less residue as fixed carbon would be left in the solid phase. For high moisture-containing feeds, the solid loading in the reactor must be high to facilitate the initial hydrolysis of polymers to monomers which are transported to the aqueous phase and may lead to the initiation of polymerization and increased solid precipitation (Robbiani, 2013). For this reason, dewatering of sewage sludge by mechanical means to a moisture level of 80–85% or less is recommended under consideration of process economics as well as yield (Wang et al., 2019).

Reaction Time

Reaction time with temperature synergistically influences the nature of hydrochar. Increasing retention time at elevated temperatures may reduce the hydrochar yield to an extent but improves the porosity. However, shorter retention time with temperature rise is mentioned as an influencing condition of increased hydrophobicity of hydrochar (He et al., 2013). Also, secondary polyaromatics may be formed if the aqueous phase is polymerized for a longer retention time, including the increased percentage of ash and coke in the solid fraction (Peng et al., 2016). It is to be noted that residence time positively influences the operating cost of the process along with the product characteristics. HTC of sewage sludge is reported to be a slow process in which the rate of reaction is mostly influenced by diffusion mechanisms originating from decomposition and polymerization (Nizamuddin et al., 2017). Although the slow rate is considered at elevated temperatures, the yield of hydrochar is enhanced. Here, it may be observed that the biomolecules have a certain rate of degradation at mild temperature and the disintegration products originating from them may require high temperature for a fast conversion. The influence of residence time for other parameters is yet to be investigated precisely for the HTC of sewage sludge.

Catalyst

Use of a catalyst in HTC of sewage sludge may increase the conversion of the organic complexes and yield a higher quantity of hydrochar by promoting deoxygenation and denitrogenation. The catalysts used in the HTC process, such as HTL, are mostly homogenous and comprise acids, alkalis, and metal salts (Neyens et al., 2003; Lynam et al., 2011). A small amount of strong acid can be used for the HTC process favouring the dehydration of carbohydrate moieties. Adding 2 mM H_2SO_4 may increase the conversion of sugars from hydrolysis to HMF and furfural, later promoting solid precipitation (Peterson et al., 2008). Also, under acidic conditions, the carbon dioxide generated from the decarboxylation reaction gets converted to carbonic acid, further functioning as an acid catalyst, but up to 260°C (Rogalinski et al., 2008). Such a situation is favourable for sludge with high carbohydrate or organic polymer content. Weak acids such as citric acid (30 g) have also been tested in HTC of 12.1–17.1 L of digested stabilized sewage sludge, and it was observed that the HHV for hydrochar increased (Escala et al., 2013). Transition metals have also been tested in the HTC of sewage sludge, although limited studies have been conducted with insights. Iron nitrate loading of up to 10 wt.% has been reported to promote hydrolysis while decreasing the transport of organic carbon and total nitrogen to the aqueous phase and increasing the elemental content of produced hydrochar (Hu et al., 2008). It can be understood that in normal conditions, it is not easy to control the diffusion of carbohydrate carbon and ammonia to the aqueous phase, but the application of a metal catalyst may induce probable characteristic advantages to hydrochar in terms of heteroatoms and metal complexes which would further enhance the applications.

10.3 ENERGY RECOVERY BY HYDROTHERMAL TREATMENT OF SEWAGE SLUDGE

Energy recovery from organic-rich waste is majorly dependent on the carbohydrate content and is also a function of efficient heat transfer in the HT system assisted by moisture (Zhai et al., 2017). The fixed carbon influences the HHV for yielded oil and char, which defines the energy conversion ratio in HT of sewage sludge (Oladejo et al., 2018). Since the HT process eliminates the drying and dewatering of sludge, which is the major restriction in recovering energy, it has been explored based on its characteristics and variable product grades. The organic fraction of sewage sludge may vary at a range of 50–80%, which can be directed majorly to biocrude by HTL or hydrochar by HTC (Prestigiacomo et al., 2019). Both processes have the specific advantage of conserving the energy density in the major product, but they differ in their integration and resource recovery potential. In HTL, the aqueous phase may retain the HMs and nutrients, which may further be required to be processed by biological conversion to recover the resources. Or it may be required to upgrade the crude to meet the market standards for application. However, in HTC, the potential for mobility of metals and nutrients to the biocrude phase and the aqueous phase is less, and it would be easier to recover the nutrients with the char fraction.

10.3.1 Energy Estimation for Hydrothermal Conversion of Sewage Sludge

Energy estimation has been represented in different forms while studying the HT of sewage sludge. Energy recovery can be represented as follows:

$$Energy\ Recovery\ (\%) = \frac{\%\ biocrude\ \ yield \times HHV_{biocrude}}{HHV_{sludge}}$$

which was suggested for conversion and mapping basic energy balance in the hydrothermal conversion of organic feedstock (Liu et al., 2018). Primary sludge has been reported to yield hydrochar at a temperature range of 140–200°C with an energy content of 21.5–23.31 MJ/kg against a typical 18.5 MJ/kg energy density for sewage sludge (Danso-Boateng et al., 2013). The solid fuel produced by HTC of sewage sludge can be more stable to convert further by thermochemical routes. It has been reported that both the activation energy and preexponential factor reduce after HT (He et al., 2013). However, it must be kept in mind that for sludge identified with high ash content, the produced char would exhibit a lower HHV value, in which case the energy recovery would be compromised (Wang et al., 2019). For a temperature range of 275–400°C, a maximum of 59% energy recovery has been reported together with aqueous phase reforming (APR) (Das et al., 2020).

For HTC, the energy recovery is majorly in terms of the compositional changes observed in the feedstock and produced hydrochar. Understanding of the energy estimation has been described in terms of gross energy recovery or output and can be estimated using feed and char elemental profile. One such method for estimation has been investigated by Channiwala and Parikh (2002) and can be represented in the form of empirical equations. The following equations consider the contribution of the feedstock composition and all the products yielded (biocrude, bio-char, and gases), during HT (Oliveira et al., 2022):

$$\begin{aligned} &HHV_{solids/liquid}\ (MJ/kg) \\ &= 0.349 \times \%C + 1.178 \times \%H + 0.1 \times \%S - 0.103 \times \%O - 0.015 \times \%N - 0.021 \times \%Ash \end{aligned}$$

Similarly, the HHV of evolved gases can be estimated considering the HHV of individual gases and their relative yield percentage as

$$HHV_{gas}\ (MJ/kg) = 141.7 \times \%H_2 + 55.5 \times \%CH_4 + 51.9 \times \%C_2H_6 + 50.4 \times \%C_3H_8$$

The energy balance of the process can then be represented as

$$\begin{aligned} &Energy\ recovery\ (\%) \\ &= \frac{(HHV_{char} \times \%yield_{char}) + (HHV_{biocrude} \times \%yield_{biocrude}) + (HHV_{gas} \times \%yield_{gas})}{HHV_{sludge}} \times 100 \end{aligned}$$

The focus for energy recovery in most of the processes is on the optimized use of internal energy within the system to reduce the expense of externally supplied energy. But it should not be ignored that the yield of products contributes significantly to the system's energy balance. Since the gases have a larger HHV to the mass fraction, it should not be confused that HTL and HTC process is less energy efficient than HTG. For HTL and HTC, the energy is distributed in decomposition and reforming, whereas dissociation energy is a significant player in HTG. Hence, the economics of the process is also dependent on the selection of sewage sludge with HHV values compatible with fuel-grade feedstocks, as well as the process condition and yield distribution of the products.

10.3.2 Enhanced Energy Recovery Potential by Process Integration

Recently, investigations have shifted to coupling the HTL process with APR and anaerobic digestion. Both strategies are suitable in terms of enhancing energy recovery, but in terms of heating value, the end product affects the energy balance. Anaerobic digestion of the aqueous phase could be preferred when a load of HMs and toxicants are admissible to environmental standards, as microbial conversion is not standardized for the reduction of bioavailability of these metal species. The abundant yield from HTL-AD would impact energy due to the heating value of methane generated, which may increase the energy efficiency by 14% (Medina-Martos et al., 2020). However, APR can be considered advantageous as that the end product is hydrogen, which has a higher energy value (120–142 MJ/kg) than methane (50–55 MJ/kg) and thus is a good competitor for energy efficiency in this process. Apart from hydrogen, alkenes are the major products of HT-APR processing (Cortright et al., 2002). APR can be carried out using a typical temperature and pressure range of 20–250°C under 15–20 bar pressure, respectively. APR can be integrated with both HTL and HTC. A recent study showed that 48.5 wt% of hydrochar can be recovered by HTC of secondary sludge originating from a membrane bioreactor at a mild temperature of 170°C. And the aqueous phase can then be reformed to yield 98.7 mmol H_2 g/TOC by applying a PtRh/CBe catalyst (Oliveira et al., 2022). However, it should be considered that the organic load is one factor that shifts the yield in APR. A high organic load from the source may contain more degradable compounds that may be refractory to APR mechanisms (Kirilin et al., 2010). Also, a high phosphate content may inhibit the activity of the catalyst applied.

Thus, the energy dynamics of sludge HT is a function of the molecular species in the feed and the process, and the parameter for yield is an influencing factor for the energy efficiency of the process at large.

10.4 RESOURCE RECOVERY BY HYDROTHERMAL TREATMENT OF SEWAGE SLUDGE

Sewage sludge is rich in nitrogen and phosphorous contents. NPK recovery from sludge is a conventional process preferred by composting. Due to the massive quantity of sludge and the process shortcomings, HT has emerged as a potential candidate for nutrient stabilization and recovery from sewage sludge. The thrust area for understanding the potential resource recovery is an investigation of the mobility and transformation of N, P, and HMs during HT of sewage sludge. Both HTL and HTC have pathways for the transformation of N, P, and metals, and they differ in their qualitative profile for stabilization. HT of sewage sludge may transform organic N from the peptides and inorganic N, present as NH^{4+}, NO_3, and NO_2, to amide and subsequently to NH^{4+} in the liquid phase. In contrast, P content as organic and polyphosphate may be transformed into inorganic and pyrophosphate.

10.4.1 Suggestive Mechanisms of Nitrogen Transformation

Nitrogen in sewage sludge is majorly from the proteins; the concentration of protein N is high in the feedstock. In the aqueous phase, the first transformation of this nitrogen occurs through hydrolysis at a temperature between 150 and 240°C. The cleavage of peptide bonds generates stable amide bonds (CO-NH), and the amide N is further converted beyond the temperature of 260°C (Ekpo et al., 2016). Raising the temperature and time of reaction further leads to the deamination of the amide compounds and initiates ring-opening reactions to yield pyridine N with some proportion of ammonia N. At high-temperature conditions beyond 300°C, the pyridine, amide, and ammonium

N are mainly converted to ammonium N due to the thermal cracking of pyridine N and residual amides. The nitrogen in biocrude has been reported to increase from 0.5% to 4.93%, with a temperature rise of 150–300°C for sewage sludge (Zhuang et al., 2017). The protein N in the sludge, such as in the aqueous phase, undergoes hydrolysis and is cleaved to amine N, which further transforms as a result of the Millard reaction in presence of reducing sugars from the carbohydrate fraction of sludge, forming heterocyclic N in the temperature range of 150°C to 270°C (Peterson et al., 2010; Déniel et al., 2016). The heterocyclic N formed can be stable at higher temperatures; even at temperatures beyond 300°C, the heterocyclic N may transform to nitrile N, but the contribution of this to TN of bio-oil would be relatively low. The char fraction does not retain much nitrogen in the HTL process and almost 80% of the TN is reported to be decomposed and mobilized to an aqueous and crude phase with less remaining in the char. This may occur as the hydrolysis of peptides at a lower temperature is fast, and most protein N is decomposed to amide N before the char formation is stabilized at higher temperatures. However, after hydrolysis, pyrrole N is formed at a temperature range of 180–240°C, which transforms to pyridine N and quaternary N, as an account of the Diels-Alder reaction (Kelemen et al., 2002). These N species finally yield ammonium N at temperatures beyond 300°C. The suggested pathways for nitrogen transformation are represented in Figure 10.3.

10.4.2 Suggestive Mechanisms of Phosphorous Transformation

Sewage sludge has a high phosphorous content originating from the food chain, as 80%–90% of phosphorous is mobilized to sewage sludge even after nutrient removal in WWTP. Thus, the fate of phosphorous in HT of sewage sludge is essential for resource recovery. Organic phosphorous (OP) is a minor contributor to the total phosphorous (TP) in sewage sludge and may reach up to 35%, whereas inorganic phosphorous (IP) is easy to recover and is present in the form of phosphates. Phosphorous in sewage sludge can be categorized into TP containing OP and IP, which is further classified into apatite phosphorous (AP), formed in association with Ca and non-apatite IP (NAIP), formed in association with oxides and hydroxides of Fe, Mn, and Al (Wang et al., 2020). The cell-bound phosphate contributor to OP content is sludge which undergoes hydrolysis to form

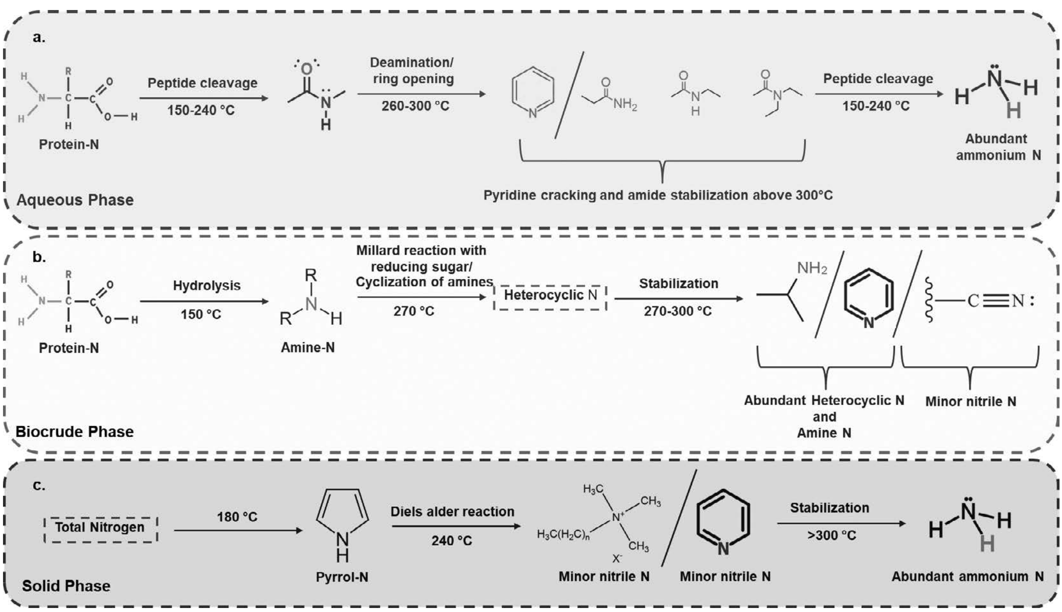

FIGURE 10.3 Suggestive pathways for nitrogen transformation; (a) aqueous phase, (b) bio-oil phase, and (c) solid phase.

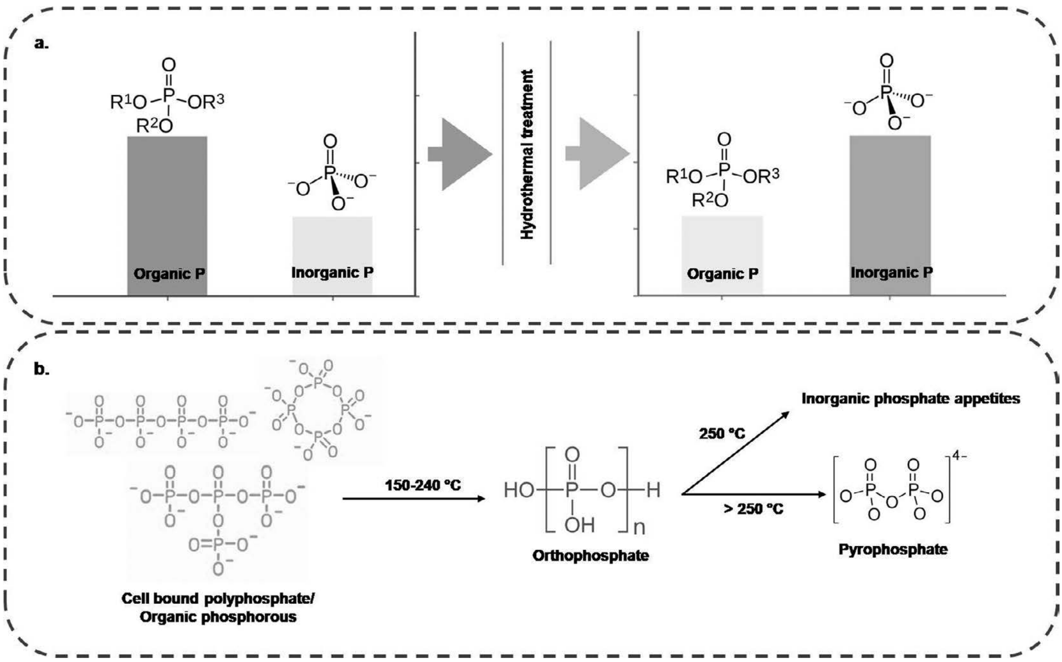

FIGURE 10.4 Suggestive pathways for phosphorous transformation; (a) trend in organic and inorganic phosphorous and (b) pathway for phosphorous transformation.

orthophosphate. Orthophosphate may react with the metal species at a temperature beyond 250°C to form inorganic phosphate salts. The alternative routes suggested for the OP and polyphosphate are dehydration and polymerization. Dehydration of OP at high temperatures, mainly when the reaction time is short, may lead to the formation of pyrophosphate, which would concentrate the P content to the solid fraction of process yield. Under normal degradation, orthophosphate could be achieved, but in certain cases, when the OP content of the feed is high, the polymerization due to the presence of active organic groups and stable organic phosphates may be formed. This, however, may not alter the content of P in the solid residue, but the characteristics may change significantly (Figure 10.4). Thus, phosphate recovery is suitable by conversion to char using HTC rather than HTL, where the aqueous phase may influence the formation of ionic salts with metals. Thus, recovery of phosphate in the form of NAIP could be viable as, in this form, the P content is abundant and most stable. Phosphate in the form of struvite can also be achieved by sewage sludge HTC with a recovery rate of 82.5% when the Mg content of sludge is significant (Becker et al., 2019). The advantage of P immobilization in sewage sludge HTC is due to the presence of varied metal species, which eases the formation of salts. Compared to algal feedstock, where phosphate may leach into the aqueous phase, the presence of metals favours its immobilization in sewage sludge (Dai et al., 2015; Ekpo et al., 2016). Hence, the P recovery potential by HT of sewage sludge would largely depend on the metal species present and their content in the sewage sludge (Zhao et al., 2018).

The HMs in the HT systems are investigated with a focus on the solid and liquid yields as in relatively lower temperatures, mobility of metals towards the gaseous phase is difficult. Many studies have been conducted to identify the fate of HM during HT of sewage sludge. Studies have suggested that the HMs are majorly immobilized to the char fraction when the temperature and pressure are high, while at longer durations of reaction, the metals may migrate to the liquid phase (Yuan et al., 2011; Li et al., 2012; Shao et al., 2015). The temperature has a significant role in the mobility of metals during HT. For HTL, the rise in reaction temperature mobilized metals towards bio-oil and a portion to hydrochar, which increases further with the increase in temperature. However, for HTC, it has been observed that increasing temperature hurts metal immobilization. Increasing HTC temperature reduces the metal content in the hydrochar fraction. Another important consideration for metal mobility is the presence of a catalyst. The catalyst NaOH has been reported to reduce the metal content from the bio-oil phase, adding it up to the char fraction (Huang and Yuan, 2015). Also, the application of solvent influences metal immobilization in the HT process. Ethanol and acetone have been shown to have the potential to concentrate the metals into solid fractions, whereas acetone has been shown to bear good capacity in mobilizing the metals (Leng et al., 2014). However, it must be noted that very few studies exist on the mechanism and mobility of metals in HT of sewage sludge, and this area still needs investigation to understand the favourable process conditions for metal recovery and its applications.

10.5 CONCLUSIONS AND PERSPECTIVES

Sewage sludge is a heterogeneous feedstock with a variable quantity of carbohydrates, proteins, cell-generated lipids, and inorganics. Characterization based on the elemental and quality profile for sludge is essential before planning for the conversion process. The characteristics of sludge are highly variable, depending not only on the region of the globe but also on the cultural differences within the regions. To address the quantity and quality of sewage sludge, thermochemical routes are better suited as they can reduce the volume while generating usable products. However, in reality, the variability in yield and targeted product quality is the challenge in designing large-scale conversion facilities. The scientific community and industrial sectors are calling for a circular economy approach, which would require mindful selection of the process and standardized conversion parameters. In the case of sewage sludge, studies have suggested that being rich in moisture, which ranges beyond 80% in most cases, this feed is suitable for HT. A direct argument is the elimination of dewatering and drying while conversion saves significant energy and time. Secondly, being rich in organics, the potential for transformation into hydrocarbon products is high.

HTL is well studied for sewage sludge which has proved to be a possible conversion process in its biorefinery design. But the major challenge is stabilizing its kinetics and optimizing the process outcomes concerning the feedstock characteristics. The next generation HTL conversion would require investigation in the co-feed system and modelling the alterations in yield with variation in sludge quality. Also, in some aspects, the sludge should not be directed to HTL, especially when volatiles are less and the nutrient or metal content is high. For such quality, conversion to solid char by HTC is preferable. Although the reactions in both processes are relatable, their mechanisms differ significantly. Hydrochar rich in metals and stabilized organic moieties can be an asset for industrial applications.

Regarding energy, the process design depends on the proportion of solid, liquid, and gaseous yields. Compounds with stability and high energy density must be traced for their formation pathways, and it may be further evaluated to standardize the yield concerning energy recovery of the process. HT should not be considered an end process for sludge conversion. Bioconversion or reforming can be coupled with HT of sewage sludge to achieve further energy recovery. The sustainability of sludge conversion through the hydrothermal process largely depends on the pathway of transformation for the metal and inorganic species. The organics transform into classified compounds, but the inorganics interact with the transforming organics to decide the properties of the end product. Hence, to think of sustainable processing of sludge using hydrothermal interventions, these issues need addressing, and this would open the area for mind work concerning other highly heterogeneous feedstocks.

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11 Gasification of Sewage Sludge

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11.1 INTRODUCTION

Increased population, rapid urbanization, and industrialization caused serious problems like climate change, air pollution, and water pollution. High quantities of wastewater from anthropogenic activities are also causing problems. Many countries have implemented wastewater treatment (WWT) methods to reduce water pollution. For this purpose, these countries installed WWT plants (WWTPs) (Werle and Sobek, 2019). There are a lot of thermochemical processes like incineration, combustion, pyrolysis, and gasification. Gasification is a thermochemical process in which carbonaceous material transforms into fuel gases, e.g., syngas. Syngas is a mixture of carbon monoxide, carbon dioxide, hydrogen, and methane hydrocarbon compounds (C_xH_y). Syngas produced by gasification can be used in power generation, chemicals, and methanol (Fericelli, 2011). In addition, disposal problems of SS can be reduced by gasification. SS gasification meets environmental and economic concerns compared to different technologies. SS is the biomass with high moisture content, low calorific value, and much less residual energy, so every technique cannot work with this feedstock. Co-gasification and gasification of SS showed the production of syngas (Gao et al., 2020). Gasification is more suitable for SS than other thermochemical technologies like combustion, incineration, and pyrolysis due to its high adoptability with SS. There are a variety of gasifiers for the gasification process, like fixed beds, fluidized beds, rotatory kilns, plasma gasifiers, and supercritical water technology.

Sewage sludge (SS) is a product that is produced after the WWT process from WWTPs. SS is a semi-solid residue released as a by-product of WWT. WWTPs purify the wastewater into useable or dischargeable form. There are a lot of WWTPs in the world which are working, as shown in Table 11.1. It is a mixture of some dissolved and undissolved particles (Demirbas et al., 2017). There are multiple sources of wastewater municipal, industrial or rural sources. The definition of sludge by the CEN (European Committee for Standardization) is a "mixture of water and solids separated from various types of water as a result of natural and artificial processes" (Twardowska et al., 2004). SS comes out from the WWTPs at the end process as waste. It belongs to the biowaste category, which can undergo aerobic decomposition, anaerobic digestion, and many thermochemical processes. Sludge may contain different minerals and organic and inorganic compounds in soluble and colloidal forms (Macedo et al., 2021).

Dumping of SS is also a problem for the community due to its presence of hazardous elements. Increased water in domestic and industrial activities increased the SS generation. Sometimes this hazardous waste is mixed up with drinking water sources, which is very hazardous for human health. In addition, entering sewage water into the drinking source will pollute and disturb aquatic life. It is very difficult to limit wastewater generation in domestic and industrial activities because, day by day, this quantity is increasing (Zhou et al., 2020). Furthermore, heavy metals, like mercury, copper, zinc, nickel, cadmium, chromium, and lead, can be found in the industrial SS (Bonfiglioli et al., 2014). This form of waste comes out from domestic and industrial WWTPs continuously. Millions of tons SS are produced annually due to the continuous flow of wastewater in treatment plants worldwide, which burdens the environment (Rulkens, 2008). Some elements present in the SS are in high quantity. Like SS is very rich in phosphorus and nitrogen as fertilizing value similar

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TABLE 11.1
WWTPs in Different Countries

Country	WWTPs	Year
Argentina	143	2020
Australia	1234	2016
Belarus	348	2020
Brazil	2820	2017
Canada	2064	2017
China	2486	2010
Europe	24,971	2020
India	816	2015
Japan	378	2020
Malaysia	484	2020
Mexico	2540	2018
New Zealand	317	2019
Peru	184	2018
Russia	1269	2020
South Africa	964	2019
Turkey	319	2020
USA	14,819	2012
Other Countries	2346	2020

to manure (Siuris, 2011). Due to its beneficial properties, it can be used in agricultural fields as a fertilizer. In this chapter, we will discuss the gasification of SS for different benefits.

11.2 SEWAGE SLUDGE COMPOSITION AND PROPERTIES

The composition of SS is very important before recovering energy from this waste through the gasification process. This waste can be categorized into six groups: (1) non-toxic compounds (found 60% in dry weight), mostly from the biological processes; (2) toxic compounds (organic and inorganic), i.e., polychlorinated biphenyls (PCBs), pesticides, dioxins, and heavy metals; (3) nitrogen and Phosphorus compounds; (4) inorganic compounds such as calcium, aluminates, silicates, and magnesium; (5) pathogens; and (6) water. The main problem with SS is that it contains all of the given previous groups in one mixture. There are very few compounds that we can use for different purposes, like nitrogen and phosphorus. Nitrogen and phosphorus are very valuable compounds present in SS. SS treatment can be a source of recovery and reuse of different valuable compounds. This sustainable treatment of SS can minimize the negative impacts of inappropriate dumping of SS on living things. Sometimes in sewage treatment, we must remove the water from SS to reduce disposal and transportation costs. The amount of nitrogen in the sludge is reduced as compared to wastewater. The amount of phosphorus compounds is dependent on the treatment process of wastewater. It is easy to concentrate all the available phosphorus in the SS. So, we can recover phosphorus from waterbodies such as waterlines (Rulkens, 2008).

11.2.1 Sewage Sludge Physical Properties

- Appearance: Raw sludge (RS) is mostly yellowish-brown in color with a strong odor. Dry sludge (DS) is slightly changed in color from yellowish-brown to ash brown with less odor. Furnace dried sludge (FDS) is also similar to DS in appearance with no odor (Raveendravarrier et al., 2020).

TABLE 11.2
Physical Properties of SS and Conventional Biomasses

Physical Property	DS	FDS	Sugarcane Bagasse	Sawdust	Coconut Shell	Rice Husk	Coir Pith	Coffee Husk
Calorific value (MJ/kg)	11.54	12.26	16.04	17.58	16.87	14.39	16.72	18.50
Bulk density (kg/m^3)	460	450	115	246	543	398	270	378
Solid density (kg/m^3)	1410	1390	746	840	876	796	1084	910

- Density: Density values are very significant characteristics of any biomass because these values give an idea of how biomass could be packed efficiently. High-density biomasses take up more space for storage and transportation purposes, ultimately costing more management expenses (Eisenbies et al., 2019). The average bulk density of RS is almost 874 kg/m^3, which is more than the bulk density of FDS (450 kg/m^3) and DS (460 kg/m^3). The average tapped densities of DS and FDS are 550 and 520 kg/m^3, respectively, whereas the solid densities of the DS and FDS are 1410 and 1390 kg/m^3 (Eisenbies et al., 2019). Some other conventional biomasses have less density. Table 11.2 shows the comparison of DS and FDS densities with other biomasses (Marrugo et al., 2019).
- Calorific value: Calorific value is a very important characteristic of any biomass because it shows the thermochemical conversion capacity. This value directly influences the feedstock thermochemical conversion calculations (Dogru et al., 2002). The average calorific values of FDS and DS are 12.26 and 11.45 MJ/kg, respectively. Relatively with other biomasses, SS has less calorific value as shown in Table 11.2.
- Residual energy: Residual energy is a factor that can indicate the thermochemical conversion capacity of any biomass. Higher residual energy in biomass indicates a high value of recoverable energy contents. Moisture content in biomass has a huge impact on energy recovery or residual energy. DS and FDS have positive residual energy ranging from 0.24 to 12.44 MJ/kg, and RS has negative residual energy ranging from 0.89 to 1.91 MJ/kg (Chan and Wang, 2016). Another study by (Eisenbies et al., 2019) showed that RS has 1.87 MJ/kg negative residual energy. On the contrary, FDS and DS samples resulted in net positive residual energy values of 10.53 and 8.73 MJ/kg, respectively. The difference among the values of residual energy of RS, DS, and FDS shows that a reduction in moisture content increases the residual value. A reduction in moisture content below the critical level can increase by more than 80% residual energy.

11.2.2 Sewage Sludge Proximate Analysis

- Moisture Content: Moisture content is essential to any biomass we convert into an energy source. SS has a lot of water content in it due to its source of generation from wastewater. The average moisture content in the RS is more than 80% of the total weight (Eisenbies et al., 2019). Some studies reported that moisture content in the RS ranged from 93% to 98% of the total weight (Chan and Wang, 2016). The average moisture content in the DS ranged from 13% to 18% by the total weight. After drying at a high temperature of 105°C, it reduces its weight. After drying, the average moisture content of FDS reduces to 5% of the weight. High moisture content in the biomass decreases the calorific value. Moreover, it will require more energy to evaporate the extra water content from the biomass. Ultimately, it will decrease the thermal efficiency of the reactors in the gasification process (Motta et al., 2018). Some studies described that the ignition process in the gasification zone is delayed due to high moisture content. High moisture content in the biomass also decreases the reactors' temperature, resulting in more tar condensation (Piippo et al., 2018). It clearly shows that moisture content should be very low to get better performance.

- Volatile Matter: Volatile matter is measured after the heating process of the biomass. Due to high temperature, the matter converts from sludge to condensable or non-condensable vapors. Fixed carbon and ash content depend on the volatile matter of the biomass. High volatile matter content decreases NOx emissions, increases carbon burnout, and makes it easy to ignite (Nzeve and Ikubano, 2021). As previously discussed, moisture content will hugely impact the volatile matter. RS has a high moisture content, so it will have less volatile matter. The average volatile matter of the RS is almost 5% by weight. On the other hand, DS and FDS have high volatile matter due to less moisture content. The average volatile matter of FDS and DS is 50% and 45%, respectively. The volatile matter of these sludges (DS and FDS) is lower than the other biomasses but can be used as biomass for energy production through gasification.
- Fixed Carbon: Solid carbon residue left after so many processes in the gasification. It forms after heating and contains elemental carbon and carbonaceous residue in it. Less fixed carbon conversion shows that it has a low yield capacity. The average values of the fixed carbon in the RS, FDS, and DS are 4.5%, 7.46%, and 6.35%, respectively. Product yield is determined by the conversion value of fixed carbon (Qian et al., 2013).
- Ash Content: It is solid residue left out after biomass combustion in the gasification process. SS has higher ash content than the other conventional biomasses. The average ash content of FDS and DS is 37.69% and 34.95%, respectively. High ash content can cause slagging problems in the gasification reactor during the thermochemical process (Yao et al., 2020). Table 11.3 represents the whole proximate analysis of SS and other conventional biomasses.

11.2.3 Sewage Sludge Chemical Properties

- pH: pH value of any biomass reveals its nature as basic, acidic, or neutral. pH of the SS depends upon the source of wastewater and the WWTP from where it releases. The pH values of RS, FDS, and DS are 6.87, 5.23, and 5.81, respectively, showing SS's acidic nature (Raveendravarrier et al., 2020). SS with neutral nature supports the growth of microbes in the biomass, but acidic nature restricts the growth.
- Ultimate Analysis: Carbon is the dominant element in the ultimate analysis of SS. FDS and DS contain carbon and oxygen as major elements with low quantities of nitrogen, hydrogen, and sulfur. Mostly FDS contains more carbon, nitrogen, and hydrogen with less sulfur and oxygen than DS. SS contains less quantity of carbon and more quantity of sulfur than the other biomasses. High carbon value in the biomass enhances the heating value of the feedstock. Hydrogen-to-carbon and oxygen-to-carbon ratios also can be calculated by the ultimate analysis and give a better perspective regarding different gases

TABLE 11.3
Proximate Analysis of SS and Conventional Biomasses

Characteristics	RS	DS	Sugarcane Bagasse	Sawdust	Coconut Shell	Rice Husk	Coir Pith	Coffee Husk	Forestry Residues
Volatile matter (%)	12.10	45.50	81.30	71.23	62.96	63.30	70.10	77.09	34.5-80
Ash content (%)	1.10	30.60	3.10	0.98	5.86	10.62	3.70	3.55	1.4-3.2
Moisture content (%)	82.30	13.25	5.40	9.98	12.56	6.60	11.60	9.06	56.8
Fixed carbon (%)	4.50	6.35	10.20	17.21	18.63	14.26	14.60	19.36	17.8-25.4

TABLE 11.4
Ultimate Analysis of SS and Conventional Biomasses

Component	FDS	DS	Sugarcane Bagasse	Sawdust	Coconut Shell	Rice Husk	Coir Pith	Coffee Husk
H (%)	4.62	4.11	6.01	4.86	5.00	5.92	3.37	6.11
C (%)	25.32	21.56	44.55	51.60	42.25	39.29	39.29	44.76
O (%)	22.84	21.43	45.80	42.10	37.45	52.45	38.81	42.93
N (%)	3.21	2.89	0.39	1.38	0.83	1.33	0.19	2.57
S (%)	1.29	1.81	0.07	0.06	0.03	0.43	0.10	0.09

with heating values in the gasification (Ghassemi and Shahsavan-Markadeh, 2014). The carbon and oxygen percentage of DS are almost equal at 21.56 and 21.43, respectively. However, FDS has a little difference between carbon (25.32%) and oxygen (22.84%) values (Raveendravarrier et al., 2020). Table 11.4 shows the ultimate analysis of SS and other conventional biomasses.

- Energy Density: The amount of energy held by any given fuel mass is called energy density. The energy density of the DS has less value 4892.51 MJ/m^3 than the comparison FDS value of 5054.61 MJ/m^3. These values show that FDS generated the same amount of energy as DS with less storage volume. The high calorific value of FDS is the reason for better energy density than the DS.
- Fuel Value Index: The fuel value index (FVI) results from many factors such as moisture content, wood density, ash content, and calorific value of any feedstock. It is also a very important factor for any feedstock going into gasification. It shows the energy per volume of the feedstock, such as energy density. SS has very less values of FVI than the other conventional biomasses. The average values of FVI of DS and FDS are 10.66 and 26.66 MJ/m^3 (Raveendravarrier et al., 2020).

11.3 FUNDAMENTALS OF GASIFICATION PROCESS

11.3.1 Mechanism of Gasification Process

Gasification of SS involves the process of incomplete oxidation of sludge in a reducing atmosphere. This process converts the carbonaceous matter of the sludge into usable fuel. This thermal treatment of biomass happens in the presence of air or steam at temperatures ranging from 800 to 1400°C (Bonfiglioli et al., 2014)). The most suitable energy recovery system for sludge is gasification because of SS physical and chemical properties. In a recent gasification technique in Germany, DS is gasified through a fluidized bed gasifier with a temperature of 850–880°C and a residence time of 30 minutes. Germany installed the first gasification project of SS in 2002 with a capacity of 2700 Mg/a (Schnell et al., 2020).

The ultimate useable fuel or product of the gasification process is synthetic gas that is called syngas. Many thermochemical conversion processes can convert biomass into different fuels such as pyrolysis, incineration, and combustion. Gasification is the most promising technique among all available techniques due to its elasticity and variety of products. This technique should convert almost every biomass into syngas, mainly H_2, CO_2, CH_4, and CO (Chen et al., 2015). Gas turbines use syngas to produce electricity as the ultimate product.

Steam gasification is the most suitable gasification technique because it enhances the syngas production and heating value. It is indirect gasification because it does not require energy to start the gasification reactions. Recently, some other gasification techniques have been under investigation, such as catalytic gasification and co-gasification, to improve the productivity of syngas. Seggiani et al.,

(2012) stated that it is possible to co-gasify SS with solid waste and conventional biomasses. This study showed the possibility of co-gasification with SS (70%) and wood pellets (30%) in a fixed-bed gasification reactor.

Inorganic matter in the biomass disturbs syngas production because of slagging. Slagging is the conversion of inorganic matter into solid substances in the gasification process (Bonfiglioli et al., 2014). This process mainly uses the air as a gasifying agent, but some studies reported CO_2, O_2, and steam as gasifying agents (Gao et al., 2020). In gasification, total produced gas is not equal to total exhaust gas because sulfur transforms into H_2S, chloride into HCl, and nitrogen into NH_3. Furthermore, SO_2, dioxins, and NO_x prevent emissions into the environment with gas-cleaning techniques (Werle and Dudziak, 2013).

Gasification includes four steps, e.g., drying process, pyrolysis, oxidation, and reduction. All of the steps are briefly discussed in the following:

- Drying: The drying process removes the feedstock's moisture content at temperatures above 100°C. Energy is required in this step to dry out the biomass. There is no chemical reaction taking place in the drying process. It is the initial step of any gasification technique to remove moisture content for further processing.
- Pyrolysis: It is the second step of gasification in which the decomposition of biomass starts in the absence of oxygen or air (oxidant) (Bridgwater, 2003). There is also an increase in temperature in this step to vaporize the feedstock using primary reactions. Biomass composition and size play a role in the distribution of product gas at the end.
- Oxidation: In this step, oxidation reactions occur in the gasification process. At elevated temperatures, chemical reactions occur between oxidants and feedstocks and produce CO and water vapors. The chemical nature of biomass, oxidant type, and physical conditions of the gasification process mainly causes oxidation. This step results in a lot of heat due to exothermic reactions, and heat can be used to sustain further processes.
- Reduction: Reduction happens as the last step of the gasification process in the absence of oxygen. Chemical reactions in this step are very high-temperature and endothermic reactions. Products of oxidation reactions and char react and form new hydrocarbons. Ash and char are the by-products of reduction (Saghir et al., 2018). The gasification process occurs in different types of reactors called gasifiers.

11.3.2 Chemistry of Gasification Process

- Drying: In this step, water is removed from the feedstock (SS) by giving a temperature 70–200°C. The temperature range in the drying process depends upon the type of gasifier and feedstock (Wang and Stiegel, 2016). High moisture content in the feedstock reduces the efficiency of the gasification process, so drying the feedstock is an important step in gasification. The drying process is an endothermic reaction because it requires heat to start. Then he vaporizes the water content from the feedstock (Ram and Mondal, 2022). The chemical reaction of the drying step is given in the following:

$$\text{Moist feedstock} + \text{Heat} \rightarrow \text{Dry Feedstock} + H_2O \tag{11.1}$$

- Pyrolysis: Pyrolysis is the process of decomposition in which temperature rises from 200 to 450°C in the absence of air. The conversion rate of feedstock into volatiles and char depends upon the nature and properties of biomass (Lappas and Heracleous, 2016). Almost 70% of the feedstock is reduced in this step due to the release of volatiles [40]. Volatile compounds consist of carbon monoxide and methane. This process is endothermic because it takes heat to start. The chemical reaction of this step is given in the following:

$$\text{Dry Feedstock} + \text{Heat} \rightarrow \text{Volatiles} + \text{Char} \tag{11.2}$$

- Combustion: The next reaction is the combustion of feedstock in the presence of oxygen or air. This reaction is exothermic. This step produces a lot of heat, which helps crack tar hydrocarbons in the next reactions. The chemical reaction is given in the following:

$$C + O_2 \rightarrow CO_2 + \text{Heat} \tag{11.3}$$

- Gasification: The gasification step has a lot of redox reactions in it. There are three main reactions, e.g., water-gas, Boudard, and hydrogasification (Seo et al., 2018). In the water-gas reaction, carbon reacts with steam (H_2O) and produces carbon mono-oxides and H_2. In the Boudard reaction, carbon reacts with carbon dioxides and produces carbon mono-oxides. In the hydrogasification reaction, carbon reacts with H_2 and produces methane. At the carbon, mono-oxides react with water and produce syngas. Mainly syngas is a mixture of CO, H_2, CH_4, water vapors, and sulfur compounds. Karaca et al., (2015) reported the syngas gas composition from SS gasification such as H_2 (25%), CO (14%), CO_2 (27%), CH_4 (10%), and other compounds (24%). All the gasification process reactions are given in the following:

$$C + H_2O + \text{Heat} \rightarrow CO + H_2 \quad (\text{water} - \text{gas reaction}) \tag{11.4}$$

$$C + 2H_2O + \text{Heat} \rightarrow CO_2 + H_2 \tag{11.5}$$

$$C + CO_2 + \text{Heat} \rightarrow 2CO\ (\text{Boudard reaction}) \tag{11.6}$$

$$C + 2H_2 \rightarrow CH_4 + \text{Heat (Hydrogasification)} \tag{11.7}$$

$$CO + 3H_2 + \text{Heat} \rightarrow CH_4 + H_2O \tag{11.8}$$

$$CO + H_2O + \text{Heat} \rightarrow H_2 + CO_2 \tag{11.9}$$

11.4 GASIFICATION TECHNOLOGIES FOR SEWAGE SLUDGE

Gasification technologies have developed for decades according to required needs and desired production. As a result, numerous studies are focused on the possible advanced levels of gasifiers. Currently, gasifiers are designed and classified according to various parameters, including to process of gasification, method of heat supply, gasification agent used in the reactor, and type of reactor. As most reactors typically fall into fixed-bed and fluidized-bed reactors for SS gasification, rotary kiln and plasma reactors are also used. The selection of these reactors depends on conditions such as the characterization of feedstocks and desired application of the produced gas (Pio and Tarelho, 2021).

11.4.1 Existing Sewage Sludge Gasification Technologies

Gasification is a thermochemical conversion of highly organic content materials into H_2 and CO, known as "synthetic gas," with CO_2, CH_4, H_2O, and other hydrocarbons with a high temperature of 800–1000°C. Usually, it occurs when a mixture of gases such as carbon dioxide, oxygen, steam, and a mixture of these gases with air combined. Some previous studies revealed that gasification agents somehow influence the calorific value of syngas that are 4 and 12 MJ/Nm, with high heating values (HHVs) by oxy-gasification (Chanthakett et al., 2021).

The produced gas as a result of the gasification process can be used for heating purposes or power generation, or it can be further processed for liquid fuel synthesis. The primary inputs of the reactor are feedstock and heat source, while the outputs are stack emissions and ash. The process has similarities with the combustion process, with some exceptions such as moisture tolerance

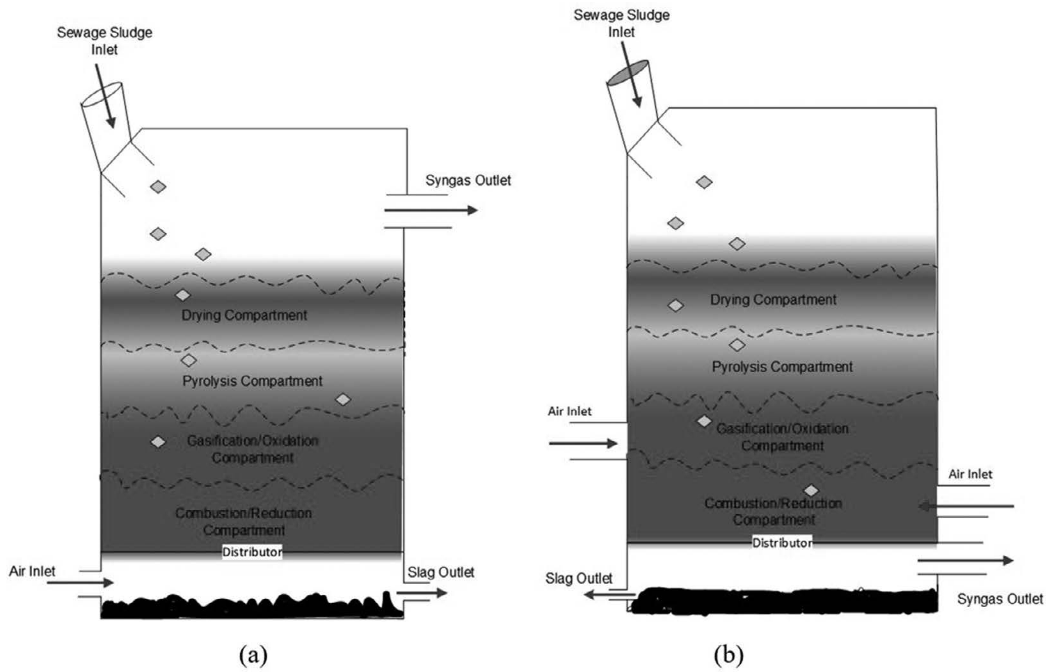

FIGURE 11.1 (a) Updraft fixed bed reactor and (b) downdraft fixed reactor.

restrictions of reactor (<15 wt%) and lake of stoichiometric oxidants essential for complete combustion. The ash generated by this process is usually disposed to the landfill site or used for agriculture and construction applications, as per chemical characterizations, qualities, and heavy metal concentration (Đurđević et al., 2019). There are four sub-stages of the gasification reaction. Initially, the appropriate sample with low moisture content of about 15 wt% was dried at a temperature of 70–200°C. The devolatilization stage, where the temperature increases to 350–600°C, volatile oxidation, and char gasification. Figure 11.1 shows the complete gasification. Herein, the delayed pyrolysis process or incomplete combustion occurs because producing more gaseous yield requires gas-to-solid, gas-to-gas, and liquid cracking processes (Oladejo et al., 2019). Two main gasifier technologies used for SS gasification, which is working on basic principles of gasification, are fixed bed and fluidized bed. The fixed bed gasifier involves the flow of gasifying agent over the feedstock while the raw material is dried, pyrolysis, and continually gasified over time. This process leads to an efficiency of reduction and shorter residence time to feedstock compared to fluidized bed gasifier that allows the feedstock less time to pass the sub-stages of the gasification process (Santos et al., 2023).

The following sections introduce several technologies available worldwide.

11.4.2 Fixed-Bed Gasifier

Fixed-bed gasifier is simple gasification technology. It is usually made up of cylindrical-shaped steel. Fixed-bed gasifier is used for different gasifying agents, including SS, with a high pressure of 900–1800s (Materazzi et al., 2013). The feedstock is introduced from the top of the reactor to the bottom, where the temperature is less than the lowest area of the reactor. Then feedstock moves from the top to the bottom of the reactor through the fixed bed. The bottom of the reactor is the high-temperature area that contains oxygen and steam to synthesize the fuel gas. Fresh feedstock, after reaching bottom, starts to devolatilize. Products after devolatilization are phenols, methane, tars, and oil. These light hydrocarbon gases leave the top of the reactor with syngas. In the bottom zone, oxygen and steam enhance the production of syngas. Temperature increases from top to bottom throughout the bed, but the bottom zone maintains the temperature to reduce slag formation (Wang and Stiegel, 2016). The process of gasification and high carbon conversion from the raw materials continues over time at high pressure between 1 and 200 bar, and approximately temperature reaches 500–1200°C in fixed-bed gasifier (Chanthakett et al., 2021).

Fixed bed gasifiers are also of two types: (1) updraft gasifiers and (2) downdraft gasifiers as shown in Figure 11.1. When feedstock is supplied from the top and air from the bottom of the reactor into gasification chamber, this gasifier is called an updraft gasifier or counter-current gasifier. In updraft gasifiers, product gas diffuses from the top of the reactor. In the downdraft gasifier, the feedstock is supplied from the top and air from the top of the reactor. As a result, product gas (syngas) comes out from the bottom of the chamber. These types of gasifiers are called downdraft or co-current gasifiers. The combustion zone of both types of gasifiers is at the bottom. Downdraft produces less tar than the updraft due to the release point of syngas at the bottom and near the combustion zone. Syngas with less tar content is preferable in turbines and gas engines (Ram and Mondal, 2022).

Previous studies show that the fixed-bed reactor can be used for different waste conversions to produce a high carbon conversion rate and less ash emission. However, the requirement of low moisture content in feedstock is one of the limitations of fixed-bed reactor, which is why it is usually not applicable to large scales (Mazaheri et al., 2019).

11.4.3 Fluidized Bed Gasifier

In the fluidized bed reactor, the SS was added with sand and injected into the fluidized medium at approximately 800–900°C. These reactors are usually made of heat-tolerant steel and can maintain a high range of temperature between 700 and 1000°C (Chanthakett et al., 2021). In this gasifier, gasifying agents behave as fluidizing medium, and feedstock is introduced into fluidized bed. The temperature profile of the gasification chamber is isothermal. In this gasifier, ash fusion temperature is higher than the bed temperature. Of this temperature difference, cyclones remove ash particles from the top of the bed. In some fluidized bed gasifiers, the biomass conversion rate is very less due to low temperatures. Some light hydrocarbons and tars also produce syngas from fluidized bed gasifiers (Wang and Stiegel, 2016). Similar to fixed-bed reactor, the fluidized bed reactor also has two configurations: the bubbling and circulating fluidized bed reactor.

Bubbling fluidized bed gasifiers are used in low gas speed conditions between 1 and 3 m/s and functionate at temperatures of 800 and 1000°C. The fly ash is separated by a cyclone and settles down to the bottom, and the resultant raw syngas transfer to the next portion of the reactor (Ruiz et al., 2013).

The circulating fluidized bed reactor is conducted in two steps: the first one is a repetition of the same bubbling process where the bubbling fluidized bed reacts with the feedstock and produces the syngas. The second step, with the help of high-speed gas, usually between 3 and 10 m/s, moves the feedstock to the next stage. Finally, the solid particles allow separation by the cyclone and the circulated fluidized bed reactor. This technology is widely used for solid waste gasification, including SS, due to its high efficiency (Chanthakett et al., 2021). A fluidized bed gasifier is very efficient reactor for highly exothermic reactions. It offers high heat exchange and high-temperature control efficiency due to continuous gas flow in the fluidized bed. This reactor can bear pressure of high velocities gases and does not drop pressure which is problem of fixed bed gasifiers. Moreover, it decreases the cost due to easy construction, less heat exchange area in the reactor. However, fluidized bed has some disadvantages, such as difficulty separating catalyst from exhaust gas and complexity in operation (Lappas and Heracleous, 2016). Energy production efficiency is the current most significant gasification application that identifies the good performance of gasifier indicators and waste type potential.

11.4.4 Rotary Kilns Reactor

The rotary kiln reactor is highly used for converting high carbon-content raw material into syngas and has wide applications in large-scale SS gasification. This reactor is cylindrical shape chamber made up of steal and operates at a temperature of approximately 300–600°C (Freda et al., 2018). The cylindrical chamber is movable when it operates, it moves downward inclination and relatively moves to the existing; hence, the feedstock passes along the chamber through the reactor for gasification. In rotary kilns, SS introduces from the top of the reactor, while the gasifying agent is from bottom of the reactor (Molino et al., 2013).

11.4.5 Plasma Gasification

Plasma gasification is an advanced technology that uses electrically ionized gas to break the SS into syngas. Plasma reactors are used in this gasification technology shown in Figure 11.2. These reactors initially produce electrically ionized gas at about 10,000°C and use plasma torches with pressure between 1 and 3 bar to divide SS into syngas. In these reactors, the feedstock is also introduced from the top of the plasma reactor, while the gasifying agent is from the side of the reactor (Molino et al., 2013). Through plasma torches, organic materials convert into syngas, while the inorganic part of waste converts into residues such as inert and glazed slag. The addition of different parameters in the reactor can increase the operational cost. In addition, the plasma reactor system requires more electricity to operate, approximately between 1200 and 2500 MJ/ton of raw material. Due to high maintenance and operational cost, these reactors are significantly less feasible from a commercialization point of view (Arena, 2012). Nevertheless, plasma torch systems can be deployed for waste-to-energy (WtE) pilot plants due to economic and technical challenges.

11.4.6 Supercritical Water Technology

Supercritical water technology is used to convert SS into syngas with high content of H_2 and CO_2 as a clean and efficient method. The carbon dioxide collected from this technique can be used further for producing hydrocarbon fuel. This gasification technology has advantages such as a high reaction rate, more gas generation, and clean syngas production. Figure 11.3 represents the supercritical technology. The supercritical water gasification of SS is usually carried out in a continuous-mode reactor. The reactor is made up of steel tube (SS316) type with different diameters and lengths. The reactor is fixed and placed in an electrical furnace. Initially, water is introduced into the reactor, and pressure is adjusted, usually at 25 MPa, through back-pressure regulator. When pressure is reached at 25 MPa, the temperature is set according to the requirement. Normally SS feeds into the reactor by a feeding system with 400 rpm agitation speed and 1.3–15 mL/min flow rate. The density of water changes continuously with the fluctuation of temperature. The feedstock is fed into the reactor after confirmation of desired conditions and experimental values of reactor. To gain the desired conditions and experimental values, the reactor usually operates 1 hour before the sample collection (Amrullah and Matsumura, 2018).

11.5 CURRENT PRACTICES OF SEWAGE SLUDGE TREATMENT WORLDWIDE

SS produced from WWTP contains high moisture content, almost 98%, posing major issues for its usage in many areas; therefore, large-volume treatment is required. For example, less than 60% moisture content is required for agricultural applications, whereas less than 50% is desired

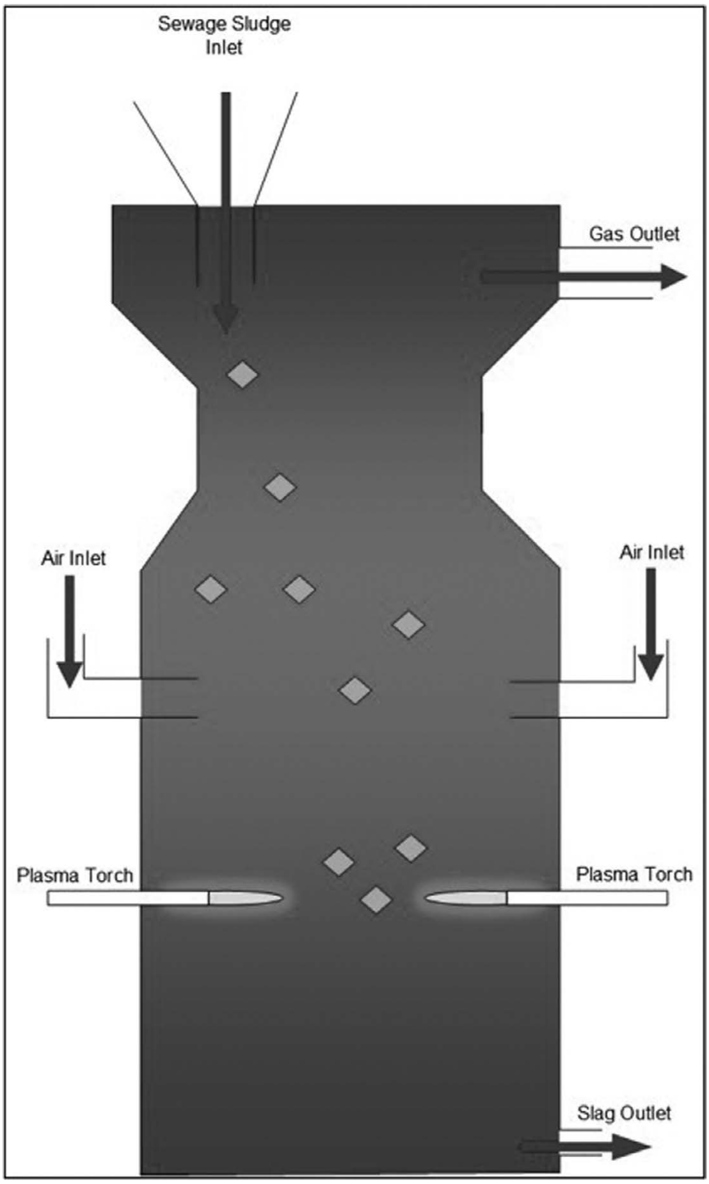

FIGURE 11.2 Schematic diagram of plasma gasifier.

for thermal processing (Zhang et al., 2018). The combined amount of SS generated in China, USA, and Europe is 240 MT/year (Wang et al., 2017). In China, during 2007–2013, the growth rate of SS was 13%, with 6.25 MT production in 2013, of which only 25% was properly treated. While, in 2015, the SS production in China and Taiwan is 30–40 MT and 77,000 T, respectively. Whereas, in 2015, five European countries, including UK, Spain, France, Italy, and Germany, produced approximately 75% of the total SS of Europe (Kacprzak et al., 2017). In 2002, in Germany, 10 MT of SS was produced, approximately 3.5 times the volume of Pyramid Giza (Kokalj and Samec, 2014).

The increased SS production in Europe is associated with increased WWT and due to the practical implementation of the Directive on the urban WW (91/271EEC), indicators, and national legislative requirements. In Europe, there are many SS gasification plants; the most

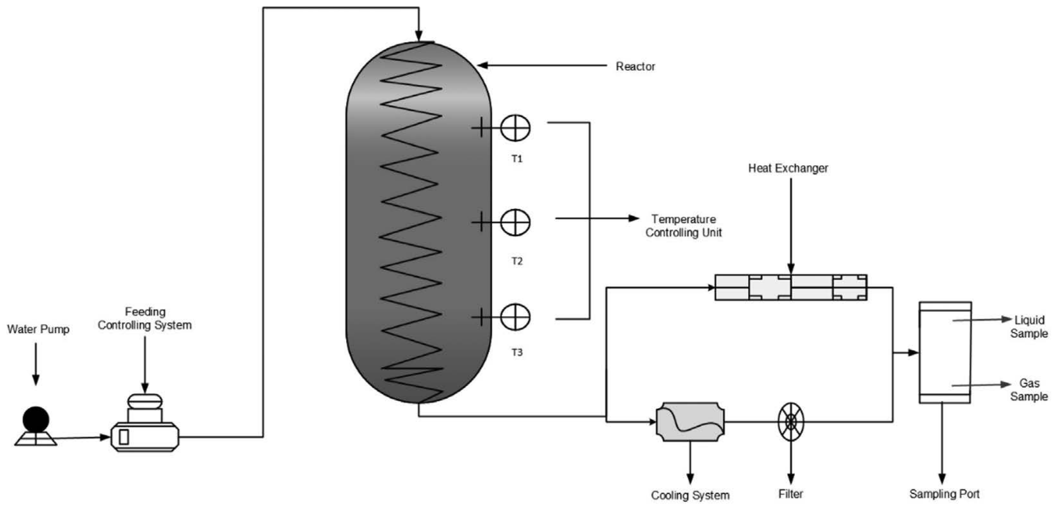

FIGURE 11.3 Experimental setup of supercritical water gasification.

famous one is in Austria, built in 2001 with 8 MW power; some other examples include plants in Kokemaki (Finland), Skive (Denmark), and Spiez (Switzerland) (Uchman and Werle, 2016). In Germany, there are two SS gasification plants. The first demonstration of plant was constructed in 2002 in Balingen with 1100 ton/day flow. After many years of experiments, it was built with 1950 ton/day flow. The SS was supplied from six nearby localities with 25,000 inhabitants. The installed power of that plant was 720 kW heat energy. The second plant was built in 2010 in Manheim with 5000 ton/day flow. The system had about 600,000 inhabitants, and the installed power generation capacity was 2.2 MW of heat energy. Both plants are based on two-step gasification process where SS with 18%–95% dry content was conveyed via silo with limestone to the thermolysis screw feeder. In the first step, thermolysis gas, products, ash, and carbon are conveyed to fluidized bed gasifier. In the second step of SS gasification, carbon is converted into gas under sub-stoichiometric conditions, and the long chain molecules of tar are cracked. After gas cleaning, the produced syngas is used to run the gas engine to generate CHP (combined heat and power), whereas the surplus gas can generate heat (Wang et al., 2017).

In 2013, the highest SS treatment rate was recorded in Malta 100%, followed by the UK, the Netherlands, Luxembourg, Spain, and Germany, with 99.5%, 99.4%, 98.2%, 97.8%, and 96.4% SS treatment rates, respectively. Besides, in Croatia, Romania, Albania, Turkey, Bosnia, Serbia, Kosovo, and Herzegovina, less than 2% of WWT is being carried out. In Portugal, Italy, Ireland, Spain, and Luxembourg, approximately three-quarters of SS is used as a fertilizer, whereas, in Finland and Lithuania, at least two-third is used as a compost (Blagojević et al., 2017). As an alternative to SS disposal or used as a fertilizer, many researchers focus on the potential of SS gasification. Many countries, including China, Poland, and Slovakia, implement the thermal methods of SS conversion. The SS treatment in different countries is mainly due to the respective legal regulations. In Germany, the German SS Ordinance provides legal framework for SS disposal. In Switzerland, SS use in agriculture was completely banned in 2003. The EU Directive 86/278/EEC sets limits for heavy metals regarding the use of SS in agricultural activities. Currently, some countries, including Germany, Belgium, and the Netherlands, have introduced much strictest regulations as the directive are now 30 years old. In the Netherlands, the limit values are strict, and the SS agricultural use is excluded altogether. Whereas, in Belgium, the heavy metal limits for Cu, Ni, and Pb are even stricter than in Germany (Hudcová et al., 2019).

By implementing such limit values in other countries, an increasing trend of thermal treatment of SS can be expected.

11.5.1 Sewage Sludge Treatment in Italy: A Case Study

In Campania region, Southern Italy fluidized bed gasification plant is installed for the treatment of SS, generated from a WWTP situated at the nearby locality. Two different samples of SS, produced at different seasons of the year, were taken and subjected to gasification at 850°C, but at different equivalence ratios (ERs). The two SS samples, SS-A and SS-B, were taken in January 2019 and April 2019, respectively. The gasification agent used was nitrogen/air at a varying rate of oxygen/fuel ER (0.1:0.2). The gasifier consists of a fluidization column (diameter 41 mm, height 1000 m) and a distribution plate (diameter 41 mm, height 600 m) that are made of stainless steel. The distribution plate separates the inlet gas chamber and acts as a gas preheater. The ceramic filter is installed at the reactor's bottom to collect fly ash. The feeding system consists of mechanical and pneumatic conveying devices and is located at the bottom of the reactor. Nitrogen steam is used as transport gas in the feeding system, whereas the remaining gasifying agent is air that introduces at a flow rate of 50 NL/h. It creates the fluidizing environment in the reactor with 3% oxygen and 0.3 m/s fluidization velocity. Silica sand (0.2:0.3 mm) is used inside the bed reactor with 180 g bed inventory. The elemental composition was determined using LECO SC-144DR and LECO CHN628 analyzer according to UNI 7584 and ASTM D5373 standards. While chlorine content was determined using 883 Basic IC plus ion chromatograph (Migliaccio et al., 2021).

The composition of syngas produced from both SS samples was determined using computer-aided simulations, particularly the impact of seasonal changes in the chemical composition of the sludge on the produced syngas. After calibration, the composition of the syngas varies with changing the gasification temperature, 700–900°C, and ER 0.1–0.5 for both SS samples. Aspen plus software with Fortran modules were used as in the previous studies (Abdelrahim et al., 2019). Ashes from SS gasification can be entered into the circular economy after proper management. It can be used in building applications, treated for phosphorous recovery, or as adsorbent material to remove hot H_2S (Gil-Lalaguna et al., 2015; Parés-Viader et al., 2017). The higher ER and gasification temperature, the lower is the tar production, which causes clogging and fouling in the pipes and filters. However, the gasification efficiency and syngas heating value decreased with increased ER value. Thereby, to have good gasification performance and low tar production, high temperature, and low ER were considered in this study. The operating conditions of the gasifier were selected based on previous research (Abdelrahim et al., 2019; Brachi et al., 2014). At 10% ER, small amount of unreacted carbon was present in the gasification ashes. The concentration of heavy metals in the SS gasification varies with ER values. Moreover, under a reducing environment, the bottom ash exhibits a relatively higher surface area (32 m^2/g), which determines this factor's strong dependence on the oxygen present in the reactor. Furthermore, the presence of phosphorous in the ashes under the current operating conditions depicts gasification as the best alternative compared to incineration with respect to phosphorous separation. Therefore, only a single-stage alkaline extraction process is required for further ash treatment. The model used in the study could not change the tar production and ash composition, therefore requiring additional efforts. The impact of operating conditions of SS gasification on heavy metals is still debatable. The results of other studies showed that heavy metal partitioning depends on raw material properties besides operating conditions.

11.5.2 Sewage Sludge Gasification Plant in Poland: A Case Study

Poland still progresses in the SS gasification technology, despite its ranking in fifth position after Czech Republic, Germany, the Netherlands, and Greece, regarding biomass use in the energy sector (Banja et al., 2019). Since biomass gasification is a promising technology, therefore, it is optimistic to look at the SS gasification. Furthermore, according to the 88 Resolution, Council of Members of August 11, 2016 in National Waste Management Plant 2022, the assumed mass of municipal SS in 2020 would be 750,000 Mg. Therefore, gasification is an eminent technology that ensures sterilization of SS, recovery of valuables, and effective mass reduction. The fixed bed gasification reactor was used in this case study. The recovery of phosphorous and phenol adsorption process was also analyzed. The key element of the gasifier is a stainless-steel gasifier pipe (diameter 150 mm, height 300 mm). The granular SS was introduced from the fuel box located at the top of the reactor, and the pressure fan injected the gasifying agent at the bottom of the reactor. For temperature measurements, 6 N-type thermocouples were installed with the vertical axis reactor and connected to the temperature recording system. Besides the reactor's temperature, the temperature of the gas released from reactor was also determined. Flow meters in the reactor measured the flow rate of gas and gasifying agent. The gas is captured through gas pipeline and cleaned by gas cleaning system. It consists of cyclone, scrubber, and drop separator. Analyzers determined the components of gasification gas. Two SS samples, SS1 and SS2, were taken from Polish WWTP that is operated in mechanical biological and mechanical biological chemical systems. Both systems consist of dewatering stage, anaerobic digestion, stabilization, and mechanical drying. Adsorption process was conducted in static environment. The study's objective was to determine the efficiency of phenol adsorption on solid waste generated from SS gasification technique and compare the obtained values with other materials. Phenol was used as an adsorbate in this study (Werle and Sobek, 2019). In the gasified material, sulfur, nitrogen, and chlorides are transformed into H_2S, NH, and HCl. Moreover, SO_2, NO_x, and dioxin formation are prevented. The gas cleaning system is small and less expansive compared to combustion (Werle, 2013). Therefore, the installation of gasification process is very effective, particularly on the WWTP site.

Temperature and air ratio in the studied process greatly impacted process gas composition. The quantity of CO, CH_4, and H_2 was increased with increasing the oxygen content and temperature of gasifying agent. Higher C and H values in SS1 and SS2 gasified fuel increase the gasification gas's LHV (lower heating value). LHV reaches its maximum value at an air ratio of 0.18 for both SS samples. Similar results were obtained in fixed bed gasifier from other studies as well, in which released gas consisted of volume fractions such as CO (10%), H_2 (5%), and CH_4 (1%) (Kim et al., 2016). Ayol et al., (2019) also presented similar results, small amount of CH_4 with 1.2% volume fraction was generated from SS fixed bed gasification. Above this value of air ratio, the process could change from gasification to combustion only. The LHV of released gas from gasification is comparable to popular gaseous fuels, 5 MJ/m^3_n. Unfortunately, it is less than CH_4 or H_2 but comparable to blast furnace gas. The results indicate that the gasification gas can be used as a fuel in power systems. Gasification was performed in a bench scale rotary kiln, with air ratio of 0.15–0.24 and at 800–850°C temperature. The resulting dry gas produced has HHV of 6–9 MJ/m^3_n and 4–6 g/m^3_n tar content [51]. Choi et al. (Choi et al., 2018) performed air gasification of SS in fluidized bed gasifier, resulting in HHV of 5 MJ/m^3_n. The results of all the discussed studies showed that besides temperature, the average value of processed gas depends on air ratio, the optimum value of this could yield an HHV of gas.

11.6 CHALLENGES AND OPPORTUNITES

11.6.1 Gasification Process Limitation

The composition of syngas' combustible elements, including H_2 and CO, defines its LHV and depends on the quantity of air supplied to the gasifier. At the air ratio of 0.18, the LHV reaches its maximum value resulting in gasification process, which produces combustible gases instead of complete combustion that mainly results in CO_2 production (Werle, 2016).

The water content in the SS, which is to be treated in gasifier, should be 10%–20%. In the gasification process, both digested and undigested sludge could be treated, but undigested sludge is preferred compared to digested sludge as it produces syngas with higher energy content. The gasification plant scale depends on gasifier type and ranges from 5 to 20 kW for small-scale downdraft fixed bed gasifiers. According to the requirement, fluidized bed gasifiers range up to 100 MW (EPA, 2012). The economic feasibility of the plant depends on electricity tariffs, and the statistics showed that at a plant capacity of 0.093 m^3/s, the SS gasification plant becomes economically feasible (Lumley et al., 2014).

Gasification processes are mostly carried out at a lab scale, only a few processes are working on a commercial scale due to high capital and production investment. Researchers are working on different techniques to improve the yield and products, such as palletization combinations of different materials blends. For commercial applications of gasification process, co-conversion of waste blends is already designed, which seems practical approach in implementation of the process. It is a proven fact that high amount of syngas can be produced by applying different techniques in the gasification process. Besides, some features, including tar production, $CO + H_2$ quality, and elimination of carbon footprints, should be carefully addressed before implementing gaseous elements into valued fuels (Hameed et al., 2021).

11.6.2 Use of Life Cycle Assessment (LCA) in Waste to Gasification

A sustainable solution assesses the environmental feasibility of a process by performing life cycle assessment (LCA), which consists of different impact categories, including global warming, eutrophication, acidification, resources, and ozone depletion. It is a systematic tool to assess the environmental performance of a process, product, or service. Further, economic feasibility can be checked by life cycle cost analysis. One of the advantages of LCA is the assessment of environmental impacts of WtE generation technologies; therefore, LCA is a decision-making tool that helps in selecting the best sustainable approach that can reduce the risks of making incorrect decisions (Hameed et al., 2021). It is very important to determine the system boundary of a process for analyzing the impacts. Moreover, a cradle-to-grave LCA can provide a complete assessment of the total energy (Corominas et al., 2013). However, most LCA-based studies have focused on the entire WWT operation rather than the sludge treatment.

Ramachandran et al., (2017) have proposed an SS and woody biomass co-gasification process for Singapore in which the Greenhouse gases emissions of the proposed system and the existing system were determined by performing LCA. The results showed that the proposed system provides a net emission reduction of 137–164.1 kT of CO_2eq. The reasons include carbon sequestration in biochar, an increase in energy recovery, and the avoidance of additional fuel for SS incineration. Moreover, the net electricity production from SS and biomass gasification increased by 3%–24%, leading to the energy recovery of 12.1–74.8 GWh/year. Whereas 34 kT/year biochar could also be produced. Besides, decentralization-reduced direct transport emissions, total transportation kg-km, and road kg-km are driven by 38%, 43%, and 42%, respectively. This depicts the lower number of trucks on roads, ultimately reducing the traffic. As a result, the annual decrease in kg-km driven by the proposed system is 4.23 MT-km.

11.6.3 Sludge to Energy: A Circular Economy Concept

The concept of circular economy is not a novel one, rather than it was introduced in 1989 by British economists, whereas it was described better after 20 years (Potocnik, 2013). It is defined as a closed loop of material flow, efficient energy and resource usage, waste reuse, and waste deposition prevention. SS produced from WWTP has huge potential for energy recovery and must be recycled under circular economy perspective. This results in legal requirements to prohibit SS storage, introduce sustainable development principles, and raise awareness among the masses for the usefulness and potential of SS as an energy source. Currently, the circular economy concept is important to the EU agenda, and many countries embrace it by effectively using SS (Werle and Sobek, 2019).

EU has developed a plan for circular economy, in which four key areas are defined. For SS, only two areas of the plan are focused: waste management and secondary raw materials (EC, 2015). Both these factors play an important role in circular economy. System thinking is very important when considering the efficiency of various SS treatment methods. The waste transformation into secondary raw materials helps achieve a circular economy where waste generation is minimized. Another advantage of waste using is the prevention of harmful pollutants, particularly in SS, associated with various environmental and health risks, including the production of pathogenic substances, endocrine disruptors, and bioaugmentation of heavy metals in living organisms (Tsybina and Wuensch, 2018). The syngas produced from SS gasification could be combined with other fuels, completely or partially reducing the burden on fossil fuels, contributing to a circular economy.

SO_2 and NO_x cleaning account for 2.9%–3.85% of total operating costs. However, mixing SS with MSW and recycling flue gas for SS drying offers economic benefits of 199.23 CYN/d. A study showed that gasification of 2.1 M gallons of raw SS with an integrated power system increased the profit to 3.5 M$ for 20-year lifespan (Chen et al., 2019). Comparing the CHP installation with a mixture of fuels, wood and gasification of SS, it was determined that among all the mentioned processes, SS was the only profitable process that had a breakeven point within 6 years, with 7% (12,475 €) and 7.52% of net present value (NPV) and internal rate of return (IRR) in case of selling electricity, respectively. If the plant is energy self-consumed, using the electricity produced from the process, NPV and IRR were increased to 206,925 € and 14.78%, respectively, and then the breakeven length decreased to 9 years. The cost of fuel contributes significantly to the total cost of the process; therefore, SS is more cost-efficient than other processes due to less fuel price (Lumley et al., 2014). The integration of CHP with SS gasification performs better with 6.8 years and 1337 €/year payback (PB) and NPV, respectively, than the integration of CHP with conventional processes such as natural gas. Approximately 340 € are required to treat 1 ton of undigested SS by gasification integrated with CHP, with 0.072 €/kWh electricity price, slightly higher than conventional methods (Di-Fraia et al., 2016). Lumley et al., (2014) reported the gasification of SS and determined the economic benefit of approximately 3.5 M$, compared to landfilling. SS management and treatment is a cost-intensive process, accounting for 50% of WWT cost. Therefore, it is important to analyze the techno-economic feasibility of SS treatment in the long term with its potential application in WtE and circular economy concept (Gherghel et al., 2019).

11.6.4 Techno-economic Barriers

Thermal treatment methods, particularly gasification, have many advantages over other treatments, but some technical and economic barriers need to be addressed. For example, high moisture content in SS decreases the temperature and thermal efficiency of the gasification process. Moreover, it also increases the cost of pretreatment as it requires mechanical dewatering and thermal drying that need high power consumption. Therefore, the selection of SS drying method depends on moisture content and operational costs. There are different types of drying, including mechanical, thermal, natural, and bio drying (Zhang et al., 2018). Solar energy can perform natural drying, whereas thermal drying is more efficient than mechanical drying. Bio drying is comparatively slow and may take many days to lessen the moisture content. Another important parameter in the SS ash is its heavy metal content, a sustainable solution for heavy metal from ash should be figured out (Gao et al., 2020)

In SS gasification, both organic and inorganic contamination are important; thereby novel techniques are continuously proposed. For example, in the raw and treated SS, organic and inorganic contaminants in the form of polycyclic aromatic hydrocarbons (PAHs) and heavy metals were determined, respectively. In SS, ash heavy metals were identified. To determine the contaminants in the SS gasification by-products, some methods, such as absorption spectrometry, gas chromatography, ecotoxicological analysis, and photoacoustic spectrometry, can be used (Werle, 2013). Large amount of CO_2 is released into the environment. To reduce operating cost and CO_2 emissions, different methods have been suggested in the literature, including heat recovery strategies and solar energy applications should be combined with SS treatment operations (Rosiek, 2020). Solar drying, O_2 purification, organic Rankine cycle, and wet carbon capture process needed to be modeled, and economic assessment should be performed to determine their efficiency as they still consume a lot of energy.

The syngas produced from SS gasification consists of CO and H_2 and also of H_2O, CH_4, and tar. The tar produced may cause many problems, such as blockage of pipelines, fuel injector nozzles, and valves. This issue needed to be solved. Moreover, catalyst is a considerable solution for tar removal; it initiates the cracking reaction in tar and destroys it, thus preventing clogging problems and increasing the combustible content of syngas. It can also increase the technological readiness level (TRL) value from 6 to 9; high TRL value indicates that it would be suitable for investors to invest in SS gasification process. Moreover, SS contains more sulfur and nitrogen pollutants, and the catalysts can reduce NO_x and SO_x emissions. Therefore, catalytic gasification is a promising technology for alleviating environmental pollutants. It is a less complicated and cost-effective method for tar removal. However, the presence of chlorine, sulfur, and blockage of pores because of fly ash can deactivate the catalyst, decreasing the reaction rate (Gao et al., 2020). Studies on supercritical steam gasification have been conducted in which H_2 rich syngas with high LHV can be produced. Furthermore, catalyst modification could be performed to acclimatize the extreme working conditions, pressure, temperature, agglomeration, and mechanical destruction. Sometimes, heavy metals act as heat exchange media or catalysts, affecting the formation of syngas that needs further investigation (Kamyab et al., 2022). Studies have been conducted to prove SS's co-gasification with biomass and waste. A study was performed by Vonk et al., (2019) on the co-gasification of SS with wood, SRF, plastics, and waste tires. A mixture of 20% dried SS and 80% wood resulted in good gasifier performance but with lower hydrogen content. This could be due to high iron in the SS, approximately 7.7%. Thomsen et al., (2017) performed a work showing that co-gasification and low-temperature circulating fluidized bed gasification is an effective method for the management of SS.

However, plugging, corrosion, and high operation cost make the commercial application of hydrothermal gasification inactive. Water can be oxidizing, reducing, acidic, or basic in hydrothermal gasification at high temperatures and pressure. The reactor materials are usually made of stainless steel and nickel-based alloys. During supercritical water gasification, the constituents of reactor wall, Ni, Cr, and Mo can be detected in the solid phase due to corrosion. Moreover, in the hydrothermal environment, the heteroatoms in SS also form corrosive acids. Hastelloy reactor is highly corrosion resistant, but it is uneconomical. Therefore, both processes-based and equipment-based approaches should be applied, including corrosion-resistant materials, chemical control, and mechanical design for corrosion control. In addition, the precipitated salts in the reactor can cause plugging issues. These issues must be addressed to make this economically and practically applicable. To overcome these issues, a two-stage hydrothermal gasification under mild temperature and pressure could be used (He et al., 2014). In SS management, there is a dire need for novelty in process intensification, resource recovery, pretreatment, sludge valorization, energy recovery, and costs.

11.6.5 Nutrient Recovery Potential

For phosphorous recovery, an amendment was made in German SS Ordinance, and the important changes include the following: it is obligatory to recover phosphorous from SS of municipal WWTP above 20 g/kg DM concentration. WWTP with less than 50,000 PE capacity can recover phosphorous with direct agricultural application. Whereas, for WWTP above 50,000 PE capacity, at least 50% and 80% of phosphorous must be recovered from SS and SS ash, respectively, irrespective of plant size. For WWTP with more than 10,000 PE capacity, a transitional period till 2029 and for all other plants, 2032 was suggested (Hamawand et al., 2015). Besides treatment cost and energy conversion, nutrient recovery from the remaining ash of SS gasification is also important in the circular economy approach. The ash contains a considerable amount of phosphorous 20.06% of the total weight, slightly lower than P_2O_5, which was 22.47% in the ash produced from SS combustion (Gorazda et al., 2018). Despite this, the content of micronutrients such as Fe, Cu, Zn, and Mn is different from natural sources of phosphorous; hence, their values should be controlled in accordance with Regulation EC no 2003. Acelas et al., (2014) reported 95% of phosphorous in dewatered SS gasification at 600°C in supercritical water. In EU, 5.5%–11% of the total phosphorous supplied to fertilizer production was recovered from SS ash. The phosphorous recovery from SS ash is an eco-friendly process, the phosphorous recovery cost calculated was approximately 1.5–4.5 €/kg, almost similar or less than the market price. However, the low quality makes it less market competitive (Jama-Rodzeńska et al., 2021).

Moreover, the ashes from SS gasification can be used as adsorbents to remove toxic substances such as phenols from wastewater. Firstly, the adsorbent obtained should be subjected to purification processes. The efficiency of the phenol adsorbent was higher than other adsorbents such as carbon, rice husk, or olive pomace. Furthermore, the solid fraction obtained after the gasification process is a source of phosphorous (20.06% P_2O_5), almost as high as 22.47% and 28.05% in SS ash and natural phosphate rocks, respectively. However, the technological parameters and chemical properties of the phosphate recovered differ from the natural phosphate. Therefore, it should be treated and managed separately (Werle and Sobek, 2019). Viader et al., (2015) showed that approximately 26% of phosphorous could be recovered from pure SS ashes using two-compartment electro dialytic setup, whereas 90% was recovered from a mixture of straw pallets and gasification of SS. Despite the progress in WWT methods, phosphorous recovery from wastewater streams still requires considerable attention. Nitrogen recovery is not possible as it is present in diluted form in syngas (Magrí et al., 2020). Moreover, slag produced from high-temperature SS gasification is non-leachable and non-hazardous, suitable to use in construction materials. In Balingen, Germany, the mineral granulates produced from slag at Kopf Gasification Plant were used for asphalt and construction materials (EPA, 2012).

11.7 CONCLUSIONS AND PERSPECTIVES

This chapter signifies the SS treatment and the technological advancements in gasification process. The conversion of SS into energy through gasification has great commercial potential and is a promising technique due to its elasticity and variety of products. SS gasification offers environmental and economic benefits than other thermochemical technologies such as combustion, incineration, and pyrolysis due to its high adoptability with SS as SS contains high moisture content, low calorific value, and much less residual energy, so every technique cannot work with this feedstock. This chapter also includes a detailed review of gasifiers, including fixed beds, fluidized beds, rotatory kilns, plasma gasifiers, and supercritical water technology. It also provides information about steam gasification, supercritical water technology, catalytic gasification, and co-gasification to improve the productivity of syngas. Case studies on SS treatment in Italy and Poland have been presented. The commercialization of SS gasification process is still at an early stage due to high capital and production investment. Researchers work on different techniques to improve the yield and products, such as palletization combinations of different materials blends. For commercial applications of gasification process, co-conversion of waste blends is already designed, which seems practical approach in implementation of the process. Although there are some challenges in the SS gasification process, such as high capital investment required for mechanical dewatering of SS, blockage of pipelines, fuel injector nozzles, and valves with tar, these issues can be solved by solar drying and catalytic gasification. Studies on supercritical steam gasification have been conducted in which H_2 rich syngas with high LHV can be produced. Furthermore, catalyst modification could be performed to acclimatize the extreme working conditions, pressure, temperature, agglomeration, and mechanical destruction. Nutrient recovery from the remaining ash of SS gasification is also of prime importance in the circular economy approach. Gasification of SS can effectively reduce the environmental impacts; therefore, a sustainable approach assesses the environmental feasibility by performing more LCA-based studies.

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Section III

Biological Processing of Sewage Sludge

12 Role of Microbes in Sewage Sludge Treatment and Management

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12.1 INTRODUCTION

In recent years, with the continuous progress of science, rapid socio-economic development, accelerated urbanization and growing population, the production of wastewater and its discharge are also showing an increasing trend. Meanwhile, with the continuous improvement of wastewater treatment capacity, the production of municipal sewage sludge is also increasing. Sewage sludge mainly comes from municipal wastewater treatment plants, which is a flocculent mixture of sediment, particulate matter and suspended matter produced in the process of treating domestic sewage (Kacprzak et al., 2017). As an inevitable product of the wastewater treatment process, sludge is a collection of microorganisms forming a colloid with their adsorbed organic and inorganic matter etc. The sludge is a collection of a large number of microorganisms and their adsorbed organic and inorganic substances. The treatment and management of sludge has become an increasingly important issue in municipal wastewater treatment. The sludge is a major environmental hazard. Untreated sludge is a major threat to the environment and it is increasingly important that it is disposed of properly. Sewage sludge from municipal wastewater treatment often contains organic matter that is difficult to degrade, pathogenic microorganisms, parasitic eggs and heavy metals such as arsenic and zinc (Ding et al., 2021). The composition of sewage sludge is extremely complex. It is extremely complex and will cause secondary pollution if not thoroughly controlled and effectively treated, posing a serious threat to the ecological environment and human health, as well as affecting the pace of social and economic development in towns and cities. It also affects the pace of social and economic development of the town. Therefore, how to safely, economically and efficiently treat municipal sludge, which is complex in composition and relatively large in volume, into a harmless resource that can be reused has become an urgent need and a major problem for the entire wastewater industry. This is a major challenge for the entire wastewater industry.

Advances in science and technology have led to the development of biological engineering, which in turn has led to an increasing interest in the use of microorganisms in sewage sludge treatment. Microorganisms themselves have a small form, a large surface area, a high reproductive capacity, a fast and strong metabolism, a large variety and number of degrading enzymes in their metabolites, a very large population and a wide distribution in nature, a high adaptability to the environment etc. Microbiological methods for sewage sludge treatment also have obvious advantages in terms of operating costs. The microorganisms are also distinguished by their ability to eat and decompose

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pollutants as food, the products of which are harmless and of great importance for the protection of the ecological environment. With the help of microorganisms, sewage sludge can be transformed into biofuels, bioplastics, fertilizers and many other useful products (Waldrop, 2021). The vast majority of pollutants in the sludge are also available to microorganisms, providing favourable conditions for microbes to survive and multiply, so they can reproduce quickly and can decompose the contaminants efficiently. Therefore, the use of microorganisms in the treatment of municipal sludge is widely used, and sludge is usually treated by means of various physical, chemical and biological techniques for thickening, conditioning, dewatering, stabilization and drying (Kelessidis and Stasinakis, 2012; Yu et al., 2013). Sludge disposal methods such as sanitary landfill, composting, agricultural use, thermal treatment, anaerobic digestion and the use of construction materials are also used to varying degrees (Hale et al., 2012; Yang et al., 2017).

Microbial-based treatment of pollutants is in fact the selection of suitable flora by using sewage sludge as a microbial culture, and then the creation of a miniaturized ecosystem in which substances are exchanged. By placing suitable microorganisms in the sewage sludge and creating an environment conducive to their growth, the microorganisms are able to use their own functions to break down the pollutants harmlessly and the sewage sludge is able to meet basic discharge standards. There are three main ways of using microorganisms to treat sewage and sludge, degradation, co-metabolism and detoxification. Degradation means that organic pollutants in sewage sludge can be degraded by bacteria, fungi and algae, for example aerobic gram-negative bacilli and cocci can effectively degrade organophosphorus pesticides and chlorobenzene (Kushkevych, 2021). The co-metabolic approach refers to the use of microorganisms to decompose organic matter, but not to use such matter as an energy source and component element; the detoxification approach refers to the various reactions carried out by microorganisms that result in a complete change in the molecular structure of the pollutant, thus effectively reducing the toxicity of the pollutant. It is important to note that there are many different types of microorganisms and that their role is therefore very complex. Some microorganisms can produce new pollutants while playing a purifying role, so when using microorganisms to purify pollutants, it is important to pay attention to the end products of the organic compounds' decomposition and to take necessary precautions for avoiding the emergence of new pollutants.

This chapter is mainly focusing on the role of microorganisms in municipal sewage sludge treatment and will specify the role of microorganisms in various technologies in relation to the processes and mechanisms.

12.2 SEWAGE SLUDGE TREATMENT TECHNOLOGY

12.2.1 Composting

Although the volume of sludge is much smaller than sewage, the construction of treatment facilities is relatively expensive, and the unreasonable treatment of sludge will not only occupy a large amount of land resources but also easily cause secondary pollution to the environment. Composting of sludge not only stabilizes and reduces the total volume of sludge but also enables the resource recovery from sewage sludge, so that the active ingredients in sludge can quickly enter the natural cycle. As a new biological sludge treatment technology, sludge composting is one of the most traditional and economical techniques for managing sewage sludge (Zhang et al., 2021). Composting is the use of biochemical reactions of microorganisms under the control of artificial participation to transform organic matter in sludge into fertilizer under specific conditions, in order to achieve an optimal disposal of sludge. Composting is the most traditional and economical process.

Sludge contains a large amount of nitrogen, phosphorus and potassium, which are needed for plant growth, and the organic humus in sludge is a soil conditioner, so composting sludge has good prospects for development. Sludge composting reduces the level of odour, kills pathogens, improves the properties of sludge and degrades many toxic and harmful substances in sludge. The composted sludge has a lower volatile content, less odour and significantly improved physical properties, such as reduced water volume, looseness, dispersion and granularity, making the compost product more suitable for use as a soil conditioner and plant nutrient source.

The research on sludge composting technology at home and abroad has been more mature; according to the different microbial growth environment, the composting process can be divided into two categories: aerobic composting and anaerobic composting, of which aerobic composting has a better overall effect and more mature technical methods and already has a certain degree of application. At present, aerobic composting is mainly used, and it is the main way to make organic fertilizer. Composting technology is generally referred to as aerobic composting unless otherwise specified.

12.2.1.1 Anaerobic Sludge Composting Technology

Anaerobic composting technology is the process of composting using anaerobic microorganisms to break down organic matter in sludge under conditions of lack of oxygen. There is no ventilation system in an anaerobic compost pile, so composting temperatures are low and decomposition and detoxification times are long. The essence of the reaction process is the transformation and stabilization of organic matter by anaerobic microorganisms in the absence of oxygen through three stages of liquefaction, acidic fermentation and alkaline fermentation of the sludge. The method is easy to operate, time and labour saving and is suitable for situations where fertilizer is not urgently required or where there is little labour. Under normal circumstances, anaerobic composting requires a turn of the pile about one month after the closure of the pile, which facilitates the multiplication of microorganisms and the decay of the pile.

12.2.1.2 Aerobic Sludge Composting Technology

Under aerobic conditions, aerobic fermentation is a process that uses the absorption and oxidative decomposition of microorganisms to convert part of the absorbed organic matter into inorganic matter through oxidation reactions. Most of the energy released from aerobic sludge composting is used for microbial growth and activities; while the organic matter that is not oxidized is synthesized into cytoplasm, providing favourable conditions for the reproduction of microorganisms (Liew et al., 2022).

Aerobic composting is a process whereby aerobic microorganisms, in full contact with air, cause a series of exothermic decomposition reactions in the organic matter in the compost material, ultimately converting the organic matter into simple and stable humus (Liew et al., 2022). The organic matter is an important material for maintaining the life activities of microorganisms. In the process of microbial life, the organic matter is first mineralized, then humified and finally decomposed. Throughout the composting process, microorganisms transform energy and material through their own life activities. Soluble small molecules can be directly absorbed and used by the microorganisms, while insoluble large molecules adsorbed outside the body need to be gradually decomposed by the microorganisms through the secretion of a series of extracellular enzymes, and then absorbed and used after decomposition into soluble small molecules. In this way, the microorganisms obtain the nutrients they need to grow and generate more microbial biomass, constantly converting part of the organic matter into simple inorganic substances for crop uptake and releasing CO_2, O_2 and energy. The final product is stable, high-fertility humus.

In the piling process, aerobic microorganisms ferment under aerobic conditions, the pile temperature is high, which can maximize the killing of pathogenic bacteria, and the degradation of organic matter is also fast, less odour, safe and stable, with high nutrient content and other excellent features. It is one of the ways to achieve organic solid waste reduction, resourcefulness and harmless treatment. In the aerobic composting process, the admixtures are then added in a certain proportion. The admixtures added can be divided into two types, one is a conditioning agent and the other is a swelling agent. Conditioning agents are mainly the products of composting formed after the decomposition of rice husks and straw, which effectively reduce the water content of the pile; swelling agents are mainly added to maize cobs, sawdust and other substances, which further increase the porosity of the pile and facilitate the smooth exchange of gas and outside air within the pile. The swelling agent is mainly added with corn cobs, sawdust and other substances, thus further increasing the porosity of the pile and facilitating the smooth exchange of gas and outside air.

The whole process of aerobic composting of sludge has four main steps, heating, disinfection and sterilization, cooling and maturation. At the beginning of the composting process, additives are added in reasonable proportions and sufficient ventilation is required, all of which are indispensable to provide suitable conditions for the microorganisms to flourish. Microorganisms are active in this process, decomposing organic matter and causing the temperature of the reactor to rise considerably, with temperatures exceeding 55°C effectively inhibiting the frequency of microbial activity within the pile, when thermophilic bacteria begin to operate. The main role of thermophilic bacteria is to kill harmful substances such as pathogenic bacteria in the pile; when the decomposition of organic matter is complete, the temperature begins to gradually drop; when the temperature drops to 40°C, the pile enters the stage of ripening, the entire composting cycle is complete.

Compared to anaerobic composting, the decomposition of organic matter in aerobic composting is fast, organic matter is more thoroughly degraded and the composting cycle is relatively short, in general the primary fermentation time for aerobic composting is about 4–12 days and the secondary fermentation time is 10–30 days. The high temperatures generated during the aerobic composting process effectively inactivate pathogens, parasite eggs and plant seeds in the sludge, thus rendering the compost harmless. Aerobic composting has better environmental conditions and the process does not produce terrible odours, so the current composting process is generally aerobic.

Composting technology can make sludge into an efficient, high-quality, safe and harmless organic fertilizer, which is beneficial to agricultural use. With the progress of science and technology, various process technologies have been improved, and the theoretical research on aerobic composting technology has been gradually improved in a large number of engineering applications.

12.2.2 Anaerobic Digestion

Anaerobic digestion is a reaction in which anaerobic microorganisms degrade organic matter under anaerobic conditions to produce CH_4, H_2, CO_2 etc. The anaerobic digestion of sludge can be used to stabilize, recycle, render harmless, reduce the volume of sludge and recover resources (Gao et al., 2019; Xu et al., 2020). Anaerobic digestion can produce clean energy while stabilizing the substrate and is widely used due to its low treatment cost, high destruction rate of pathogenic bacteria and methane production (Wang et al., 2020). For these reasons, it is seen as a sludge treatment technology to prevent pollution and achieve sustainable energy and has been successfully applied to treat municipal sewage sludge.

The anaerobic digestion process, as accepted by scholars at home and abroad, is accomplished in three successive stages: hydrolytic acidification, hydrogen and acetic acid production and methane production, with various microorganisms converting the organic matter into products such as CH_4 and H_2O through their respective metabolic processes. In the first stage of hydrolytic fermentation, hydrolytic acidifying bacteria convert complex large molecules of organic matter (cellulose, sugars, proteins etc.) in the substrate into simple small molecules (amino acids, monosaccharides, propionic acid etc.), which can be used by microorganisms in the next stage. The conversion of large molecules such as proteins, fats, carbohydrates (polysaccharides such as hemicellulose, cellulose, pectin and starch) and nucleic acids, into monosaccharides (and disaccharides), polypeptides (then amino acids), polyols, long-chain fatty acids, glycerol, pyrimidines and purine bases etc. is mediated by corresponding microorganisms, accompanying the production of H_2 and CO_2. In the second stage of acid production, hydrogen- and acetic acid-producing bacteria convert the intermediate products of the first stage into acetic acid, hydrogen and CO_2. In the third stage of methanogenesis, alkyl-producing bacteria convert the acetic acid, H_2 and CO_2 produced in the first and second stages into methane, and methanogenic bacteria mainly include hydrogen-, acetic acid- and methyl-forming methanogenic bacteria. Anaerobic digestion is one of the common processes used to resource, reduce and render harmless sewage sludge.

12.2.2.1 Acidogenic Fermentation

Anaerobic digestion of sludge for short-chain fatty acids (SCFA) production is an important choice for resource recovery from sludge. SCFA are important high value-added intermediates in the anaerobic digestion of sludge and are widely used as raw materials for chemical synthesis, as substrates for the production of clean energy such as biogas, as a source of organic carbon for nitrogen and phosphorus removal microorganisms in wastewater treatment (Lv et al., 2021). It can also be used as a raw material to produce a range of high value-added products, such as carbon for the synthesis of the biodegradable plastic polyhydroxy fatty acid esters (PHA) (Wang et al., 2020), or extracted for the synthesis of other chemicals, such as acetic acid, which can be chemically synthesized into cellulose acetate, which can be further converted into aspirin, latex paint, pigments etc. Therefore, using the acid production phase of anaerobic digestion of sludge to obtain SCFA is an important way to resource sludge and has great potential for application. The SCFAs recovered from sludge are obtained in the acidification phase and in the hydrogen and acetic acid production phase.

A variety of microorganisms are present in the anaerobic sludge digestion system, mainly bacteria, fungi, methanogens, methanogens, thermogens etc. The anaerobic digestion process is actually a series of coupled biochemical reactions carried out by microorganisms, bacteria are the main microorganisms that complete hydrolysis and acidification, bacteria involved in the process with SCFA as end products are collectively known as fermentative acid-producing bacteria, according to their physiological metabolic functions can be divided into cellulolytic bacteria, carbohydrate-degrading bacteria, proteolytic bacteria, lipolytic bacteria, hydrogen-producing acetic acid-producing bacteria and homotypic acetic acid-producing bacteria. Most of these bacteria are exclusively anaerobic, but there are also parthenogenic anaerobes. The hydrogen-producing acetic acid bacteria mainly degrade volatile fatty acids and alcohols to produce acetic acid and hydrogen, which will increase the hydrogen partial pressure of the system and therefore form a symbiotic relationship with the hydrogen-consuming methanogenic bacteria or desulphurization bacteria, while the homoacetogenic bacteria use CO_2 and hydrogen as substrates and compete with other hydrogen-consuming bacteria in the anaerobic system. The various species of fermentative acidogenic bacteria are interdependent and work together to produce acids via anaerobic digestion. SCFA recovery from anaerobic digestion of sludge is an important process for sludge resource recovery and has a promising application potential.

12.2.2.2 Co-digestion

Anaerobic digestion of sludge is a complex process with high investment and operation costs, the system is susceptible to the changes in external environmental conditions. The efficiency of sludge digestion alone is poor due to the lack of sufficient substrate, low sludge activity, low biodegradability, resulting in long digestion cycles, low volatile solids removal and less gas production. Due to the difficult nature of sludge degradation during hydrolysis, the low hydrolysis rate of single digestion makes it difficult to achieve maximum methane production potential (BMP); thus, low biogas production from single sludge digestion is recovered. Therefore, appropriate treatment technologies are needed to improve the digestion performance of the residual sludge and to increase the biogas yield, aiming at maximizing the resource recovery from sewage sludge. In addition, the carbon-to-nitrogen ratio in sludge is generally low, which makes it difficult to meet the normal operational requirements of anaerobic digestion (C/N ratio of 20–30). Therefore, this is why co-digestion is becoming an increasingly popular method of research.

Co-digestion is a biodegradation process in which two or more organic solids are used together as digestion substrates to promote and complement each other. It is an important means of addressing the inadequacy of anaerobic digestion of a single substrate. Co-digestion can overcome the disadvantages of low organic matter conversion, long residence time, accumulated intermediate products causing process poisoning and low biogas yield of single substrate digestion, and significantly improve the stability and gas production characteristics of the anaerobic fermentation system. In addition, the synergy between substrates brought about by co-digestion allows for a more optimal growth and metabolic environment for the microbial community (suitable C/N, ideal nutrient and water content), diluting toxic and hazardous substances, improving organic matter removal and, in most cases, producing more methane from co-digestion than the sum of the anaerobic digestion of the various substrates alone (Fountoulakis and Manios, 2009; Li et al., 2008). In co-digestion, the choice of substrate is very important, if two or more substrates complement each other in nature, and if the balance of macronutrients and trace elements in the system can be achieved, maintaining a suitable carbon-to-nitrogen ratio (C/N) for the growth of anaerobic digestion microorganisms, avoiding acid inhibition and ammonia inhibition, diluting toxic and hazardous substances (Mata-Alvarez et al., 2011). The synergistic effect of "1+1>2" can be achieved by maintaining the right carbon-to-nitrogen ratio (C/N) for anaerobic digestion, avoiding acid inhibition and ammonia inhibition, and diluting toxic and harmful substance. Among the studies carried out so far, municipal waste, kitchen waste and grease trap substrate and fats and oils have been used for co-digestion with sewage sludge.

Co-digestion of sludge with other substrates not only treats these pollutants simultaneously and reduces the waste treatment branch process, but also improves the efficiency of anaerobic digestion, recovers resource energy more efficiently and generates economic benefits, so co-digestion has become a hot research topic.

12.2.2.3 Microbial Electrolysis Cell

As the microbial cell walls in sludge are stable, complex organic matter is not easily hydrolysed and the decomposition rates are slow, therefore often limiting the efficiency of sludge hydrolysis and acidification, which in turn reduces the degree of sludge reduction and methane production. Furthermore, conventional anaerobic digestion relies on interspecies hydrogen transfer between acid-producing bacteria and methanogenic bacteria, a fragile metabolic approach that can easily disrupt the balance of methanogenic metabolism, thus affecting the overall anaerobic digestion effect. Recent studies have found that microbial electrolysis cell systems can shorten the duration of anaerobic digestion, breaking the bottleneck of anaerobic digestion in treating residual sludge that cannot effectively use protein, and that the coupling of microbial electrolysis with anaerobic digestion can promote the efficiency of anaerobic digestion of sludge, including hydrolytic acidification.

Microbial electrolysis cell is a new type of anaerobic treatment technology for environmental microorganisms, which can promote the hydrolysis and acidification of the substrate by using electrochemically active microorganisms to oxidize and degrade a wide range of organic substances (volatile acids, glucose, proteins, cellulose etc.) to generate electrons for transfer to the cathode. At the same time, the electrons and protons generated by oxidation are combined at the cathode to produce hydrogen gas (Wang et al., 2022). This enhances the activity and abundance of hydrogenophilic methanogenic bacteria in the anaerobic system, ensuring the smooth production of hydrogen and acetic acid in the system, providing substrate for the main pathway of methanogenic methanogenesis, significantly enhancing the efficiency of methanogenesis and increasing methane production (Wang et al., 2021). The MEC is an effective way to balance the acidity of the anaerobic treatment system and to protect the anaerobic methanogenic system from external environmental factors (Wang et al., 2022). By integrating two bioelectrodes into the anaerobic digestion process (MEC-AD), MEC not only enriches functional electroactive microorganisms but also enhances information communication and extracellular electron transfer between the microbial cells and the solid electrode at the same time (Lin et al., 2019).

The coupling of microbial electrolytic cell technology with anaerobic digestion has obvious advantages in terms of pollutant treatment effectiveness and sludge resource recovery and has broad research and application prospects.

12.2.3 Other Technologies

Sewage sludge treatment is a rapidly changing field. Long-standing methods of sludge treatment, such as land use, agricultural use, composting or landfilling, do not quite meet today's requirements for the degree of sludge treatment. Looking to the future, organic pollutants, microplastics and heavy metals from agricultural use and composting will become constraints due to environmental concerns, and landfills lead to the loss of nutrients and raw materials and no longer follow the current trend of material recovery in a circular economy.

Thermal treatment is an effective and safe method of treating sewage sludge and is considered one of the most promising processes for sewage sludge treatment. Heating of the sludge allows the decomposition of some organic matter and the hydrolysis of hydrophilic organic colloidal substances, as well as the decomposition and destruction of microorganisms in the sludge and the freeing of water from the cell membranes, thus improving the thickening and dewatering properties of the sludge. Heat treatment of sludge has the advantages of stabilization, rapid treatment, small footprint, harmlessness, reduction and resource efficiency. Heat treatment of sludge causes chemical reactions in the organic matter in the sludge by heating, oxidizing toxic and harmful pollutants such as PAHs and PCBs and toxic microorganisms such as pathogenic bacteria. Heat treatment destroys the cellular structure by heating, so that the internal water in the sludge is released and removed, for example the incineration process reduces the water content of the treated sludge (actually the ash after incineration) to zero, achieving maximum reduction. And with municipal sludge after thermal treatment, the amount of waste can be reduced by up to 50% in order to save costs and recover biofuels or chemicals from sewage sludge for sustainable management. On the one hand, the stabilization process allows for the relevant resource use, on the other hand, a large amount of organic matter in the sludge can be converted into combustible oil, gas and other fuels, while the thermal treatment also enables the immobilization of heavy metals and other harmful substances. Furthermore, the results of studies have shown that the initially neglected dirty peat from pyrolysis (a type of biochar) is not only convenient for subsequent incineration but also suitable for use as a soil conditioner. Pyrolysis units are also promising in remote areas with low heavy metal content sludge, but the process should be maintained at a safe treatment temperature of 600°C or more and operated under stable conditions to remove organic pollutants. The process should be operated at a safe temperature above 600°C and under stable conditions to remove organic pollutants.

12.3 MICROBES FOR IMPROVED SEWAGE SLUDGE TREATMENT

Microbial enhancement is based on the fact that not all microorganisms in natural systems are the most effective microorganisms or present in the desired concentrations. The concentration and metabolic activity of bacteria in the system can be increased by selective placement of some microbial strains or genetically improved microbial engineered strains into the system and can only work in close integration with the wastewater treatment facility in practical applications. Such selected placed strains should be able to maintain and enhance the activity of naturally occurring strains, thus optimizing and controlling the balance of microbial populations and improving the efficiency of the treatment plant. The conditions that need to be met by the placed strains are generally as follows:

a. Ensure the efficacy of sewage treatment
b. Inclusion of indigenous microorganisms
c. Adaptability to process facilities

Population studies are one of the first elements of microbial treatment of sewage sludge. In the survey of microbial populations in wastewater reactors using 16S rRNA gene analysis technique, 36 major communities were found, indicating the diversity of wastewater microorganisms. The introduction of

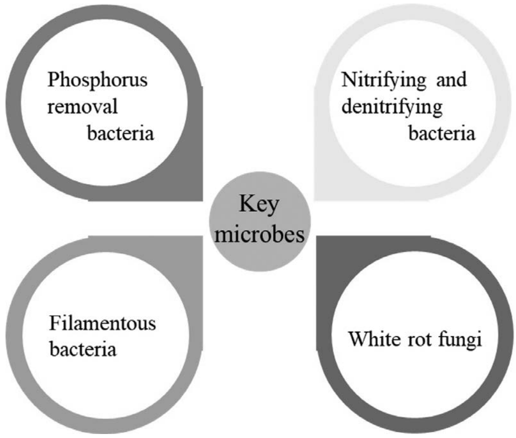

FIGURE 12.1 Main categories of functional microbes for sewage sludge treatment.

molecular biology and environmental engineering detection techniques in the study of microbial community structure and function will be more helpful for the development and use of microbial communities in wastewater biological treatment systems. Major categories of functional microorganisms for enhanced sewage sludge treatment and resource recovery are presented in Figure 12.1.

12.3.1 Phosphorus Removal Bacteria

The amount of phosphorus in sewage sludge is the main indicator of its contamination, and phosphorus-removing bacteria can enable biological phosphorus removal to purify sewage sludge. Most of the identified isolates with the capacity of phosphorus removal are belonged to the genus *Acinetobacter* in the r-*Proteobacteria*. Many other groups isolated using different selective media by researchers were also identified as *Acinetobacter* spp. (Chen et al., 2021). There are also bacteria that have the capability of both nitrogen and phosphorus removal. Several strains from various genera have been reported to accomplish nitrogen and phosphorus removal, which are primarily affiliated with *Pseudomonas*, *Bacillus*, *Paracoccus*, and *Arthrobacter* (Dai et al., 2022). The mechanism of simultaneous nitrogen and phosphorus removal by denitrifying phosphorus accumulating organisms is different from the traditional biological nitrogen and phosphorus removal. The denitrifying phosphorus removal technology based on the metabolic function of denitrifying phosphorus accumulating organisms can overcome the problem of carbon source competition and sludge age contradiction in traditional biological nitrogen and phosphorus removal processes and can be applied to the treatment of urban sewage with low C/N ratio.

12.3.2 Nitrifying and Denitrifying Bacteria

With the help of heterotrophic microorganisms, sewage sludge produces ammonia by ammonification, which is then oxidized by nitrifying and nitrosating bacteria to nitrite and nitrate, thus playing a detoxifying role. Nitrifying bacteria play an important role in wastewater treatment, agriculture and other fields, and have become a hot topic of research in the world in recent years. The analysis showed that 50% of the membranes belonged to nitrifying bacteria and the remaining 50% were heterotrophic bacteria, which were 23% of a subclass of Amoeba, 13% of the y subclass, 9% of the green non-sulphur bacteria, 2% of the cytophaga–flavobacterium–bacteroides group and 3% of the undefined taxa. This result indicated that the nitrifying bacteria supported the heterotrophic bacteria through the production of soluble products and the heterotrophic bacteria also ensured the ecological stability of the biofilm in terms of metabolic diversity. Most members of denitrifying bacteria, such as *Alcaligenes*, *Pseudomonas*, *Methylobacterium*, *Paracoccus* and *Hyphomicrobium*, have been isolated from wastewater plants as denitrifying microbiota. *Hyphomicrobium* etc., have been isolated from wastewater plants as denitrification microbiota. Whether these bacterial genera have in situ denitrification activity in wastewater plants is not known. A combination of Fluorescence in situ hybridization and microscopic autoradiography was used to identify nitrogen-degrading bacteria in situ. This technique, combined with the full-cycle rRNA method, revealed new, as yet uncultured, nitrogen-fixing Vibrio spp. (Azoarcu) related bacteria, which are important denitrifying bacteria in wastewater plants.

12.3.3 Filamentous Bacteria

Filamentous bacteria are used as the skeleton of sludge flocs, and the surface of filamentous bacteria is adhered to the bacterial colloid, forming a sludge floc with a tight structure and good settling performance, which has a high purification efficiency; on the other hand, when the floc size increases to a certain critical value, the filamentous bacteria stretch out and significantly affect the settling properties of flocculent activated sludge (sludge swelling) or cause biomass changes and This significantly affects the settling properties of flocculent activated sludge (sludge swelling) or causes changes in biomass and foam formation (sludge foaming), thus seriously affecting the treatment efficiency of activated sludge. The use of rRNA target oligonucleotide probes allowed the rapid identification of most filamentous bacteria, revealing that some filamentous bacteria in activated sludge exhibit polymorphism. Filamentous Chloroflexi received considerable attention in sewage sludge treatment systems due to their contribution to the nutrient transformation (Nierychlo et al., 2019).

12.3.4 White Rot Fungi

White rot fungi are a peculiar group of filamentous fungi that decay on trees or wood and are capable of degrading lignin and causing wood decay, which to some extent excludes barriers to the carbon cycle in the Earth's biosphere. White rot fungi are directly involved in the degradation of various difficult-to-degrade organic pollutants and toxic substances through the production of extracellular oxidases, such as lignin peroxidase, manganese peroxidase and laccase. This unique degradation ability and degradation mechanism has been highly valued by scientific and industrial communities worldwide for many years. White rot fungi possess unique oxidative and extracellular ligninolytic systems, which exhibit low substrate specificity, enable them to degrade a great number of pollutants in sewage sludge (Zhuo and Fan, 2021).

In addition, sewage sludge is subjected to different environments, and the study of microorganisms and their functions in extreme environments is highly sought after. At present, the following categories are known: I. Thermophiles: microorganisms that can survive in high-temperature environments from 50°C to 110°C; II. Psychrophiles: microorganisms that can survive in low-temperature environments from 0°C to 15°C with the maximum growth temperature not exceeding 20°C; III. Alkaliphiles: microorganisms that can survive in environments with pH greater than 9, usually pH 10–12 in the environment to survive, and in neutral conditions grow slowly or do not grow microorganisms; IV. Halophilic bacteria: at least 2 mol/L (3% to 20%) of salt in the presence of microorganisms; V. acidophilic bacteria (Acidophiles): in the environmental pH value is not higher than 2.0 to survive microorganisms; VI. pressure bacteria Piezophiles: microorganisms that can survive under a pressure greater than one atmosphere. Extreme environmental microorganisms produce extreme enzymes and active substances such as antibiotics that have a broad application for sewage sludge treatment.

12.4 SEWAGE SLUDGE MANAGEMENT

12.4.1 Current Practices

The sludge consists of two basic forms, sludge and secondary sludge, also known as activated sludge in the case of activated sludge process. Sludge from biological treatment operations is sometimes

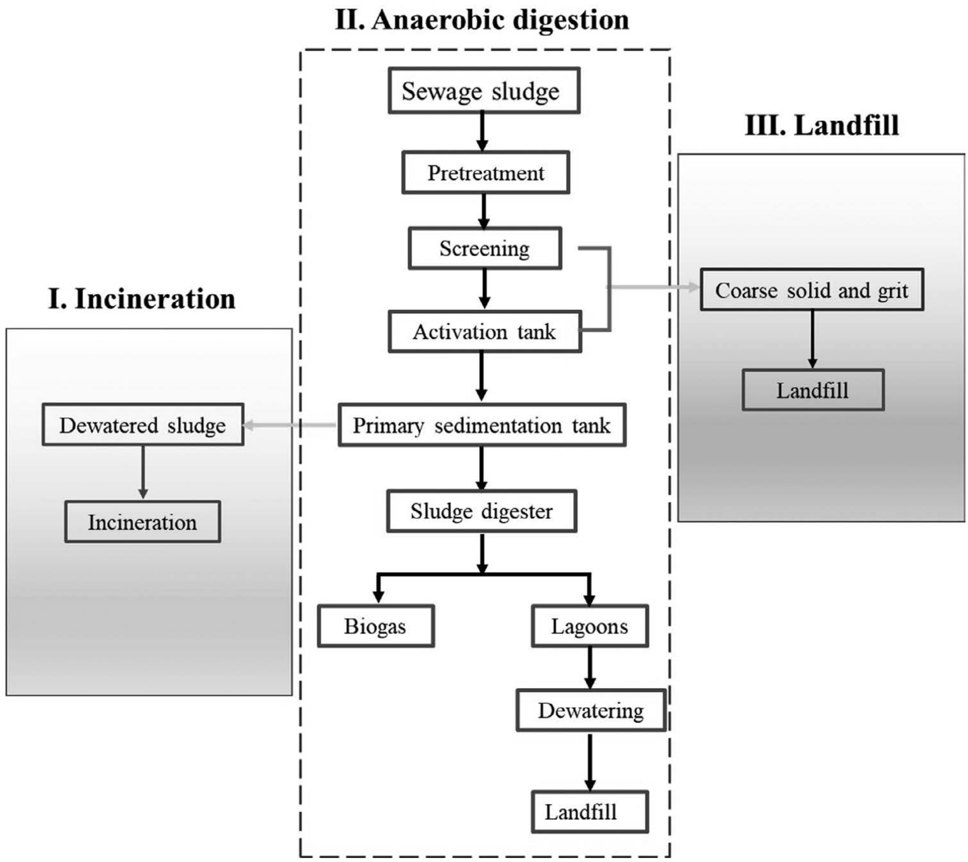

FIGURE 12.2 Simple flow chart of sewage sludge treatment.

referred to as wastewater biosolids. Wastewater sludge contains a variety of organic and inorganic compounds. The generation of municipal sewage sludge has increased in parallel with rapid industrialization. Sewage treatment is the process of removing contaminants from wastewater, primarily from household sewage. It includes physical, chemical and biological processes to remove these contaminants and produce environmentally safe treated wastewater. Sewage treatment is mainly divided into three stages: preliminary treatment or pretreatment, primary treatment, and secondary treatment.

Figure 12.2 shows a simple flow chart of sewage sludge treatment. In pretreatment, coarse solids (with a diameter of more than 2 cm) and grit (heavy solids) are removed by screening. These coarse materials are not included in biosolids. Preliminary treatment may include a sand or grit channel or chamber, where the velocity of the incoming sewage is adjusted to allow the settlement of sand, grit, stones, and broken glass. These coarse solids are removed because they may damage pumps and other equipment. The influent in sewage water passes through a bar screen to remove all large objects such as cans, rags, sticks, and plastic packets carried in the sewage stream. The solids are collected and later disposed of in a landfill or incinerated. Bar screens or mesh screens of varying sizes may be used to optimize solids removal.

In primary treatment grit (fine, hard solids), suspended solids and scum are removed in two stages: pre-aeration and sedimentation. The settled and floating materials are removed, and the remaining liquid may be discharged or subjected to secondary treatment. Grease and oil from the floating material can sometimes be recovered for saponification or biodiesel production. The wastewater is aerated by air pumped through perforated pipes near the floor of the tanks. This aeration makes the wastewater less dense, causing the grit to settle out. As the air jets are positioned such that the water is swirling as it moves down the tanks, the suspended solids are prevented from settling out. The air also provides dissolved oxygen for the bacteria to use later in the process, but the wastewater is not in these tanks long enough for bacterial action to occur in the process. The solids are removed from the bottom of the tanks by scrapers, and scum is washed off with water jets. The scum and solids are brought to a common collection point where they are combined to form sludge and sent off for anaerobic digestion.

Secondary sludge accumulated from secondary wastewater treatment process is also added to the digester tank. The organic compounds within the sludge are converted to carboxylic acids and then finally to carbon dioxide and methane by anaerobic fermentation. The obtained biogas is a valuable source of fuel. When the sludge leaves the digesters, it has undergone a 50% volume reduction. Careful management throughout the entire treatment process allows plant operators to control solid ingredients, nutrients, and other components of biosolids. Micropollutants such as pharmaceuticals, ingredients of household chemicals, chemicals used in small businesses or industries, environmental persistent pharmaceutical pollutant, or pesticides may not be eliminated in the conventional treatment process (primary, secondary, and tertiary treatment) and therefore lead to water pollution. The micropollutants eliminate via a fourth treatment stage during sewage treatment; however, since those techniques are still costly, they are not yet applied on a regular basis.

12.4.2 Legislation and Regulations

Environmental legislation and regulation is a key driver to improve sewage sludge treatment and management standards and pursue high energy and resource recovery. European Union countries established their directives much earlier, for example the Council Directive dated on 19 December 2002 on the establishment of criteria and procedures for acceptance to store a given type waste. However, the current legislation and regulation frameworks are mostly tailored to the current structures. Thus, proper adjustment is needed for the adaption to the challenges in future. Take nutrient phosphorus as an example. In 2016 and 2017, Switzerland and Germany announced their legal requirements for phosphorus recovery, respectively. The fertilizer regulation (EC 2003/2003) of European Union is an example of such a regulation that can be used to support the quicker transition towards nutrient recovery from sewage sludge. The legislation and regulations are not specified in a single legal act and distinct depending on the purpose of treatment and management. Hence, the way of monitoring and enforcing the discharge limits should be relevant and flexible.

12.5 CONCLUSIONS AND PERSPECTIVES

As the global population expands and urbanization accelerates, the amount of sewage sludge generated continues to increase. Traditional sludge disposal methods. such as landfilling and land tillage, are no longer suitable for future sludge management and are being banned in an increasing number of countries and regions. Therefore, there is an urgent need for environmentally friendly and economically viable technologies to properly treat sewage sludge. Anaerobic digestion of sewage sludge from wastewater treatment plants is one of the most promising technologies for bioenergy production. However, depending on the implemented substrates and facilities, these techniques have shown variations in sludge solubilization and biogas production. These existing pretreatment methods require further research to address the economic and energy concerns. Finally, there is an utmost necessity to establish standardized techniques for each pretreatment technique in terms of energy balance and environmental sustainability perspectives.

The research on the use of microorganisms in wastewater and sludge treatment at home and abroad has made great development, but there are also some weak points, according to the current development trend, it is appropriate to carry out and pay attention to the following research work in the future:

1. Application of the latest molecular biology techniques for the detection and ecological study of microorganisms in sewage sludge treatment can be used to track and analyse the dynamics of the microbial population and determine the number, community structure and activity of functional microorganisms in the system, so as to achieve the purpose of microbial treatment of sewage sludge. There is still an urgent need for the development of appropriate bacterial culture strategies.
2. The differences in temperature, humidity and dissolved oxygen in water in different climatic zones induce large differences in microbial decomposition ability. Firstly, temperature and precipitation changes affect the growth rate of microorganisms, and secondly, climate changes affect the enzymatic activity of microorganisms and thus change the degradation rate of organic matter in sewage sludge. There are few studies on this topic at home and abroad, so it is necessary to further investigate the mechanism of microbial action at different spatial and temporal scales in order to apply microbial treatment of sewage sludge more effectively.
3. To investigate the interfacial morphology of microorganisms and to explore their microscopic mechanisms at the nanoscale and higher levels by using atomic force microscopy and other sophisticated instrumentation; to further reveal the structural changes of microbial responses to pollutant stress by combining emerging technologies such as proteomics, Raman spectroscopy and other interdisciplinary research methods and knowledge.
4. Stable isotope ratio signal is a good tracer in sewage sludge treatment process, and the study of microbial-secreted enzymes and nutrient removal can be studied with the help of stable isotopes. Typical sewage sludge can be selected for in situ study to more accurately investigate the changes of microorganisms in the sludge treatment.

DECLARATION OF COMPETING INTEREST

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this chapter.

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13 Emerging Resources Recover Technology

Sustainable Treatment and Management of Sewage Sludge

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13.1 INTRODUCTION

The current scenario has faced a great challenge allied with sewage sludge management. Improper sewage sludge management will produce the threat linked with the toxic elements. Furthermore, the inadequate storage, treatment, and transport of the same in a frequent manner will also instigate the chance of subsoil water contamination. In this regard, the recent research has focussed on a plethora of industrial as well as technological solutions to recover energy, water, and other products from sewage sludge. The major drivers of implementing these kinds of technological approaches have ranged from the resource recovery potential to the large environmental footprints and high energy demands. The majority of the conventional techniques currently employed for this purpose have principally known to possess certain significant concerns; hence, the research community has

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predominantly followed the advanced technologies to rectify such concerns and thereby offer sustainable development in an eco-attractive way (Kehrein et al., 2020).

Sustainable development has been recognized as the overall goal of sewage sludge management and to get an overview regarding the particular degree of treatment, it is essential to analyse the concept of advanced developments in water quality management approaches in the context of reducing deleterious effects towards the environment (Khan and Malik, 2014; Potter, 1914; Wei et al., 2003). Even though it has been done empirically for greater than 5000 years, from the various advantageous effects obtained in these regards, the recovery of various elements from the sewage sludge in a modern technology-based perspective can be recognized as a novel inference in many communities across the world (Dalezios et al., 2018). It has also been suggested that the reuse of reclaimed wastewater during the twentieth century could be reported as a result of technology-based practices. There is currently a substantial volume of wastewater accessible for treatment, and the aforesaid volume will definitely grow to an extreme level in future (Alawode et al., 2019; Jarcu et al., 2018; Salgot and Folch, 2018). The perspectives of many of the previous research articles clearly argue that advancements in various technologies of wastewater management and practices will be considered to be instrumental in various circumstances especially in providing clean water, sanitation, and environmental protection to growing economies and populations (Lofrano and Brown, 2010).

The promising developments in wastewater management and water purification allow the scientific community to recover various valuable resources, including biosolids, electricity, biodiesel, nutrients, and recycled water (Kuwayama and Olmstead, 2020). The treatment has usually focussed on diminishing the rate of toxic elements released into water bodies, thereby enhancing the water quality as additional societal benefit. The emerging technologies (Table 13.1) let the facilities of wastewater treatment recover essential valuable resources, including nutrients, recycled water, and energy, during the treatment process (Mo and Zhang, 2013). The membrane filtration approaches

TABLE 13.1
Various Technologies and Major Contaminants

Sl. No.	Technology	Contaminants	Reference
1.	Sedimentation	Settleable solids	Ghernaout and Ghernaout (2012)
2.	Coagulation	Colloids	
3.	Flocculation	Colloids	
4.	Sand filtration	Settleable solids	Swapnil et al. (2018)
5.	Membrane filtration and Biological filters	Nitrogen compounds	
6.	Screen filter	Settleable solids	
7.	Chemical oxidation	Iron and manganese	
8.	Biological filters	Iron and manganese	
9.	Chemical oxidation	Organic compounds	
10.	Chemical precipitation and Activated carbon adsorption	Arsenic	Devda et al. (2021); Sorlini et al. (2015)
11.	Activated carbon adsorption	Organic compounds	
12.	Membrane filtration	Arsenic	
13.	Micro-screen filter	Cyanobacteria	
14.	Thermal processes		
15.	Chemical oxidation	Cyanobacteria	
16.	Coagulation flocculation	Cyanobacteria	Devda et al. (2021)
17.	Ion exchange	Arsenic	
18.	Dilution with rainwater	Salinity	
19.	Sand filtration	Cyanobacteria	

have recently been witnessed to be a key technology in the aforesaid regard, which allows the consistent advanced treatment for the reclamation and reuse of wastewater and sewage sludge management. Producing efficient output followed by acting as a physical barrier counter to particle material and the need for relatively less space has been recognized to be the major advantage of the membrane filtration approaches. The use of ultrafiltration (UF) membranes offers researchers to remove the polysaccharides, colloids, bacteria, and proteins even if certain viruses recently, which augment the output quality of the treated water. Activated carbon filtration, extraction, advanced oxidation processes, ion exchange, adsorption, distillation, hybrid and intensified processes, precipitation with the aid of bioelectrochemical systems, thermal energy, and hydropower have also produced the same kind of output (Kehrein et al., 2020). In this regard, the current chapter discusses the major treatment process along with the aforesaid perspectives and the recovery of various elements with special inference on the emerging technologies for the sludge treatment process.

13.2 NECESSITIES FOR THE SEWAGE SLUDGE MANAGEMENT

By 2050, we must address intensive water and nutrient scarcity (Lal, 2016; Verstraete and Vlaeminck, 2011). The inferences concerning the Zero Waste Water concept would be known to represent a centralized technology with prominent advantageous effects on the society. Surprisingly, Zero Wastewater is predicted to be profitable approach (Verstraete and Vlaeminck, 2011). Water pollution can occur from a variety of sources, including residences, industry, and mining; however, one of the most significant facts in this regard is the widespread use of the same by industry (Crini and Lichtfouse, 2019; Schwarzenbach et al., 2010), which generates the sewage sludge.

In general, process waters (also known as wastewaters or effluents) are noted to be the most problematic. In one crucial way, the wastewaters usually exhibit a great range of differences from drinking water sources; when compared to wastewater originating from industrial-type activities, contamination levels in most drinking water sources are found to be fairly low.

The toxicity of the same, however, is determined by their composition, which is determined by the origin of the resource. The challenges instigated during the treatment are extremely complex because effluent comprises a variety of pollutants depending on its source (Crini and Lichtfouse, 2019).

The development of less expensive, more effective, and new decontamination technologies has also been found to be an active area of research in the current scenario, as seen in many of the previous studies. The preservation of the environment, particularly in the context of water pollution, has become a serious concern for the public, scientists, industry, and researchers, including national and international decision-makers (Crini and Lichtfouse, 2019).

The public's desire for pollutant-free waste discharge has made industrial wastewater cleansing a major priority (Shack and Moore, 2014). However, this fact has been noted to be a demanding and difficult task. It is therefore hard to identify a general method for the removal of all major and prominent toxic elements from wastewater. This assessment discusses the benefits and drawbacks of available technology. A wide range of approaches can be applied, including traditional methods, proven recovery processes, and developing removal technologies (Hsu et al., 2019). However, only a handful of the diverse wastewater treatment procedures are routinely used by the industrial sector in recent epochs. Despite this, adsorption onto activated carbons has frequently been regarded as the preferred method for removing a wide range of contaminants since it provides the greatest outputs in terms of productivity (Crini et al., 2018).

Domestic wastewater contamination is a severe problem (DeGeorges et al., 2010) in both developed and developing countries, affecting water bodies, groundwater, and the ecosystem. Domestic wastewater treatment devices consume a significant amount of energy. This has significantly increased greenhouse gas (GHG) emissions (Hendrickson et al., 2015) as well as the spread of several diseases. Furthermore, with the consumption of energy at a high rate, there exists an increase in the chance of global warming, which is maybe the most significant challenge that humanity confronts today.

From the 1990s to the present, the life cycle assessment (LCA) has been employed in the field of environmental research (Reich, 2005). Various organizations, governmental organisations, municipal employees, and residents, along with the industrial sector, are currently concerned about the environmental impact of many items (Sabeen et al., 2018). These concerns are about the manufacturing processes, associated services, waste disposal, and the goods themselves on persons and the surrounding. Almost every product that leaves a household, whether as solid trash or as domestic wastewater from activities, such as washing clothing, cooking, or bathing, has an impact on the stability of our environment. The LCA is a comprehensive way that considers the impact on all forms of natural environments as well as human health. It is an environmental technique that examines the environmental consequences of a product's life cycle. As a result, it can aid in providing a full assessment of product sustainability from three important perspectives: social, environmental, and economic.

13.3 ADVANCED DEVELOPMENTS IN SPECIFIC TECHNOLOGIES FOR THE TREATMENT

Rapid urbanization along with industrial development is recognized to be the most important tool for the exploitation of freshwater resources (Ako et al., 2010). Climate change and terrain use patterns are major concerns for urban water ecosystems and humanity. Increased hard surface area, GHG emissions, flooding, wastewater generation, and mismanagement, along with a variety of other variables, are wreaking havoc on freshwater resources in both qualitative and quantitative ways (Zouboulis and Tolkou, 2015).

Temperature rises and other human activities contribute to water contamination and diminish the exploitation of natural water resources. Due to the aforementioned considerations, urban planners and governing bodies have a major dilemma due to limited water availability and massive water demand (Nguyen et al., 2019). In order to reveal the complicated dynamics belonging to various circumstances, a substantial study has recently been dedicated to performing field-based studies thereby developing new techniques to induce the influence of climate change on aquatic environments. However, given the critical importance of water resources to humanity, it is still necessary to conduct research on all aspects of the impact of climate change and anthropogenic activities, including biogeochemical cycles, eutrophication, emerging pollutants in the flood resilience, and the water environment (Ho and Goethals, 2019).

13.4 MAJOR TREATMENT PROCESS

The by-product of treated wastewater is usually referred to as sewage sludge and is consisted of various following elements, including organic and inorganic elements: organic chemicals, plant nutrients, and pathogens (Singh and Agrawal, 2008). For this reason, it is intensively significant to treat the above-said wastewater resources in a proper way to diminish the various consequences instigated over the environment, thereby offering an effective way for nature conservation perspectives. In this regard, it is essential to discuss the basic sludge treatment process, consisting of Stages I–IV (Figure 13.1).

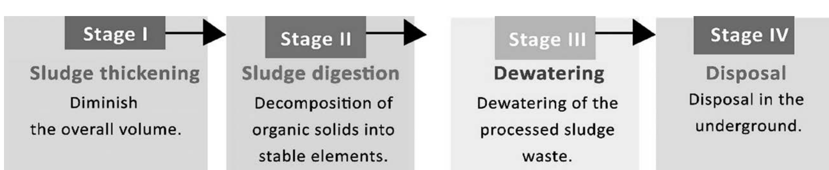

FIGURE 13.1 Sludge thickening.

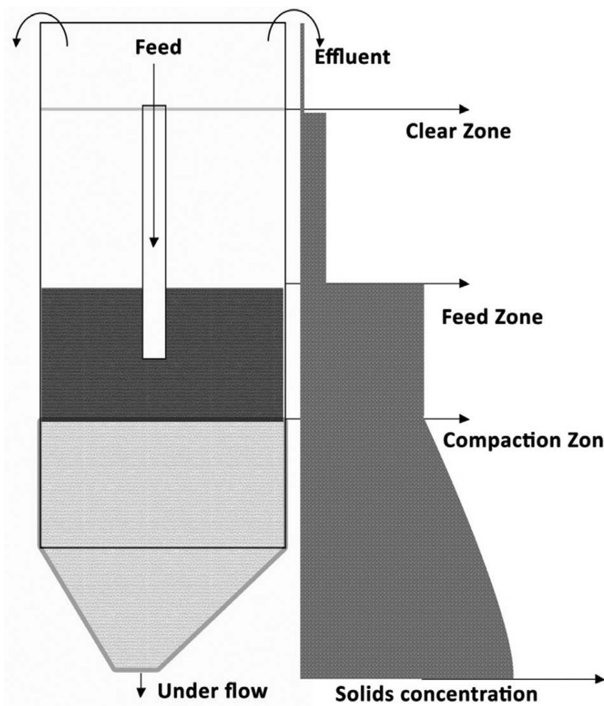

FIGURE 13.2 Sludge digestion and anaerobic granular technology.

Sewage sludge is a by-product of treated wastewater (Feng et al., 2015). The presence of organic and inorganic materials along with certain other critical elements such as pathogens and nutrients are also witnessed in the aforesaid sewage sludge (Fijalkowski et al., 2017). Therefore, it is extremely imperative to properly treat the same to minimize its repercussions over the environment. Here the authors have added a brief overview concerning the sludge treatment process to attain and accomplish a better understanding regarding the various treatment techniques followed by the most important requirements.

13.5 STAGE I

Sludge thickening (Figure 13.2) has been noted to be the first step in the treatment process (Zhao et al., 2022). This step includes the thickening of the sewage sludge to diminish the overall volume of the same, thus allowing easy handling. In addition to this, dissolved air flotation has also been used as an alternative to accomplishing the above-said target with the aid of air bubbles (Pachaiappan et al., 2022).

13.6 STAGE II

The process of sludge digestion (Figure 13.3) is primarily recognized as the second stage, involving the decomposition of organic solids into stable elements (Rajendran et al., 2020). The stage also witnessed a diminishing of the total solid's mass and the entire process has been completed in two stages. The process is always noted as biological; hence, the bacteria present in it hydrolyze the protein molecules and lipids in the sludge thereby forming smaller water-soluble molecules (Gerardi, 2006). After this, the processed sludge enters the second chamber where a mixture of methane and carbon dioxide (CO_2) has formed (Feodorov, 2016). The formed methane has now been reused to power the digestion chamber in the system to generate enough power for the various processes involved (Rulkens, 2008).

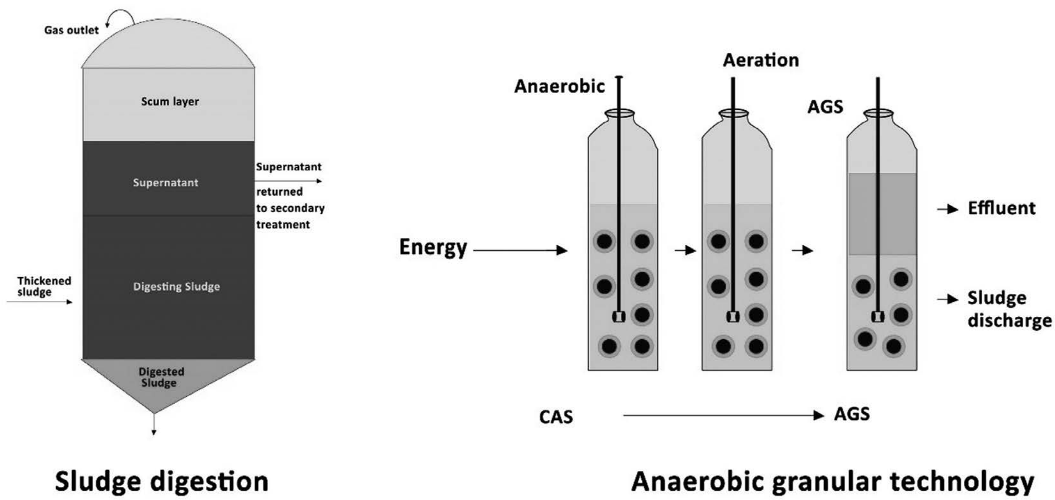

FIGURE 13.3 Major stages of sludge treatment process (Stages I–IV).

13.7 STAGE III

The dewatering of the processed sludge waste was recognized as the third stage (Yu et al., 2019). The use of sludge drying beds for drying is noted to be a most predominant method to complete stage III (Yoshida et al., 2013). The solid-liquid separation devices have also been used in many circumstances to fasten the process. The belt filter press along with the rotary drum vacuum filter has also been recognized as an alternative strategy to retrieve all the water thereby offering an easier approach to handling sludge waste at reduced costs in shorter durations (Tunçal and Mujumdar, 2022).

13.8 STAGE IV

The final stage usually referred to as the disposal stage involves the disposal of the same in the underground (Demirbas et al., 2017). The sludge waste in this form can also be used as a fertilizer (Usman et al., 2012). If the sludge waste was found to be extremely toxic, in such situations, the same will be incinerated and converted into ash (Xue and Liu, 2021).

13.9 EMERGING RESOURCES RECOVER TECHNOLOGY AND MANAGEMENT OF SEWAGE SLUDGE

Human survival depends on the availability of clean water. To fulfil the demands for potable and agricultural water, there is an instant need to treat the water from various waste resources, specifically sewage sludges and slimes principally from industrial effluents. These unwanted sewages are previously known to be reported as highly and potentially hazardous in nature. For this reason, it is essential to consider the fact that, the toxic elements must be removed from wastewater before they may be reused (Levine et al., 1985). The removal of pollutants from these effluents includes the process of flocculation and coagulation. The process of flocculation as well as coagulation can be employed in this regard.

From a human life cycle perspective, it appears that the mountains' natural waterfalls are merging with the oceans and the human population maintains this water for industrial, municipal, and agricultural needs (Kumar et al., 2017). But the constant and frequent exploitation of natural water resources without prioritizing their conservation may primarily result in the formation of wastewater.

The following factors principally aid the conversion of natural water into wastewater form: sewage from homes, population increase, urbanization, industry, institutions, and hospitals (Corcoran, 2010).

13.10 WASTEWATER TREATMENT AND BASIC APPROACHES

Because of the presence of a plethora of toxic elements, including biological substances, poisonous inorganic compounds, and the presence of hazardous materials, in the water, it was noted that it can be harmful to the public (Kumar et al., 2017). Coagulant chemicals and their associated products are useful, but they can modify the physical and chemical properties of water, complicating sludge disposal. Hence, the current research has principally focussed on advanced technologies, including the membrane-based approaches.

13.11 MEMBRANE-BASED PROCESSES

The membrane-based strategies are recently attaining prominent attention since they offered the following benefits: great fitness for the recovery of energy and various other elements. The output of the separation approaches is primarily influenced by the type of pores, which is classified based on the size of the same. Among the different types, the macropores (yeast, bacteria, and fungi), mesopores (proteins, viruses, polysaccharides, and nucleic acids), and micropores (common antibiotics, water, and inorganic ions) depict a pore size of >50, 2–50, and 0.2–2 nm, respectively (Udugama et al., 2017). The dairy industry has been witnessed to be one of the major instances, where a resource recovery approach can offer plenty of advantageous effects by using membrane-based methods. The advanced novel approaches focusing on integrating the hybrid perspectives in membrane-based research have recently achieved much more interest than the conventional approaches. Such membrane-based approaches could productively replace technologies that prefer conventional operations, including distillation, thereby eliminating the chance of decomposition of certain specific side products that are previously known for their temperature-sensitive characteristic properties (Udugama et al., 2017).

13.12 PRECIPITATION AND EXTRACTION

Minerals' precipitation profoundly cooperates with other chemical processes and directly impacts the composition of water through sequestering soluble. The basic mechanism allied with the precipitation has been relied upon to recover P from various aqueous wastes. The extraction approach is one that primarily separates a particular chemical element or substance when it is dissolved with others. The solvent extraction, ion exchange, as well as adsorption approaches are primarily belonging to this class of technologies (Udugama et al., 2017).

13.13 NUTRIENT REMOVAL PROCESSES

The nutrient removal processes in wastewater treatment have enhanced treatment procedures to provide environmental protection (Wilsenach et al., 2003). This consumes energy while disposing of resources. The application of integrated management of assets in the recent investigation made wastewater management more sustainable (Wilsenach et al., 2003).

13.14 CONVENTIONAL ACTIVATED SLUDGE (CAS) SYSTEMS

Sewage treatment is mostly based on traditional activated sludge systems, which provide a suitably reduced level of pollutant effluent. CAS, on the other hand, has an economically friendly approach with strong recovery potential (Verstraete and Vlaeminck, 2011), as well as large electricity demand and environmental imprint.

13.15 BIOGRANULATION TECHNOLOGY

One of the major approaches for the treatment of wastewater of sludge origin is granulation technology (Liu and Tay, 2004). The principle behind the biogranulation technology primarily relied on the theory of granulation. Anaerobic and aerobic granulation methods are used in biogranulation technology (Liu and Tay, 2004). The anaerobic granulation is a popular one, while research on aerobic granulation found recently begun. A plethora of full-scale anaerobic granular sludge facilities have been operational around the world, but no analogous units for aerobic granulation have been reported. Cell-to-cell interactions involving the hollowing pattern are involved in granulation: biological, physical, and chemical (Liu and Tay, 2004). Aerobic and anaerobic granulations are the two types of biogranulation. Microorganisms self-immobilize to generate biogranules (Quan et al., 2015). These granules are formed or consisted of dense microbial consortia packed with several bacterial species, and each gram of biomass generally contains millions of organisms. These microorganisms have several functions in the degradation of complicated industrial wastes. Biogranules have a regular and sturdy structure, as well as good settling capabilities, as compared to traditional activated sludge (Liu and Tay, 2004). They have a great biomass retention rate and can handle high-strength wastewater. Anaerobic granule formation has been extensively explored and is most likely best recognized in the up-flow anaerobic sludge blanket (UASB) reactor. Anaerobic granulation technology has already been employed in several treatment plants (Abbasi and Abbasi, 2012). Individual species in the above-said habitat are unable to degrade the influent completely. The complete decomposition of waste always necessitates complex interaction among the organisms inhabited over there (Tay et al., 2009).

13.16 MICROWAVE (MW) CHEMISTRY

Microwave (MW) chemistry has received a lot of attention in the field of wastewater treatment (Wang and Wang, 2016). It has been used successfully with a variety of organic wastewater treatment technologies. Understanding the current state of research on MW-based treatment procedures could benefit the advancement of wastewater treatment technologies connected with MW irradiation (Wang and Wang, 2016).

When compared to traditional heating methods, the performance of MW-based approaches was noted to be superior. Higher heating rates followed by lower activation energy, increased reaction rates, economic perspectives, and smaller equipment are the principal benefits of MW-based approaches. Furthermore, MW-based approaches also exhibit the following positive outputs: selective and volumetric heating capabilities, distinct thermodynamic functions than conventional heating, and good controllability (Wang and Wang, 2016). Because of the benefits listed earlier, MW-based approaches have received extensive attention for a variety of analytical and chemical applications, including soil remediation, polymerization processes, extraction in analytical chemistry, air purification, macromolecule degradation, and chemical catalysis (Wang and Wang, 2016).

Wastewater treatment and reuse were found to be a critical concern, and scientists are seeking low-cost, appropriate technology (Kivaisi, 2001). Water treatment technologies are primarily used for the following purposes: treating wastewater, reducing water sources, and recycling (Gupta et al., 2012). Unit functions and processes are currently merged to offer what is known as primary, secondary, and tertiary therapy. Primary treatment covers physical and chemical purification techniques, whereas secondary treatment is concerned with the biological treatment of wastewater (Gatidou et al., 2019). Wastewater after the first and second stages has then turned into high-quality water that can be used for a variety of purposes, including drinking, industrial, and medicinal supplies, in tertiary treatment processes. The tertiary process removes up to 99% of the contaminants and converts the water to a safe quality for a specified usage (Gupta et al., 2012). All three procedures are combined in a full water treatment facility to produce high-quality, safe water.

13.17 PRACTICAL IMPLEMENTATION OF TECHNOLOGIES FOR COMMERCIAL SCALE WATER TREATMENT

Despite the advancement of a plethora of technologies for water treatment, commercial scale treatment remains to be a difficult problem in the current scenario (Ahmad et al., 2020). It is important to consider how to manage the toxic elements that have been removed from the waste materials.

The water treatment approaches are primarily divided into three categories: primary, secondary, and finally, tertiary water treatment (Demirbas et al., 2017). Water is treated at the primary level employing the following processes: screening, filtration, centrifugation, coagulation, sedimentation, gravity, and flotation processes (Gupta et al., 2012). These procedures are typically employed when water is heavily contaminated. The matter has been converted into carbon dioxide, water, and ammonia gas by microbes, which are often bacterial and fungal strains. Organic matter is sometimes transformed into various compounds, such as alcohol, glucose, and nitrate. Furthermore, bacteria detoxify harmful inorganic materials. After that, the effluent should be devoid of hazardous organics and inorganics. The aerobic and anaerobic digestions of wastewater are included in biological treatment. The tertiary treatments are critical since they are employed to obtain healthy drinking water.

The major goal of the recommendations is to aid in the development of legislation for the management of wastewater while taking into account the unique characteristics of each country. These guidelines included an important microbiological examination for threat assessment, which included data gathering on pathogens found in water. Furthermore, the guidelines offer health risk management and preventive estimates of the level of wastewater usage (Jaramillo and Restrepo, 2017).

13.18 URANIUM REMOVAL AND NANOTECHNOLOGY-BASED TREATMENT

The remediation in this regard from drinking water can be achieved using different various technologies including adsorption and various biological strategies. The reduced initial cost of implementing the strategy for the same followed by ease of operation along with the reduced power requirement made the approach more prominent (Zhang et al., 2022). Carbon along with the multiple-walled carbon nanotubes as well as the covalent organic frameworks are noted as the common kind of nano-based approach mainly used. The adsorption capacity and mechanical stability of adsorbent can be enhanced by the introduction of graphene oxide (Zhao et al., 2018). Recent years have also witnessed the fact that there have been many studies focusing on graphene being used as an adsorbent for heavy metals.

13.19 REMOVAL OF EMERGING CONTAMINANTS (ECs)

The emerging contaminants, principally synthetic organic chemicals, are known to possess drastic environmental impacts. The toxic elements found in nature at relevant concentrations cause deadly effects on both human and aquatic populations. The main constituents of municipal sewage consist of the pharmaceutical industry waste, products of household, hospital waste, and wastes from natural aquatic habitats. The biological activated carbon treatment process can be used for the removal of the wide range of ECs and residual disinfection/oxidation products without generating toxic active products. In order to fight against the toxic elements, through the removal of various contaminants in the natural ecosystem, the application of the biological activated carbon treatment process in various patterns has been discussed in previous literature. Moreover, the environmental contaminants also exhibited a strong affinity towards the ozone and which can be explored for adsorbing such threatening contaminants (Yu et al., 2020).

13.20 MICROPLASTIC REMOVAL

The membrane bioreactor followed by the rapid sand filtration approaches efficiently removes microplastics. The membrane bioreactor usually explored the combination of membrane filtration processes towards the suspended growth biological reactors. Furthermore, the rapid sand filtration approach would be noted as one of the cost-effective ways that permit rapid removal of pollutants; it has the main disadvantage that it can fragment microplastics into smaller particles (Talvitie et al., 2017).

13.21 ELECTRODIALYSIS

Heavy metals are getting much more significant in the current scenario since such harmful pollutants possess carcinogenicity, toxicity, non-biodegradability, and persistence in the natural habitat. The electrodialysis of effluent from metal finishing and leather industries can be used for removing such heavy metal ions, including chromium, copper, nickel, and finally, zinc. Solutions that can be reused could be obtained through electrodialysis. Effluents from refineries, various industries, and power plants could be subjected to electrodialysis before it used. This approach is essential for the desalination of salty effluent (Gautam et al., 2016).

13.22 REMOVAL OF SELENIUM

Toxic selenite was distributed in the environment through natural and anthropogenic activity. Nanofiltration and reverse osmosis are implemented for the treatment process. He et al. (2016) developed a thin-film composite membrane encompassing zwitterionic co-polymer that has the ability to be soluble in water and has improved the efficiency. Reverse osmosis was also reported with greater output in this regard (Linares et al., 2016). A study concerning drinking water illustrated the fact the expensive operational cost of nanofiltration and reverse osmosis is not much more feasible than mesoporous-activated alumina.

13.23 REUSE OF WASTEWATER

One of the most well-known advantages of using wastewater in agriculture is the reduction of demand for freshwater supplies. As a result, wastewater is reported to serve as a substitute irrigation source, particularly for agronomy, which consumes approximately 70% of water available in nature (Jaramillo et al., 2017).

Furthermore, the reuse of wastewater boosts agricultural outputs in water-stressed areas. Hunger affects approximately 805 million population over the globe. To be successful, however, an inclusive way that merges both public and private investments focussed on enhancing agricultural output is required. Another advantage of wastewater reuse for agricultural needs is that the specific individual can reduce the cost required for extracting the groundwater. In addition, it is worth noting from this perspective that the energy which necessitates pumping the groundwater was usually noted to be 65% of the total expenditures required for the irrigation actions (Jaramillo and Restrepo, 2017). Another advantage of wastewater reuse in agriculture would be the reduction of water contamination. Low-cost wastewater treatment systems, accomplished through specific scientific alternatives that fulfil the goal of reuse in agricultural perspectives, may potentially be a viable option in locations where climatic and geographic conditions allow (Jaramillo and Restrepo, 2017).

Wastewater usage in agriculture aids in the liberation of capital resources of various countries. Some of the countries have got advantageous effects in this regard, since the reuse of wastewater may help to reduce municipal costs associated with finding novel water sources with the aid of more expensive approaches. Based on regulatory considerations, the reuse of wastewater in agricultural fields can help to justify appropriate investment plans and funding mechanisms for pollution management and prevention (Jaramillo and Restrepo, 2017).

13.24 SLUDGE MANAGEMENT

Sludge management generated by wastewater treatment plants is recognized as one of the most predominant problems that is to be addressed in the current scenario in developing countries (Spinosa et al., 2011). This is due to the fact that sludge produced by wastewater treatment plants (WWTPs) accounts for just a small percentage of the wastewater that has been previously treated with various technologies. Furthermore, the necessity to develop an effective sustainable sludge management approach has become a major fear in recent years. In this point of view, the development of new methods to maximize the recovery of valuable materials in a sustainable line has been noted to be essential.

When discussing the advancement of human activities, the term "sustainability" is now commonly employed. Three elements are critical to achieving effective sustainability and cannot be considered separately: the environmental, economic, and social elements (Kibert et al., 2000).

> The approach concerning management of water resources must be approached from a multifaceted standpoint. In this regard, the European Water Framework Directive has focused on economic research new policies for efficient water resources management (Riegels et al., 2013). Because it has been previously reported as a reasonable and methodical decision-making assistive tool. Furthermore, the various processes involved in wastewater treatment also exhibited substantial environmental profits. Currently, sludge management solutions are frequently inefficient and unsustainable, owing to the fact that no single procedure or treatment can handle all three of the aforementioned elements (Neumann et al., 2016). As a result, the development of efficient procedures for sustainable sludge management should necessitate the evaluation routes of management, capable of maximizing recycle output through systems with low energy impact. Depending on the local context, burning dry sludge with the probability of phosphorus recovery from ash is a feasible option.
>
> (Spinosa et al., 2011)

Based on specific operations for sludge management, a group of scientists from various countries over the globe with varying backgrounds volunteered their knowledge and began the integrated system construction for sustainable sludge management (Strauss and Montangero, 2002). The system incorporates numerous processes, each of which is capable of recovering different materials followed by energy from sludge and was designed to use and reuse alternatives rather than disposal options. The system is easily adaptable to varied local settings, with the ability to preferentially generate one or more materials over others, depending on local variables.

13.25 NUTRIENTS RECOVERY

Wastewater nutrient recovery has the probability to improve the sustainability of the water and agricultural industries (Xie et al., 2016). Forward osmosis and membrane distillation, followed by electrodialysis, are the important three novel membrane techniques that potentially increase wastewater nutrient recovery (Xie et al., 2016). Forward osmosis is always witnessed as a process in which the water is passed from a feed solution through a semipermeable membrane because of the osmotic pressure gradient found across the semipermeable membrane. Membrane distillation usually allows the researcher to treat the wastewater possessing high salinity. According to the major principle behind the same, it was evident that a vapour pressure gradient is generated by a specific temperature differential over the porous membrane which is previously known to possess hydrophobic nature. The vapour generated from the feed distributes over the selected membrane and finally results in permeate side condensation. The hydrophobic nature of the membrane also prevented the water molecules from moving over the pores.

The electrodialysis comprises passing the wastewater between the two specific plates having opposite electrical charges. The non-metals in the wastewater are primarily attracted towards the positive pole, while the metals are prominently migrated towards the negative charge. Finally, both negative and positive ions can be detached and discarded from the plates.

Another aspect of wastewater treatment is based on human urine. Human urine records for approximately 80% of the total nitrogen load and 40–50% of the overall phosphate burden (Theregowda et al., 2019). An integrated approach for urine and wastewater treatment process with single high-rate ammonium removal over nitrite and anaerobic ammonium oxidation processes was previously reported by Wilsenach and van Loosdrecht (2006). The model study revealed that integrated wastewater treatment with more compact and energy-efficient procedures can be viable in certain circumstances where the urine was collected (50% or more) and treated separately (Park et al., 2015). The fundamental advantage of urine separation is that it produces the same high-quality effluent while saving a significant number of resources (Wilsenach and van Loosdrecht, 2006).

Recover technology is another major aspect of wastewater treatment. Normally, wastewater contains several kinds of components mainly metals nutrients, etc.; with the aid of recover technology, we can recover certain kinds of useful products from the wastewater and indirectly it helps in wastewater treatment also. One of the best examples was the recovery of phosphorous from waste. Phosphorus is abundant in the ash produced by the combustion of sewage treatment sludge. However, no practical phosphorus-recovery technique has been discovered to date; therefore, these ashes must be disposed of according to the requirements. Because phosphorus extraction is limited to a very narrow geographic area on Earth, alarms concerning the shortages of the same in the near future are largely justified. Several investigations on the recovery of phosphorus from ash (the final part of the wastewater treatment sludge) have been performed in an effort to lower the costs allied with waste disposal (Shaddel et al., 2019).

Research on recovery technology for phosphorus primarily uses the solvent-extraction method. However, the approach requires an extensive volume of organic solvents along with critical difficult operation processes. While comparing the same with other approaches, it was also noted as a less expensive and simpler approach. Sewage treatment sludge incinerated ash is mostly composed of the following elements: Al, Ca, O, Si, and Fe; there are several applications for inorganic raw materials (Xu et al., 2008). The formed ash is regarded to be the most readily obtainable for use. However, this ash is known to encompass significant levels of toxic elements, including heavy metals, phosphorus, aluminium, and the components in the ash have negative impacts on the characteristics of cement, thus eliminating these elements is critical. Many acids, including sulphuric acid and hydrochloric acid, can dissolve phosphorus, aluminium, and heavy metals at pH levels lower than 2.0. The aluminium phosphate eluted can be precipitated by adding the following alkalis: sodium carbonate calcium, sodium bicarbonate, and sodium hydroxide (Swami et al., 2019). In a similar way, the heavy metals dissolved in various elements, including phosphorus and nickel, can be precipitated as hydroxide by adding alkalis at pH 4–10. As a result, based on the basic considerations of such ash usage technology, the elimination of phosphorus, aluminium, and a plethora of heavy metals from these kinds of ashes was anticipated to be efficient.

The significance of technology for reusing recovered materials cannot be overstated. The phosphorus-free acid residue can be used as a raw element for the cement industry. The recovered phosphorus using the above-said approaches, primarily encompassing aluminium phosphate, has a variety of following potential applications: as an absorbent and a raw element in glass manufacture. However, it was also evident that the usage of aluminium phosphate at industrial frequency, recovered from the above-mentioned approaches, has not been fully unveiled, and the use has been extremely restricted to date (Takahashi et al., 2001). For these reasons, it was suggested that, either new applications or interventions should be developed to convert the aluminium phosphate to more beneficial phosphate compounds. In light of probable future perspectives concerning the phosphorus resource constraint due to declining worldwide supplies, the strategies that combined the removal and recovery of phosphorus from wastewater is a logical and sustainable solution. Precisely, the recovery of phosphorus from wastewater or any other waste resources could extend the life of accessible stocks thereby improving environmental sustainability (Castaño-Trias et al., 2020).

The great demand for various elements, including ammonium as well as phosphate, which are used in a plethora of industries (fertilizer), is ascribed to industrial nutrient shortages, especially given the extremely growing population rate over the globe. Many technologies are now being researched for enhancing the efficacy of nutrient recovery process. The conventional approaches, including chemical precipitation and adsorption, as well as more modern strategies, such as osmotic membrane bioreactors and bioelectrochemical systems, have gained much more significance in the aforesaid context (Ye et al., 2017).

Also, Increased food consumption places a great strain on natural resources. Excessive N and P fertilizer uses have a wide range of severe environmental consequences, including eutrophication. The recovery of nutrients in a concentrated solid form, such as sparsely water-soluble crystalline struvite ($MgNH_4PO_4H_2O$), is attractive due to the simplicity of handling, shipping, storage, and regulated reuse, as well as the reduced infection risk. Mg source is the key to cost-effective and long-term struvite recovery. Although Mg is a common element of many minerals, accounting for 2% of the earth's crust, the majority of soil Mg (98%) is incorporated in the crystal lattice structure of poor solubility minerals, making it inaccessible to plants (Ash et al., 2019). Recent studies also validated that phosphorus recovery using chemical approaches necessitates much more resources than other conventional approaches, which justified the need for developing an effective strategy in this regard. According to the LCA analysis by certain previous investigations, it was evident that choosing an effective recovery method for phosphorus should take into account local conditions as well as the environmental implications (Bradford-Hartke et al., 2015).

13.26 ENVIRONMENTAL BENEFITS AND CHALLENGES

As we know, the health of the human population has been always regarded as a critical factor for the existence of many organisms, including the human population, indicating the relevance of maintaining better health for the human population. The emergence of advanced hospitals over the globe for human population and other animals, including various pets, verified the aforesaid perspective. However, the emergence of such approaches for better health has also instigated drastic effects on the environment also. In this regard, this chapter conclusively analysed the environmental benefits of waste management and future perspectives with special reference to the recovery technology and the negative impacts of such things against the environment. Due to the increased population rate, the entire planet faces the extreme issue of freshwater availability and it demands the wastewater treatment (Bouwer, 2000). With minimal adverse environmental impact, the contaminated water can be treated in a sufficiently clean state. As mentioned in the former sections, the natural water resources that have been primarily contaminated by humans and animals or by any means are restored to desirable quality through chemical, biological, physical, or a combination of all is clearly depicted as wastewater treatment. The length of the process depends on the quality of water required (Di Iaconi et al., 2020).

Due to the constant rise in population along with the overdependence on diminishing water resources over the last few years, quantity of municipal wastewater has drastically increased. The discharge of contaminated water not only threatens the micro-macroflora and fauna but also affects the good quality of water essential for all socio-economic functions. Wastewater treatments help in preventing unnecessary water loss and thereby saving water (Bouwer, 2000). At present the reuse of wastewater is essential in order to meet the water requirement as the population increases and recycling polluted water ensures environmental health. A significant portion of pharmaceuticals from hospitals and drug-manufacturing industries are being released into the water bodies causing a major issue of contamination. The introduction of these pharmaceutical compounds as well as resistant microbes, these companies causes an extreme hazard to the environment.

Pharmaceutically active compounds, from a plethora of organisms belonging to the group of microbial consortia, have been found in the wastewater discharged from hospitals (Majumder et al., 2021). Compared to domestic wastewater, hospital wastewater has previously reported greater biochemical oxygen demand followed by chemical oxygen demand, and high levels of nitrogen and ammonia (Majumder et al., 2021). It is extremely difficult to treat hospital wastewater by conventional biological methods since the biodegradability index of wastewater from the hospital is lower than municipal wastewater. The viruses, as well as the bacteria and various pathogens from the hospital wastewater, still remain a great threat to the environment because of their release into the aquatic ecosystem (Majumder et al., 2021). The most widely used biological processes in wastewater treatment plants worldwide include activated sludge processes (AS) and it has been used for the elimination of contaminants for more than a century due to their great nutrient removal, toxin degradation, and biomass retention capabilities. The performance and functional stability of WWTPs depend on the microbial community structure and diversity. Collectively, this part also remembers us the fact that the life of various animals, including the human population, is primarily threatened by our uncontrolled and worsening activities, which directs the scientific community to rectify the concerns formed from the aforesaid activities through a simple, but effective approach, precisely the recovery technology.

13.27 CONCLUSIONS AND PERSPECTIVES

The nutrient reuse as well as the recovery of various forms of energy from the sewage sludge has traditionally got much more significance and hence it has been accomplished through a variety of approaches. Sewage sludge usually represents the by-product of treated wastewater which consists of various following elements, including organic and inorganic elements, organic chemicals, plant nutrients, and pathogens. The treatment has been primarily accomplished through four stages depicting thickening, digestion, dewatering, and disposal. The implementations of membrane-based approaches, precipitation, extraction, biogranulation technology, MW chemistry, nanotechnology, and electrodialysis have also provided better output in the current scenario. Efforts based on these kinds of approaches, especially the special focus on recovery technology, direct the scientific community to rectify the concerns allied with sludge waste management.

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14 Biostabilization of Sewage Sludge

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14.1 INTRODUCTION

Globally, approximately 48 million dry tons of sewage sludge, a by-product of primary and secondary wastewater treatment, are produced each year (Posligua et al., 2018; Bagheri et al., 2023). As the global population increases, the quantity of sewage sludge is expected to increase, posing a threat to the environment, climate, and human health. It has been reported that of the total amount of wastewater, only 1–2% is sewage sludge; however, it contains almost all the original compounds that are responsible for pollution and health problems. Among the most prominent compounds, pathogens, putrescible organic waste, antibiotics, and heavy metals are the most abundant; therefore, an improper treatment of sewage sludge before disposal can lead to water bodies' contamination, uncontrolled methane emissions, and human diseases. (Fijalkowski et al., 2017; Spinosa & Doshi, 2021).

Sewage sludge contains 50–70% organic matter, 30–50% mineral compounds, 3.4–4.0% nitrogen, 0.5–2.5% phosphorus, and micronutrients making it a potential substrate for renewable energy and material recovery (Fijalkowski et al., 2017; Rorat et al., 2019). In developed countries, the fate of sewage sludge is predominantly agricultural use, with 40% and 47% of all sludge in the European Union and the United States, respectively, being applied to agricultural land (Kelessidis & Stasinakis, 2012; Di Giacomo & Romano, 2022). Policies that reflect a shift toward a more circular economy, such as Best Practicable Environmental Option in the United Kingdom, encourage agricultural land application compared to other disposal methods, such as incineration, landfilling, and discharge into water bodies, and thus allow the recovery-reutilization of nutrients present in sewage sludge (Christodoulou & Stamatelatou, 2015; Lombardi et al., 2022). However, every disposal method, excluding incineration, requires treatment or stabilization, to ensure that the surrounding environment is not negatively impacted.

Particularly, the stabilization process in sewage sludge aims for the reduction of pathogens, elimination odors, and prevention of decomposition by rendering the biodegradable fraction resistant to decomposition. Disposal methods such as incineration are examples of thermal stabilization since the organic matter in the sludge is being destroyed at high temperatures. Other methods to achieve stability included chemical stabilization, where a high pH environment created by lime eliminates microbiological risk, and biostabilization, which relies on microorganisms to digest the biodegradable fraction (Rorat et al., 2019). The process used for stabilization depends on factors such as geography and demographics as well as the fate of the sludge (Christodoulou & Stamatelatou, 2015). For example, due to its high population density, Japan relies heavily upon incineration to decrease the volume of produced sludge. Ultimately, from a sustainability perspective, there are two goals of stabilization: (i) decrease the negative impact in the environment and (ii) harness the energy and resource potential that sewage sludge offers.

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Anaerobic and aerobic digestion are the two types of biostabilization, where organic matter is rendered resistant to decomposition by organisms, which can be assessed using biological oxygen demand and chemical oxygen demand (Zheng et al., 2022a). Given its ability to eliminate pathogens while retaining similar nutrient characteristics to the raw sludge, biostabilization is the most favored method to stabilize sludge around the world. It's well known that there are lot of researches performed in both mentioned methods, but still are challenges and research gaps that need to be addressed during its use to stabilize sewage sludge more efficiently. Among the main challenges that need to be addressed is the energy use and recovery to make the stabilization process more feasible, the use of sequential and alternative methods, the decrease or elimination of pathogens, maximize the nutrients recovery, as well as the use of stabilized sludge to decrease the environmental pollution. In this sense, the main aim of this chapter is to offer an overview of sewage sludge stabilization through anaerobic, aerobic, and sequential digestion processes.

14.2 ANAEROBIC DIGESTION

Anaerobic digestion is a biostabilization process that occurs in the absence of oxygen (El-Fadel et al., 2013). During this process, the symbiotic obligate microorganism and facultative anaerobes decompose the organic matter in sewage sludge into simpler compounds, decreasing the sludge volume, with the concomitant production of a nutrient-rich digestate and biogas rich in methane (Babson et al., 2013). Anaerobic digestion was first used as a method of wastewater treatment in the late 1800s; this process has been applied to numerous organic wastes, including municipal solid waste, animal manure, and food residues among others (Abbasi et al., 2012). Currently, anaerobic digestion continues to be one of the most effective methods used for the biostabilization due to the following characteristics:

- Biogas as by-products can be captured and used as a renewable energy source to produce electricity and heat to replace fossil fuels (Møller et al., 2009)
- Nutrients such as nitrogen and phosphorus in anaerobic digestate can be recovered through agricultural land application or extraction processes (Rorat et al., 2019)
- Potential odor impact of digestate can be reduced by up to 98% from sewage sludge (Orzi et al., 2015)
- Pathogen levels are significantly reduced without the need for additional treatments (Liew et al., 2022)

Anaerobic digestion is a series of four metabolic stages, defined by the microorganisms and reactions responsible for each transformation of carbon (Figure 14.1). Methanogenesis is the key to successful biostabilization, as the production of methane and carbon dioxide shows that the organic matter originally present in the sewage sludge has reached a state where it is resistant to further decomposition (Lü et al., 2018). In fact, the biochemical methane potential test is used to evaluate the biostability of anaerobically digested sludge, using methane generation to estimate the degree of biodegradability (Filer et al., 2019). Another method of evaluating biostability is the detection of odor intensity, which correlates to the presence of organic matter. The above is possible during acetogenesis where the volatile organic compounds that cause odors are degraded to acetate (Orzi et al., 2015). While the focus of biostabilization is the conversion of organic carbon into biogas, anaerobic digestion also leads to the production of ammonia from organic nitrogen (Babson et al., 2013). The production of ammonia can contribute to the development of toxic conditions for the microorganisms, affecting negatively the anaerobic digestion, which in conjunction with substrate competition leads to the reduction of pathogens in the digestate (Orzi et al., 2015).

The anaerobic digestion process begins with hydrolysis, where extracellular enzymes secreted by microorganisms convert complex molecules into simpler compounds (Richard et al., 2019). Then, during acidogenesis, compounds produced during hydrolysis are converted into volatile fatty

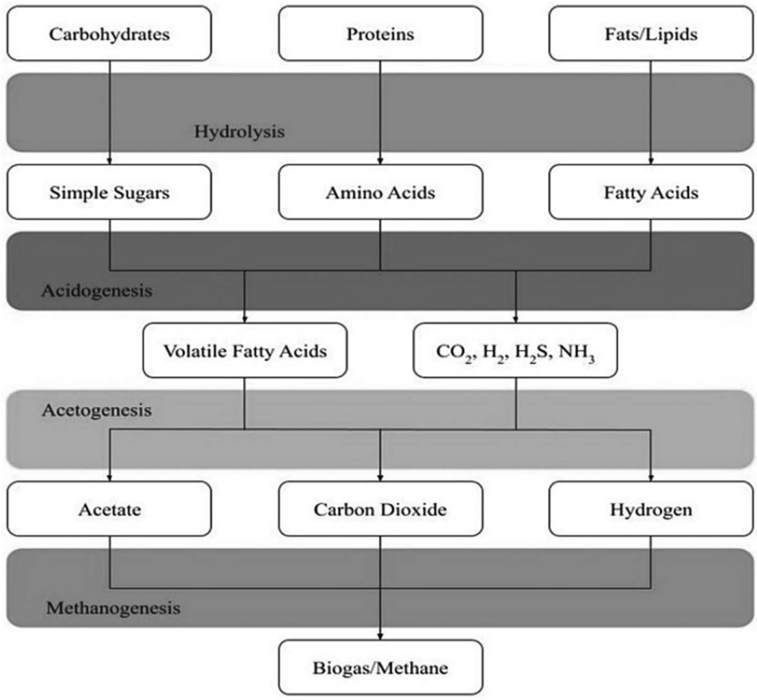

FIGURE 14.1 Biogas production pathway using sewage sludge as a substrate.

acids (VFAs) and inorganic compounds by facultative and obligate anaerobic bacteria. After that, acetogenic bacteria convert VFAs into acetate and hydrogen. In the final step, obligate anaerobic methanogens produce methane and other gases through the acetolactic or hydrogenotrophic methanogenesis (Meegoda et al., 2018).

Methanogenesis is the stage of anaerobic digestion that is most sensitive to environmental changes. As a result, conventional anaerobic digestion (CAD) takes place under mesophilic conditions (30–40°C) and has a total solid (TS) content of 3–6% (Li et al., 2017). Compared to thermophilic conditions (50–60°C), the microbial community present in mesophilic conditions is more diverse and dynamic, making the process more stable to environmental changes (Gebreeyessus & Jenicek, 2016). Due to the sensitivity of essential microorganisms, system parameters such as pH, mixing, and organic loading rate are controlled throughout CAD (Li et al., 2022). CAD is a one-phase process, where the entire process occurs under the same process conditions in the same reactor (Møller et al., 2009). Work has been done to develop two-phase systems, which would have different operating conditions for the acidogenic and methanogenic stages and increase the reaction efficiency (Blumensaat & Keller, 2005). However, high operational costs, technological constraints, and increased potential process instability have prevented the implementation of two-phase anaerobic digestion systems.

Thermophilic anaerobic digestion (TAD) is carried out in the same way as CAD, with a difference in operational temperature. In this process, the higher temperature (50–60°C) accelerates the biochemical reactions, shortening the hydraulic retention time required for biostabilization. Higher temperatures also contribute to enhanced pathogens removal, both by increasing reaction rates and creating a more hostile environment. Additionally, it has been reported that TAD has higher biogas yields, making it a better choice for waste conversion to produce renewable energy (Yu et al., 2014).

Both processes (conventional and thermophilic) are examples of wet anaerobic digestion because both involve the treatment of sludge with a TS content of 10%. High-solid anaerobic digestion

(HSAD) and thermophilic HSAD (THSAD) are both types of dry anaerobic digestion where the TS content in the sludge is higher than 20% (Abbassi-Guendouz et al., 2012). Operationally, dry AD introduces a dewatering step before the digestion begins, increasing the TS content and reducing the volume of sludge. An advantage to having lower volumes of sludge is the decrease in digester size, making the implementation of anaerobic digestion in wastewater treatment plants more feasible (Di Capua et al., 2020). However, HSAD and THSAD tend toward instability, as high solid content increases the risk of high ammonia accumulation, which is responsible for inhibiting the microbial activity. Due to the above, the integration of in situ ammonia removal processes has been shown to be necessary, but effective, to remove the excess ammonia and enhance the microbial activity (Bi et al., 2020). As is expected with thermophilic conditions, THSAD exhibits a better reduction of volatile solids and consequently a higher biogas production (Xu et al., 2020).

Regardless of the TS content of sewage sludge, hydrolysis is the rate-limiting step in anaerobic digestion. Hydrolysis is essential for biostabilization due to the presence of complex biopolymers, for instance, cellulose is degraded into monomers that can undergo mineralization, humification, and gasification during the later stages of AD (Lü et al., 2018). However, breaking these large molecules takes time, which increases the rate of hydrolysis and increases the hydraulic retention time required for biostabilization. Figure 14.2 shows a strategy to decrease the quantity of unhydrolyzed organic matter and prepare the organic matter for degradation before anaerobic digestion, pretreated sludge is used to reduce the initial organic matter in the process, where the CAD systems involve a mesophilic or thermophilic digestion step, shown by paths 1 or 2, then a dewatering and disposal step, shown by path 3. Alternative methods toward improved biostabilization include a pretreatment step before digestion (1a or 2a) and nutrient recovery before disposal (3a) (Neumann et al., 2016).

Table 14.1 shows the four types of pretreatment strategies to optimize the hydrolysis step in anaerobic digestion (Richard et al., 2019). Although all the pretreatment strategies result in an increase in AD efficiency, they typically increase the energy demands of the system. However, historically, physical as well as chemical pretreatment has been the predominant strategies, but in

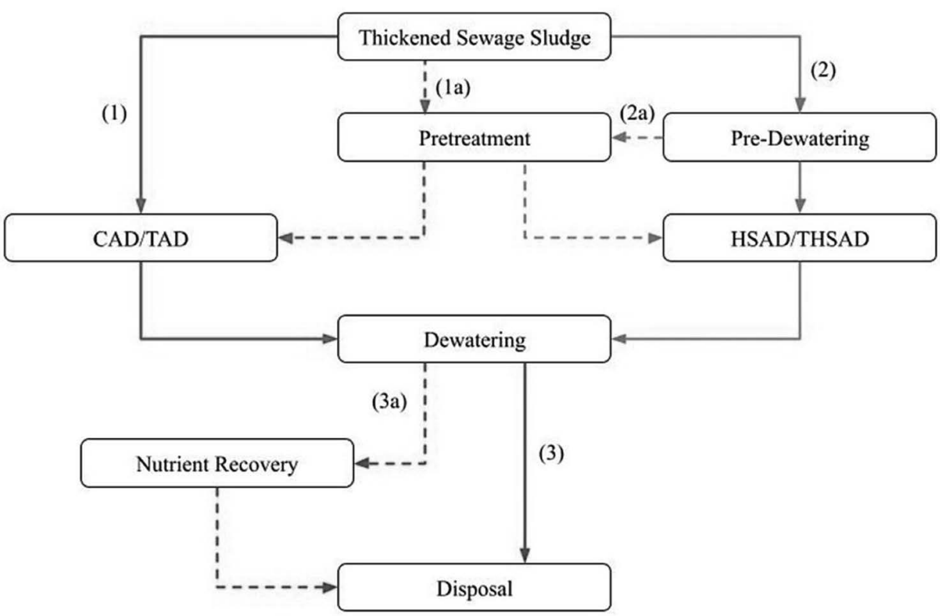

FIGURE 14.2 Differences in anaerobic digestion systems and the potential pathways to improved biostabilization. (Modified from Li et al. 2017.)

TABLE 14.1
Use of Pretreatment on Anaerobic Digestion

Strategy	Examples	Advantages	Disadvantages
Mechanical	Ultrasound, microwave, electrolysis	• Reduction in particle size increasing area available for biodegradation • Increased removal of some organic pollutant	• High energy demand
Chemical	Acid or alkaline treatment, ozonation, Fenton oxidation	• Enhances removal of lignin • Enhances chemical oxygen demand, accelerating methane production	• Increased possibility of inhibitory chemical production • High cost
Thermal	Low-/high-temperature treatment, freezing/thawing	• Accelerate lignin solubilization reducing hydraulic retention time • Pathogen removal	• Leads to increased digestate ammonia concentration
Microbial	Enzyme addition, aerobic digestion	• Increased methane yield • Low energy demand • Increased microbial diversity, increasing process stability	• Slowest pretreatment method

recent years the usage of these with microbial pretreatment has increased, due to a need to decrease the energy demands in the process (Neumann et al., 2016).

In addition to biogas generation, anaerobic digestion is valued as a method of biostabilization because it produces a nutrient-rich digestate by-product that can be used as biofertilizer to replace inorganic chemical fertilizers (Cheong et al., 2020). Compared to its raw form, digested sewage sludge can have higher levels of ammonia, increasing its effectiveness as a fertilizer (Walsh et al., 2012). Unfortunately, anaerobic digestate has considerably high concentrations of heavy metals compared to raw sludge, due to the decrease in volume throughout the process (Zheng et al., 2022b). For this reason, the application of digestate on agricultural land, or even the disposal of digestate in landfills, can lead to the contamination of groundwater and soils with considerable amounts of heavy metals, impacting human health (Fijalkowski et al., 2017). The removal of heavy metals is not a straightforward process, as each metal behaves differently. As an alternative, it has been reported that the addition of biochar to immobilize heavy metals is the most prominent method discussed in the literature (Liew et al., 2022). Alternatively, nutrient recovery can decrease the heavy metal contamination, by removing key nutrients from digestate that can be used for later agricultural use. One common method of recovering phosphorus and nitrogen from digestate is struvite precipitation, which produces magnesium ammonium phosphate that can be used as a fertilizer (Uludag-Demirer & Othman, 2009).

14.3 AEROBIC DIGESTION

Aerobic digestion is a biostabilization that occurs in the presence of oxygen. During aerobic digestion, bacteria and fungi decompose the multimolecular organic matter in sewage sludge into a digestate rich in humic substances with carbon dioxide, water, and ammonia as a by-products (Sang et al., 2012; Katipoglu-Yazan, 2015). This biostabilization method is ideal for small wastewater treatment plants, primarily because they lack the capital necessary to invest in more expensive technologies such as anaerobic digestion (Hallas et al., 2019). Compared to anaerobic digestion, aerobic digestion is less prone to process failure and has more straightforward operational controls, making it easier to implement on a smaller scale (Tas, 2010). For instance, in wastewater treatment plants operating at

20,000 m^3 or less per day, aerobic digestion is the most environmentally friendly choice for biostabilization, as the final product can be better controlled (Bernard, 2000; Ranieri et al., 2021). Additionally, aerobic digestion has several advantages making it a useful method of sewage sludge stabilization for three main reasons: (i) the sludge retention time is much shorter than anaerobic digestion, even without pretreatment, (ii) the heat generated through exothermic biochemical reactions naturally decreases the biopolymer degradation times, and (iii) the pathogen inactivation is effective enough to produce digestate that conforms to high biosolid quality standards (Zhang et al., 2016).

During aerobic digestion, different microbial processes occur in parallel. Complex biopolymers such as lignin and cellulose are broken down by fungi and bacteria, which produce carbon dioxide and water, and ultimately stabilized humic compounds (Sang et al., 2012). Meanwhile, nitrifying bacteria convert ammonia into nitrate and nitrite, which significantly reduces the odor of the sewage sludge. Specifically, anoxic conditions within the sewage sludge can allow denitrifying bacteria convert nitrate and nitrate into nitrogen gas, reducing the environmental impact of digestate upon disposal. However, once the carbon source from sewage sludge has been depleted, microorganisms begin to consume their own protoplasm for cell maintenance, in a process known as endogenous respiration, in which during aerobic digestion, microorganism go through a lag, logarithmic, and stationary phase, consuming the carbon present in the sewage sludge. Instead of undergoing a typical death phase, microorganism go through endogenous respiration, where they consume their own biomass as a carbon source (Ikumi & Harding, 2020). This type of "cannibalization" is ultimately the mechanism responsible for the reduction of digestate volume as well as the sludge stabilization due to the biomass has been reduced to a very low concentration (Friedrich and Takacs, 2013). As a matter of fact, the most common method for analyzing the biostability of aerobically digested sewage sludge is the respiration index, which measures the consumption of oxygen in terms of biomass concentration (Lü et al., 2018). The respiration index (static and dynamic) is measured during the aeration of sludge stabilization. However, it has been reported that the dynamic respiration index is the most accurate method to evaluate the stabilization process over time (Scaglia et al., 2013; Policastro & Cesaro, 2023).

On another hand, the two operational parameters that have the largest impact on the progression and success of aerobic digestion are aeration and waste loading. For most of the biochemical reactions carried out during aerobic digestion, oxygen is required, converting the oxygen concentration as the limiting factor in the stabilization process. Low oxygen concentrations during the stabilization process result in a deficient microbial metabolism, where the microorganism breaks down organic material, resulting in the accumulation of VFAs (Obeta Ugwuanyi et al., 2005). High concentrations of VFAs are detrimental to microbial activity, decreasing the performance efficiency of aerobic digestion systems, leading to the retention of biodegradable organic matter. Unlike anaerobic digestion where VFAs are simply an intermediate, in aerobic digestion, VFAs are a by-product that is responsible for unwanted odors, highlighting the fact that a proper aeration is essential for overall sewage sludge odor reduction. Similarly, the feeding rate into the bioreactors can lead to thermal or osmotic shock conditions, due to the high solid content, which causes an increase in oxygen demand, impacting directly the microbial community and its metabolic capacity to growth (Layden et al., 2015). For example, thermal shock causes a sharp drop in temperature, shocking thermophilic microorganisms, which highly affects the microbial performance during the stabilization process (Kirilova et al., 2020). In this sense, for both situations, a decrease in microbial activity and an increase in VFA concentration have been observed, leading to a poor biostabilization performance (Obeta Ugwuanyi et al., 2005).

Traditional aerobic digestion is composting, which uses a bulking agent to create space for aeration. Typically, biochar is used for this purpose, as its physical properties promote aeration in the composting process, with the concomitant improvement of microbial activity as well as the prevention of nitrogen lost (Li et al., 2019). While the implementation of composting is simple and effective, the process is lengthy and results in the loss of nutrients. Conventional aerobic digestion is a more industrially applicable process known as autothermal thermophilic aerobic digestion

(ATAD). Since heat is a major by-product of aerobic digestion, the thermophilic conditions have an advantage over the heat generated by the biological activity (Zhang et al., 2021). These naturally occurring high temperatures not just help to accelerate biomass degradation, but also shortening the sludge retention time, decreasing the energy input required for temperature control. Additionally, high temperatures are the primary mechanism for pathogen inactivation in the aerobic digestion (Liu et al., 2012). Specifically, a reliable achievement of temperatures higher than 45°C makes it possible for digestate produced from ATAD to be applied to land without additional processing (Kirilova et al., 2020). In practice, ATAD is typically a two-stage process, where the first and second stages are operated at different temperatures. However, in recent years, the one-stage ATAD systems have gained interest because they are simpler units that occupy less area while achieving equivalent stabilization to two-stage systems (Liu et al., 2012). Nevertheless, ATAD systems require high investments and their operation is costly, as significant aeration is necessary for its successful performance (Zhang et al., 2022).

Other potential biostabilization processes include vermicomposting, a type of aerobic digestion that uses earthworms as a biological factor. The interest in vermicomposting for stabilization of sewage sludge is mainly due to its being a lower tech method and as a consequence more economical compared to autothermal thermophilic aerobic digestion, but faster than traditional composting (Gupta & Garg, 2008). Unlike other forms of aerobic digestion where mechanical pretreatment of sludge to reduce particle size is a separate energy-consuming step, vermicomposting naturally incorporates both. Earthworms added to sewage sludge consume and digest the organic matter, leading to particle fragmentation which leads to the production of a fine humified product (Vuković et al., 2021). Aerobic and anaerobic microorganisms from the earthworm gut helps to the microorganisms already present in the sewage sludge to promote and enhance the conversion of organic matter into a stabilized product. Moreover, due to earthworms' characteristics to be adapted to survive in severe environments, vermicomposting does not result in thermophilic temperatures, making the reduction of pathogens highly dependent on the production of antibacterial coelomic fluids and the digestion of protozoa, bacteria, and fungi by earthworms (Sinha et al., 2010). Vermicomposting can sufficiently stabilize sewage sludge, achieving volatile solids reductions of up to 66.6%, which is similar to other forms of aerobic digestion and even anaerobic digestion (Zhao et al., 2010).

A primary goal of aerobic digestion is obtaining a final biostabilized product that can be safely disposed to the environment. Likewise, the biostabilized aerobic digestate is valued as a biofertilizer and soil conditioner, where not only do the humic compounds improve the soil structure and increase humus content but also contribute to the nitrogen fixation making the nutrients present in the digestate plentiful and accessible (Lloret et al., 2012). While methods such as autothermal thermophilic aerobic digestion produce digestate that can be applied to agricultural land without restriction, environmental concerns still exist. For instance, the accumulation of heavy metals throughout the aerobic digestion processes can affect human health as well as plant growth when the sewage sludge is disposed of or in agriculture. A common option to mitigate the accumulation of heavy metals is by increasing the pH of sewage sludge using alkaline materials such as coal fly ash, which can decrease the metal availability to form metal complex at the expense of microbial activity (Wong & Fang, 2000; Kumar et al., 2019). Conversely, the addition of amendments materials during vermicomposting can lead to a decrease in heavy metal concentration without impacting the earthworm activity. During vermicomposting, heavy metals are regulated inside of earthworm tissues and excreted to form complexes with amendment materials, reducing the metal availability and mobility (He et al., 2016).

14.4 SEQUENTIAL DIGESTION

Independently, aerobic, and anaerobic digestion can meet the requirements for the biostabilization of sewage sludge by reducing the biodegradable fraction and improving sludge quality. Compared to aerobic digestion, anaerobic digestion is a more energy-efficient process, as energy can be recovered

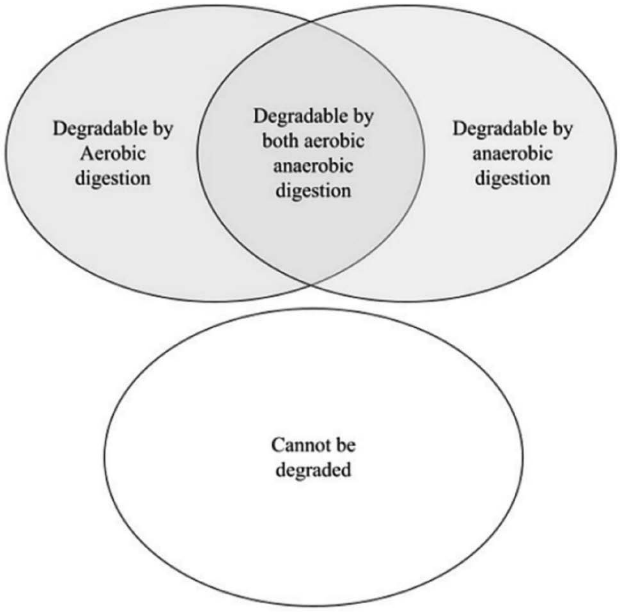

FIGURE 14.3 Representation of sewage sludge stabilization using sequential process.

in the form of biogas. Meanwhile, sensitive operating parameters make anaerobic digestion more difficult and costly to operate, making the aerobic process a more reliable choice for biostabilization. Figure 14.3 shows the integration of aerobic and anaerobic digestion into multi-stage sequential digestion systems that can help to alleviate some of the limitations observed in both processes and can be improved independently, to enhance the process performance (Tomei et al., 2011). Aerobic and anaerobic digestion degrades organic matter in different environmental conditions, which means that certain fractions are degraded under only anaerobic or aerobic conditions. However, neither aerobic nor anaerobic digestion is independently capable of degrading all of the organic compounds in sewage sludge. While the integration of a combined system (aerobic and anaerobic digestion) as a sequential process has the ability to achieve a higher level of stabilization, by degrading a greater portion of organic matter, (Kim & Novak, 2011).

Sequential anaerobic-aerobic digestion is the operation performed firstly in anaerobic reactors and then in aerobic reactors, independently but in series. One of the main differences between anaerobic and aerobic digestion is the presence of nitrifying and denitrifying bacteria that is higher during aerobic process in comparison with anaerobic conditions where the metagenomic consortia are highly abundant. For this reason, the anaerobic digestate can be rich in ammonia which in the agricultural application is less preferred instead of nitrate and nitrite. Nonetheless, an excessive amount of nitrogen in any digestate disposed of through land application can also lead to the accumulation of nitrogen in watersheds, causing eutrophication, as well as through filtration, which can contaminate groundwater sources decreasing the water quality. Running sewage sludge through aerobic digestion after anaerobic digestion makes it possible to reduce the ammonia concentration and overall nitrogen concentration. For instance, Kim & Novak (2011) found that through sequential digestion, 90% of ammonia and 50% of all nitrogen can be removed during the aerobic digestion step. Furthermore, it has been reported that CAD and sequential anaerobic-aerobic digestion increase the removal of organic matter from sewage sludge from 50 to 62% (Novak et al., 2011). Overall, anaerobic-aerobic digestion retains the biogas production from single-stage aerobic digestion processes while achieving better stabilization and a better suited digestate for land application, making it the most promising digestion configuration (Wang et al., 2021).

During sequential aerobic-anaerobic digestion, sewage sludge is subject to aerobic conditions before undergoing anaerobic digestion. In aerobic-anaerobic digestion, often the primary role of aerobic digestion is to act as a pretreatment for anaerobic digestion. Most simply, aerobic digestion can begin the hydrolysis process, decreasing the solid retention time during the anaerobic digestion (Jang et al., 2014). Likewise, the unwanted production of VFAs during aerobic digestion is beneficial in a sequential system because their presence can support the establishment of anaerobic populations, accelerating the digestion process by reaching the methanogenic phase in a short period of time (Obeta Ugwuanyi et al., 2005; Xu et al., 2016). In terms of energy recovery, achieving the methanogenic phase in a short time is beneficial for biogas production, with an increase of 42% in methane production over processes using a single anaerobic digestion (Jang et al., 2014; Xu et al., 2016). Furthermore, aerobic-anaerobic digestion can improve the performance of anaerobic digestion, by decreasing up to 15% of the total oxygen demand when thermophilic aerobic digestion was used in series with CAD. One additional benefit of sequential aerobic-anaerobic digestion is pathogens removal. While anaerobic digestion does eliminate pathogens from sewage sludge, it does not compare with the effective removal using aerobic digestion. Typically, aerobic digestate has a Class A solids designation by the US government, where the pathogen levels are below a certain "safe" standard, whereas anaerobic digestate is considered to be a Class B solid (Wang et al., 2018). Including the first step of aerobic digestion leads to an increasing in pathogens elimination, making the final digestate a more suitable alternative for agricultural purposes or to be disposed (Naserian et al., 2021). Moreover, sequential digestion can be expanded beyond just two-stage systems. Multi-stage sequential digestion has the potential to further increase process performance. For example, in a sequential anaerobic-aerobic-anaerobic digestion system, organic matter reduction was increased by 20% over conventional one-stage aerobic digestion. Even though the addition of more process stages would seem to increase the overall cost of a stabilization system, the value of the biogas produced during the process is great enough to mitigate both the additional processing and the cost of aeration used in the aerobic digestion (Tomei et al., 2011).

14.5 CONCLUSIONS AND PERSPECTIVES

Out of all the methods of sludge stabilization, biostabilization has the greatest potential to contribute to a more sustainable residue, because sewage sludge is treated as a resource rather than a residue. Through this chapter, anaerobic digestion is the biostabilization process with the greatest interest. However, this system is complex and economically expensive to operate that can be prone to failure, but one of the main advantages that this system provides is its high potential for energy recovery through biogas capture. Comparatively, aerobic digestion only produces digestate with a good nutrient profile and consumes copious quantities of energy, which explains as to why the research interest is oriented toward anaerobic digestion. Sequential digestion systems should be the future of biostabilization. Regardless of the system configuration, the combination of anaerobic and aerobic digestion almost eliminates the limitations of the two independent systems but maintains the best qualities of each process. Finally, the integration of the processes has the capability to decrease the environmental impact of the sewage sludge, helping to accomplish the main goals of sludge stabilization.

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15 Antibiotic Resistance Genes in Sewage Sludge

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15.1 INTRODUCTION

The antibiotic resistance genes (ARGs) spread into the environment is a major issue around the world, posing extremely serious human health risks (Aljeldah, 2022). Due to the extensive use of antibiotics (ABs) in medicine, a large number of antibiotics have been released into municipal wastewater (Omuferen et al., 2022). Antibiotic resistance is a worldwide concern due to the spread and development of resistance in most common bacteria to the most inexpensive generic antibiotics. Antibiotic resistance is now universally recognized as a public health priority, and a strategy to combat resistance should be developed. Wastewater treatment plants (WWTPs) receive a diversity of ABs and are thought to be possible hotspots for ABs and ARGs to spread. AB resistance may be influenced by biological treatment processes in WWTPs, and sewage sludge (SS) is serving as a reservoir for the spread of human pathogens and ARGs. Sludge with a high concentration of microorganisms promotes horizontal gene transfer (HGT) through mobile genetic elements (MGEs) (Chen et al., 2021). There is evidence that several ARGs and antibiotic-resistant bacteria (ARB) exist in SS, raising the risk to public health (Ondon et al., 2021). As a result, removing pathogens and ARGs from sewage sludge becomes a major concern (Qiu et al., 2021).

WWTPs collect and enrich ABs and ARG from sewage from factories, homes, hospitals, and several industries. Sub-inhibitory antibiotic concentrations, high microbial densities, and nutrient contents in WWTPs promote ARB survival as well as the distribution and transformation of ARGs (Osiska et al., 2019). In addition to ABs, selection pressure from co-exposure to heavy metals and biocides in sewage can cause ARG mutations to increase (Bengtsson-Palme et al., 2016). Furthermore, two major pathways for ARG distribution among bacteria, namely HGT and vertical gene transfer (VGT) (Yue et al., 2022), in WWTPs exacerbate the dissemination of ARG and ARB multiplications. HGT is a method of transferring resistance genes between bacterial species *via* conjugation, transduction, or transformation (Lerminiaux and Cameron, 2019).

SS typically enriches ABs via adsorption and electrostatic gravity, permitting microbes in the sludge focus to be constantly exposed to ABs at the sub-inhibitory concentration (Osiska et al., 2019). As an outcome, ARGs are highly likely to be induced under the selective pressure of ABs.

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Adsorption by activated sludge would decrease extracellular DNA carrying ARGs in wastewater, resulting in far more high concentrations of ARG in excess sludge compared to wastewater (Hou et al., 2019). Total ARG levels in the discharge from the sludge dewatering process were reported to be 16- to –638-fold high than in the effluents and 7- to –308-fold high than in the influents (Qiao et al., 2018). As a result, it is widely assumed that SS contributes more ARG to the environment than WWTP effluents. Sludge disposal and treatment are serious to reduce ARG risk that causes "secondary pollution" in the downstream environment. There are various proofs that the use of sludge as a soil variation is an essential mechanism for ARG introduction as well as spread into the soil (Sun et al., 2021). With the increasing demand for reclaiming sewage sludge for land application, much focus has been given to the elimination potential of predictable contaminants such as pathogenic microbes and heavy metals during the treatment of sludge. However, variations in ARGs diversity and abundance during the process of sludge treatment have yet to be thoroughly investigated. This chapter systematically summarized and compared the fate of ARGs due to various sludge digestion strategies and in combination with relevant pretreatment technologies and additives.

15.2 ARG OCCURRENCE AND DISSEMINATION IN SEWAGE SLUDGE

There is already some knowledge that SS contains considerable quantities of ABs from almost all the major classes (except labile-lactams) (Zhang et al., 2021). Pharmaceuticals have been shown to be adsorbed on the particles of SS. The rate at which this process occurs is determined by the drug's chemical structure, biodegradation, hydrophobicity, and mobility. Furthermore, pharmaceuticals adsorbed to sludge have been shown to be more stable than those present in wastewater (Ihsanullah et al., 2022). Research suggests that even ppb (parts per billion) antimicrobial concentrations can keep ARGs in bacterial populations and may promote plasmid transfer (Zhou et al., 2021). Because of the AB content, it is easy to assume that few microbes in SS are intrinsically resistant to these compounds and/or exhibit AB resistance conferred by clinically relevant mechanisms. In fact, irrespective of the method used, treated SS is high in ARB. However, innovative technologies such as lime stabilization and anaerobic digestion substantially decrease ARB numbers when compared to gravity thickening and simple dewatering (Uluseker et al., 2021). Surprisingly, even if ARBs lose sustainability while treatment, the ARGs frequency may rise at the same time. One hundred fifty-six distinctive ARGs and MGEs encoded resistance to nearly every known AB group in composted SS, implying that this byproduct is a potential reservoir of AB resistance determinants (Su et al., 2015). In WWTPs, commonly detected antibiotics along with their associated ARGs have been provided in Tables 15.1 and 15.2.

15.3 TYPES OF ARGS

The commonly detected antibiotics in WWTPs are classified into four groups based on their chemical structures and characteristics: tetracycline, quinolone, β-lactam, macrolide, as well as sulfonamide ABs, which were all detected at levels of micrograms/liter in raw sewage (Pazda et al., 2019). Their respective resistance genes are, tetracycline (*tetX, tetW, tetT, tetS, tetQ, tetO, tetA*) (Xu et al., 2022), a quinolone (*parC, gyrB, gyrA*) (Lee et al., 2022), β-lactam (*ampC, ampR, blaTEM, blaCIT, blaFOX, blaCTX-M*) (Li et al., 2007), macrolide (*msrA, mefA, mph[B], mph[A]*) (Dayao et al., 2016), and sulfonamide (*sulIII, sulII, sulI*) (Hong et al., 2018), respectively. Aside from these, multidrug resistance genes are frequently found (Algammal et al., 2021). Furthermore, the class 1 integron integrase gene (*intI1*) is regarded as a favorable indicator for assessing ARG horizontal migration potential (Zheng et al., 2020) and is usually found in WWTPs, whereas the ARGs absolute abundance in SS from WWTPs varies by order of magnitude; the concentration values of total ARGs detected range between 108 and 1014 copies/g dry sludge.

TABLE 15.1
In WWTPs Commonly Detected Antibiotics Along with Their Associated ARGs

Sampling Location	Antibiotic Compounds (Type)	Antibiotic Class	Antibiotic-Resistant Genes (Subtype)	References
Activated sludge/tertiary effluent/raw influent	Amoxicillin, penicillin V, cloxacillin, ampicillin	β-lactam	bla_{TEM}, $bla_{CTX\text{-}M}$, bla_{SHV}, $bla_{OXA\text{-}A}$, *mecA*	Ziembińska-Buczyńska et al. (2015); Zhang et al. (2019)
Sewage sludge	Tobramycin, Kanamycin, gentamicin	Aminoglycoside	*aadA, aacA4, aadB, aadE, strB*	Tang et al. (2017)
Digested sludge/secondary effluent/raw influent	Ciprofloxacin, Ofloxacin, norfloxacin	Quinolone	*qnrS, qnrD, qnrC*	Castrignanò et al. (2020a); Castrignanò et al. (2020b)
Digested sludge/secondary effluent/raw influent	Tetracycline	Tetracyclines	*tetE, tetB, tetH, tetG, tetT, tetS, tetA, tetX*	Huang et al. (2016); Wang et al. (2020)
Secondary effluent/raw influent	Clarithromycin, azithromycin, erythromycin/erythromycin-H2O, roxithromycin	Macrolides	*mefC, ermB, ereA, erm43, ermC, and mphG*	Sugimoto et al. (2017); Wang et al. (2020)
Activated sludge	Trimethoprim	Trimethoprim	*dhfr14, dhfrA1*	Ziembińska-Buczyńska et al. (2015)
Raw influent/secondary effluent/activated sludge	Sulfamethoxazole	Sulfonamides	*sul2, sul1*	Lye et al. (2019); Rolbiecki et al. (2020)

TABLE 15.2
Priority of Pathogens by WHO, Which Require New Antibiotics

Priority	Antibiotic Resistance	Bacteria
Critical	Carbapenem	*Acinetobacter baumannii*
	Carbapenem, ESBL[a] – producing	*Enterobacteriaceae*
	Carbapenem	*Pseudomonas aeruginosa*
High	vancomycin-intermediate, and Methicillin	*Staphylococcus aureus*
	Fluoroquinolone	*Campylobacter* spp.
	Faecium Vancomycin	*Enterococcus*
	Clarithromycin	*Helicobacter pylori*
	Fluoroquinolone, Cephalosporin,	*Neisseria gonorrhoeae*
	Fluoroquinolone	*Salmonellae*
Medium	Ampicillin	*Haemophilus influenzae*
	Penicillin	*Streptococcus pneumoniae*
	Fluoroquinolone	*Shigella* spp.

Source: Lawe-Davies and Bennett (2017).

15.4 ARG OCCURRENCE AND DISSEMINATION

ARGs are divided into two types: acquired resistance gene and intrinsic resistance gene (Zhang et al., 2021). Microbes generate intrinsic ARGs to compete against natural ABs long before the environmental selection pressure caused by recent clinical AB use. DNA collection encoding resistance to tetracycline ABs and β-lactam (Jian et al., 2021). Unlike intrinsic resistance genes, acquires resistance genes are incorporated through HGT (McInnes et al., 2020). Free DNA from the extracellular environment is absorbed by microbes and inserted into their genomes during transformation (Rizzo et al., 2013). The transfer of genetic material is transduction between the recipient and donor bacteria through bacteriophage intermediates. MGEs such as integrative and conjugative elements (ICE) and plasmids mediate the conjugation process (Cabezón et al., 2014; Ilangovan et al., 2015; Chiang et al., 2019). Because HGT occurs in both the same and different species (McInnes et al., 2020), HGT is thought to contribute considerably to ARGs worldwide spread when compared to VGT (von Wintersdorff et al., 2016).

15.5 ANTIBIOTIC RESISTANCE MECHANISMS

The common mechanisms by which bacteria resist antibiotics via ARGs can be divided into four groups:

1. Reduced cell permeability—prevents entry of ABs into bacterial cells can be accomplished by developing an alternative metabolic pathway selecting more selective channels or decreasing drug entry channels. It can be also done by modifying the cell surface to limit drug interaction and reduce ABs entry (Schaenzer and Wright, 2020).
2. Direct medicine inactivation-ABs that entered the cell can be rendered inactive through hydrolysis or chemical group transfer; that is, bacterial enzymes add chemical groups to AB molecules' susceptible sites to prevent the ABs from binding to their target protein, resulting in antibiotics (Munita and Arias, 2016).
3. Antibiotic target modification—DNA absorption can confer AB resistance by modifying target proteins via the formation of "mosaic" genes. Another method of changing the target is to acquire genes that are homologous to the original target. Furthermore, target modification by chemical group addition can prevent AB binding (Blair et al., 2015).

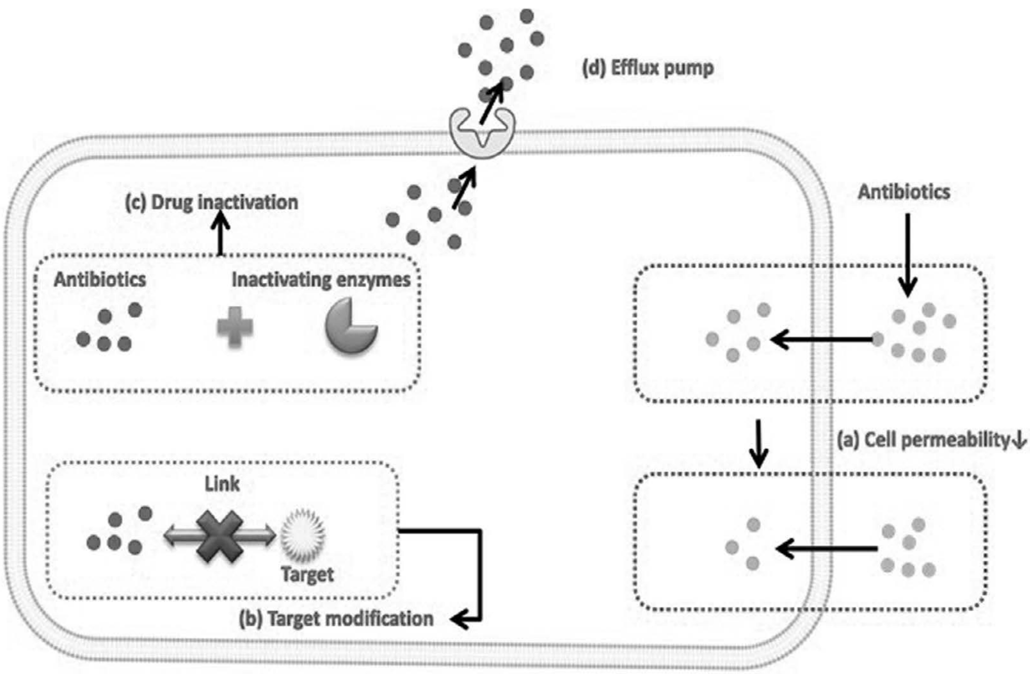

FIGURE 15.1 Antibiotic resistance mechanisms: (a) cell permeability decrease, (b) target modification, (c) drug inactivation, and (d) efflux pump.

4. An efflux pump is used to remove active ABs from bacterial cells. In almost every bacterial species, efflux pumps are found and allow the direct extrusion of different drugs from the periplasmic or cytosol space to the outside of bacterial cells (Poole, 2007).

Antibiotic resistance mechanisms have been provided in Figures 15.1 and 15.2.

15.6 DETECTION TECHNIQUES OF ARGS IN SLUDGE

Until now, the most important clinical method for detecting AB resistance has been the isolation of pure cultures. This method was also useful for detecting AB resistance in sludge or wastewater, and it played a crucial role in AB resistance identifying in WWTPs. For environmental bacteria, drug sensitivity tests and cultures have limitations because only a small quantity of environmental bacteria (*Enterococci* and *Pseudomonas*) can be grown in the lab condition (McLain et al., 2016; Karkman et al., 2018). To detect AB resistance in sludge, culture-independent molecular biology techniques are extensively used. The presence, as well as identification of ARGs in microbes, are frequently based on genetic information taken from samples (Miłobedzka et al., 2022). Thus, in order to detect ARGs using culture-independent molecular biology techniques, RNA and/or DNA were extracted from sludge samples using extraction kits, i.e. RNeasy Mini Kit for RNA and Fast DNA™ Spin Kit for DNA (Wang et al., 2019; Xu et al., 2020). Because RNA is unstable, the RNA extracted is usually synthesized into cDNA and then used for ARG detection via RT-qPCR, metagenomics, or DNA microarray (Galhano et al., 2021).

15.6.1 Real-time Quantitative Reverse Transcription (RT-QPCR)

The extensively used culture-independent method for determining targeted genes for numerous ARGs is RT-qPCR. This method could determine ARGs' absolute as well as relative abundance by

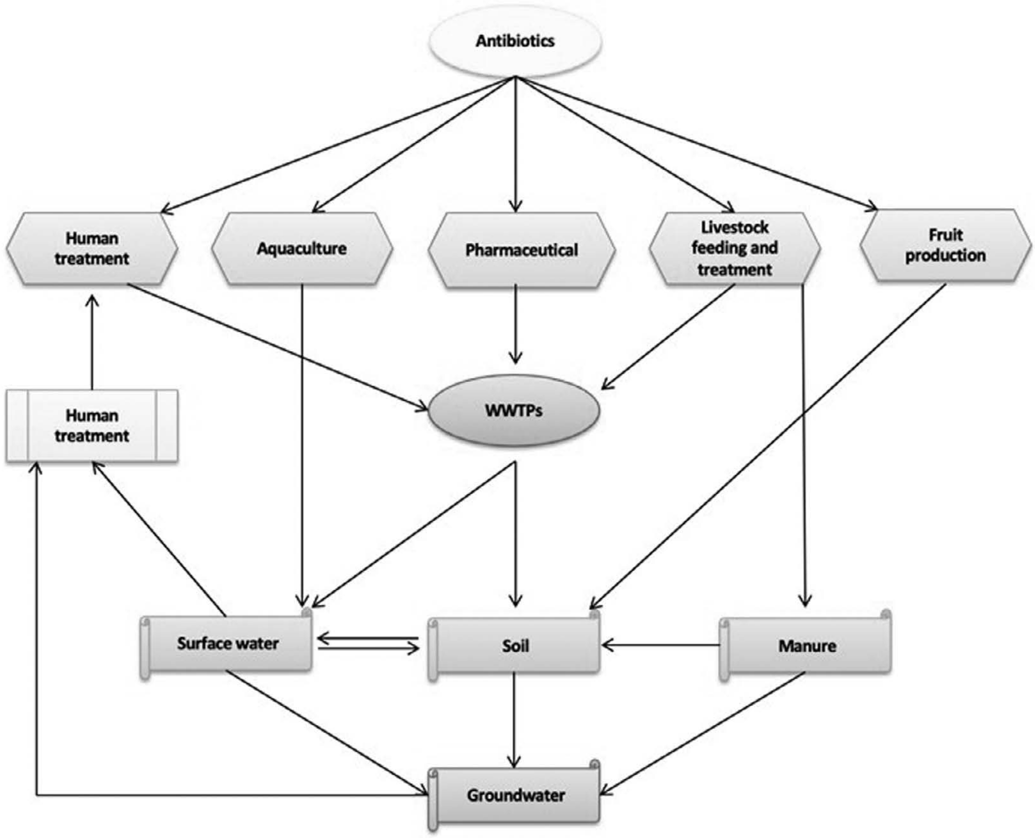

FIGURE 15.2 Transmission routes for antibiotic-resistant genes and resistant bacteria in the environment.

monitoring the amplification reaction with fluorescence. In WWTP sludge, resistance genes to common AB, such as sulfonamide, β-lactam, and tetracycline, have been detected (Paruch, 2022). The main advantages of this technique are (1) rapid examination, (2) high specificity (within 24 hours), (3) a low limit of quantitation (LOQ) and limit of detection (LOD) of about 3 copies/μL, and (4) providing absolute abundance (Karkman et al., 2018).

As a result, it has higher specificity. The RT-qPCR results can be achieved within 24 hours if the target gene primers are available. The standard curve method is used in RT-qPCR to determine absolute abundance. The standard curves are created by quantifying the target genes' known number in the sample. Gene copies/g-TS, gene copies/mL, and gene copies/g-DW (DW: dry weight) are common units in relative research. Another unit frequently used for quantifying ARGs is relative abundance. Relative abundance is the percentage of target gene abundance in each sample's total DNA reads. The ARG's relative abundance is normalized to 16S rRNA abundance as gene copies/16S rRNA for ease of comparison. In related articles, 16S rRNA is typically expressed as biomass (Tong et al., 2017).

Furthermore, some amplification errors may increase the target genes' amplified number. This could be attributable to editing errors that arise during DNA polymerase-catalyzed enzyme replication or errors caused by thermal damage to DNA (Pienaar et al., 2006). Moreover, flux is a significant drawback of this method. At a time using RT-qPCR, only one ARG target can be detected, which severely restricts its throughput (Stedtfeld et al., 2008). The test steps must be repeated for each target gene to quantify multiple ARGs in the same sample.

15.6.2 High-throughput Real-Time Quantitative Reverse Transcription (HT-qPCR)

Presently, HT-qPCR has gained popularity for determining the ARGs abundance in sludge. It is a platform that allows for the miniaturization of conventional RT-qPCR and the processing of larger sample numbers (Lamas et al., 2016). It has the same accuracy as traditional RT-qPCR with quality control (Keenum et al., 2021). HT-qPCR has all the benefits of RT-qPCR. When compared to traditional RT-qPCR, HT-qPCR unquestionably overcomes the detection flux limitation (Waseem et al., 2019). Subject to experimental equipment, HT-qPCR could measure hundreds of ARGs in sludge at the same time, covering more AB classes in a single test (An et al., 2018). However, because this technique is also based on RT-qPCR, it suffers from the same drawbacks as RT-qPCR, such as the requirement to create primers before detection, as well as the incapability of providing host information.

15.6.3 Next-generation Sequencing

Metagenomics, as opposed to HT-qPCR and RT-qPCR, which only collects the genetic data of microbial communities, measures the abundance of targeted ARGs. The shotgun high-throughput sequencing is currently broadly applied in sequencing platforms and has been extensively used in the detection of ARGs in a variety of environments, which include sludge (Wani et al., 2021). The major advantages of this technique include (1) providing ARG host information via network analysis and (2) obtaining information for every ARG in the sample (Cheng et al., 2021). Because metagenomics collects every genetic information in a sample, it could theoretically provide all species relative abundance and ARGs in a sample (Majeed et al., 2021). ARG potential hosts in the sample can be identified further using network analysis.

Furthermore, the target gene detection limit is narrowly associated with sequencing depth that significantly raises data size. However, this increases accuracy and enhances handling time, expenses, as well as computational resource consumption. More notably, metagenomics sequencing typically provides information on the target ARGs' relative abundance. Recent studies have transformed the relative abundance obtained from metagenomic analysis by adding the Internal Standard into absolute abundance. The Internal Standard, on the other hand, must be calibrated and measured using RT-qPCR (Crossette et al., 2021), which adds to the methodological optimization effort and may introduce biases.

15.6.4 Emerging Techniques

Traditional molecular biology techniques offer an exact method for identifying as well as quantifying ARGs in sludge. Though, because of the read length limitation, these techniques cannot directly show the ARGs' true host in sludge. There is a crucial requirement to understand the ARGs' true host, which may aid in uncovering the diversity of ARG hosts and implementing interventions to decrease ARG spread in sludge. As a result, some new methods have recently emerged, such as single-cell genome sequencing, third-generation sequencing platforms, and Emulsion paired isolation and concatenation (Epic) PCR.

15.7 ARGS REMOVAL IN WWTPs

15.7.1 Wastewater Treatment Process

ARGs are reduced by up to four orders of magnitude in numerous wastewater treatment processes that are considered partial reductions. However, studies have discovered a potential positive correlation between the concentration of ARGs and 16S rDNA content that describes bacteria's total number. This suggests that WWTPs decrease ARG abundance by reducing total bacterial biomass in effluent rather than eliminating ARGs selectively from bacterial cells. Higher levels of ARGs as well as ARBs have been detected in the effluent of WWTP. The removal of ARGs varies significantly across WWTPs (Jäger et al., 2018).

15.7.2 Biological Treatment

ARGs are primarily removed by biological treatment units via activated sludge adsorption and microbe degradation in wastewater (Shao and Wu, 2020). Following that, the ARGs are transferred to sludge *via* sedimentation. Because removal of ABs and ARGs is not targeted in the biological treatment process, in this treatment, ARGs are not efficiently reduced. However, ARG abundances in biological treatment effluents increase in some cases. The relative abundance of tet genes (*tetA, tetB, and tetM*) and *sul* genes (*sul2 and sul1*) in the effluent was greater than in the influent after treatment with activated sludge. A higher relative abundance of ARGs, such as *tetC, tetG, tetM, tetX, sul1*, and *intI1*, was detected in effluent from an A/O WWTP (Li et al., 2015). The upgraded A/O process also failed to reduce ARGs effectively, and the anaerobic tank improved ARG abundance. This is due to the higher nutrients and microorganisms content in activated sludge during the biological treatment stage that promotes HGT between ARB and ARGs (Li et al., 2016). During processing, ARB horizontal transfer allows for their horizontal transfer to new species, enhancing their relative abundance. Furthermore, the hydraulic retention time (HRT) was discovered to be considerably correlated with ARGs along with ARB, explaining their low elimination efficiency (Korzeniewska and Harnisz, 2018). ARGs and ARB have more time to multiply and reproduce when the HRT is longer.

15.7.3 Mechanical Treatment

Mechanical treatment units, such as sand filtration, grids, and primary sedimentation, have an insignificant role in ARG elimination, ranging from 0.15 to 1.75 orders of magnitude (Wen et al., 2016). However, after biological treatment, the secondary sedimentation unit plays an important role in ARG reduction. It was discovered that 37.70% of the ARGs in the biological unit's effluent could be eliminated in the secondary sedimentation tank. The tet gene was largely removed from Bacteroides, the common host of tet. Bacteroides are easily absorbed by sludge precipitation because they can form flocs by excreting extracellular polymers, resulting in the removal of tet genes (Lee et al., 2017).

15.7.4 Advanced Treatment

UV disinfection, chlorine disinfection, biological activated carbon (BAC), constructed wetlands, and chemical oxidation are advanced treatment methods for WWTPs. ARGs are reduced slightly after UV disinfection. The UV method of removing ARGs works by forming pyrimidine dimers, which can straightly destroy ARG-containing DNA fragments in cells without affecting other cell components (Guo and Kong, 2019). WWTPs' UV disinfection doses, on the other hand, have no discernible elimination effect on *erm, sul, tet*, or other ARGs. Due to the high levels of background contaminants and chromaticity in wastewater, a high UV dose is needed to inactivate ARGs, far above the conventional UV disinfection dose (Zhang et al., 2015). The ARGs' removal was found to be 0.56 log copies/mL when the UV disinfection dose reached 250 mJ/cm2, and several types of ARGs concentration were reduced to some level (McConnell et al., 2018). UVA-TiO_2 heterogeneous photocatalysis demonstrated greater benefits in inactivating ARB and preventing ARG propagation in water treatment. Photocatalysis can cause DNA damage in cells, resulting in the elimination of *ampC* and *ecfx* gene reduction in *Pseudomonas aeruginosa* (Karaolia et al., 2018).

BAC eliminates ARGs by adsorbing ARB-containing ARGs on the surface of the ARB. ARGs are intercepted by the biofilm formed on the surface of activated carbon (Zhu et al., 2018).

Chemical oxidation and chlorine disinfection promote the propagation of ARG after ARGs are removed from reclaimed water. Chlorine disinfection is frequently used as a pretreatment step in the reused water reverse osmosis process (Luo et al., 2021). The cumulative relative abundance of 14 common ARGs in membrane foulants increased by 49.60%, including *tolC, acrA*, and *acrB*, when the chlorine dose was increased from 0 to 5 mg/L. (Wu et al., 2022).

15.8 DIFFERENT ARG REMOVAL EFFECTS

The impacts of eliminating various types of ARGs differ; *tet* and *ermB* are relatively well-reduced ARGs (Wen et al., 2016). The *tet* genes could be lowered by two to three orders of magnitude which has been attributed to the ease, with which tetracycline-resistant bacteria could be removed (Chen and Zhang, 2013). The *ermB* gene is found primarily in gram-positive bacteria Firmicutes (*Streptococcus, Staphylococcus*, and *Enterococcus*), but it can also be transferred to gram-negative bacteria, particularly Bacteroidetes. The *ermB* gene was found to decrease by 0.40–1.50 orders of magnitude, probably because of a decrease in Firmicutes abundance (Rafraf et al., 2016). The *sul* gene is a difficult ARG to remove, leading to a high concentration in the effluent (Du et al., 2020). This is most likely due to the presence of *sul* genes in all MGEs, which can be transferred between heterogeneous and homogeneous bacteria (Michael-Kordatou et al., 2018). Furthermore, the relative abundance of korB that also targets plasmids from Incompatibility Group P-1 (IncP-1) was discovered to be expressively higher in wastewater after treatment (Pallares-Vega et al., 2019). The plasmid IncP-1 spreads into multiple bacteria through HGT (Klümper et al., 2015). IncP-1 plasmids, on the other hand, frequently contain genes encoding metal resistance and xenobiotic compound degradation, possibly enhancing their transmission, and enhancing bacterial metabolic dominance in activated sludge (Klümper et al., 2015).

15.9 CONCLUSIONS AND PERSPECTIVES

As ARGs have been found in sewage sludge, wastewater treatment processes need to address this issue. With the innovation of detection techniques, ARGs occurrence and elimination in WWTP sludge have upgraded significantly over the past few years. Though recent advances in metagenomic and other emerging molecular-based techniques have permitted ARGs' comprehensive assessment in sludge, RT-qPCR remains the most frequently used technique for ARG detection because of its accuracy, cost, and handling time. Many studies used RT-qPCR to determine the abundance of 5–25 ARGs. These ARGs were chosen based on primers and probes accessibility, the diversity of AB classes, and the mechanism of resistance. These ARGs' targets cannot represent every ARG in the sample, so the results may not fully reveal the total ARG distribution and abundance in the sludge sample. As a result, further advancements in the detection approaches are still required to quantify a broad spectrum of ARGs efficiently and accurately in sludge.

Furthermore, the efficacy of sludge digestion technologies in eliminating ARG is presently being assessed based on the change in ARG abundance before and after sludge digestion. Sludge reuse risk for ARG transfer in the environment is thus determined by ARG abundance in digested sludge. It is obvious that ARGs can be transferred from sludge to the natural environment because of sludge reuse, resulting in ARG proliferation. However, in addition to ARG changes, various sludge digestion techniques altered the microbial community in the sludge. The microbial community effect in sludge on ARG transfer into the natural environment is largely unknown. To protect public health, ecosystems, and beneficial microorganisms, ABGs in sewage sludge must be addressed in wastewater treatment. Forthcoming research should focus on the changes in soil ARGs caused by the treated sludge application.

15.10 DECLARATION OF COMPETING INTEREST

The authors declare that they have no known competing financial interests or personal that could have appeared to influence the work reported in this chapter.

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16 Bioleaching of Sewage Sludge for Enhanced Dewaterability

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16.1 INTRODUCTION

Reducing the volume of sewage sludge is an important factor affecting the scale and cost of subsequent sludge treatment and disposal (Cao et al., 2021). Sewage sludge is mainly produced during the biological treatment of wastewater and usually contains significant amounts of microbial cell residues and organic materials. The moisture content of sludge is more than 99% owing to the internal water in residual cells and the large amount of moisture absorbed on the surface of sludge flocs (Zhang et al., 2022). However, reducing the sludge moisture content to 97% through feasible technology can reduce the sludge volume by two thirds (Li et al., 2022). Therefore, sludge dewatering directly determines the choice of treatment technology and economic outcomes. Mechanical dewatering technology is currently the most widely used process. However, mechanical dewatering does not effectively reduce the moisture content of sludge if undertaken as the only process due to the different states of water in sludge flocs (Vaxelaire and Cézac, 2004). In addition, sludge flocs contain a considerable amount of organic matter, that is, extracellular polymeric substances (EPS), comprising many hydrophilic components that can absorb a significant amount of water (Wu et al., 2017). Therefore, it is necessary to adopt an appropriate pre-treatment technology to improve sludge dewatering performance.

Bioleaching technology, as a biological pre-treatment technology, has received increasing attention from researchers in recent years (Liu et al., 2012). It can not only separate heavy metals in sludge but also improve the dewatering process and conserve most fertilizer nutrients. However, bioleaching technology has the typical disadvantages associated with bio-conditioning, including a long reaction cycle and low efficiency (Murugesan et al., 2017). Therefore, improving the efficiency of bioleaching technology and exploring the dewatering mechanism are significant challenges for future applications. In this chapter, factors of sludge that intrinsically result in dewatering difficulties are described. Second, different pre-treatment technologies, including physical, chemical, and microbial processes, are introduced. Third, the general principles and applications of bioleaching technology for heavy metal removal are presented. In addition, the applications of bioleaching to improve sludge dewaterability and recent developments are described. Finally, technical limitations and future research prospects are discussed.

16.2 INFLUENCING FACTORS OF SLUDGE DEWATERING

16.2.1 Extracellular Polymeric Substances

EPS are a major component of sludge flocs and are regarded as one of the most important factors influencing sludge dewatering performance because of their high hydrophilicity (Wu et al., 2022). As shown in Figure 16.1a, EPS can be divided into three types: tightly bound EPS (TB-EPS),

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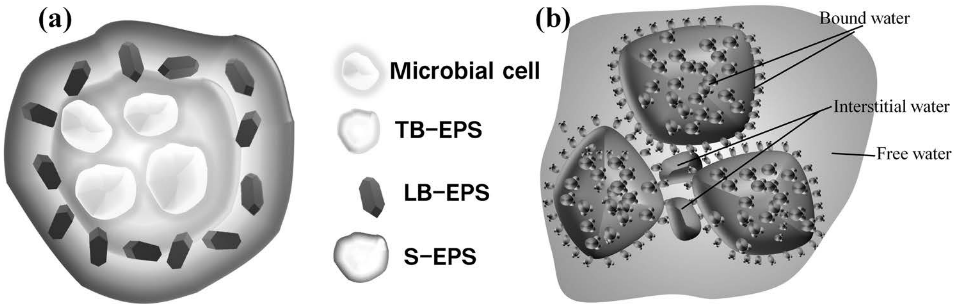

FIGURE 16.1 Distribution of extracellular polymeric substances (TB-EPS: tightly bound extracellular polymeric substances, loosely bound extracellular polymeric substances, and soluble extracellular polymeric substances) (a) and distribution of water in sludge floc (b).

loosely bound EPS, and soluble EPS. It contains a large amount of hydrophilic organic matter, accounting for approximately 80% of the total biomass of the activated sludge (Lin et al., 2022). These hydrophilic radicals trap large amounts of water and form stable flocculating structures.

16.2.2 Water Distribution in Sludge

Sludge water can be divided into three types: free water, interstitial water, and bound water, as shown in Figure 16.1b. Free water flows easily through the sludge system and does not bind to sludge solids; therefore, it can be easily separated by simple mechanical methods (Wu et al., 2021). Bound water accounts for only ~10% of the total water content of sludge; however, it cannot be easily removed using mechanical dewatering because of its strong binding force with particles (Wu et al., 2022). Interstitial water does not flow freely in the sludge suspension and accounts for approximately 10%–25% of the total water. Therefore, the key method to improve the capacity for dewatering sludge is to remove the existing forms of bound water and interstitial water through pre-treatment to convert it into free water, which can then be separated using current mechanical methods (Zhang et al., 2022).

16.2.3 Surface Charge

The hydrolysis and ionization of carboxyl and phosphate groups on the surface of negatively charged flocs can produce electrostatic repulsion, which can stop flocculation and settlement (Wu et al., 2022). The electrical charge of flocs can be characterized by their zeta potential, which is generally between −30 and −10 mV. Positively charged H^+ ions in water molecules are absorbed onto the surface of sludge particles. However, due to the electrical double layer, electrostatic repulsion also hinders the flocculation of sludge particles (Zhang et al., 2022). Therefore, during pre-treatment, electrical neutralization and double-layer compression are the major mechanisms for reducing electrostatic repulsion.

16.3 SLUDGE PRE-TREATMENT TECHNOLOGY

Sludge pre-treatment is a process that occurs prior to mechanical dewatering to improve the characteristics of the sludge for final disposal. Efficient pre-treatment can alter the microstructure and occurrence state of water in sludge, change the bound water in sludge into free water, damage the EPS network, and eventually improve the dewatering performance (Wei et al., 2018).

16.3.1 Chemical Process

Chemical conditioning improves the dewatering process by changing the surface charge, bridging, and oxidation and includes inorganic or organic flocculant addition, pH adjustment, and advanced oxidation (Kamizela and Kowalczyk, 2019). Chemical flocculants are widely used for sludge pre-conditioning in urban wastewater treatment plants due to their ease of use and low cost. However, excessive dosage of flocculant and the subsequently generated chemical sludge may cause the final amount of dry sludge to increase (Cao et al., 2021). Advanced oxidation processes can produce free radicals that destroy cells and release bound water while simultaneously reducing sludge volume, eliminating pathogenic microorganisms, and decomposing micro-pollutants. However, advanced oxidation processes have rarely been reported in engineering applications. There are strict regulations regarding the storing and use of chemical reagents, and the operational cost of advanced oxidation processes is much higher than that of other methods used under similar conditions (Cao et al., 2021).

16.3.2 Physical Process

Physical methods, including freeze-thaw, microwave, ultrasonic, and electrical conditioning, improve sludge dewatering performance by destroying microbial cells, altering the structure of sludge flocs, and releasing intracellular materials (Huang et al., 2020). Freeze-thaw treatment uses ice crystals formed during freezing to destroy sludge cells, change floc structure, and release bound water. However, freeze-thaw treatment is only cost-effective and efficient in cold regions (Rao et al., 2019). Ultrasonic treatment applies a certain frequency of ultrasonic waves to the sludge to produce a cavitation effect and a large number of cavitation bubbles, which can be broken within several microseconds to produce high temperatures and strong shear forces which are the main factors causing the destruction of sludge flocs. High temperatures changing the state of water in sludge can be beneficial for subsequent dewatering treatments (Feng et al., 2009). However, ultrasonic processing is still at the laboratory stage, with limited applications, and ultrasonic equipment is still being developed. Electro-dewatering applies an electric field during mechanical dewatering to remove water. Heavy metal ions and organic compounds also migrate to the electrode, which is conducive to wider sludge use. However, because the electrode was immersed in sludge for a longer duration, it was easily corroded. Therefore, the anti-corrosion characteristics of an electrode are critical for its suitability.

16.3.3 Biological Process

Bio-conditioning involves the use of microorganisms or microbial products to improve sludge dewatering performance and include enzyme conditioning and the addition of specific functional microorganisms (Kurade et al., 2016). The direct addition of biological enzymes or enzyme-producing microorganisms can improve dewatering performance by degrading macromolecules, such as polysaccharides (PS) and proteins (PN) in EPS and so release bound water. The addition of specific functional microorganisms, such as those that can synthesize bioflocculants, can also improve the dewatering process (Murugesan et al., 2017). Microbial flocculants are organic materials with flocculating activity obtained through microbial fermentation, extraction, and refinement. Their main components are proteins, polysaccharides, cellulose, and DNA, which have three main types: microbial cells, microbial cell extract material, and microbial cell metabolites (Murugesan et al., 2017). Microbial flocculants are non-toxic, biodegradable and can compact flocs. They may act as flocculants, promote the aggregation of sludge particles, and improve dewatering performance.

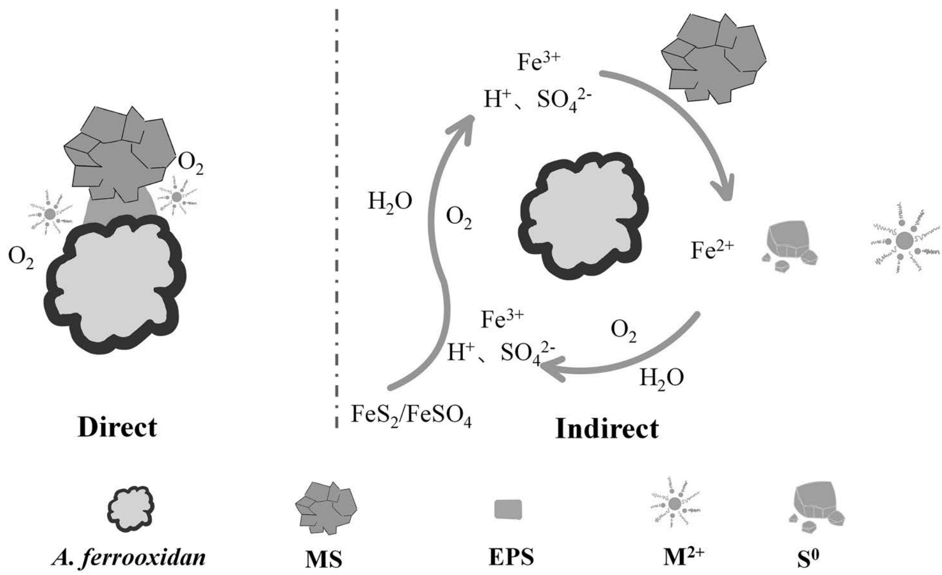

FIGURE 16.2 Bioleaching mechanism for heavy metal removal.

16.4 BIOLEACHING TECHNOLOGY FOR REMOVAL OF HEAVY METALS

Bioleaching was first used in the 1950s to extract heavy metals from mines (Bosecker, 1997). Bioleaching microorganisms can reduce pH by providing S^0 and Fe^{2+} as substrates. Mechanisms for the removal of heavy metals via bioleaching can be classified as either direct or indirect (Gu et al., 2017), as shown in Figure 16.2.

Direct mechanism: under aerobic conditions, the bacterium *Acidithiobacillus ferrooxidans* oxidizes insoluble metal sulfides to soluble metal sulfates via direct contact (Natarajan, 2018). The reaction is shown in Eq. 16.1.

$$MS + 2O_2 \xrightarrow{A.\,ferrooxidans} M^{2+} + SO_4^{2-} \tag{16.1}$$

Indirect oxidation refers to the process of dissolving heavy metals in metal sulfides via the oxidation of the ferric iron (Fe^{3+}) (Eq. 16.2). This process does not require the participation of *A. ferrooxidans*. Fe^{3+} is generated from the bacterial oxidation of FeS_2 (Eq. 16.3) and $FeSO_4{\cdot}7H_2O$ (Eq. 16.4). The elemental sulfur produced, as shown in Eq. (16.2), is oxidized to sulfuric acid by bioleaching bacteria (Eq. 16.5). The production of sulfuric acid decreases sludge pH, which further improves metal solubilization (Natarajan, 2018).

$$2Fe^{3+} + MS \xrightarrow{A.\,ferrooxidan} M^{2+} + 2Fe^{2+} + S^0 \tag{16.2}$$

$$FeS_2 + 3.5O_2 + H_2O \xrightarrow{A.\,ferrooxidan} FeSO_4 + 2H_2SO_4 \tag{16.3}$$

$$2FeSO_4 + 0.5O_2 + H_2SO_4 \xrightarrow{A.\,ferrooxidan} Fe_2\left(SO_4\right)_3 + H_2O \tag{16.4}$$

$$S^0 + 3O_2 + 2H_2O \xrightarrow{A.\,ferrooxidan} 2H_2SO_4 \tag{16.5}$$

Generally, microorganisms used for sludge bioleaching can be classified into mesophiles and thermophiles, according to the temperature range for their growth (Kumar and Yaashikaa, 2020). The most dominant mesophilic microorganisms used in sludge bioleaching are *Acidithiobacillus* sp., including the sulfur-oxidizing bacteria *A. thiooxidans*, and iron-oxidizing bacteria *A. ferrooxidans* (Gu et al., 2017). Both are chemolithotrophic bacteria that derive energy from Fe^{2+} or S^0 oxidation. *A. ferrooxidans* can grow in the temperature range of 20–40°C; however, the optimum temperature is approximately 33°C. *A. ferrooxidans* can grow in the pH range of 1.0–4.5, with an optimum range of 2.0–2.3 (Pathak et al., 2009). In addition to autotrophic microorganisms, heterotrophic microbes can be used for bioleaching by their production of organic acids (oxalic acid, citric acid, and malic acid) that can provide protons and metal-complexing anions. Bacteria such as *Acetobacter*, *Acidophilum*, *Arthrobacter, Pseudomonas*, and *Trichoderma* can be used to bioleach ores and minerals (Valix, 2017). However, little information is available regarding the use of these microorganisms for sludge bioleaching.

16.5 BIOLEACHING TECHNOLOGY FOR ENHANCING SLUDGE DEWATERABILITY

In recent years, bioleaching has been used to improve sludge dewatering performance and has attracted considerable attention. Wong et al. (2004) were the first to introduce bioleaching in sludge dewatering and found that bioacidification promoted the dissolution of heavy metals and improved dewatering efficiency. Similar to the separation of heavy metals, two types of bacterial inoculations were used for dewatering conditioning: *A. ferrooxidans* and *A. thiooxidans*. Most notably, as reported by Song and Zhou (2008), bioleaching has been applied in engineering to improve the sludge dewatering performance in Nanjing, China.

16.5.1 Factors Influencing Dewatering Effects of Bioleaching

The effectiveness of bioleaching depends largely on the type and dose of energy source, the bioleaching microorganism species used, initial pH, temperature, and O_2 and CO_2 concentrations. Microorganisms are the key players in bioleaching, inferring that if they maintain a high activity, more dewatering occurs.

16.5.1.1 Energy Sources

Bioleaching microorganisms obtain the energy needed for their growth and reproduction through the oxidation of energy-reducing materials (Natarajan, 2018). The addition of different types of energy sources or different mixing ratios of energy sources will affect the activity of bioleaching microorganisms, thereby affecting the reduction in sludge pH and the removal efficiency of EPS (Li et al., 2021). Fe^{2+} is used as an energy source in the bioleaching process to improve performance. Both Fe^{2+} and its oxidation product Fe^{3+} can increase dewatering, but S^0 does not exhibit this characteristic (Liu et al., 2019).

16.5.1.2 Bioleaching of Microbial Species

The effect of bioleaching also depends on the microorganism used. The microorganisms used in bioleaching can be divided into iron-oxidizing and sulfide-oxidizing bacteria, with iron-oxidizing bacteria being more superior. With iron-oxidizing bacteria, the pH can be reduced to approximately 2.0 within two days, compared to sulfide-oxidizing bacteria which reach the same acidification effect after approximately six days (Liu et al., 2019). In addition, sulfide-oxidizing bacteria can acidize sludge to a pH lower than 2.0, which may affect subsequent processing (Kumar and Yaashikaa, 2020). However, iron-oxidizing bacteria do not have this effect because they use two processes: acid production and acid consumption. In addition, Fe^{2+} and Fe^{3+} were used as typical chemical flocculants in sludge pre-treatment.

16.5.1.3 pH

Different initial pH values affect the starting rate of bioleaching, use of energy sources, and overall bioleaching effect. Jain and Tyagi (1992) found that there are at least two types of *Acidithiobacillus* sp. with different degrees of acidophilicity in sludge, with optimal pH of 7.0 and 4.0. In the process of bioleaching, the weak *Acidthiobacillus* sp. proliferates first, reducing the pH to a certain extent, and the strong *Acidthiobacillus* sp. proliferates gradually, leading to a further decrease in the pH (Wang et al., 2018). Some researchers have attempted to reduce the initial pH of the sludge to an appropriate range before bioleaching. Although this improved the bioleaching effect, the overall cost of treatment increased due to the higher cost of acid consumed to reduce the sludge pH.

16.5.1.4 Temperature

Optimal temperatures differ between types of bioleaching microorganisms; therefore, suitable temperatures need to be considered to promote rapid propagation of leaching microorganisms in order to accelerate sludge acidification rates (Natarajan, 2018). Optimum temperatures for iron and sulfide oxidation by *A. ferrooxidans* ranged from 28 to 30°C. Tyagi et al. (1994) studied the acidification capacities of bioleaching microorganisms at various temperatures. Their results showed that when the reaction temperature was from 7 to 35°C, the time required for the sludge pH to decrease to 2.0 was 120 to 336 h. When the reaction temperature was above 28°C, the time required was the shortest (120 h), and when the temperature continued to increase to 42°C, pH decreased to below 2.0.

16.5.1.5 O_2 and CO_2

A. ferrooxidans is gram-negative rod-shaped bacteria which use carbon by fixing atmospheric CO_2. Sufficient oxygen supply can promote the biological oxidation of bioleaching microorganisms. Li et al. (2021) reported that an increase in oxygen concentration during bioleaching improved the dewatering performance and shortened the bioleaching time. In laboratory experiments, this was achieved through shaking and stirring; however, for large-scale engineering applications, maintaining a sufficient oxygen supply may be difficult.

16.5.2 Conventional Dewatering Mechanism for Bioleaching

At present, existing research on bioleaching for sludge dewatering mainly focuses on the dewatering effect and influencing factors. The possible dewatering mechanism of sludge flocs is discussed based on the traditional flocculation theory, namely, the electrical double layer theory, with respect to the characteristics of sludge flocs.

16.5.2.1 Charge Neutralization

The essence of the chemical conditioning method is to add positively charged substances to the sludge to neutralize the negative charges on the surface of particles, weaken the electrostatic repulsion between them, allowing them to easily aggregate and settle (Zhang et al., 2022). Bioleaching microorganisms can oxidize Fe^{2+} to Fe^{3+}, and, at the same time, Fe^{3+} hydrolyzes to form H^+ and produce acid (Natarajan, 2018). As shown in Figure 16.3, both Fe^{3+} and H^+ have positive charges, which is similar to the effect of adding inorganic acid and inorganic ferric salts, neutralizes the negative charges on the surface of particles, and finally improves the settling and dewaterability of the sludge.

16.5.2.2 Biooxidation and EPS Damage

The acidic environment provided by H^+ produced during bioleaching can destroy microbial cells and release inner water (Li et al., 2022). However, it can disintegrate EPS when in contact with microbial cells, as shown in Figure 16.3. Because EPS contain a large number of hydrophilic groups, most of the free water is restricted and difficult to remove. Under intense acidic conditions, the EPS was destabilized, the structure of the hydrophilic group was destroyed, and the bound water was released.

16.5.2.3 Changes in the Microbial Community Structure

Bioleaching microorganisms are autotrophic microorganisms that can grow well in acidic environments, whereas the microbial community in wastewater sludge consists mainly of heterotrophic

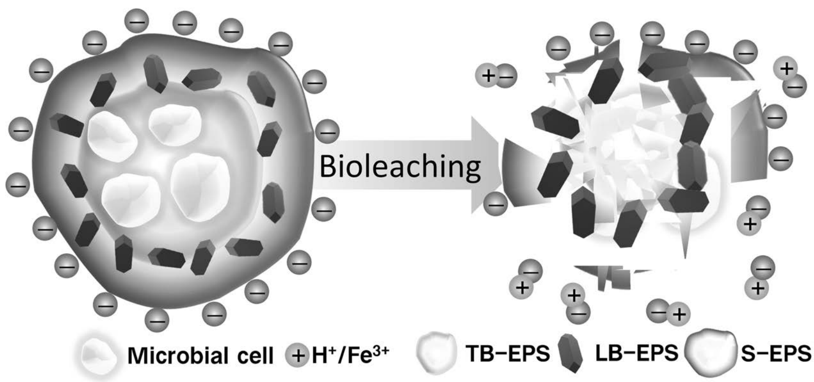

FIGURE 16.3 Conventional dewatering mechanism of bioleaching (TB-EPS: tightly bound extracellular polymeric substances, loosely bound extracellular polymeric substances, and soluble extracellular polymeric substances).

microorganisms, most of which are suited to a neutral environment. When the sludge treatment system is extremely acidic, heterotrophic microorganisms cannot tolerate the conditions and are replaced by *Acidthiobacillus* sp., altering the structure of the microbial community (Huang et al., 2020; Li et al., 2021). Compared with heterotrophic microorganisms with high EPS secretion, autotrophic microorganisms secrete low EPS content (Natarajan, 2018). Therefore, the microstructure of the sludge flocs change and the content of hydrophilic substances decreases considerably.

16.6 RECENT DEVELOPMENTS IN BIOLEACHING

Although bioleaching technology has many advantages in improving sludge dewatering performance, it has some limitations, such as the poor adaptability of inoculum exogenous microorganisms and a low growth rate compared to indigenous microorganisms found in sludge (Huang et al., 2020). Therefore, in practical applications, a higher amount of inoculum may be required to improve treatment efficiency and ensure sludge dewatering. Recently, novel technologies have been developed to address the problems of low efficiency and long bioleaching cycles, for example, a combination of other physicochemical pre-treatment technologies and inoculation of mixed cultures during bioleaching.

16.6.1 Bioleaching Combined with Physicochemical Method

Several processes, including the Fenton, ultrasonic, and surfactant methods, can be used to enhance the dewatering effect of bioleaching. The advantages of bioleaching combined with Fenton's method are that it can provide the acidic conditions required for the Fenton reaction, and the iron-containing substances generated by bioleaching can be used, in turn, as catalysts for the Fenton reaction. Furthermore, the bioleaching cycle was shortened, and the cost was reduced. Zeng et al. (2015) found that a combination of Fenton and bioleaching shortened the bioleaching time to five days and reduced the EPS content and sludge-specific resistance (SRF) by 68.7% and 83.8%, respectively. Fontmorin and Sillanpää (2015) reduced the leaching time to six days and improved the dewatering performance with a 99.5% reduction in sludge SRF and 98.4% reduction in sludge capillary suction time (CST) using a Fenton-like process coupled with bioleaching. Ultrasonic treatment can break the structure of sludge flocs and can be used for bioleaching pre-treatment to improve reaction efficiency. Huang et al. (2020) studied an ultrasonic-coupled bioleach to enhance sludge dewatering performance and found that ultrasonic treatment played a significant role in destroying the sludge structure. The organic matter in the TB-EPS layer was then transferred to the external EPS layer.

Compared to bioleaching alone, the sludge structure and cell walls were more easily destroyed. Liu et al. (2019) found that when a surfactant was used as a conditioner for bioleaching, the improvement in sludge dewatering was mainly due to the fragmentation of sludge flocs, protonation of surface charges, and release of EPS.

16.6.2 Bioleaching of Sludge Based on Mixed Culture

The sludge dewatering effect can be improved by inoculation with pure *Acidithiobacillus* sp. However, when indigenous bacteria are used, other native bacteria in sludge cannot be ruled out. Acidophilic bacteria are widely found in terrestrial and aquatic habitats. The inoculation of pure bacteria increases the operational cost; however, each treatment needs to be inoculated with new microorganisms to ensure the effect, which has certain limitations in engineering applications (Li et al., 2021, 2022). Thus, bioleaching microbes can be enriched and used for bioleaching through specific acclimation. However, few studies have been conducted on the dewatering performance of mixed bacteria for bioleaching. Fontmorin and Sillanpää (2015) used $FeSO_4{\cdot}7H_2O$ to enrich indigenous iron-oxidizing bacteria in fresh sludge, and the sludge obtained at the end of enrichment was used as the inoculum. They reported that compared with the inoculation of pure strains and indigenous microorganisms, the dewatering performance of bioleaching based on mixed culture was stable and efficient. Li et al. (2021) attempted to acclimate sewage sludge by repeated use of Fe^{2+} and obtained a mixed culture named acidified sludge (AS). When AS was used in sludge conditioning, the CST and SRF of the treated sludge decreased by 76.5% and 95.7%, respectively, and the total protein (PN) content in the TB-EPS decreased by 82.5%.

16.6.3 Novel Insights into Dewatering Mechanism of Sludge Bioleaching

In addition to the distribution of EPS, chargeability, and microbial community substitution, several researchers have recently proposed that the molecular structure of EPS, such as secondary structure of extracellular proteins, is closely related to sludge dewatering (Ding et al., 2022). Furthermore, with the development of microecology, metagenomic sequencing technology has been used to study microbial mechanisms of bioleaching (Huang et al., 2020).

16.6.3.1 Effects of Protein Secondary Structure

PN comprises a macromolecular hydrophilic substance that plays a more important role than PS in determining dewaterability (Liu et al., 2020). Due to hydrogen bonding induced by nitrogenous and oxygen-containing functional groups, water molecules are trapped in the gaps between chain macromolecules, resulting in a high water-containing capacity (Li et al., 2021; Zhang et al., 2019). During bioleaching, with a decline in pH and generation of iron-containing bio-flocculates, the PN structure is disrupted, leading to a decrease in protein hydrophilicity. The concentration of PN is related to sludge dewatering performance, and the transformation of the PN structure also affects dewatering. Generally, the secondary structures of PN include β-sheets, aggregated strands, α-helices, random coils, 3-turn helices, and antiparallel β-sheet/aggregated strands. Hydrophobicity of a protein is related to its secondary structure. A low α-helix content, high β-sheet content, and random coil will lead to a loose structure. The aggregation, adsorption, and flocculation of microbial aggregates can be enhanced by increasing their α-helix content. Although there must be a necessary relationship between loose structure and sludge dewatering performance, a description of this relationship has not been determined. The loose structure may damage sludge dewatering by increasing the water retention capacity of sludge flocs; however, it may also lead to the leakage of more hydrophobic groups, thus improving the dewatering ability. Wu et al. (2017) reported that the disintegration of the secondary structure of PNs by the removal of disulfide bonds may be a key factor in eliminating the inhibitory effect of extracellular PN on sludge interstitial water removal. As reported by Li et al. (2020), a low α-helix content, high β-sheet content, and random coils can lead to a loose EPS structure. In addition, the α-helix/(β-sheet + random coil) ratio is related to the aggregation, adsorption, and flocculation of sludge. As shown by Li et al. (2022), bioleaching based on a mixed culture can facilitate the release of intracellular water by degrading the secondary structure of the proteins in TB-EPS and enhancing the hydrophobicity of sludge flocs, thus improving dewaterability. Therefore, focused investigations on the secondary structure of extracellular proteins under different conditioning techniques are needed to help understand the molecular mechanism of sludge dewatering.

16.6.3.2 Effect of Microbial Structural Composition and Functional Diversity

Microorganisms play a key role in improving sludge dewatering via bioleaching. Recently, high-throughput sequencing technology has been used to analyze the compositional characteristics of microorganisms during bioleaching, particularly for bioleaching based on mixed cultures.

In bioleaching treatment for sludge dewatering, *Acidthiobacillus* sp. is a non-indigenous organism, and its ability to compete with indigenous organisms is unclear. Microbial community structure analysis can provide a comprehensive understanding of vital microbes. Huang et al. (2020) used *A. ferrooxidans* and *A. thiooxidans* as the inoculum and found that an increase in *Firmicutes* and reduction in *Proteobacteria* may improve sludge dewaterability. In contrast to this, Li et al. (2022) used AS obtained by Fe^{2+} acclimation as an inoculum during bioleaching and indicated that the relative abundances of the genera *Acidocella, Thiomonas, Acidithiobacillus*, and *Metallibacterium* increased with the addition of Fe^{2+} and AS. These microorganisms are acidophilic bacteria that can grow in acidic environments and adapt to the presence of metals at certain concentrations. It can be deduced that inoculation with AS altered the microbial community in the sludge and increased the relative abundance of specific acidophilic and metal-oxidizing microorganisms. Although the inoculum and sludge used in various studies were different, *Acidthiobacillus* sp. was observed in the samples, the diversity of eosinophilic bacteria was richer, and the microbial community structure was more stable under inoculated indigenous mixed cultures. This finding supports the potential for engineering applications of bioleaching.

16.7 CONCLUSIONS AND PERSPECTIVES

Reducing moisture in sludge content can effectively reduce its volume and subsequent treatment costs. Compared with traditional conditioning techniques, bioleaching is generally considered a more sustainable and environmental-friendly process that can improve sludge dewatering while separating some heavy metals. Bioleaching technology changes the microstructure of sludge flocs through bio-acidification and bio-oxidation, promotes the flocculation of sludge flocs, and transfers internal water to free water for removal. The dewatering mechanism of bioleaching is no longer limited to the study of sludge floc structures and EPS distributions. An increased number of microscopic analysis, such as the analyses of protein secondary structure, hydrophobic/hydrophilic properties, and amino acid composition, enables researchers to understand dewatering mechanisms. Bioleaching processes based on a mixed culture result in more efficient, stable, and easily engineered applications. Although bioleaching has moved from laboratory to engineering applications, some problems still need to be addressed.

1. Development of a mixed culture with higher stability and stronger competitiveness can improve bioleaching efficiency, such as the co-inoculation of *Acidthiobacillus* sp. and acidophilic heterotrophic bacteria, or inoculation with indigenous bioleaching microorganisms.
2. Exploring the bioleaching treatment system based on the continuous operation mode can improve the treatment efficiency in practical applications.
3. Genetic engineering is a powerful technique for improving the performance of bioleaching microorganisms. The challenges of process acceleration, resistance to biological contamination, and competition with the native wild bacteria in sludge can be addressed through genetic technology interventions.

4. Studies at the omics level will effectively manipulate the cellular metabolic processes of bioleaching microorganisms and improve their efficiency. For example, a combination of metagenomics, proteomics, and transcriptomics can deepen our understanding of the metabolic mechanisms of bioleaching microorganisms from a microscopic perspective.
5. More attention should be given to sludge recycling after bioleaching conditioning to provide support for the entire process, such as the production of soil amendment through compost treatment and the preparation of biochar for the advanced oxidation of organic matter.

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17 Detoxification of Sewage Sludge by Natural Attenuation

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17.1 INTRODUCTION

The most important risk associated with sewage sludge (SS) is high heavy metal content that is equally persistent in soil and toxic to crops. Many researchers have shown that final release of SS which are generated from wastewater treatment plants (WWTP) and households constitute serious challenges to environmental safety (Mazzeo et al., 2015; Carita et al., 2019). SS is commonly used in soil amendment owing to its richness in essential plant nutrients that can substitute chemical fertilizers. However, they are equally rich in complex mixture of pollutants that can be harmful to various organisms. When viewed from another perspective, SS is found to be very useful as a source of organic matter and nutrient that can be used to enrich the soil (Mazzeo et al., 2016).

Notwithstanding, the application and disposal methods, several toxic agents present in SS threaten the safety of humans and plants that come in contact with it (Suchkova et al., 2014; Yan et al., 2020). The toxic potential of SS due to diverse organisms at different trophic levels needs to be assessed before final disposal. It is therefore pertinent to find ways to reduce the toxicity of SS prior to its disposal or application as a source of fertilizer (Shou et al., 2019). Sewage detoxification can be handled by many processes which could be physical, chemical or biological. For years, heavy metals have been extracted from SS, with acid washing as a standard test to evaluate the extent of heavy metal uptake by plants grown on sludge amended soils. Other conventional techniques include electrochemical oxidation, thermal treatments and remediation processes such as disposal in landfills, soil replacement amongst others (Mazzeo et al., 2016). Studies have proven that sludge acidification is inconsistent in achieving rapid solubilization of the heavy metals (Mazzeo et al., 2016). Moreover, these technologies are expensive and, in most cases, simply transfer pollutants from one phase to another, thereby making bioremediation approach less efficient and cost-effective. Among all the processes, the application of biological method (bioremediation) is found to be the most environmental friendly, cheap and efficient (Makadia et al., 2011). A holistic knowledge on natural attenuation methods is very vital to advance the knowledge of this research area. In doing this, information on recent researches on various methods of natural attenuation as applied in SS is necessary while the recent global perspective on the application of artificial intelligence in all aspects of studies and technological benefits is indispensable.

Natural attenuation of SS involves biological, physical and chemical process (Dania et al., 2015). Natural attenuation means reduction in the concentration and mass of a substance and the products into which the substance breaks down due to naturally occurring physical, chemical and biological process. Although a viable alternative to the use of SS is its application as a reconditioner of agricultural soils, it may contaminate the soil due to its toxicity of contaminants. Hence, monitored natural

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attenuation (MNA) is a process that is applied to decontaminate the SS before its disposal into the environment (Mazzeo et al., 2016).

This chapter therefore focuses on the detoxification of SS using the natural attenuation methods that do not add more toxics to the environment. The different sources of SS, possible contaminants present in SS and their hazardous effects, different remediation techniques and application of artificial intelligence methods in SS treatment are evaluated. More so, suggestion on the possible proactive control and toxicity minimization approaches are presented.

17.2 NATURAL ATTENUATION

Natural attenuation or bio-attenuation is considered the reduction of the concentration of pollutant in the environment using biological processes such as aerobic and anaerobic biodegradation, plant and animal uptake (Dania et al., 2015). Also, it involves physical phenomena such as dispersion, advection, sorption/desorption, diffusion, volatilization dilution, and chemical events such as ion exchange, complexation, abiotic transformation (Ulrich & Joachim, 2001). In the definition of general natural attenuation, terms such as intrinsic remediation or biotransformation are included. Although one of the crucial components of natural attenuation is biodegradation, which is the breakdown or change in form of compounds orchestrated by living organisms, natural attenuation takes place gradually at most contaminated sites. However, appropriate conditions must exist underground for proper clean-up of polluted sites; otherwise, the clean-up process will be slow or incomplete. Scientists monitor these conditions and processes to ensure that natural attenuation is working and is called MNA. In other words, MNA is a technique used for monitoring or testing the progress of natural attenuation processes with capacity to degrade soil or groundwater pollutants (Mazzeo et al., 2016). If the natural attenuation process is too slow or incomplete, bioremediation could be enhanced either by bio-stimulation or bio-augmentation.

17.2.1 Natural Attenuation Methods

Natural attenuation of SS involves biological, physical and chemical process (Dania et al., 2015). Natural attenuation means reduction in the concentration and mass of a substance and the products into which the substance breaks down due to naturally occurring physical, chemical and biological process. It could be the following destructive chemical processes: aerobic, hypoxic and anaerobic biodegradation and chemical degradation such as abiotic oxidation and hydrolysis. It could also be non-destructive physical process such as adsorption, absorption, adsorption, dispersion, diffusion, dilution and volatilization (Ulrich & Joachim, 2001). Enhanced natural attenuation means the promotion of natural attenuation sites by the addition of chemical, biota or other substances or processes while MNA involves monitoring the natural attenuation as it occurs. When sites need fast removal of pollutants, bioremediation is categorized into bio-stimulation and bio-augmentation (Kouzuma & Watanabe, 2019). Recent methods such as column simulation experiment, reactive transport model and 16S rRNA gene clone library have been applied successfully (Lu et al., 2015).

17.2.2 Stages of Sewage Sludge (SS) Natural Attenuation

The stages of SS natural attenuation are as follows: collecting of SS samples, dewatering of SS samples, detoxification, chemical analyses of aqueous extracts – high power liquid chromatography (HPLC) (to determine dioxins), processing and preparation of SS extracts – Soxhlet method, recombination of yeast assay and assay with Danio rerio embryo (Mazzeo et al., 2015).

Although a viable alternative to the use of SS is its application as a reconditioner of agricultural soils, it may contaminate the soil due to its toxicity of contaminants. Hence, MNA is a process that is applied to decontaminate the SS before its disposal into the environment (Mazzeo et al., 2016). MNA has the following advantages: it is a process that takes place under favourable environmental conditions that result in decrease of mass, toxicity, mobility, volume or concentration of contaminants. It is very efficient, inexpensive and environmentally friendly.

It is as a result of many physical, chemical and biological processes. Chemical processes involve characterizing damage in contaminated areas to ascertain the amount of toxic substances in the environment. It has a limitation of not being able to ascertain the bioavailability of the chemical to the related biota (Moreira et al., 2008). Biological process involves the application of bioassays to establish the actual effect of the contaminants on the environment. The bioassays allow the observation of isolated, or the interactive impacts of substances present in the environment. It can reveal the complexity of the related biochemical and physiological processes (Dania et al., 2015). Examples of such processes mentioned earlier are as follows: biodegradation, dispersion, dilution, adsorption, volatilization, transformation and weathering (Mazzeo et al., 2016).

17.2.3 Bioremediation

Bioremediation is the process that uses microorganisms, green plants or their enzymes to treat the polluted sites to regain their original condition (Megharaj et al., 2011). It may be either aerobic or anaerobic (Weigel & Wu, 2000). It depends on the metabolic potential of microorganisms to detoxify or transform the pollutant molecule. The metabolic potential of the microorganism depends on both bioavailability and accessibility. There are basically two types of bioremediation based on removal of waste: *in situ* bioremediation and *ex situ* bioremediation (Juwarkar et al., 2014).

17.2.3.1 *In Situ* Bioremediation

This is the method whereby organic pollutants are biologically degraded under natural conditions to either carbon(di)oxide and water at the site. The advantages of *in situ* bioremediation include low cost, low maintenance, environmentally friendly and sustainable approach (Juwarkar et al., 2014). *In situ*-bioremediation is preferred to *ex situ* in handling water environment such as wastewater and SS. Three different types of *in situ* bioremediation process are bio-attenuation, bio-stimulation and bio-augmentation (Juwarkar et al., 2014). The method to apply depends on on-site conditions, quantity and toxicity of pollutant chemical species present, indigenous population of microorganism and type of microorganism etc.

17.2.3.2 Bio-Attenuation

This process transforms or immobilizes the pollutants into less harmful forms largely through biodegradation by microorganisms (Smets & Pritchard, 2003). It is the most preferred for non-aggressive approach demanding pollution sites. It is efficient and cost-effective. The major challenges to bio-attenuation are that it is not only adequate and sustainable in many soils or sites that are oligotrophic or has inappropriate microorganisms.

17.2.3.3 Bio-Augmentation

In this approach, microorganisms are amended to a polluted site to hasten detoxification and for degradation (Cheng et al., 2021). Microorganisms of different physiological groups and divisions are brought together to enhance efficiency of bio-stimulation. Table 17.1 represents the recent applications microbes such as trametes, wild living archaea, bacterial in bio-augmentation for detoxification of pharmaceutical wastes, metals, and activated sludge.

17.2.3.4 Bio-Stimulation

This involves the increase of microbial turnover of chemical pollutants through the supply of carbon, nutrients, available oxygen, soil pH, temperature, redox potential and type/of concentration of organic pollutants. The challenge with bio-stimulation is that the additions may be inaccurate and not enough for polluted sites with different kinds of pollutants. However, resource-ratio approach to ascertain the ecophysiological station of pollutants degrading microorganisms helps to

TABLE 17.1
Recent Applications of Bio-Augmentation of Sewage Sludge

S/N	Microbes	Results Obtained	Ref.
1	Bacteroidales and Proteobacteria	NH_3 and H_2S reduction and conservation of N and sulphate	Cheng et al. (2021)
2	Wild living archaea and bacteria (MAB)	Cd, Cr, Ca, Cu, Fe, Pb, Mn, Mo, Ni and Zn removal	Montusiewicz et al. (2021)
3	*Trametes versicolor*	Pharmaceuticals removal at or vstation dichlogene nac-hydrochlorothiaride and ranitidine ferrofibrate	Rodriguez-Rodriguez et al. (2012)
4	Bacteria and Archaea (*Cytophaga* sp. and *Methanoculleus* sp.)	Improved biogas yield/methane yield, improved fermentation.	Lebiocka et al. (2018)

provide theoretical framework for the nutrient formulation optimization in bio-stimulation methods (Ahmed, 2020). Bio-stimulation which often results in its failure are as follows: lesser efficiency, competitiveness and adaptability, relative to the indigenous members of natural communities (Megharaj et al., 2011).

17.2.3.5 Phytoremediation

Also, microbe-assisted phytoremediation has been very a bioremediation technique that uses the ability of plants to promote dissipation of organic pollutants by immobilization, removal and promotion of microbial degradation (Yan et al., 2020). The phenomenon applies the following strategies: phyto-volatilization, phyto-degradation, phyto-stabilization and phyto-extraction. Many plants such as Eucalyptus, willow, *Jatropha curcas*, *Cymbopogon martinii*, *Sedum alfredii*, *Zea mays* etc. have found great applications in detoxification of many metal contaminants found in SS (Table 17.2). The following metals chromium, arsenic, zinc, lead, cadmium, magnesium, potassium nickel, manganese have been successfully removed through the process of phytoremediation using different plants (Suchkova et al., 2014). Table 17.2 contains recent reports on the application of phytoremediation.

TABLE 17.2
Recent Applications of Phytoremediation of Sewage Sludge

S/N	Plant(s)	Contaminants	Ref.
1	Bedstraw, cow vetch, field daisy, silverweed cinquefoil, base vervain and winter cress	Cu, Zn, Mn	Zykova and Isakov (2020)
2	*Cymbopogon martinii*	Cd, Cr, Pb, Ni, Zn, CU, Mn, Fe	Sirigh et al. (2020)
3	Eucalyptus, willow	Cr, As, Cu, Zn, alkanes, PCBs	Nissim et al. (2018)
4	Scots pine, Norway Spruce and oak	Heavy metals	Grobelak et al. (2017)
5	*Sedum alfredii* and *Zea mays*	Cu, Zn, Pb and Cd	Xu et al. (2015).
6	*Amaranthus albus* L., *Amaranthus viridis* L., *Cardaria draba* (L.), (Desr., *Chenopodium album* L., *Cynodon dactylon* (L.) Pers., *Cyperus rotundus* L., *Lolium perenne* L., *Lycopersicon esculentum* mill, molva, parnflora Li, Dortulaca, oleracea	Ca, Mg, K, P, Mn, Cu, Zn, Fe, Cr, Ni, Pb and NG.	Suchkova et al. (2014)
7	Jatropha circus	Zn, Pb, Cr, Cd, Cu	Awalla (2013)

17.2.4 Bioremediation Methods

Based on bio-augmentation and bio-stimulation methods, bioremediation technologies include the following: (i) bioventing, (ii) land farming, (iii) bioreactor and (iv) composting (Li et al., 2017; Shou et al., 2019).

Basically, composting and addition of composted material is meant to reduce volume and water content of waste, destroy pathogens and remove odour-generating compounds. Also, another important hybrid technology of bioremediation is electro-bioremediation (Alba & Seastia, 2021). This is applied for the treatment of hydrophobic organic compounds. Some of the limitations of electro-bioremediation are as follows:

i. toxic electrode effects on microbes metabolism
ii. the availability of the right microorganisms at the site of contamination
iii. the ratio between target and non-target on concentration.
iv. (solubility of the right pollutant and its desorption from the soil matrix.

17.3 SEWAGE SLUDGE SOURCES AND POSSIBLE CONTAMINANTS

17.3.1 Sources Sewage Sludge

SS is generated during wastewater treatment process. The major source of SS is WWTP. However, different WWTPs have different types of treated effluents and sewage treatment processes (Carita et al., 2019). Also, wastewater from industries such as oil and gas (refineries, petrochemicals and natural gas), WWTP, bottling company plants, homes and restaurants and power plants is generated after consuming fresh water for various applications (Figure 17.1). The wastewater is treated for safe discharge into the environment or for recovery of water for reuse. Depending on the process industry, different contaminants are present. Hydrocarbons, oil grease and organic matter [Biochemical oxygen demand (BOD) and Chemical oxygen demand (COD)] come from refineries and petrochemical industries. Heavy metals originate

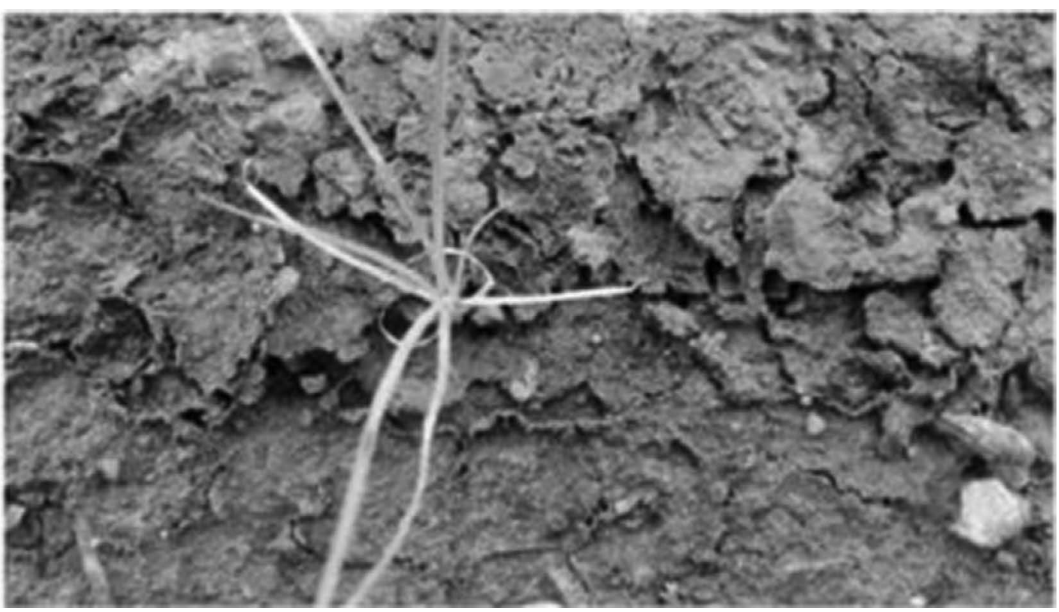

FIGURE 17.1 A pictorial representation of a sewage sludge.

from metallurgical industries. Also, sewage comes from household and sanitation usage of water. Treating the wastewater helps to recover fresh water and a solid waste generated as SS. Examples of the treatment units are as follows: effluent treatment plant at steel industry, membrane bioreactor at a construction company, upflow anaerobic sludge blanket (UASB) at a paper industry and zero liquid discharge plant at a synthetic rubber industry. Figures 17.2a–c shows various sources of SS.

FIGURE 17.2 (a) A domestic wastewater, (b) An agricultural sludge and (c) An industrial wastewater source. *(Continued)*

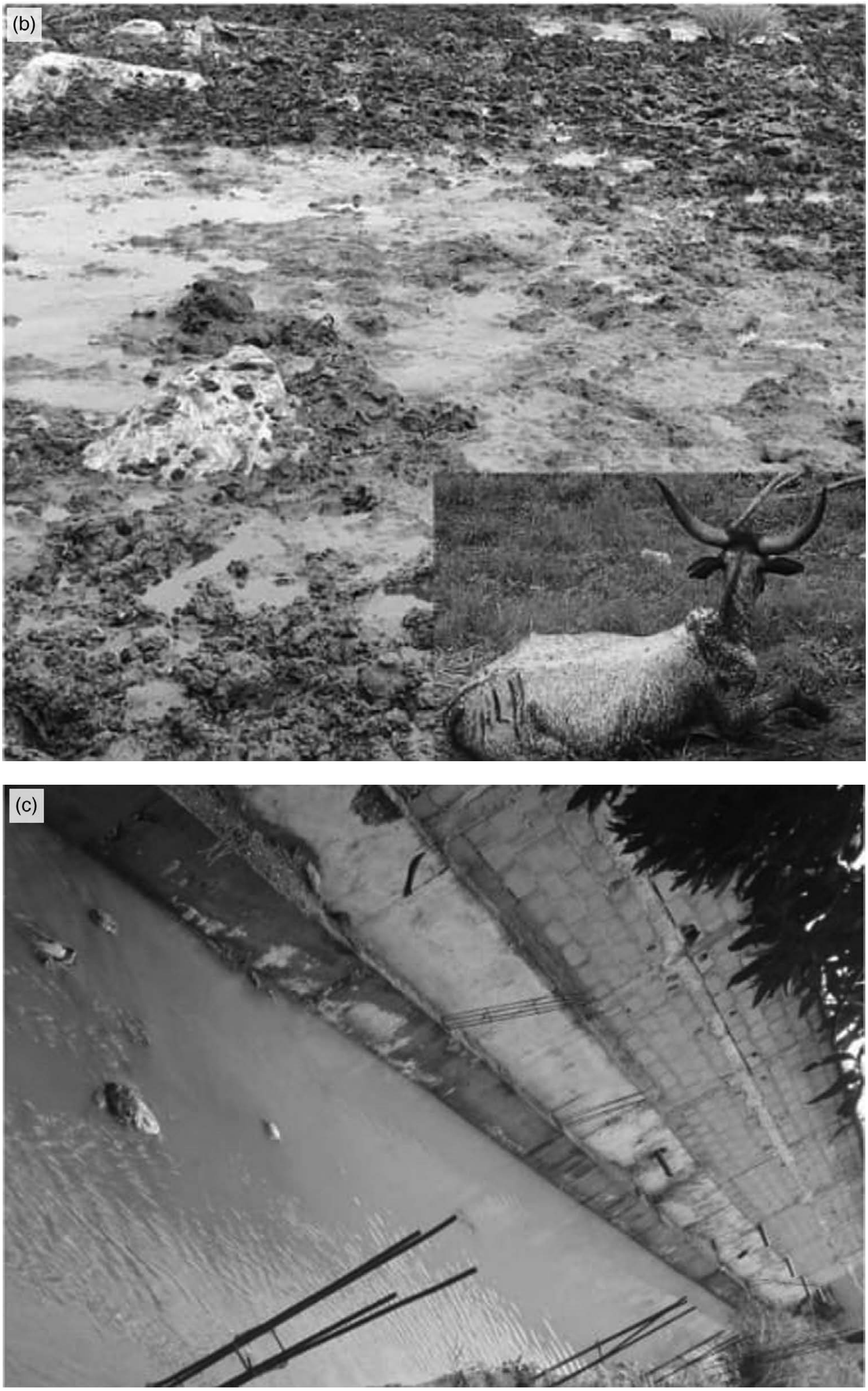

FIGURE 17.2 *(Continued)*

The major sources of SS and their generating processes are as follows:

1. Urban and industrial effluents (food, cable and wood manufacturing)
   Aerated lagoons → sedimentary lagoons → centrifugation of generated sludge → Air drying in piles with mixing
2. Sanitary and industrial sewage.
   Conventional activated sludge → digester → filter process
3. Sanitary and industrial effluents (small tannery and textile industry)
   Biological filter → digester → centrifugation → Drying on a bed
4. Sanitary and industrial sewage (from chemical, pharmaceutical, metallurgical and textile industries)
   Conventional activated sludge → digester → filter process.

17.3.2 Sewage Sludge Contaminants.

SS contains high accumulation of variety of undesirable organic and inorganic contaminants as well as pathogenic microorganisms. It is composed of inorganic and organic as well as toxic (harmful) substances while the toxic substances can be categorized into potential and organic pollutants (Ambika et al., 2022). The contaminants include the following:

- Heavy metals (arsenic, iron, barium, nickel, cadmium, selenium, chromium, silver, copper, zinc and lead)
- Polycyclic aromatic hydrocarbons
- Halogenated organic compounds
- Linear alkyl benzene sulphonates
- Pathogens (*Salmonella* species, *Giardia lamblia*, *Rotavirus*, *Ascaris lumbricoides*)
- Endocrine disruptors (potential risk for human)
- Dioxin-like compounds population
- Oil and grease

Some of the toxic elements can combine and generate the following:

- Aryl hydrocarbon receptor (AhR)
- 2,3,7,8-tetrachlorodibenzo-(*p*)-dioxin (TCDD)
- 10-planar PCBs
- Benzopyrenes

17.3.3 Characterization of Toxic Substances in Sewage Sludge

SS can be characterized using physical, chemical and biological methods (Ambika et al., 2022). Many bioassays and biosensors have been developed for effective toxicity measurements in SS. Integrated approach can be applied for complex SS toxics effluent using chemical analysis through characterization and identification. Also, it has been made possible to combine sample preparation and chemical analysis with biological measurement (Farre & Barcelo, 2003). The toxicity measurement can be carried out by either bioluminescence inhibition method or whole-cell bacterial biosensor. Bacterial acute toxicity assays are applied for determination of phenols, polyethoxylate surfactants, benzene sulphonates, naphthalene, polycyclic, aromatic hydrocarbon (PADs), pharmaceutical drugs, pesticides, linear alkylbenzensulphorates (LABS). The following techniques have been found very useful.

17.3.4 Toxicity Analyses

SS toxicity analysis can be conducted using physical/chemical assays, Bioassays and some recent technique. Details of these techniques have been reported by Hasan et al. (2018). However, useful highlights are presented later.

17.3.4.1 Physical and Chemical Assays

This involves the application of chromatographic techniques such as HPLC – mass spectrometry (MS) (HPLC-MS), gas chromatography – MS (GC-MS), solid phase extraction (SPE), atomic absorption spectroscopy (AAS) (Hasan et al., 2018). The advantages of physical and chemical assays are high sensitivity and accuracy while it equally suffers the following challenges: time consuming, laboratory specific, inability to detect toxicity of multiple contaminants in a sample, requires skilled personnel and highly expensive.

17.3.4.2 Bioassays

Bioassays use the following assay organism such as fish, invertebrate plants, algae and bacteria. Bioassay measurement allows for the assessments of toxicity of SS towards target organism. It is helpful in assessing the risk associated with contaminated sample. Also, it is simple and gives rapid response. It is specific, sensitive and cost-effective. However, bioassays toxic chemicals are not clearly identified and rely on physiological response of living organism which can be interfered.

17.3.4.3 Emerging Toxicity Analyses Techniques

The recent approaches in toxicity analysis involve the application of Daphnids, bioluminescence bacterial and microtox assay. They are more sensitive, selective, portable and eco-friendly and less electrical power demanding.

17.3.5 Effects of Sewage Sludge Contaminants

Among the options for SS disposal, its application as fertilizer in the soil is the best (Sharma et al., 2022). Long-time exposure and frequent contact with SS by workers in the SS treatment plants and farmers make them exposed to infection diseases (Liu et al., 2005; Claxton et al., 2010). Hence, there is need to detoxify the SS before application on the soil. The following adverse effects are associated with SS contaminants: immune dysfunction, endocrine disruption, reproductive toxicity, developmental defects, cancer in vertebrates, soil contamination, aquatic environment pollution etc. (Ngodhe & Odhiambo, 2018).

17.3.6 Sewage Sludge Decontamination Methods

The two main objectives of SS treatment are to reduce the volume of sewage and to stabilize SS. The method used to decontaminate SS depends on the amount of formed solids. Conventionally, larger plants apply anaerobic digestion in bioreactors and aerobic decomposition is suitably applied for small plants. The two methods have their pros and cons depending on energy requirement, degree of treatment, rate of sludge production, process stability and other critical factors as contained in Table 17.3. Decontamination of sludge can be done using chemicals such as formaldehyde (Popora & Baykov, 2014), thermal methods furnace (Dubora et al., 2020). The basic steps are sludge thickening, digestion, dewatering and disposal. *Thickening is achieved* by gravity thickener to reduce the volume to about half. *Digestion* involves microorganisms where the organic matter is converted into simpler substances mainly bacteria using either aerobic or anaerobic digestions to convert about 60% of the sludge into liquids and gases. *Dewatering* is done in an open field mainly in the rural areas. *Disposal* is done either by land fill or as fertilizer of incinerated. Also, *advanced methods*

TABLE 17.3
Anaerobic versus Aerobic Sewage Sludge Treatment (Sharma, 2014)

S/N	Parameter	Anaerobic	Aerobic
1.	Energy requirement	Low	High
2.	Degree of treatment	Moderate (60–90%)	High (95%)
3.	Sludge production	Low	High
4.	Process stability	Low to moderate	Moderate to high
5.	Start-up time	2–4 months	2–4 weeks
6.	Nutrient requirement	Low	High for some
7.	Odour	Potential odour problems	Less opportunity for odour
8.	Alkalinity requirement	High for certain industries	Low
9.	Biogas production	Yes	No

such as membrane bioreactor, aerobic granular sludge system and biological predation have successfully been applied. *Anaerobic process* uses the bacteria that live and reproduce in an environment containing no free oxygen to treat sludge that is a by-product of the wastewater. *Thermo-chemical* methods involve the application of pyrolysis, hydrothermal liquefaction, gasification and wet oxidation. *SS can be disposed through landfills,* incineration, ocean disposal, *application in* production of cement and bricks and agricultural fertilizer and soil conditioning (Carita et al., 2019). Table 17.3 shows comparison of anaerobic and aerobic SS treatment.

17.4 MODELS FOR THE EVALUATION OF EFFECTS OF HEAVY METAL POLLUTION

Due to high content of organic matter and nutrients in SS, they are commonly used as a reconditioner for agricultural soils. However, a major hindrance for its use as agricultural conditioning points to the fact that the SS can harbour complex mixture of toxic compounds that, if introduced to the soil agriculture or water ecosystem, can be made available to the exposed organisms (Engwall & Hjelm, 2000). Owing to the diversity of pollutants contained in SS and the challenge of chemical profiling, bioassays are considered ideal approach for the evaluation of the toxic potential of SS (Chenon et al., 2003). In order to prevent such hazard, concerted efforts have been made in the development of biomonitoring model systems that evaluates the presence of toxic agents in the environment. For example, the effectiveness of natural attenuation of an SS or the cytotoxicity and genotoxicity potential of SS extract can be tested on plant, bacteria, marine and animal models of infection.

17.4.1 Plant Models of infection/Phytotoxicity

Plant bioassays are considered very important because most treated SS is introduced into agricultural soils. Among the diverse plants assays, *Allium cepa* is considered a biological marker for cellular and DNA damages, and has often been used to evaluate the effect of variety of substances, mostly as a result of its high sensitivity, reproducibility and easy manipulation, thus permitting the successful assessment of diverse parameters at several levels, such as, cytotoxicity, toxicity, genotoxicity, and mutagenicity (Mazzeo et al., 2015; Sommaggio et al., 2018).

Anacleto et al. (2017) used the mitotic index (MI) which is related to number of cells undergoing cell division, as well as number of cells nearing cell death (cytoplasmic and/or nuclei vacuolization, heteropyknotic nuclei among others) as a parameter for evaluating cytotoxicity. Genotoxicity was assessed through analysis of chromosomal aberrations based on the different types of abnormalities such as losses, breaks, bridges and delay among others and usually observed at different phases of cell division, whereas mutagenicity was evaluated by analysing the mean of micronucleated cells (Anacleto et al., 2017).

In an attempt to clearly understand the risks involved with the incorporation of SS in agricultural soils and the treatment efficiency, Caritá et al. (2019) also compared the ability of SS obtained from five different WWTP to induce cellular and chromosomal alterations in meristematic and F1 cells of *A. cepa*. Furthermore, the authors ascertained cytotoxicity by calculating the MI for each treatment and counting the number of cell division in meristematic root cells of *A. cepa*. Additionally, genotoxicity was evaluated by counting the chromosomal aberrations and cellular abnormalities in different phases of cell division, whereas mutagenicity was accessed by quantifying the micronuclei in meristematic and F1 root cells of *A. cepa*. Also, Tobacco plant *Nicotiana tabacum* L. var. *Xanthi* – Dulieu was used to evaluate *in vivo* genotoxic potential of sludge and sludge-amended soil samples (Chenon et al., 2003), whereas *Tradescantia* micronucleus (Trad-MN) assay with pollen tetrads was employed by in the evaluation of the genotoxicity of sludge samples and heavy metals, respectively (Majer et al., 2002; Mielli et al., 2009).

Another study used *Vicia faba* micronucleus (MN) test in evaluating the genotoxic potential of the raw SS polluted with hexavalent chromium and composts of SS residue mixed with palm waste (El Fels et al., 2015). *V. faba* micronucleus test is mostly used to monitor the genotoxic effect of organic and inorganic pollutants in soils, sewages and wastewater (Shahid et al., 2011; Kapanen et al., 2013) among others.

For phytotoxicity tests, Walter et al. (2006) assessed the impact of anaerobically digested, heat dried and composted SS on seed germination of Cress (*Lepidium sativum* L.), barley, (*Hordeum vulgare* L.) and oats (*Avenasterilis*) treated with SS. This study observed that germination of the seeds was affected in different ways, thus suggesting the involvement of phytotoxic compounds in the various sludge. In addition, root elongation of the seeds was affected in all assays with the sludge extracts. The authors equally noted that the method of sludge processing influenced the availability of individual metals and caused significant variations in the phytotoxicity test results (Walter et al., 2006).

17.4.2 Animal Infection Models

In the case of animal infection models, Wistar rats were fed with rations of composite SS for 90 days and then monitored the frequency of micronucleated polychromatic erythrocytes (micronucleus test) and DNA damage index by comet assay (Solano et al., 2009). While Chenon et al. (2003) evaluated teratogenic potential of a municipal SS using Frog embryo Teratogenesis Assay-Xenopus (FETAX).

17.4.3 Microbial, Marine and Nematodeinfection Models

Microorganisms can equally be used in bioassay. Kummrow et al. (2010) studied the strains of *Salmonella typhimurium* TA 98 and TA 100 to verify the genotoxic potential of five WWTP and the results exhibited that the aqueous extracts of the different sludges were not genotoxic for the test organism. However, the organic extracts from the five WWTP exhibited mutagenic potential, when tested with *S. typhimurium* TA 98 strain (Caritá et al., 2019).

Yeast based bioassays have equally been employed in monitoring toxic potentials of SS pollutants. Mazzeo et al. (2016) and colleagues used genetically modified yeast strains to reproduce the vertebrate signalling response to either oestrogen receptor or aryl hydrocarbon receptor ligands. Some of these pollutants are able to bind and activate these receptors, thereby compromising fertility and disrupting the functioning of the endocrine system of exposed animals (Mazzeo et al., 2016). Same author equally employed zebrafish embryo assays in evaluation of toxicity of SS disposal in aquatic environment, owing to the sensitivity of zebrafish to xenobiotics. The embryos of zebrafish can be used to monitor specific toxic responses (embryotoxicity) by analysing the induction of deformities, contaminant- or stress-related genes. The cytochrome P450 1A gene, CYP1A, is an important marker in this regard whose expression increases upon exposure of aryl hydrocarbon receptor ligands to zebrafish embryo and adult (Voelker et al., 2007; Olivares et al., 2013).

The strategies of *Rhinocricus padbergi* (a neotropical diplopod specie and a bioindicator of environmental pollution) and *Xiphophorus maculatus* (popularly known as platyfish) to address polluted substrates were investigated by(da Silva Souza et al., 2020). The histological assessment of the midgut and fat body for *R. padbergi* and gills for *X. maculatus* after exposition to SS and biosolids (SS treated with lime) showed alterations such as cytoplasmic vacuolization and hyper-proliferation of regenerative cells in the diplopod and congestion of secondary lamellae and filament epithelium proliferation in the fish gills.

17.5 REMEDIATION TECHNIQUES

Toxic pollutants are of environmental concern owing to non-degradability and possibility of bioaccumulation (Soni et al., 2014), thereby constituting serious harm to humans and the ecosystem by negatively affecting the food chain, water, land use among others (Wuana & Okieimen, 2011). Over the years, environmental pollution, especially with heavy metals has been on the rise, hence remediation processes are important to provide solutions for removing these pollutants so as to protect human health and environment (Martin & Ruby, 2004). Bioremediation is the application of living microorganisms such as bacteria, fungi and plants in degrading or cleaning up harmful environmental pollutants to less toxic forms. Using living organisms such as microorganisms and plants for remediation processes offer an attractive alternative to physicochemical methods for the reduction or elimination of heavy metals by converting environmental pollutants into less harmful forms (Kensa, 2011).

The detoxification and degradation of these toxic pollutants could be achieved through enzymatic transformation of a toxic compound to a lesser or nontoxic component or intracellular accumulation (Jada et al., 2020), which in turn is based on two processes: growth and co-metabolism. In growth, an organic pollutant serves as the only source of carbon and energy which ultimately results in total mineralization of the organic pollutants. In co-metabolism, an organic compound is metabolized in the presence of a growth substrate that acts as the primary source of carbon and energy (Fritsche & Hofrichter, 2008b). Primary principles of bioremediation involve the alteration of pH, redox reactions and adsorption of pollutants from polluted environment in order to minimize its solubility and subsequent conversion to less toxic or inert and more stable products.

Effective bioremediation relies on numerous factors such as suitability of environmental conditions for growth and metabolism which comprises level or concentration of pollutants, adequate temperature, pH, and moisture content (Azubuike et al., 2016; Verma & Jaiswal, 2016). There are three main approaches of bioremediation include the use of microbes, plants and enzymatic remediation (Rayu et al., 2012). The first line of defence against environmental pollutants such as heavy metals are microorganisms, which possess numerous strategies for survival in these polluted environments (Agarwal et al., 2018, 2019). There are numerous reports with evidence of microbial ability to detoxify SS, industrial pollutants and the remediation of soils contaminated with heavy metals (Guarino et al., 2017; Mateos et al., 2017; Dhaliwal et al., 2020). These microbes do not necessarily degrade the heavy metals but can transform them by changing their physical and chemical attributes. One of the adaptive mechanisms for microbial survival in heavy metal contaminated environments is through the variation of genetic materials such as possession of mer operon for mercury tolerance normally found on the chromosome(s), plasmid(s) or as a component of transposons (Bosecker, 1999). The detoxification mechanism adopted by microbes could be by bioaccumulation, biosorption, biotransformation and bio-mineralization which are exploited for bioremediation process due to their cost-effectiveness, easiness to handle and higher efficiency (Cappello et al., 2015). Fungi, bacteria and yeasts are involved in the degradation of pollutants but reports of the involvement of algae and protozoa are few. The various approaches for bioremediation are shown as a sketch in Figure 17.3.

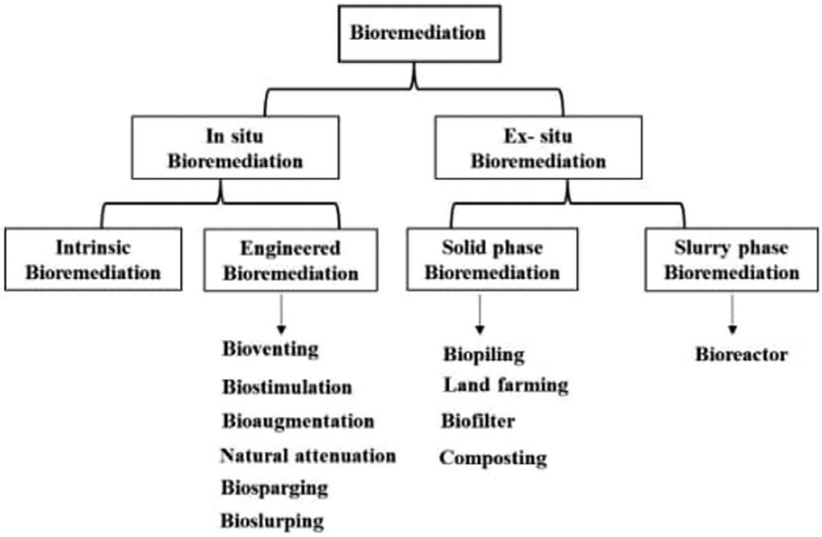

FIGURE 17.3 Various approaches for bioremediation.

Bioremediation process can be classified into three phases, natural attenuation, bio-stimulation and bio-augmentation. Natural attenuation is a situation whereby contaminants are reduced solely by autochthonous microbes and rely on aerobic and anaerobic processes of microbial pollutant degradation without external intervention. Although no external force is applied, the process is monitored so as to establish an ongoing and sustainable bioremediation process, hence it is termed MNA.

In bio-stimulation process, growth and viability of the indigenous microbes are stimulated by supplementing the system with growth-optimizing factors such as temperature, nutrients, oxygenation, pH, and biosurfactants in order to accelerate biodegradation (Mateos et al., 2017). In bio-augmentation, enriched mixed microbial consortium or recombinant organisms with better remediation capacity is introduced into contaminants (Mariano et al., 2009). Many factors influence microorganisms to use pollutants as substrates or co-metabolize them, such as, the genetic potential and certain environmental factors such as temperature, pH, and available nitrogen and phosphorus sources, then, seem to determine the rate and the extent of degradation (Fritsche & Hofrichter, 2008a). Therefore, applications of genetically engineered microorganisms (GEMs) in bioremediation have received a great deal of attention. Examples of some microorganisms implicated in bioremediation of environmental pollutants include Bacteria genera such as *Pseudomonas*, *Arthrobacter*, *Alcaligenes*, *Xanthobacter*, *Corynebacterium*, *Flavobacterium*, *Mycobacterium*, *Achromobacter*, *Nitrosomonas*, *Bacillus*, Fungi species such as *Phanerochaete chrysosporium*, *Penicillium*, *Graphium*and yeast species such as *Candida lipolytica*, *Rhodotorula mucilaginosa*, *Yarrowia*, and *Pichia* among others (Singh et al., 2014).

17.5.1 Myco-remediation

Myco-remediation is a type of bioremediation wherein fungal species are used to decontaminate or recover polluted areas. This term was coined by Paul Stamets and specifically entails the use of fungal mycelia in bioremediation. Fungi are crucial in bioremediation because, such as bacteria, they are able to metabolize dissolved organic matter and principally responsible for the decomposition of carbon which is achieved by the mycelium that secretes extra-cellular enzymes and acids that break down the main components of plant fibre, lignin and cellulose. Lignin and cellulose are organic compounds made up of long chain of carbon and hydrogen with structural similarity to many organic pollutants. However, unlike bacteria, fungi can thrive in low pH solutions and moisture areas which assist them in the breakdown of organic matter. Mycorrhiza is a symbiotic association that occurs between fungi and the root of a vascular plant. In mycorrhizal interaction, the fungi colonize the plant roots, either intracellularly as in arbuscular mycorrhizal fungi (AMF), or extracellularly as in ectomycorrhizal fungi. They remain crucial component of soil life and soil chemistry. Bioremediation through application of mycorrhiza is referred to as mycorrhizoremediation.

17.5.2 Bioremedition Strategies

In the selection of any bioremediation technique, it is imperative to consider the nature of pollutant, depth and amount of pollution, nature of environment, cost, location and environmental policies.

Bioremediation can be carried out *in situ* or *ex situ*. The procedure for developing bioremediation techniques may entail the following steps:

a. Isolation and characterization autochthonous microorganisms with bioremediation potential.
b. Laboratory cultivation to develop viable populations.
c. Studies on the catabolic activities of these microbes in polluted material through bench scale experimental assays.
d. Monitoring and measurement of bioremediation progress through chemical analysis and toxicity tests in chemically contaminated media.

17.5.3 *In Situ* Bioremediation

In situ bioremediation involves on site clean-up process of contaminated environment (Rayu et al., 2012; Elekwachi et al., 2014). Little or no movement, excavation or the disruption of soil construction is needed. There are two types of *in situ* bioremediation Intrinsic or natural attenuation and engineered bioremediation. This technique is most desired due to minimal cost and less disturbance since the treatment is performed in place, therefore, excavation and transportation of contaminants are not necessary.

17.5.3.1 Intrinsic Bioremediation

Intrinsic bioremediation involves passive remediation of polluted sites, without any external human assistance. It deals with the stimulation of autochthonous microbial population in biodegradation of environmental pollutants based on microbial aerobic and anaerobic processes. This technique is less expensive compared to other *in situ* techniques since no external intervention or force is needed.

17.5.3.2 Engineered *In Situ* Bioremediation

This approach incorporates the introduction of specific microbes to the polluted site. The use of GEMs in *in situ* bioremediation speeds up the degradation process by enhancing the physicochemical conditions that encourage microbial growth.

17.5.3.3 Bioventing

This technique entails controlled stimulation of air flow by supplying oxygen to unsaturated or vadose region with the aim of increasing activities of autochthonous microorganisms for bioremediation. Amendments can be done by the addition of nutrients and moisture. The primary parameters include air flow rates and air intervals; the success of this technique in bioremediation process depends on the number of air injection points, which assists in maintaining uniform air distribution. It is applicable for simple hydrocarbons and can be applied in areas where contamination is deep under the surface.

17.5.3.4 Bioslurping

This technique achieves remediation of polluted soil and water by a combination of bioventing soil vapour extraction and vacuum-enhanced pumping through indirect provision of oxygen and stimulation of contaminant biodegradation (Gidarakos & Aivalioti, 2007). The design of this technique is intended for free product recovery such as light non-aqueous phase liquids (LNAPLs), hence, remediating both unsaturated and saturated zones. This technique is equally used to recover soils polluted with volatile and semi-volatile organic compounds. The system makes use of a "slurp" that stretches into the free product layer, which takes up liquids from this layer. Through an upward movement, the pumping machine transports LNAPLs to the surface where it is separated from air and water. Excessive soil moisture prevents air permeability and reduces the rate of oxygen transfer, thus limiting microbial activities.

17.5.3.5 Biosparging

Biosparging entails the injection of air under pressure below the water table to increase the oxygen concentrations in groundwater. This technique is slightly similar to bioventing in that there is an injection of air into sub-surface to enhance microbial activities, but contrary to bioventing, air is injected at the saturated area which can lead to upward movement of volatile organic compounds to unsaturated area for the promotion of biodegradation. The efficiency of this technique relies on two factors: soil permeability and pollutant biodegradability (Philp & Atlas, 2014).

17.5.3.6 Permeable Reactive Barrier

Permeable reactive barrier is an *in situ* technique used in the recovery of groundwater contaminated with diverse pollutants, including heavy metals and chlorinated compounds. In this approach, a semi-permanent or permanent reactive barrier (medium) that is mostly composed of zero-valent iron (Garcıa et al., 2014; Zhou et al., 2014) is submerged in the trajectory of polluted groundwater. As the contaminated water flows across the reactive barrier under its natural gradient, these contaminants are trapped and they equally pass through a series of reactions which results in clean water in the flow through (Obiri-Nyarko et al., 2014).

17.5.3.7 Phytoremediation

Phytoremediation is defined as the process of treating polluted environment with plants in order to eliminate the pollutants. The main principle of phytoremediation entails breaking the pollutant by the roots of plants to less toxic element or absorption of the pollutant, and storing it in stems and leaves of the plant (Kaur et al., 2018). Phytoremediation focuses on two general categories: the identification of plant species with capacity to tolerate high levels of metals (also known as hyperaccumulators) and the use of high-biomass crops combined with chelating agents that can solubilize soil heavy metals and enhance its uptake by plants for the removal of the pollutant. A lot of plant species have shown the capacity to tolerate high levels of heavy metals with great importance for phytoextraction purposes (Memon et al., 2001; Memon & Schröder, 2009). About 400 hyperaccumulator plant species have been identified and mainly belong to *Euphorbiaceae*, *Asteraceae*, *Fabaceae Flacourtiaceae*, *Caryophyllaceae*, *Lamiaceae*, *Violaceae*. *Poaceae* and *Brassicacea*e plant family among others.

Phytoremediation approach has numerous benefits, including minimized cost, public acceptance and most importantly the ability to simultaneously remove organic and inorganic pollutants. This technique exploits the natural trait of some plants to hyper-accumulate essential heavy metals in various tissues (Rascio & Navari-Izzo, 2011). In addition, plants possess several secondary metabolites (Hadacek, 2002) that are considered pollutant analogues within the network of supra metabolism, with implications for forecasting the fate of contaminants (Singer et al., 2004). Secondary metabolites such as cymene, limonene, carvone and pinene have been reported to show expression of catabolic genes by the rhizosphere or plant-colonizing bacteria leading to enhanced biodegradation of polychlorinated biphenyls (Singer et al., 2003).

Plants such as cotton (*Gossypium hirsutum*), a non-edible important fibre crop can be very efficient for the uptake of heavy metals from metal-polluted soils (Kaur et al., 2018).

The use of metal accumulating plants for removal of heavy metals from polluted water and soil has numerous benefits such as minimized cost, production of a recyclable metal-rich plant residue, less environmental perturbation, applicability to various ranges of toxic metals and radionuclides, removal of secondary air or water-borne contaminant, and general acceptance. There are five types of phytoremediation techniques, and these are classified on the basis of the contamination fate. phytoextraction, phytostabilization, phytotransformation, rhizofiltration, phytodegradation. In addition, the combination of these techniques is possible in nature. They are further discussed as follows:

a. *Phytoextraction or phytoaccumulation* entails the procedure used by the plants to accumulate pollutants into the roots and aboveground shoots or leaves. It saves remediation costs by accumulating low levels of pollutants from a widespread zone. Contrary to the degradation mechanisms, this technique produces a mass of plants and pollutants (normally metals) that can be transported for disposal or recycling.
b. *Phytotransformation or phytodegradation* involves the uptake of organic pollutants from soil, water or sediments and their subsequent transformation to a more stable, less toxic, or less mobile form. For example, metal chromium can be reduced from hexavalent to trivalent chromium, which is a less carcinogenesis and mobile form.
c. *Phytostabilization* is an approach wherein plants minimize the mobility of polluted soil. Leachable constituents are adsorbed and bound into the plant structure to form a stable mass of plant from which the pollutants cannot go back into the environment.
d. *Phytodegradation or rhizodegradation* refers to the breakdown of pollutants through the activity of the existing rhizosphere. This activity is a result of the presence of proteins and enzymes produced by the plants or by soil microbes. Rhizodegradation is a symbiotic relationship existing between plants and microbes in soil wherein plants provide nutrients that are essential for these microbes to thrive, while in return, the microbes provide a healthier soil environment for plant growth.
e. *Rhizofiltration* is a water remediation technique that entails the uptake of pollutants by plant roots. Rhizofiltration is used to minimize pollution in natural wetlands and estuary areas.

Phytoremediation is mainly appropriate for use at large field sites where other remediation methods are not practicable or more expensive; at sites with a low pollutant concentration where only polish treatment is needed for long periods; and in combination with other techniques where vegetation is applied as a final cap and closure of the site. However, there are limitations to this technique which include, long duration of time for remediation, possibility of contamination of the vegetation and food chain, and difficulty in maintaining vegetation at some sites with high level of toxic pollutants. Several reports have stated that the bioavailability of metals and their uptake by plants can be enhanced through the addition of fertilizers, organic supplements, chelating agents, and adjustment of pH.

17.5.4 *Ex Situ* Bioremediation Techniques

Ex situ bioremediation technique involves the excavation of pollutants from polluted sites and subsequent transport to another site for proper treatment. The choice of this technique is based on the type of contaminant, degree or depth of pollution, cost of treatment and geographical location of the polluted site. Some examples of *ex situ* bioremediation technique are mentioned later.

17.5.4.1 Biopile

Biopiles are hybrid of composting and land farming and involve piling of excavated contaminated soil above ground with subsequent fortification of nutrients and oxygen to enhance metabolic activities for effective bioremediation. Irrigation, leachate collection, nutrients and treatment beds are the main components of this technique. The biopile is mostly used to remediate low molecular weight volatile contaminants in extremely cold regions (Gomez & Sartaj, 2014; Dias et al., 2015).

17.5.4.2 Windrows

Windrows rely on periodic rotation of pied contaminated soil to augment bioremediation process by increasing activities of autochthonous and transient hydro-carbonoclastic microbes that are present in contaminated soil.

17.5.4.3 Bioreactor

Bioreactor is a vessel in which raw materials are transformed to specific products following a series of biological reactions. Diverse operational modes of bioreactors exist and include: batch, fed-batch, sequencing batch, continuous and multistage. The conditions in a bioreactor favour natural process of cells by mimicking and maintaining their natural ecosystem to provide optimum growing conditions. The contaminated samples can be introduced into a bioreactor as a slurry or dry matter.

Compared to other *ex situ* bioremediation techniques, one of the major advantages of a bioreactor is the possibility of controlling the bioprocess parameters (pH, agitation, temperature, substrate and innoculum concentrations and aeration rates). The flexibility in the design of a bioreactor permits maximum biological degradation while minimizing abiotic losses (Venkata Mohan et al., 2007).

17.5.4.4 Land Farming

Land farming is the simplest bioremediation technique owing to its low cost and minimal equipment needed for its operation. It is still debated as to whether it should be considered *in situ* or *ex situ* technique due to the treatment site. The depth of the pollutant determines whether this technique can be performed *in situ* or *ex situ*. Usually in land farming, contaminated soils are tilled, but the treatment site determines the type of bioremediation. Generally, tilled contaminated soils are placed on a fixed layer support slightly elevated above the ground surface to permit.

aerobic biodegradation of contaminant by indigenous microbes (Philp & Atlas, 2014; Silva-Castro et al., 2015). In Land farming, tillage, which enhances aeration, nutrient supplementation (nitrogen, phosphorus and potassium) and irrigation are the main operations that stimulate activities of indigenous microbes.

17.5.5 Application of Microorganisms/genetically Engineered Microorganisms

GEM is a microbe whose genetic information has been manipulated using genetic engineering techniques that are inspired by natural genetic exchange between microbes. By the application of recombinant DNA technology, the autochthonous microorganisms are enhanced to degrade specific pollutants or new recombinant bacteria with the capacity to tolerate metal stress by over-expressing metal-chelating proteins and peptides; hence, the ability of metal accumulation is produced for bioremediation application (Menn et al., 2008; Singh et al., 2020) and they consist as follows:

1. Modification of enzyme specificity and affinity
2. Pathway construction and regulation
3. Bioprocess development, monitoring and control
4. Bio-affinity bioreporter sensor applications for chemical sensing, toxicity reduction and end-point analysis

Genes involved in the degradation of environmental contaminants, for example, toxic wastes, chlorobenzene acids, toluene and other halogenated pesticides have been identified. A single plasmid is required for every compound, but there is potential for creating microbial strain with the capacity of degrading various types of hydrocarbon (for example, a multiplasmid-containing *Pseudomonas* *putida* strain capable of oxidizing aliphatic, aromatic, terpenic and polyaromatic hydrocarbons. The plasmids involved in the degradation of toxic compounds can be grouped into four categories:

1. CAM plasmid involved in the decomposition of camphor
2. XYL plasmid involved in the degradation of xylene and toluenes
3. OCT plasmid which is involved in the degradation of octane, hexane and decane
4. NAH plasmid that degrades naphthalene

Furthermore, the recent advances in recombinant DNA technologies have also made way for conceptualizing "suicidal genetically engineered microorganisms" (S-GEMS) in order to reduce possible hazards and to achieve effective and safer removal of contaminants in polluted sites (Pandey et al., 2005).

17.6 APPLICATION OF ARTIFICIAL INTELLIGENCE TECHNIQUES IN NATURAL ATTENUATION OF SEWAGE SLUDGE

Artificial intelligence, machine learning and deep learning are major interrelated tools for system monitoring and control, design, optimization and data analyses (Onukwuli et al., 2021). Effective modeling, design and optimization performance requires the application of artificial intelligence, machine learning and deep learning (Onukwuli et al., 2021). However, artificial intelligence has been successfully applied in almost all facets of learning, research and development. Critically, many artificial intelligence techniques are available through different computer software such as design expert, Minitab, mat lab etc. The most widely used modeling and optimization techniques are response surface methodology (RSM), artificial neural networks (ANN), adaptive neuro-fuzzy inference system (ANFIS) etc. These could be coupled with some algorithms such as particle swam, desirably function, genetic algorithm to enhance modeling and optimization of desired responses while considering some mostly influencing process conditions or independent variables.

SS natural attenuation has received some attention in the application of artificial intelligence. The major statistical optimization technique has been RSM based on tacchuchi or box-behnken (BB) design of experiment through central composite design (CCD). The response variables that have been considered include biohydrogen (Gnanabal et al., 2018), biethanol (Alam et al., 2006), biogas (Leangliang et al., 2019), optimal elasticity demand and maximum revenue (Liu et al., 2021), COD removal, efficient sludge quality (Valid et al., 2016), hydraulic retention and sub-state influent concentration (Rahman et al., 2016). Table 17.4 contains more details on the application of RSM in the optimization of bioremediation of SS.

17.7 CONCLUSIONS AND PERSPECTIVES

SS from WWTP constitutes a possible alternative to agricultural fertilizer. However, their use is limited by the presence of toxic substances that may be hazardous to humans and environment. There are well-developed studies on various approaches to remediate SS matrices, and impressive results have been obtained, but there are still many gaps. For instance, the mechanistic underpinnings of methods are still underexplored and the systematic evaluation standards for methods are not well developed. The mechanisms of the current removal of heavy-metal pollutants from SS need to be deeply studied and optimized in addition to innovation of agents, process improvement and the integration of diverse methods. Each method possesses its advantage and disadvantage. The natural attenuation method for SS detoxification is effective in reducing the presence of pollutants. The first step to a successful bioremediation is the characterization of the site which will help in establishing the most suitable bioremediation technique. *Ex situ*-bioremediation technique seems to be more expensive owing to the additional costs of excavation. However, cost of installing equipment on site may render *in situ* technique less efficient. This review equally demonstrates an array of bioassays for detecting toxic potential and the risks associated with SS.

TABLE 17.4
Overview of AI/Modeling Techniques Application in (Wastewater) and Sewage Sludge Natural Attenuation

S/N	AI/Modeling Techniques	Independent Variables	Dependent Variables	Wastewater/Sewage Sludge	Types of Reactor	Ref.
1	Factor analysis	Biogas yield	Biovine dung, primary sludge; pH, volatile fatty acid (VFA), volatile solids (VS)	–	–	Nikiema et al. (2022)
2	Taquchi and RSM/BB DoE	Nutrient removals (nitrate and phosphate)	Zinc chloride, impregnation ratio and attraction times using activated carbon from banana trunk	Bio sludge extracted from cow	–	Sarva et al. (2022)
3	Simulation model – (MILP) and system dynamics (SD)	Economic feasibility of co-digestion based on optimal electricity demand. Net present value (NPV), maximum revenue	Volume of digester, unit of food waste collection transportation cost, economic life span, labour cost, electricity, price for sewage plant, methane energy content, electricity demand of sewage plant in a day	Sewage sludge	–	Liu et al. (2021)
4	RSM/CCD	Methane yield	C/N ratio F/M ratio pH	Rice straw (*Hydrilla verticillata*)	Batch	Jasini et al. (2020)
5	RSM/CCD	HRT	COD influent	Yogurt influent	Hybrid EGSB-E bioreactor	Jasini et al. (2020)
6	RSM/CCD Box-Behken	Strong time	Temperature	Cattle manure and canola residues	Batch reactor	
7	RSM/CCD	Biomass support	Sorbent dosage	Textile wastewater	MSBR	Jasini et al. (2020)
8	RSM/CCD	Methane yield	• Total solid • Proportion of co-support • Inoculation concentration	Potato waste aquatic weed	BR	Jasini et al. (2020)
9	RSM/CCD	Biogas yields	Temp, pH, substrate concentration	Rice straw	Floating drum anaerobic digester	Jasini et al. (2020)
10	RSM/CCD	Biogas production and methane yield	OLR, temperature mixing level	Cow manure	Mixed plug flow reactor	Jasini et al. (2020)
11	RSM/CCD	Biogas production	pH, substrate concentration TOC	Organic faction municipal waste	BR	Jasini et al. (2020)

(Continued)

TABLE 17.4 ***(Continued)***
Overview of AI/Modeling Techniques Application in (Wastewater) and Sewage Sludge Natural Attenuation

S/N	AI/Modeling Techniques	Independent Variables	Dependent Variables	Wastewater/Sewage Sludge	Types of Reactor	Ref.
12	RSM/CCD	Decolourization of Ramazol Brilliant using oxidation enzymes.	Initial substrate cone (30–100 mg/L), incubation period (5–25 days) and pH (5–7). Optimum conditions 57.15 mg/L, pH–6, 8.55 days. Max removal 78.34%, Lac 0.22 µ/mg, MnP 0.24 µ/mg and LIP 14.22 µ/mg.	Sewage Sludge	–	Efaq et al. (2020)
13	RSM/CCD	Biogas production (mL)	Sewage sludge ratio (%) and Feed concentration (g VS/L) as factors. Optimum conditions for maximizing methane potential were an SS:CM:Ms ratio of 30:35:35 and bulk VS concentration of 15.0 g VS/L to give 8047.31 mL methane (C/N ratio of 127).	Sewage Sludge	–	Leangliang et al. (2019)
14	Taguehi orthogonal array of L/6 DoE/ CCD of RSM.	Biohydrogen	Optimum conditions = temp = 25°C, pH = 6.3, innoculum volume = 4.7% and substrate concentration = 1.8% at 24 h incubation	–	–	Gnanabal et al. (2018)
15	RSM on CCD/full factorial	Removal of starch (COD removal), effluent and TSS of 38 mg/L and SVI of 57 mL/g	Impeller diameter, reactor geometry	Sewage Sludge	–	Valid et al. (2016)
16	RSM/BB	Bio decolorization of synthetic dye solution calvalaria Sp.	Initial dye concentration (20–100 mg/L), Ph (2–8) and Temp (25–40°C). Optimum conditions: 60 mg/L, 5, 32.5°C, 100% decolourization	Sewage Sludge	–	Senthilkumar et al. (2015)
17	RSM/CCD	1. Urban wastewater/SBR – adsorbent = powdered Zehae (Pz). Pz = Portland cement + zeohe	Contact time (11.7 h), aeralin rate (2.87 L/mn) and leachate to wastewater ratio (20.13%). Removal efficiency – Te = 79.57, Mn = 73.38, Ni = 79.29 and Cd = 76.96.	Sewage Sludge	–	Anun et al. (2014)
18	RSM/CCD	Carbon and Nitrogen removal	mixed liquor suspended solids, chemical oxygen demand L COD J/ N ratio aerobic time and cycling time, COD	Sewage Sludge	–	Maliche et al. (2012)
19	RSM/CCD	Bioethanol yield	Temperature, pH, innoculum volume = and substrate concentration at 24 h incubation		–	Alam et al. (2006)

Controlling the discharge of pollutants from specific commercial sectors and putting in place remedial measures to minimize the input from some domestic and diffusible sources can assist in minimizing inputs to SS production. There is a need for increased awareness on household refurbishment (e.g., for old lead paint and piping) and proper education on the disposal of potential pollutants down the sinks, and to raise the curiosity of the ecological implications of various processes and products in urban wastewater. Most times, the reports of inputs of potentially toxic elements to urban wastewater quote data from earlier research studies. There is little or absence of recent quantitative information on the sources and input of these potentially toxic elements to urban wastewater. Furthermore, the sources of the pollutants in wastewater need to be identified and more work is required to clearly understand partitioning and speciation of these pollutants, especially for compounds, such as nonylphenol ethoxylates (NPE), that could become more toxic through wastewater treatment. Another area of concern is the evaluation of the impact of cocktail effects of several pollutants present at the same time. For instance, the interactions between metals and organics are complex and may be synergistic or antagonistic interactions and clearly need to be understood. In addition, diverse range of ecotoxicological tests using various organisms at different trophic levels need to be adopted to clearly understand the effects of SS prior to disposal in soils or water stream.

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