Biochemical Evaluation of Landfill Leachate and Gas Generation Dynamics Followed by Treatment Under Actual Landfill Conditions

A thesis submitted

In partial fulfillment of the requirement for the degree of

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by

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This is to certify that the thesis entitled "**Biochemical Evaluation of Landfill Leachate and Landfill Gas Generation Dynamics Followed by Treatment Under Actual Landfill Conditions**" submitted by Chejarla Venkatesh Reddy (156104038), a Research Scholar in the Department of Civil Engineering, Indian Institute of Technology Guwahati, for the award of the degree of Doctor of Philosophy, is a record of an original research work carried out by him under my supervision and guidance. The thesis has fulfilled all requirements as per the regulations of the institute and, in my opinion, has reached the standard needed for submission. The results embodied in this thesis have not been submitted to any other University or Institute for the award of any degree or diploma.

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Abstract

Landfilling is one of the major municipal solid waste (MSW) disposal methods practiced worldwide. Though it is considered the most cost-effective means of waste disposal, poor management practices in developing countries such as India are the major cause of environmental problems such as groundwater, surface water, and air pollution. The solid waste characteristics in India, the organic fraction of waste is up to 40 to 80% depending on the income and lifestyle of the population. The current situation of landfills/open dumpsites leads to the generation of uncontrolled gases and leachate generation. Therefore, it is very challenging to describe the current and future leachate pollution of existing landfills due to the high amount of fresh organic waste and uncontrolled anaerobic biochemical processes in the landfills. In the future, the requirement for leachate and gas collection systems, operation of treatment technologies in developing countries would be safeguarded for environmental and human livelihood. Thus, quantifying leachate and gas composition from the landfill system is a prerequisite for landfill operation and treatment. Models for predicting leachate constituents as design parameters for leachate treatment, monitoring, and operation of existing and new landfills.

The environmental impacts of landfills depend on several factors, including waste composition, technical barriers, landfill operation, and climatic conditions. A profound evaluation of all factors and their impact is necessary in order to evaluate the environmental threats emanating from landfills. The composition of landfill leachate and gas generation and the extraction of potential pollutants from the waste depend upon several factors, including solid waste composition, degree of compaction, the absorptive capacity of the waste, waste age, seasonal weather variations, and levels of precipitation. However, the available literature deals with biologically/pre-treated waste, leachate recirculation with water addition, and controlled temperature. Information on operational aspects, leachate, and gas dynamics for fresh landfill leachate are rather limited. Additionally, very limited investigations have been made on the combination of physico-chemical and biological treatment of fresh landfill leachate. Therefore, this research deals with the simulation of fresh municipal solid waste in existing operational aspects (no addition of any inoculum and recirculation of leachate) of landfills/open dumpsites (actual conditions) and a comprehensive understanding of the biochemical assessment of leachate and gas dynamics

within the landfills. An investigation was done on the combined process for the removal efficiency of fresh landfill leachate pollutants using a combination of physico-chemical and biological treatment methods.

In order to accomplish the objectives, studies were carried out on solid waste treatment technologies for Guwahati city using a life cycle assessment tool (LCA) and assessment of landfill leachate and gas dynamics using landfill simulation reactors operated under actual landfill conditions. The research work was divided into three phases. In **phase-I**, the adopted methodology was a life cycle assessment of different solid waste treatment strategies for Guwahati city. Primarily, the study was considered four different scenarios based on current and future solid waste treatment strategies, i.e., scenario 1 (S1): Windrow composting + open dumping (COMP_OD), scenario 2 (S2): Windrow composting + sanitary landfill with gas utilization (COMP_SLF) scenario 3 (S3): Refuse-derived fuel + windrow composting + sanitary landfill with gas utilization (RDF_COMP_SLF) and scenario 4 (S4): Manual sorting + windrow composting + sanitary landfill with gas utilization (MSP_COMP_SLF). This study aimed to analyze and compare the environmental impacts due to different waste treatment technologies and the life cycle costing of each scenario to promote the more suitable waste management option. LCA study concluded that S4 is the best choice from the perspective of the environment because of its minimum environmental impacts due to methane emissions, SO_X, NO_X, and heavy metal pollution can be avoided by separating waste materials and found to be suitable for the Guwahati municipal solid waste management. S1 has an economic advantage compared to other scenarios because of less maintenance and operating cost, while future scenarios improve the overall sustainability compared to the current situation. In **phase-II**, the study primes to understand the biochemical evaluation of fresh municipal solid waste landfills in terms of the anaerobic degradation of mixed solid waste and examine the seasonal variation on solid waste stabilization rates. Fabrication of a laboratory-scale landfill simulation reactor (LSR) of dimensions 1m×1m×1.1m (length, width, and height, respectively). The reactors were named as reactor 1 (R1) and reactor 2 (R2), and it represents landfills/open dumpsites in India, where R1 operated without rainfall, had a high amount of wet waste (73%) and dry (27%); and R2 operated without rainfall had a high amount of dry waste (62%) and wet (38 %) respectively. Reactor 3 (R3), operated at the weighted average actual rainfall rate of 50 years, containing the waste composition of wet waste (73%) and dry waste (27%). The LSR was then fed with un-shredded mixed MSW of the composition. Monitoring of leachate from the simulated reactors was done once a week, and gas was

done twice a week during operating phases. The exponential model was developed using simulated landfill leachate data to predict the future trends of landfill leachate pollution with time. Results concluded that R1 leachate of organic compounds such as BOD_5 (13980-50250 mg/L) and COD (19056-92160 mg/L) had decreased significantly within one year, but the strength of the leachate was high. It was observed that there is no leachate production in the R2, which contains 38% of wet waste and 62% of dry waste due to the high amount of dry waste-filled that may not reach field capacity and signifies that developed countries have less problem handling the leachate issues. The fresh MSW landfill reactor operated with rainfall was evident that concentrations of leachate parameters were high in initial phases and a significant decrease in BOD_5 (7041-39310 mg/L), COD (15692-71630 mg/L), and TS (9077-33200 mg/L), respectively. Therefore, rainfall had a direct influence on leachate quality. A high amount of CO_2 (56.73%) in R1 and 59.85% of CO_2 in R3 keeps the landfill system in an acidic phase due to the dissolution of CO_2 in landfill leachate to form carbonic acid. Therefore, it would be a better representative of landfill cells in open dumpsite conditions with no leachate and gas collection systems that mostly have confined conditions except in the top layers or below the final cover. The developed nonlinear curve fit (exponential decay model) showed the goodness of fit of the leachate parameters. The model performance is suitable for cities where the wet waste composition is greater than 70%. Leachates from the dry season have shown a much higher concentration of pollutants than those produced during the rainy season, and higher precipitation leads to a higher rate of waste decomposition in the landfill system. In **phase-III**, treatment of fresh landfill leachate from the high amount of organic waste contained MSW landfill using combined treatment system, i.e., the coagulation-flocculation process (pretreatment) followed by an upflow anaerobic filter under conventional way and evaluation of the treatment performance based on removal efficiency of organic pollutants. The pretreatment process was used for R1 leachate (L1) to obtain the best coagulant with the optimal dosage and pH for the various coagulants that are available, and the upflow anaerobic filter was applied to pre-treated leachate for the acclimatization phase (individual organism adjusts to a change in its environment). The obtained best coagulant was employed to R3 leachate (L3), and pre-treated leachate was used in an upflow anaerobic filter for the treatment phase to check the treatment performance. Furthermore, a metagenomics study was conducted to obtain the microbial diversity in raw landfill leachate and anaerobic filter effluent leachate. Lastly, leachate biomethane potential (inoculum study) was attempted to acquire the best ratio of inoculum addition to

treating the landfill leachate. Coagulation-flocculation results indicated that the chemical oxygen demand (COD) removal percentage was higher for $FeCl_3$ with 55.4% at pH 5 for the L1. R3 leachate (L3) was treated before using it for the treatment phase. COD's removal efficiency was 35% in coagulation treatment. While in the upflow anaerobic filter (acclimatization phase), maximum COD removal was 23.03%. 35% of COD was removed in the R3 leachate (L3) during the treatment phase, while in the anaerobic filter, 20.03% of COD removal was achieved. Overall removal was 51.52%, including both the treatment technologies. As a result, it is possible to treat acidic landfill leachate with an anaerobic filter, which is the least alteration treatment method in field applications. Metagenomics analysis showed a high abundance of proteobacteria followed by firmicutes in anaerobic filter effluent leachate. The maximum abundance of proteobacteria and firmicutes is potentially useful in the degradation of mixed solid waste. The maximum removal efficiency of COD of landfill leachate (78%) was obtained in 50% (v/v) of digested cow dung (DCD) digester. Therefore, this microbial seeding was found to have potential low-cost applications in leachate treatment with high COD. This entire research outcome provides useful information for the design and management of landfill leachate for the realistic prediction of future trends.

Keywords: Fresh municipal solid waste; Landfill leachate; Landfill gas; Landfill simulation reactor; Coagulation-flocculation process; Upflow anaerobic filter

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LIST OF ABBREVIATIONS

AAS	Atomic absorption spectroscopy
AF	Anaerobic filter
AnMBR	Anaerobic membrane bioreactor
APHA	American public health association
BIS	Bureau of Indian standards
CO	Carbon monoxide
COD	Chemical oxygen demand
CPCB	Central pollution control board
CPHEEO	Central public health and environmental engineering organization
CW	Constructive wetland
DCD	Digested cow dung
GHGs	Greenhouse gases
HA	Humic acids
HRT	Hydraulic retention time
LFL	Landfill leachate
LSR	Landfill simulation reactor
MBR	Membrane bioreactor
MEF	Micro electrolysis-fenton processes
MSW	Municipal solid waste
NF	Nano-filtration
OLR	Organic loading rate
PAC	Powdered active carbon
RO	Reverse osmosis
SBR	Sequential batch reactor
SWM	Solid waste management
TPD	Tons per day
USEPA	United states environmental protection agency

CHAPTER

1

Introduction

This chapter mainly deals with solid waste management in India, problems associated with landfills/open dumps, life cycle assessment of landfills, the significance of landfill leachate & gases, and perspectives on technologies for landfill leachate treatment.

1.1 INTRODUCTION

India is one of the world's fastest-growing economies. For the first time in more than a century, India's economy has exceeded that of the United Kingdom and the sixth-largest economy by gross domestic product (GDP) and the third-largest economy by purchasing power parity (PPP) (Gramer, 2016). India's urban growth rate rose from 27.8 to 31.6% between 2001 and 2011, and it is projected that up to 50% of the Indian population will live in cities over the next decade (Gupta and Arora, 2016; Sharma and Jain, 2019). According to the 2011 census, India's population was 1,027 million, with 377 million urban residents (approximately 31% of the total) residing in 7,936 towns/cities. However, India's economy gradually transforms from an agricultural one to an industrial and service-oriented focus (Joshi and Ahmed, 2016). India's municipal solid waste (MSW) generation is expected to rise dramatically in the future as the country strives to achieve developed nation status by 2020 (Gidde et al., 2008). Rapid industrialization, population expansion, and economic development in India resulted in the movement of citizens from villages to towns, producing thousands of metric tons of MSW per capita per day due to changing lifestyles and social status (Sharma and Jain, 2019). Municipal solid waste management (MSWM) is highly dependent on the nature of the MSW. Attaining a viable waste management scenario in India is difficult due to the increasing waste composition and varying waste generation rates. Ineffective waste management may have local, regional, or global effects, as shown by climate change and environmental deterioration (Agamuthu et al., 2007). The central pollution control board (CPCB) report indicated that the solid waste production rate lies

between 200 and 300 g/capita/day in small towns/cities with less than 0.2 million populations. It is usually 300-350, 350-400, and 400-600 g/capita/day in cities with population ranges of 200,000-500,000, 500,000-1 million, and above 1 million, respectively (CPCB, 2016; MNRE, 2016). The effect of population development on solid waste production until 2041 is summarized in Table 1.1. In 2001, 31.6 million tons of waste were produced in 366 Indian cities; this figure increased to 47.3 million tons (Mt) in 2011. According to estimates, these 366 cities will generate 161 Mt of MSW in 2041 or about five times as much in four decades. At this pace, gross urban MSW would reach 165 million tons by 2030, 230 million tons by 2041, and 436 million tons by 2050 (Sharma and Jain, 2019).

Table 1.1. Calculated and expected population growth and national waste generation rate

Year	Urban population (million)	Per capita waste generation rate (kg/day)	National waste generation rate (million tons/year)
2001	197.3	0.439	31.63
2011	260.1	0.498	47.3
2021	342.8	0.569	71.15
2031	451.8	0.649	107.01
2036	518.6	0.693	131.24
2041	595.4	0.741	160.96

MSW composition in India is approximately 40-60% compostable, 30-50% inert, and 10-30% recyclable. According to the National Environmental Engineering Research Institute (NEERI), Indian waste consists of 0.64 ± 0.8% nitrogen, 0.67 ± 0.15% phosphorus, and 0.68 ± 0.15% potassium, and has a 26 ± 5 C:N ratio (Gupta et al., 2015; Joshi and Ahmed, 2016). The current municipal solid waste management (MSWM) in India is poor because the best and most appropriate methods from waste collection to disposal are not being used. According to the CPCB, the Indian population has generated 1,52,076.70 t/day of MSW in 2019. The non-sanitary landfills and waste collection efficiency were recorded as 3126 and 95% of total waste generated. About 36.66 % of the collected waste is treated, while the remaining 32.98 % is dumped openly in India. (CPCB, 2019). In India, solid waste management (SWM) determines solid waste management system gaps, such as insufficient waste collection, poor door-to-door collection, and poor waste management. This leads to increasing open dumpsites, and people nearby the dumpsites affected poor wastewater and air quality. The lack of strategic MSW plans, waste

collection/segregation, and a government finance regulatory framework are major barriers to achieving effective SWM in India (Goswami, 2019). Therefore, MSW management is one of the most challenging environmental issues in India. In Indian cities, MSW is disposed of unscientifically in low-lying areas or open dumps without taking proper precautions or operational controls, which has adverse impacts on all components of the environment and human health (Kaushal et al., 2012; Kalyani and Pandey, 2014; Alam and Kulkarni, 2016; Sharma and Jain, 2019).

The mixture of residential and commercial waste in India is predominantly organic waste (70-80%) remaining waste is inorganic (Gupta et al., 2015; Ramachandra et al., 2018). As a result, direct waste disposal in open dumpsites or landfills without pretreatment of MSW has several geo-environmental consequences such as greenhouse gas (GHG's) emissions and organic and inorganic compounds leaching from solid waste pollutes surface and groundwater (Gouveia and Prado, 2010; Patil and Singh, 2017). The total quantity of methane estimated in India was 4612.69 Mt/day due to an open dumping system and deliberated as the third-largest source of greenhouse gases (GHGs) (Singh et al., 2011). Except for a few cities in India, the direct combustion of waste and dumping of garbage in low-lying areas without any treatment practices are common in the Indian scenario (Rana et al., 2019; Rathore et al., 2020). In the future and now, two crucial problems are seen as being responsible for inadequate MSW management: collection, segregation, and recycling, as well as a shortage of treatment options (Babu et al., 2014; Botello-Álvarez et al., 2018). Uncontrolled dumping and open waste burning are standard waste disposal practices in India and other developing countries, contributing to environmental toxicity to human health and aquatic life (Yadav and Samadder, 2017). Inventory levels of CH_4 from landfills are highly uncertain in developed countries such as India due to a lack of data on management and emissions. MSW intended for the landfill undergoes many intermediate phases, including separating recyclable and compostable items. This can alter the amount and composition of waste that eventually reaches landfill sites, affecting GHG's emissions. Measurements of GHGs pollution from landfills are critical for reducing uncertainty in inventory estimates from this source. Life cycle assessment (LCA) is a conceptual model which analyses and verifies the input, performance, and environmental impacts induced by a life cycle of the material. As a result, an LCA must be investigated systematically, define, and identify potential changes in mitigating and managing the consequences (effects) caused by solid waste management activities (Ozeler et al., 2006). Therefore, developed and developing countries are finding an MSW alternative with minimal environmental

effects. Still, the problem can be solved sustainably to handle India's tremendous amount of waste using LCA.

Landfilling is the most usual MSW disposal practice globally, probably because it is the most economical and does not require skilled workers (Li et al., 2014). In India, older landfills do not have a barrier system/liner and leachate collection system to restrict leachate migration into groundwater. Leachate, a liquid origin from MSW, has been seen as a significant pollutant that affects environmental resources such as surface and groundwater, human health, and hygiene. The liquid generated due to biodegradation and precipitation travels downwards under self-weight (Naveen et al., 2017). These landfills are often observed over permeable soils with shallow water tables beneath, enhancing the potential of the leachate to contaminate groundwater. Currently, there are no scientific leachate collection systems in India and other developing countries (Kumar and Alappat, 2005). Landfill leachate (LFL) may contain vast amounts of organic matter (biodegradable and refractory to biodegradation), where humic-type constituents are ammonia-nitrogen, heavy metals, chlorinated organic, and inorganic salts (Renou et al., 2008). Other processes occur during the percolation of leachate in solid waste, including dissolution, sorption, ion exchange, oxidation-reduction, complexation, and metal diffusion. (Morling, 2007). In addition to the leachate, the MSW landfill also produces gas, mainly methane gas (CH_4) and carbon dioxide (CO_2). These gases are classified as greenhouse gas that causes an increase in atmospheric temperature, the so-called global warming effect (Hayati et al., 2018). While most organic substances decay into gases, predominantly methane and carbon dioxide, some refractory organic substances and some organic materials are stabilized in the landfill or leach out through leachate. The refractory organics in leachate are primarily composed of humic substances (HSs), such as humic and fulvic acids, produced through the chemical and biological degradation of organic compounds (Xiaoli et al., 2013). Municipal solid waste (MSW) decomposes in landfills, producing methane (CH_4) and carbon dioxide (CO_2) gases, small amounts of radioactive compounds, and a foul odor as byproducts (Yang et al., 2015). Landfill gas is the third-largest source of anthropogenic methane pollution globally, accounting for about 9-12% of those emissions in 2005 Intergovernmental Panel on Climate Change (IPCC) (Change, 2007). Methane is the second most abundant greenhouse gas and is responsible for 14% of global GHG emissions and, in turn, climate change (GMI, 2011; Lagodimos et al., 2007). In particular, the potential contribution of methane to global warming is 21 times higher than that of carbon dioxide (Gewald et al., 2012).

Several variables may influence the leachate characteristics of a landfill. To continue, the interphase of impact is the sources and composition of the solid waste. After that, a chemical or biological reaction happens, and external factors such as precipitation play a significant role. Due to anaerobic fermentation, young/fresh leachate, i.e., in the acidogenic process, produces a large concentration of volatile fatty acids (VFA). As a result of the aging of leachate and the degradation of organics, the BOD/COD ratio declines, whereas the pH and ammonia concentrations rise. It is one reason why regulatory bodies do not favor leachate treatment with municipal sewage. It is much harder to dispose of sludge from biological treatment. Another reason to justify different treatment is conventional municipal treatment process does not efficiently remove many refractory compounds (Henderson and Atwater, 1995). Many other methods have been investigated for treating leachate generated from municipal landfill wastes. The leachate treatment system's systematic approach may be complex because of the high variability of characteristics and selection methods based on leachate properties (Tchobanoglous et al., 1993). In general, leachate treatment is usually performed and requires a combination of treatment technologies. Most of the processes, which developed for the treatment of water and wastewater has also been adopted for the treatment of landfill leachates includes: (i) biological processes like aerobic, anaerobic, and nitrification/denitrification to organic and nitrogen compounds (Pelkonen et al., 1999; Galvez et al., 2006; Wei et al., 2012; Tałałaj et al., 2019). The physical and chemical process is effective in removing suspended solids, biological parameters (Aziz et al., 2007; Rui et al., 2012; Xu et al., 2017), integrated membrane system with coagulation- nanofiltration (Smol and Włodarczyk-Makuła, 2017), and chemical oxidation process (Derco et al., 2010). The coagulation-flocculation process (CFs) has been widely used as pretreatment and post-treatment to remove leachate parameters such as color, suspended solids, high concentrations of organic pollutants before other biological treatments (Kamaruddin et al., 2017).

Few studies have been carried out based on the addition of sludge and controlled temperature to enhance the degradation of solid waste. Pretreatment method and biological treatment were used to treat stabilized and fresh landfill leachate under a controlled environment. Due to the lack of information on the significant factors and conditions of fresh solid waste degradation in landfills, it is very difficult to identify the dynamics of leachate quality during the active phase of landfills. Thus, it is important to know the leachate characteristics for effective solid waste management in developing and developed countries during the active phase and control their potential impact on the water quality of

surrounding surface and groundwater resources. Careful management of landfills is very important for the reduction of leachate quantity. The most common treatment methods for leachate are biological and physicochemical techniques, which possess differential removal capacities for various pollutants. The effluents must meet the discharge standards or be recycled within landfills without discharge into the environment. The novelty and aim of the research were to assess the landfill leachate and gas dynamics with time in simulated landfills that contained a high amount of wet waste. Multistage treatment technology was adapted for fresh landfill leachate to evaluate the removal efficiency of organic matter. Research is also linked to raw and pretreated leachate microbial community dynamics to understand the different functions for degradation. Therefore, this is the first of its kind study in India that simulated fresh solid waste landfills and treatment of fresh landfill leachate, i.e., coagulation and flocculation, and upflow anaerobic filter under actual landfill conditions.

1.2 THESIS ORGANIZATION

The thesis organization is as follows

Chapter 1 concerns problems associated with landfills/open dumps, life cycle assessment of landfills, treatment of landfill leachate, a summary of introduction.

Chapter 2 gives the detailed literature on solid waste management in India and problems associated with landfill leachate and gases, studies on characterization on leachate from municipal solid waste landfills, life cycle studies on landfills/open dumps in India as well as globally, and the treatment technologies applied to treat the landfill leachate.

Chapter 3 gives the objectives, need of the study, and scope of the thesis

Chapter 4 deals with materials and methods of the various phases of this study; a collection of primary data, secondary data from CPCB; fabrication of landfill simulation reactors; design details of the upflow anaerobic filter (UAF); and detailed procedures of physical, chemical, and biological analyses.

Chapter 5 is about results and discussions of phase-I, includes a life cycle assessment on solid waste management system in Guwahati; methodology to assess the environmental impact across various scenarios for waste management.

Chapter 6 gives the results and discussions of phase-II, including part-I (simulated without rainfall) & Part-II (simulated with rainfall), variation of physical, chemical, and biological properties of landfill leachate, and landfill gas composition with time.

Chapter 7 is about results and discussions of phase-III, including removal efficiency of pretreatment and upflow anaerobic filter treatment of fresh landfill leachate.

Chapter 8 lists the overall conclusions of the research work and future scope from this study.

CHAPTER

2

Literature review

This chapter deals with the available relevant literature concerning the problems associated with solid waste management in India, factors affecting the biodegradation process of solid waste in landfills, characterization of landfill leachate and gas, life cycle studies on landfills/open dumps in India, as well as globally, statistical modeling for predicting leachate parameters and lastly application of landfill leachate treatment technologies are also discussed.

2.1 SCENARIO OF SOLID WASTE MANAGEMENT IN INDIA

2.1.1 Functional elements of municipal solid waste

Solid waste is the unwanted or useless solid materials generated from human activities in residential, industrial, or commercial areas. The activities associated with the management of municipal solid waste from generation to final disposal can be grouped into the six functional elements as shown in Fig. 2.1 (CPHEEO, 2000; Gupta et al., 2015).

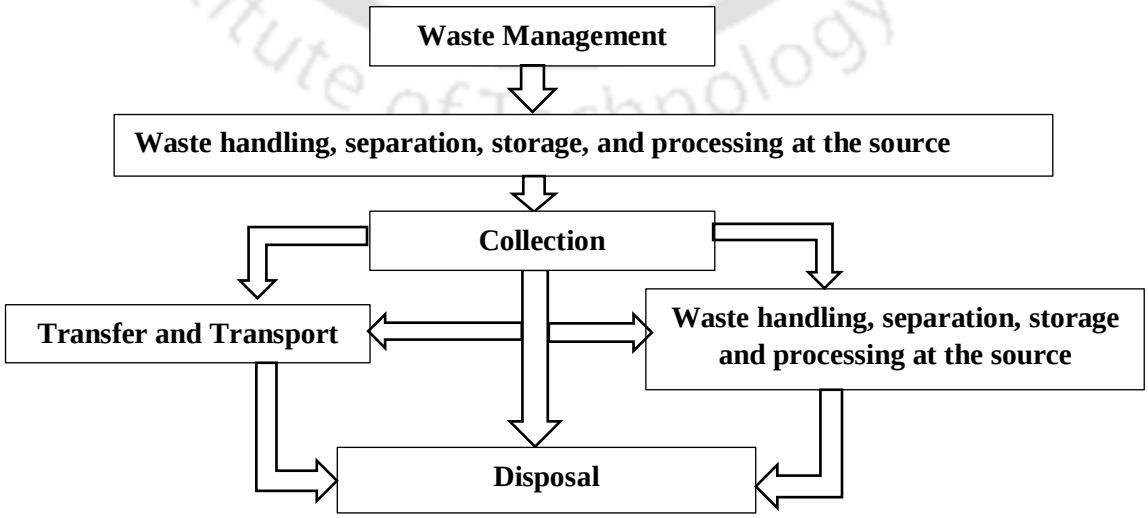


Fig. 2.1. Functional elements of solid waste management

- **Waste generation**

Waste generation encompasses activities in which materials are identified as no longer being of value (in their present form) and are thrown either away or gathered together for disposal. Although not controlled by solid waste managers, reducing waste at the source is now included in system evaluations to limit the quantity of waste generated.

- **Waste handling, sorting, storage, and processing at the source**

The second of the six functional elements in the solid waste management system is waste handling, sorting, storage, and processing. Waste handling and sorting involve the activities associated with managing waste until they are placed in storage containers for collection. Sorting waste components is an important step in handling and storing solid waste at the source. For example, the best place to separate waste materials for reuse and recycling is at the source of generation. Households are becoming more aware of the importance of separating newspaper and cardboard, bottles/glass, kitchen wastes, and ferrous and non-ferrous materials.

- **On-site storage**

It is critical for public health and aesthetic reasons. Unpleasant improvised containers and even open ground storage, both unacceptable, are often seen in various residential and business locations. Processing at the source includes activities such as composting yard waste.

- **Collection**

The functional element of the collection includes not only the gathering of solid wastes and recyclable materials but also the transport of these materials, after collection, to the location where the collection vehicle is emptied. This location may be a materials processing facility, a transfer station, or a landfill disposal site.

- **Sorting, processing, and transformation of solid waste**

The sorting, processing, and transformation of solid waste materials are the fourth functional element. This functional element encompasses the recovery of sorted materials, solid waste processing, and solid waste transformation that occurs primarily in locations away from the source of waste generation. For example, sorting commingled (mixed) wastes usually occurs at a materials recovery facility, transfer stations, combustion facilities, and disposal sites. Sorting often includes separating bulky items, separating waste components by size using screens, manually separating waste components, and separating ferrous and non-ferrous metals.

- **Waste processing**

It is undertaken to recover conversion products and energy. Various biological and thermal processes can transform the organic fraction of municipal solid waste (MSW).

- **Waste transformation**

It is undertaken to reduce waste's volume, weight, size, or toxicity without resource recovery. A variety of mechanisms may do transformation (e.g., shredding), thermal (e.g., incineration without energy recovery), or chemical (e.g., encapsulation) techniques.

- **Transfer and transport**

The functional element of transfer and transport involves two steps: (i) the transfer of wastes from the smaller collection vehicle to the larger transport equipment and (ii) the subsequent transport of the wastes to a processing or disposal site.

- **Disposal**

The final functional element in the solid waste management system is disposal. Today the disposal of wastes by landfilling or uncontrolled dumping is the ultimate fate of all solid wastes, whether residential wastes collected and transported directly to a landfill site, residual materials from Materials.

- **Recovery facilities (MRFs)**

Residue from solid waste combustion rejects of composting or other substances from various solid waste-processing facilities. A municipal solid waste landfill plant is an engineered facility used to dispose of solid wastes on land or within the earth's mantle without creating nuisance or hazard to public health or safety, such as breeding rodents, insects, and groundwater contamination.

2.1.2 Current status of SWM in India

Municipal solid waste management (MSWM) methods primarily address health, environmental, aesthetic, land use, resource, and economic challenges associated with inappropriate waste disposal (Ferronato et al., 2018). MSW is disposed of in an unscientific manner in low-lying regions or open dumps in Indian cities without enough measures or operational controls, resulting in negative effects on all environmental and human health components (Kaushal et al., 2012; Kalyani and Pandey, 2014; Gupta et al., 2015; Alam and Kulkarni, 2016). MSWM is governed in India under the Solid Waste Management Rules, 2016. According to these laws, all municipal authorities or urban local bodies (ULBs) are responsible for the generation, on-site storage, collection, transportation, processing, and treatment of MSW in an environmentally sustainable manner while considering economic,

energy, aesthetic, and preservation standards (Sastry, 2015; Mani and Singh, 2016). However, in most Indian cities, the MSWM system consists of only four activities, i.e., waste generation, collection, transportation, and disposal (Kaushal et al., 2012). Inadequate collection and transportation contribute to the buildup of MSW in every nook and cranny of cities (Sharholy et al., 2008; Gupta et al., 2015). Statewise MSW statistics in India are illustrated in Table 2.1.

Table 2.1. Current status of state-wise MSW statistics in India (CPCB, 2019)

Solid waste generation status (2018-2019)						
S.No.	Indian States	Solid waste generation (TPD)	Collected (TPD)	Collection efficiency (%)	Treated (TPD)	Landfilled (TPD)
1	Andaman Nicobar	120	117	99.975	65.1	37.9
2	Andhra Pradesh	6440	6140	95.34	548	203
3	Arunachal Pradesh	270.96	215	79.34	-	-
4	Assam	1293.63	1119.37	86.52	-	-
5	Bihar	2272	Yes		-	No
6	Chandigarh	470	458.52	97.55	150	361.28
7	Chhattisgarh	1650	1386	84	1271	115
8	Daman Diu	98	94.5	96.42	5	89.5
9	Delhi	10817	10614	98.12	5714	5225
10	Goa	236.41	235.9	99.9	154.71	1.49
11	Gujarat	-	11,119		1,127	9,992
12	Haryana	4635.79	4430.25	95.56	815.93	3614.32
13	Himachal Pradesh	389	340	87.4	150	190
14	Jammu & Kashmir	1530.53	1452.86	94.92	-	-
15	Jharkhand	2205	2043.4	92.67	836.69	0

16	Karnataka	11958	10011	83.71	4515	-
17	Kerala	3903.023	742.23	38.03	437.74	-
18	Nagaland	339.5	216.9	63.88	135.8	33.95
19	Lakshadweep	35	18	51.42	18	-
20	Madhya Pradesh	8000	7500	93.75	6100	1400
21	Maharashtra	23844.55	23675.7	99.29	12623.33	11052.37
22	Manipur	218.6	126.63	57.92	80	46.63
23	Mizoram	251.42	213.07	84.99	29.22	-
24	Meghalaya	170.63	170.63	100	8.72	161.91
25	Orissa	2564.43	2255.32	87.94	91.63	2163.69
26	Punjab	4634.48	4574.93	98.71	917.56	3657.37
27	Pondicherry	599.25	505	84.27	24	481
28	Rajasthan	6625.56	6475.39	97.73	780.18	4187.16
29	Sikkim	75.1	67.1	89.34	13.05	51.4
30	Tamil Nadu	13968	12850	91.99	7196	5654
31	Telangana	8497	8360	98.38	5747	869
32	Tripura	445.72	389.46	87.37	150.1	239.36
33	Uttarakhand	1527.458	1437.4	94.1	524	-
34	Uttar Pradesh	17377.3	17329.4	99.72	4615	0
35	West Bengal	14613.3	13064.63	89.4	916	334
	Total	152076.70	149748.60	99.99	55759.60 (36.66%)	50161.33 (32.98%)

MSW collection efficiency for Indian states is shown in Table 2.1, based on the data provided by CPCB for 2018-2019. Most Indian states are unable to collect the total quantity of generated waste. The waste collection efficiency in India ranges between 50 and 100%. The treated quantity of generated waste was only 36.66% in 2018-2019, and in point of fact, nothing is scientifically managed in sanitary landfills. In 2019, Maharashtra generated the largest quantity of MSW (23844.55 t/day) among all states. The waste collection efficiency was 82.84% in India. Low collection efficiency was reported in Kerala is 38%, and generated waste was 3903.023 t/day. Metropolitan cities reported around 100% collection efficiency.

2.1.3 Comparative view of MSWM in developed and developing countries

Countries are grouped based on their gross national income, including high-income countries (HICs), e.g., France, Germany, Spain, the United Kingdom, the United States, and Italy; upper-middle-income countries (UMICs), e.g., Brazil, Mexico, Poland, Lithuania, and Latvia; lower-middle-income countries (LMICs), e.g., Algeria, China, India, Jordan, Turkey, Bulgaria, and Thailand; and lower-income countries (LICs), e.g., Bangladesh, Ghana, Nepal, Pakistan, Nigeria, Uganda, and Zimbabwe. In HICs, most of the waste is collected and treated scientifically, whereas LICs are still working on open dumping. The concept of the Five Rs approach is effectively adopted in HICs, whereas LICs and LMICs are still fighting for the 100% collection efficiency of MSW. HICs focus more on reducing and recycling instead of landfilling and incineration of MSW, whereas open dumping and sanitary landfilling are the preferable options in LICs and LMICs. The total percentage of recycling and composting is highest in Germany (66.2%), followed by Nepal (55%), Lithuania (48%), Italy (45.1%), the United Kingdom (44.5%), Poland (44%), France (41.7%), the United States (34.6%), Bulgaria (31.8%), and Spain (29.7%). After investing millions of dollars in MSWM, Indian performance is still very poor among LMIC, UMIC, and HIC countries. The waste collection efficiency of India (82%) is the lowest in the LMIC group, behind that of China (90%), Turkey (84%), Thailand (100%), Bulgaria (94%), Jordan (90%), and Algeria (92%). The quantity of waste treated in India is only 36.66%, whereas 92% in China, 100% in Turkey and Bulgaria, 55% in Jordan, and 46% in Thailand. Algeria, Pakistan, India, and Thailand have higher percentages of open and illegal waste dumping: 96, 90, 72, and 54%, respectively. As a large waste-generating country, India needs to focus its MSWM heavily on high waste recycling, composting, and anaerobic digestion to reduce the load on waste disposal through incineration and sanitary landfilling. (Sharma and Jain, 2019).

2.2 LIFE CYCLE ASSESSMENT (LCA) APPROACH

2.2.1 Conceptual framework

Between 1960 and 2000, the global ecological footprint of humans, which takes into account human resource usage, increased by 80%. At present, the globe consumes 1.2 times more resources than it can replenish (Reap et al., 2008). Increases in the price of raw materials and natural substances indicate the scarcity of natural resources. As public awareness of environmental issues grows, industries and corporations evaluate how their operations impact the environment (Jorgensen et al., 2008). LCA is one such tool. A

"cradle-to-grave" method to examine industrial systems is known as LCA. The term "cradle-to-grave" refers to the process of collecting raw materials from the land to make a product and ending with the restoration of all elements to the earth (Ayres, 1995; Jorgensen et al., 2008; Cherubini et al., 2009; Lima et al., 2018). LCA considers all phases of a product's life cycle from the viewpoint of interdependence, i.e., one operation leads to the next. LCA allows for assessing cumulative environmental effects from all phases of a product's life cycle, including those typically neglected in more conventional evaluations (e.g., raw material extraction, material transportation, final product disposal, and so on). LCA gives a more accurate picture of the underlying environmental trade-offs in product and process choices by providing a complete perspective of the environmental elements of the product or process (Hochschorner and Finnveden, 2003). The first series of standards appeared between 1997 and 2000 (ISO 14040-ISO 14043) (Standardization, 2006). In 2006, after these standards had been revised, content changed minute, but the standard specifications were restructured. International standard organization (ISO) 14040 (1997) (Kruse et al., 2009), ISO 14041 (1998), ISO 14042 (2000) (Sander and Murthy, 2010), and ISO 14043 (2000) (Nguyen and Gheewala, 2008) were divided into the classification: ISO 14040 (Pehnt, 2006) and ISO 14044 (Roy et al., 2009) that is used currently. Therefore, LCA is the primary globally defined approach for analyzing and evaluating environmental factors and their potential implications. The basic framework and the iterations involved in LCA are depicted in Fig. 2.2. The LCA process is a systematic, phased approach and consists of four components: goal definition and scoping, inventory analysis, impact assessment, and interpretation (Khandelwal et al., 2019).

(i) Scope and goal

The scope determines the product system and the setting and boundaries in which the study is evaluating. Specifically, the scope includes the product being studied with a definitive functional unit, the system boundary, methodological choices, and analysis details. The goal of the study guides the selection of the appropriate impact categories, and the number of categories analyzed is limited by the practicality and depth of the study (Pieragostini et al., 2012). The entire life cycle can be examined using the system boundary shown in Fig. 2.3, which covers product development, manufacture, usage, and end-of-life stages. The scope, on the other hand, may be adjusted to fit the study's objectives.

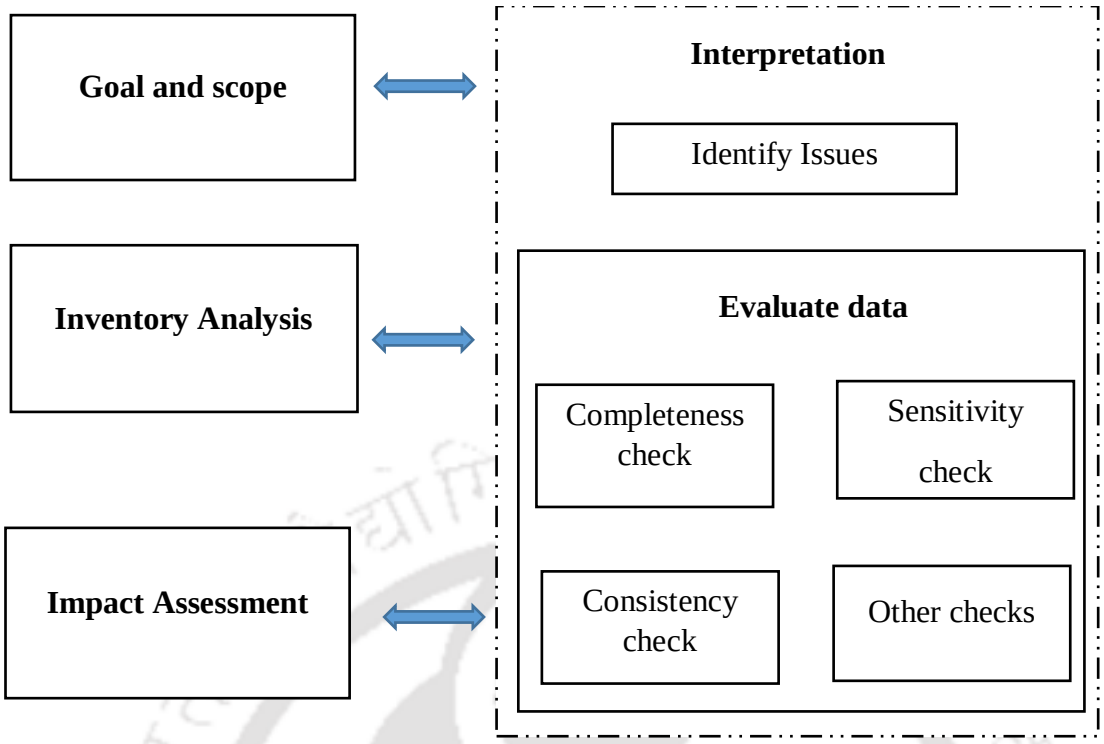


Fig. 2.2. Process-based LCA framework and steps

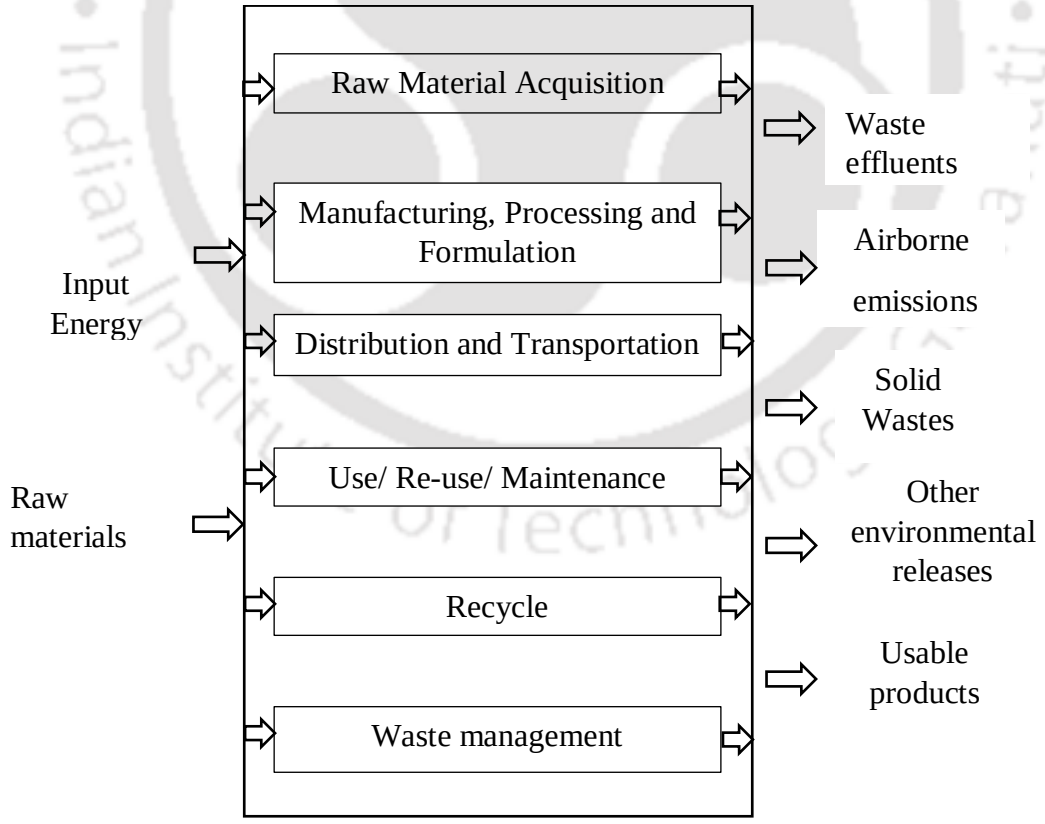


Fig. 2.3. Life cycle assessment full system boundary

(ii) Life cycle inventory analysis

A life-cycle inventory is a set of data and material and energy flow calculations that quantify a product life-cycle's inputs and outputs. Estimating material and energy inputs, wastes, and emissions data caused by a functional product unit defined in the goal and scope are all part of this stage. The data is particular to the study's system boundaries and takes into account the system's environment. After data collection, the calculation of the emissions relative to the considered functional unit is recorded (Pieragostini et al., 2012).

(iii) Life cycle impact assessment (LCIA)

The study of possible human health and environmental implications of environmental resources and emissions discovered during the LCI phase of an LCA is known as the Life Cycle Impact Assessment (LCIA). Environmental and human health implications and resource depletion should be included in an impact assessment (Rana et al., 2019). A life cycle impact assessment attempts to establish a linkage between the product or process and its potential environmental impacts.

(iv) Life cycle interpretation (LCI)

Life cycle interpretation is a systematic technique to identify, quantify, check, and evaluate information from the LCI and LCIA results and communicate them effectively. Life cycle interpretation is the last phase of the LCA process. ISO has defined the following objectives of life cycle interpretation: (a) Analyze results, reach conclusions, explain limitations, provide recommendations based on the findings of the preceding phases of the LCA, and report the life cycle interpretation results transparently. (b) Provide a readily understandable, complete, and consistent presentation of the results of an LCA study, according to the study's goal and scope (Organization, 1997).

2.2.2 Impact indicators

A typical LCA addresses between five and nine impact categories. Most commonly, acidification potential, aquatic eutrophication potential, global warming potential (embodied carbon), human health particulate, ozone depletion. These impact categories are shown in Table 2.2 with their units of measure. To fully understand the environmental impacts of a product or process, all impact categories should be addressed; however, the indicators studied are ultimately dependent on the goal and scope of the LCA. The relationship between all impact indicators is complex, and many interconnections (Standardization, 2006). Because emissions can cause environmental impacts to multiple indicators, environmental indicators are classified as midpoint indicators that translate

impacts into environmental themes. The ISO methodology aggregates the midpoint indicators into three areas of endpoint damage: (1) human health damage, (2) natural resource consumption, and (3) ecosystem damage (Pieragostini et al., 2012). All midpoint indicators directly connect to an endpoint impact and have secondary impacts on others. The relationship flow chart, Fig. 2.4, shows these connections (Pieragostini et al., 2012).

Table 2.2. Life cycle assessment impact category indicators

Midpoint LCIA Indicators	Units
Global Warming Potential	kg-CO_2-eq
Ozone Depletion (air) potential	kg CFC-11-eq
Acidification Potential	kg-SO_2-eq
Eutrophication Potential	kg-N-eq
Smog (air) Potential	kg-O_3-eq

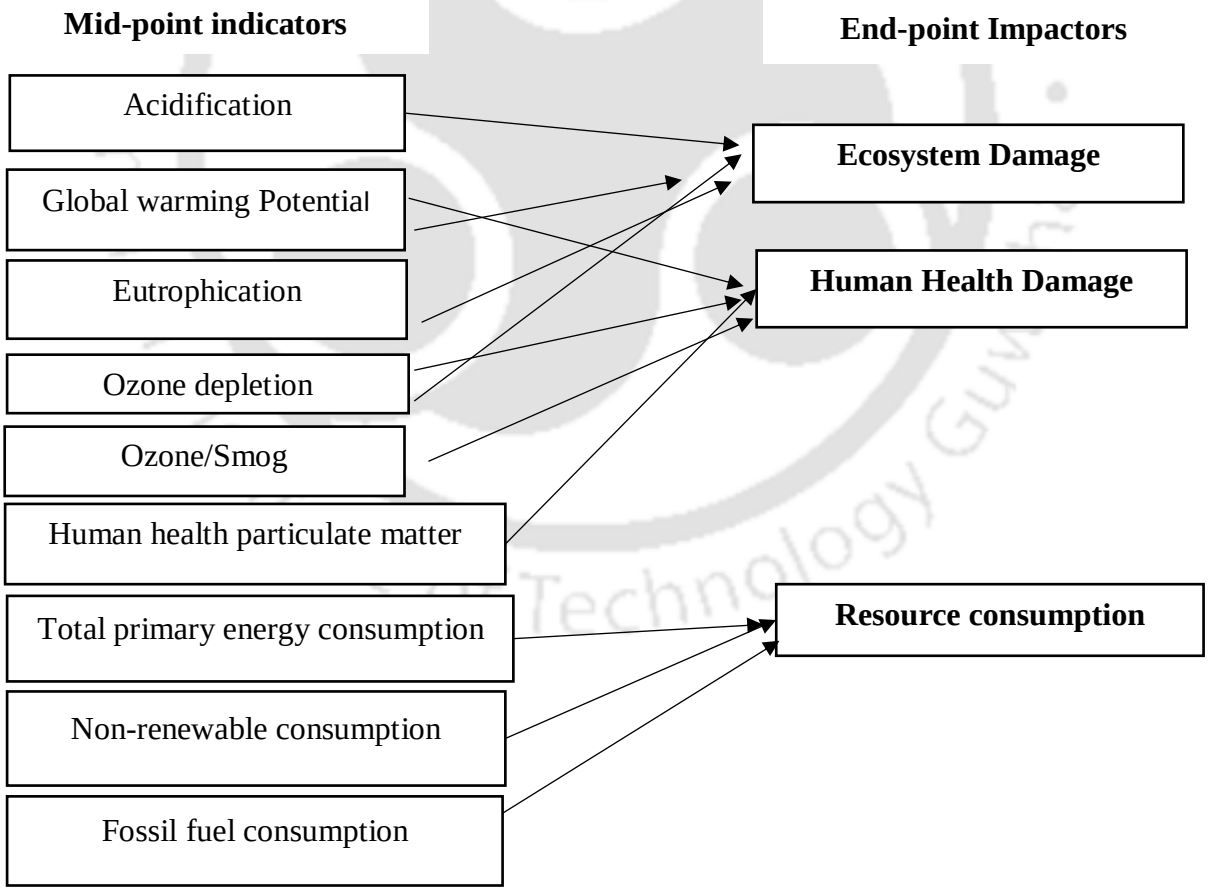


Fig. 2.4. LCA midpoint and endpoint relationships

2.2.3 Ecosystem and human health damage indicators

2.2.3.1 Global warming potential (GWP)

Global warming potential (GWP), also commonly known as "Embodied Carbon," measures the quantity of greenhouse gas emissions that could become trapped in the atmosphere. GWP is calculated by the amount of energy the emissions of one ton of a gas will absorb over a given period relative to the emissions of one ton of carbon dioxide (CO_2) (Talmage, 2018). Greenhouse gasses impart many different impacts to the earth, including changes in precipitation, sea-level rise, ocean currents, storms, hurricanes, and possible additional impacts on human health and biotic natural resources (SETAC, 2017). By combining these impacts, GWP provides a simple midpoint indicator representation of the relative radiative forcing resulting from a unit mass emission of greenhouse gas (Earth, 1995).

2.2.3.2 Ozone depletion (air) potential (ODP)

The ozone high in the earth's stratosphere protects its surface from the damage caused by the sun's ultraviolet (UV) rays. Chlorofluorocarbons (CFCs) are released into the atmosphere through (1) the use of aerosols containing them, (2) refrigeration equipment, or (3) the industry producing and using refrigeration equipment. When CFCs are released into the atmosphere, the sunlight causes halocarbons to release chlorine atoms, which break down the ozone molecules (Talmage, 2018). When the ozone in the earth's stratosphere is broken down, the ozone layer allows UV-B radiation to reach the surface. Ozone depletion potential (ODP) measures the expected impact on the ozone per unit mass emission of gas compared to expected from the same mass emission of CFC-11 integrated over time (Earth, 1995). Thus, ODP is commonly used as a comparison of relative impacts of different gases upon the ozone. When the sun's UV rays pass the depleted ozone layer, the UV rays can cause significant damage, including skin cancer, eye damage, reduction of plankton, and crop loss (Organization, 1997).

2.2.3.3 Acidification potential (AP)

Acidification potential measures the disposition of a unit of mass of a component to release hydrogen ions (H^+) to a receiving medium. The addition of hydrogen ions alters the pH of that medium and causes undesirable effects (SETAC, 2017). When water and soil pH balance is changed, many consequences affect plants and animals that thrive on a specific pH level. Additionally, a change in pH can reduce soil productivity, which can cause a supplementary decline in plant and animal health. The change in plant and animal health can also impact the liveliness of these species and reduce their potential as food

sources for humans (Organization, 1997). Acidification-causing substances are sulfur dioxide and nitrogen oxide that are transformed into either sulfuric acid or nitric acid through oxidization and photochemical reactions that eventually deposit on the earth's surface (SETAC, 2017).

2.2.3.4 Eutrophication potential (EP)

Eutrophication is the surplus of biological activity in an aquatic system that stems from adding nutrients to the system. Eutrophication can inhibit the water system by depleting the available oxygen needed for species and threatening biodiversity. Eutrophication occurs when nitrogen and phosphorus are added to an aquatic system. Typically, this occurs from septic field seepage, storm and wastewater runoff, fossil fuel combustion, and rainwater runoff after contact with fertilized agriculture, manure, and aquaculture, all of which contain nitrogen and phosphorus (SETAC, 2017).

2.2.3.5 Smog (air) potential (POCP)

Smog is the formation of ground-level ozone, which poses significant threats to air quality. The formation of smog is caused by the release of natural and manmade substances into the atmosphere, volatile organic substances (VOCs), and oxides of nitrogen, which react to sunlight. VOCs are emitted from building materials and building maintenance products, while oxides of nitrogen are emissions related to fossil fuel consumption. Ground-level ozone and airborne particles are created when the sunlight reacts with these emissions. Ground-level ozone formation is reported in the weight of ozone (O_3). Ground-level ozone poses a significant threat to human health respiratory systems, including reduced lung function, aggravation asthma, and permanent lung damage (SETAC, 2017).

2.2.4 Limitations of LCA

The adoption of LCA has provided a way to quantify measures of environmental impacts, helped create a way to make comparative assertions, and made a method in which people can incorporate sustainability into design. LCA has many strengths, but there are also limitations. The data and assumptions that LCAs require add a significant amount of uncertainty, which can create gaps and errors in the outcome of the LCA results (Organization, 1997). LCA does not address occupational exposure limits, indoor air quality or other legal emission limits, resource management, or social equity. These other factors are important in decision making and for complete understanding and evaluating the sustainability of a system or process (O Connor et al., 2012).

2.3 LIFE CYCLE COST ANALYSIS

The economic accounting of costs associated with a product or process throughout its full life cycle, known as life cycle cost analysis (LCCA), is the standardized method used by many organizations, including the US transportation department. LCCA estimates the economic costs associated with all phases of a product or process, including initial, operating, and refurbishment/replacement costs. In addition, LCCA includes the total discounted dollar cost of a product or process over a specified life cycle (Mearig et al., 1999). The goal of any LCCA is typically to reduce the economic cost of any product or process and identify high-cost contributors and alternatives that can decrease the overall cost. This method is often used with LCA to establish and compare the economic and environmental cost of a product or process for enhanced decision-making (Durairaj, 2002).

2.3.1 Optimization and trade-off of impact categories and economic cost

LCA has the potential to be used in many ways. In its simplest form, LCA can be used to quantify and evaluate the environmental performance of a product or Process. LCA can also act as part of a much broader decision process and can understand general trade-offs of environmental impacts and select among alternative products or processes (Organization, 1997). The method provides the ability to assess trade-offs between impacts and, by employing optimization techniques, clarify and help designers make complex decisions. Additionally, LCA provides a basis for assessing potential improvements in the environmental performance of a product system (Pieragostini et al., 2012).

Similarly, LCCA has the potential to be used to quantify and evaluate the economic performance of a product or process. LCCA can be used to make decisions on alternatives and understand the broad or general trade-offs of materials of Processes involved in the system (Rebitzer et al., 2004). Economic optimization tools are often used for LCCA to make these decisions and address improvements on the overall economic cost (Standardization, 2006). The value of the information developed from both LCAs and LCCAs can be heightened inapplicability in different contexts if they are both used not only on their own but also simultaneously as part of a comprehensive decision-making process that weighs economic and environmental tradeoffs against each other Although LCA and LCCA can provide data and results that are presented with options that have the lowest impacts or smallest economic costs, trade-offs between economic growth and environmental protection often occur. More specifically, establishing a functionally equivalent product or process with lower environmental impacts can add additional

economical costs and vice-versa (Den Butter and Verbruggen, 1994). In comparative LCAs, where functionally equivalent products are being directly compared, optimization techniques can be used to determine the tradeoffs between the use of different materials. Depending on the goal and scope of the study, the LCA and LCCA can be used together to select which product or process has the lowest desired costs (environmental and economic) (Organization, 1997), simplistically shown in Fig. 2.5.

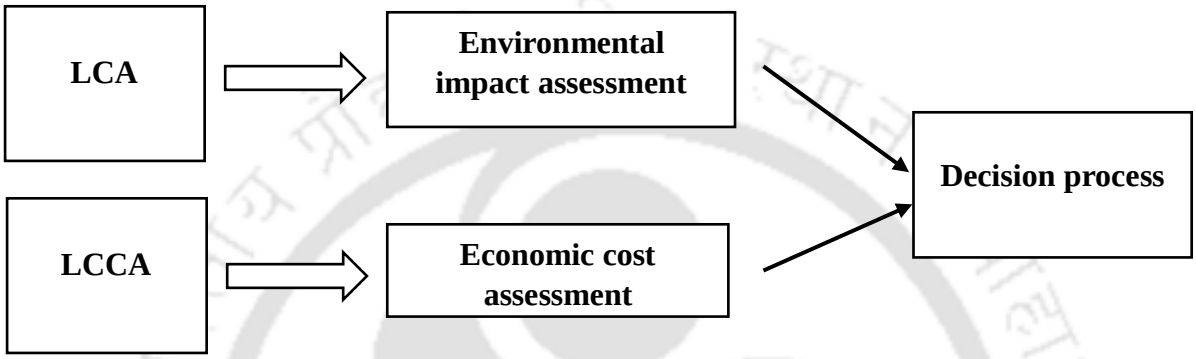


Fig. 2.5. LCA and LCCA decision process

2.4 APPLICATION OF LCA IN SOLID WASTE MANAGEMENT

The application of LCA in MSW management is a very challenging task due to the following reasons:

Every single waste management facility is considered environmentally friendly. However, solid waste management facilities require land (a lot of land in the case of landfills), consume nonrenewable natural resources for their operation (e.g., fuels and electricity), and emit a sequence of air pollutants and leachates. Therefore, waste management facilities put an environmental burden of their own on the natural environment. The trade-offs between environmental gains and burdens have to be assessed. On the other hand, solid waste management facilities generate many useful "products"; material reclamation facilities produce different sorts of paper and cardboard, glass, plastics, etc. A mechanical biological treatment facility generates refuse-derived fuel (RDF), which can be used as a solid fuel in cement kilns, and compost, which can be used as a fertilizer substitute. Thermal treatment facilities, the so-called waste-to-energy, produce electricity and heat. Therefore, solid waste management facilities have to be credited for all those useful "products." There is a lot of uncertainty in a lot of the major solid waste treatment processes. The lack of quality data with respect to waste management

practices is a recognized problem of LCA (McDougall et al., 2008). Landfilling, the most widely used MSW management option, has many uncertainties related to the time frame of the impacts. Data relating to processes with direct measurements (such as collection, recycling, and treatment) are more reliable than data from landfills which partially have to be modeled and where estimations are necessary (Obersteiner et al., 2007).

2.4.1 Life cycle of municipal solid waste (MSW)

Fig. 2.6 represents the MSW life cycle. The LCA system border interfaces connect the waste management system, environment, and other product systems. When a product or product becomes waste, its user discards it in the garbage collecting containers, the life cycle begins. MSW is collected in two ways: in mixed bags or separately. Infrastructure, like specific bins and collection trucks, is required for each collection method. Following that is the step of transportation. The mixed bag garbage may be disposed of in a landfill, a waste-to-energy facility, or a mechanical biological treatment plant in industrialized nations' MSW management systems (MBT). If it's a dry stream (paper and cardboard, plastics, glass, tin, aluminum, and so on), the source-separated waste may proceed to the material reclamation plant; if it's a wet stream (kitchen leftovers, garden trimmings, and so on), it may go straight to the biological treatment facility. Next, LCA assesses the use of resources and the release of emissions to air, water, land, and the generation of useful products. All these inputs (material and energy resources) and outputs (emissions and products) have to be identified and quantified during the life cycle inventory (LCI) phase of the LCA. In the following sections, the most important LCI components of each management stage are identified and presented. Inputs from natural resources and output emissions are identified in red, while the useful products are green. The functional unit (FU) is the reference to which the inputs and outputs are related (Organization, 1997).

(i) Collection and transport

The collection of MSW can either be in mixed bags or separate bins. Mixed bag collection is the most widely applied method; however, the separate collection is a prerequisite for successful material recovery. Fig. 2.7 presents the inputs and outputs to the collection and transport stages of MSW management. The inputs are MSW and the materials and energy for the required infrastructure (MSW temporary storage containers and vehicles needed for collection and transportation). The outputs of these processes are again MSW (with altered physical properties such as density) and air, water, and solid emissions.

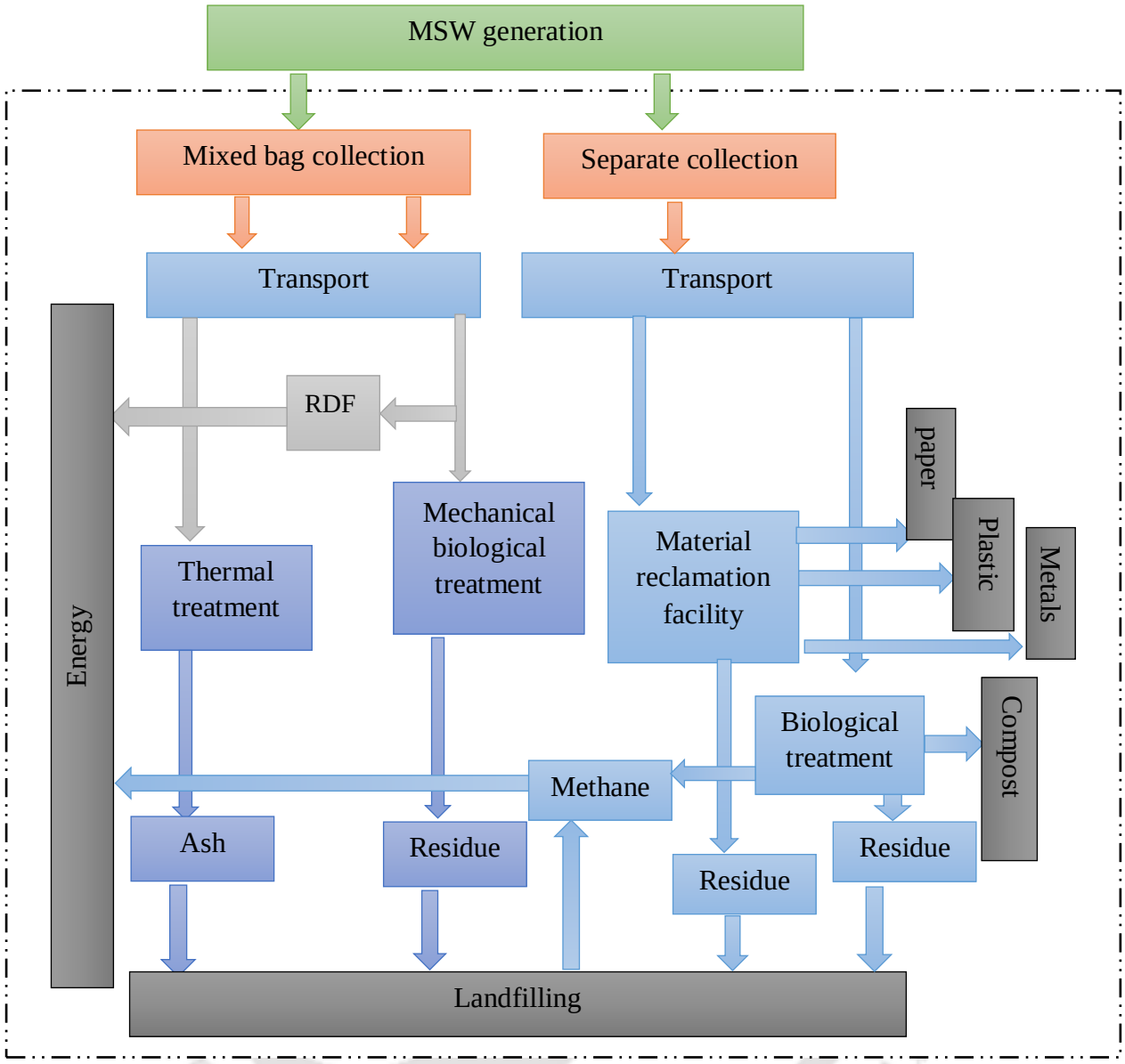


Fig. 2.6. The complete life cycle of MSW

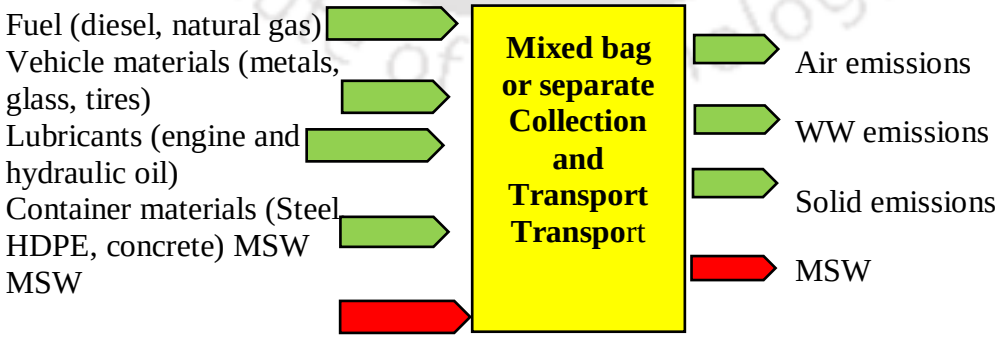


Fig. 2.7. Life cycle inventory components for the collection and transportation stages

The following parameters must be taken into account for the compilation of an effective LCI

in the collection and transportation stages of an LCA:

- Selective collection system,
- Material of containers (HDPE, steel, and fiberglass),
- Collection frequency,
- Distance covered,
- Type of collection truck (pneumatic, top loader, rear loader, side loader),
- The fuel of collection truck (diesel, natural gas),
- The density of the waste fractions in containers and collection trucks,
- Size of containers,
- Filling percentage of the waste containers

(ii) Mechanical and biological treatment

Mechanical and biological treatment is a process that generates many useful "products" illustrated in Fig. 2.8. Its inputs include mixed-bag MSW, electricity, fuels (e.g., diesel and natural gas), water, and materials for the required infrastructure (e.g., lubricants and strapping). The outputs are recovered metals (Fe and Al), RDF (which ultimately can be used as an energy source), compost (which can substitute chemical fertilizers), emissions to air and water, and finally, a fraction of residue that ends up in the landfill (Tarantini et al., 2009).

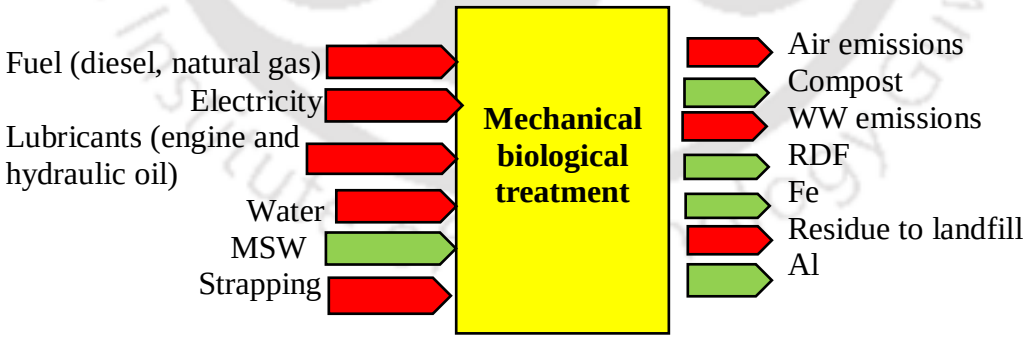


Fig. 2.8. Life cycle inventory components for the MBT plant

(iii) Thermal treatment

The major inputs and outputs considered when compiling the LCI of an incineration plant are the following Fig. 2.9. The major inputs are MSW, electricity, other fuels (diesel, natural gas, or even coal), water, and activated carbon (for air pollution control). On the

other hand, the outputs are flue gas (HCl, SO_2, NOx, dioxins, CO, PM_{10}, HF), bottom ash, iron scrap, electricity generated, water discharge, and air pollution control residues (Khandelwal et al., 2019).

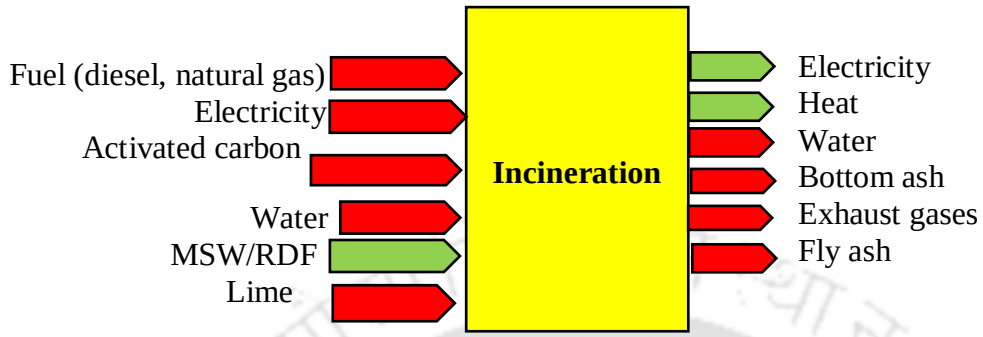


Fig. 2.9. Life cycle inventory components for an incineration plant

(iv) Biological treatment

Fig. 2.10 presents the major inputs and outputs for the life cycle inventory of MSW biological treatment. There are two processes included under the term "biological treatment" in MSW management: composting and anaerobic digestion. The biodegradable fraction of the MSW is involved in both of the processes. Composting is an anaerobic process, and degradable organic carbon convert in the MSW is converted into CO_2 (Obersteiner et al., 2007).

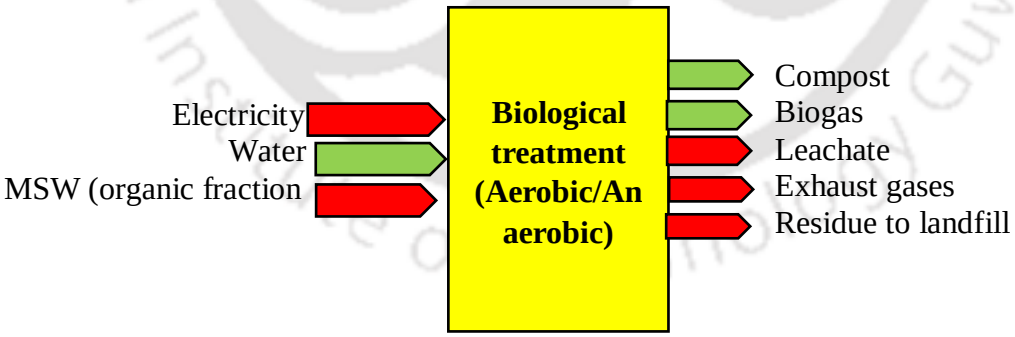


Fig. 2.10. Life cycle inventory components for biological treatment

(v) Landfilling

Landfilling is the first and oldest MSW treatment option. The types of landfilling facilities worldwide range from uncontrolled dumpsites to highly engineered facilities with leachate and landfill gas (LFG) management. Fig. 2.11 presents the major inputs and outputs for the life cycle inventory of landfilling. When MSW is landfilled directly,

anaerobic biological degradation produces landfill gas and leachate. As a result, over 90% of the converted organic carbon is released as CO_2 and CH_4. The remainder is released in the leachate (Obersteiner et al., 2007).

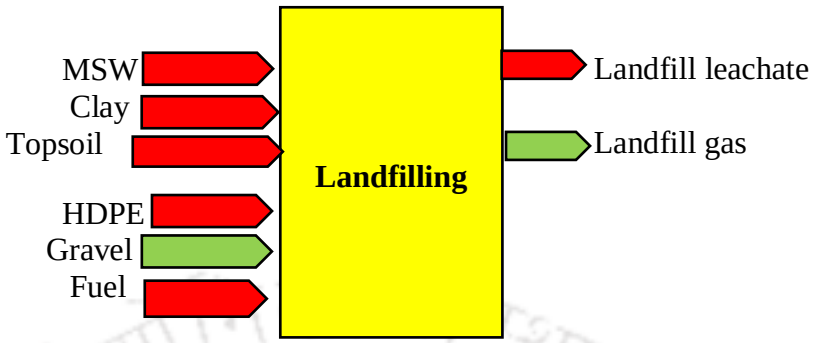


Fig. 2.11. Life cycle inventory components for landfilling

2.5 LIFE CYCLE ASSESSMENT STUDIES

The studies were classified into four categories (Bank, 2016), including (1) low income (LI) countries (GDP $1005), (2) lower middle income (LMI) countries ($1006 GDP $3955), (3) upper middle income (UMI) countries ($3956 GDP $12,235), and (4) higher income (HI) countries (GDP $12,236). The HI category has the most nations that have done LCA, followed by LMI and UMI nations currently being studied. The present research attempts to determine the global distribution of LCA on MSWM. The research systematically analyses the methodology and main conclusions of LCA research, taking into account the gaps and inclusion of social and economic LCA in the chosen research. LCA studies on construction and demolition waste, hazardous and e-waste, and sewage sludge management were primarily focused on LCA studies related to the MSWM system, which covered MSW generated from residential and commercial (non-residential) areas, excluding LCA studies on construction and demolition waste, hazardous and e-waste, and sewage sludge management.

Table 2.3. A detailed review of LCA research studies on waste management technologies in different countries

Income Group, Country, Reference	Functional unit (FU), model & method	Scenarios	Critical findings
Sao paul, Brazil, UMI, (Mendes et al., 2004)	FU: Treatment & disposal of 1 ton of waste Model: NM Method: NM	i) LF, ii) LFGTE iii) COM iv) composting followed by gas treatment (compost with bio filter), and v) bio gasification.	Landfilling was the scenario with the highest environmental impacts, except in the case of acidification potential, in which composting presented the highest potential. In addition, composting without gas treatment presented higher environmental impacts than bio gasification. Finally, both composting and bio gasification can decrease the impacts significantly compared to landfilling.
Pudong, China, UMI, (Hong et al., 2006)	FU: Treatment of 2200 t/day of MSW Model: NM Method: NM	i) LF ii) INC iii) MBT + COM, iv) BMT + INC and v)	LCA results show that the incineration process of MSW presents the highest

		BMT + LF	acidification potential while the landfill presents both the highest global warming and eutrophication potential.
Ankara, Turkey, UMI, (Ozeler et al., 2006)	FU: Amount of municipal solid waste generated Model: IWM-1 Method: NM	S1: COLL + TRANS+ LF S2: SR + COLL + TRANS + LF S3: COLL+ MRF TRANS + LF, S4: COLL + MRF TRANS+ LF + INC, S5: COLL + MRF TRANS+ LF+ AD	The scenario, which included source reduction, collection, transport, and landfilling, was the one with the minimum contribution in all the impact categories but global warming and final solid waste as hazardous and non-hazardous, due to the source reduction process and subsequent recycling of the sorted materials in addition to less solid waste input to landfill. The lowest contribution to GWP was calculated for the anaerobic digestion process.

Thailand, UMI, (Wanichpongpan and Gheewala, 2007)	FU: 1 ton of collected MSW Model: NM Method: NM	S1: Included a single landfill using the methane emitted for electricity production. 2: Two small landfills without electricity production and with flaring of the collected methane.	The authors were primarily interested in the possibility of global warming as an impact category. Main conclusions: With landfill gas-to-energy projects, centralized landfills are practical and preferable to the current small waste management approach.
Urban area, (Iriarte et al., 2009)	FU: 1500 tons of MSW Model: SimaPro 7.0.2 Method: CML 2 baseline 2000 method	Collection systems modeled on densely populated urban areas: i) the mobile pneumatic, ii) the multi-container, and iii) the door-to-door.	At a distance of 20 kilometers between cities, the mobile pneumatic system has the largest environmental effect and energy consumption. Therefore, inter-city travel is critical; the findings indicate that this becomes the sub-stage from a distance of 11 kilometers between cities, which greatly influences

			global warming and energy consumption in all systems.
Italy, HI, (Cherubini et al., 2009)	FU: NM Model: SimaPro 7.1 Method: NM	S0: LF without any further treatment; S1: LF with biogas utilization; S2: Sorting plant coupled with biogas production; S3: Direct INC	Results showed that Scenarios (0 and 1) are the worst waste management options and scenario 2 is likely to be the best option with biogas production. Scenario 3 is better than Scenarios (0 and 1) from an environmental point of view.
Tehran, Iran, UMI, (Abduli et al., 2011)	FU: 1 ton of MSW Model: NM Method: NM	S1: LF S2: LF +COM	The landfill scenario is the preferred alternative for Tehran in the current situation, both in terms of environmental and economic factors.
Vietnam, LMI, (Thanh and Matsui, 2013)	FU: NM Model: NM Method: IPCC	S1: Open dumping S2: LF S3: LFGTE S4: COM + LF S5: INC +LF	Incineration with energy recovery seems to be the best solution.

		S6: AD + LF + INC	
China, UMI, (Yang et al., 2013)	FU: 1 ton of MSW Model: NM Method: IPCC	S1: Open dumping S2: LF without LFG collection S3: LF S4: LF with LFG utilization	Among all the scenarios evaluated, S4 has the least environmental impact.
United kingdom, HI, (Jeswani et al., 2013)	FU: 1 ton of MSW Model: GaBi Method: NM	S1: INC S2: LF with LFG to energy	Waste incineration saves significant amounts of greenhouse gases (GHG) compared to LF.
Bangalore, India, LMI, (Babu et al., 2014)	FU: NM Model: NM Method: CML	S1: Open dumping S2: LF without gas recovery S3: LF S4: Bioreactor landfill	The bioreactor landfill had an environmental and economic advantage over the other approaches. However, the open dump scenario was the least preferred choice since the environmental impacts were so significant.
United Kingdom, HI, (Parkes et al., 2015)	FU: 1 ton of MSW Model: GaBi Method: CML	S1: COM + RECY + LF S2: COM + RECY + INC S3: AD + RECY + LF with energy recovery	The incineration with energy recovery treatment scenarios

		S4: AD + RECY + INC S5: RECY + LF with energy recovery S6: RECY + INC S7: COM + RECY + gas S8: AD + RECY + gas S9: MBT + COM + RECY + INC S10: MBT + AD+ RECY + INC	provides the best environmental solution.
Kolhapur city, India, LMI, (Mali and Patil, 2016)	FU: 1 ton of MSW Model: Simapro Method: CML	S1: Open dumping S2: RECY + COM + LF S3: RECY+ AD + LF S4: RECY + PY	S2 was found to be the option with minimum environmental impacts.
Mumbai, India, LMI, (Sharma and Chandel, 2017)	FU: 1 ton of MSW Model: GaBi Method: IPCC	S1: 31% Bioreactor landfill + 69% open dumping S2: 3.2% RECY + 96.8% SLF S3: 3.2% RECY + 32% COM + 64.8% SLF S4. 3.2% RECY+ 32% AD +64.8% SLF S5. 3.2% RECY+ 16% COM + 16% AD + 64.8% SLF S6: 3.2% RECY + 8% COM+	The scenario with material recovery facility (MRF), composting, anaerobic digestion, and incineration had the least environmental effect.

		88.8% INC S7: 3.2% RECY+ 96.8% INC	
China, UMI, (Havukainen et al., 2017)	FU: MSW generated in 2013 Model: GaBi Method: CML	S1: INC + LF S2: RDF + INC + COM + LF S3: RDF +INC + Bio drying S4: RDF + INC + AD S5: RDF + INC + ethanol S6-S9: Same as S2-S5, the difference being the Incineration plants are new and have higher electric efficiency.	According to the findings, RDF production and incineration might enhance Hangzhou's MSWM global warming, acidification, and eutrophication compared to MSW incineration with coal.
Spain, HI, (Fernández-Gonzalez et al., 2017)	FU: NM Model: SimaPro Method: Impact 2002	S1: MBT and LF S2: MBT and LF with LFG to energy S3: MBT, AD, and LF with LFG to energy S4: MBT, RDF, LF with LFG to energy S5: MBT, GAS, and LF S6: MBT, INC, and LF	The WtE energy technologies, including incineration, had a better advantage in environmental terms, and anaerobic digestion is the optimal solution.
Italy, HI, (Cremiato et al., 2018)	FU: 1 ton of MSW Model: GaBi Method: CML	S1: Diversion rate ¼ 50%. Material recovery facility MRF (unsorted waste), waste to energy (WTE), composting,	The most significant outcome is that the increased separate collection of recyclable materials used as raw material

		Landfill S2: Same as S1, diversion rate ¼ 60% S3: Diversion rate ¼ 50%. MRF, WtE, AD, LF S4: Same as S3, diversion rate ¼ 60 MSWM by WtE	substitutes and bio-waste used for renewable energy generation helps to reduce direct and indirect burdens associated with the total life cycle of product manufacturing.
Dhanbad City, India, LMI, (Yadav and Samadder, 2018)	FU: 1 ton of MSW Model: SimaPro Method: CML 2 baseline 2000 method	S1: COLL + TRANS S2: RECY + Open burning + open dumping, + UnLF S3: COM + LF without gas recovery S4: RECY + COM + Disposal of inert waste at LF without gas recovery	The worst disposal option for mixed solid waste was found to be landfilling without energy recovery. The most environmentally sustainable technology for the study area was found to be scenario S4.
Nagpur city, Mumbai, LMI, (Khandelwal et al., 2019)	FU: 1 metric ton of MSW Model: GaBi Method: CML-1A method	S1: 17% COM + 83% LF S2: 44% LF + 48.1% COM + 7.5% MRF S3: 44% LF + 48.1% AD + 7.5% MRF S4: 44% LF + 24.5% AD + 25.4% COM + 7.5% MRF	S2 was found to have the least adverse effects on the environment in the areas of global warming, human toxicity, eutrophication, and photochemical ozone formation.
USA, HI, (Lee et al., 2020)	FU: 1 ton of wet waste Model: SimaPro	S1: Hs-AcD S2: LF with LFG utilization	The HS-AcD is the best choice to manage biosolids and

	Method: NM	S3: LF without LFG utilization S4: COM S5: INC	organic waste in environmental and economic sustainability.
Brasilia, Brazil, HI, (Silva et al., 2021)	FU: Amount of waste generated Model: NM Method: NM	S1: Landfill S2: RDF in the south wing S3: RDF + MBT in sector p sul S4: RDF + MBT in the south wing	Results concluded that environmental benefits through RDF plant scenarios resulting in a reduction of fossil fuels are included in this analysis.
Mumbai, India, LMI, (Sharma and Chandel, 2021)	FU: I ton of MSW Model: NM Method: LCC method	S1: (20% of recyclable materials) and SLF (remaining fraction) S2: MRF (20% of recyclable materials), COM (80% of biodegradable waste), and SLF (remaining fraction) S3: MRF (20% of recyclable materials), AD (80% of biodegradable waste), and SLF (remaining fraction) S4: MRF (20% of recyclable	Due to reduced operational costs, the scenario combining recycling and sanitary landfill was the most economically feasible alternative, with a net LCC of US$19 per ton of MSW.

	materials), COM (40% of biodegradable waste), AD (40% of biodegradable waste), and SLF (remaining fraction) S5: (20% of recyclable materials), COM (20% of biodegradable waste), and INC (remaining fraction) S6: (20% of recyclable materials) and INC (remaining fraction)

S1, S2, S3: stands for scenarios chosen in respective studies.

Abbreviations used in Table 2.3: LF (Landfill); COMP (Composting); RDF (Refuse Derived Fuel); PY (Pyrolysis); INC (Incineration); RECY (Recycling); AD (Anaerobic Digestion); MBT (Mechanical and Biological Treatment); WTE (Waste to Energy); UnLF (Unsanitary Landfill); SLF (Sanitary Landfill); TRANS (Transportation); COLL (Collection); LFG (Landfill gas); LFGTE (LFG to energy); GHG (Greenhouse gas); NM (Not Mentioned); LCC (life cycle cost); LCA (Life cycle assessment).

2.6 MSW OPEN DUMPING

Unsustainable techniques that enhance environmental pollution and disease spread aggravate solid waste management (SWM) in developing countries. The primary difficulties identified include open dumping at unregulated locations, open burning of waste fractions, and mishandling of the leachate generated at final disposal sites (Abd Manaf et al., 2009; Wilson et al., 2015). The problem is caused in slum regions, which face additional challenges such as high human density, traffic, and air and water pollution. In these circumstances, uncontrolled dumping in open places near water bodies is a daily problem, which correlates to public health concerns (Abd Manaf et al., 2009). The principal environmental consequences evident in ultimate open-air disposal are:

- Visual impacts,
- Air contamination, odors and greenhouse gasses (GHG) emission,
- Vectors of diseases,
- Surface water and groundwater pollution.

Organic carbons, ammonia, chloride, and heavy metals are found in landfill leachate generated at open dumpsites (Torretta et al., 2017), as well as significant quantities of fluoride, chloride, ammonium–nitrogen, biological oxygen demand (BOD), and chemical oxygen demand (COD) (Karak et al., 2013). For example, MSW dumped at the Mathkal dump site (Kolkata, India) is affecting water quality in and around the dumpsite area: Cd and Ni are detectable in leachate, improving groundwater contamination; the metals Pb, Cd, Cr, and Ni are characterized as toxic for drinking water, and the concentration of these components increases near an unsanitary landfill, potentially causing serious health effects; Indeed, it has

been found that leachate increased the concentrations of Cr, Cd, and Mn in groundwater, harming the population's health and the environment's quality (Biswas et al., 2010).

2.7 LANDFILL LEACHATE GENERATION

Annual precipitation, runoff, infiltration, evaporation, transpiration, mean ambient temperature, waste composition, waste density, initial moisture content, and landfill depth are some of the most important parameters that impact leachate formation. Landfills go through various phases. Leachate is initially created only as long as the landfill fills up with waste. In other words, leachate is generated after the wastes are saturated with moisture or when it can no longer retain more moisture without leakage. If the gravitational forces are stronger than the holding capacity, percolation has occurred (Leckie et al., 1979). However, this depends on the initial moisture content of the landfilled solid wastes (Chu et al., 1994). The more water that travels through the waste, the more contaminants are leached in general. As a result, developing methods to quantify the quantity of leachate generated in landfills is essential. The leachate generation cycle is illustrated in Fig. 2.12.

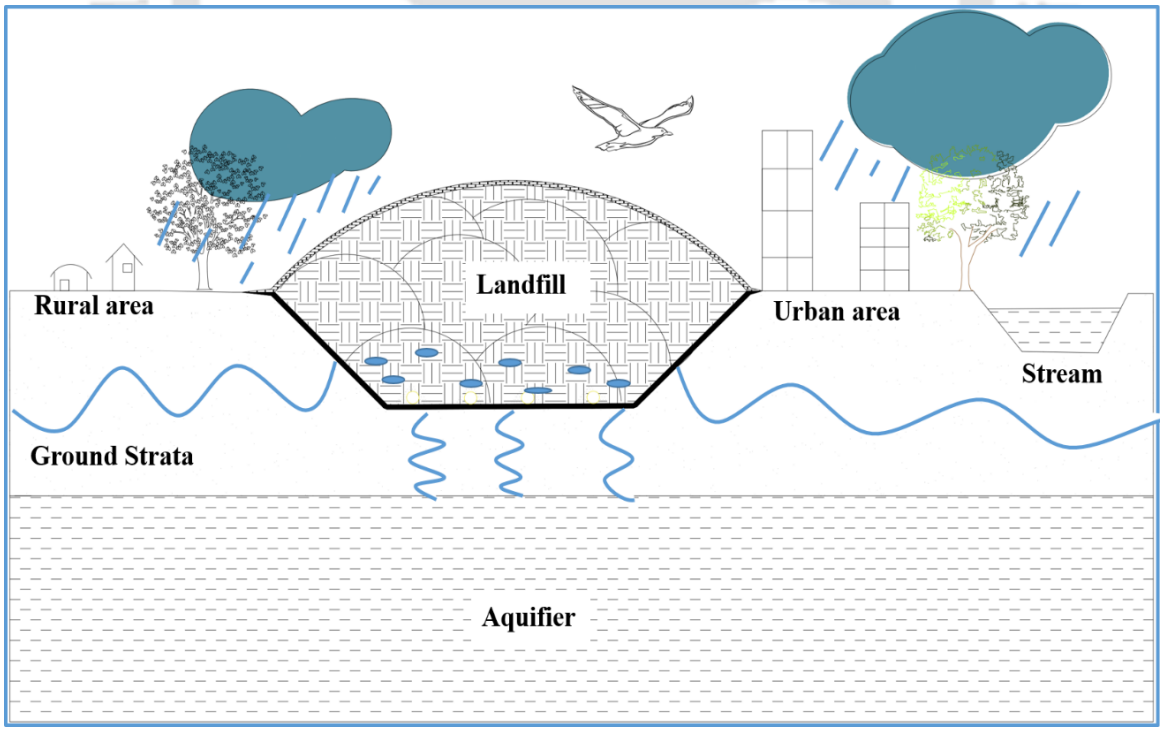


Fig. 2.12. Leachate generation cycle

2.7.1 Factors affecting leachate quality in landfills

In most cases, the quality of the leachate is highly varied. Various complex elements, such as waste composition, depth, moisture content, dissolved oxygen, landfill design and management, and waste age, influence the level of variance in leachate quality. However, researchers have identified the major elements as contributing to variance in leachate quality in general.

2.7.1.1 Waste composition

The amount of municipal waste can vary significantly, with different compositions and characteristics. In general, the waste composition determines the extent of biological activity within the landfill sites (Wimalasuriya et al., 2011). For example, organic material in leachate comprises garbage, food, and garden wastes, agricultural and animal residues, and Inorganic constituents in leachate are often derived from ash wastes and construction and demolition debris (Pohland et al., 1985). Therefore, solid waste is heterogeneous; only basic assumptions about the relationship between waste composition and leachate quality can be made.

2.7.1.2 Depth of waste

At comparable precipitation and percolation conditions, leachates from deeper landfills have higher amounts of contaminants (Qasim and Chiang, 1994). Deeper fills need more water to attain saturation, take longer to decompose, and spread the leached material across a longer period (Lu et al., 1985). Water that enters the fill will pass through the waste. As water percolates through the landfill, it comes into touch with the garbage and leaches chemicals from it. Deep landfills enhance the contact period between the liquid and solid phases, increasing the leachate's strength (McBean, 1995).

2.7.1.3 Moisture availability

The most important component influencing waste stabilization and leachate quality is moisture. Moisture addition has been shown consistently to stimulate methanogenesis (Barlaz et al., 1990). while other research indicates that moisture transport through the waste is just as significant as water addition (Klink and Ham, 1982). Moisture in landfills acts as a reactant in hydrolysis processes, transports nutrients and enzymes, dissolves metabolites, offers pH buffering, dilutes inhibitory chemicals, exposes the surface area to microbial action, and regulates microbial cell growth (Noble and Arnold, 1991). High moisture flow rates can wash

soluble organics and microbial cells out of the landfill, and that microbial activity has a reduced role in determining leachate quality in such scenarios (Lu et al., 1985). Therefore, high moisture application rates may remove most waste pollutants early stage. Unfortunately, only a small amount of moisture to support biological breakdown in dry landfills (i.e., 20 to 40% water) have extremely poor stabilization rates. The recommended moisture content in the literature varies from a minimum of 25% (wet basis) to optimal values of 40 to 70% (Chen and Bowerman, 1974; Barlaz et al., 1990).

2.7.1.4 Available oxygen

The amount of free oxygen in a landfill determines the kind of decomposition (anaerobic vs. aerobic). While there is oxygen present, the aerobic breakdown happens during the first dumping of waste. After that, aerobic degradation may continue to occur below the fill's surface (Barlaz et al., 1990; McBean, 1995). Chemicals generated by aerobic decomposition vary significantly from those generated by anaerobic decomposition (Bagchi, 1994). Microorganisms decompose organic matter to CO_2, H_2O, and partially decomposed residual organics during aerobic decomposition, producing significant heat. During anaerobic decomposition, high quantities of organic acids, ammonia, hydrogen, carbon dioxide, methane, and water are generated (McBean, 1995).

2.7.1.5 Temperature

Landfill temperature, a mostly uncontrollable factor impacting leachate quality, has also changed with seasonal ambient temperature fluctuations. Various salts (for example, $Ca_3(PO_4)_2$ and NaCl) become more soluble as the temperature rises. However, the solubility of several chemicals in leachates, such as $CaCO_3$ and $CaSO_4$, decreases with rising temperatures. (Lu et al., 1985). The temperature has an impact on bacterial growth and chemical reactions in landfills. Microorganisms have been reported to persist at temperatures ranging from -5 to 80 °C. Anaerobic reactions occur best at 30-38°C for mesophilic organisms and 50-65°C for thermophilic species (Parkin and Owen, 1986).

2.7.1.6 Processed waste

Waste shredding may have a significant impact on leachate properties. Shredded waste leachate is more polluted during the early stages of waste stabilization and less polluted during the latter stages than unshredded waste leachate (Reinhardt and Ham, 1974). Authors have agreed that leachate from shredded fills contains much more contaminants than leachate from

unshredded landfills (Lu et al., 1985; Qasim and Chiang, 1994). This higher leachate strength may be attributed to greater surface area and, as a result, greater biodegradation rates in shredded waste landfills (Lu et al., 1985). Researchers have found that shredding did not affect waste decomposition in laboratory-scale lysimeters (Tittlebaum, 1982). While field capacity (i.e., the maximum amount of moisture that can be retained without continuous downward percolation by gravity) was delayed in shredded waste landfills, the rate of pollutant removal, solid waste decomposition, and cumulative mass of pollutants released per unit volume of leachate was significantly increased when compared to unshredded waste landfills (Qasim and Chiang, 1994).

2.7.1.7 Age of landfill

The quality of leachate is strongly impacted by the amount of time that has passed after garbage dumping. Because the amount of chemicals in waste is limited, the quality of the leachate reaches a high after approximately two to three years, followed by a gradual decline over the subsequent years (Lu et al., 1985; McBean, 1995). Generally, leachate from new landfills will have high levels of BOD_5 and COD for the first few years before gradually declining and flattening out after around ten years (Akyurek, 1995). However, not all pollutants reach their maximum levels at the same time. Organic chemicals degrade more quickly than inorganic chemicals as the age of the landfill increases (Service et al., 1977). Inorganics are only removed by rainfall infiltration due to washout (Qasim and Chiang, 1994). Organic compounds, however, decrease in concentration through decomposition as well as a washout.

2.7.2 Biodegradation processes

Heterotrophic microbes use soluble organic molecules as a source of carbon and energy (bacteria, fungi). Certain heterotrophic and chemoautotrophic bacteria (nitrifying and methanogenic bacteria) use hydrocarbons primarily as an energy source and carbon dioxide as a carbon source. Bacteria obtain the energy they need for growth (anabolism) through redox reactions (catabolism); the quantity of energy produced varies according to the reaction, as shown in Table 2.4 (Anonymous, 2017); aerobic respiration activities give much more energy than methanogenesis and anaerobic processes in general. Invariably, microorganisms will select an energetically favorable response. Facultative bacteria can alter their catabolism in response to changing environmental conditions (Cossu et al., 2018). Due to the heterogeneity

of the waste and the different conditions present in a landfill, all the reactions may occur in parallel (Berge et al., 2005).

Table 2.4. Energetic behavior of the main biochemical redox reactions (Cossu et al., 2018).

Biochemical Redox Reaction Name	Oxidation	Formula	Energy Yield (kcal/mol)
Aerobic respiration	O_2	$O_2 + CO_2$	686
Denitrification	NO_3^-	$NO_3^- + H^+ \rightarrow N_2$	649
Iron reduction	Fe^{3+}	$Fe_2(OH)_3 + 2H^+ \rightarrow Fe_2^+$	300
Sulfate reduction	SO_4^{2-}	$SO_4^{2-} + 2H^+ \rightarrow HS^-$	190
Methanogenesis	CO_2	$H_2 + CO_2 \rightarrow CH_4$	8.3

Biodegradable organics are hydrolyzed and ultimately transformed to methane and carbon dioxide in anaerobic conditions. Although the generation of biological heat is minimal, it enables the waste body to reach and maintain mesophilic temperatures (35-45°C) or even higher) because the waste acts as an insulator. In comparison to the carbon metabolized, the biomass created is less than 5%. On the other end, this signifies a high rate of carbon compound reduction, while on the other, it predicts a much slower rate of metabolism than aerobic breakdown. The reactions listed below indicate the general formula for aerobic (2.1) and anaerobic (2.2) glucose degradation:

$C_6H_{12}O_6$ (1 kg) + O_2 (0.64 kg) → CO_2 (0.88 kg)+ H_2O (0.34 kg)+ Biomass (0.4 kg) + Heat (9300 kJ) (2.1)

$C_6H_{12}O_6$ (1kg)) → CH_4 (0.25 kg) + CO_2 (0.69 kg) + Biomass (0.056 kg) + Heat (632 kJ) (2.2)

Since bacteria grow relatively quickly in the presence of oxygen, biochemical processes are accelerated; further, some refractory complex organic polymers may be hydrolyzed (cometabolized) and ammonia nitrogen nitrified. Oxygen is not distributed evenly throughout the landfill and may not diffuse into all solid waste particles. As a result, methane production

may continue locally, albeit it will almost certainly be oxidized in the presence of oxygen. Due to the slow methane oxidation rate, methane at low concentrations may remain in the off-gases recovered during in situ aeration. The biochemical processes can be subdivided into aerobic and anaerobic processes, as shown in Fig. 2.13.

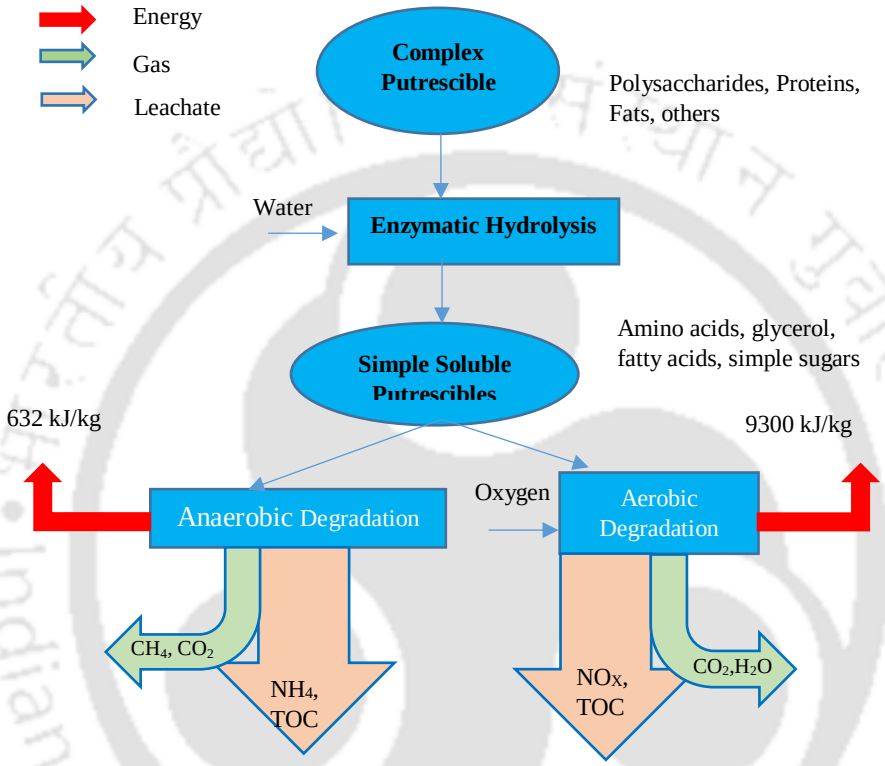


Fig. 2.13. Overview of aerobic and anaerobic processes in a young landfill with average energy production and main catalytic products (Cossu et al., 2018)

Organic chemicals are degraded, but new big molecules may form simultaneously. Humic and fulvic-like compounds, which are big molecules with a high molecular weight. Polyphenols as precursors are released from cell walls of microorganisms, particularly fungi, and are released during the breakdown process in substances such as lignins, resulting in humic compounds that are not well characterized.

2.7.2.1 Landfill microorganisms

A diverse spectrum of active microorganisms may be found in municipal solid waste (MSW) landfills; certain bacteria can catabolize just certain substrates in a limited pH or

temperature range and do not form spores (i.e., methanogenic bacteria). On the other hand, some are more adaptable and can live in various situations (i.e., facultative bacteria). The major bacterial groups are shown in Table 2.5 (Sleat et al., 1989; Berge et al., 2005). In addition, fecal bacteria and viruses may be found in new landfilled waste, but their numbers decline quickly owing to competition with other organisms and high temperatures when present; their presence in leachate is regarded as minimal (Andreottola and Cannas, 1992).

Table 2.5. Key bacterial groups involved in the biochemical processes of landfills indicate their main activity

Bacterial Group	Main Biochemical Activity
Amylolytic bacteria	Hydrolysis and degradation of starch
Proteolytic bacteria	Hydrolysis and degradation of proteins
Cellulolytic bacteria	Hydrolysis and degradation of cellulose
Hemicellulolytic bacteria	Hydrolysis and degradation of hemicellulose
Hydrogen-oxidizing methanogenic bacteria	Methane production from hydrogen and carbon dioxide
Acetoclastic methanogenic bacteria	Ethane and carbon dioxide production from acetic acid
Sulfate-reducing bacteria	Sulfate reduction to sulfite through acetic acid
Nitrosomonas	Converts ammonia nitrogen into nitrite
Nitrobacter	Converts nitrite into nitrate
Thiobacillus denitrificans	Uses inorganic sulfur to reduce nitrates in autotrophic denitrification

2.7.2.2 Anaerobic degradation

The main processes are hydrolysis of large-molecular-mass substances (carbohydrates, proteins, and lipids) into soluble sugars, amino acids, long-chain carboxylic acids, and glycerol. Because polymers are difficult to transfer across microbial cell membranes, biological hydrolysis of polymers in the waste is mediated by extracellular enzymes generated

by landfill bacteria (Palmisano et al., 1993). Fermentative microorganisms then break down these hydrolysis products into short-chain carboxylic acids, ammonia, carbon dioxide, hydrogen, acetic acid, and proton-reducing acetogens oxidize fermentation products propionate, and butyrate to acetate, carbon dioxide, and hydrogen. However, this process is thermodynamically favorable only at low hydrogen concentrations (Zehnder, 1978). The overall process of anaerobic decomposition of biological polymers is shown in Fig. 2.14.

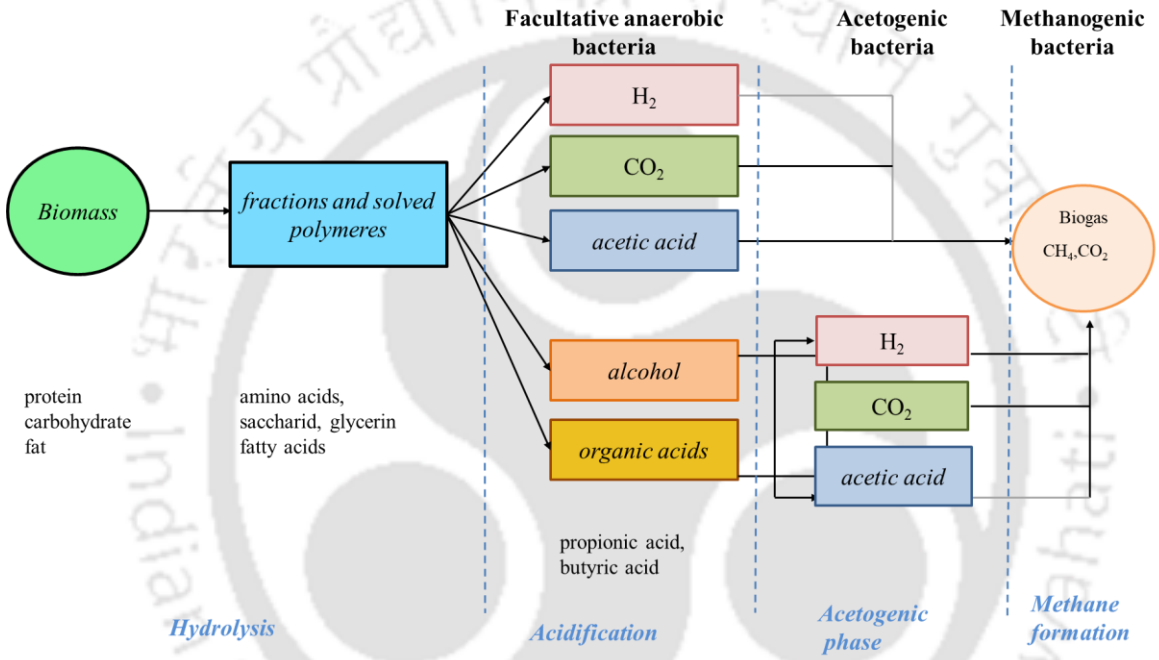


Fig. 2.14. The overall process of anaerobic decomposition of biological polymers (Brock et al., 1994)

Anaerobic degradation can be subdivided into four subsequential phases: Hydrolysis, Acidogenic fermentation, Acetogenic fermentation, and Methanogenesis, in addition to sulfate reduction, which may compete with the last three processes in substrate utilization. The stabilization of waste proceeds in five sequential and distinct phases:

1) Aerobic phase: The initial phase of the degradation process. Aerobic bacteria consume oxygen while breaking down the long molecular chains of complex carbohydrates, proteins, and lipids that composite organic waste. For example, Eq. 2.3 indicates organic waste broken down into a simple sugar compound (Ostrem and Themelis, 2004).

$C_6H_{10}O_4 + 2H_2O \rightarrow C_6H_{12}O_6 + 2H_2$ (2.3)

2) Acidogenic phase: It is the second phase of the degradation process. Acidogenic bacteria covert the products into volatile acids, ketones, alcohols, hydrogen, and carbon dioxide. The main products of the stage are propionic acid, butyric acid, and acetic acid. Hydrogen, carbon dioxide, and acetic acid will skip the third stage and are utilized by methanogenic bacteria are represented in Eq. 2.4, 2.5 & 2.6 (Ostrem and Themelis, 2004).

$C_6H_{12}O_6 \leftrightarrow 2CH_3CH_2OH + 2CO_2$ (2.4)

$C_6H_{12}O_6 + 2H_2 \leftrightarrow 2CH_3CH_2COOH + 2H_2O$ (2.5)

$C_6H_{12}O_6 \rightarrow 3CH_3COOH$ (2.6)

3) Acetogenic phase: In the third phase, acetogenic bacteria consume the acidogenic phase of acids and convert them into hydrogen, carbon dioxide, and acetic acid, represented in Eq. 2.7 (propionate to acetate), 2.8 (Glucose) & 2.9 (ethanol) (Ostrem and Themelis, 2004).

$CH_3CH_2COO^- + 3H_2O \leftrightarrow CH3COO^- + H^+ + HCO_3^- + 3H_2$ (2.7)

$C_6H_{12}O_6 + 2H_2O \leftrightarrow 2CH_3COOH + 2CO_2 + 4H_2$ (2.8)

$CH_3CH_2OH + 2H_2O \leftrightarrow CH_3COO^- + 2H_2 + H^+$ (2.9)

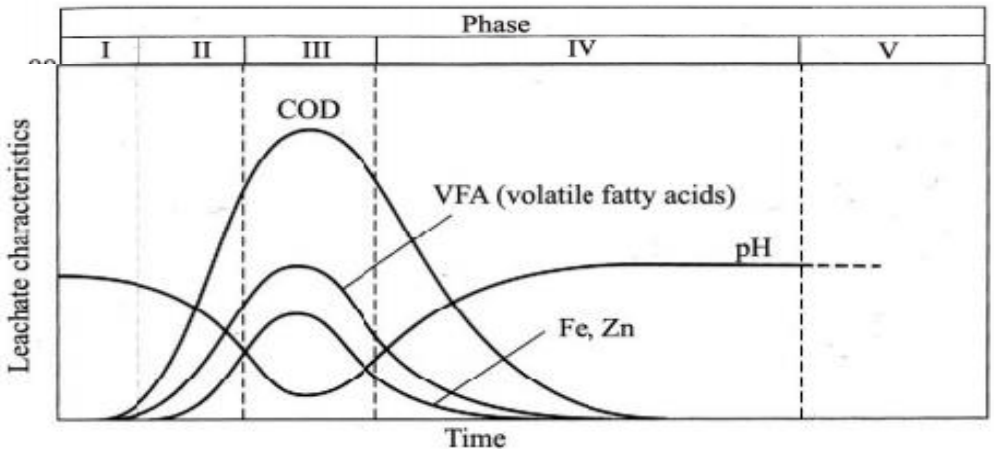


Fig. 2.15. Generalized phases in the generation of landfill gases (I = initial adjustment, II = transition phases, III = acid phase, IV = methane fermentation, and V = maturation phase) (Pohland, 1991; Pohland, 1982)

4) Methanogenic phase: The final phase is methanogenesis. In this phase, methanogenic bacteria consume acetic acid and produce methane and carbon dioxide. Hydrogenotrophic methanogens use hydrogen as a source of energy to produce methane. Methanogenic activity is optimal around pH 7. The activity of fermenting organisms exceeds that of carboxylic acid degraders and methanogens. The accumulation of H_2 and carboxylic acids leads to pH drops and inhibits the methanogens in Eq. 2.10, 2.11 & 2.12.

$$CO_2 + 4H_2 \rightarrow CH_4 + 2H_2O \quad (2.10)$$

$$2C_2H_5OH + CO_2 \rightarrow CH_4 + 2CH_3COOH \quad (2.11)$$

$$CH_3COOH \rightarrow CH_4 + CO_2 \quad (2.12)$$

2.7.2.3 Leachate characteristics

In the case of landfill leachate, it was found that the composition varies significantly depending on the age of the landfill (Baig et al., 1999). Generally, the age of a landfill may be measured by two stages of decomposition: acid and methanogenic. For new landfills containing a high concentration of biodegradable organic waste, a fast anaerobic fermentation occurs. The primary result is the volatile fatty acids (VFA) early phase, referred to as the acid phase (Welander et al., 1997). The methanogenic phase happens when a landfill grows. Methanogenic bacteria proliferate in the garbage during this phase, and the VFA is transformed into landfill gas (CH_4, CO_2) (Renou et al., 2008). Additionally, the organic part of the leachate becomes dominated by non-biodegradable refractory compounds such as humic chemicals (Chian and DeWalle, 1976). Table 2.6, by which the general behavior tendency can be observed and defined (Kjeldsen et al., 2002). The leachate properties are dependent on the waste components and moisture content and their sources. For landfill leachate, rainfall significantly affects the leachate's properties. Rainwater penetrates the landfill and transports soluble contaminants from the solid phase to the liquid phase. Simultaneously, organic matter in trash decomposes into soluble organic matter (such as volatile fatty acids), incorporated into the leachate by microbial activity. Not only does leachate have a high concentration of organic pollutants, but it also includes a wide range of organic and inorganic contaminants, as well as toxic components (Tränkler et al., 2005).

Table 2.6. The typical range of leachate composition in municipal solid waste (Tränkler et al., 2005).

Leachate parameter	Typical range (mg/L)
pH	4.5-9
Specific conductivity(µs/cm)	2,500-35,000
Total solids	2,000-60,000
Heavy Metals	Range (mg/L)
Arsenic	0.01-1
Cadmium	0.0001-0.4
Chromium	0.02-1.5
Cobalt	0.005-1.5
Copper	0.005-10
Lead	0.001-5
Mercury	0.00005-0.16
Nickel	0.015-13
Zinc	0.03-1,000
Inorganic Macro components	Range (mg/L)
Total phosphorous	0.1-23
Chloride	150-4,500
Sulphate	8-7,750
Carbonic acid	610-7,320
Sodium	70-7,700
Potassium	50-3,700
Ammonium-N	50-2,200
Calcium	10-7,200
Magnesium	30-1,50,00
Iron	3-5,500
Manganese	0.03-1,400
Silica	4-70
Organic Matter	Range (mg/L)
Total organic carbon	30-29,000

Biochemical oxygen demand	20-57,000
Chemical oxygen demand	140-1,52,000
BOD/COD (ratio)	0.02-0.80
Organic nitrogen	14-2,500

Authors have investigated 33 landfills in Northern Germany; the leachate concentrations mainly derived from the late eighties and early nineties. Three characteristic periods were defined according to the BOD/COD ratio is illustrated in Table 2.7.

Table 2.7. Classification of landfill waste degradation phases

Degradation phases	BOD/COD range
Acid phase:	BOD/COD $\geq$ 0.4
Transient phase:	0.4 > BOD/COD > 0.2
Methanogenic phase:	BOD/COD $\leq$ 0.2

2.7.3 Laboratory scale landfill simulation studies on leachate characterization

The microbiological degradation of waste could lead to long–term landfill emissions behavior is controlled by chemical and physical conditions (temperature, moisture content, and substrate) (Ab Ghani et al., 2017). Recently, studying the mechanism, characteristics, and trends of landfill emissions has become an important aspect of waste management studies. To develop a more realistic perspective of waste degradation and leaching, landfill simulation reactors (LSRs) have been used to generate reliable information on long-term landfill behavior with time and early-stage waste degradation (Ejlertsson et al., 2003; Sponza and Ağdağ, 2004).

Ehrig (1983) measured the quantity and quality of leachate from 13 sanitary landfills in Germany. The quality and quantity of leachate from 13 sanitary landfills in Germany were measured. The organic content of landfills ranges between 30 and 40%. In the acid phase, the average COD and BOD_5 values were 22000 and 13000 mg/L, respectively. Because of variations in pH, the concentrations of certain inorganic components, such as Fe and Ca, are comparable. Ammonia levels gradually rise as landfills age. Heavy metal concentrations are typically less than 1 mg/L.

Pohland et al. (1993) demonstrated three simulated landfill columns consisting of a 20-gauge steel cylinder, 0.9 1m in diameter and 3.25 m in length. The cylinders were prepared with an internal coating of epoxy paint and equipped with a leachate collection/recycle system, volumetric gas meters, and lifting plates to facilitate the eventual removal and retrieval of the stabilized refuse and co-disposed diapers. The 627 kgs of shredded municipal solid waste and 20 diapers were used in each column. After loading, the columns were covered, and tap water was added (8.4 L/week) to simulate natural rainfall infiltration (1.27 cm/week). The mode of operation of the reactor was recirculation of leachate, and collection of leachate was once a week. The results indicated that a high amount of VFA was produced in the acidic phase due to anaerobic conditions.

Bae et al. (1998) fabricated three lysimeters with PVC cylinders, and gas and leachate sampling ports were installed at three different depths of the lysimeter. The composition of the wet waste was 53.5%, and the dry part was 46.5%. The waste was collected from residential areas and operated with and without recirculation. The findings showed that the initial pH of the leachate was approximately 4 for all lysimeters, increasing to 6 at t=60 days. Even with high COD during the early stages of the experiment, the rapid increase in pH can be attributable to the formation of NH_4^+-N (2,000-5,000 mg/l) from food waste degradation. As a result, it was shown that COD values increased gradually and that a long period was required for COD concentrations to decrease in an analyzer operating with no-leachate recirculation.

Rodriguez-Iglesias et al. (1998) designed an opaque PVC pilot-scale research with a diameter of 0.5 m and a height of 2.83 m. Recirculation of leachate to the reactor was the mode of operation, and the controlled temperature was 36ºC. At the landfill site, water was added in the same amount as the daily rainfall. The organic part of waste accounted for 52%, with the remaining dry waste accounted for 48%. The study's main findings were that the pH of the leachates was low (about 5.5-6.1) from days 40 to 170 when it started to rise to neutral. During the early phase of waste degradation, a significant amount of volatile fatty acids was formed, and metals including Fe, Mn, and Zn were found in the leachates.

Rodríguez Iglesias et al. (2000) fabricated a pilot plant was constructed in opaque PVC, the diameter and height being 0.5 and 3.6 m, respectively. The organic fraction of waste was 52%. The reactor's operation with leachate recirculation and the controlled temperature was kept constant at 36 ± 1ºC using an external water jacket. The results indicated that COD

concentration (80,000 mg/L and decreased to 40,000 mg/L until 150 days) from the landfill was very high during the dry season since no water addition. However, the decrease in COD of the leachates with time being slower in the landfill than the pilot plant. The ammoniacal-N content of the leachates generated in both the landfill and the pilot plant was high, reaching values of around 2,000 mg/L in the early and 1500 mg/L in the latter. Ammoniacal-N is not eliminated under anaerobic conditions. It must be noted that the metal content in the leachate from the pilot plant and the sanitary landfill was low. This may be because the dissolved metals were retained in the solid phase and possibly precipitated as sulfides.

San and Onay (2001) constructed in the laboratory using two PVC pipe columns with a length of 1 m and a diameter of 0.35 m. The reactor was operated with and without recirculation and kept the constant temperature at 34^{o}C. Interestingly, the pH of both the recirculation and single-pass reactors showed a similar trend throughout the first five phases of the experiment. An average pH value of 5.5 was not suitable for establishing the methanogenic conditions. The COD concentrations from the single-pass and recycle reactors increased from 5,000 to 45,000 and 39,000 mg/L, respectively. However, the addition of water to the recycle reactor on day 125 caused a decrease in the COD concentration.

Chan et al. (2002) fabricated triplicated stainless steel columns of 15 cm (diameter)150 cm (length) were employed to study the influence of leachate recirculation and without recirculation on anaerobic degradation of a mixture composed of MSW sewage sludge and marine dredging (75:20:5, fresh weight basis). The system operation of test columns was placed in a laboratory with a constant temperature at 38^{o}C, which is close to the optimum mesophilic temperature of 42^{o}C for anaerobic fermentation of MSW. The study's key findings were ammonium concentrations varied between 1,000 and 1,200 mg/L after 250 days of operation in a landfill reactor operated without recirculation.

Ejlertsson et al. (2003) constructed a lysimeter of 3 m height and 1.3 m diameter was using reinforced cement concrete rings and filled with 20 cm gravel, 2.1 m height. The organic fraction was 52%, and the dry part was 49%. The operation of LSR (landfill simulated reactor) by pre-composting, the acidogenic stage was avoided, allowing methanogenesis to develop as soon as the air was switched off. Leachate samples were collected every month and analyzed for pH, EC, TS, TVS, TDS, COD, BOD_5, VFA, chloride, nitrate, nitrite, sulfate, phosphate, calcium, magnesium, TOC, ammonia, sodium, and potassium. Reduction in leachate organics

as indicated by the decrease in leachate BOD_5 (99%), COD (88%), and TOC (81%) improved organic removal efficiency achieved through leachate recirculation.

Sponza and Agdag (2004) constructed a cylindrical reactor of 30 cm in diameter and 100 cm in height. The reactor has an electronic heater to provide a temperature of 35ºC and was hermetically sealed with a gasket and silicone sealant to maintain anaerobic conditions. A single-pass reactor was operated without leachate recirculation, while the other two reactors were operated with leachate recirculation. The reactor contained 76% of organic waste, and operation times are 222 days. This study showed that the COD concentration in a single pass reactor (without recirculation of leachate) was 90,000mg/L, and the highest COD in this reactor was about 1,04,000 mg/L was reported during the dry season. The initial VFA concentration in the single-pass reactor was approximately 10,000 mg/l, and it was reported that the reactor is in the acid phase. Ammonia concentrations in the single-pass reactors displayed evidence of washout, although the concentrations were maintained above 500 mg/l during the experimental period. Therefore, high concentrations of leachate parameters were observed in a single-pass reactor.

Swati et al. (2005) constructed two lysimeters of 3 m height and 1.3 m diameter were constructed using reinforced cement concrete rings. One lysimeter operated with leachate recirculation, while the other reactor operated like an open dump. Manually segregated MSW collected from Chennai, used as substrate in two lysimeters. The two reactors were maintained with moisture at the field capacity of the waste. The results indicated that 312 L of leachate was generated in lysimeter 1 (L1) over a period of 120 days, and a reduction of 32% and 26% were observed in open dump and bioreactor lysimeters, respectively. COD concentration was 83,729 mg/L, while the pH ranges from 5.60-5.88.

Swati et al. (2011) fabricated four pilot-scale lysimeters of 3.0 m height and 1.3 m diameter were constructed outdoors using reinforced cement concrete rings. Reactor 1 (R1): Filled with fresh MSW to simulate a young controlled dump without leachate recirculation. The biodegradable fraction of fresh MSW consisted of a major portion of food waste (46%), and the dry part was 54%. The results showed that leachate calcium concentrations from young landfill lysimeters ranged from 3,500-4,900 mg/L in R1. High organic loads were evident in leachates from young landfill lysimeters (COD levels reaching up to 95000 mg/L, and BOD_5 levels up to 35,000 mg/L).

Rafizul and Alamgir (2012) studied three different landfill situations that were considered. The open dump lysimeter having a baseliner and sanitary landfill lysimeter-B and C at two different types of cap liner were simulated. The MSW deposited in the lysimeter mainly consists of 93 (w/w) organic (food and vegetables), 3 (w/w) of plastic/polythene and 2 (w/w) of leather/rubber, 1 (w/w) of animal bone, and rubber/leather as well as 1 (w/w) of rope/straw and egg pill. Leachate characterization, leachate generation, and climatic influence are continually monitored. The results indicated that COD was initially about 60,000 mg/L for both the open dump and sanitary lysimeter, but it was dropped steadily with the range of 60,000-10,240, 60,000-3840, and 60,000-35,550 mg/L until the end of the dry season. Moreover, the seasonal variation of the leachate generation pattern indicates that the highest degradation of MSW occurred during the rainy season and the lowest degradation of MSW during the dry season due to lack of moisture in the lysimeter. It is interesting to note that high strength leachate quality during the dry season even though low leaching activity was reported.

Wang et al. (2012) A demonstration pilot with seven anaerobic landfill simulators (LSRs) was used to study the impact of temperature in the range of 20-46^{o}C on long-term landfill emissions, i.e., (psychrophilic (20^{o}C), mesophilic (40^{o}C), and lower thermophilic ranges (50^{o}C). Approximately 85 kg of this originally municipal solid waste (MSW) containing mostly organic matter was filled in the reactor. The reactor's operation was recirculation of leachate, water addition, maintained liquid to solid ratio, and temperature. The results showed that higher temperatures accelerated the waste degradation and resulted in higher leachate COD and $NH4^{+}$-N concentrations. Hence, the increase in temperature can accelerate waste degradation and gas generation but cannot decrease the length of the waste stabilization. It is interesting to note that the fraction of carbon removed through the water path has been the highest at the lowest temperature, which is related to the lower biological activity than in the mesophilic range. The temperature changed from mesophilic to thermophilic, the value of the COD increased by 1.5%/^{o}C, which was 67% higher than when the temperature changed from psychrophilic to mesophilic. This confirms that a high temperature is advantageous for waste stabilization and organics leaching.

Zhao et al. (2013) investigated the fresh leachate that was sampled from a refuse transfer station in Shanghai, China. The station has a daily transferring capacity of approximately 800 tons of MSW, which generates about 100 tons of leachate per day. Seasonal effect on leachate

generation and characterization was studied. In general, fresh leachate during the summer had lower pH values than fresh leachate during the winter. The presence of dominating hydrolytic and acidogenic bacteria in the leachate resulted in a higher concentration of volatile acids during the summer, resulting in a lower pH of the leachate. COD concentrations mainly ranged from 15000 to 92000 mg/L. Therefore, seasonal variations directly affected fundamental characteristics of raw leachate, and it could provide a useful potential reference for fresh leachate bio-treatment.

Zhao et al. (2016) conducted a comprehensive laboratory test, and the biodegradation properties of MSW were studied. The waste collected from Chongqing landfill with representative organic content of 50% was prepared, and different constant temperature fields were considered by keeping water temperatures at different constant values (20, 25, 30, 35, 40, 45, and 50°C). The results indicated that 22 to 45°C was a more proper temperature range, favoring the degradation velocity of waste in the anaerobic phase.

Bhatt et al. (2017) investigated a laboratory-scale study with various waste compositions (0–100%), controlled temperature, and rainfall rates (2, 6, 12 mm/day) was conducted. The fresh waste is collected from different sources, and sludge is mixed with solid waste before feeding the reactor. Results indicated that 100% of food waste reactors of BOD_5 and COD concentrations were high because food is the most degraded material.

2.8 LANDFILL GAS GENERATION

Municipal solid waste (MSW) in landfills decomposes and produces methane (CH_4) and carbon dioxide (CO_2) gases, trace amounts of toxic substances, and bad odor, which are the by-products of the decomposition (Shin et al., 2005). Landfill gas is ranked as the third-highest source of global anthropogenic methane emissions, responsible for approximately 9-12% of those emissions in 2005 (Intergovernmental Panel on Climate Change (IPCC) (Change, 2007). In particular, the potential contribution of methane to global warming is 21 times higher than that of carbon dioxide (Gewald et al., 2012). Therefore, carbon emission reduction has become a global issue with the increase of methane and carbon dioxide emissions. India, one of the world's largest emitter of CH_4 from landfills, currently produces about 16 tons of CO_2 equivalent per year, which is predicted to increase to almost 20 tons of CO_2 equivalent per year by 2020 (Singh et al., 2016). CH_4 alone constitutes about 29% of the total GHG emissions in India, which is near twice the worldwide average of 15%. Moreover, the emission from wastes

is also twice (6%) than the global average of (3%) (Siddiqui and Khan, 2011). A major part of MSW constitutes biodegradable organic materials, which undergo an aerobic and anaerobic decomposition in landfills, generating various gases collectively called landfills gases (LFGs). Four phases of bacterial decomposition and the gases produced during each phase. Fig. 2.16 shows gas production at each of the four stages.

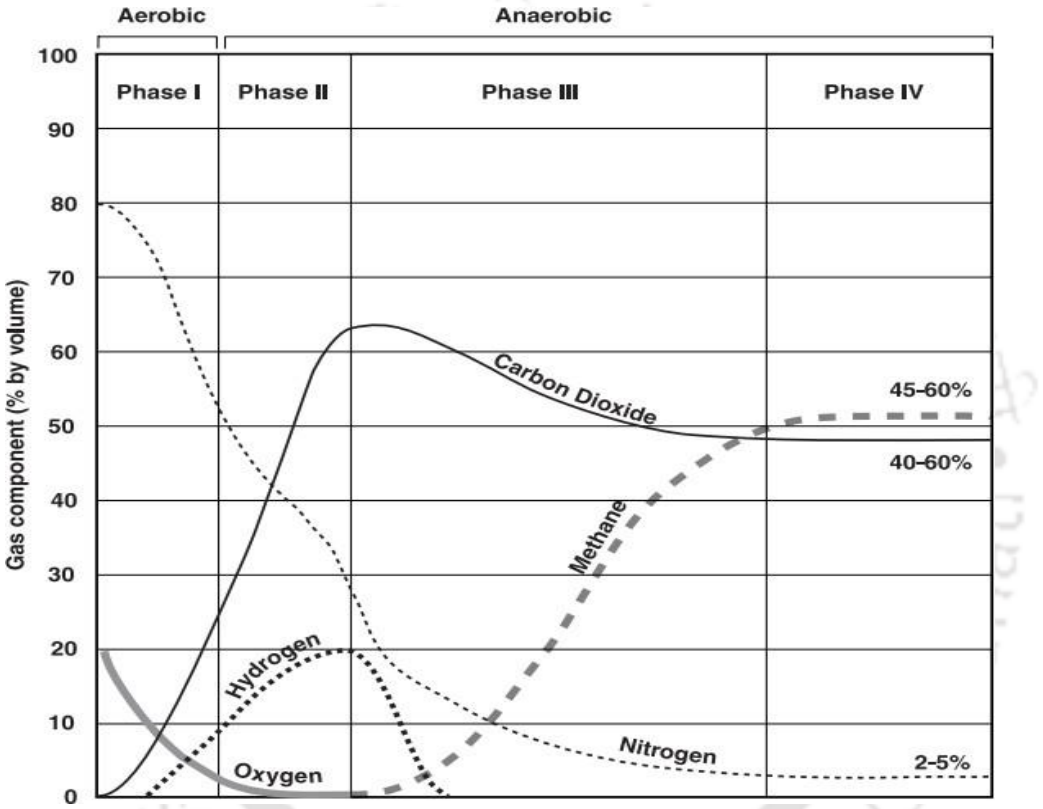


Fig. 2.16. production of gases at different phases (Change, 2007)

Note: phase duration varies with landfill conditions

Landfill gas is composed of a mixture of hundreds of different gases. By volume, landfill gas typically contains 45% to 60% methane and 40% to 60% carbon dioxide. Landfill gas also includes small amounts of nitrogen, oxygen, ammonia, sulfides, hydrogen, carbon monoxide, and non-methane organic compounds (NMOCs) such as trichloroethylene and benzene vinyl chloride. Table 2.8 lists "typical" landfill gases, their percent by volume, and their characteristics.

(i) The four phases of bacterial decomposition of landfill waste

Phase 1: Decomposition can last for days or months, depending on how much oxygen is present when the waste is disposed of in the landfill. Oxygen levels will vary according to factors such as how loose or compressed the waste was when it was buried.

Phase 2: As the acids mix with the moisture present in the land-fill, they cause certain nutrients to dissolve, making nitrogen and phosphorus available to the increasingly diverse species of bacteria in the landfill. The gaseous byproducts of these processes are carbon dioxide and hydrogen.

Phase 3: Acid-producing bacteria create compounds for the methanogenic bacteria to consume. Methanogenic bacteria consume carbon dioxide and acetate, too much of which would be toxic to the acid-producing bacteria.

Phase 4: Landfill gas usually contains approximately 45 to 60% methane by volume, 40 to 60% carbon dioxide, and 2 to 9% other gases, such as sulfides.

Table 2.8. Typical landfill gas components (Change, 2007)

Component	Percent by Volume	Characteristics
Methane	45–60	Methane is a naturally occurring gas. It is colorless and odorless. Landfills are the single largest source of U.S. man-made methane emissions
Carbon dioxide	40–60	Carbon dioxide is naturally found at small concentrations in the atmosphere (0.03%). It is colorless, odorless, and slightly acidic.
Nitrogen	2–5	Nitrogen comprises approximately 79% of the atmosphere. It is odorless, tasteless, and colorless.
Oxygen	0.1–1	Oxygen comprises approximately 21% of the atmosphere. It is odorless, tasteless, and colorless.
Ammonia	0.1–1	Ammonia is a colorless gas with a pungent odor.

NMOCs	0.01–0.6	NMOCs are organic compounds (i.e., compounds that contain carbon). (Methane is an organic compound but is not considered an NMOC.) NMOCs may occur naturally or be formed by synthetic chemical processes. NMOCs most commonly found in landfills include acrylonitrile, benzene, 1,1-dichloroethane, 1,2-cis dichloroethylene, dichloromethane, carbonyl sulfide, ethyl-benzene, hexane, methyl ethyl ketone, tetrachloroethylene, toluene, trichloroethylene, vinyl chloride, and xylenes.
Sulfides	0–1	Sulfides (e.g., hydrogen sulfide, dimethyl sulfide, mercaptans) are naturally occurring gases that give the landfill gas mixture its rotten-egg smell. Sulfides can cause unpleasant odors even at very low concentrations.
Hydrogen	0–0.2	Hydrogen is an odorless, colorless gas.

2.8.1 Laboratory scale simulation studies on landfill gas

Pohland et al. (1993) demonstrated three simulated landfill columns consisting of a 20-gauge steel cylinder, 1 m in diameter and 3.25 m in length. The cylinders were prepared with an internal coating of epoxy paint and equipped with a leachate collection/recycle system, volumetric gas meters, and lifting plates to facilitate the eventual removal and retrieval of the stabilized refuse and co-disposed diapers. The 627 kgs of shredded municipal solid waste and 20 diapers were used in each column. After loading, the columns were covered, and tap water was added (8.4 L/week) to simulate natural rainfall infiltration (1.27 cm/week). The mode of operation of the reactor was recirculation of leachate, and collection of leachate was once a week. The results indicated that initial adjustment and transition were evidenced by the initial production of large quantities of CO_2 and a subsequent period of low total gas production (Days 0 to 200), and hydrogen (H_2) was also detected in the gas phase (2%).

San and Onay (2001) constructed in the laboratory using two PVC pipe columns with a length of 1 m and a diameter of 0.35 m. The reactor was operated with and without recirculation and kept the constant temperature at 34°C. The landfill gas composition in the single-pass reactor (without recirculation) shows the initial stages of decomposition 30% of CO_2 produced while the methane produced 1-2% may be due to a high amount of VFA production in the reactor. As time increases, CO_2 increases to approximately 50%, while methane production is below 10% after 250 days due to lack of moisture and the leaching process carrying out the nutrients and carbon from the waste.

Sponza and Agdag (2004) constructed a cylindrical reactor of 30 cm in diameter and 100 cm in height. The reactor has an electronic heater to provide a temperature of 35°C and was hermetically sealed with a gasket and silicone sealant to maintain anaerobic conditions. A single-pass reactor was operated without leachate recirculation, while the other two reactors were operated with leachate recirculation. The reactor contained 76% of organic waste, and operation times are 222 days. The key findings of the study were methane content was approximately 10–25% in the single-pass reactor. The gas production rate of the single-pass reactor was small peaks around the 11^{th} day may be related to the fermentation of easily biodegradable organics in solid wastes. Therefore, the low methane production considered additional evidence of washout, indicating the mobility of the system to develop an active methanogenic population and enhance waste stabilization. The recycled reactors exhibited a steady pattern of methane productions with a gradual increase in methane percentage compared to the single-pass reactor. The increase in VFA concentrations caused a decrease in the COD concentrations removed in leachate samples.

Mahar et al. (2007) studied the pretreated mixed un-shredded MSW by the natural convection of air was employed to assess the aerobic pretreatment suitability. The nature of the mixed MSW contains about 60% of organic content, and the remaining part was dry waste. The operation of the reactor was controlled temperature of 35-40°C and pre-treated MSW. The gas analysis was done for CH_4 and CO_2. Carbon dioxide decreases to the range of 39 to 40%. As pH increases, the proportion of methane also increases; at pH 7, methane proportion was about 35%, and at pH 8, methane proportion was reached 50%.

Swati et al. (2011) fabricated four pilot-scale lysimeters of 3.0 m height and 1.3 m diameter were constructed outdoors using reinforced cement concrete rings. Reactor 1 (R1): Filled with

fresh MSW to simulate a young controlled dump without leachate recirculation. The biodegradable fraction of fresh MSW consisted of a major portion of food waste (46%), and the dry part was 54%. The controlled dumps R1 shows 16-35% of methane content was generated, and a similar trend was obtained for CO_2. Therefore, smaller methane concentrations in the controlled dumps are considered to be proof of washout, indicating the inability of the system to develop an active methanogenic population to degrade the substrate to give biogas rich in methane anaerobically.

Zhao et al. (2016) studied a comprehensive laboratory test was performed, and the biodegradation properties of MSW were studied. The reactor operation was controlled temperature and recirculation of leachate. The waste collected from Chongqing landfill with representative organic content of 50% was prepared, and different constant temperature fields were considered by keeping water temperatures at different constant values (20, 25, 30, 35, 40, 45, and 50^{o}C). The percentage of carbon dioxide in the biogas reached a peak value of 38% during the initial stages.

2.9 STATISTICAL MODELING OF LEACHATE PARAMETERS

Past studies have reported modeling leachate constituents (Wigh, 1979; Daskalopoulos et al., 1998; Gau and Chow, 1998; Gonollu, 1994;). In past studies, two main modeling methods have been used. The first simply involves fitting empirical formulae to the different concentration curves of the leachate (Wigh, 1979). The second modeling method is complex, including the biological process in landfills (Gonullu, 1994). Lab-scale data of COD from reactors has been simulated using the model for estimating COD. The study exerted that mass transfer, substrate utilization, and microbial mass production have been considered to develop an equation, but no goodness of fit was reported (Gonollo, 1994). Two models were developed for 'with and without' leachate recirculation conditions. COD concentration was modeled with correlation coefficients 0.90 and k_1, and k_2 parameters were found to be 0.0027 and 0.0055 day-1 (Kouzeli-Katsiri et al., 1999). Another study compared model equations used to relate leachate concentrations with time (Rowe, 1995). In general, practically every researcher formulated their modeling equation and incorporated the factors they considered were most significant or relative to their experiment conducted. The following studies were selected from the literature to explain various modeling approaches based on laboratory data.

Lu et al. (1985) attempted to establish relationships between landfill age and leachate constituents. This study used data from over 50 years of field cell tests to develop relationships based on the first-order rate equation 2.13.

$$C = C_o 10^{-kt} \quad (2.13)$$

where:

C is the pollutant concentration, C_o is the initial pollutant concentration, k is the first-order reaction rate constant, and t is time. Values of the reaction rate constant, k, for BOD and COD were found to be 0.043 and 0.045 yr^{-1}, respectively.

Gonullu (1994) developed a chemical oxygen demand (COD) model in leachates from solid waste disposal areas. The model contains dissolution/dilution, mass transfer, substrate utilization, and microbial mass production processes. Three columns comprised of Plexiglas columns 0.30 m in diameter and 1 m in height. The mixtures of wastes were obtained by grinding each waste material group one by one in a coarse grinder (diameter~2 cm) and mixing completely and equation 2.14.

$$C(t) = e^{-(t)}\left\{C_o + \oint (R + R_1 + R_2)e^{-G(t)}dt\right\} \quad (2.14)$$

The model takes into account processes like dilution (G), mass transfer (R), substrate utilization (R_1), and microbial mass production (R_2). However, one of the model weaknesses is that it requires the input of assumed values for some parameters such as maximum COD value (C_0), mass transfer rate constant, yield coefficient, leachate velocity constant, and others. In addition, although the author provided graphs of good COD simulation curves, no statistical values for the goodness of fits were reported.

Rowe (1995) modeled strength of leachate parameters decreases with time. This decrease may result from the biological breakdown of organics into simpler compounds or the precipitation of soluble compounds. It has been recognized that the decline in concentration, c with time, t can be empirically by the first-order rate equation 2.15.

$$C(t) = C_o e^{-kt} \quad (2.15)$$

where:

C is the pollutant concentration, C_o is the maximum/initial pollutant concentration, k is the first-order reaction rate constant, and t is time.

Kouzeli-Katsiri et al. (1999) fabricated six lysimeters made of PVC. The lysimeters were 2.5 m high with an internal diameter of 0.30 m and were insulated externally with a 3.5 cm thickness of polyurethane foam. The wet composition of 59.8% and the dry part was 40.2%. Two models were developed with and without leachate recirculation data from six lab-scale lysimeters and verified using actual landfill data from the literature. The models involve two main processes: the exchange of organic matter between solid and liquid phases and the depletion of dissolved organics due to biodegradation and flushing and equation 2.16.

Model 1: without recycling

$$C = k_1 M_{so}\left(\frac{1}{Q + k_2 v - k_1 v}\right)\left(e^{-k_1 t} - \exp\left[-\left(\frac{Q}{V} + V_2\right)t\right]\right) \quad (2.16)$$

where:

C (g/L) is COD concentration in leachate, k1 (yr^{-1}) is the rate constant of the decrease of organic matter in solid phase due to solubilization, k2 (yr^{-1}) is the rate constant of biological decomposition of biodegradable COD in the liquid phase, M is mass of COD in the liquid phase, Mso (g) is the initial mass of leachable COD in the solid phase, Q (L^3/yr) is leachate flow rate, t (yr) is time, and V (L^3) is reactor volume. Simulations of lab data and data from literature gave very good fits with correlation coefficients greater than 0.90 in all cases. Therefore, the authors suggested using k1 and k2 value ranges of 0.0027 to 0.0055 day^{-1} and 0.0016 to 0.0027 day^{-1}, respectively, for new landfills.

Bhatt et al. (2016) preliminary investigation models for estimating first-order rate constants for removing BOD_5 and COD from landfill leachate were studied. The experimental condition was MSW mixed with sludge and controlled temperature to enhance the degradation. In this research, the interest was in the declining part of the BOD_5 and COD curves and modeled using first-order reaction kinetic in equation 2.16.

$$C(t) = C_o e^{-kt} \quad (2.16)$$

where:

C is the pollutant concentration, C_0 is the maximum/initial pollutant concentration, k is the first-order reaction rate constant, and t is time.

BOD ($k= 0.026\ day^{-1}$ & $R^2= 0.89$) and COD ($k= 0.028\ day^{-1}$& $R^2= 0.964$) for reactor contained 60% of food waste.

2.10 LANDFILL LEACHATE TREATMENT

The leachate treatment system's systematic approach may be complex because of the high variability of characteristics and selection methods based on leachate properties (Tchobanoglous et al., 1993). In general, leachate treatment is usually performed and requires a combination of treatment technologies. Most of the processes developed for the treatment of water and wastewater has been adapted to the treatment of landfill leachates includes: (i) biological processes like aerobic, anaerobic, and nitrification/denitrification to organic and nitrogen compounds (Pelkonen et al., 1999; Galvez et al., 2006; Wei et al., 2012; Tałałaj et al., 2019). The physico-chemical process is effective in removing suspended solids, biological parameters (Aziz et al., 2007; Rui et al., 2012; Xu et al., 2017), integrated membrane system with coagulation-nanofiltration (Smol and Włodarczyk-Makuła, 2017), chemical oxidation process (Derco et al., 2010). Coagulation -flocculation process (CFs) has been widely used as pretreatment and post-treatment to remove leachate parameters like color, suspended solids, high concentrations of organic pollutants before other biological treatments (Kamaruddin et al., 2017). Coagulation is a cost-effective way to treat leachate using aluminum and other compounds (Huda et al., 2017). Studies of the CFs practiced to MSW leachate (Ehrig, 1984; Diamadopoulos, 1994; Sletten et al., 1995; Aziz et al., 2007; Smol and Włodarczyk-Makuła, 2017). Iron-based coagulants are used to treat old leachate, poly ferric sulfate (PFS) compared with ferric chloride 6-hydrate ($FeCl_3 \cdot 6H_2O$) and ferric sulfate 7-hydrate ($FeSO_4 \cdot 7H_2O$), with an initial COD concentration of 10623 mg/L was found to be higher removal of organic compound 56.38% for PFS, 68.65% for ferric chloride 6-hydrate and 55.87% for ferric sulfate 7-hydrate. Therefore, appreciable COD removal specified that CFs as a pretreatment could remove refractory compounds in landfill leachate (Liu et al., 2012). Current common treatment methods used for leachate treatment are either single or combined methods and are classified into three main categories: biological processes (aerobic or anaerobic), chemical and physical processes, and a combination of Physico-chemical and biological processes. Independently,

each treatment process has its advantages and disadvantages (Zhang et al., 2013). Strong knowledge of chemistry and processes involved is necessary to choose the appropriate treatment methods before the characteristics of the wastewater and the treatment objectives. Coagulation -flocculation is known as one of the oldest treatment methods in landfill leachate. Apart from that, it has been widely used in treating stabilized (Al-Hamadani et al., 2011) and matured landfill leachate (Vedrenne et al., 2012). The characteristics of the landfill leachate can usually be represented by the basic parameters COD, BOD_5, the BOD/COD ratio, pH, suspended solids (SS), ammonium nitrogen (NH_4^+-N), total Kjeldahl nitrogen (TKN), and heavy metals (Renou et al., 2008). Based on the target of the specific content, all the available options for leachate treatment were mentioned in Table 2.9.

Table 2.9. Landfill leachate treatment options for specific pollutants (Johannessen, 1999)

Treatment Objectives	Main Treatment Options
Removal of degradable organic (BOD_5)	Aerobic biological Aerated lagoon/extended aeration Activated sludge Sequencing batch reactor (SBR) Anaerobic biological: Up-flow sludge blanket
Denitrification	Anoxic biological Sequencing batch reactor Vegetated ditch (artificial wetlands) Ammonia stripping
Removal of non-degradable organic and color	Lime / coagulant addition Activated lagoon Reverse osmosis Chemical oxidation

Removal of hazardous trace organic	Activated carbon Reverse osmosis Chemical oxidation
Removal of dissolved iron and heavy metals and suspended solids	Lime/coagulant addition, aeration, and setting
Final polishing	Artificial wetlands (e.g., reed beds, ponds)
Disinfection	Hypochlorite
Volume reduction / pre-concentration	Reverse osmosis evaporation

2.10.1 Coagulation-flocculation studies

A study was conducted in the Merida landfill leachate with low suspended solids. Ferric sulfate indicated higher COD removal values (>40%) at optimum dosage (300 mg/L) and pH (<2) (Mendez-Novelo et al., 2005). In water and wastewater, alum, ferric chloride, ferrous sulphate were commonly used (Ehrig, 1984; Amokrane et al., 1997). Compared with aluminum salts, iron salts are more efficient with a lower dosage and a wider range of pH (Amokrane et al., 1997; Maranon et al., 2008; Li et al., 2010). Furthermore, aluminum salts have been suspected to be carcinogenic, mutagenic, and more toxic to fishes than organic complexes. Moreover, overdosage of aluminum leads to pollution of surface and groundwater (Srinivasan et al., 1999), and iron salts are less health risk than aluminum salts (Cheng and Chi, 2002). The coagulation-flocculation process conducted for the treatment of stabilized leachate and ferric chloride was used. The removal efficiency of COD was 76.9% at pH 6 (Ishak et al., 2018). Different coagulants were used to treat the old leachate, i.e., ferric chloride, aluminum sulphate, aluminum polychloride (PAX), and polyacrylamide polyelectrolytes. Ferric chloride exhibited better COD efficiency (73%) with an actual COD of 4814 mg/L than other coagulants (Maranon et al., 2010). Ferric chloride is the best coagulant for treating stabilized landfill leachate with variable dosage and pH since the strength of leachate varies with time and the composition of solid waste, and changes in dosage and pH of coagulants also occur. (Gandhimathi et al., 2013; Yadav and Dikshit, 2016; Kumar and Bishnoi, 2017; Verma and Kumar, 2018). Coagulation-flocculation studies are shown in Table 2.10.

Table 2.10. Application of various coagulants in landfill leachate treatment

S.No.	Type of Coagulant	Optimum dosage & pH	Initial Characteristics (parameter concern)	Removal efficiency	Remarks	References
1	Lime	6 g/L	pH: 5.8 Turbidity: 130 NTU COD: 22900 mg/L	Turbidity: 98% COD: 10%	High strength industrial landfill leachate	(Slater et al., 1983)
2	Iron, Aluminum,	15 mmol/l & 3.5, 5 mmol/l & 5.6	pH: 7.5 COD: 1140 mg/L	Iron (COD: 56%) & Aluminum (COD: 39%)	Stabilized leachate	(Diamadopoulos, 1994)
3	Bioflocculant	NM	pH 7.5, COD 750 mg/l	COD : 30%	Stabilized leachate	(Zouboulis et al., 2004)
4	Ferric chloride, ferric sulphate, aluminum polychloride, and aluminum sulphate	Ferric sulphate (dosage: 0.3 g/L & 2)	pH: 8.40 , COD: 5740 mg/L, TSS:342 mg/L	COD: 40%,	Stabilized leachate	(Méndez-Novelo et al., 2005)

5	Ferric chloride ($FeCl_3$), Aluminum sulphate ($Al_2(SO_4)_3$), Aluminum polychloride (PAX)	0.5 g/L & 3.8, 0.5 g/L & 6, 6 g/L & 8.3	COD: 18500–20000 mg/L, pH: 8.2–8.3 Turbidity >4000 NTU	Ferric chloride (COD: 28.1%, Turbidity: 90.2%), Aluminum sulphate (COD: 27.1%, Turbidity: 93.1%) , polychloride (COD: 36.8%, Turbidity: 96.4)	Young leachate	(Marañón et al., 2008)
6	$Al_2(SO_4)_3$, $FeCl_3$, PACl, and poly ferric sulfate (PFS).	6 g/L & 5.5, 0.6 g/L & 6, 0.5 g/L & 5.5	COD: 2817 mg/L, turbidity: 374 NTU, suspended solids (SS): 676 mg/L	$Al_2(SO_4)_3$ (COD: 68%, turbidity: 97%, suspended solids (SS): 93%), $FeCl_3$ (COD: 53%, turbidity: 95%, suspended solids (SS): 87%), poly ferric sulfate (COD: 56%, turbidity: 99%, suspended solids (SS): 93%	Stabilized leachate	(Li et al., 2010)
7	Calcium hydroxide and Alum	25 g/L & 8, 15 g/L	pH: 6, COD:2451 mg/L, TSS: 533 mg/L	Calcium hydroxide (COD: 69%, turbidity : 99%), Alum (COD: 54%, turbidity : 94%)	Stabilized leachate	(Shabiimam and Dikshit, 2011)

8	Alum and $FeCl_3$	7.6 g/L & 5.15, 10.9 g/L & 7.05	PH: 7.4. COD: 16896 mg/L	Alum (COD: 55%), $FeCl_3$ (COD: 35%)	Fresh leachate	(Gandhimathi et al., 2013)
9	Polyaluminium chloride (PACl) and $FeCl_3$	7 g/L & 8, 10 g/L & 7	pH: 8, COD:2340 mg/L	PAC (COD: 59%), $FeCl_3$ (COD: 51%)	Stabilized leachate	(Yadav and Dikshit, 2016)
10	$FeCl_3$	3 g/L & 8	pH: 8.68, COD: 9600 mg/L	COD: 71.3%	Stabilized leachate	(Kumar and Bishnoi, 2017)
11	Alum	3.8 g/L & 6	Stabilized leachate (pH: 7.8, COD: 3850 mg/L), Municipal wastewater (pH: 7, COD: 500 mg/L	COD, turbidity, and TSS removal were 80, 88, and 81%, respectively.	Municipal wastewater with stabilized leachate (ratio of 1:5)	(Verma and Kumar, 2018)

2.10.2 Biological treatment (upflow anaerobic treatment)

Biofilm, or fixed film, reactors depend on the natural tendency of mixed microbial populations to adsorb to surfaces and collect in biofilms. Adsorbed bacteria thrive, reproduce, and create extracellular polymeric compounds that frequently spread from the cell, forming the gelatinous matrix called a biofilm. A polymeric material mediates the bacterial attachment, primarily polysaccharides, which extend from the cell to form a tangled mass of glycocalyx fibers. The entire deposit is called a biofilm (Jimeno et al., 1990). The packing is fixed in the upflow anaerobic packed-bed reactor, and the wastewater flows up through the interstitial spaces between the packing and growth. While the first upflow anaerobic packed-bed processes contained rock, various synthetic plastic packing designs are currently used. A large portion of the biomass responsible for treatment in the upflow attached growth anaerobic processes is loosely held in the packing void spaces and not attached to the packing material (Bodik et al., 2000a). An up-flow anaerobic filter is a contact unit in which wastewater passes through a mass of biological solids contained inside the reactor by a support medium. The biomass is contained in the reactor, by

1) biomass attached to the support media's surface as a thin biofilm;
2) biomass entrapped within the media matrix; and
3) biomass held as a granulated or flocculated sludge mass beneath the media.

Researches have indicated that the start-up of anaerobic reactors is more time-consuming and is subjected to disturbances more than that of aerobic reactors. The startup of the anaerobic process is still considered a major area of research. Many researchers have reported long start-up periods of 2-3 months to 1 year (or even more) for the anaerobic reactors (Ramakrishnan and Gupta, 2006). Low up-flow velocities are generally used to prevent washing out the biomass, as mentioned by (Kavitah, 2009). Authors have reported that the C: N:P ratio of 100:2:1 is optimal for start-up anaerobic fixed-film reactors (Jimeno et al., 1990).

2.10.2.1 Advantages of upflow anaerobic filter

The advantages of the anaerobic filter (an attached-growth system) over a suspended-growth anaerobic high-rate reactor are as follows:

- Biofilm reactors are especially useful when slow-growing organisms have to be kept in wastewater treatment (Bodik et al., 2003)
- It has relatively good load fluctuation resistance (Dahab, 1982)
- Due to efficient biomass retention, long sludge ages, and more compact

reactors can easily be achieved (Bodík et al., 2003; Kobayashi et al., 1983).

- Unlike the anaerobic activated sludge process (Bodik et al., 2000b). Therefore the cost of energy for sludge returning is not necessary.

2.10.2.2 Disadvantages of upflow anaerobic filter

However, the anaerobic filter has some drawbacks too. The disadvantages of the anaerobic filter are as follows (Bodik et al., 2000b)

- Channeling can occur, i.e., the formation of preferential paths of liquid flow through the reactor.
- Dead-zone formation is caused by sludge compaction or clogging of interstitial matrix spaces by solids.
- Clogging due to poorly designed distribution systems.

2.10.2.3 Packing media

Several types of materials have been used as packing media in biological reactors, including quartz, ceramic blocks, oysters and mussel shells, limestone, plastic rings, hollow cylinders, PVC modular blocks, granite, polyethylene balls, and bamboo. The packing media have been designed to occupy from the total depth of the reactor to approximately 50 to 70% of the height of the reactor. Different types of plastic packing media are available in the market, ranging from corrugated rings to corrugated plate blocks. The specific surface areas of these plastic materials usually range from 100 to 200 m^2/m^3 (Song and Young, 1986). However, some types of packing media are more efficient than others in biomass retention. The final choice will depend on the specific local conditions, economic considerations, and operational factors (Young and Yang, 1989). Authors have indicated that specific surface area, porosity, surface roughness, pore size, and orientation of the packing material were important factors influencing the anaerobic filter reactor performance. High surface area and porosity, large pore size, and rough surface area for packing material improved performance of an anaerobic filter (AF) reactor small-medium are employed AFs may suffer from blockages and to minimize blockages, filter media tend to have relatively large diameters (>20 mm) (Elmitwalli et al., 2000).

2.10.3 Anaerobic filter studies of landfill leachate

Studies have been conducted on anaerobic treatment of young landfill leachate and stabilized leachate because of the advantage of low energy consumption, low amount of sludge generation, and high performance in high strength wastewaters (Gourari and Achkari-Begdouri, 1997; Henry et al., 1987b; Kennedy et al., 1988; Wu et al., 1988; Henry

and Prasad, 2000; Inanc et al., 2000; Wang and Banks, 2007; Luo et al., 2020). Clogging problems in the operation of anaerobic filters due to suspended solids (SS) in wastewater have been found by several investigators and have suggested that pretreatment is required before anaerobic treatment, and SS control is essential (Elmitwalli et al., 1999; Jahren et al., 1999; Show and Tay, 1999; Ghangrekar and Kahalekar, 2003). The stabilized leachate (COD 3,750 mg/L) and new landfill leachate (COD 14,000 mg/L) are treated with anaerobic filters having gravel media to remove COD, and removal efficiency was obtained 90% with an OLR of 1-2 kg/m^3 d. Therefore, studies that showed that high loading rates (>5 kg COD /m^3 d) of COD removals decrease substantially (Crawford et al., 1980; Hall et al., 1982; Young, 1983).

Henry et al. (1987a) demonstrated a laboratory-scale study using anaerobic filters on stabilized leachate (COD 2,000mg/L). Therefore, the removal efficiency was achieved 90% at 10^oC with a 72-hr detention time.

Wu et al. (1988) studied the treatment of high strength leachate treatment by anaerobic filters with an initial COD of 15000 mg/L, and removal efficiency was 50-60% achieved at Organic loading rate (OLR) 0.7-2.8kg/m^3. However, at an OLR of 6 kg/m^3, the removal efficiency was 27%. The OLR up to 4 g COD/day m^2, the removal efficiency was 90% with an initial COD of 17,450 mg/L and a final value of 1560 mg/L. The plastic media was used to form biofilm and remove pollutants in the landfill leachate.

Henry and Prasad (2000) investigate the effectiveness of the sulfate-reduction pathway in the anaerobic treatment of landfill leachate. Low strength leachate (COD =1,500-3,300 mg/L) was collected from the Brock West Landfill. About 60% COD removal was achieved at a loading of 2.8 kg COD/m^3d and a COD/SO_4^{2-} ratio of 1.0. However, at a loading of 6 kg COD/m^3 d, only 27% COD removal was achieved, all of it through the sulfate-reduction pathway. From this study, it can be concluded that there is no advantage to the sulfate-reduction pathway in the anaerobic treatment of landfill leachate.

Wang and Banks (2007) Sulfate-rich alkaline landfill leachate was treated with an anaerobic filter, and $Fecl_3$ was used as a reactant and sludge as seeding to the leachate. Therefore, the removal efficiency was between 75% and 90%, with an OLR of 0.76-7.63 kg COD /m^3 d.

2.11 INFERENCE FROM THE LITERATURE REVIEW

MSW is disposed of in an unscientific manner in low-lying regions or open dumps in Indian cities without enough measures or operational controls, resulting in negative effects

on all environmental and human health components. Solid waste management is a major issue in developing countries like India because the best and most appropriate methods, from waste collection to disposal, are not being used and insufficient waste collection, poor door-to-door collection, and poor management of waste lead to increasing numbers of open dumpsites; people living near such dumpsites are affected by poor water and air quality. Life cycle assessment of waste management is one of the key analyses for sustainable waste management. LCA approach for comparing environmental impacts by different waste management alternatives and determining the feasible waste management scenario with minimum impacts can be assessed. As the literature says, similar waste management options may not be suitable for other regions/countries because the waste composition and climatic conditions differ from region to region and city to city. It must be noted that the biodegradation factors were computed by researchers using lab-scale anaerobic degradation studies (e.g., lab-scale simulated landfills), thereby creating "ideal" conditions for biodegradation through shredding, controlling pH, or optimizing the C/N ratio. These factors can help compute the maximum amount of leachate quantity and quality generated from solid waste under anaerobic conditions. However, in practice, the actual conditions may differ from ideal conditions, so the uncertainty of data analysis is introduced. Moreover, no leachate quality and quantity data are available on the high amount of organic waste contained in landfills. To date, various studies have been carried out based on the waste composition of the country/region, sludge addition, and controlled temperature environment to understand the degradation behavior of solid waste landfills. But very few studies are available on actual landfill operational conditions on the degradation of fresh solid waste in an active phase. The seasonal impacts on leachate quality provide a preliminary basis for treatment process options and operation of the fresh landfill leachate. In general, leachate treatment is usually performed and requires a combination of treatment technologies. The leachate treatment system's systematic approach may be complex because of the high variability of characteristics and selection methods based on leachate properties. Very limited studies were carried out on the treatment of fresh landfill leachate using coagulation-flocculation as a pretreatment. No studies are available on the application of upflow anaerobic filters in treating fresh landfill leachate (acidic condition).

CHAPTER

3

Research Objectives

This chapter mainly deals with the knowledge gap, research objectives, the need of the study, the scope of the study.

3.1 KNOWLEDGE GAP

- Most of the LCA studies conducted in India focused on megacities such as Bangalore, Mumbai, and Delhi, but no single research was conducted in the northeastern part of India. Technologies recommended for megacities may not be optimal for application in other cities and regions.
- The previous research studies were conducted on a global perspective, i.e., biologically/pre-treated waste, recirculation of leachate to enhance the biodegradation, partially degraded waste, release behavior of specific leachate parameters from MSW with different operational modes of the reactor, interactions between recirculated leachate and the stabilized waste and addition of sludge to the solid waste.
- The studies in India have focused on the recirculation of leachate with water addition and controlled temperature conditions. None of the studies have carried out the research based on actual conditions (no addition of sludge and uncontrolled temperature) of the landfills.
- The previous research studies were conducted on a global perspective, i.e., focused on stabilized leachate using the coagulation-flocculation process as pretreatment and anaerobic filters as a secondary treatment, sludge addition to leachate to increase the removal efficiency. No studies in India have focused on fresh/young leachate of age less than one year and having a high concentration of organic pollutants, COD, etc. This study attempts the much-needed multistage treatment of such leachate through coagulation-flocculation followed by an upflow anaerobic filter. No single study is

associated with both the pretreatment method and biological treatment. To achieve the above research gaps, detailed research has been conducted.

3.2 RESEARCH OBJECTIVES

The improper management of solid waste in India resulted in the uncontrollable generation of gases and leachate from the dumpsites. Moreover, waste composition, temperature, and moisture content have large impacts on the degradation of waste. It is difficult to generalize the chemical composition of leachate, as leachate varies over time and depends on waste composition and climatic conditions. The broad research objective is to study the influence of tropical conditions on the degradation of MSW in real-time scenarios of open dumpsites in India and the feasibility of treating fresh leachate from the high amount of organic waste continued MSW landfill using a combined system, i.e., the coagulation-flocculation process followed by an upflow anaerobic filter. The entire research work is divided into three phases and five objectives as follows:

- **Objective-I:** Identification of possible environmental impacts of different municipal solid waste treatment scenarios on various impact categories and life cycle cost of each scenario for the study area of Guwahati City using the LCA approach.
- **Objective-II (Part-I):** Comprehensive characterization of fresh landfill leachate and landfill gas composition from the unshredded mixed MSW dumped in a lab-scale landfill simulation reactor and investigation of alterations of leachate quality with time **(without the addition of rainfall).**
- **Objective-III (Part-II):** Comprehensive characterization of fresh landfill leachate and landfill gas composition from the unshredded mixed MSW dumped in a lab-scale landfill simulation reactor and investigation of alterations of leachate quality with time **(with the addition of rainfall).**
- **Objective-IV:** Feasibility of treating fresh landfill leachate from the high amount of organic waste in MSW landfill using a combined treatment system, i.e., the coagulation-flocculation process followed by an upflow anaerobic filter. (a) Comparing the removal efficiency of different coagulants and their optimum dosages and (b) Treatability performance of high organic load with an upflow anaerobic filter having support media of plastic rings.
- **Objective-V:** Biomethane potential test (BMP) (1 L capacity) with different ratios of digested cow dung (DCD) as inoculum, i.e., 10, 20, 40, 50 & 80%, to attain the best ratio for microbial addition to the treatment of fresh landfill leachate.

3.3 NEED OF THE STUDY

Solid waste management (SWM) is one of the most challenging tasks in developing nations like India. Municipal solid waste (MSW) is composed of biodegradable, non-biodegradable, and inert components. The source segregation is negligible or non-exist in developing nations like India (Matter et al., 2013). Inventory estimates of CH_4 emissions from landfills have large uncertainties due to inadequate MSW management and emissions data availability. During the cradle to grave process, MSW passes through various stages such as sorting recyclable and compostable materials before final disposal to landfills. These stages may change the quantity and properties of waste, ultimately reaching the landfill sites, thus influencing GHGs emissions and leachate quality. The potential generation of gases depends on waste composition and the rate of waste degradation (Kim and Townsend 2012). This creates environmental and health hazards along with social problems. Generation of leachate develops as rainwater passes through the waste, contaminants dissolve in the liquid and leach out from the landfill. Therefore Contaminated leachate is high-strength wastewater containing high pollution levels like BOD_5, COD, ammonium nitrogen, etc. (Fuller, 1978; Koshy et al., 2007). In landfills, highly polluted leachate and landfill gases are produced during the waste degradation process through physical, chemical, and biological reactions and are considered the most threatening by-products of the landfill. Therefore, its management is the most critical issue addressed in landfills design and operation (Abunama et al., 2018). This kind of study is needed to comprehensively understand the behavior of high organic waste landfills for waste stabilization as a function of waste composition, the actual temperature inside the waste, ambient temperature, moisture content, and pH.

3.4 SCOPE OF THE STUDY

Presently, unscientific management of solid waste landfills leads to leachate and gas generation in landfills/open dumpsites. The pollution from landfills differs with different solid waste compositions and waste age. Firstly, the collection of fresh solid waste is a challenging task as mixed solid waste contains paper that is absorbent and quickly absorbs nutrients from the food waste. To avoid such mixtures, individual waste is collected from different sources. The fabrication of reactors, the experimental setup for the physico-chemical and biological analysis, learning the protocols, and operation of instruments were executed. Furthermore, the rigorous monitoring of simulated landfill systems,

characterization of landfill leachate, and gas composition with time is necessary to understand the behavior of fresh solid waste degradation in landfills under actual landfill conditions and landfill leachate treatment to avoid environmental externalities.

CHAPTER

4

Materials and methods

Different experimental approaches were used to accomplish the stipulated objectives. The research work was carried out in different phases using various solid waste material combinations. The detailed methodology is given in this chapter

4.1 EXPERIMENTAL FLOWCHART

In order to accomplish the objectives, the study was proposed to carry out in different phases, as summarized below. Fig. 4.1 shows the experimental design of the thesis work. In **Phase-I**, the primary and secondary sources were collected from the municipal board of Guwahati city and other sources of solid waste management to evaluate the environmental impacts and economic analysis of proposed scenarios for the solid waste management of Guwahati city. In **phase-II**, the 59 cities of solid waste composition data obtained from the CPCB, later design of experiments was used to optimize the data. Initially, two different solid waste compositions were selected based on high wet to low dry (represents developing countries) and high dry to low wet waste ratio composition (represents developed countries) collected from the optimized data. Wet and dry fresh waste was collected from different sources, and a $1m^3$ landfill simulated reactor was fabricated to simulate the behavior of waste degradation under the actual landfill conditions. The experimental design was divided into two parts, i.e., part-I simulated without rainfall and part-II simulated with rainfall under an anaerobic condition to assess the seasonal influence of tropical conditions on the degradation of solid waste. Landfill leachate and gas were collected once a week to assess the degree of biodegradability of solid waste. In **phase-III**, treatment of young/unstabilized leachate using chemical treatment followed by anaerobic treatment to evaluate the removal efficiency of organic matter. A metagenomics study was

carried out to obtain the microbial diversity in raw and pretreated leachate. BMP study was performed to attain the best ratio for microbial addition to the treatment of fresh landfill leachate.

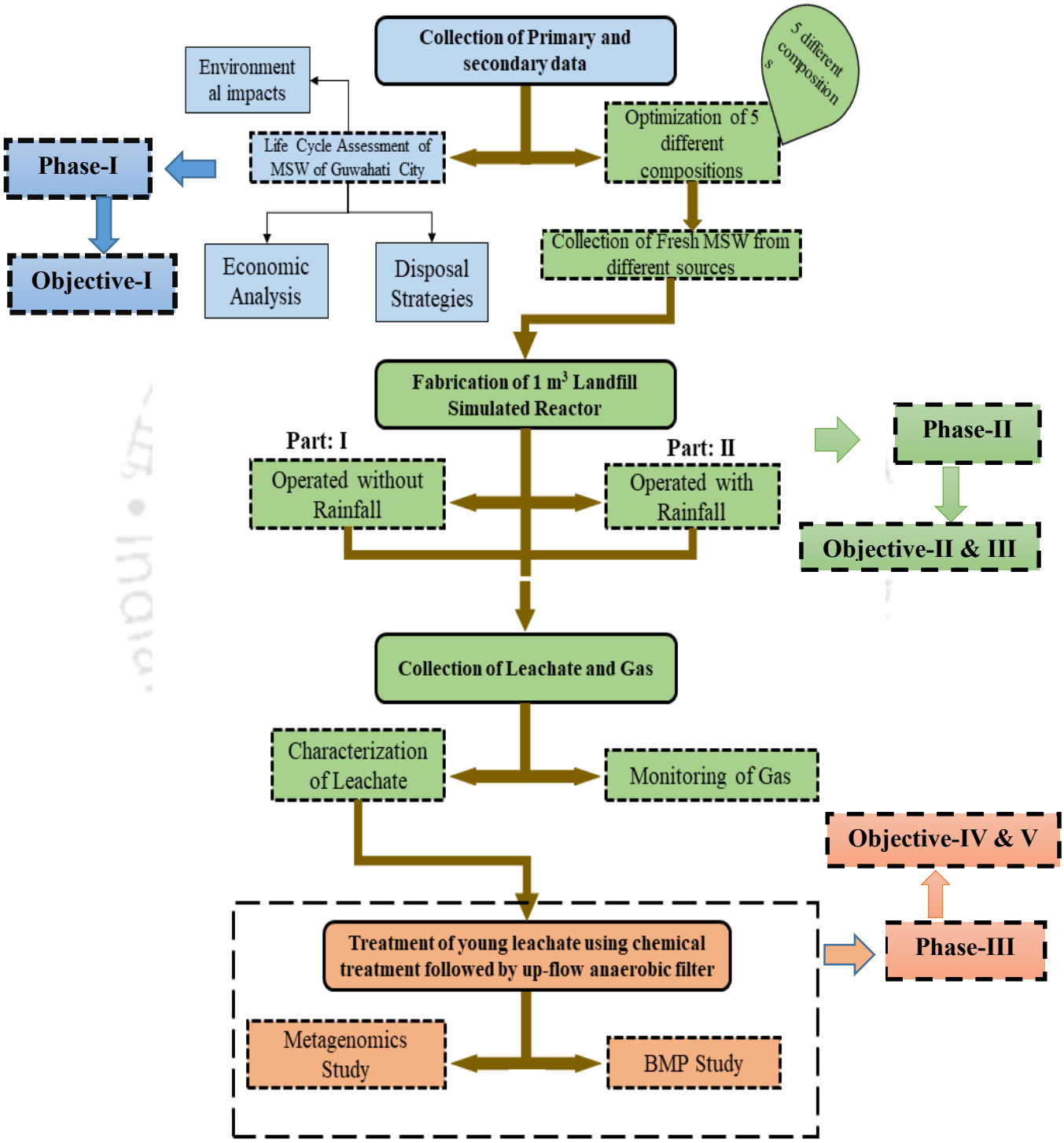


Fig. 4.1. Experimental flow chart

PHASE-I (OBJECTIVE-I): PRIMARY AND SECONDARY DATA ACQUISITION FROM THE MUNICIPAL BOARD AND OPEN DUMPSITE OF GUWAHATI CITY

4.2 STUDY AREA AND COMPOSITION OF MSW

Assam is well-known for its one-horned rhinoceros and an Indian state and functions as the entry point to northeastern India. Guwahati is the state's primary commercial capital and has a population of 9.69 lakh people. The city is located between 26.1445°N, 91.7362°E. The area covered all 31 wards administratively, and the area is around 216 km^2 (Chandramouli and General, 2011), as shown in Fig. 4.2. About 15-20 kg of waste was gathered from the overall amount of waste, which was thoroughly mixed and then quartered before samples of a size that could be treated in the laboratory. The composition of MSW generated in Guwahati city is 37.42% food waste, 17.44% plastic, 16.41% paper, 4.94% textiles, 4.14% glass, 1.97% leather, 0.45% rubber, 0.37% metals, 5.25% lawn, 2.45% wood scraps and 9.16% miscellaneous. The major portion of solid waste generated in Guwahati is organic, plastic, and paper.

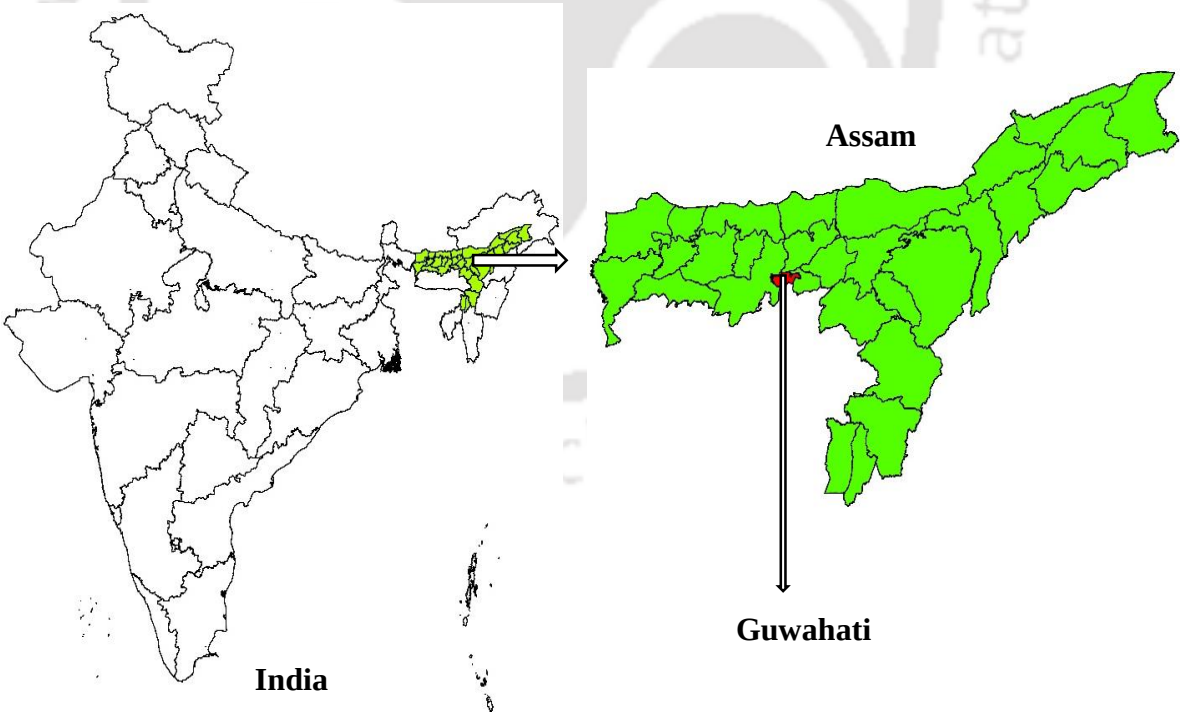


Fig. 4.2. Study area and map of Assam

4.2.1 Chemical characteristics of MSW

The chemical composition of MSW components aids in the evaluation of alternative processing and recovery options. The most crucial properties to understand are proximate analysis, ultimate analysis, and energy content, critical in assessing the combustion properties of waste or waste-derived fuel (refuse-derived fuel). The moisture content, the ash content, and the fixed carbon content were determined. Based on the results of bomb calorimeter experiments, mixed solid waste analysis was performed to determine the energy content of organic components in MSW, and metal analysis was also performed. The constituents of waste used in waste management technologies are listed in Table 4.1.

Table 4.1. Proximate analysis, calorific value analysis, and heavy metals of mixed solid waste collected from the Guwahati disposal site

Characteristics	Average (%)
Moisture content	64.83 ± 6.05
Ash content	45.27 ± 12.52
Volatile matter	51.31 ± 10.84
Fixed carbon	3.61 ± 3.01
Parameters	**Average values**
Calorific value (kcal/kg)	2531.70 ± 553.4
Energy content (MJ/kg)	10.60 ± 2.30
Energy content (BTU/lb)	4557.1 ± 996.0
Metals	**Content (mg/kg of waste)**
Zinc (Zn)	116.60± 50.8
Copper (Cu)	38.40 ± 14.5
Lead (Pd)	69.20 ± 25.2
Arsenic (As)	1555.20 ± 364.60
Cadmium (Cd)	5.70 ± 9.6
Iron (Fe)	10201.20 ± 3326.30

The ultimate analysis is important as mass balances for chemical and thermal processes are calculated. The ultimate waste analysis was performed to determine the carbon, hydrogen, oxygen, nitrogen, and sulfur content (C, H, O, N and S). Metal content (e.g., Cd, Cr, Hg, Ni, Zn, and Mn, etc.) was calculated due to the possibility of adverse environmental effects. Fig. 4.3 shows the C, H, O, N, S portions with percentage concentration in the waste.

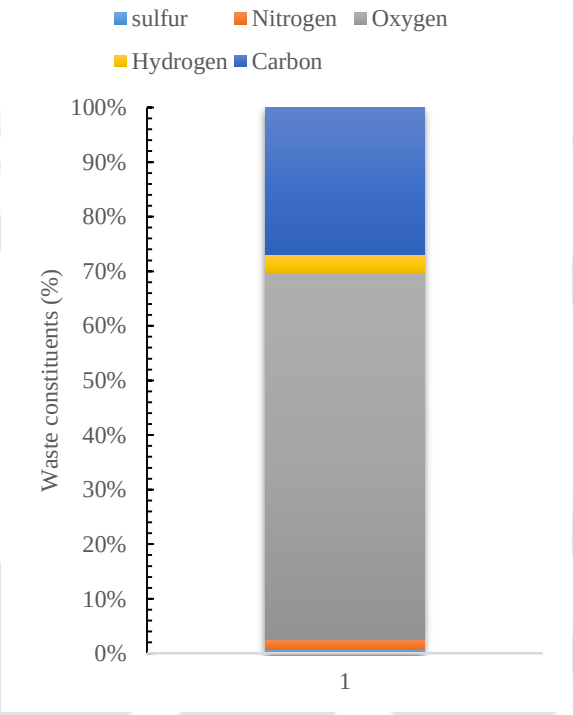


Fig. 4.3. Ultimate analysis of mixed solid waste

4.3 LIFE CYCLE ASSESSMENT

LCA is used as an environmental protection method to measure the material from the cradle to the grave, as defined in ISO 14,040 and ISO 14044. The life cycle of a product from its raw materials, production, usage, and end-use to disposal (Khandelwal et al., 2019). The current research work has involved four phases: goal and scope definition, inventory analysis, impact evaluation, and interpretation.

4.3.1 Goal and scope definition

The study's primary goal is to measure and evaluate the environmental effects of four different waste disposal strategies and perform an economic assessment of proposed strategies. The first scenario was chosen based on Guwahati's current MSW composition, which includes a lot of wet waste that is composted and open dumping. The second scenario

was windrow composting and sanitary landfill with gas utilization. Based on the physicochemical properties of solid waste, the third scenario was chosen: refuse-derived fuel, windrow composting, and sanitary landfill with gas consumption. Selection of the fourth scenario based on waste composition, physicochemical properties of solid waste: manual sorting, windrow composting, and sanitary landfill with gas utilization. The analysis was conducted using the Gabi program and the CML 2001 system.

4.3.2 Functional unit

To illustrate the impact of the present waste system on the environment, the economy uses life cycle assessment and costing analysis for the Guwahati waste management system. Four commonly used MSW treatment technologies are compared to analyze the efficiency and suitability best disposal option. To evaluate the alternative scenarios, the management of one metric tons of MSW in Guwahati, Assam, India, was selected as the functional unit.

i) ***Scenario 1 (S1):*** This is the baseline scenario which consists of landfilling into an open dump at the disposal site and windrow composting. Composting would be favored due to the high moisture content of food waste and the remaining waste transported to open dumping in this study.

ii) ***Scenario 2 (S2):*** This scenario involves segregating the MSW into two different streams and including a composting process for the biodegradable organic waste while the non-biodegradable inorganic waste and biodegradable organic waste in a sanitary landfill for gas utilization. However, gas is produced when there is a high amount of organic waste in a landfill

iii) ***Scenario 3 (S3):*** This scenario will include, in addition to composting and sanitary landfilling, a refused derived fuel (RDF). High calorific value products were used to make RDF pellets and transported to waste to energy plants.

iv) ***Scenario 4 (S4):*** In this scenario, the biodegradable fraction of MSW is sent for composting. At the same time, the non-biodegradable waste like plastic, wood, and textile waste is segregated by manual sorting, and the resulting residue is landfilled along with other inerts.

4.3.3 System boundaries

The emission levels associated with different waste management options were calculated using the system boundary framework, a critical component of the life cycle assessment methodology. A subjective system boundary was used to establish the system

boundary for this analysis. Fig. 4.4 showed the system boundary accounts for all inputs of natural resources and outputs into the air, water, and land, as well as economic analysis (fixed cost and variable cost). It starts with the disposal strategies of MSW and the application of different MSW technologies to provide the best disposal method to Guwahati municipal solid waste management.

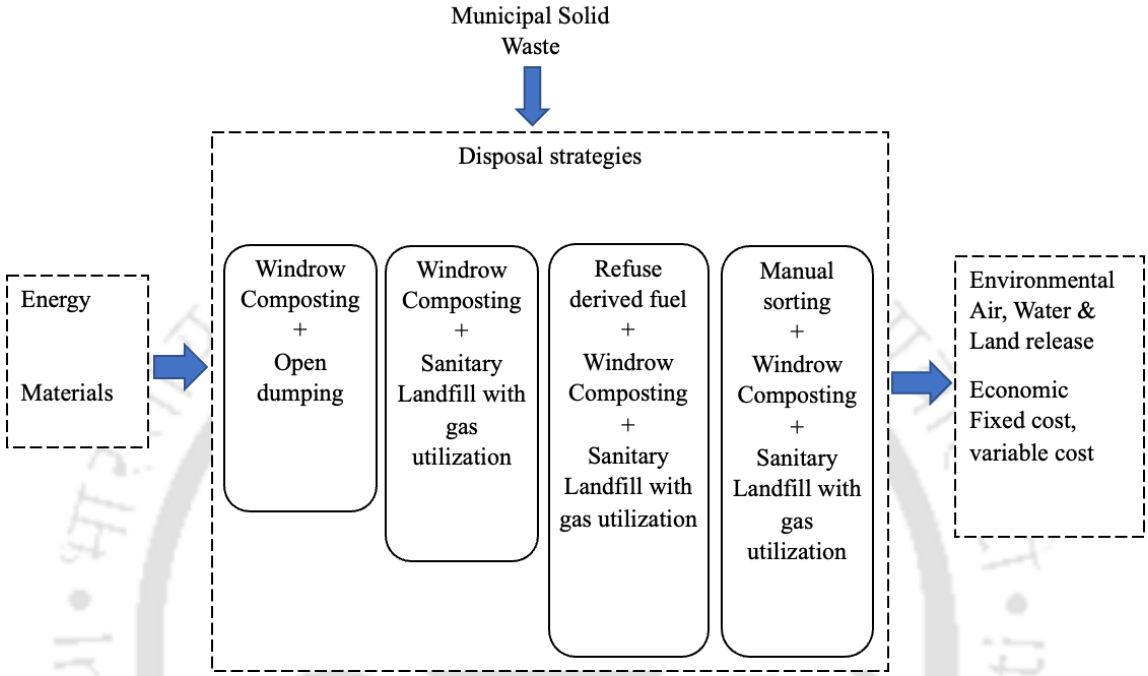


Fig. 4.4. The system boundary of scenarios of waste treatment strategies

4.4 LIFE CYCLE INVENTORY

4.4.1 Composting

4.4.1.1 Environmental impact (Air emissions)

Composting is the third most preferred choice in the hierarchy of advanced solid waste management. Aerobic compost occurs in the presence of oxygen which provides macro and micro-nutrients for the plants. MSW has a low heating value due to its high moisture content, which reduces its combustion ability. As a result, S1, S2, S3, and S4 include windrow composting (food and yard waste). The composting method assumptions were made using data from the IPCC Guidelines for National Greenhouse Gas Inventories and an overview of the waste composition in the study area. The environmental effect of windrow composting (air emissions) was measured using the fractional mass of the generated waste. CO_2 and CH_4 concentrations are maximum in the center of the windrows, where O_2 is scarce (Hao et al., 2001). Methanotrophic bacteria oxidize 46-98% of CH_4

formed in windrows before exiting the window composting (Jackel et al., 2005). Table 4.2 shows the default emission factors for CH_4 and N_2O emissions from waste biological treatment. Vehicular emissions from the movement of vehicles to compost plant data collected from the municipal officials of Guwahati Corporation as shown in Table 4.3.

Table 4.2. Default factors for CH_4 and N_2O emissions from biological treatment for the tier 1 method (DPR, 2008)

Composting	CH_4 Emission Factors (g CH_4/kg waste treated)	N_2O Emission Factors (g N_2O/kg waste treated)
Dry weight basis	10 (0.08 - 20)	0.6 (0.2 - 1.6)
Wet weight basis	4 (0.03 - 8)	0.3 (0.06 - 0.6)

Table 4.3. Vehicular emissions from the movement of vehicles to compost plant (DPR, 2008)

Parameters	Indicators	Emission factors /Amount	Converted Amount (kg/ton of waste)
Plant capacity		50 TPD	-
Material consumption	Diesel	50	41.6 kg
	Fuel economy	-	-
	Land use area		7000 m^2
Direct gas emissions	CH_4	4	6.424×10^{-7}
	N_2O	0.3	4.818×10^{-8}
	CO_2	515.2 g/km	2.06
	CO	3.6 g/km	0.014
	NO_x	6.3 g/km	0.25
	CH_4	0.09 g/km	3.6×10^{-4}
	SO_2	1.42 g/km	5.68×10^{-3}

HC	0.28 g/km	1.12×10^{-3}
PM	0.87 g/km	3.48×10^{-3}
NMVOC	-	7.2×10^{-3}
N_2O	-	2.16×10^{-5}
1,3-Butadiene	0.0074 mg/km	2.9×10^{-8}
Benzene	0.0049 mg/km	1.9×10^{-8}
Formaldehyde	0.0610 mg/km	2.4×10^{-7}
Acetaldehyde	0.0	-
Total Aldehyde	0.0837 mg/km	3.3×10^{-7}
Total PAH	3.9707 mg/km	1.5×10^{-5}

4.4.1.2 Economic analysis of composting

Scenarios of S1, S2, S3 & S4 were a mix of windrow composting technology as presented in the system boundary of waste disposal strategies. All biodegradables, primary food, and yard waste were assumed to be sent to windrow composting, other waste materials to the RDF plant, manual sorting (separation of recyclables). The components contributing to composting costs are: machinery cost includes the purchase, erection and commissioning cost, transportation, operating, and maintenance cost. The Full cost accounting and life cycle costing are applied to study the system's lifetime investment and cost. The estimate is made for both baseline and other combinations of technologies. Table 4.4 shows the primary data collected from the municipal officials of Guwahati Corporation. The life cycle costing of composting is carried out by estimating the initial operating, maintenance, and decommissions cost for the compost plant. The initial costs are determined using municipal data. The civil estimate is per the central public works department schedule of rates, 2013-14, and machinery cost, including the purchase, erection, and commissioning costs. The operational cost is estimated from the data obtained from the municipality officials. The subtracting salvage value calculates the net present value from all the estimated net present values, as shown in Table 4.5.

(i) To calculate the Net Present Value (NPV) of estimates

$$PV = \frac{C\left(1+\frac{i}{100}\right)^{(n-1)} - 1}{\left(1+\frac{d}{100}\right)^{n}} \quad (4.1)$$

Where,

C = any cost element at n^{th} year

I = inflation rate (5% for India rupee)

d = discount rate/ interest rate (8%)

Net Present Value (NPV) = Initial cost + Operating cost + Maintenance cost – Decommission cost (4.2)

Table 4.4. Inventory data of transportation, energy covered, and staff (DPR, 2008)

Data	Quantity
Number of employees	14
Plant Capacity	200 tons
Fuel consumed	50 lt
Vehicle economy	4 kmpl
Electricity consumed	0.01 MWh/ton
Salary	Operators-14,000
	Workers- 8,000

Table 4.5. Initial cost, annual operation cost, maintenance cost, and decommission cost for compost plant (DPR, 2008)

Initial cost (Ic) for the setup of compost plant			
Cost Element	Value / year	Time period	Remarks
Land cost (L)	4,92,80,000	0-1	38,500 m^2
Building cost (B)	15,74,28,943	0-1	Complete construction work
Machinery cost(M)	3,45,00,000	0-1	Trommels, blowers carry vehicle etc.,

Annual operation cost (Oc) for the compost plant			
Fuel cost (F)	10,80,500	2-10	Diesel
Electricity cost (E)	6,60,000	2-10	0.01 MWh/t
Labour cost (L)	30,24,000	2-10	Salary
Miscellaneous (M)	4,50,000	2-10	(1 % of Oc) Compost Testing, packing etc
Annual maintenance cost (Mc) for the plant			
Cost Element	Value / year	Time period	Remarks
Annual maintenance	3,45,000	2-10	10% of purchase
Decommission cost (Dc)			
Cost Element	Value / year	Time period	Remarks
Salvage cost	50,55,837	2-10	All machinery

4.4.2 Refuse derived fuel (RDF)

4.4.2.1 Environmental impact (Air emissions)

RDF is used as an alternate fuel in the cement industry to reduce the usage of fossil fuels and the overall effect on electricity production, GHG pollution, and environmental impacts. (Lima et al., 2018). RDF manufacturing and usage demand less energy (4.7 GJ/ton clinker) than hard coal, resulting in CO_2 emissions reductions ranging from 863-888 kg per tonne of clinker output, based on waste composition (Reza et al., 2013). Fuel emissions are measured using annual pollutant emissions as provided in Eq. 4.3.

$$E_{j,i} = Q_f \text{ x (pollutant concentration in fuel /100)} \times (MW_p/EW_f) \times \text{Opt hrs} \quad (4.3)$$

where

$E_{j,i}$ = Annual emissions of pollutant i, kg/day

Q_f = Fuel use, kg/hr

Opt hrs = Operating hours, hrs/day

MW_p = Molecular weight of pollutant emitted, kg/kg-mole

EW_f = Elemental weight of pollutant in fuel, kg/kg-mole

i = Concentration of pollutant i in fuel expressed as weight percent, %

The emissions inventory of the RDF plant was calculated using Eq.1. The plant capacity was assumed to be 200 TPD; the material consumption is diesel, and the required area is 14,000 m^2. Potential air Emissions were estimated and presented in Table 4.6.

Table 4.6. Emission inventory of RDF plant (DPR, 2008)

Parameters	Indicators	Emission factors /Amount	Converted Amount (kg/ton of waste)
Plant capacity		200 TPD	
Material consumption	Diesel	50	194.6 kg
	Land use area	-	14,000 m^2
Direct gas emissions	CH_4	56	0.18
	CO	98	0.32
	CO_2	154	0.51
	SO_2	1.8	6×10^{-3}
	N_2O	9.4	0.03
	CO_2	515.2 g/km	4.90
	CO	3.6 g/km	0.033
	NO_x	6.3 g/km	0.59
	CH_4	0.09 g/km	8.56×10^{-4}
	SO_2	1.42 g/km	1.35×10^{-2}
	HC	0.28 g/km	2.66×10^{-3}
	PM	0.87 g/km	8.28×10^{-3}
	NMVOC	-	1.71×10^{-2}
	N_2O	-	5.14×10^{-5}
	1,3-Butadiene	0.0074 mg/km	6.9×10^{-8}
	Benzene	0.0049 mg/km	4.52×10^{-8}
	Formaldehyde	0.0610 mg/km	5.71×10^{-7}
	Acetaldehyde	0.0	-
	Total Aldehyde	0.0837 mg/km	7.8×10^{-7}
	Total PAH	3.9707 mg/km	3.57×10^{-5}

4.4.2.2 Economic analysis for RDF plant

Economic analysis was carried out based on accounting for one year. The life cycle costing study of RDF is carried out by estimating the initial cost, operating cost, maintenance cost, and decommission cost for the RDF plant. The initial costs are determined using municipal data. The civil estimate is per the central public works department schedule of rates, 2013-14, and machinery cost, including the purchase, erection, and commissioning costs. The operational cost is estimated for the setup plant for a year. The subtracting salvage value calculates the net present value from all the estimated net present values, as shown in Table 4.7.

Table 4.7. Initial cost, annual operation cost, maintenance cost, and decommission cost for RDF plant (DPR, 2008)

Cost Element	Value / year	Time phase	Remarks
Initial cost (Ic) for the setup of the RDF plant			
Land cost (L)	4,92,80,000	2-10 year	38,500 m^2
Building cost (B)	10,74,28,943	2-10 year	Complete construction work
Machinery cost (M)	12,40,00,000	2-10	All machinery
Estimate of operational cost (Oc) of RDF plant			
Electricity cost (E)	1,30,39,020	2-10 year	0.01 MWh/t
Labour cost (L)	30,24,000	2-10 year	Salary
Miscellaneous (M)	1,60,000	2-10	(1 % of Oc) Testing, packing etc
Estimate of maintenance cost (Mc) of RDF plant			
Annual maintenance	10,00,000	2-10	10 % of purchase
Decommission cost (Dc)			
Annual maintenance	1,50,75,505	2-10	All machinery

4.4.3 Manual sorting plant (MSP)

4.4.3.1 Economic analysis for the manual sorting plant

This scenario is focused on a sorting plant that can classify waste into many parts, including organic from inorganic (main flows), ferrous metals from lighter materials, and the heaviest portion of the waste. The organic fraction is essential to kitchen and vegetable waste, while plastic, paper, cardboard, wood, and textiles are inorganic fractions. To reduce the burden on landfills sorting plants has been combined with windrow composting and sanitary landfill. After sorting, organic material is transported to windrow composting, while plastic is delivered to the RDF plant to generate electricity. The life cycle costing study of MSP is carried out by estimating the initial cost, operating cost, maintenance cost, and decommission cost for the MSP plant. The initial costs are determined using municipal data. The civil estimate is per the central public works department schedule of rates, 2013-14, and machinery cost, including the purchase, erection, and commissioning costs. The operational cost is estimated for the setup plant for a year. The subtracting salvage value calculates the net present value from all the estimated net present values, as shown in Table 4.8.

Table 4.8. Initial cost, Annual operation cost, maintenance cost, and decommission cost for MSP (DPR, 2008)

Cost Element	Value / year	Time phase	Remarks
Initial cost (Ic) for the setup of the MSP plant			
Land cost (L)	92,80,000	-	38,500 m^2
Building cost (B)	10,74,28,943	0-1 year	Complete construction work
Machinery cost (M)	18,00,00,000	0-1 year	All machinery
Estimate of operational cost (Oc) of RDF plant			
Electricity cost (E)	37,23,000	2-10 year	0.01 MWh/t
Labour cost (L)	15,00,03,000	2-10 year	Salary
Miscellaneous (M)	10,00,000	-	(1 % of Oc) Testing, packing, etc

Estimate of maintenance cost (Mc) of RDF plant			
Annual maintenance	15,47,260	2-10	10 % of purchase
Decommission cost (Dc)			
Salvage cost	2,18,83,797	2-10	All Machinery

4.4.4 Sanitary landfill/ open dumping

4.4.4.1 Environmental impact (Air emissions)

Landfilling is the most traditional and often used form of MSW treatment. Landfilling operations worldwide vary from uncontrolled dumpsites to heavily engineered facilities that store Leachate and landfill gas (LFG). The organic fraction of waste decomposes in this environment, producing so-called landfill biogas. This is composed mostly of methane (CH_4) (58%) and carbon dioxide (CO_2) (41%) but can also include traces of hydrogen sulfide (H_2S), hydrochloride (HCl), hydrogen fluoride (HF), and other chemical compounds (Cherubini et al., 2009). Leachate is a liquid that percolates from landfills after the biodegradation of solid waste. Landfill gas and leachate are the two major issues of landfills and their impacts on the environment. Therefore, landfill gas is estimated using the flux chamber method (Gollapalli and Kota, 2018). The circular stainless steel chamber has a width of 40 cm and a height of 35 cm, with one side closed and two holes, as shown in Fig. 4.5. One hole at the center is made for sampling, and the other is close to rims for temperature measurement. The 10 cm of the 35 cm high chamber is inserted into the soil or waste bed, and the upper section is placed on top. Sampling points were established based on the age (2-4 years) of the MSW on the surface layer and deposition height (5-15 feet) (Jha et al., 2008). The chamber was embedded in the landfills, and appropriate installation ensured that the surrounding atmosphere was retained. A displacement procedure was used to take the gas in the sampling chamber and transfer it to vials. The chamber was positioned from 9 am to 4 pm (day time); samples were taken at daily intervals of one hour and measured landfill temperature at 5 cm shallow depth and atmospheric temperature. Methane concentration was analyzed using the Porapak Q column of Gas Chromatograph (GC). The emission flux rates were calculated using Eq. 4.2 (Gollapalli and Kota, 2018).

$$EF = \frac{\Delta c}{\Delta t} \times h \qquad (4.4)$$

where

$\Delta C/\Delta t$ denotes the slope of the linear relationship between concentrations and sampling period, and h denotes the chamber's height above the soil level.

The gas samples were collected at different intervals, as shown in Table 4.9, and concentration vs. time was plotted to obtain the slope and presented in Fig. 4.6. The CO_2 emission was calculated using the above Eq. 2, and obtained value was 4.484 mg/ m^2 d.

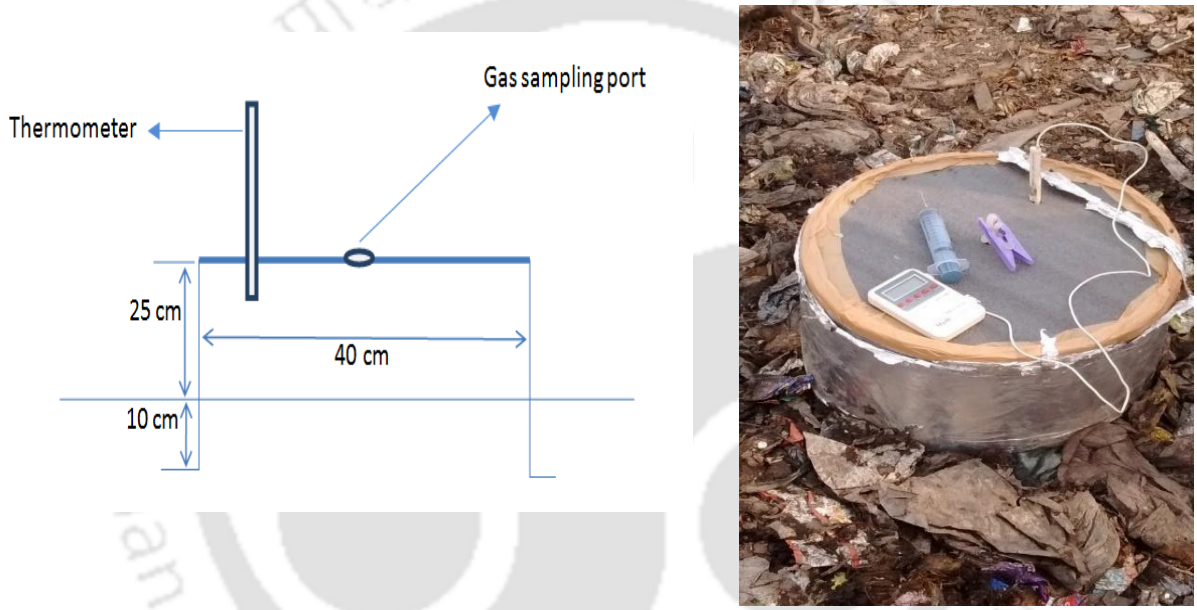


Fig. 4.5. Field setup of stainless steel chamber for gas collection

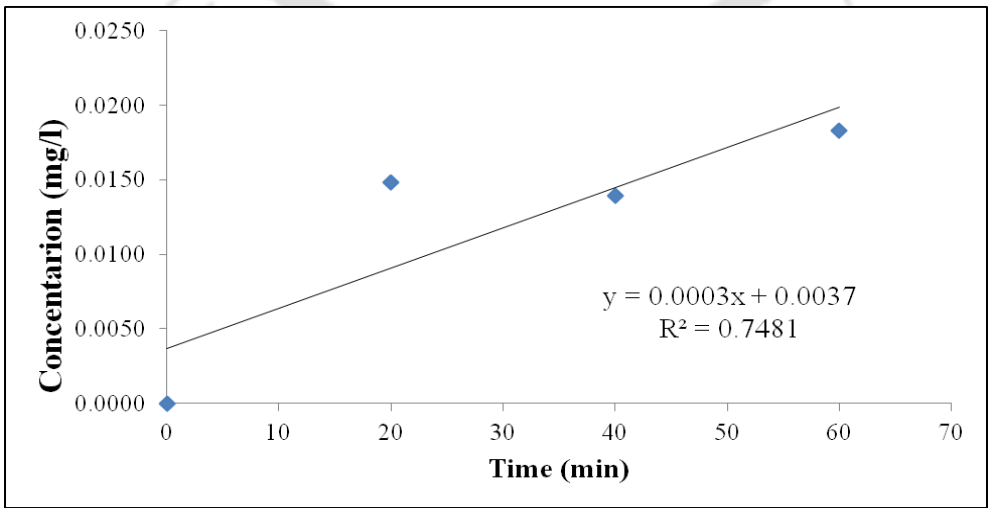


Fig. 4.6. The linear diagram for CO_2 analysis

Table.4.9. Gas samples from landfill

Sample No.	Time	Volume	Temperature
Sample 1	0	0.0	25
Sample 2	20	3.259	21
Sample 3	40	3.073	22
Sample 4	60	4.059	23

(i) Determination of methane theoretical using IPCC default method

The United Nations Intergovernmental Panel on Climate Change developed a multi-phase model in 2006 to measure global methane pollution for all nations. Waste generation rates, population, waste degradable organic carbon (DOC) composition, DOC_f fraction, waste decay rate (k), and CH_4 correction factor may be used for default values or actual data. The IPCC model makes use of the first-order effects. The IPCC model relies on using first-order decay models, the DOC, and k-values for various kinds of waste, including paper, food, furniture, etc. (Gollapalli and Kota, 2018). Emissions were estimated using Eq. 4.5, 4.6, 4.7 & 4.8.

Methane emission (Gg/yr)=

$$\left\{\left(MSW_T \times MSW_f \times MCF \times DOC \times DOC_f \times F \times \left(\frac{16}{12}\right) \times (1-OX)\right)\right\} \quad (4.5)$$

where:

MSW_T: total MSW generated (Gg/yr)

MSW_F: fraction of MSW disposed to solid waste disposal sites

MCF: methane correction factor (fraction)

DOC: degradable organic carbon (fraction) (kg C/ kg SW)

DOC_f: fraction DOC dissimilated

F: fraction of CH_4 in landfill gas (IPCC default is 0.5)

16/12: conversion of C to CH_4

OX: Oxidation factor (fraction – IPCC default is 0)

(ii) The methane emissions from landfills are theoretical calculated

Methane emissions (Gg/yr) = {(200 Gg/yr)×(0.7)×(0.4)×(0.157) × (0.77) × (0.5) × $\left(\frac{16}{12}-0\right)$} × (1-0) = 1.89 Gg/yr. (4.6)

(iii) Volatile organic compound emission (as hexane)

$$Q_{VOC}=\left(1+\left(\frac{C_{CO_2}(\%)}{C_{CH_4}(\%)}\right)\times Q_{CH_4}\times\left(\frac{C_{VOC}}{10^6}\right)=1229.35\frac{m^3}{yr}\right) \quad (4.7)$$

where

Q_{VOC} = Emission rate of pollutant VOC,

Q_{CH_4} = methane generation rate, m^3/yr (from Eq.1)

$C_{CH_4}(\%)$ = the concentration of CH_4 as a percentage of the total landfill gas. If unknown, assume 55% CH_4

$C_{CO_2}(\%)$ = the concentration of CO_2 and other gas constituents as a percentage of the total landfill gas. If unknown, assume 45%

106 = conversion from ppmv

(iv) Mass calculation of VOC (as hexane)

$$M\ (kg/yr)=Q_{VOC}\left[\frac{MW\times 1atm}{8.205\times 10^{-5}}\right] = 4332.9\ kg/yr \quad (4.8)$$

where

MW = Molecular weight (as hexane), g/mol

8.205×10^{-5}= constant to convert emissions of VOC to kg/yr, m^3-atm/g mol-k

- Air emissions (site-specific data) were estimated from landfills using the above equations, as shown in Table 4.10

Table 4.10. Air emission inventory from landfill (DPR, 2008)

Parameters	Indicators	Emission factors /Amount	Converted Amount (kg/ton of waste)
Material consumption	Land use	86,000 m^2	-

Fuel consumption	Diesel	200	166.4 kg
Direct gas emission (air)	CO_2	4.484 mg/m^2 day	1.87×10^{-4}
	VOC	4332.9 kg/year (Hexane)	0.02
	CH_4	1.89 Gg /yr	9.41×10^{-6}
	CO_2	515.2 g/km	0.749
	CO	3.6 g/km	5.2×10^{-3}
	NO_x	6.3 g/km	9.1×10^{-3}
	CH_4	0.09 g/km	1.3×10^{-4}
	SO_2	1.42 g/km	2.06×10^{-3}
	HC	0.28 g/km	4.07×10^{-4}
	PM	0.87 g/km	1.26×10^{-3}
	NMVOC	-	2.52×10^{-3}
	N_2O	-	7.56×10^{-6}
	1,3-Butadiene	0.0074 mg/km	1.07×10^{-8}
	Benzene	0.0049 mg/km	7.12×10^{-9}
	Formaldehyde	0.0610 mg/km	8.8×10^{-8}
	Acetaldehyde	0.0	0
	Total Aldehyde	0.0837 mg/km	1.2×10^{-7}
	Total PAH	3.9707 mg/km	5.6×10^{-6}

- The Leachate generation from the landfill site was observed. Leachate and water samples are assessed to evaluate the contaminant flow to local water bodies. The parameters were tested for COD, BOD_5, and heavy metals like arsenic, lead, zinc, and cadmium, illustrated in Table 4.11.

Table 4.11. Leachate emission inventory from landfill

Parameters	Indicators	Emission factors/Amount	Converted Amount (kg/ton of waste)
Direct water emission	Color	-	Black
	Sample volume	500 mL	-
	pH	6.5	-
	Alkalinity	72.5 mg/L	6.59×10^{-8}
	TDS	1394.5 mg/L	1.26×10^{-6}
	EC	652.25 mS/cm	-
	BOD_5	24,750 mg/L	2.25×10^{-5}
	COD	46,450 mg/L	4.22×10^{-5}
	BOD/COD	0.538	-
	TN	302.5 mg/ L	2.75×10^{-7}
	NH_4^+-N (converted as ammonia)	206.5 (251.081 as ammonia) mg/L	2.28×10^{-7} -
	Zn	443 µg/L	4.02×10^{-10}
	Cd	2.6925 µg/L	2.44×10^{-12}
	Pb	49.5 µg/L	4.5×10^{-11}
	Cu	115.75 µg/L	$1.01\ 5 \times 10^{-10}$
	Ni	0.29 µg/L	2.63×10^{-13}
	Cr	337 µg/L	3.06×10^{-10}

4.4.4.2 Economic analysis for landfill

The Economical study using full cost accounting and life cycle costing involves the check for affordability in maintaining the landfill over initial cost, maintenance cost, etc. Table 4.12 shows primary data for the analysis.

Table 4.12. Primary data of labor, vehicles, and consumption (DPR, 2008)

Data	Quantity
Land area	24 hectare
No of vehicles	4
Number of drivers	8
Salary of drivers	9000 per month
No. of supervisor	2
Salary of supervisor	15,000 per month
Fuel provided to vehicles	200 L/day
Fuel price	Rs 60/L
Total cost over fuel for all auto tippers	12,000/day

The life cycle costing of the sanitary landfill was estimated investment for the start of landfill includes land cost, site design, development and assessment, material cost, machinery cost. The initial costs are estimated based on the data collected from the municipality. The civil estimate is per the central public works department schedule of rates, 2013 -14, and machinery costs, including purchase. The detailed costing is presented in Table 4.13.

Table 4.13. Initial cost, covering cost and post-closure maintenance cost for landfill (DPR, 2008)

Cost Element	Value / year	Year	Remarks
Initial investment (Ii) for the start of landfill			
Land cost (L)	25,60,00,000 Cr	-	2,00,000 m^2
Site design, development and assessment (Ma)	1.04,00,000 Cr	0-1	Earthwork,

Material cost	38,58,50,000 Cr	0-1	Geomembrane, Leachate Pipes, Sumps and Pumps
Machinery cost (M)	31,50,000 L	0-1	vehicular
Covering cost (Cc) for the start of landfill			
Fuel cost (F)	36,50,000 L	0-10 year	Diesel
Complete Cover system	28,33,50,000 Cr	0-10 year	Clay, sand, plantation
Labour cost (L)	8,64,000 L	0-10 year	Salary
Miscellaneous (M)	7,02,00,000 Cr	-	1% of Oc
Estimation of post-closure maintenance cost			
Post closure cost	3,58,064 L	-	10% of the purchase

PHASE-II (PART-I) & (PART-II): OPTIMIZATION OF WASTE COMPOSITIONS, FABRICATION OF SIMULATED REACTOR, CHARACTERIZATION OF LANDFILL LEACHATE & MONITORING OF LANDFILL GAS

4.5 PART-I (OBJECTIVE-II) (SIMULATED REACTOR OPERATED WITHOUT RAINFALL)

4.5.1 Experimental design

The experiment (DOE) design was used for the experimental setup to minimize the number of reactors. 59 cities of wet and dry compositions of solid waste data of India were considered (CPCB, 2005) and used to optimize the experiments instead of conducting several experiments with available solid waste compositions. The exponential population growth, the high density of urban areas, diverse cultures, changing food habits, and lifestyles have seen an unresolved problem in terms of municipal solid waste management (MSWM) in India, which is the main reason for the selection of different cities of solid waste composition rather than that of a specific city-data. As a result, it would represent entire India. A full factorial design quadratic model in Matlab enabled exploration of all main effects and interaction effects that have been used to optimize the waste composition.

This design enables us to minimize the time involved in measuring the leachate parameters and monitoring the reactor.

(i) Design considerations:

- Waste compositions: wet and dry compositions
- Linear constraint: $w + d \leq 100$

The specific combined design determined by full factorial design was that the sum of two components should be 100. Based on the model, five different proportions of waste were determined to conduct the experiments, as shown in Table 4.14.

Table 4.14. Five different compositions of waste

Wet	Dry
73	27
65	35
56	44
47	53
38	62

4.5.2 Waste components: collection and preparation

The waste components considered in this study comprised 73% wet waste (i.e., food) and 27% dry waste (i.e., paper, plastic wood, and metals), and the composition of waste data was obtained from the central pollution control board report (CPCB, 2005). Fresh waste components were obtained from individual waste sources. Food waste was obtained from the fancy Bazar and mostly contained fruits and vegetables, and kitchen waste was collected from IITG hostels. Mixtures of paper, plastic, wood, and metals were collected from recycling bins and local stores. Individual components were mixed and fed into the reactor.

4.5.3 Initial characterization of mixed solid waste samples

The moisture content of samples was determined by drying them in a hot air oven for 24 hours at 105°C. (Tiquia and Tam, 1998). A muffle furnace was used to determine the quality of volatile solids (VS) (Tiquia and Tam, 1998; Kalamdhad and Kazmi, 2008). The pH of the samples was calculated by rotating them in distilled water for two hours with waste to water extract proportion of 1:10 (w/v). The total organic carbon calculation is shown below Eq.4.7.

$$\%\,\text{Total organic carbon} = \frac{\%\,\text{Voltaile solids}}{1.8} \quad (4.9)$$

4.5.4 Setup of simulated landfill reactors

Two landfill simulation reactors (LSR) were employed in the experiment, coded as reactor 1 (R1) and reactor 2 (R2). R1 and R2 reactors represented conventional landfills in India where R1 operated without rainfall, had a high amount of wet waste with a 73% and 27% ratio, and R2 operated without rainfall had a high amount of dry waste with a ratio of 38 % and 62% respectively as shown in Fig. 4.7. The simulated landfill with 1m×1m×1.1m was constructed using mild steel with a thickness of 0.012m. the 10cm headspace was provided even rainfall distribution to the waste. After loading waste to the reactor, the simulating reactors were sealed with m-seal and silicone sealant and operated in a laboratory environment. The leachate was collected once a week in each of the simulated reactors, coded as leachate 1 (L1) from R1 and leachate (L2) from R2. Leachate was stored in a collection tank for further analysis and treatment. Therefore, the simulated reactors replicate an anaerobic cell's actual condition (unshredded mixed solid waste) in India's open dumpsites. Landfill temperature was measured using the multi thermometer to evaluate the influence of degradation of waste process.

4.5.5 Collection of landfill leachate and gas

Landfill Leachate samples were collected in a measuring container to monitor the amount of leachate produced and for characterization and transferred to a refrigerator in specimen tubes for storage at 4°C. The reactors were monitored once a week. All the collected samples were analyzed in duplicate in the laboratory. The Gas collection system is arranged at surface level, representing the entire gas composition of the reactor. These data were used to represent landfill gas composition. Gas composition was measured by pumping gas out of the reactor and collected into a Tedlar bag through a standard SKC grab air sampler at 1.0 L/min. Gas chromatography was used for measuring % Methane (CH_4), % Carbon Dioxide (CO_2), %Oxygen (O_2), %Nitrogen (N_2), and Carbon Monoxide (CO). The frequency of gas sampling was two times a week. The gas samples were collected from R1 (without rainfall) & R3 (with rainfall). As degradation progressed, the gas production rate decreased, and the frequency of sampling was reduced accordingly. The actual temperature due to the degradation process measured using a thermometer is illustrated in Fig. 4.

Fig. 4.7. Pictorial view of landfill simulation reactor

Fig. 4.8. Pictorial view of mixing of solid waste and feeding of waste to the reactor

Fig. 4.9. Pictorial view of the collection of landfill leachate and gas samples

4.5.6 Part-II (Objective-III) (operated with rainfall)-an operational procedure

The LSR was fabricated with mild steel sheets. The landfill reactor was operated without leachate recirculation and controlled temperature, representing a conventional landfill operation. The landfill reactor operated with rainfall is coded as reactor (R3) and leachate (L3). The reactor setup with dimensions of 1.1m ×1m ×1m was used to simulate the current practice of open dumping in India under the effect of rainfall using a large-scale anaerobic landfill reactor in laboratory conditions. Headspace between the top surface of the waste and the top lid was kept at 10 cm to accommodate the water distribution pipe to distribute the tap water as rainfall uniformly over the landfill area. Unshredded fresh mixed (organic and inorganic) MSW was fed into the reactor. The weighted average rainfall according to area received by India was 1155 mm in 50 years (IMD, 2014); based on the information of rainfall and dimensions of the reactor, the amount of tap water to be added was calculated; 22.4 L of water was added to the simulator reactor through the showerhead. This ensures the even distribution of the rainwater across the waste. Once a week, rainfall (tap water) was added to the reactor. Distilled water was not used for rainfall simulation as it has a higher affinity to absorb the contaminants than tap water, resulting in high carbon washout (Karanjekar et al., 2015). Landfill and ambient temperature were measured with a multi thermometer to assess the effect of temperature on the waste degradation process. The reactor was sealed entirely with a sealant to ensure no leakage, and the anaerobic condition was maintained. Collection of waste, feeding procedure is the same as section 4.4.

4.6 MODEL DEVELOPMENT (EXPONENTIAL DECAY)

A model curve was developed to extrapolate leachate emissions. The approach of other research studies (Gonullu, 1994; Kouzeli-Katsiri et al., 1999) attempted to develop models for organic compounds like BOD_5 and COD. In this study, interest was in the declining part of measured leachate parameters curves. Two basic processes were considered 1) the solubilization of organic waste from a solid phase to liquid phase and 2) degradation by microbial decomposition and flushing of dissolved organic matter through leachate. The first-order degradation model was used to model the leachate parameter curves. The aim was to estimate the rate constants k values for leachate parameters in origin software, using the Levenberg–Marquardt Algorithm, and function is first-order exponential decay. The fitting parameter, ‘a,’ is coefficient, and x represents time (week). Parameter ‘a’ and the

degradation factor k= b are obtained by fitting the experimental data by exponential decay, while y denotes the simulated value presented in Eq. 4.7.

$$y = a * e^{(-b*x)} \tag{4.10}$$

After obtaining the k values, the model was evaluated to get the R^2 value, which gives the goodness of fit of the model curves.

4.7 PARAMETER CORRELATION ANALYSIS

The degree of interaction between measured parameters and their phenomenon is explained by correlation analysis. Pearson correlation gives output -1 to +1 as a relation between parameters. The positive value indicates a positive relationship between parameters, and the negative value suggests a negative relationship between measured parameters (McBean & Rovers, 2000). The Pearson correlation (r) is represented in Eq. 4.8.

$$r = \frac{\sum_{i=1}^{n}(X_i - \overline{X})\,(Y_i - \overline{Y})}{(n-1)S_X S_Y} \tag{4.11}$$

Which $\overline{X}$ and $\overline{Y}$ Were several means. S_X and S_Y Respective standard deviations of the X and Y.

4.8 PHASE- III: CONVENTIONAL TREATMENT OF FRESH LEACHATE FROM SIMULATED REACTORS USING COAGULATION-FLOCCULATION FOLLOWED BY UP-FLOW ANAEROBIC FILTER

4.8.1 Landfill Leachate used for conventional treatment

Two simulated landfills were employed in the experiment, coded as reactor 1 (R1) and reactor3 (R3). The schematic configuration of reactors is shown in Fig. 4.6. R1 and R3 reactors represented conventional landfills, where R1 operated without rainfall and R3 operated with rainfall. The exact ratio of fresh refuse was wet (73%) and dry (27%). The leachate was collected once a week in each simulated reactor, coded as leachate 1 (L1) from R1 and Leachate (L3) from R3. Weekly addition of rainfall in the R3. Leachate was stored in a collection tank for further analysis and treatment. Therefore, the simulated

reactors replicate an anaerobic cell's actual condition (unshredded mixed solid waste) in open dumpsites in India.

4.8.2 Pretreatment of high strength leachate using the coagulation-flocculation process

All coagulant chemicals used in the study, including potash-alum, ferric chloride, ferrous sulfate heptahydrate, bentonite clay, and calcium hydroxide, are used to treat wastewater. Fresh leachate from reactor 1 (R1) was used to establish titration curves on the raw leachate. Titrations had to be done after many of the sampling occurrences due to leachate variability. Many coagulants also influence pH in relation to the landfill leachate's pH variability, making it challenging to achieve the aimed pH during coagulation-flocculation. Hence, those were selected and conducted the laboratory analysis for analyzing the optimum dosage and pH values. The fresh/young leachate generated from reactor 1 has been used as a pretreatment process. Coagulation was carried out using each coagulant at different pH and dosages to perform the above analysis at a laboratory scale. The velocity of the mixing and sedimentation was the same for all coagulants. The standard 1 L glass beakers were used, and the stirring rate was 120rpm at one minute and 30rpm at 20mins, the floc allowed to settle. The jar tester also included a fluorescent lamp to observe floc formation, as shown in Fig. 4.10.

Fig. 4.10. Jar test apparatus

4.8.3 Biological treatment of young/high strength landfill leachate

4.8.3.1 Upflow anaerobic filter- system design and operational condition

Laboratory-scale up-flow anaerobic filters packed with plastic rings were constructed in series. One reactor was designed with a column height of 6 feet, a total liquid volume of 20 L, and a working volume of 16L. One column consisted of an acrylic glass column with an inside diameter of 0.114 m and 0.003 m. The corrugated plastic rings were used as media of ETIP 25 grade with a one-inch diameter and 1 mm gauge, with 85 kg/m^3 of bulk density. Plastic media was selected due to a specific surface area to grow the biofilm over the media, and the void ratio is 91%. The anaerobic filters operated at room temperature (15-40^oC), and the filter's specific surface area was 213 (m^2/m^3). On average, the specific packing surface area is 100 m^2/ m^3 (Song and Young, 1986). The dimensions of the plastic pall rings were diameter x height x thickness: 0.025x0.025x0.0012 m^3. Peristaltic pumps were used to feed the leachate into the reactor and maintain the required recirculation flow. The 5days HRT was maintained during the operation period. The OLR was 5 kg COD/m^3 d as the literature suggested that 5-22 kg COD/m^3 d have been successfully applied to anaerobic filters treating other wastes (El-Shafie and Bloodgood, 1973). The acclimatization phase/startup phase was done for 5months to develop biofilm, which consumes the organic solids in high-strength leachate. The biomass requires 2-4 months to develop the biofilm at favorable conditions, consuming the organic solids available in it by filtration mechanism (Young and McCarty, 1969). The treatment phase was done with Reactor (R3) leachate, which was operated with rainfall. The pretreatment was given to R3 leachate before treating it with an anaerobic filter because the concentration of leachates changes with time. Besides that optimum dosage, and pH also changes. Therefore, before an anaerobic filter, leachate pretreatment is necessary to remove suspended solids not to clog anaerobic filters. The OLR calculation is shown in Eq. 4.9.

This design is calculated considering the lowest flow rate practically possible in the lab through a peristaltic pump: 3 mL/min

Organic load suggested in literature according to the substrate and its concentration: 5 $\frac{kg}{m^3.day}$ (El-Shafie and Bloodgood, 1973).

Flow rate for 5 day HRT =3.5 mL/min

$$\text{Organic loading rate} = \frac{\text{flow rate} \times \text{concentration}}{\text{reactor volume}} = \frac{3(\frac{\text{mL}}{\text{min}}) \times 40000\ (\frac{\text{mg}}{\text{L}})}{\text{V}\ (\text{m}^3)} = 5 \quad (4.9$$

$$\frac{\text{kg}}{\text{m}^3.\text{day}}$$

$$V \approx 16\ L$$

Considering 12.5% volume to be occupied by biomass and media;

The volume of filter required: $\frac{16}{0.8} = 20\ L$

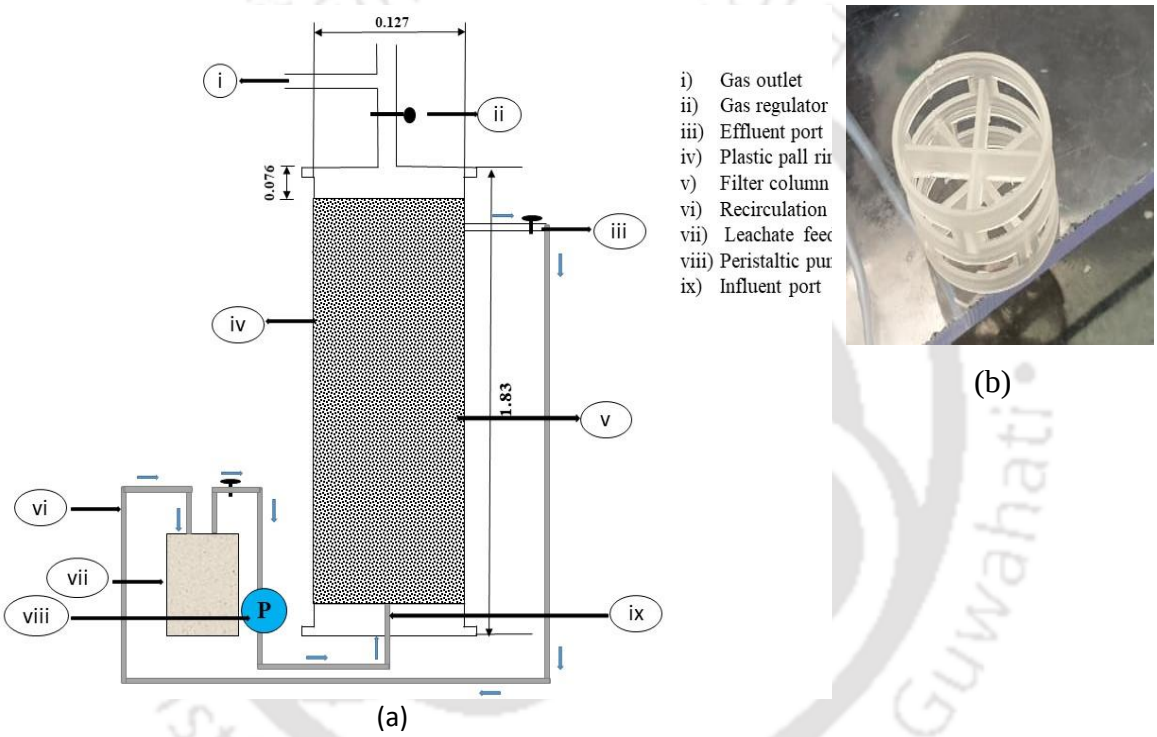


Fig. 4.11. (a) schematic diagram of upflow anaerobic filter and (b) plastic pall rings

4.9 METAGENOMIC STUDY

The two samples were collected from the simulated landfill reactor and pretreated landfill leachate to be analyzed microbiologically. Sample 1 (S1) corresponds to raw leachate, and sample (S2) corresponds to leachate incoming the biological treatment after coagulation treatment. The efficiency of the biological removal of carbon from leachate was determined by the activity of microbial populations present in biological reactors. Therefore, a complete characterization of bacterial communities revealed by 16S rRNA sequencing has been carried out.

4.9.1 Metagenomic DNA isolation, qualitative and quantitative analysis

The metagenomic DNA was isolated from the received soil and water samples by a commercially available Kit (Nucleospin). The qualities of the isolated metagenomic DNA sample were quantified using NanoDrop.

4.9.2 Preparation of 2 x 300 MiSeq library

The amplicon libraries were prepared using Nextera XT Index Kit (Illumina inc.) as per the 16S metagenomic sequencing library preparation protocol (Part # 15044223 Rev. B). Primers for the amplification of the bacterial 16S V3-V4 region were designed and synthesized at Eurofins Genomics Lab. Amplification of the 16s gene was carried out. 3µl of PCR product was resolved on 1.2% Agarose gel at 120V for approximately 60 min or till the samples reached 3/4th of the gel. Primers used in the present study, i.e., 16S rRNA F (GCCTACGGGNGGCWGCAG) and 16S rRNA (R ACTACHVGGGTATCTAATCC). The QC passed amplicons with the Illumina adaptor were amplified using i5 and i7 primers that add multiplexing index sequences and common adapters required for cluster generation (P5 and P7) as per the standard Illumina protocol. The amplicon libraries were purified by AMPure XP beads and quantified using Qubit Fluorometer.

4.9.3 Quantity and quality check (QC) of the library on Agilent 4200 tape station

The amplified libraries were analyzed on a 4200 Tape Station system (Agilent Technologies) using D1000 Screen tape per manufacturer instructions.

4.10 ANAEROBIC BMP SETUP OF LANDFILL LEACHATE WITH DIFFERENT RATIOS

Leachate Biomethane potential test (LBMP) is a useful tool for establishing methane production potential of liquid substrates and removal efficiency of COD. Digested cow dung was used as inoculum and obtained from local people nearby IIT Guwahati. The batch reactor was prepared using 1 L reagent glass bottles with rubber corks for closing the bottles. In the inoculum study, 10 batch reactors were used. Five different ratios were considered, i.e., 10, 20, 40, 50, and 80 % (v/v) of the total volume of cow dung added to landfill leachate, as shown in Table 4.15. The cow dung and leachate were divided equally and added to 1 L reagent bottles. The maintained volume was 800 mL in the reagent bottles, with duplicate reactor bottles were placed. The anaerobic reactor was coupled to another aspirator bottle having a 1.5 N sodium hydroxide (NaOH) solution that absorbs CO_2 and H_2S. The displaced NaOH solution represents CH_4 production. The purpose of Thymol

blue indicator. It is a pH indicator and, in this case, is used to make sure that the NaOH solution is still capable of removing CO_2 and H_2S. Initially, the aspirator bottle was blue due to the addition of an indicator. Once the capacity of the NaOH solution is exhausted, it turns into white color. All the reactor was maintained at room temperature varies from 30 to 38ºC. Manual mixing was done for less than one minute (10 times forward and backward) three to four times a day. Significant parameters were monitored during the entire experiment period, such as pH and COD. Fig. 4.12 shows the schematic diagram and experimental anaerobic BMP setup for the current study.

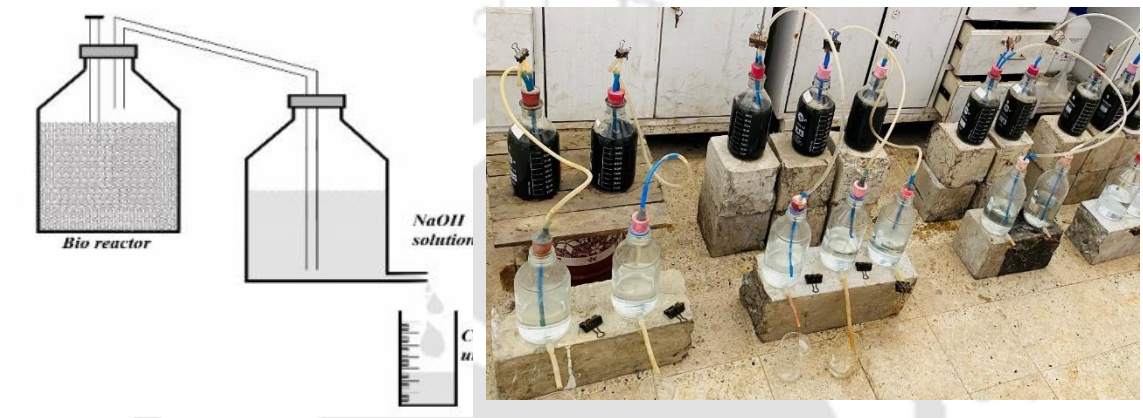


Fig. 4.12. The schematic and experimental anaerobic setup used for the current study

Table 4.15. Composition of feedstock used for BMP study

	Dosage of each material (%v/v)		
S.No.	Cow dung (%)	Landfill leachate (%)	Total volume filled in the digester (mL)
1	10	90	800
2	20	80	800
3	40	60	800
4	50	50	800
5	80	20	800

4.11 MONITORING AND ANALYSIS

4.11.1 Monitoring of landfill simulated reactor

The leachate sampling technique involves the collecting of data on the physical and chemical characteristics of the leachate. All leachate samples were collected in a measuring

container to determine the volume of leachate generated and then transported to the refrigerator in specimen tubes to be stored at 4°C before adding 2-3 drops of concentrated nitric acid metal analysis (APHA, 1998). All the collected samples were analyzed in duplicate at the laboratory. The sampling strategy was deliberate to cover various physicochemical parameters once a week. Gas samples were collected twice a week on consecutive days and analyzed in gas chromatography.

4.11.2 Analytical procedures

Analytical procedures of Standard Methods for the Examinations of Water and Wastewaters 20th edition, published by (DiLallo and Albertson, 1961; APHA, 1998), have been followed throughout the analysis. Certain parameters such as pH, EC, BOD_5, COD were analyzed as early as possible in the laboratory. These non-conservative parameters change over time. Later, samples were placed in coolers and kept at 4°C until the next analyses were scheduled to be carried out. A quality control procedure was maintained throughout, including the recalibration of instruments. Reagents were prepared as recommended by APHA Standard Methods (20th edition, 1998). All chemicals and reagents used in the analyses were of analytical grade unless otherwise stated. Deionized water was used for dilution. Standard solutions were prepared by diluting the stock solutions. The measured parameters are discussed below:

(i) pH

The pH meter was calibrated using two standard buffer solutions with pH values of 4.0 and 7.0. The electrodes were gently cleaned with distilled water and wiped clean with tissue paper. Immerse the electrode in the water sample (to measure the pH) and wait up to a few minutes for a consistent reading.

(ii) Electrical Conductivity (EC)

The temperature of 0.1 N KCl solution is determined. The cell was immersed in 0.1 N KCl solution, the instrument was turned on, and a switch with a range of 10 mho was chosen. The temperature knob was set to the solution's actual temperature. The CALIB knob was used to adjust the display to the correct conductivity value for the solution's temperature based on the chart included with the instrument (0.1 N KCl). Now that the device has been calibrated, it is possible to measure unknown conductivity.

(iii) Solid Analysis

Total solids, total dissolved solids, total suspended solids, total volatile solids, and total fixed solids were determined following APHA method 2540. To determine total solids, 50 mL of water was placed in an evaporating dish and heated to 104°C for 24 hours. The initial and final weights of the evaporating dish were determined to quantify the amount of residue left in the dish after evaporation to depict the total solids (TS). Total dissolved solids (TDS) were determined by heating another water sample, filtered with standard filter paper, at 180 °C for 24 hours in an evaporating dish. The residual remaining is the concentration of TDS. The water sample was heated to 180^{o}C rather than 104^{o}C to convert bicarbonate to carbonate. TSS is calculated as the difference between TS and TDS. Total volatile solids were determined as the difference in weights of total solids evaporating dishes after two hours at 550°C in a muffle furnace. By removing total volatile solids from total solids, total fixed solids were determined.

(iv) Biochemical Oxygen Demand (BOD_5)

The ultimate BOD test can determine the BOD_5, measured using initial and final DO (after keeping the BOD bottles in an incubator at 20°C) readings:

Procedure for measurement of BOD_5

- Prepare BOD dilutions. Use dilution water (it contains nutrients, the exact contents are described in Standard Methods): Blank (only dilution water); 10 mL sample in 300 mL BOD bottle, fill up with dilution water.
- Take a 300 mL sample in a BOD bottle. Prepare two sets of this sample. Keep one set for DO analysis for day 0 (i.e., Sample 0th day) and another sample in BOD incubator for 5 days at 20 °C (Sample 5th day).
- Measure DO in different samples at t=0.
- Incubate samples at 20 °C for 5 days, after 5 days, and record dissolved oxygen.

$$BOD_5 = \frac{(DO_5 - DO_0)}{\text{mL of sample}/300} \text{ mg } O_2/L \qquad (4.10)$$

(v) Procedure for measurement of DO

- Make dilution water by adding 1mL/L of the following reagents in distilled water:
- Phosphate buffer solution
- Magnesium sulfate solution
- Calcium chloride solution
- Ferric chloride solution

Add 1 mL alkali azide and then 1 mL manganese sulfate solution to a specified sample container. Shake the container vigorously and leave it open for 5 minutes to allow the precipitate to settle. Add 2 mL concentrated H_2SO_4 and close the container with the cap. Shake the container vigorously until all precipitate is dissolved. Titrate with 203 mL of normal sodium thiosulfate solution (0.025N) until the color changes from dark yellow to pale yellow. Then, titrate the solution with a few drops of the starch indicator until it becomes colorless or returns to its original sample color. Maintain track of the amount of 0.025N sodium thiosulfate consumed.

(vi) Chemical Oxygen Demand

The COD can be determined using the closed reflux method as follows:

- To avoid contamination, wash culture tubes and caps with 20% H_2SO_4 before use. Add digestion solution to the culture tube or ampule containing the sample. Carefully stream sulfuric acid reagent down the vessel's interior, forming an acid layer underneath the sample digestion solution layer, then cover tubes or ampules snugly, inverting each several times to mix thoroughly. In a block digester prepared to 150°C, place tubes or ampules and reflux for 2 hours under a protective barrier. Bring containers to room temperature and put on test tube rack. Certain amounts of mercuric sulfate may precipitate out, but this will not affect the analysis. Add 0.05 to 0.10 mL (1 to 2 drops) ferroin indicator and titrate with standardized 0.10M FAS. The endpoint is a sharp color change from blue-green to reddish-brown, although the blue-green may reappear within minutes. Similarly, reflux and titrate a blank containing the reagents and a volume of distilled water equal to that of the sample.

$$\text{COD is given by, COD} = \frac{(A-B)\times M\times 8000}{\text{mL of smaple}} = \text{mg } O_2 / L \qquad (4.11)$$

$$M = \frac{\text{Volume of } K_2Cr_2O_7}{\text{Volume of FAS}} \times 0.1$$

A = mL of FAS used for titrating the blank

B = mL of FAS used for titrating the sample

(vii) Sodium (Na^+), Potassium (K^+) & Calcium (Ca^{+2})

The concentration of sodium and calcium was determined using a Flame Photometer (SYSTRONICS Flame Photometer 128) in the laboratory and SYSTRONIC FPM Compressor 126.

(viii) Total Kjeldhal Nitrogen

The TKN can be determined using the kelpus distillation unit as follows:

- Take the 10 mL sample in the digestion tube. Take 25 mL sample in conical flak, 40% NaOH in a beaker. Start the distillation using the kelpus distillation unit. After the distillation process, titrate with 0.02N H_2SO_4.

$$\text{TKN in \%} = \frac{(\text{mL of } H_2SO_4) \times 14 \times \text{N of } H_2SO_4}{\text{mL of sample}} \quad (4.12)$$

(ix) Chloride

In the laboratory, Argentometric Method was used to determine chloride, where the sample was titrated against 0.0141 N solution of silver nitrate, with potassium chromate as an indicator.

(x) Fluoride

Preparation of standard curve*:*

- Fluoride standard solution in the range of 0-1.4 mg/L was created by diluting suitable amounts of Standard fluoride solution in distilled water to 50mL. Each standard is pipetted and well mixed with 5mL SPADNS solution and 5 mL zirconyl-acid reagent. With distilled water, the photometer is set to the zero absorbance point. After obtaining absorbance values of standards, a curve plotting mg F/L against absorbance is made.

Sample color development:

- 50mL of the sample or a portion diluted to 50mL with distilled water is used 0.5mL each spadns solution, and zirconyl-acid reagent is added and mixed well, and the absorbance is read.

(xi) Sulphate

Formation of barium sulfate turbidity:

- Measure 100mL sample or a suitable portion made up to 100 mL into a 250 mL flask. Add 20 mL buffer solution and mix in stirring apparatus. While stirring, add a spoonful of $BaCl_2$ crystals and begin timing immediately. Stir for 60 ± 2 s at a constant speed.

Measurement of barium sulfate turbidity:

- After the stirring period has ended, pour the solution into the turbidity cell of the turbidity meter and measure turbidity at 5 ± 0.5 min.

Preparation of calibration curve:

- Estimate SO_4^{2-} concentration in sample by comparing turbidity reading with a calibration curve prepared by carrying SO_4^{2-} standards through the entire procedure. Space standards at 10 mg/L increments in the 0-40 mg/L SO_4^{2-} range.

(xii) Nitrate

- **Treatment of sample:** 50 mL of clear sample is taken, added 1 mL 1 N HCl solution, and mixed thoroughly.
- **Preparation of standard curve:** Nitrate calibration standard in the range 0 to 7 mg NO_3^--N/L is prepared. Dilute the following volumes of intermediate nitrate solution, i.e., 0, 5, 15, 25, and 35 mL to 50 mL with distilled water. The standards are treated in the same manner as a sample.
- **Spectrophotometric measurement:** Absorbance is read against distilled water set at zero absorbance. A wavelength of 220 nm is used to obtain NO_3^--N reading, and a wavelength of 275 nm are used to determine interference due to dissolved organic matter.

(xiii) Ammonia

- **Treatment of sample**: 25 mL of clear sample is taken and added 1 mL phenol solution, 1 mL sodium nitroprusside, and 2.5 mL oxidizing agent.
- **Preparation of standard curve**: Ammonia calibration standard in the range 0 to 6 mg NH_3-N/L is prepared.
- **Spectrophotometric measurement**: Absorbance is read against distilled water set at zero absorbance. A wavelength of 640 nm is used to obtain an Ammonia reading.

(xiv) Phosphate

A series of 10 mL volumetric flasks were stacked in a grid pattern. 0.5 mL 0.387M ammonium molybdate, 3 mL 0.25N sulfuric acid, and aliquots of disodium hydrogen phosphate equivalent to 0.3-12.24 ppm (0.1,.2, 0.3, 0.5, 1.0, 1.5, 2.0, 2.5, 3.0, 3.5 and 4.0 mL) were added to each flask. Finally, 1 mL of 2.0833×10^{-3} M sodium sulphide solution was added to each flask. After about 20 minutes at room temperature, the solution was tested for adsorbance at 715 nm.

(xv) Volatile Fatty Acids (VFA)

Take a 50 mL sample and make it pH 3.3 using 0.05 N H_2SO_4. Boil it for 3 minutes and make it pH 4 using 0.05 N NaOH. Note the reading how much mL of 0.05N NaOH was consumed in bringing pH 4 to 7.

$$\text{VFA (mg/L)} = \frac{0.05\text{N NaOH consumed} \times 2500 \times \text{Dilution factor}}{\text{Volume of smaple (mL)}} \quad (4.13)$$

If VFA is greater than 180 mg/L, then multiply by 1.5.

(xvi) Heavy Metals (Cd, Cr, Cu, Ni, Pb, Fe, Mg & Zn)

All the heavy metals were analyzed using AAS by preparing standards in a certain range against absorbance at a particular wavelength.

(xvii) Gas chromatography

Gas samples were collected from the simulated landfill reactors, i.e., R1 & R3. The gas was collected in Teflon bags. Gas samples were measured using gas chromatography (GC, Dhruva company Pvt Ltd) equipped with molecular sieve 5A and TCD detector. 1 mL samples were used for analysis. The argon gas was used as a carrier to carry the gas to the TCD detector. Initially, known gas was taken from the gas mixture to obtain retention time of CO_2, CH_4, H_2, N_2, O_2. Later, unknown samples were injected into GC to attain the concentration of targeted samples. The area of the peak was calculated using Eq. 4.14.

Area= height of the peak × ½ width (4.14)

4.12 Instruments used

Table 4.16 Instruments used in the analysis

Parameter tested	Instrument	Model/Manufacturer
pH	μ pH system 361	132, Systronics, India
EC	Digital conductivity meter	VSI-04-Deluxe
Na^+, K^+ & Ca^{2+}	Flame Photometer 128	Systronics
Heavy Metals	AAS	Varian Spectra 55B
Nitrite, Ammonia, chloride, fluoride, sulphate	Spectrophotometer	MRC spectro V-110
Nitrate	UV-Spectrophotometer	CARY 50 Bio, VARIAN
TKN	Kelpus distillation unit	Pelican kelpus – Digital EM VA
Turbidity	Digital Nephelo-turbidity meter	132, Systronics, India
COD	Cod digester	Hach DRB 200, Hach, USA

Weight	Weighing balance	SL-234, Denver Instrument
Drying	Hot air oven	ICT, Calcutta, India
VS,FS	Muffle furnace	ICT, Calcutta, India
Sample preservation	Refrigerator	MRC scientific instruments, India
Biogas	Gas chromatography	Dhruva Gas chromatography

CHAPTER

5

Life cycle assessment

This chapter's insights explain the current research conducted in Guwahati city and the capital of Assam. This is a small representative city in India that is concerned with various solid waste management issues. Besides that, previous research has shown that the environmental impacts of multiple technologies differ from city to city and region to region due to waste composition and environmental conditions. Currently, Guwahati city was facing problems due to the continuous increase in solid waste, and there is no scientific disposal of waste for transportation, collection, and disposal process. As a result, not all cities can use the same technology. The analysis was carried out to evaluate the environmental impacts of different waste treatment technologies scenarios and determine the possible scenarios to reduce the effects on the environment for Guwahati city. The scenarios include open dumping, composting, sanitary landfill, RDF, and manual sorting and impact categories such as global warming potential, acidification, eutrophication, freshwater toxicity, photochemical ozone creation potential, and fossil depletion potential were also analyzed. The life cycle costing approach is used to the economic performance of different scenarios.

5.1 LIFE CYCLE ASSESSMENT OF DIFFERENT PROPOSED SCENARIOS

5.1.1 Baseline impact analysis

The multiple scenarios' life cycle risk assessments (LCIA) are carried out using the CML 2001-April 2015 methodology for global warming, acidification, eutrophication, freshwater toxicity, and the ReCiPe 1.08 methodology impacts on fossil depletion. The LCIA outcome for the baseline scenario is based on different effect categories, as shown in Fig. 5.1. CO_2 is a major driver of climate change. The baseline scenario 1 (S1) had the

maximum release of CO_2 impact (7.30E+ 00 kg of CO_2 eq) compared with other categories. Pollutant contaminants such as particulate matter (PM), oxides of sulfur (SO_X), oxides of nitrogen (NO_X), and heavy metals cause human toxicity. S1 had a moderate impact on Human toxicity (4.00E-01 kg of 1,4-dichlorobenzene (DCB) eq). Acidifying substances cause acidification like SO_X, NO_X. Scenario 1 had the least impact on acidification and freshwater aquatic toxicity (1.00E-02 kg SO_2 Eq). NO_X and VOCs have the potential to generate photochemical ozone since ozone is found at lower levels of the atmosphere under sunlight. The baseline scenario shows maximum photochemical oxidation potential (5.00E-02 kg of NMVOC eq) since NO_X is the primary pollutant generated by diesel engines. Another study in Nagpur, India, explained that the baseline considered composting and remaining material to landfilling. The findings showed that the baseline situation had the least effect on acidification (1.92E-01 kg SO_2 eq), eutrophication also highest impact (7.9E-02 kg PO_4^- eq). Photochemical has a maximum impact (3.04E-01 kg C_2H_4 eq) (Khandelwal et al., 2019).

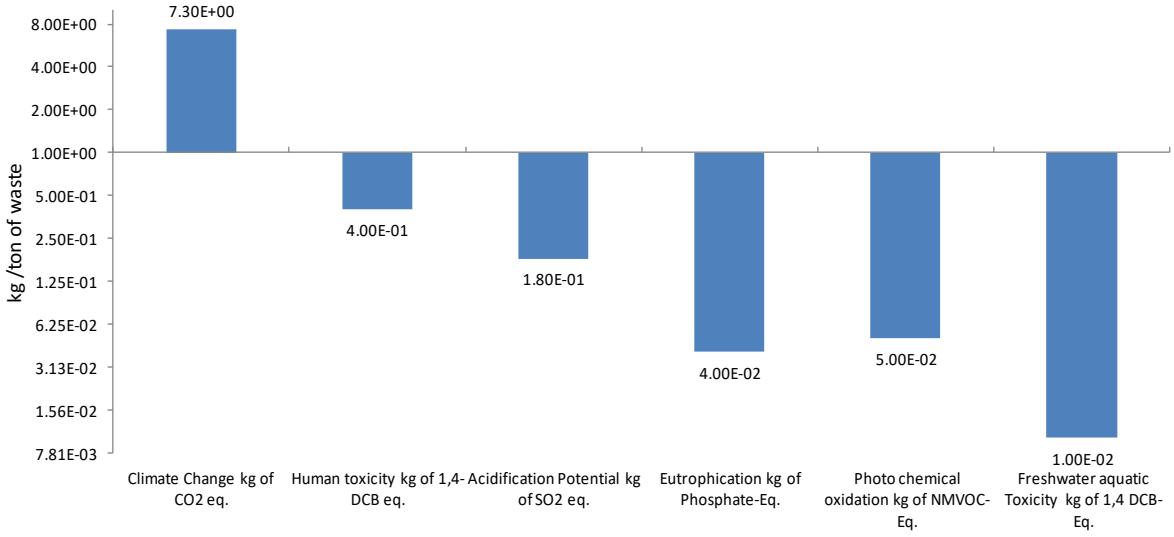


Fig. 5.1. Impact assessment of baseline study using CML 2001-April 2013

5.1.2 Environmental impacts of proposed solid waste treatment scenarios for Guwahati MSW

The LCIA of the various scenarios was conducted using the CML 2001-April 2015 methodology for global warming, acidification, eutrophication, freshwater toxicity, and the ReCiPe 1.08 methodology impacts on fossil depletion. The LCIA result for the different proposed scenarios according to the different impact categories is presented in Fig. 5.2. The

MSW composition in Guwahati city is 37.42% food waste, 17.44% plastic, 16.41% paper, 4.94% textiles, 4.14% glass, 1.97% leather, 0.45% rubber, 0.37% metals, 5.25% lawn, 2.45% wood scraps and 9.16% miscellaneous. The majority of waste generated in Guwahati is organic, plastic, and paper found in significant amounts in the present study. Food waste had an overall moisture content of 64.83 percent, which was higher than that of China (61%) (Zhen-Shan et al., 2009) and turkey (57%) (Yay, 2015), and Dhanbad (65%) (Yadav and Samadder, 2018). Therefore, it concluded that the moisture content of waste depends on biodegradable material in the landfill. The proposed scenarios for Guwahati municipal solid waste as follows:

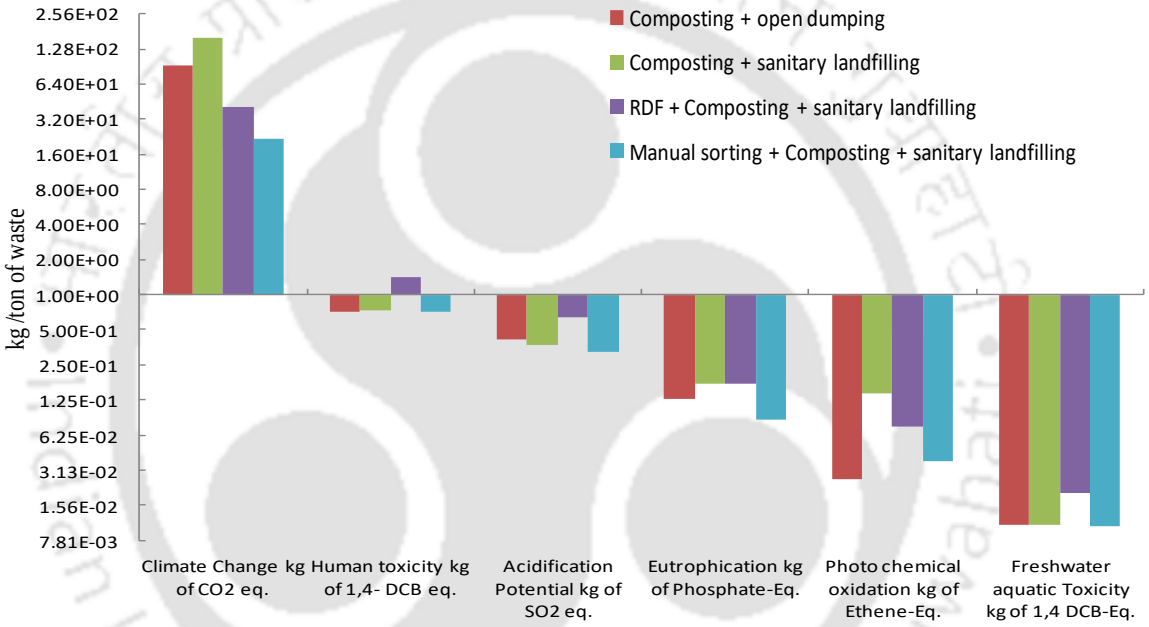


Fig. 5.2. Environmental impact assessment of different scenarios using CML 2001-April 2013

- ***Scenario 1 (S1):*** Windrow composting + open dumping (COMP_OD)
- ***Scenario 2 (S2):*** Windrow composting + sanitary landfill with gas utilization (COMP_SLF)
- ***Scenario 3 (S3):*** Refuse derived fuel + windrow composting + sanitary landfill with gas utilization (RDF_COMP_SLF)
- ***Scenario 4 (S4):*** Manual sorting + windrow composting + sanitary landfill with gas utilization (MSP_COMP_SLF)

5.1.2.1 Global warming potential (GWP)

Global warming impacts will be mainly due to greenhouse gas emissions (GHGs) in CO_2, CH_4, and N_2O. The results for the global warming potential for the baseline scenario

S2 indicate that compositing + sanitary landfilling system is a very high impact on climate change GWP 1.7E+02 kg CO_2 equivalent, which is mostly due to the large amounts of waste that are currently being landfilled without any prior treatment given especially to the large amounts of organic waste that are disposed of. The emission of huge quantities of CH_4 is mostly from the degradation processes of organic waste in the landfill. Also, CH_4 is a highly potent greenhouse gas that accounts for up to 25 times the GWP than CO_2. The percentage contribution to GWP by the waste collection process is negligible compared to emissions from the landfill site. The results for the baseline scenario have been compared with the baseline emissions from the city of Nagpur, i.e., 1259.69 kg CO_2 eq. (Khandelwal et al., 2019) and city of Dhanbad, i.e., 1.24 E + 01 kg CO2 eq. respectively, and is found to be much higher than the Dhanbad and lower than the Nagpur city. This may be because the disposal site in Guwahati is quite old and subjected to heavy rainfall compared to the two cities, which affects the rate of decomposition in the landfills. Among the alternatives (S2, S3, and S4), while the emission control system in alternatives S2, S3, and S4 may cause any methane that is formed in the landfill to be utilized and also waste segregation can be done in other three scenarios, the emissions in both cases will be much lower (1.70 E+02 kg CO_2 eq, 4.31E+01 kg CO_2 eq and 23.29E+01 kg CO_2 eq. respectively) than the baseline due to the organic waste being diverted for biological treatment through composting. Although composting is a completely biological process, the net emissions of GHGs and fuel consumed are required by the different types of machinery in the compost plant. The GWP in the case of alternative S2 will be higher (1.70 E+02 kg CO_2 eq.) then the two previous scenarios due to the usage of fossil fuels such as diesel in various moving vehicles. The S3 (RDF_COMP_SLF) and S4 (MSP_COMP_SLF) had the least impact on global warming due to waste segregation, separation of organic waste, and utilization of landfill gases.

5.1.2.2 Acidification potential (AP)

The acidification potential arises mainly due to ammonia (NH_3), NO_x, and SO_2 emissions. The acidification potential of the baseline scenario is about 3.48E-01 kg SO_2 eq. However, when comparing the different alternate scenarios with the baseline, there will be significant impacts from the three scenarios compared to the baseline. In comparison, scenarios S3 and S4 had the greatest impact on AP. However, in scenario S2 (COMP_SLF) (3.93E-01 kg SO_2 eq), mainly due to releases of acidifying compounds like NO_x and SO_x in the form of inorganic emissions into the atmosphere during the composting process. S2 (3.93E-01 kg SO_2 eq), S3 (6.67 E-01 kg SO_2 eq), and S4 (4.29E-01 kg SO_2 eq) have a

higher impact on acidification compared with baseline scenario S1 due to compost and movement of vehicles to open dumping and the lowest acidification among all the scenarios because, in this case, sulphur and nitrogen compounds get oxidized in the lesser amount resulting in, the lower emissions of SOx and NOx. In another study in Dhanbad, S3 (RDF_COMP_SLF) had the lowest impact (2.72E-06 kg SO_2 eq.) on acidification compared with other alternative scenarios (Yadav and Samadder, 2018).

5.1.2.3 Eutrophication potential (EP)

Eutrophication of aquatic systems due to nutrient enrichment by compounds such as NH_3- N and total phosphorus will occur mainly from leachate pollution from landfills. As seen from the results, the eutrophication potential of the baseline scenario1 is about 1.31E-01 kg PO_4^- eq and, S2 (1.87E-01 kg PO_4^- eq) and S3 (1.87E-01 kg PO_4^- eq) shows the maximum potential for nutrient enrichment, while S4 shows the least (8.44E-02 kg PO_4^- eq) has the lowest eutrophication among the alternative scenarios owing to the presence of impermeable synthetic bottom liners in sanitary landfills. An explanation for the EP in S2 may be attributed to the excessive amount of nitrogen and phosphorus nutrient present in the compost and NH_3 gas both from the composting and landfill. However, with proper collection facilities and energy recovery, as seen in S3, the impact can be compensated.

5.1.2.4 Freshwater toxicity (W_t)

Toxicity-related impacts to water will be mainly in organic, inorganic, and heavy metal emissions measured in terms of kg 1,4-DCB equivalents. The baseline scenario's water toxicity impacts are much higher (1.05E-02 kg 1,4-DCB eq) than all the three alternate scenarios. This shows that heavy metals in the leachate samples analyzed significantly affect the quality of water bodies compared to inorganic or organic emissions, respectively. Among the alternate scenarios, S1, S2, and S3 had almost the same impact on water toxicity which may be mainly due to the emission of heavy metals from the landfill, but scenario S4 (1.00E-02 kg 1,4-DCB eq) had a better performance due to recycling and separating recyclables from the main waste stream.

5.1.2.5 Photochemical ozone creation potential (POCP)

VOCs, CO, and CH_4 are the main contributors to POCP. The S2 (1.37E-01 kg C_2H_4 eq had the greatest impact mainly due to the release of VOCs, methane, carbon monoxide, and other inorganic air pollutants. The major cause of VOCs is landfilling and disposal of waste. Comparing the results of POCP for the baseline scenario (2.55E-02 kg C_2H_4 eq) showed a low impact on POCP of the present study with the S2; however, S4 (3.77E-02 kg C_2H_4eq) is found to be much better than the other scenarios. The S3 (7.74E-02 kg C_2H_4 eq) and S4

(5.47 E-02 kg C_2H_4 eq) were less impacted than S2 due to sanitary landfilling, manual sorting of materials, and use of high calorific value products. This reduction in effect in S4 is attributed to avoiding methane pollution from the system by composting the food and residual in a sanitary landfill.

5.1.3 Life cycle costing Guwahati waste disposal strategies

Fig. 5.3 depicts the life cycle costing for Guwahati waste disposal strategies. The findings indicate that scenario 3 is the most expensive option, with scenario 4 coming in second and scenario 1 being the least expensive. In scenarios 2, 3, and 4, which are not considered, credits from produced electricity play a positive role in lowering overall costs. Scenarios 4 and 3 have similar investment costs, with scenario one having the lowest due to the lack of maintenance and operational costs. However, the disparity between scenarios is heavily affected by the activity expense. Scenario 3 and 4 had the highest operating costs, owing to the high initial expenditure and input energy. Therefore, scenario 1 produces the best economic outcomes, scenarios 2 and 4 come in second and third place, respectively.

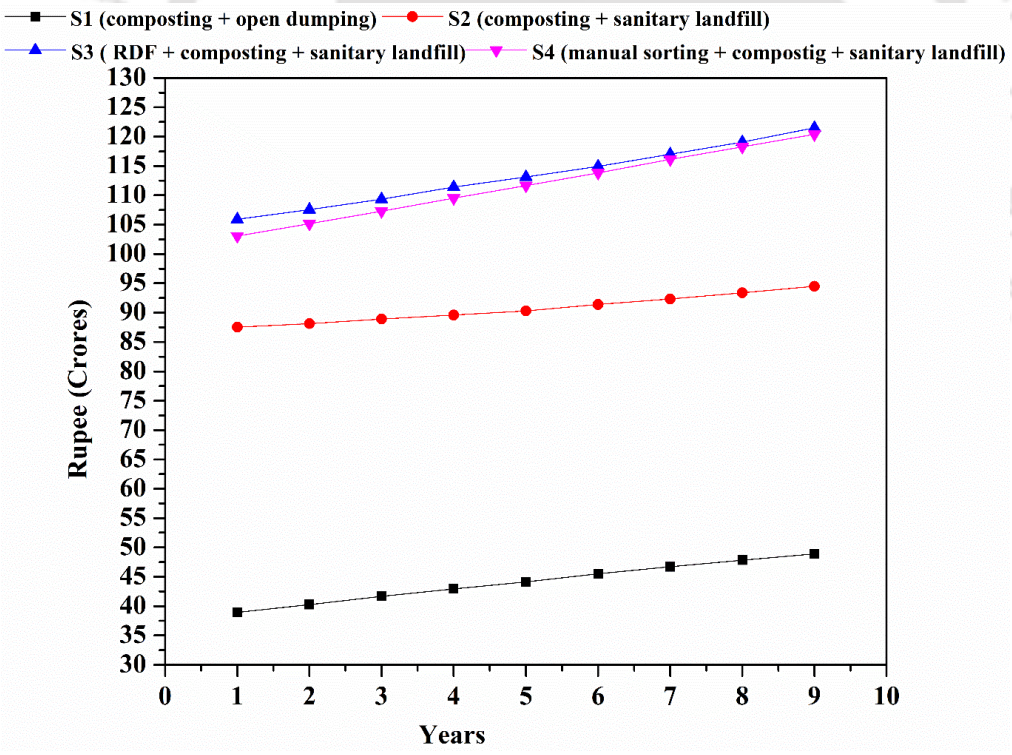


Fig. 5.3. The life cycle costing of waste disposal strategies

The average life cycle cost of 10 years for each waste treatment scenario were S1 (44.07 crores), S2 (90.68 crores), S3 (113.31 crores), and S4 (111.78 crores), respectively. S1 has

an economic advantage compared to other scenarios because a developing country like India has a high amount of food waste, while future scenarios improve the overall sustainability compared to the current situation. S1 (COMP_OD) performs better because of less investment in equipment, maintenance cost, labor cost, and other expenses. S4 signifies the expensive waste treatment technology because of high investment in equipment, maintenance cost, labor cost, and other expenses.

5.2 SUMMARY OF PHASE-I (OBJECTIVE –I)

- The findings showed that the baseline situation had the least effect on acidification (1.80E-01 kg SO_2 eq), the moderate impact on eutrophication (4.00E-02 kg PO_4^{3-} eq). Climate change has a maximum impact (7.30E+00 kg of CO_2 eq) because the maximum amount of CO_2 and CH_4 was released from the composting and open dumping methods.

- **Global warming potential (GWP)**

The S3 (RDF_COMP_SLF) and S4 (MSP_COMP_SLF) had the least impact on global warming due to waste segregation, separation of organic waste, and utilization of landfill gases.

- **Acidification potential (AP)**

S2 (3.93E-01 kg SO_2 eq), S3 (6.67 E-01 kg SO_2 eq), and S4 (4.29E-01 kg SO_2 eq) have a higher impact on acidification compared with baseline scenario S1 due to compost and movement of vehicles to open dumping and the lowest acidification among all the scenarios because, in this case, sulphur and nitrogen compounds get oxidized in the lesser amount resulting in, the lower emissions of SO_x and NO_x.

- **Eutrophication potential (EP)**

S2 (1.87E-01 kg $PO4^-$ eq) and S3 (1.87E-01 kg $PO4^-$ eq) shows the maximum potential for nutrient enrichment, while S4 shows the least (8.44E-02 kg $PO4^-$ eq.) has the lowest eutrophication among the alternative scenarios owing to the presence of impermeable synthetic bottom liners in sanitary landfills.

- **Freshwater toxicity (W_t)**

S4 (1.00E-02 kg 1,4-DCB eq) had a better performance due to recycling and separating recyclables from the main waste stream.

- **Photochemical ozone creation potential (POCP)**

The S3 (7.74E-02 kg C_2H_4 eq) and S4 (5.47 E-02 kg C_2H_4 eq) were less impacted than S2 due to sanitary landfilling, manual sorting of materials, and use of high calorific value products.

❖ **Life cycle costing Guwahati waste disposal strategies**

The average life cycle cost of 10 years for each waste treatment scenario were S1 (44.07 crores), S2 (90.68 crores), S3 (113.31 crores), and S3 (111.78 crores), respectively. Therefore, S1 has an economic advantage compared to other scenarios because of less maintenance and operating cost, while future scenarios improve the overall sustainability compared to the current situation.

CHAPTER

6

Biochemical Evaluation of Landfill Leachate and Gas Generation Dynamics

This chapter discusses the assessment of landfill leachate and landfill gas dynamics under actual conditions of a landfill. It is difficult to generalize the chemical composition of leachate, as leachate varies over time and depends on waste composition and climatic conditions and is divided into two parts, i.e., part-I (objective-II) (operated without rainfall) and part-II (objective-III) (operated with rainfall). The research focuses on the real-time scenario of existing open dumpsite conditions, characterizes the leachate and gas composition, and examines the effect of rainfall on leachate and gas dynamics generated from MSW deposited in a large-scale anaerobic landfill simulator with time. The first-order exponential decay model was developed for leachate constituents of fresh mixed MSW with respect to time to predict the future concentrations of leachate based on laboratory data.

6.1 PART-I: RESULTS AND DISCUSSIONS OF SIMULATED REACTOR OPERATED WITHOUT RAINFALL

6.1.1 Initial characterization of mixed unshredded MSW

The initial characterization was done for deposited MSW in the large-scale simulated landfill reactors. Two reactors were fabricated and coded as R1 and R2. In this study, reactor (R1) contained a waste composition of 73% wet and 27% of dry waste; similarly, R2 contained 68% wet and 32% dry waste. Initially, 100 kgs of waste were mixed according

to waste composition obtained from experts' design since the amount of waste that can be accommodated in the reactor is unknown. As a result, the R1 received 292 kg of wet and 102 kg of dry waste and 136 kgs of dry and 64 kgs of wet fed into the R2. Moisture content is a significant parameter that varies with solid waste and is an environmental factor to consider when producing leachate. To better understand solid waste decomposition behavior, mixed solid waste's physical and chemical composition was investigated. (Rafizul and Alamgir, 2012) reported that when a landfill contains a high amount of organic waste of 93 (w/w), the moisture content of waste ranges from 52-65%. R1 had a moisture content of 79.6%, and R2 had a moisture content of 54.26% in this research. The moisture content findings for both the reactors show that moisture quality is affected by the ratio of organic waste. The average pH for R1 was 5.64 and 6.94 for R2, and it was reported that there was a high volume of food waste in landfills, ranging from 4.7 to 6.1 (Sundberg et al., 2011). The solid waste samples were analyzed in duplicates (n=2). The mean and standard deviation values of pH, moisture content, total organic carbon, and volatile Solids are represented in Table 6.1. This MSW characterization is essential because it not only provides the framework for all subsequent decomposition processes and analysis of leachate, but it also represents India's highest wet: dry ratio as well as lowest wet: dry ratio.

Table 6.1. Initial characterization of fresh MSW

Parameters	R1 (mean ± stdev)	R2 (mean ± stdev)
Moisture content (%)	79.6 ± 1.41	55.2 ± 5.09
pH	5.64 ± 0.26	6.94 ± 0.01
EC (mS/cm)	5.82 ± 1.22	3.6±0.45
Density (kg/m^3)	676.24 ± 196.69	316.873 ± 22.32
VS(%)	56.22 ± 1.15	64.05 ± 7.84
TOC(%)	31.23 ± 0.64	35.83 ± 4.360
*Wet (kg)	292	136
*Dry (kg)	108	64

*Amount of solid waste fed into the reactor

6.1.2 Influence of tropical climate (dry season) on the degradation of varying solid waste composition

Analysis of Leachate quality in terms of BOD_5, COD, pH, EC, solids, ammoniacal nitrogen (NH_4^+-N), nitrate (NO_3^-), nitrite (NO_2^-), TKN, sodium (Na), potassium (K), and calcium (Ca), and heavy metals by time in relation to variations of humid climate, weekly monitoring of leachate quality was performed in a research lab. However, simulated landfill examination consists of the dry season, age of MSW, and the operational mode of landfill reactor also played a significant role, which leads to a variation of leachate quality. Therefore, a detailed way of variation of leachate quality from reactor 1 (L1) with time was discussed. There is no leachate production in the landfill. It contains 38% of wet waste and 62% of dry waste due to the high amount of dry waste that may not reach field capacity. It signifies that developed countries may have less problem handling the leachate issue because of low biodegradable waste.

6.1.2.1 Landfill leachate color

The physical appearance of leachate samples was orange, brown or dark brown, or black color. Leachate produces a malodorous smell, mainly due to the presence of organic acid, which arises because of the high concentration of organic matter when decomposed (McBean and Frank,1995). High organic substances are responsible for the high concentration of color in a landfill. Such leachate contains high levels of organic substances such as humic and fulvic compounds, indicated by leachate color (Asadi, 2008). The humic substances are natural organic matter composed of complex structures of polymerized organic acids, carboxylic acids, and carbohydrates (Adhikari et al., 2013). Fig. 6.1 shows variation in color of leachate with time (weeks) and represents the degradation of solid waste inside the landfill reactor. As literature reported, leachate varies from pale yellow to dark black due to humic and fulvic acids. Young leachate consists of a high range of humic acids and is predominantly constructed of linked aromatic rings with numerous phenolic and carboxylic acid functional groups and linear and branched alkyl substituents. Summary of some of the main changes in chemical characteristics of humic substances as they undergo humification has been prepared by (Van zomeren, 2008). Fig. 6.2 shows the degree of degradation based on color. The blackish color represents the organic matter dead in a landfill, other than actively growing microbes and degraded material that washes out through landfills. The yellow color represents the initial degradation of waste that carries

the organic compounds through the waste out of the landfill system. Therefore, the degree of degradation could be seen in the physical appearance of leachate color.

Fig. 6.1. Variations in landfill leachate color with time

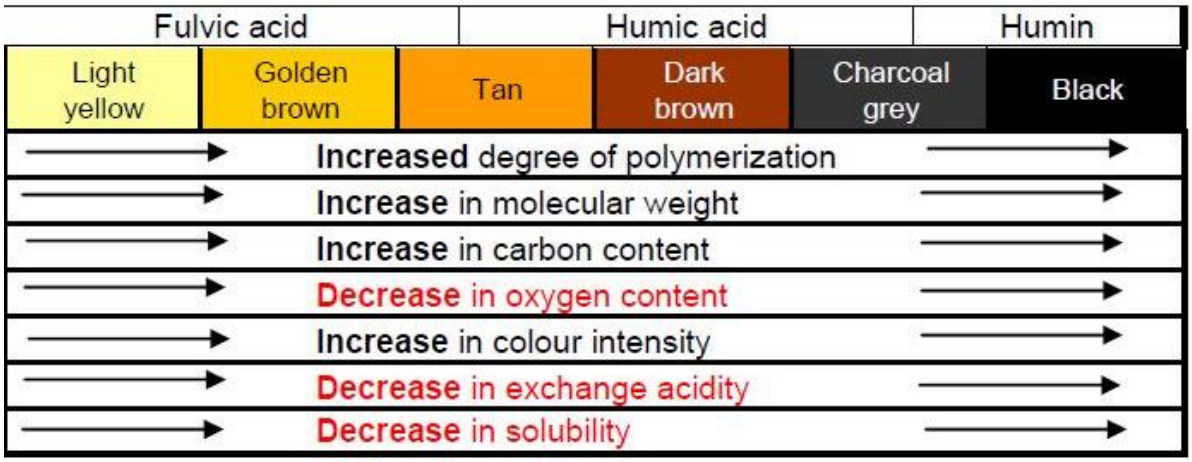


Fig. 6.2. General properties of humic substances

6.1.2.2 Landfill leachate quantity

The current total quantity of leachate was 217.6 liters. The leachate was drained each week completely, and the volume was measured. The initial leachate quantity was 22 liters and gradually decreased to 1.5 liters. It indicates more biodegradable waste present in landfills leads to a high leachate generation due to the degradation of solid waste and available moisture within the waste. Leachate quantity decreased with time due to the major source of moisture loss from the landfill was the leachate drainage through degradation. Other mechanisms of degradation like sorption and precipitation took place inside the waste

mass. Therefore, moisture plays a significant role in leachate generation from waste landfills. The trend of leachate quantity is shown in Fig. 6.3.

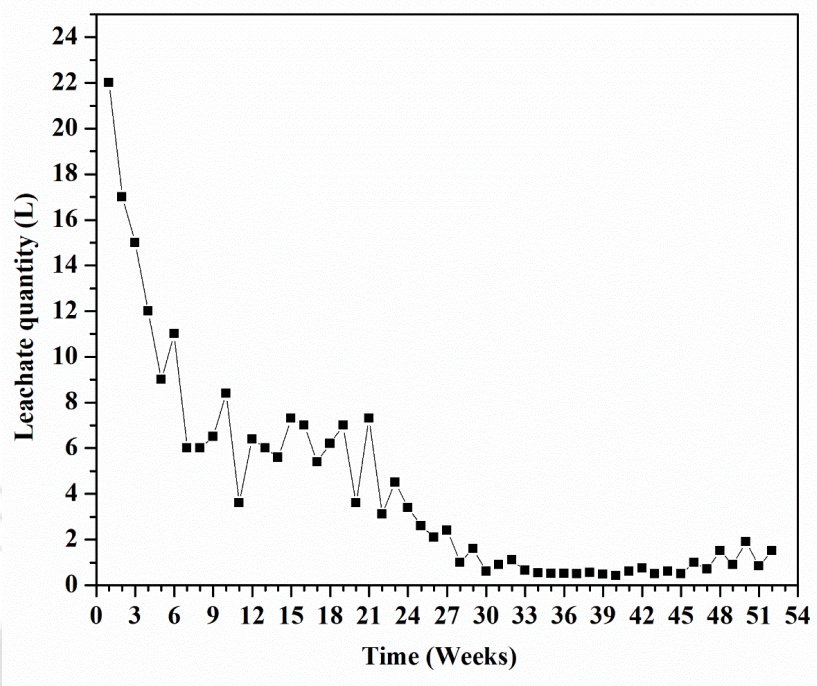


Fig. 6.3. Plotting of landfill leachate quantity against with time

6.1.2.3 Relation between pH and electrical conductivity (EC)

The variation of pH and EC profile with time from the simulated landfill reactor is illustrated in Fig. 6.4. The pH of the young leachate was observed in this study ranged from 5.33-6.25. The initial weeks of pH were decreased due to volatile fatty acids (VFA) like carboxylic acids, leading to a pH drop. Acid phase leachate is chemically active and enhances many compounds' solubilization since it is acidic, subsequently, a rise in pH to 6.25 (Kjeldsen et al., 2002). The literature suggested that pH changes in the landfill due to calcium carbonate are used to buffer pH changes throughout life (Statom, R. et al., 2004). The researcher (Rafizul and Alamgir, 2012) operated two MSW cells in an open dump and sanitary lysimeter. The results indicated that the initial pH of leachate was in the acidic phase due to the accumulation of VFA, later increased to alkaline. Therefore, it could be concluded that the pH of leachate is also close to neutral, which is consistent with the literature (Tatsi et al., 2003). Therefore, it specifies that pH is buffered naturally. There were no significant improvements in the leachate's pH, assessed between 5 and 6. The validity of this study's findings and those results of closed lysimeter and obtained initial range of pH 5-6 and stabilized to 6.25 after one year of operation (Trankler et al., 2005).

The results followed the data specified by (Visvanathan et al., 2007). The landfill's pH has risen to near neutral as VFA concentrations have decreased, but alkaline pH has not been reached. This could be due to a large amount of organic waste in the landfill and the continued stimulation of acidogenic activity. EC values increased with the decreased over time, as shown in Fig. 6.1. The EC values varied from 14.57-20.60 mS/cm, where high biological activity is related to the formation of carbonates and bicarbonates with their counterions. After a decrease in biological activity, dilution becomes more dominant, and further leaching results in decreasing buffer capacity, pH, and conductivity (Wang and Pelkonen, 2009). EC defined total organic and inorganic matter in the leachate and used it to evaluate salinity and mineral content. The EC values in the dry season were higher than the rainy season due to dilution of rainfall, and a similar trend of EC values (12.2-13 mS/cm) was obtained by (Mangimbulude et al., 2009). A simulated anaerobic landfill showed the highest value of 20.21 mS/cm, while the lowest pH was 6.25. it mentioned that EC decreases while pH increases, but the same trend has not been followed throughout the process since metals in anaerobic environments appear to form hydroxides, and most of the compounds are insoluble (Rich et al., 2008). EC values were reduced with time due to the flushing out of inorganic compounds and found that EC value was highest during the dry season (Qasim and Chiang, 1994; Yuen et al., 2001).

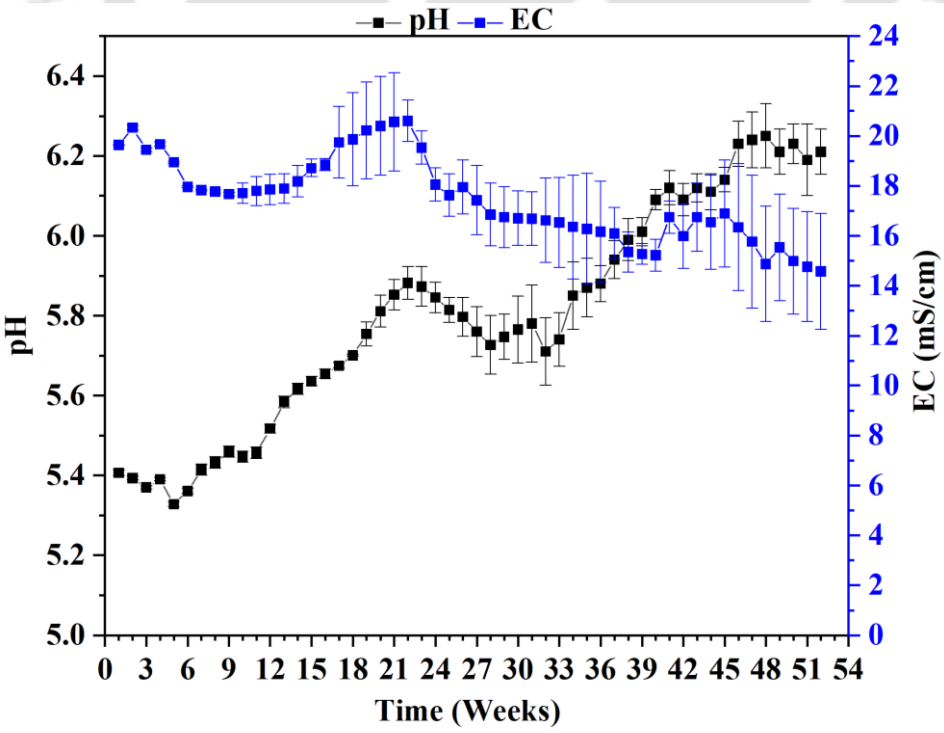


Fig. 6.4. Weekly variation of pH and EC with time

6.1.2.4 Relation between solid particles

Variability of solid particles in terms of total solids (TS), total dissolved solids (TDS), total suspended solids (TSS), total fixed solids (TFS), and total volatile solids (TVS) ranged from 3,163-57,540.67, 2,332-39,370, 679-21,882.67, 1,009-4,5911.67 and 2,467-19,298 mg/L respectively as shown in Fig. 6.5. TDS and EC found a relationship in this analysis, with high TDS and conductivity found in the early weeks of waste degradation, followed by a decrease in TDS due to the flushing out of inorganics from the waste. During the acidic phase of degradation, a high amount of VFA is produced, leading to breaking down the organic matter and precipitation of soluble organics and insoluble inorganic compounds. Waste stabilization rates observed in terms of percentage from the simulated landfill were 94.5, 96.9, 94.1, 87.2, and 97.8, respectively, by the end of 52 weeks. According to the researchers, TS concentrations were initially high, but as the landfill pH increased to a near-neutral state, solids concentrations were decreased (Rafizul and Alamgir, 2012; Yuen et al., 2001). As the age of leachate processing rises, organic compounds degrade faster than inorganic compounds. (Naveen et al., 2014). A similar trend was followed in the current study as significant solids concentrations were reduced due to the high degradation rate.

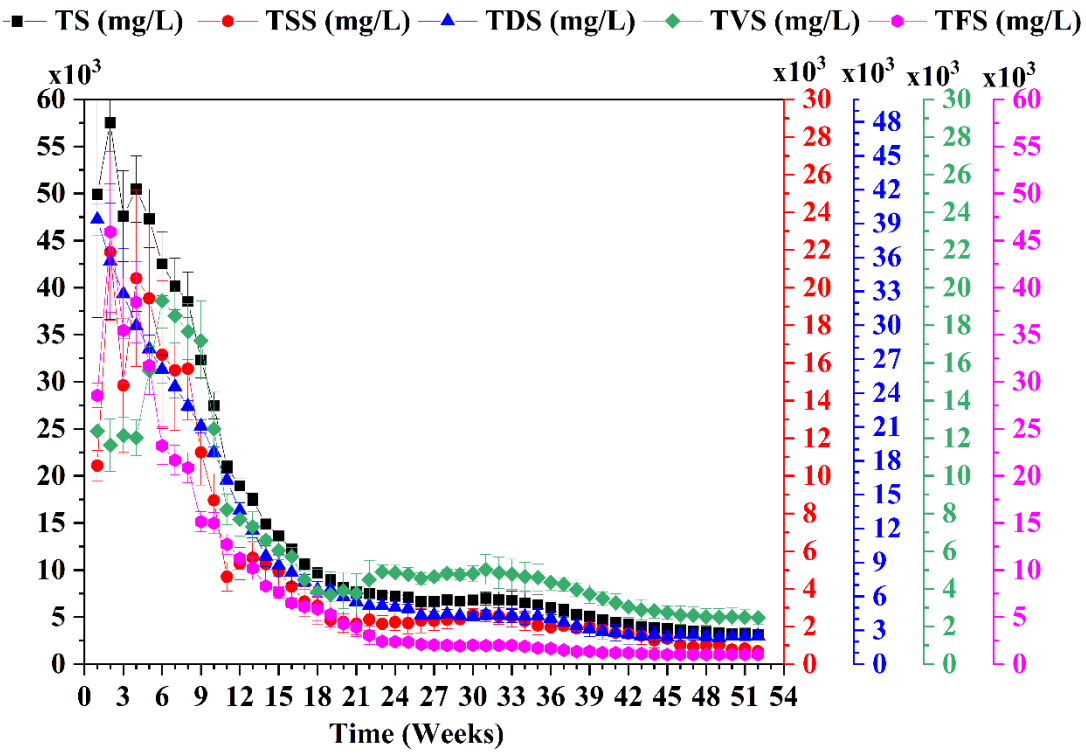


Fig. 6.5. Weekly variation of solids with time

6.1.2.5 Relation between Volatile fatty acids (VFA) and pH

Variability of VFA and pH of leachate is illustrated in Fig. 6.6. The landfill reactor's young leachate is acidic, signifying that MSW is in the acid phase for the degradation process. In this study, VFA concentrations ranged from 2,464- 37,500 mg/L. (Swati et al., 2011) noticed a similarly high level of VFA initially ranging from 14,000–30,000 mg/L. Authors have observed a similar drop in pH caused by the formation of VFA during the first weeks of waste degradation; later, when pH increases with VFA decrease with an elapsed time (Khattabi et al., 2002a). The percentage removal of VFA from the landfill was 93.4, indicating pH and VFA play a key role in the decomposition of solid waste. As evidenced by the continued high COD concentrations, the acidic environment inhibited methanogenesis (Sanphoti et al., 2006). Higher COD concentrations (19,056-92,160 mg/L) and VFA (2,464-37,500 mg/L) since there isn't any water to dilute and wash out the enzymatic and acidic compounds. As a result, a build-up of high organic and a gradual decomposition of waste (Petchsri et al., 2006). Even though the simulated landfill reactor had a high level of organic waste and moisture, still water addition is required to provide microorganisms with favorable conditions. Acid production increases when the solid waste's moisture content is high. Therefore, a high VFA content is expected (Wang et al., 2012).

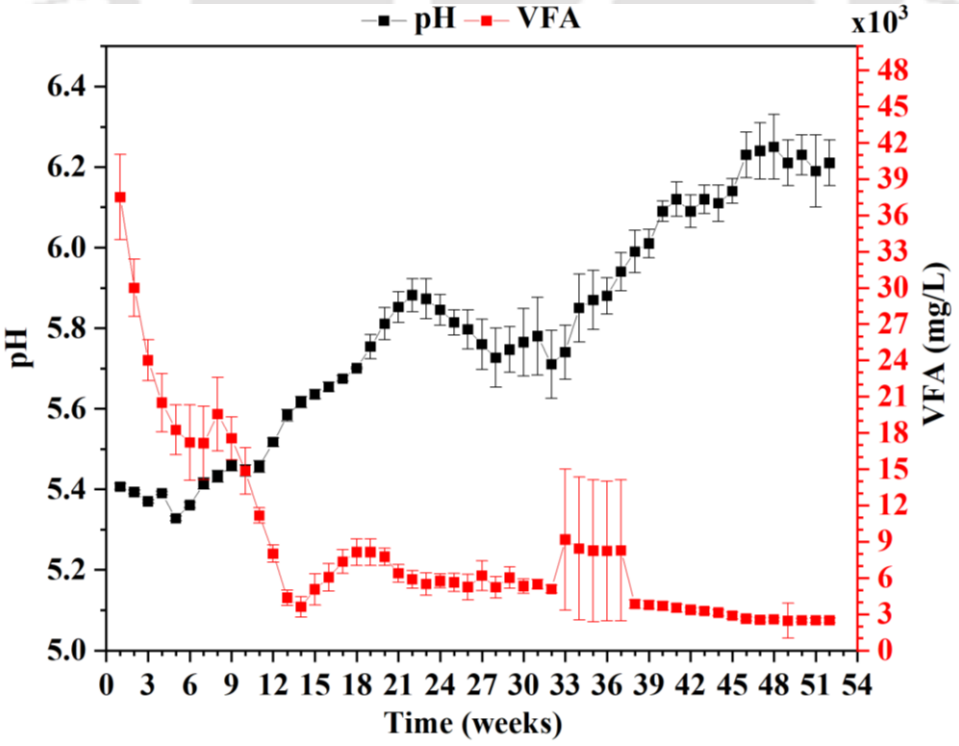


Fig. 6.6. Weekly variation of VFA with time

6.1.2.6 Relation between biochemical oxygen demand (BOD_5) and landfill temperature

The most commonly used organic pollution parameter is BOD_5, and it's applied to wastewater. Variation of organic components is very high at the initial stages of biodegradation of waste, ranging from 13980-50250 mg/L, as shown in Fig. 6.7. BOD_5 concentrations remained high throughout the dry period and then gradually decreased. Higher temperatures (21.30-49.40^oC) accelerated the degradation of organics, which resulted in higher BOD_5 concentrations at the early stage of degradation after filling was observed in this study. (Wang and Pelkonen, 2009) reported that during thermophilic temperatures, the BOD_5 stabilization rate from landfills was 62.3%, and according to the study's findings, it was 72.2%. BOD_5 concentrations increase sharply as leachate passes through the waste during this phase, and pH drops below 6. The formation of dissolved organic compounds is exceptionally high; the decreasing curve begins as the dissolved organic matter is absorbed by bacteria and released from the landfill. (Kjeldsen et al., 2002).

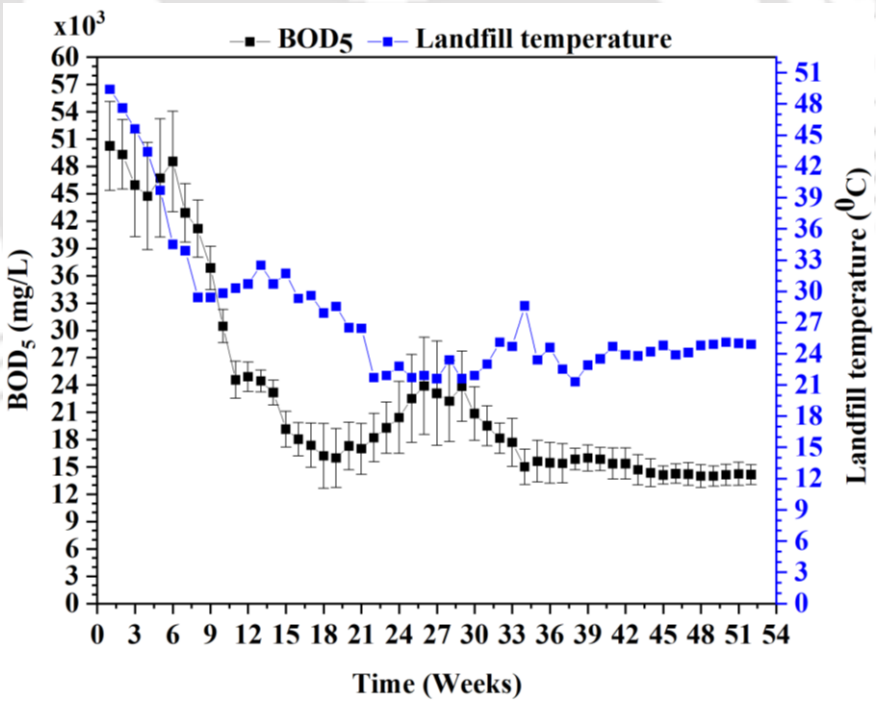


Fig. 6.7. Variable BOD_5 concentrations and landfill temperature with time

The landfill temperature at the initial stages was 49.4^oC, which indicates the thermophilic stage; later, the temperature decreased to below 30^oC due to lack of moisture, mass transfer, enzymatic access, microbial activity (microbial activity is also decreasing because of the reduced availability of substrate and increased concentration/solubilization

of inhibitors.) can be reduced in landfills. This validates the commonly reported idea that temperature plays an essential role in activating bacterial enzymes that degrade organic matter (Khattabi et al., 2002a). Fig. 6.7. Variable BOD_5 concentrations and landfill temperature with time BOD concentrations were decreased as pH increased and bacteria consumed acids. Authors have observed an 82-86% reduction of BOD concentrations during six months of landfill lysimeter operation as a bioreactor (recirculation of leachate) (Chiemchaisri et al., 2009; Kouzeli-Katsiri et al., 1999).

6.1.2.7 Relation between chemical oxygen demand (COD) and landfill temperature

COD is a measure of the combined organic and inorganic content of a substance. (Drever, 1988). The COD concentrations ranged from 19,056-92,160 mg/L was observed in this study, as shown in Fig. 6.8. COD levels peaked at the initial phases of waste degradation, where the temperature (49.4^{o}C) of the landfill was high. COD concentrations decreased at different rates depending on temperature, implying that leachate concentrations decreased faster at higher temperatures (Wang et al., 2012). Generally, a lesser volume of more concentrated leachate is produced during the dry season since only a small amount of rainwater is input. Therefore, higher COD and VFA at initial weeks (6-9) of degradation as VFA contributes to COD. The findings show that due to a lack of moisture for dilution and leaching out of acidogenic compounds, the accumulation of high-strength organic leachate within the waste reduces waste decomposition. According to the findings, pH is inversely proportional to COD values, increasing as COD values decrease due to decreased VFA concentrations over time. Other studies were reported that COD concentrations were ranged from 7,640-60,000 mg/L for open dump lysimeter (Chian and Dewalle, 1976; Rafizul and Alamgir, 2012). It is important to note that the initial concentrations were high due to the organic material's degradable nature; organic material degrades much faster than inorganic compounds (Ehrig, 1983). Faster degradation could be attributed to higher temperatures reported in this study at 49.4^{o}C during the first weeks. The major organic components of leachate are humic acids (humic and fulvic). Since they withstand degradation and absorb heavy metals, large levels of non-degradable humic substances may adversely affect landfill leachate treatment (Calace and Petronio, 1997; Christensen et al., 2001). Fresh leachate COD concentrations ranged from 15,000-92,000 mg/L due to high organic waste in the landfill during the summer (Zhao et al., 2013). The results of this study revealed a similar range of COD concentrations. The stabilization rate of COD concentrations in fresh leachate was 79.3%.

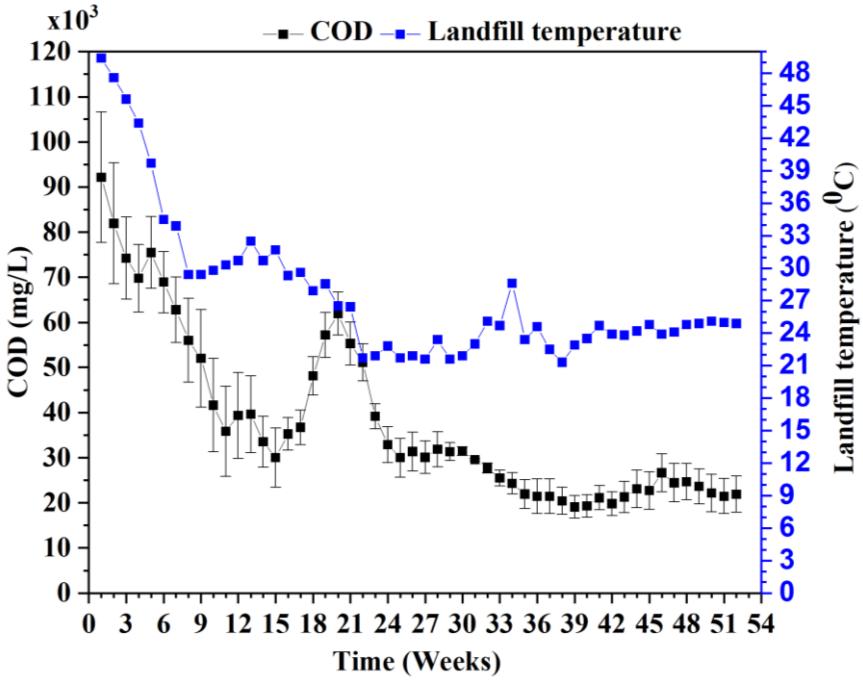


Fig. 6.8. Variable COD concentrations and landfill temperature with time

6.1.2.8 BOD/COD ratio

BOD/COD ratio is a fraction of completely organic material to biodegradable material (Reinhart and Grosh, 1998). Fig. 6.9 shows BOD/COD ratio in fresh leachate ranged from 0.29-0.84. Other researchers have reported that the unstabilized fresh waste of BOD/COD ratio was high (around 0.9), and The ratio of moderately stabilized leachates ranges from 0.1 to 0.5 (Swati et al., 2011; Tränkler et al., 2005). Thus, the fresh/young leachate of the present study represents young unstabilized to partially stabilized leachate.

6.1.2.9 Relation of nitrogen compounds ((ammonical nitrogen (NH_4^+–N), nitrate (NO_3^-), total Kjeldahl nitrogen (TKN), and nitrite (NO_2^-)) in fresh leachate

Fig. 6.10 shows that variable nitrogen compounds (NH_4^+– N, NO_3^-, TKN, and NO_2^-) in fresh leachate with time ranged from 111.27-455.63 mg/L, 11.12-205.20 mg/L, 0.05-0.59 %, and 45.97-210.54 mg/L, respectively. The results were in accordance with a difference of Ammonical nitrogen in fresh leachate during the summer season was 39 to 568 mg/L. TkN represents the sum of organic nitrogen, and NH_4^+– N signifies inorganic matter. The landfill has a low organic nitrogen concentration and a high ammonical nitrogen concentration. Another critical constituent in the leachate is nitrogen, which can pollute water and soil. Other authors have found that NH_4^+– N values were ranged from 500-1,400, 283-2,040 mg/L for different landfills (Bagchi, 1994; Aghamohammadi et al., 2007; Aziz et al., 2010). The fresh landfills ranged from 10-800 mg/L and 20-40 mg/L for matured

landfills, respectively (Tchobanoglous et al., 1993). Higher temperatures accelerated the removal of ammonical nitrogen from organic matter, resulting in higher NH_4^+-N concentrations in the leachate. However, the NH_4^+-N concentrations showed a slower decline, but more attention is required to remove them from the landfill due to a consistent long-term trend in leachate (Wang and Pelkonen, 2009). The study results showed higher temperature-driven organics' high degradation and washout of NH_4^+–N through leachate. Furthermore, the temperature tends to be a significant determinant of nitrogen dynamics in an MSW landfill, and temperatures above 40^oC limit ammonia and nitrite-oxidizing agents' activity. The average temperature rises and falls at various depths inside the landfill, and the highest temperature was observed up to 48 degrees Celsius. The TKN concentration levels were also decreased while pH reached close to neutral. The release of soluble nitrogen from waste into leachate lasts longer than the release of dissolved organics. As a result, ammonical nitrogen and TKN decrease with an increased concentration due to hydrolysis and fermentation of nitrogen compounds in the biodegradable form in waste (Abbas et al., 2009). Leaching is the only way to decrease nitrogen in landfills because there is no degradation pathway (Tchobanoglous et al., 1993). High concentrations were found in fresh leachate is 161 mg/L. A small amount of ammonium would be nitrified during the leachate passed through the solid waste in landfills, which is discussed (Karthikeyan et al., 2008b).

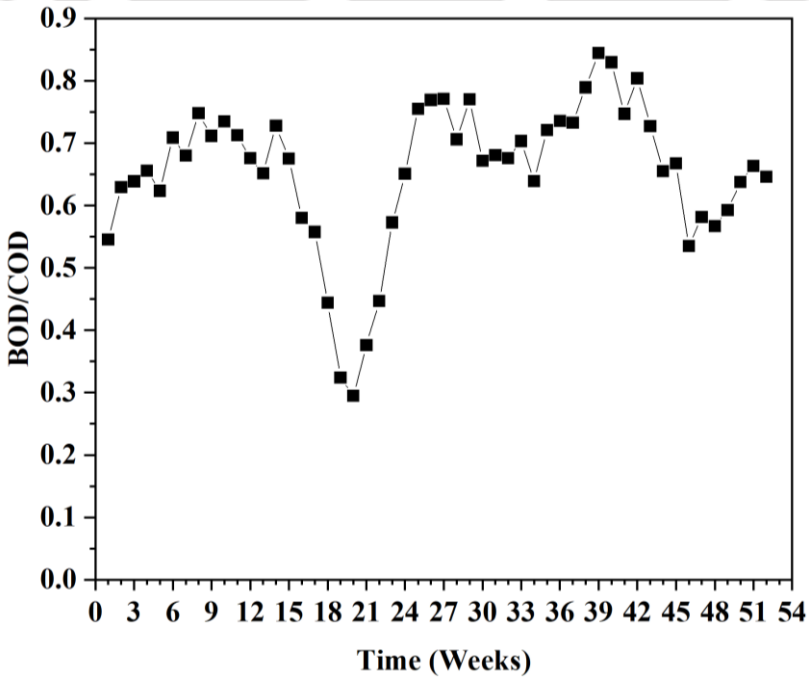


Fig. 6.9. Variable BOD/COD ratio in fresh leachate with time

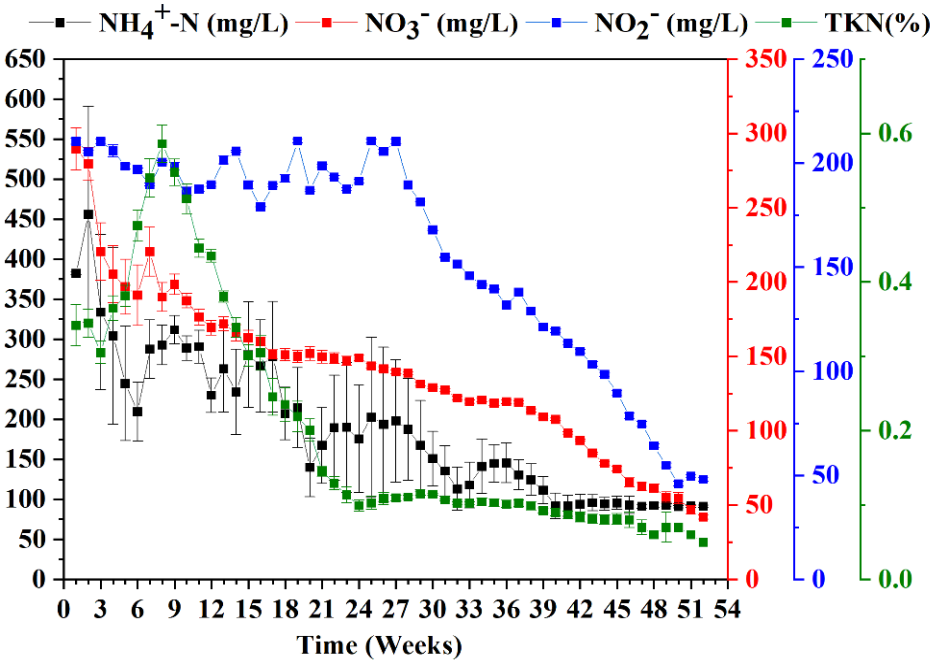


Fig. 6.10. Variable nitrogen compounds in fresh leachate with time

6.1.2.10 Relation of sodium (Na), potassium (K), and calcium (Ca) in fresh leachate

Fig. 6.11 shows the variation of micronutrients like Na, K, and Ca in fresh leachate with time. The average Na, K, and Ca concentrations ranged from 211.98, 1675.11, and 982.29 mg/L, respectively. These are the essential cations present in the landfill leachate. The concentration of these ionic species in leachate is dependent on the waste volume composition. The most important stage of landfill stabilization is extracted from waste material through mass transfer processes. (Christensen et al., 2001). Cation concentrations are lower during the methanogenic process due to higher pH, promoting sorption and precipitation while lowering dissolved organic matter and potentially complex cations (Kjeldsen et al., 2002). A decreasing trend with time was observed in this study due to the washout effect by leaching (Ehrig, 1989). Potassium is ionized quickly, degradable organic matter, and dissolved during decomposition, while Na and K are leached from fresh waste. The potassium ion has the simplest forming potential and is correlated with the other ions leached from the waste. Calcium is another essential ion in establishing the buffering ability of leachates during waste degradation. Its buffering effect throughout the waste degradation is well documented (Karthikeyan et al., 2008b).

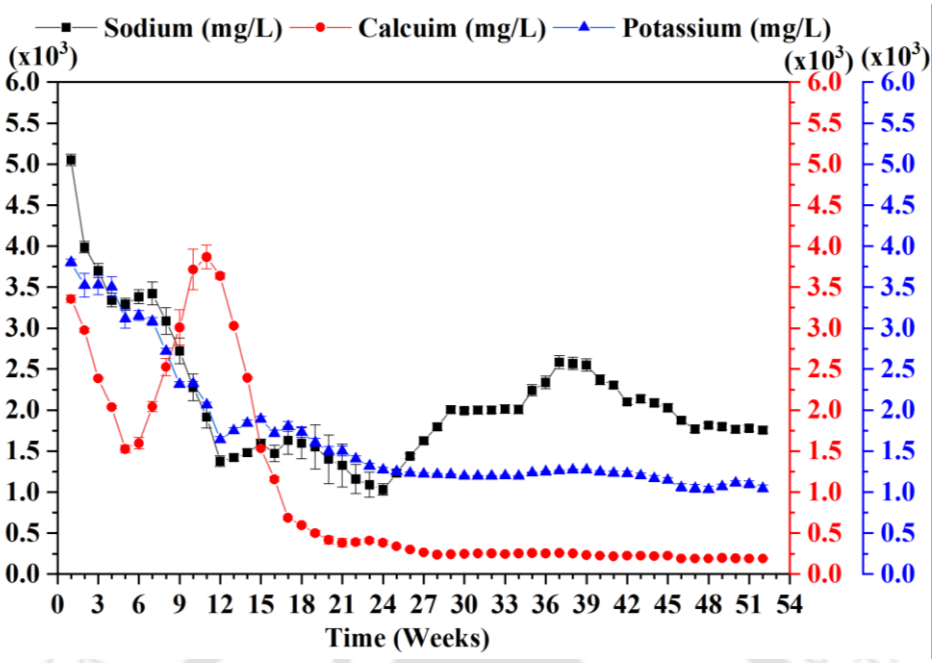


Fig. 6.11. Variable micronutrients in fresh leachate with time

6.1.2.11 Relation of inorganic nonmetallic constituents (Cl^-, PO_4^{3-}, SO_4^{2-} & F^-) in fresh leachate

Fig. 6.12 shows Inorganic Nonmetallic Constituents (chloride (Cl^-), phosphate (PO_4^{3-}), sulfate (SO_4^{2-}) & fluoride (F^-)) in fresh leachate with a function of time. The initial concentrations were observed to be high due to more waste degradation chanced during the acidic phase. The Cl^-, PO_4^{3-}, SO_4^{2-} & F^- concentrations values ranged from 21.32–131.82, 50.43-123.86, 13.65-368.80, 13.43-493.96 mg/L respectively. Chloride is a very moderate anion in leachate that is only oxidized during dilution. The study results showed that fresh leachate chloride (Cl-) increased after being deposited in an MSW landfill. (Rafizul and Alamgir, 2012) reported that initial concentrations were high, with elapsed time concentrations decreasing because the high amount of leachate production leads to chloride dilution. In this context, Cl^- is a non-biodegradable compound, but the variation of its concentrations was decreased slowly due to the washout effect (Rafizul et al., 2009). It can be concluded that COD and Cl^- are proportional because COD decreases, and Cl- is also the same with time. Three parameters were found to be high, including COD, pH, and chloride, but pH was found to be low. It was found that pH increases, concentrations of COD, and Cl^- were low. However, it illustrates that the study's findings were valid with the results obtained in the case of a closed lysimeter by (Rafizul and Alamgir, 2012; Tränkler et al., 2005). SO_4^{2-} to Cl^-, ratio and pH in leachate increasing and decreasing with

time from deposited in closed landfill simulator. It is noted that sulfate–chloride increases up to the 10th week and decreases continuously because of a gradual decrease in sulfate, while the chloride decreases slowly. SO_4^{2-}/Cl^- ratio is a biological process, sulfate degradation in the first phase before methane production starts. In anaerobic conditions, sulfate reduction rapidly due to SO_4^{2-} is reduced to S^{2-} and then precipitates with various metals. SO_4^{2-} decreases over time, which corresponds to a reduction in VFAs. (Chian and Dewalle, 1976). After a certain time under anaerobic conditions, sulphate-reducing bacteria established themselves and used the hydrogen from anaerobic corrosion as an electron donor to reduce sulphate with the associated release of hydrogen sulphide. Sugars or organic acids can stimulate microorganisms to reduce SO_4^{2-} to S^{2-} by the following reaction in Eq. 6.1.

$$SO_4^{2-} + 2CH_2O \rightarrow H_2 + 2HCO_3^- + S^{-2} \qquad 6.1$$

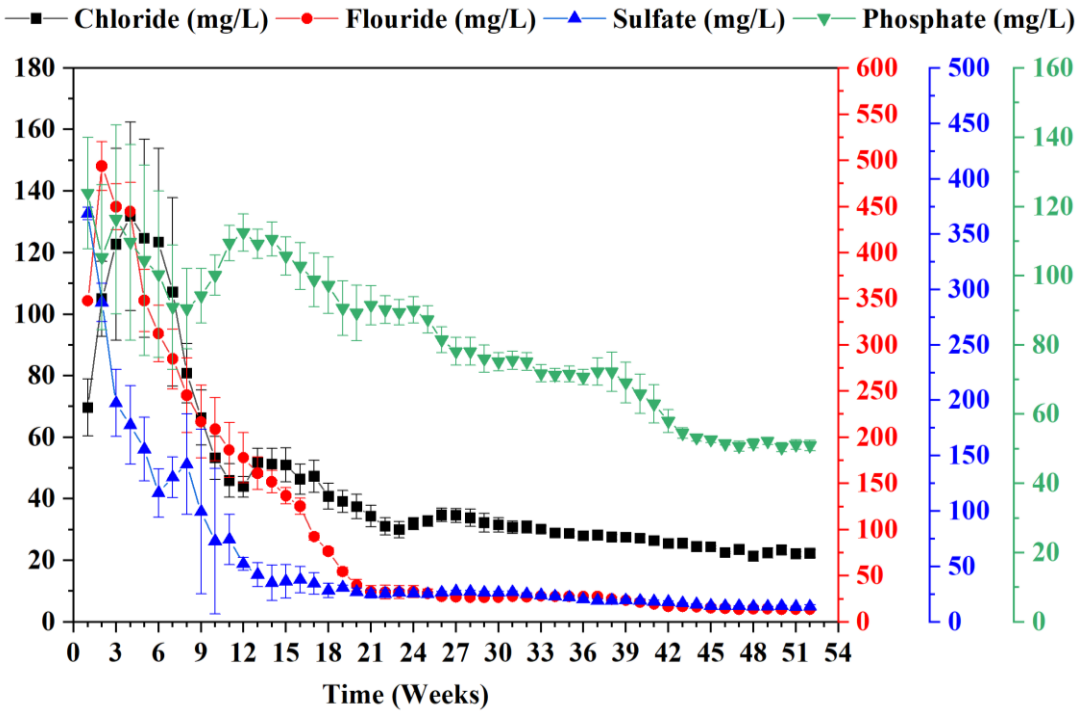


Fig. 6.12. Variable inorganic, nonmetallic constituents in fresh leachate with time

6.1.2.12 Relation of heavy metals iron (Fe), manganese (Mn), magnesium (Mg), zinc (Zn), copper (Cu), cadmium (Cd), and chromium (Cr) in fresh leachate

Fig. 6.13 shows the variation of heavy metals in fresh leachate. The heavy metals concentrations recorded from fresh leachate were ranged from 9.76-400.15, 96.89-341.58,

1.02-19.25, 0.43-11.654, 0.02-15.8, 0.02-1.6, 2.12-47.25 mg/L for Fe, Mg, Mn, Zn, Cr, Cu, and Cd respectively. By increasing concentrations in the transient aqueous medium, many processes, including complexation to inorganic and organic linkers and sorption to colloids, can mobilize heavy metals and significantly immobilize heavy metals from the landfill due to sorption and precipitation. Also, metal solubility with the sulfides and carbonates is low and capable of forming precipitates with heavy metals like Cd, Ni, Zn, Cu, and Pb (Christensen et al., 2001). The simulated landfill's initial pH was acidic, and heavy metal concentrations were high during this period. As time progressed, metal concentrations were decreased because pH was close to neutral. Organic matter has a high sorptive potential at neutral to high pH, according to researchers. (Christensen et al., 1994; Ishchenko, 2018; Kjeldsen and Christophersen, 2001; Reinhart and Grosh, 1998). In acidic conditions, a decrease in heavy metal solubility is possible because of complex anionic adsorption (Flyhammar and Hakansson, 1999). The order of heavy metals in this study was Fe>Mg>Cd>Mn>Zn>Cu>Cr. The authors investigated that effective decomposition of organic matter can increase the solubility and mobility of heavy metals in landfills by forming dissolved compounds of high acids with heavy metals (Ciavatta et al., 1993). However, the analysis may conclude that a high organic matter content in MSW supports the leaching of heavy metals from a landfill body. An increased concentration of OH^- would be favorable to form hydroxide precipitates with heavy metals, such as $Mg(OH)_2$, Fe(OH)2, $Fe(OH)_3$, etc.; the reaction can been described as follows in Eq. 6.2.

$$Me^{n+} + OH \rightarrow Me(OH_n) \quad 6.2$$

Carbonate (CO_3^{2-}) exhibits acid-base behavior, and reactions occur relevant to pH. The process can be described in Eq. 6.3 and 6.4.

$$H_2CO_3 + OH^- \rightarrow HCO_3^- + H_2O \quad 6.3$$

$$HCO_3^- + OH \rightarrow CO_3^{2-} + H_2O \quad 6.4$$

An increase in CO_3^{2-} and OH^- can precipitate with Fe, Ca, Mg, and other heavy metals. In this way, some toxic metals are removed from the groundwater. Several metal contaminants can form solid precipitates with CO_3^{2-} and OH^-. More HCO_3^- is in the reaction

zone in the form of divalent iron carbonate. As heavy metal precipitates, the release of H^+ decreases the pH value and Ca as an example as follows in Eq. 6.5.

$$Ca^{2+} + HCO_3^- \rightarrow CaCO_3(S) + H^+ \quad 6.5$$

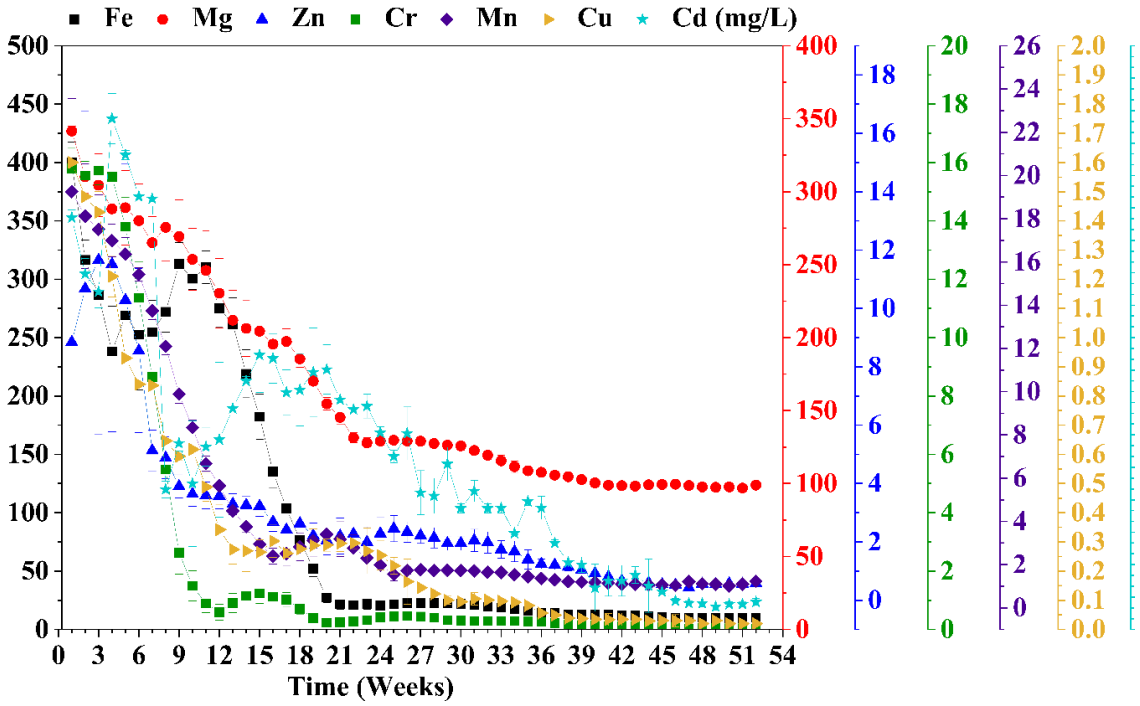


Fig. 6.13. Variation of heavy metals in fresh leachate with time

6.1.3 Statistical analysis

6.1.3.1 Descriptive statistics

Table 6.2 shows the maximum, minimum, mean, and standard deviation of leachate parameters. Because of the high volume of organic waste inside the landfill reactor, the values of leachate parameters showed maximum concentrations during the acidic process, resulting in faster waste degradation. BOD_5 and COD signify the main vital parameters of the pollution load organic of the leachate. The maximum value of BOD_5 obtained in this study was 50,250 mg/L, and COD was 92,160 mg/L for the fresh MSW. The maximum values of ambient and landfill temperature were 49.30 and 33.60°C, respectively. Landfill temperature is more than ambient temperature due to biodegradation of waste-producing heat (exothermic condition) within the landfill reactor. The ratio of BOD/COD showed that leachate is an unstabilized phase because anaerobic condition requires time and favorable

conditions. Therefore, a lower BOD/COD ratio was recorded, and the value is equal to 0.85 for young leachate represents more polluted (Chofqi et al., 2004).

Table 6.2. Descriptive statistics for the ambient temperature, landfill temperature, and leachate parameters

Parameters	Max	Min	Mean	Stdev
Ambient temp	33.60	18.30	24.91	4.33
Landfill temperature	49.40	21.30	27.67	6.77
pH	8.25	5.41	7.13	1.10
EC	39.53	15.12	30.95	9.42
BOD	50,250	13,980	2,2813.63	11,073.80
COD	92,160	19,056	3,7694.64	18,682.29
VFA	37,500	2,464.00	8,595.19	7,377.09
BOD/COD ratio	0.84	0.29	0.65	0.12
TS	57,540.67	3163	14,137.21	14,945.38
TSS	21,882.67	679	4,715.71	5,608.54
TDS	39,370	2,332	9,421.50	9,914.07
TVS	19,298	2,467	6,321.50	4,593.70
TFS	4,5911.67	1,009	7,838.13	10,980.57
Na	505	103.00	211.98	79.92
Ca	3,865.50	188.97	982.30	1,155.95
k	3,800	1,032	1,675.12	773.41
Cl^-	131.85	21.32	44.98	29.99
F^-	493.69	13.43	101.85	130.10
SO_4^{2-}	368.80	13.65	54.08	71.45
PO_4^{3-}	123.87	50.43	81.84	20.85
NH_4^+-N	455.63	90.76	186.68	87.82
NO_3^-	289.40	41.94	138.48	54.14
NO_2^-	210.54	45.97	157.16	51.68
TKN	0.59	0.05	0.20	0.16
Fe	400.15	9.76	97.16	120.47
Mg	341.58	96.89	162.70	72.98
Zn	11.65	0.43	2.91	2.93
Cr	15.80	0.02	2.33	4.73
Mn	19.25	1.02	4.49	5.41
Cu	1.60	0.02	0.31	0.40
Cd	47.25	2.12	16.02	11.74

*All values in mg/L except pH, EC, BOD/COD ratio, ambient, and landfill temperature.

6.1.3.2 Correlation analysis

Correlation coefficients were analyzed between leachate parameters, ambient temperature (AT), and landfill temperature (LT). The correlation coefficients vary from +1

(positive correlation) and -1 (negative correlation). Based on similarity, the parameters are classified as strong, moderate, or weak. The parameter is high (± 0.8 to 1.0), moderate (± 0.5 to 0.8) and weak (± 0.0 to 0.5) (Nair et al., 2005). Table 6.4 shows that the ambient and landfill temperature relationship showed a weak correlation (0.34) and R^2=0.12. According to one study, there is no correlation between ambient and landfill temperature (R^2=0.0038) (Kumar, 2007). All leachate parameters showed a strong correlation with time except ambient temperature. Relation between ambient temperature and other parameters showed a weak correlation. As a result, the effect of ambient temperature on microbial activity was minor. All of the leachate parameters showed a strong correlation with landfill temperature. As a result of the biodegradation phase of solid waste-producing heat, the landfill temperature rises and falls over time, potentially affecting the waste temperature within the landfill. All parameters have a significant negative correlation with pH, and Ec has a medium correlation with the leachate parameters. The leachate parameters are significantly positively associated with BOD_5 and COD because organic and inorganic loading rates are the primary pollutants. When these parameters increase, other organic and inorganic parameters also increase. VFA and solids have a significant positive relationship as waste degraded faster in acidic conditions, and solids were leached from the waste. Cations, anions, and other heavy metals reported a significantly strong positive association with solids. pH is essential in the decomposition of solid waste. The correlation coefficients revealed that the leachate parameters had a substantial relationship. The variables analyzed during the period of natural degradation were characterized by behavior with a high Pearson correlation. This correlation demonstrates that based on the determination of one of the parameters, the others can be estimated and controlled during the leachate treatment process.

6.1.3.3 Obtaining rate constants (k) for leachate parameters

The First-order Levenberg-Marquardt algorithm, an iterative procedure, was used to obtain exponential decay constants (k) employed to solve non-linear problems. This algorithm was applied to BOD_5, COD, TS, and VFA to get the decay constant k value to predict leachate strength. Table 6.4 shows that the model parameters and coefficients of BOD_5, COD, TS, and VFA, ammonium nitrogen, chloride, magnesium, manganese, phosphate, sulfate, nitrate, TKN, fluoride, Fe, Cr, Cu, Zn and Cd were 0.031, 0.031, 0.086, 0.067, 0.025, 0.042, 0.037, 0.101, 0.015, 0.152, 0.091, 0.043, 0.097, 0.076, 0.171, 0.109, 0.078 and 0.042 $week^{-1}$. Fig. 6.14 shows the exponential curve fit and experimental values of the rising and declining part of the plot. Other studies have been done based on

recirculation and without recirculation of leachate. The coefficients k1 for without recirculation are 1.2 to 1.9 yr^{-1} and 2.1 to 5.1 for recirculation. Higher coefficients represent high flow rates and high water content in landfills (Pohland and Maye, 1973). Another study was carried out by the addition of rainfall inside the landfill reactor. The decay coefficients for BOD_5 (0.033), COD (0.036) , and TS (0.04 $week^{-1}$) respectively. Therefore the observation was higher k value represents faster degradation material. In this research, natural attenuation of the waste degradation process was observed without rainfall to the solid waste. For 100% food waste, the estimated k values for kBOD (0.018) and kCOD (0.017 day^{-1}), respectively, while the estimated k values for 60% of food waste were BOD (0.026) and COD (0.028 day^{-1}) (Bhatt et al., 2016). The fitted models were showed goodness of fit of BOD_5 (R^2= 0.77), COD (R^2= 0.78), TS (R^2=0.95) , VFA (R^2=0.78), ammonium nitrogen (R^2=0.83), chloride (R^2=0.77), magnesium (R^2=0.95), manganese (R^2=0.95), phosphate (R^2=0.89), sulfate (R^2=0.93), nitrate (R^2=0.66), TKN (R^2=0.74), fluoride (R^2=0.95), Fe (R^2=0.85), Cr (R^2=0.91), Cu (R^2=0.97), Zn (R^2=0.87) and Cd (R^2=0.77) respectively as shown in Table 6.3. According to the model built in this analysis, the observed data fit significantly. The model is likely to perform well even in landfills with a high proportion of wet waste in tropical countries.

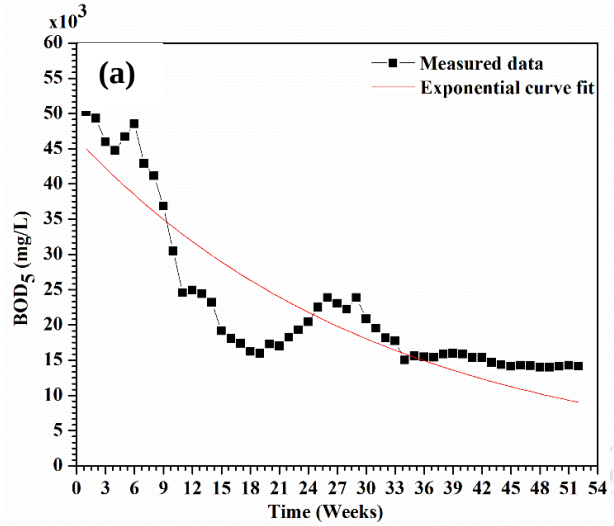
Table 6.3. Model parameters and coefficients for leachate parameters

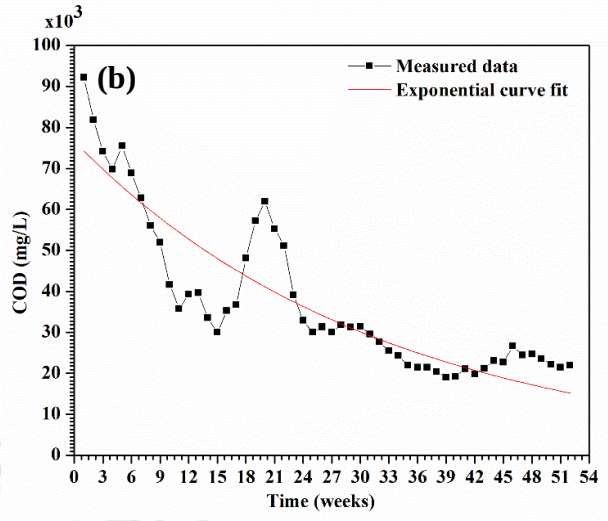
Model parameters and coefficients	a	b (k)	R^2	Model equation
BOD_5	46.45	0.031	0.77	
COD	76.53	0.031	0.78	
TS	64.46	0.086	0.95	
VFA	29.53	0.067	0.78	
Ammonium nitrogen	357.09	0.025	0.83	$y = a * e^{(-b*x)}$
Chloride	41.14	0.042	0.77	
Magnesium	328.94	0.037	0.95	
Manganese	23.61	0.101	0.95	
Phosphate	120.66	0.015	0.89	
Sulfate	375.3	0.152	0.93	

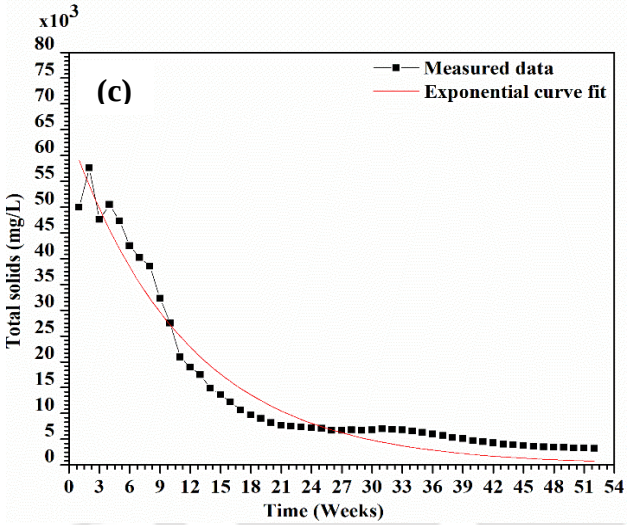
Nitrate	188.13	0.091	0.66
TKN	0.53	0.043	0.74
Fluoride	540.97	0.097	0.95
Fe	430.93	0.076	0.85
Cr	23.31	0.171	0.91
Cu	1.77	0.109	0.97
Zn	11.85	0.078	0.87
Cd	41.14	0.042	0.77

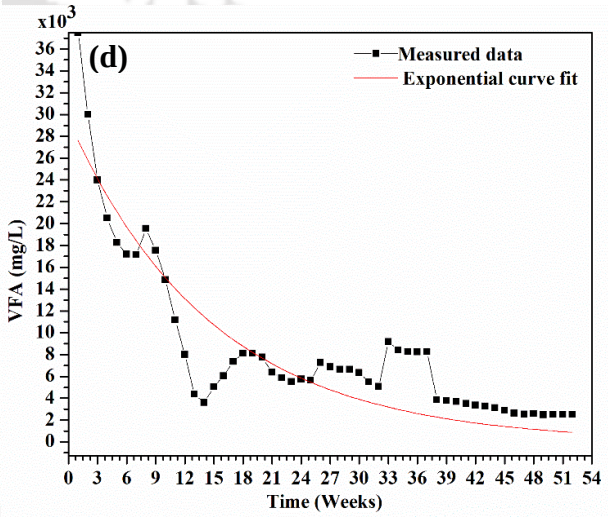
Table 6.4. Correlation analysis between leachate parameters, ambient, and landfill temperature

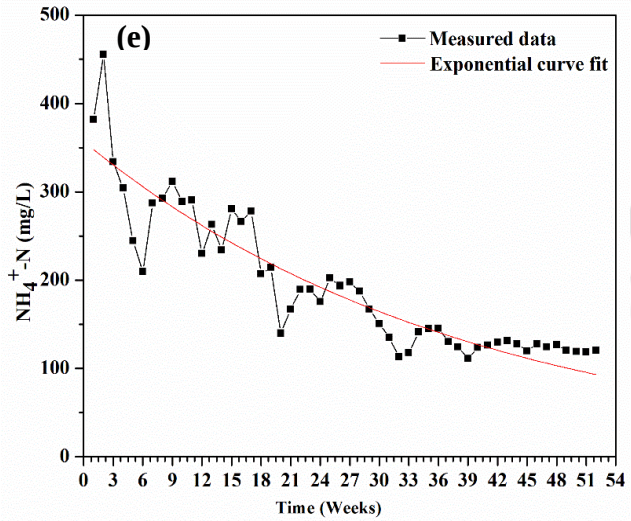
parameters	Ambient temp	Landfill temp	pH	EC	BOD	COD	VFA	TS	TSS	TDS	TVS	TFS	Na	Ca	k	Cl^-	F^-	SO_4^{2-}	PO_4^{3-}	NH_4+-N	NO_3^-	NO_2^-	TKN	Fe	Mg	Zn	Cr	Mn	Cu	Cd
Ambient temp	1																													
Landfill temp	0.32	1																												
pH	-0.04	-0.76	1																											
EC	-0.26	-0.75	0.90	1																										
BOD	0.13	0.82	-0.84	-0.71	1																									
COD	0.02	0.83	-0.84	-0.63	0.85	1																								
VFA	0.10	0.86	-0.75	-0.62	0.91	0.86	1																							
TS	0.24	0.89	-0.87	-0.78	0.97	0.86	0.90	1																						
TSS	0.23	0.74	-0.78	-0.71	0.87	0.73	0.72	0.93	1																					
TDS	0.23	0.92	-0.87	-0.78	0.96	0.88	0.94	0.98	0.84	1																				
TVS	0.18	0.67	-0.85	-0.78	0.91	0.74	0.79	0.90	0.86	0.87	1																			
TFS	0.25	0.93	-0.82	-0.74	0.93	0.86	0.89	0.98	0.91	0.97	0.80	1																		
Na	0.36	0.73	-0.42	-0.42	0.77	0.59	0.83	0.76	0.63	0.79	0.67	0.75	1																	
Ca	0.21	0.72	-0.87	-0.91	0.70	0.60	0.67	0.75	0.61	0.79	0.72	0.72	0.44	1																
k	0.23	0.92	-0.87	-0.79	0.95	0.89	0.92	0.98	0.86	0.98	0.86	0.96	0.78	0.74	1															
Cl^-	0.22	0.83	-0.82	-0.74	0.93	0.83	0.79	0.96	0.95	0.91	0.87	0.94	0.69	0.62	0.94	1														
F^-	0.27	0.93	-0.87	-0.81	0.93	0.84	0.88	0.98	0.90	0.97	0.84	0.99	0.73	0.78	0.97	0.94	1													
SO_4^{2-}	0.20	0.91	-0.72	-0.61	0.89	0.85	0.96	0.89	0.69	0.95	0.72	0.91	0.85	0.67	0.92	0.77	0.89	1												
PO_4^{3-}	-0.17	0.68	-0.90	-0.79	0.68	0.74	0.65	0.69	0.57	0.72	0.63	0.68	0.28	0.76	0.74	0.67	0.73	0.62	1											
NH_4+-N	0.00	0.79	-0.91	-0.82	0.80	0.78	0.81	0.83	0.70	0.85	0.72	0.82	0.48	0.81	0.84	0.74	0.85	0.79	0.88	1										
NO_3^-	-0.15	0.77	-0.89	-0.74	0.85	0.85	0.86	0.84	0.70	0.87	0.77	0.81	0.56	0.75	0.86	0.77	0.83	0.82	0.91	0.93	1									
NO_2^-	-0.49	0.43	-0.77	-0.56	0.59	0.66	0.55	0.55	0.49	0.55	0.55	0.51	0.15	0.53	0.57	0.55	0.54	0.46	0.89	0.77	0.86	1								
TKN	0.18	0.65	-0.92	-0.95	0.76	0.67	0.66	0.80	0.75	0.79	0.89	0.72	0.44	0.89	0.79	0.75	0.78	0.60	0.76	0.80	0.77	0.61	1							
Fe	0.25	0.83	-0.92	-0.95	0.84	0.74	0.79	0.88	0.74	0.91	0.84	0.84	0.59	0.96	0.88	0.78	0.89	0.79	0.80	0.87	0.83	0.57	0.92	1						
Mg	0.17	0.88	-0.96	-0.91	0.91	0.87	0.86	0.94	0.81	0.95	0.87	0.91	0.62	0.87	0.95	0.88	0.94	0.85	0.85	0.92	0.90	0.67	0.90	0.96	1					
Zn	0.14	0.90	-0.82	-0.69	0.93	0.87	0.87	0.95	0.87	0.94	0.78	0.97	0.71	0.64	0.95	0.95	0.96	0.87	0.73	0.79	0.83	0.60	0.66	0.79	0.89	1				
Cr	0.25	0.91	-0.69	-0.58	0.92	0.84	0.89	0.93	0.82	0.93	0.74	0.95	0.84	0.54	0.94	0.91	0.93	0.92	0.57	0.70	0.74	0.42	0.56	0.72	0.82	0.96	1			
Mn	0.24	0.91	-0.84	-0.73	0.97	0.90	0.93	0.98	0.87	0.99	0.88	0.97	0.81	0.72	0.98	0.94	0.97	0.93	0.67	0.79	0.83	0.52	0.75	0.86	0.92	0.95	0.96	1		
Cu	0.16	0.93	-0.84	-0.70	0.93	0.92	0.95	0.95	0.79	0.98	0.78	0.96	0.76	0.71	0.97	0.88	0.95	0.96	0.73	0.85	0.87	0.58	0.69	0.84	0.92	0.95	0.94	0.97	1	
Cd	-0.07	0.76	-0.84	-0.67	0.79	0.87	0.72	0.80	0.73	0.79	0.71	0.79	0.45	0.54	0.84	0.85	0.80	0.70	0.84	0.78	0.85	0.77	0.66	0.69	0.83	0.86	0.78	0.80	0.82	1

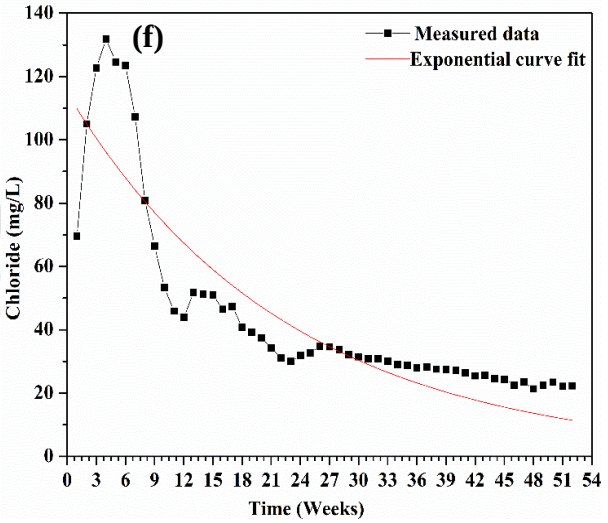

x10^3
(a)
Measured data
Exponential curve fit
BOD5 (mg/L)
Time (Weeks)

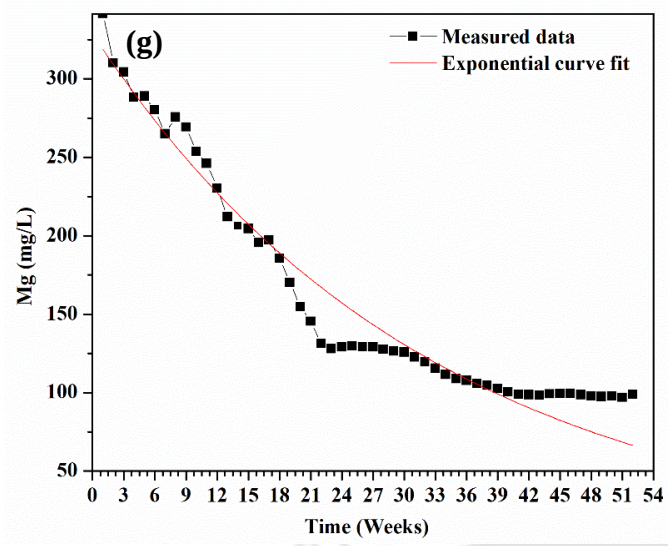

x10^3
(b)
Measured data
Exponential curve fit
COD (mg/L)
Time (weeks)

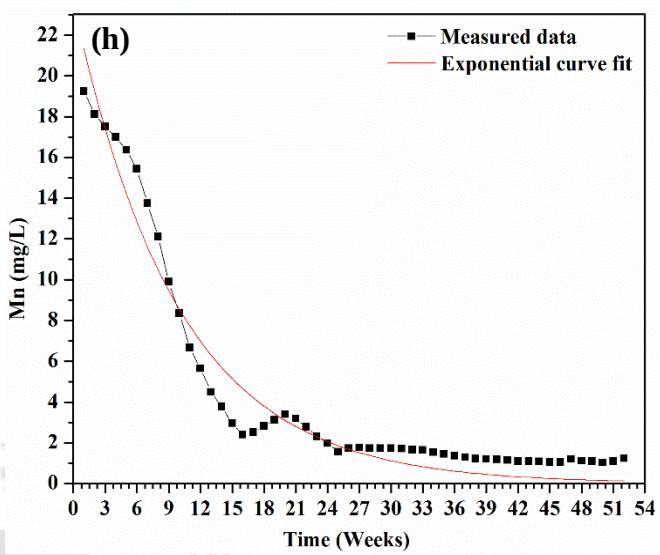

x10^3
(c)
Measured data
Exponential curve fit
Total solids (mg/L)
Time (Weeks)

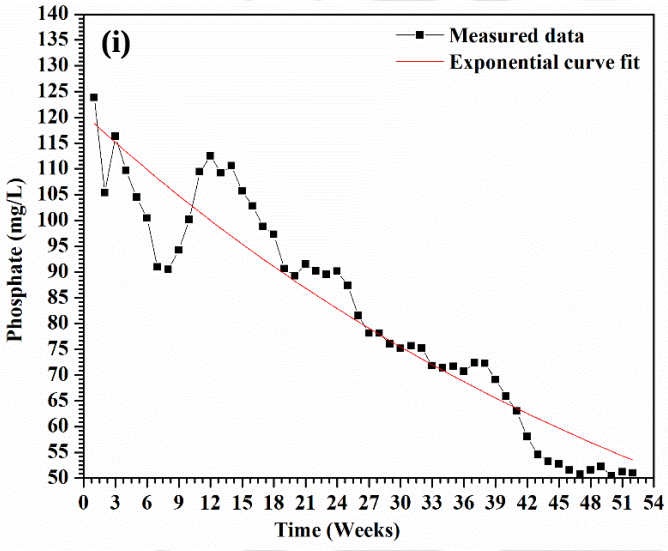

x10^3
(d)
Measured data
Exponential curve fit
VFA (mg/L)
Time (Weeks)

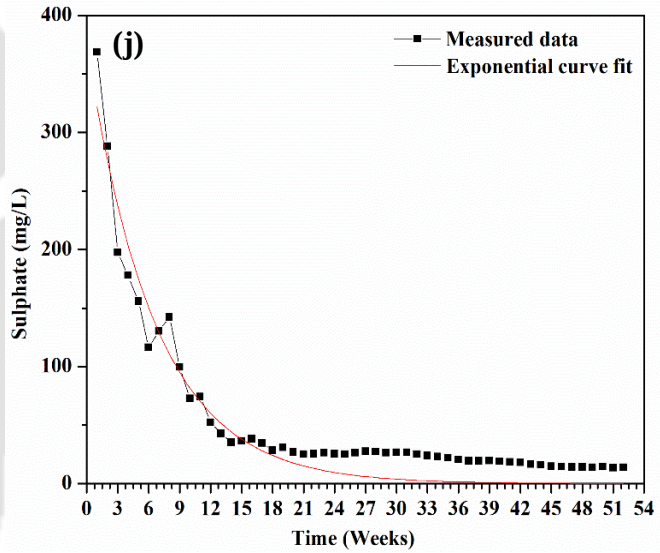

(e)
Measured data
Exponential curve fit
NH4+-N (mg/L)
Time (Weeks)

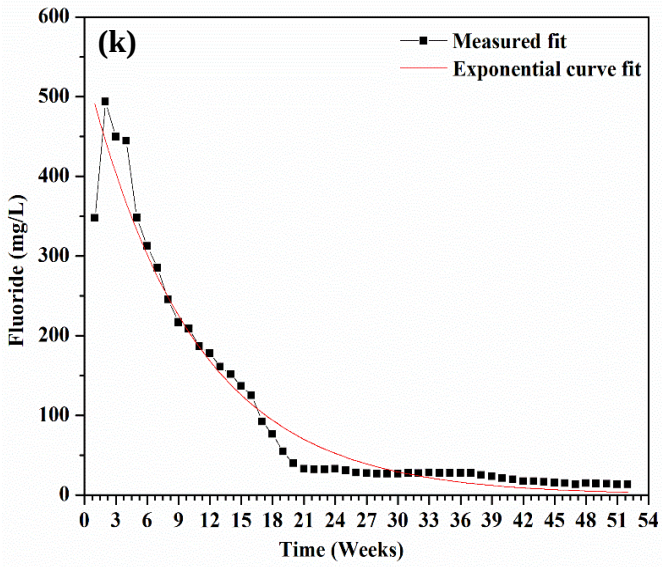

(f)
Measured data
Exponential curve fit
Chloride (mg/L)
Time (Weeks)

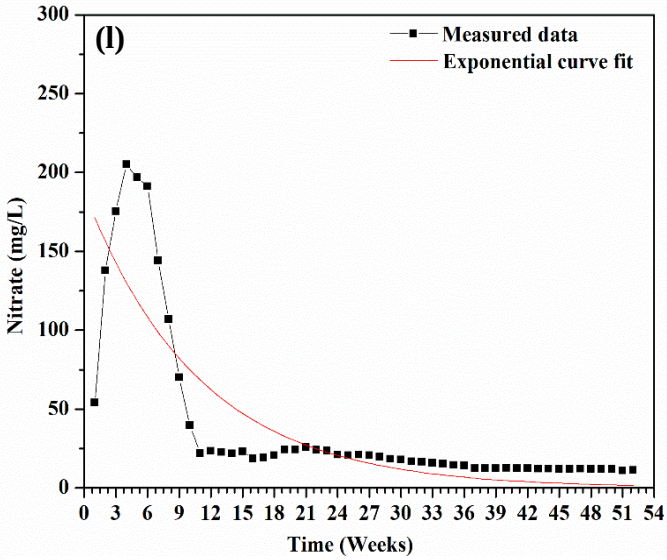

(g)
Measured data
Exponential curve fit
Mg (mg/L)
Time (Weeks)


(h)
Measured data
Exponential curve fit
Mn (mg/L)
Time (Weeks)


(i)
Measured data
Exponential curve fit
Phosphate (mg/L)
Time (Weeks)


(j)
Measured data
Exponential curve fit
Sulphate (mg/L)
Time (Weeks)


(k)
Measured fit
Exponential curve fit
Fluoride (mg/L)
Time (Weeks)


(l)
Measured data
Exponential curve fit
Nitrate (mg/L)
Time (Weeks)

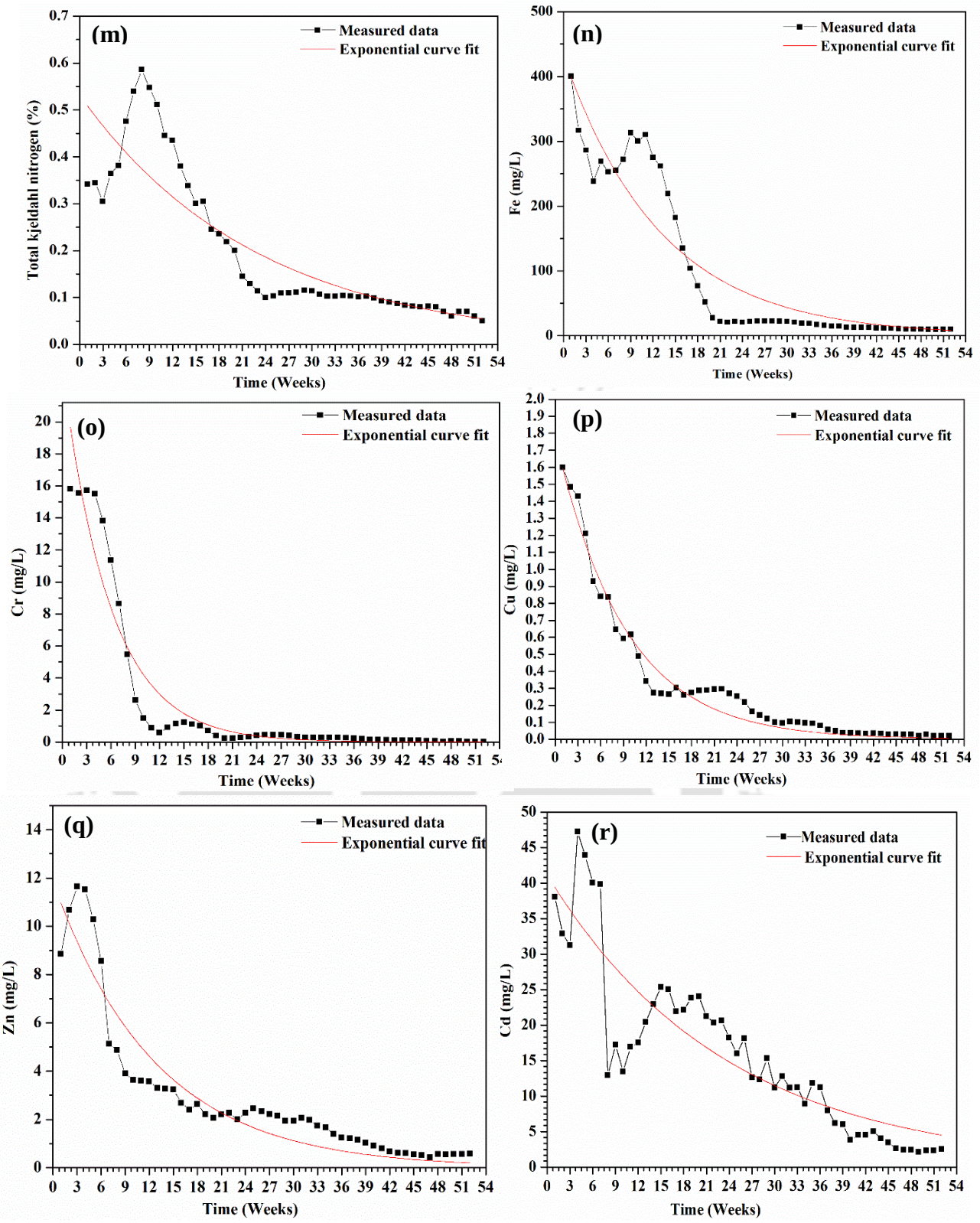


Fig .6.14. Exponential decay models for (a) BOD_5, (b) COD, (c) TS, (d) VFA, (e) ammonium nitrogen, (f) chloride, (g) magnesium, (h) manganese, (i) phosphate, (j) sulfate, (k) fluoride, (l) nitrate, (m) TKN, (n) Fe, (o) Cr, (p) Cu, (q) Zn and (r)

6.1.4 Comparison of landfill gas emissions (LFG) distribution pattern in R1

The gas composition for the reactor operated without rainfall is given in Fig. 6.15. The production of waste degradation was principally CO_2 and was likely due to the high rate of carbon reduction in the waste decomposition process, as presented in Eq. 6.6. The maximum CO_2 was found that 56.73% signifies the breakdown of carbohydrates, proteins, and lipids that composite organic waste produces a high amount of CO_2 in a landfill. The methane production was the less initial phase of degradation due to pH value not being favorable for methanogenesis and a high VFA toxic to methanogens (pH should be in the range of 6-8). Evolved CO_2, due to substantial degradation, exerts high partial pressure on leachate, and it dissolves, forming carbonic acid, which decreases the pH (Khattabi et al., 2002a). H_2 was found to a maximum of 2.12% during the early transition stage, and its concentration was decreased after methane production was slowly increased from 0.23% to 13.68%. The nitrogen was found in the landfill was a maximum of 39.63% and a minimum of 1.34%. Interestingly, empirical data from landfill biogas recovery systems have also periodically documented N_2 recovered from deep landfills without any possibility of atmospheric input, suggesting that ANAMMOX processes may be naturally occurring in deep landfill settings (Meyer-Dombard et al., 2020). Another reason could be the denitrification of nitrate reduction to form nitrogen gas from landfills, as presented in Eq. 6.7. It was hypothesized that active and well-established methanogenic populations could tolerate and function under acidic pH conditions in landfill ecosystems (Kasali et al., 1988). The authors were able to isolate acid-tolerant methanogens that could grow at a pH of 5.5 (Juhasz et al., 1997). Both the hydrolytic and acetogenic bacteria produce hydrogen, which is a substrate for methanogens. The accumulation of hydrogen was found to be an indication of an imbalance in the microbial population (Barlaz et al., 1989). Researchers have established that utilizing H_2 concentrations to monitor landfill status in high-temperature landfills. They found that samples with higher rates of methanogenesis had lower hydrogen concentrations and more neutral pH values than those with lower rates of methanogenesis because methanogens or other organisms rapidly consumed H_2, and an apparent steady-state was achieved (Mormile et al., 1996). Other important intermediates are organic acids. It has been found that high methane-producing samples have low organic acid concentrations, while low methane-producing samples accumulate high concentrations of organic acids during the initial fermentation reactions.

$$C_6H_{12}O_6 + O_2 \rightarrow CO_2 + H_2O + \text{Biomass} + \text{Heat} \quad (6.6)$$

$$NH_4^+ + NO_2^- \rightarrow N_2 + 2H_2O \quad (6.7)$$

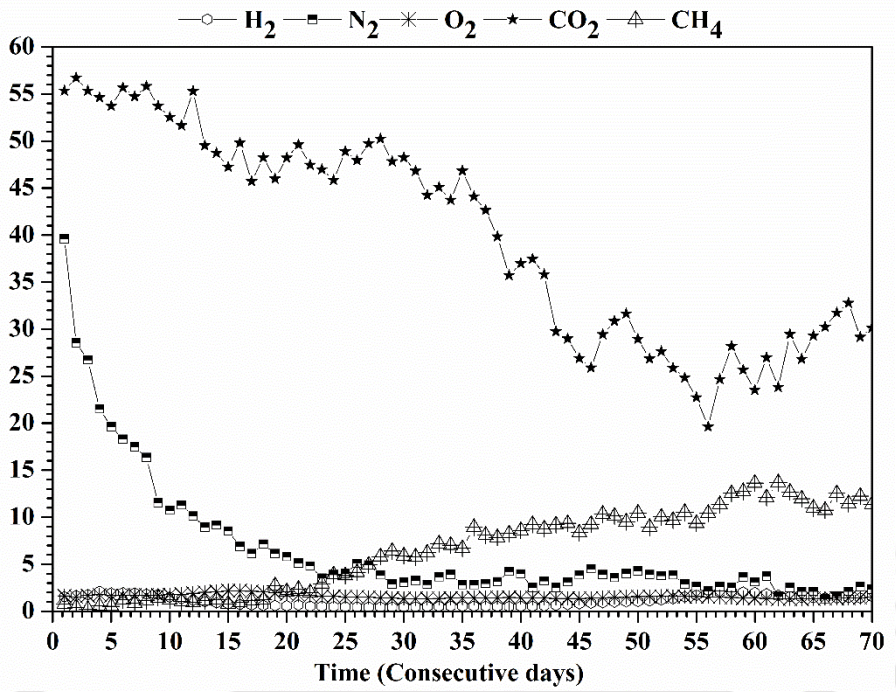


Fig. 6.15. Distribution pattern of LFG emissions in R1 (operated without rainfall)

6.1.4.1 Statistical analysis

- ***Descriptive analysis of gas components, ambient and Landfill temperature***

The reactor has a higher inside (waste) temperature than ambient temperature because exothermic reactions increase the landfill temperature during the degradation process. Table 6.5 shows the descriptive analysis of gas compositions and external factors. The high amount of CO_2 produced during the waste degradation process and methane concentration was less due to acidic pH in the reactor. It was confirmed that the decrease in CO_2 and methane concentration ow methane concentrations were considered to be another proof of washout, indicating the inability of the system to develop an active methanogenic population and enhance waste stabilization.

Table 6.5. Descriptive analysis of gas components, ambient, and landfill temperature (^oC)

Parameters	Max	Min	Mean	Stdev
Ambient temp	49.40	21.60	28.56	6.51
Landfill temp	30.40	17.30	21.80	3.18
H_2	2.12	0.44	1.10	0.56
N_2	39.63	1.34	6.55	7.06
O_2	2.18	1.31	1.56	0.22
CO_2	56.73	19.63	39.80	11.21
CH_4	13.68	0.23	6.60	4.35

- ***Correlation coefficients matrix between gas components, ambient, landfill temperature (^oC) in R1***

Correlation coefficients of gas components, ambient, waste temperature (^oC) are represented in Table 6.6. The effect of ambient waste temperature on gas components and among the parameters were examined. The positive correlation (r=0.74) between waste and ambient temperature, while other parameters like CH_4 (-0.83) and CO_2 (0.83) have a strong correlation with waste temperature and H_2 (r=0.15), N2 (r =0.87). Ambient temperature has a strong correlation with N_2 (r=0.76), and hydrogen evolves through biodegradation. CO_2 strongly correlates with waste temperature (0.83) because the temperature increases through the degradation process and evolves CO_2. There is a moderate correlation between ambient temperature and CO_2 (r=0.48). Hydrogen has a weak correlation with CO_2 (r=-0.21) and N_2 (r=0.15), respectively. Nitrogen has moderate correlation with CO_2 (r=0.61) and CH_4 (r=-0.65). Oxygen had a strong negative correlation with CH_4 (r=-0.73).

Table 6.6. Correlation coefficients matrix between gas components, ambient, and landfill temperature

Parameters	Landfill temp	Ambient temp	H_2	N_2	O_2	CO_2	CH_4
Landfill temp	1						
Ambient temp	0.76	1					
H_2	0.15	0.41	1				
N_2	0.87	0.76	0.37	1			
O_2	0.41	0.37	0.02	0.37	1		
CO_2	0.83	0.57	-0.21	0.61	0.51	1	
CH_4	-0.80	-0.56	0.15	-0.65	-0.73	-0.91	1

6.2 PART-II: RESULTS AND DISCUSSIONS OF SIMULATED REACTOR OPERATED WITH RAINFALL (R3)

6.2.1 Initial characterization of mixed unshredded MSW

The MSW was characterized in pH and moisture content as indicators of microbial activities (Chian and Dewalle, 1976). In this study, the waste components were mixed according to wet (73%) and dry (27%) weights. Initially, 100 kgs of waste were mixed. The total of wet and dry waste fed into the reactor was 292 and 108 kgs, respectively. The wet waste consists of mixed food waste, and dry waste consists of plastic, paper, textiles, yard, and wood waste. The physical and chemical composition of mixed solid waste was examined to understand the behavior of waste as moisture content is one of the important parameters which varies with solid waste, and environmental factors for leachate production and volatile solids are governing factors for gas emissions. The findings from other literature specified that moisture content values for developed countries varied from 15-40% for urban waste because it contains more dry waste(Tchobanoglous et al., 1993). Moisture content in the Istanbul landfill was 59.49% because 54.09% of mixed solid waste was composed of organic material. (Yildiz et al., 2013). The average moisture content in the Urali Devachi landfill was 48.08%, and it contained organic waste 32. 83% (Mali et al., 2011). However, the average moisture content in this study was 79.6%. The results indicate that the proportion of organic content affects the moisture content. Initial decomposable volatile solids of waste were 55% (Lakshmikanthan and Sivakumar Babu, 2017), and the obtained average value in this study was 56.22%. This study's average pH of mixed waste was 5.12 because of the high amount of food waste in landfills. The pH of food waste ranged from 4.7-6.1 (Sundberg et al., 2011). Table 6.7 indicates the average and standard deviation of pH, moisture content, total organic carbon (TOC), and volatile solids. The significance of this MSW characterization is that apart from establishing the basis of all the further degradation processes and leachate characterization, this MSW characteristic is also representative of India's highest wet: dry ratio. The reactor is named as reactor 3 (R3).

Table 6.7. Initial characterization of fresh mixed solid waste

Parameters	Mean ± Stdev
Moisture content (%)	81.6 ± 1.41
pH	5.44 ± 0.26
EC (mS/cm)	5.92 ± 1.22
Density (kg/m^3)	743.24 ± 196.69
VS (%)	59.6.22 ± 1.15
TOC (%)	33.11 ± 0.64
*Wet (kg)	292
*Dry (kg)	108

6.2.2 Influence of tropical climate (rainy season) on the degradation of varying solid waste composition

Analysis of leachate quality in terms of BOD_5, COD, pH, EC, solids, ammoniacal nitrogen (NH_4^+-N), nitrate (NO_3^-), nitrite (NO_2^-), TKN, sodium (Na), potassium (K), and calcium (Ca), and Heavy metals by time in relation to variations of humid climate, weekly monitoring of leachate quality was performed in a research lab. However, simulated landfill examination consists of the rainfall season, age of MSW, and the operational mode of landfill reactor also played a significant role, which leads to a variation of leachate quality. Therefore, a detailed way of variation of leachate quality from reactor 3 (L3) with time was discussed.

6.2.2.1 Landfill leachate quantity

The current total quantity of leachate was 911.89 liters. The initial leachate quantity was 22.5 liters. It indicates more biodegradable waste present in landfills leads to a high leachate generation due to the degradation of solid waste and available moisture within the waste. Moisture addition to the waste degradation generates a high amount of leachate. It proves that moisture plays a significant role in degradation because bacteria require favorable conditions like temperature, moisture availability for biological activity. The leaching behavior of major contaminant elements is strongly influenced by pH, solubility, and sorption properties, forming major interconnected mechanisms that control the release of pollutants in the leachate under different pH conditions inside the waste mass. Therefore, moisture plays a significant role in leachate generation from waste landfills. The trend of leachate quantity is shown in Fig. 6.16.

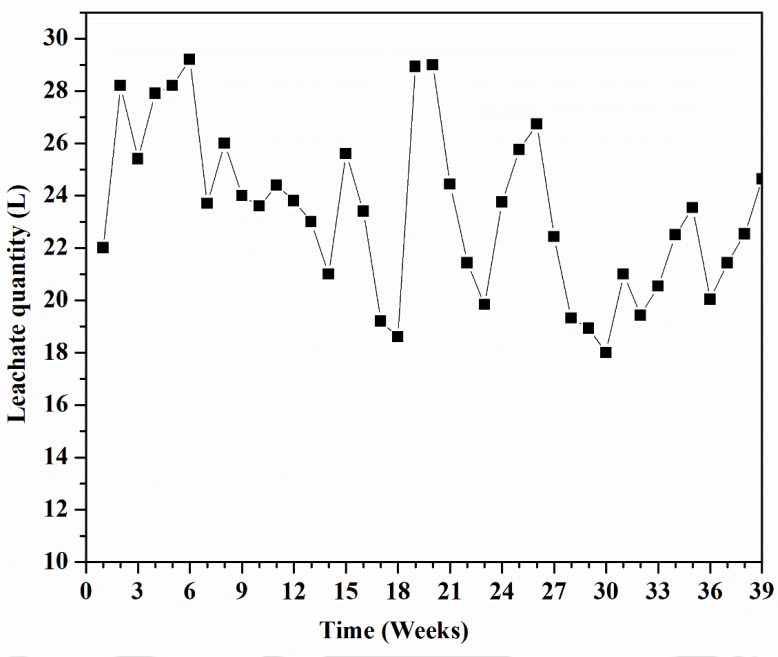


Fig. 6.16. Plotting of landfill leachate quantity against with time

6.2.2.2 Relation between pH and electrical conductivity (EC)

Fig. 6.17 shows the trend of pH and EC as a function of time for the simulated reactor. In the initial weeks, pH was in the acidic phase due to volatile organic acids' production, resulting in the accumulation of carboxylic acids and pH drops. The pH of young leachate ranged from 4.91-5.35. pH values increased very gradually to 5.35. Such an increase might be attributed to the flushing out of VFA in the system. The increase in pH of leachate with time decreased in VFA concentrations (Chian and Dewalle, 1976). The fresh leachates presented relatively low pH values (around 6), rather low ratio of BOD: COD (about 1:3). The other researchers also stated a similar decrease in pH during continuous rainfall, ranging from 4.4-6.2, with an average of 5.3 at different rainfall events (Chiemchaisri and Srisukphun, 2003; Rafizul and Alamgir, 2012). The initial concentration of calcium was as high as 2,193 mg/L. It indicates the presence of $CaCO_3$ at adequate levels to buffer pH changes in solid waste (Statom et al., 2004). Bhatt et al. (2017) and Swati et al. (2011) observed that pH reached the alkaline range within 6 months and 120 days, respectively, in their studies. While we observed a very gradual increase in pH, which did not reach the alkaline range, might be attributed to the carbonic acid accumulation (due to CO_2) that actually would be a better representative of landfill cells in open dumpsite conditions, which are mostly in confined conditions except those in top layers or just below the final cover. Evolved CO_2, due to substantial degradation, exerts high partial pressure on leachate,

and it dissolves, forming carbonic acid, which decreases the pH (Khattabi et al., 2002a). The pH values below 7 are usually softer waters, and the acidity is due to carbonic, humic, fulvic, and other organic acids (Mahapatra et al., 2011a; Mahapatra et al., 2011b).

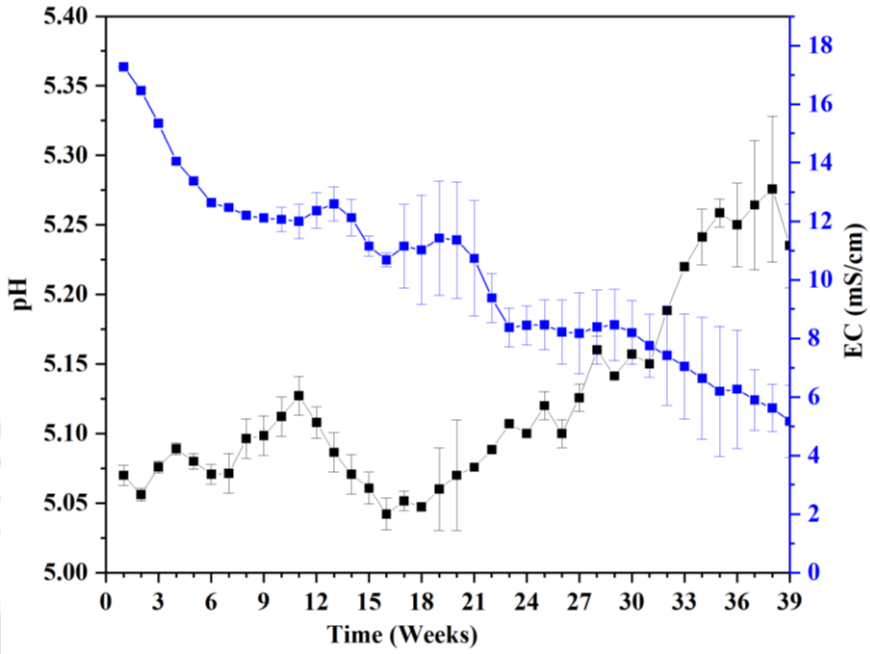


Fig. 6.17. Weekly variation of pH and EC with time

EC is influenced by the amount of dissolved organic and inorganic matter present in the leachate to indicate the degree of salinity and mineral content. EC values ranged from 5.05-16.28 mS/cm due to the production of carbonates and bicarbonates due to high biological activity (Wang and Pelkonen, 2009). The values were found to be similar to the reported values during the acidogenic phase (1.6-17mS/cm) (Reinhart and Grosh, 1998). These could be due to the presence of a high amount of inorganic salts like sodium and chloride in young leachate in the initial weeks of biodegradation of waste (Swati et al., 2011). High initial EC values dropped rapidly due to the washout of inorganics from solid waste in landfills by rainwater infiltration (Qasim and Chiang, 1994; Yuen et al., 2001). The EC values were reduced significantly due to the dilution, and a similar trend was observed (Rafizul and Alamgir, 2012).

6.2.2.3 Relation between solids

The variation of solids is presented in Fig. 6.18. Total solids, TDS, Total Suspended Solids (TSS), Total Fixed Solids (TFS), and total volatile solids (TVS) ranged from 9,077-33,200, 5067-20899, 1,285-9,246, 890-15,720, and 8,247-21,870 mg /L respectively. TDS

and EC have no relation observed in this study, where high concentrations of TDS in initial weeks and low EC were found due to dilution of water and leaching out of inorganics. High concentrations of total dissolved solids may reduce water clarity, which contributes to light limitation, decreasing photosynthesis, and increasing water temperature. It affects the growth and development of biotic components, such as photosynthetic bacteria and algae (Naveen et al., 2017). Volatile solids indicate the changes in the organic strength of leachate, while fixed solids indicate the inorganic strength of leachate. The average ratio of VS to FS in fresh waste was 1.4. The average ratios of VS to FS for fresh and mined waste leachate samples were 1.5 and 0.4, respectively, and decreased with landfill age (Karthikeyan et al., 2008a). A high amount of VFA is because of the degradation of organic compounds by microorganisms in the acidic phase. Degradation observed in solids was 72.6, 75.75, 86.10, 94.33, and 62% for TS, TDS, TSS, TFS, and TVS, respectively, by the end of 39 weeks. The 94% reduction of TFS indicates substantial removal of inorganic solids through leachate and is indicative of rapid biodegradation, which separates inorganic constituents from waste. So even in the field, it is expected that TFS would be the fastest to be leached out.

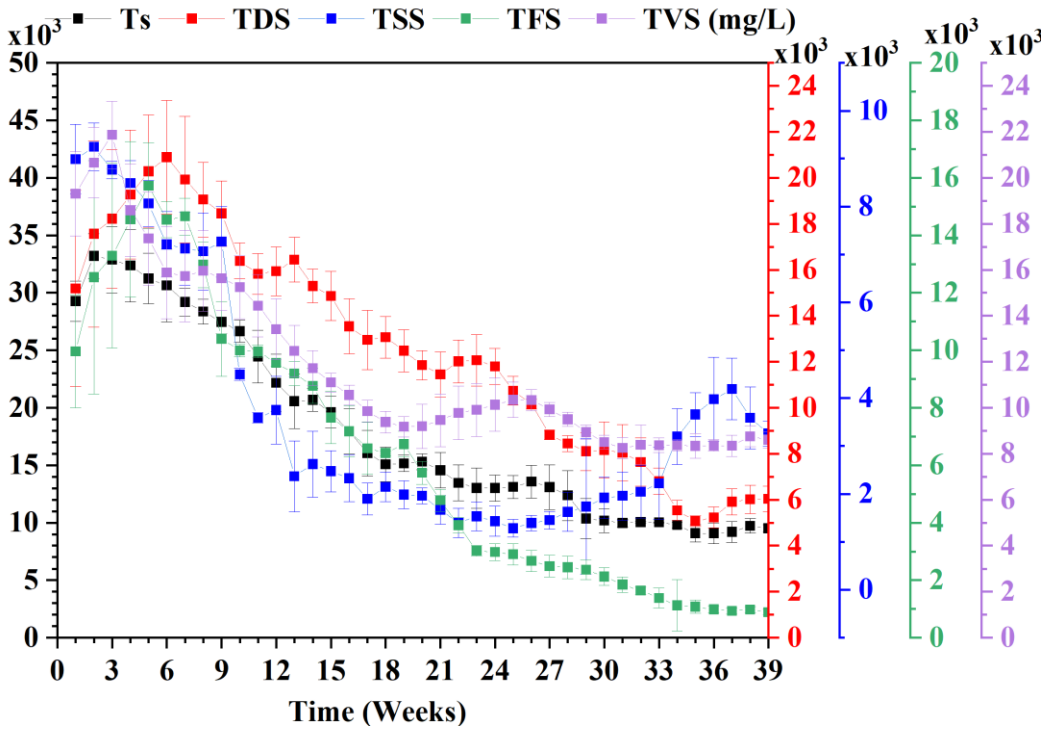


Fig. 6.18 Weekly variation of solids with time

The leachate is highly toxic in the acidic phase due to the high amount of food waste in landfills, and there is a more significant dissolution of metals at low pH (Karthikeyan et al., 2008a). The total solid concentrations were removed because of the washout by the percolating of rainwater. However, solid concertations were recorded highest at initial stages and reduced with time (Chian and Dewalle, 1976). The MSW landfills having high total solids represent the presence of loose materials that are dissolved in leachate while passing through solid waste (Qasim and Chiang, 1994; Kouzeli-Katsiri et al., 1999). A similar trend of TS in leachate was high at the initial stage, but it decreased with time by washout effect and decomposition of waste (Rafizul and Alamgir, 2012).

6.2.2.4 Volatile fatty acids (VFA)

Variation of Volatile fatty acids is presented in Fig. 6.19. VFA values ranged from 5024.39-13477.13 mg /L. Initial weeks of VFA concentrations were low, which might be due to the water addition and the lag time for biological activity. Weeks 6-14 were with significant degradation; hence, more production of VFA was observed, which was accompanied by greater inside temperatures.

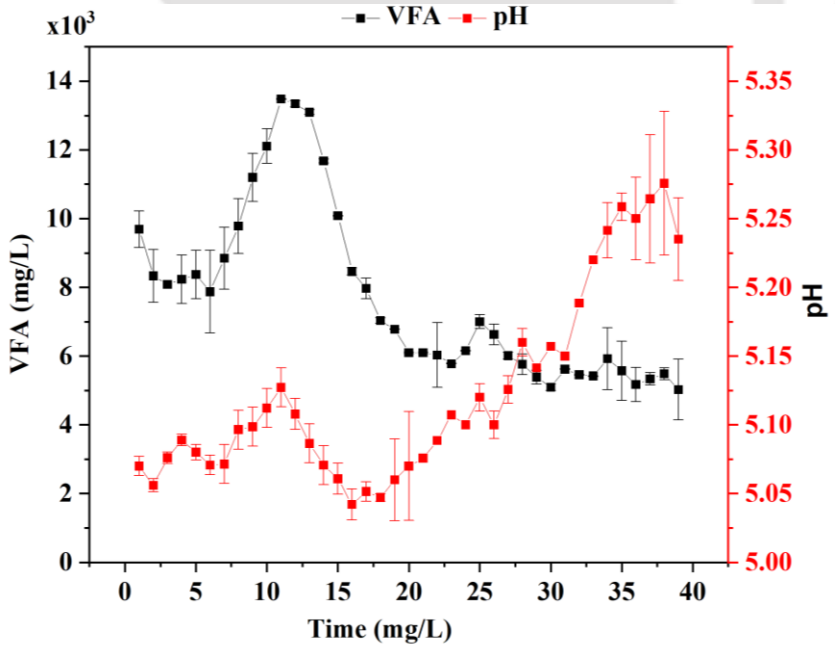


Fig. 6.19. Weekly variation of VFA with time

The acidogenic phase led to a build-up of VFAs, and the decomposition of waste is limited. It was seen that the addition of water could positively affect the degradation of waste because of the resulting pH increase and decrease in VFA with time, which is

significant for providing favorable conditions for biodegradation. Khattabi et al. (2002b) observed that the VFA of mined waste (old waste) was in the range of 1-2 g /L initially and reached levels up to 4 g/L after the incidences of rain. Multiple peaks in the reactor could be due to the degradation of wastes at different times. Literature reported that VFA greater than 10000 mg /L is toxic to methanogenic activity and that pH should be in the range of 6-8 (Petchsri et al., 2006). High VFA content is expected as acid production is enhanced if the moisture of the solid waste is high (Wang et al., 2009).

6.2.2.5 Relation between biochemical oxygen demand (BOD_5) and landfill temperature

Fresh MSW generates a high quantity of leachates, which have characteristically high organic loads (El-Fadel et al., 2002; Tatsi and Zouboulis, 2002; Warith, 2002; Swati et al., 2007). Fig. 6.20 shows the degradation pattern of BOD_5 with time. BOD_5 values ranged from 7041–39130 mg /L as a high amount of organic matter present in the landfill indicates a high concentration of BOD_5 in the initial weeks of waste degradation, which diminished over time. In the acidogenic phase (pH at 5), BOD_5 values were higher. As pH increases, BOD_5 values decrease due to the breakdown of organic matter and leached out the high concentrations and the effect of adding water resulting in dilution of liquid. BOD_5 concentrations with time represent a hydro-type curve, high point, a peak concentration, decline part (Kouzeli-Katsiri et al., 1999). The initial phase of the landfill is acidic; anaerobic bacteria decompose organic matter and convert them into volatile fatty acids. As leachate passes through waste during this stage, BOD_5 concentrations increase drastically, and pH falls below 6 (Kjeldsen et al., 2002). At this level, the rate of generation of dissolved organic compounds is very high. The declining curve begins while the dissolved organic matter is consumed by bacteria and removed from the landfill. The initial temperature inside the landfill was 45.67^{o}C temperature. At the same time, BOD_5 concentrations increased to 39,130 mg /L, and by the end of the 39 weeks, the temperature decreased to 34.50^{o}C, while BOD_5 concentrations decreased to 7,041 mg /L. The process of degradation of waste releases heat, and temperature increases significantly. The field study shows that the initial temperature of landfill increases with depth (top, middle & bottom), and temperatures were 42, 51, and 53^{o}C from top to bottom, respectively (Kjeldsen et al., 2002). In this study, measured temperature ranges from 32.5-45.67^{o}C, indicating the landfill has been mesophilic. At the end of the study, the span observed the stabilized temperature. The reason could be attributed to better degradation due to moisture addition and the cooling effect of water addition. The percentage reduction of BOD_5 concentration

after 39 weeks was 82.01, and its reduction of strength was indicative of biodegradation and organic strength removal through leachate.

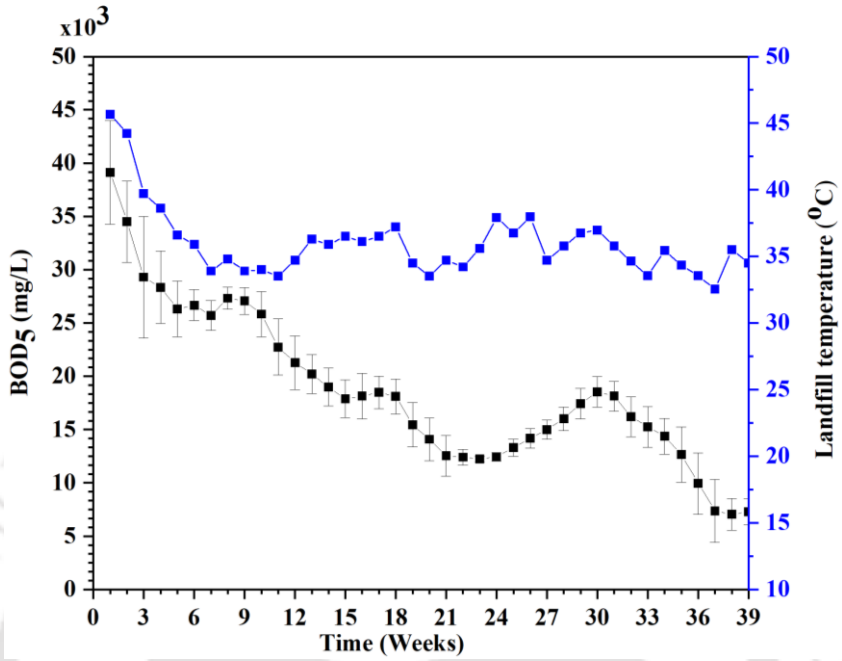


Fig. 6.20. Variable BOD_5 concentrations and landfill temperature with time

6.2.2.6 Relation between chemical oxygen demand (COD) and landfill temperature

Fig. 6.21 shows the degradation pattern of COD with time. COD ranged from 15,692 - 71,630 mg /L as a high amount of organic matter was present in the landfill, and waste was hydrolyzed to organic and inorganic constituents. In contrast, organics get converted to simpler compounds or gases, and inert materials settle out in the leachate and bottom of the reactor. It has been observed that the COD value varies inversely with the pH. COD values were higher in the acidogenic phase, and as pH increased, COD values decreased due to the breakdown of organic matter, which leached out the high concentrations. As VFA contributes to COD, the increased VFA concentrations seen in weeks 6-9 parallel the corresponding COD values spike. This result indicates the possibility of higher COD values when VFA is high. The main reason for this phenomenon - Soluble Microbial Product (SMP) contributes to COD in the effluent (Shoeybi and Salvacion, 2012). SMP is a product of bacterial metabolism. Various researchers established that the generation of SMP happens in the anaerobic degradation of organic waste, such as proteins and polysaccharides (Knox and Goddard, 2013). Temperature also plays a big role in the biodegradation of waste; as temperature increases degradation process increases. The percentage reduction of COD concentration after 39 weeks was 78%, and it signifies a

reduction of an organic pollutant, not inorganic pollutants, which are leaching out from the landfill. The COD values were decreased initially due to degradable nature and washout due to rainfall from the landfill (Chian and Dewalle, 1976; Monteiro et al., 2002; Trankler et al., 2005). The organic compounds decrease promptly than inorganic compounds with the increasing age of the landfill, and it can be attributed to a combination of a decrease in organic contaminants and an increase in the degradation of organic compounds (Chian and Dewalle, 1976; Krug and Ham, 1995; Rafizul and Alamgir, 2012).

6.2.2.7 BOD/COD ratio

The BOD/COD ratio represents the proportion of biodegradable organics in leachate. BOD/COD ratio is presented in Fig. 6.22. The average BOD/COD ratio of the landfill was 0.53. The results showed that landfill is in the acidogenic phase. This ratio is normally higher in young landfills (fresh waste) than in older or stabilized ones. Unstabilized Young leachate has a BOD/COD ratio of 0.5 or more (Swati et al., 2011), as observed in this study also. BOD_5 and COD were found to decline in concentrations due to organic matter removed by the washout effect. Partially stabilized leachates have a ratio ranging from 0.1–0.5 (Adhikari and Khanal, 2015).

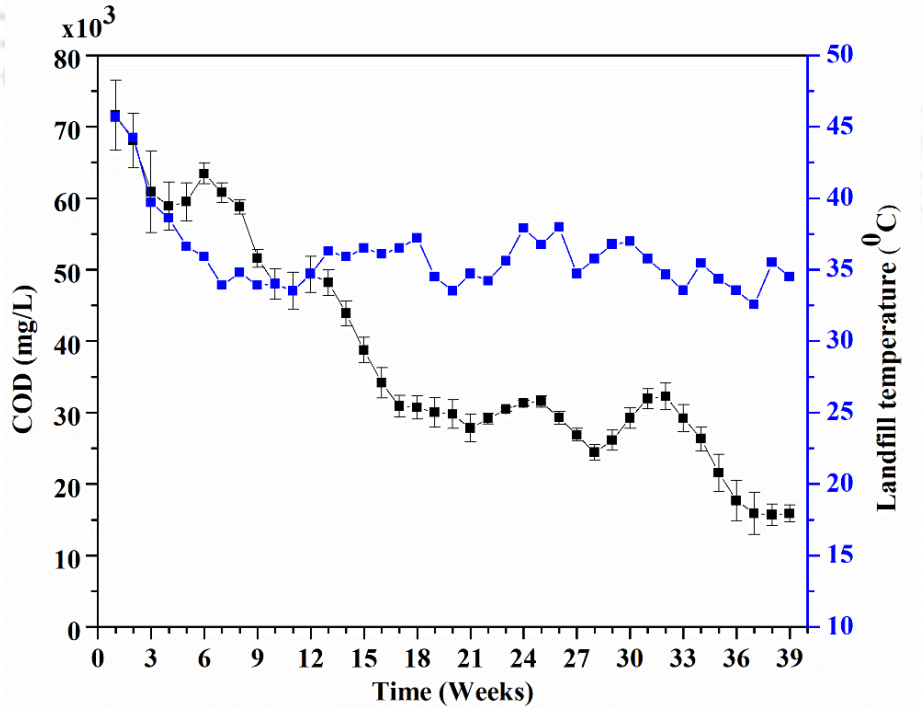


Fig. 6.21. Variable COD concentrations and landfill temperature with time

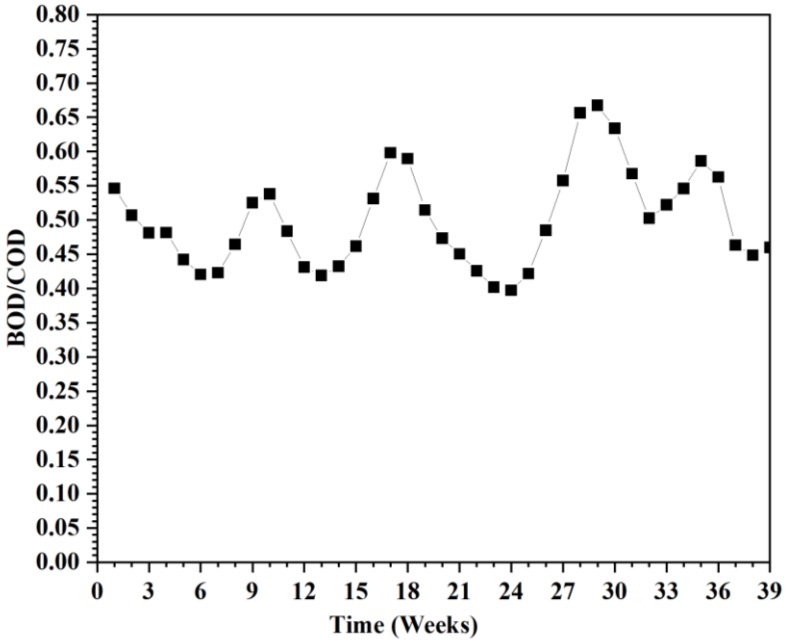


Fig. 6.22. Variable BOD/COD ratio in fresh leachate with time

6.2.2.8 Relation of nitrogen compounds ((ammonical nitrogen (NH_4^+– N), nitrate (NO_3), total Kjeldahl nitrogen (TKN), and nitrite (NO_2^-)) in fresh leachate

Fig. 6.23 shows that variable nitrogen compounds (NH_4^+– N, NO_3^-, NO_2^-, TKN) in fresh leachate with time ranged from 749.16-254.66, 319-65.85, 220.30-116.36 mg/L, and 0.76 -0.11%, respectively. During weeks 21 to 27, ammoniacal nitrogen levels were high as the temperature raised from 34 to 37°C. High temperature represented a high degradation of organics and NH_4^+-N. High concentrations of ammoniacal nitrogen inhibit nitrifying activity (Alkalay et al., 1998). As the amount of rainfall increases, ammoniacal nitrogen increases. Concentrations of ammoniacal nitrogen increase as ammonia gas are trapped in the rain and mixes with leachate, therefore leaching out from the landfill (Salleh and Hamid, 2013). Korean landfills recorded that ammoniacal nitrogen concentrations were low for closed landfills at 860 mg /L, and an operational landfill recorded 1,075 mg/L (Robinson, 2007). High NH_4^+-N concentrations were recorded for closed landfill cells. It is known that acetate utilizing methanogens is more sensitive than hydrogen utilizers, predominantly at high ammoniacal nitrogen levels (Hansen et al., 1998; Calli et al., 2005), and high concentrations were possible in anaerobic conditions (Van Velsen, 1979). The release of soluble nitrogen from waste into leachate continues as compared to soluble organics. Therefore, NH_4^+-N decreases with an increase in the age of the landfill, which is caused by hydrolysis and nitrogen fraction fermentation substances of biodegradable waste

(Abbas et al., 2009). Ammonia is produced from waste mainly by decomposition of proteins, and concentration can decrease only through the leaching process, there is no pathway to degrade the ammonia in the methanogenic phase (Robinson and Grantham, 1988). Therefore, only leaching can decrease the NH_4^+-N (Kjeldsen et al., 2002). The TKN concentration levels were also decreased while pH reaches close to neutral. The release of soluble nitrogen from waste into leachate lasts longer than the release of dissolved organics. As a result, ammonical nitrogen and TKN decrease with an increased concentration due to hydrolysis and fermentation of nitrogen compounds in the biodegradable form in waste (Abbas et al., 2009). Leaching is the only way to decrease nitrogen in landfills because there is no degradation pathway (Tchobanoglous et al., 1993). NO_3^- consumption rate was reduced in well decomposed and speculated the reduced NO_3^- depletion rate was caused by reduction in leachate VFA (Price et al., 2003). The 60% of NH_4^+– N, NO_2^- (47.18%), TKN (85.50%), and NO_3^- (79.39%) respectively was removed within 39 weeks of the degradation process.

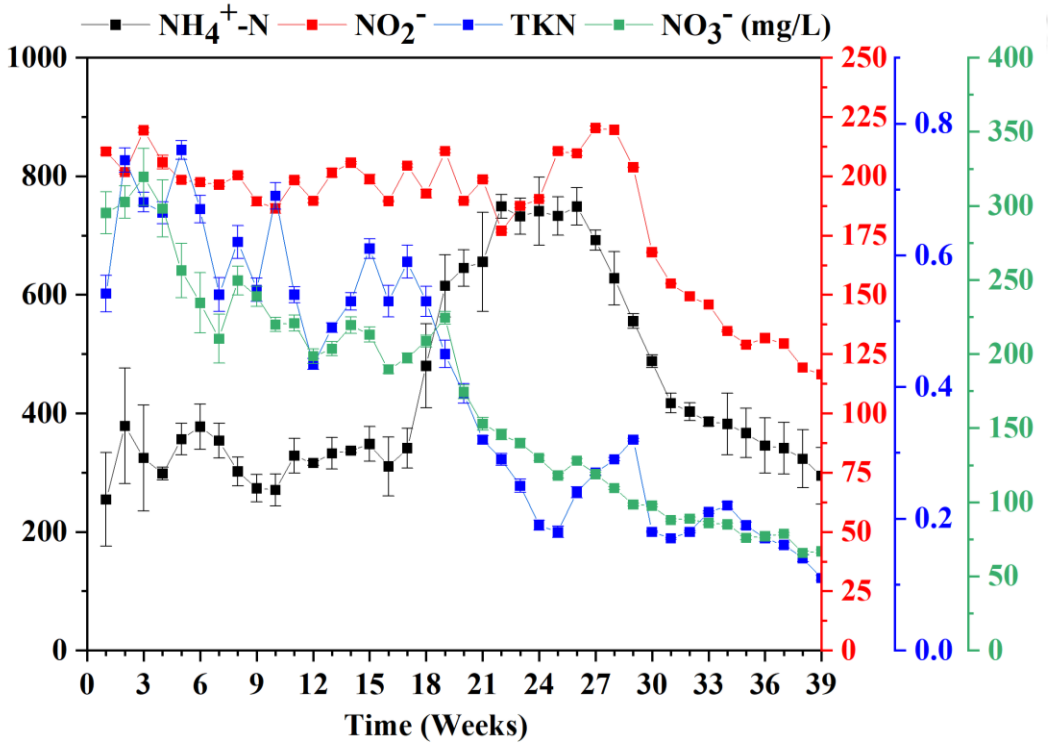


Fig. 6.23. Variable nitrogen compounds in fresh leachate with time

6.2.2.9 Relation of sodium (Na), potassium (K), and calcium (Ca) in fresh leachate

Initially, high concentrations of sodium, calcium, and potassium concentrations of 1,662, 2,193, and 4,442 mg /L, respectively, were observed in the leachate, as shown in Fig. 6.24. However, in the acidic phase, during the degradation of waste, microorganisms

consume sodium, potassium, and calcium as they form their nutrition, released during decomposition. Therefore, concentrations decreased with time were observed in this study. The waste stabilization rates were observed for sodium (66.07%), potassium (80.70%), and calcium (46.05%) within 39 weeks of the degradation process. Cation concentrations are lower during the methanogenic process due to higher pH, which promotes sorption and precipitation while also lowering the amount of dissolved organic matter, potentially complex the cations (Kjeldsen et al., 2002). A decreasing trend with time was observed in this study due to the washout effect by leaching. (Ehrig, 1989). Potassium is ionized quickly, degradable organic matter, and dissolved during decomposition, while Na and K are leached from fresh waste. The potassium ion has the simplest forming potential and is correlated with the other ions leached from the waste. Calcium is another essential ion in establishing the buffering ability of leachates during waste degradation. It buffers throughout the waste degradation is well documented (Karthikeyan et al., 2008b).

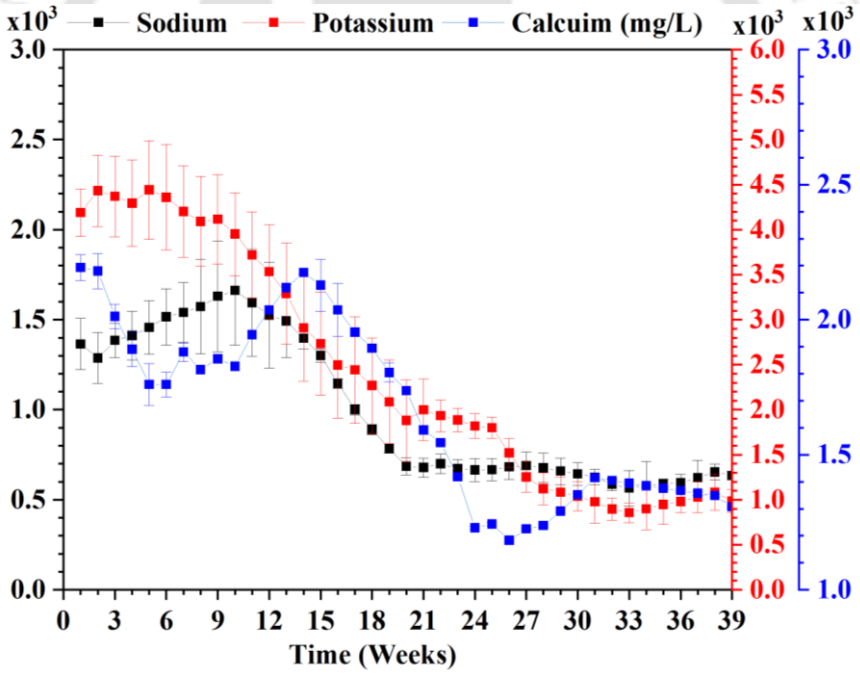


Fig. 6.24. Variable macronutrients in fresh leachate with time

6.2.2.10 Relation of inorganic nonmetallic constituents (Cl^-, PO_4^{3-}, SO_4^{2-} & F^-) in fresh leachate

Fig. 6.25 shows Inorganic Nonmetallic Constituents (chloride (Cl^-), phosphate (PO_4^{3-}), sulfate (SO_4^{2-}) & fluoride (F^-)) in fresh leachate with a function of time. The initial concentrations were observed to be high due to more waste degradation chanced during the acidic phase. The Cl^-, PO_4^{3-}, SO_4^{2-} & F^- concentrations values ranged from 59.22-138.94,

71.36-184.24, 38.65-445.32, 38.92-493.69 mg/L respectively. Chloride is a very moderate anion in leachate that is only oxidized during dilution. The study results showed that fresh leachate chloride (Cl-) increased after being deposited in an MSW landfill. (Rafizul and Alamgir, 2012) reported that initial concentrations were high, with elapsed time concentrations decreasing because the high amount of leachate production leads to chloride dilution. In this context, Cl^- is a non-biodegradable compound, but the variation of its concentrations was decreased slowly due to the washout effect (Rafizul et al., 2009). It can be concluded that COD and Cl^- are proportional because COD decreases, and Cl- is also the same with time. Three parameters were found to be high, including COD, pH, and chloride, but pH was found to be low. It was found that pH increases, concentrations of COD, and Cl^- were low. However, it illustrates that the study's findings were valid with the results obtained in the case of a closed lysimeter by (Rafizul and Alamgir, 2012; Tränkler et al., 2005). SO_4^{2-} to Cl^-, ratio and pH in leachate increasing and decreasing with time from deposited in closed landfill simulator.

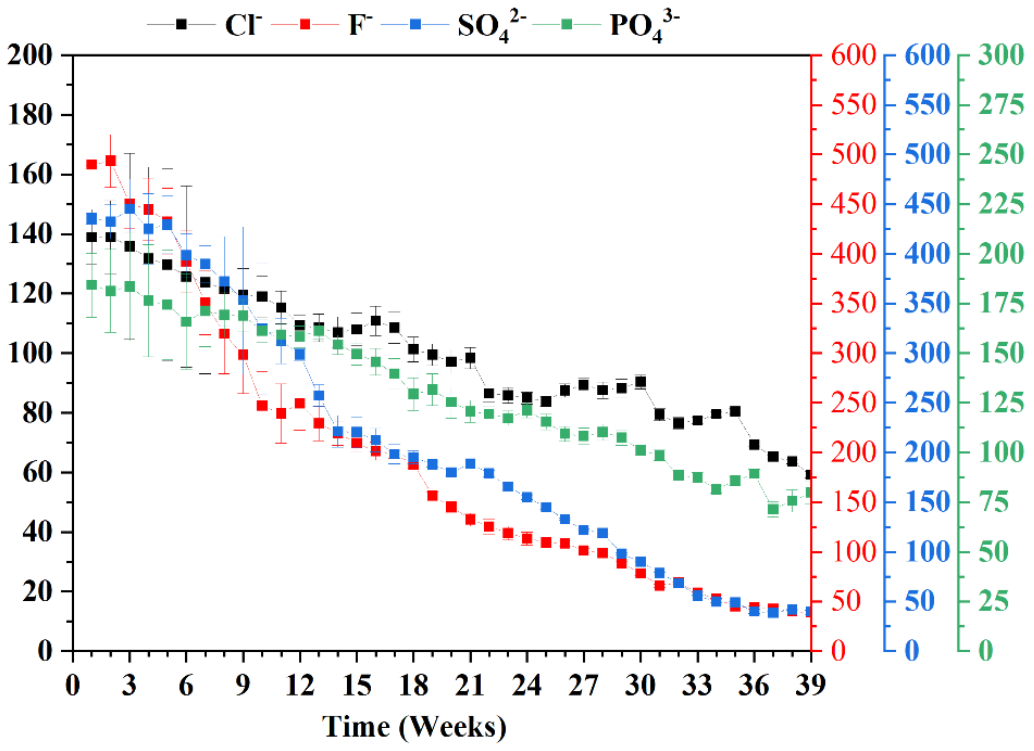


Fig. 6.25. Variable inorganic nonmetallic constituents in fresh leachate with time

It is noted that sulfate-chloride increases up to the 10th week and decreases continuously because of a gradual decrease in sulfate, while the chloride decreases slowly. SO_4^{2-}/Cl^- ratio

is a biological process, sulfate degradation in the first phase before methane production starts. In anaerobic conditions, sulfate reduction rapidly due to SO_4^{2-} is reduced to S^{2-} and then precipitates with various metals. SO_4^{2-} decreases over time, which corresponds to a reduction in VFAs. (Chian and Dewalle, 1976). The waste stabilization rates were Cl^- (57.3%), PO_4^{3-} (61.26%), SO_4^{2-} (91.32%) & F^- (92.11%), respectively, was removed within 39 weeks of the degradation process

6.2.2.11 Relation of heavy metals iron (Fe), manganese (Mn), magnesium (Mg), zinc (Zn), copper (Cu), cadmium (Cd), and chromium (Cr) in fresh leachate

Fig. 6.26 shows the variation of heavy metals in fresh leachate. The heavy metals concentrations recorded from fresh leachate were ranged from 69.33-598.93, 52.42-349.42, 2.53-28.53, 1.04-26.24, 0.42-29.83, 0.10-3.52, 8.90-50.24 mg/L for Fe, Mg, Mn, Zn, Cr, Cu, and Cd respectively. By increasing concentrations in the transient aqueous medium, many processes, including complexation to inorganic and organic linkers and sorption to colloids, can mobilize heavy metals and significantly immobilize heavy metals from the landfill due to sorption and precipitation. Also, metal solubility with the sulfides and carbonates is low and capable of forming precipitates with heavy metals like Cd, Ni, Zn, Cu, and Pb (Christensen et al., 2001). The simulated landfill's initial pH was acidic, and heavy metal concentrations were high during this period. As time progressed, metal concentrations were decreased because pH was close to neutral.

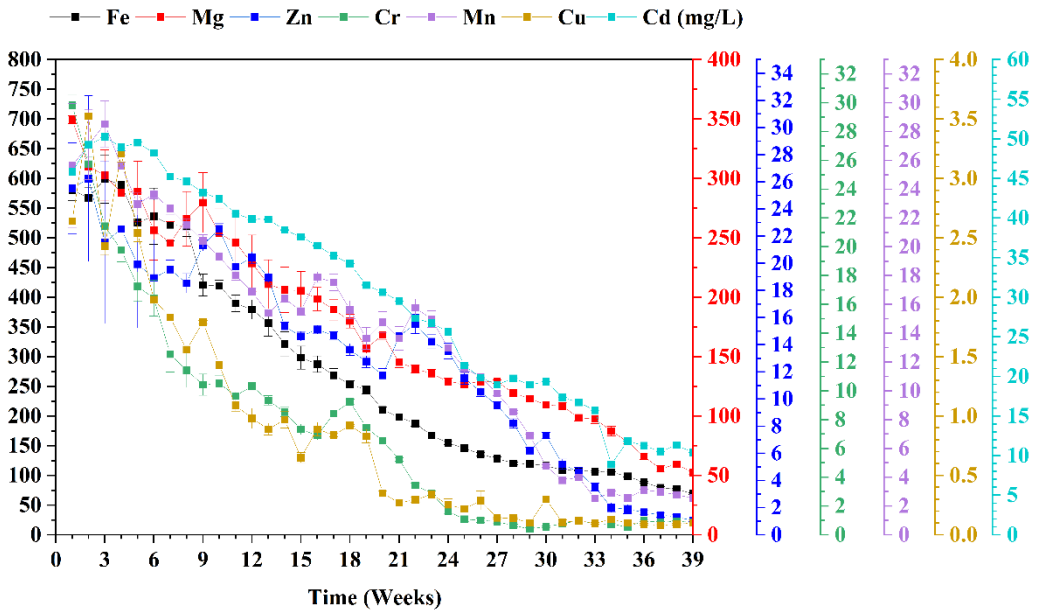


Fig. 6.26. Variation of heavy metals in fresh leachate with time

Organic matter has a high sorptive potential at neutral to high pH, according to researchers. (Reinhart and Grosh, 1998; Christensen et al., 1994; Kjeldsen and Christophersen, 2001; Ishchenko, 2018). Due to complex anionic adsorption, a decrease in heavy metal solubility in acidic conditions is possible (Flyhammar and Håkansson, 1999). The order of heavy metals in this study was Fe>Mg>Cd>Cr>Mn>Zn>Cu. The authors investigated that effective decomposition of organic matter can increase the solubility and mobility of heavy metals in landfills by forming dissolved compounds of high acids with heavy metals (Ciavatta et al., 1993). However, the analysis may conclude that a high organic matter content in MSW supports the leaching of heavy metals from a landfill body.

6.2.3 Statistical analysis

6.2.3.1 Descriptive statistics

Descriptive statistics include maximum, minimum, mean, and standard deviation of the leachate parameters presented in Table 6.8. The average values of ambient and landfill temperatures were 23.22^{o}C and 35.98^{o}C, respectively. The inside temperature is more than the outside temperature since the waste degradation process releases more heat, which leads to an increase in temperature inside the landfill. The BOD_5 and COD are the essential parameters that indicate the organic strength of leachate as the average concentration of 18660 and 37862 mg /L, respectively.

Table 6.8. Descriptive statistics for the ambient temperature, landfill temperature, and leachate parameters.

Parameters	Max	Min	Mean	Stdev
Ambient temp (^{o}C)	30.60	18.20	23.22	3.81
Landfill temp (^{o}C)	45.67	32.54	35.98	2.63
pH	5.28	5.04	5.12	0.07
EC (mS/cm)	17.28	5.16	10.22	3.02
BOD_5	39,130	7,041	18,660.79	7,386.12
COD	71,630	15,692	37,862.96	15,651.38
VFA	13,477.13	5,024.39	7,677.97	2,518.49
BOD/COD ratio	0.67	0.40	0.50	0.07
TS	33,200	9,077.00	17,984.56	8,204.38
TDS	20,899.67	5,067.00	12,448.15	4,820.70
TSS	9,246	1,285.00	3,727.28	2,557.98
TFS	15,720	890.00	6,331.80	4,767.80
TVS	21,870	8,247.37	11,797.41	3,892.28
Na	1,662	563.80	995.47	405.21
k	4,442	857	2,407.96	1,319.62
Ca	2,193	1183.12	1,658.18	329.59

NH_4^+ -N	749.16	254.66	441.66	162.66
NO_2^-	220.30	116.36	181.59	31.25
TKN	0.76	0.11	0.40	0.20
NO_3^-	319.53	65.85	169.89	74.83
Fe	598.93	69.33	271.94	176.50
Mg	349.42	52.42	174.70	81.30
Zn	26.24	1.04	13.02	7.44
Cr	29.83	0.42	7.29	7.57
Mn	28.53	2.53	13.75	7.91
Cu	3.52	0.10	0.96	0.92
Cd	50.24	8.90	30.09	13.43
C^{l-}	138.94	59.22	99.57	21.94
F^-	493.69	38.92	192.01	138.61
SO_4^{2-}	445.32	38.65	208.89	134.59
PO_4^{3-}	184.24	71.36	130.23	34.88

*All values in mg/L except pH, EC, BOD_5/COD ratio, ambient, and landfill temperature.

- ***Correlation analysis***

Correlation coefficients of leachate parameters, ambient, and inside temperature are represented in Table 6.9. Values for the relationship between ambient and inside temperature indicated that there is no relation between two temperatures because the addition of rainfall cools or controls the temperature, and heat is produced through biological activity. The coefficient (r=0.27) represents a low correlation significance between two variables. Other literature reported that there is no relation between ambient and waste temperature (R^2=0.0038) (Kumar, 2007). pH indicates a very weak correlation with ambient and inside temperature, and temperatures have no relation. EC has a moderate positive correlation that described temperature increases with EC increases. Ambient and inside temperature moderately correlate with biological parameters, as the optimum temperature is required for biological activity. pH has a weak negative correlation with leachate parameters like ammoniacal nitrogen (r =-0.14) and TSS (r =-0.19). EC has a strong positive correlation with solids, cations and a negative correlation with ammoniacal nitrogen. Biological parameters like BOD_5 and COD give a strong correlation (r=0.91), which indicates both biological and chemical degradation in a pronounced manner (Karthik, 2012). BOD_5 and COD have a strong correlation with other leachate parameters. VFA has a moderate correlation with solids, as most VFA is soluble (Marañón et al., 2008). Solids parameters show a strong positive correlation with all parameters; however, pH had a negative correlation with solids. Cations also showed a strong correlation with biological and solids parameters. Still, pH has a negative correlation with cations, as cations tend to

form hydroxides and other precipitates at higher pH. The variables analyzed during the period of volumetric precipitation were characterized by behavior with a high Pearson correlation. This correlation demonstrates that based on the determination of one of the parameters, the others can be estimated and controlled during the treatment process.

Table 6.9. Correlation analysis between leachate parameters, ambient, and landfill temperature

parameters	Ambient temp	Landfill temp	pH	EC	BOD_5	COD	VFA	TS	TDS	TSS	TFS	TVS	Na	k	Ca	NH_4^+-N	NO_2^-	TKN	NO_3^-	Fe	Mg	Zn	Cr	Mn	Cu	Cd	C^{l-}	F^-	SO_4^{2-}	PO_4^{3-}
Ambient temp	1																													
Landfill temp	0.34	1																												
pH	-0.09	-0.39	1																											
EC	0.53	0.58	-0.79	1																										
BOD_5	0.57	0.61	-0.57	0.89	1																									
COD	0.63	0.52	-0.61	0.91	0.95	1																								
VFA	0.50	0.09	-0.48	0.66	0.59	0.66	1																							
TS	0.68	0.43	-0.62	0.91	0.89	0.95	0.67	1																						
TDS	0.53	0.30	-0.78	0.88	0.79	0.90	0.71	0.94	1																					
TSS	0.76	0.45	-0.19	0.67	0.79	0.80	0.35	0.84	0.64	1																				
TFS	0.63	0.32	-0.67	0.89	0.83	0.92	0.68	0.97	0.97	0.76	1																			
TVS	0.70	0.56	-0.50	0.87	0.90	0.93	0.58	0.96	0.84	0.89	0.89	1																		
Na	0.66	0.21	-0.56	0.81	0.79	0.86	0.89	0.91	0.90	0.67	0.91	0.81	1																	
k	0.66	0.38	-0.66	0.91	0.86	0.94	0.75	0.98	0.96	0.78	0.97	0.93	0.94	1																
Ca	0.58	0.36	-0.66	0.85	0.70	0.73	0.77	0.75	0.76	0.51	0.78	0.66	0.81	0.79	1															
NH_4^+-N	-0.71	-0.07	-0.14	-0.30	-0.46	-0.42	-0.48	-0.44	-0.29	-0.61	-0.43	-0.43	-0.58	-0.44	-0.55	1														
NO_2^-	0.11	0.39	-0.87	0.73	0.56	0.61	0.49	0.60	0.71	0.21	0.62	0.54	0.53	0.63	0.54	0.17	1													
TKN	0.55	0.36	-0.72	0.89	0.81	0.83	0.68	0.91	0.90	0.65	0.93	0.81	0.87	0.91	0.84	-0.44	0.64	1												
NO_3^-	0.56	0.50	-0.77	0.97	0.86	0.89	0.66	0.93	0.91	0.70	0.92	0.88	0.84	0.94	0.85	-0.34	0.72	0.93	1											
Fe	0.67	0.46	-0.64	0.93	0.91	0.96	0.67	0.99	0.93	0.83	0.97	0.95	0.90	0.98	0.79	-0.47	0.61	0.91	0.95	1										
Mg	0.59	0.51	-0.73	0.97	0.93	0.95	0.72	0.96	0.93	0.74	0.94	0.92	0.89	0.96	0.82	-0.39	0.69	0.92	0.96	0.97	1									
Zn	0.47	0.47	-0.81	0.94	0.83	0.88	0.73	0.90	0.93	0.59	0.88	0.85	0.84	0.93	0.78	-0.19	0.76	0.86	0.93	0.90	0.95	1								
Cr	0.69	0.66	-0.59	0.94	0.90	0.91	0.54	0.91	0.80	0.84	0.86	0.92	0.76	0.89	0.79	-0.45	0.55	0.84	0.92	0.94	0.93	0.85	1							
Mn	0.49	0.48	-0.82	0.94	0.82	0.88	0.62	0.93	0.94	0.67	0.92	0.88	0.82	0.94	0.77	-0.21	0.75	0.90	0.96	0.94	0.95	0.96	0.89	1						
Cu	0.71	0.57	-0.42	0.81	0.84	0.84	0.44	0.89	0.75	0.92	0.84	0.91	0.72	0.85	0.66	-0.51	0.41	0.78	0.83	0.89	0.84	0.74	0.92	0.80	1					
Cd	0.55	0.40	-0.80	0.95	0.84	0.91	0.72	0.95	0.98	0.66	0.96	0.86	0.90	0.97	0.84	-0.34	0.72	0.94	0.96	0.95	0.97	0.96	0.87	0.97	0.79	1				
C^{l-}	0.54	0.51	-0.75	0.97	0.93	0.93	0.67	0.95	0.92	0.72	0.94	0.90	0.86	0.94	0.81	-0.36	0.71	0.94	0.96	0.96	0.98	0.93	0.91	0.95	0.83	0.96	1			
F^-	0.66	0.59	-0.65	0.94	0.92	0.96	0.58	0.97	0.90	0.85	0.94	0.96	0.83	0.95	0.76	-0.41	0.62	0.89	0.94	0.98	0.96	0.89	0.97	0.94	0.92	0.93	0.96	1		
SO_4^{2-}	0.61	0.46	-0.70	0.94	0.90	0.95	0.66	0.99	0.95	0.79	0.97	0.95	0.88	0.98	0.76	-0.35	0.67	0.90	0.95	0.99	0.98	0.94	0.92	0.97	0.86	0.97	0.97	0.98	1	
PO_4^{3-}	0.55	0.45	-0.79	0.96	0.87	0.92	0.76	0.95	0.96	0.66	0.95	0.88	0.91	0.97	0.83	-0.34	0.74	0.92	0.96	0.96	0.98	0.96	0.88	0.96	0.79	0.99	0.97	0.94	0.97	1

6.2.3.3 Obtaining k values for leachate parameters

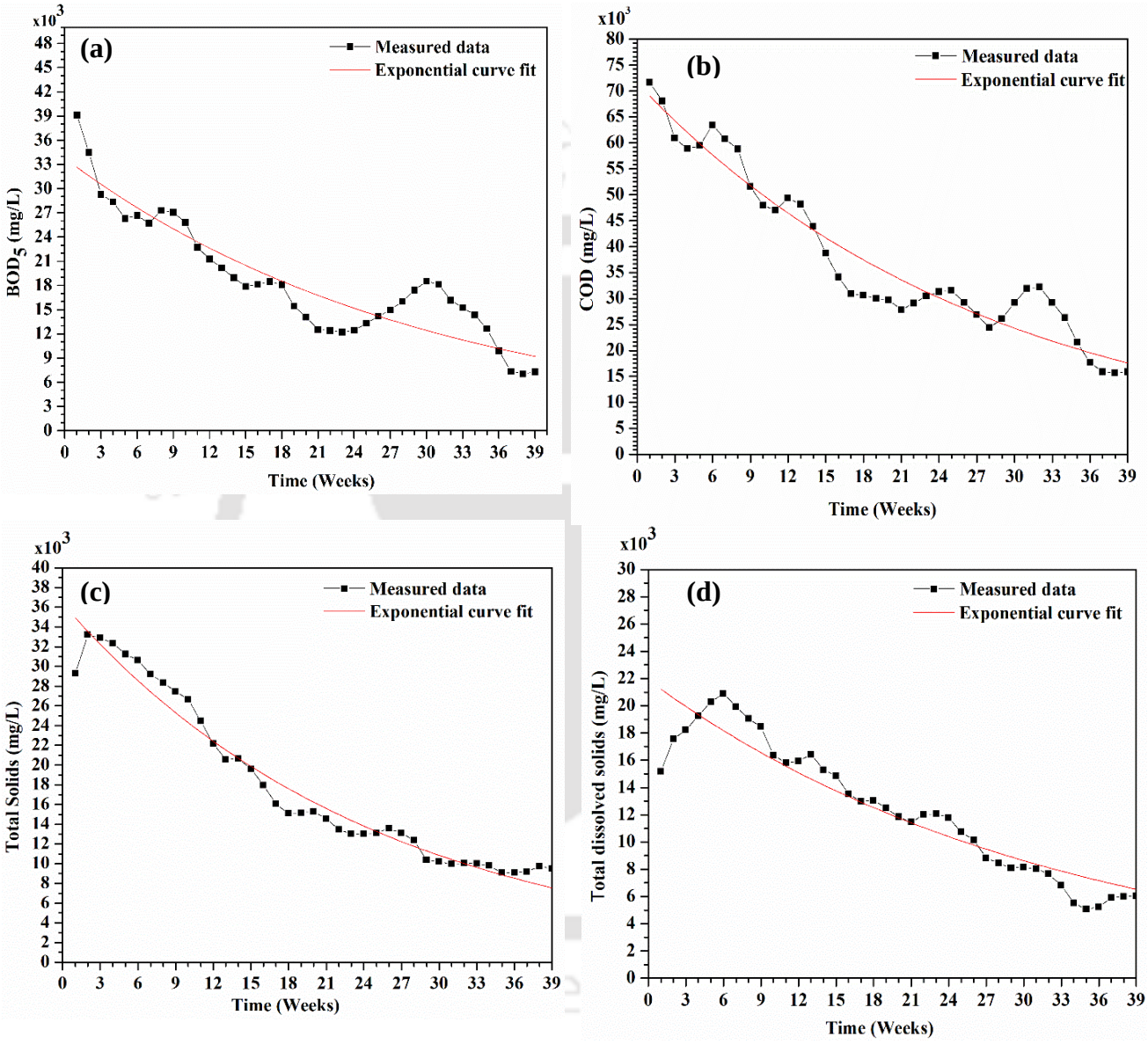
The first-order exponential decay constants (k) were obtained using the Levenberg–Marquardt algorithm, which is an iterative method that is used to solve non-linear problems. The decay constant k is the constant b. This algorithm was applied to leachate parameters to get the best exponential curve fits. The data shows a rising part and a declining part in the plot. In this study, the data used to fit the exponential curve started from the peak BOD_5, COD, total solids, and other leachate parameters. Therefore, the fitted curve simulates only a declining part of the data, as shown in Fig. 6.26. Obtained k values for BOD_5, COD, and total solids were 0.033, 0.036, and 0.04 $week^{-1,}$ respectively, as shown in Table 6.10 along with the values of 'a'. The k value represents the rate of biodegradation for the mass of waste in the landfill. Leachate parameters show a higher k value that can be

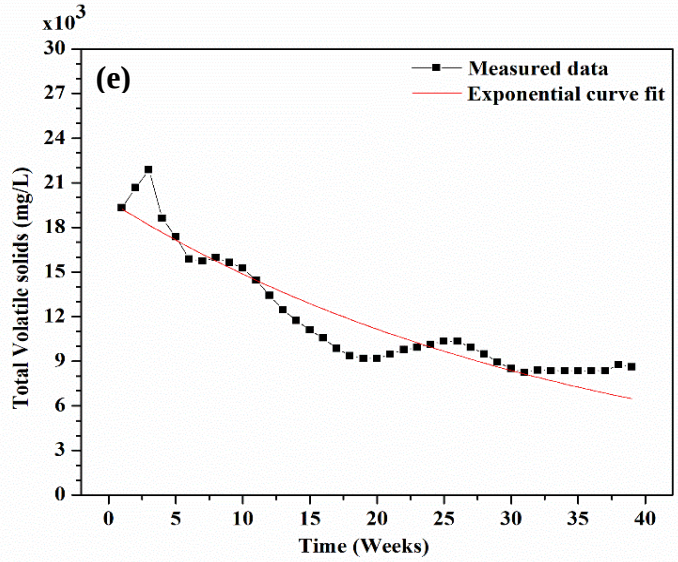
explained by a high proportion of food waste deposited in the landfill. In this study, it was observed that the addition of rainfall accelerates the decomposition of waste, and higher k values of parameters were obtained, resulting due to the fast degradation of waste. Another field study by (Ozkaya et al., 2006) also presented a first-order exponential decay equation to simulate measured BOD_5 and COD data. Furthermore, (Kouzeli-Katsiri et al., 1999) developed two models using first-order kinetics for COD with and without leachate recirculation from a lab-scale study with excellent correlation coefficients greater than 0.90. The models, which were developed in the past studies, included added nutrients and recirculation of leachate to enhance the biodegradation of waste and estimated k values (Gonullu, 1994; Kouzeli-Katsiri et al., 1999; Wigh, 1979), but in this study, real-life conditions (addition of rainfall, waste composition) were considered and k values were estimated. A laboratory-scale study was conducted, which operated landfill with controlled temperature and sludge addition to the waste. The estimated k values for BOD_5 and COD of 100% food waste were 0.018 day^{-1} and 0.017 $day^{-1,}$ respectively, and estimated k values for 60% food waste were found to be 0.026 day^{-1} for BOD 0.028 day^{-1} for COD. Therefore, concentrations were affected by microbial degradation, dilution, and dissolution (Bhatt et al., 2016). The correlation coefficient of the models showed the goodness of fit of the curve, i.e., BOD_5 (R^2=0.83), COD (R^2 =0.92), and total solids (R^2 =0.96), respectively. The model, which was developed in this study, indicates that observed data fitted significantly.

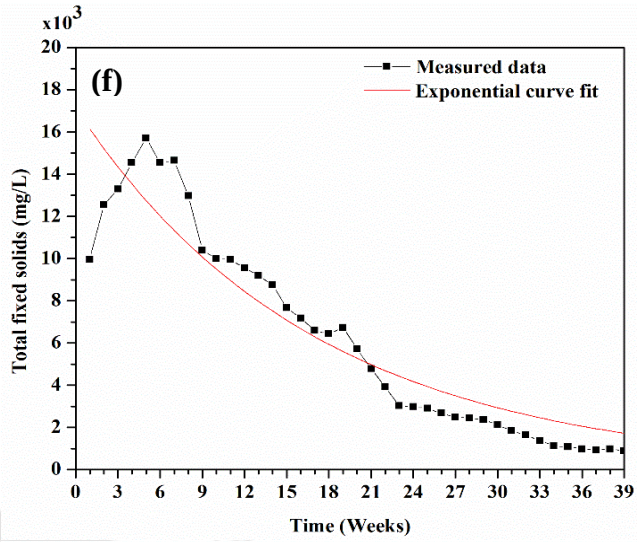
Table 6.10. Model parameters and coefficients for leachate parameters

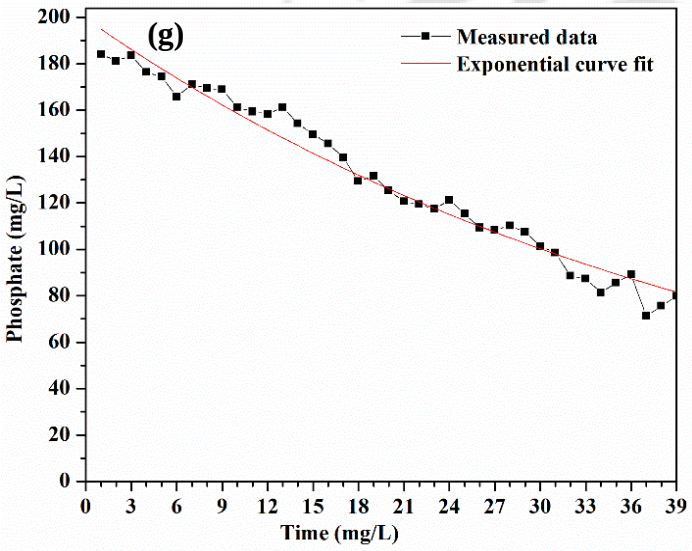
Parameters	a	b	R^2	
BOD_5	33.75	0.033	0.83	
COD	71.59	0.036	0.92	
Ca	2.17	0.014	0.66	
K	5.36	0.046	0.94	
Na	1.75	0.031	0.77	
TS	36.36	0.04	0.96	
TDS	21.89	0.031	0.88	
TFS	17.11	0.058	0.87	
TVS	19.79	0.028	0.87	$y = a * e^{(-b*x)}$
TKN	0.8	0.039	0.82	
NO_3^-	328.48	0.037	0.93	
chloride	142.42	0.019	0.96	
fluoride	553.86	0.066	0.99	
sulphate	529.53	0.055	0.96	
phosphate	199.44	0.022	0.96	
Fe	688.12	0.059	0.98	
Mg	354.72	0.04	0.97	

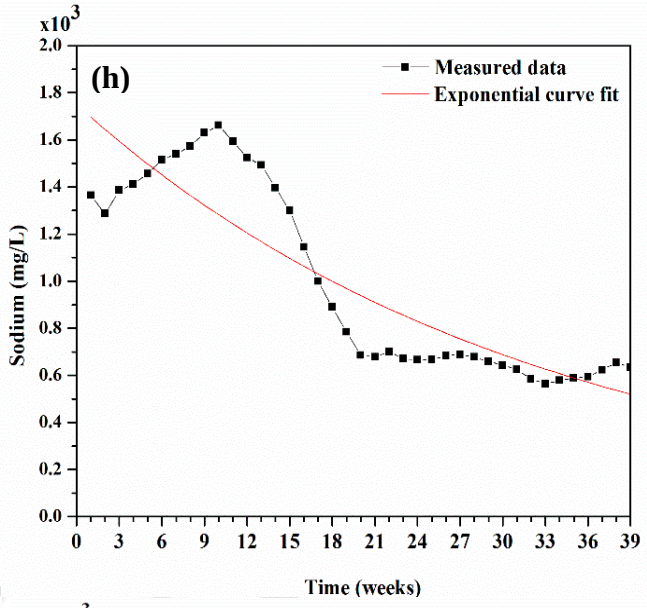
Zn	28.33	0.044	0.86
Cr	30.08	0.098	0.96
Mn	30.87	0.046	0.9
Cu	3.58	0.091	0.86
Cd	58.06	0.036	0.93

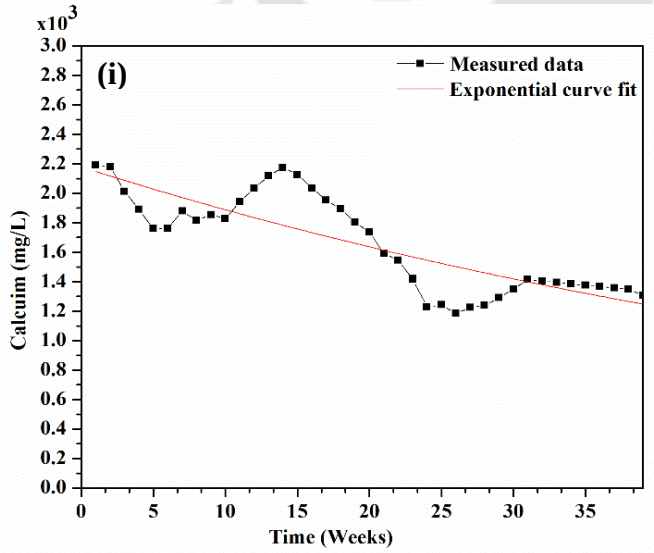


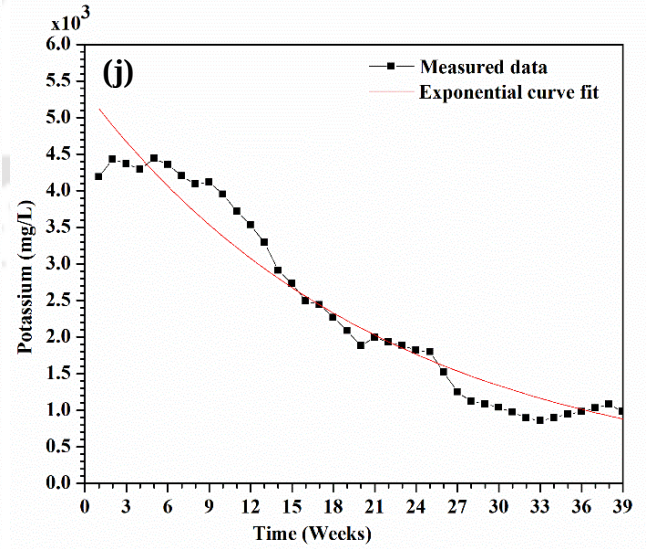

(e)
Measured data
Exponential curve fit
x10^3
Total Volatile solids (mg/L)
Time (Weeks)

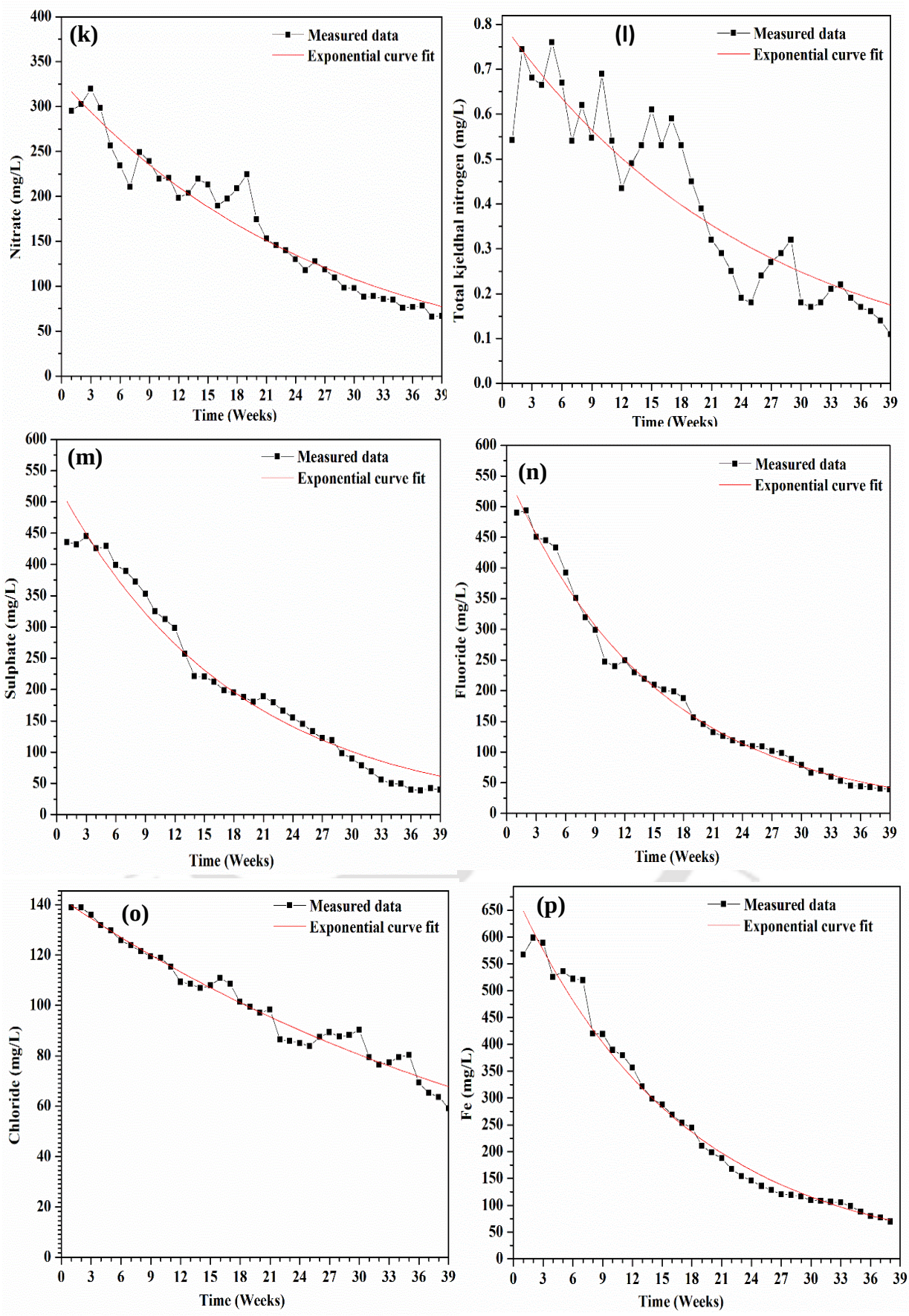

(f)
Measured data
Exponential curve fit
x10^3
Total fixed solids (mg/L)
Time (Weeks)


(g)
Measured data
Exponential curve fit
Phosphate (mg/L)
Time (mg/L)


(h)
Measured data
Exponential curve fit
x10^3
Sodium (mg/L)
Time (weeks)


(i)
Measured data
Exponential curve fit
x10^3
Calcuim (mg/L)
Time (Weeks)


(j)
Measured data
Exponential curve fit
x10^3
Potassium (mg/L)
Time (Weeks)


(k)
Measured data
Exponential curve fit
Nitrate (mg/L)
Time (Weeks)
(l)
Measured data
Exponential curve fit
Total kjeldhal nitrogen (mg/L)
Time (Weeks)
(m)
Measured data
Exponential curve fit
Sulphate (mg/L)
Time (Weeks)
(n)
Measured data
Exponential curve fit
Fluoride (mg/L)
Time (Weeks)
(o)
Measured data
Exponential curve fit
Chloride (mg/L)
Time (Weeks)
(p)
Measured data
Exponential curve fit
Fe (mg/L)
Time (Weeks)

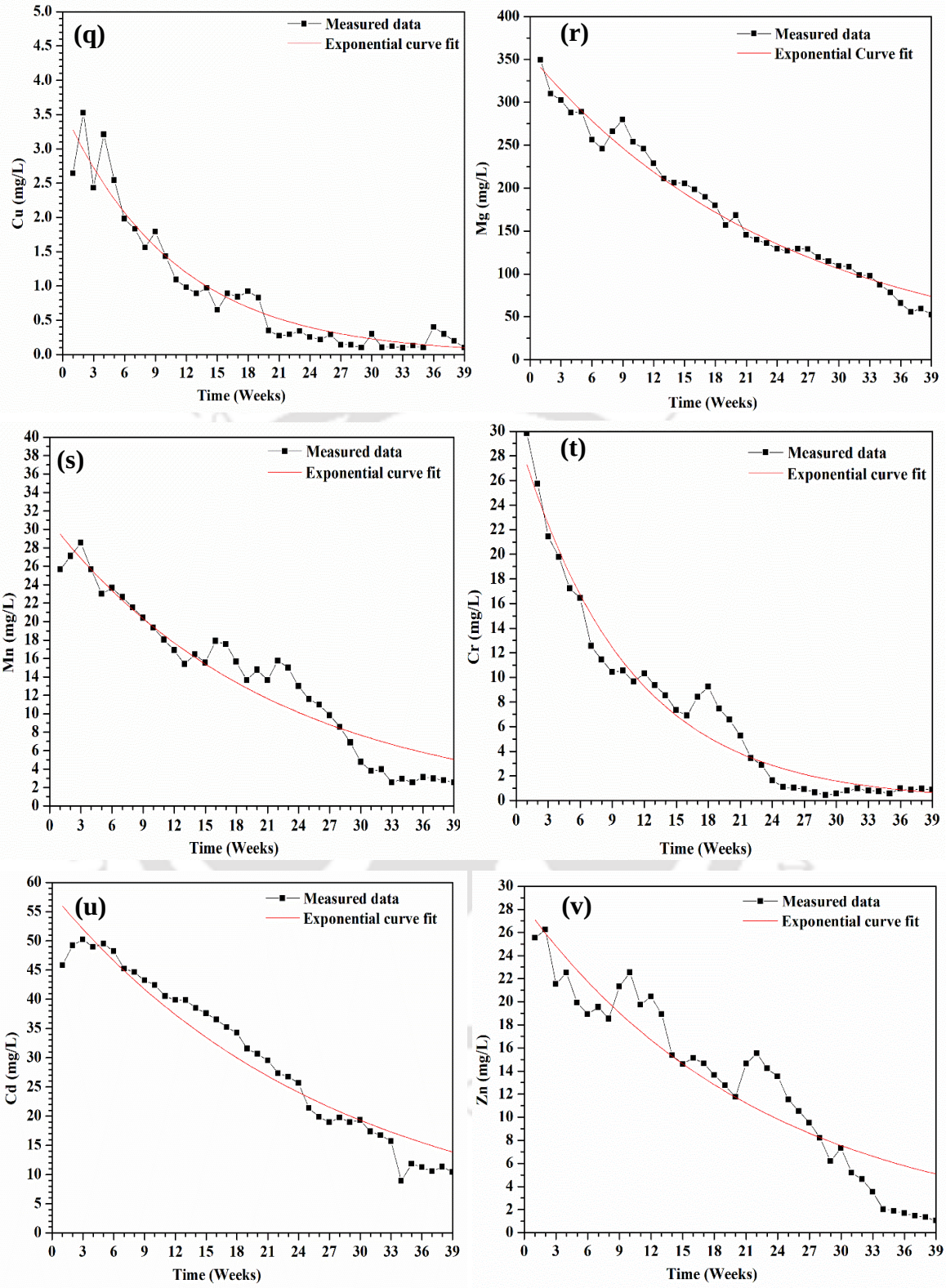


Fig. 6. 27. Exponential decay models for (a) BOD_5, (b) COD, (c) TS, (d) TS, (e) TVS, (f) TFS, (g) phosphate, (h) sodium, (i) calcium, (j) potassium, (k) nitrate, (l) TKN, (m) sulphate, (n) fluoride, (o) chloride, (p) Fe, (q) Mg, (r) Mn, (s) Cr, (t) Cu, (u) Zn, and (v) Cd

6.2.4 Comparison of landfill gas emissions (LFG) distribution pattern in R3

The gas composition for the reactor operated with rainfall is given in Fig. 6.28. The production of waste degradation was principally CO_2 and was likely due to the high rate of carbon reduction in the waste decomposition process, as presented in Eq. 6.8. The maximum CO_2 was found that 59.85% signifies the breakdown of carbohydrates, proteins, and lipids that composite organic waste produces a high amount of CO_2 in a landfill. The methane production was the less initial phase of degradation due to pH value not being favorable for methanogenesis and a high VFA toxic to methanogens (pH should be in the range of 6-8). Evolved CO_2, due to substantial degradation, exerts high partial pressure on leachate, and it dissolves, forming carbonic acid, which decreases the pH (Khattabi et al., 2002a). H_2 was found to have a maximum of 2.92% during the early transition stage, and its concentration was decreased after methane production was slowly increased from 1.56% to 20.63%. The nitrogen was found in the landfill was a maximum of 31% and a minimum of 1.53%. Interestingly, empirical data from landfill biogas recovery systems have also periodically documented N_2 recovered from deep landfills without any possibility of atmospheric input, suggesting that ANAMMOX processes may be naturally occurring in deep landfill settings (Meyer-Dombard et al., 2020). Another reason could be the denitrification of nitrate reduction to form nitrogen gas from landfills, as presented in Eq. 6.9. It was hypothesized that active and well-established methanogenic populations could tolerate and function under acidic pH conditions in landfill ecosystems (Kasali et al., 1988). The authors were able to isolate acid-tolerant methanogens that could grow at a pH of 5.5 (Juhasz et al., 1997). Both the hydrolytic and acetogenic bacteria produce hydrogen, which is a substrate for methanogens. The accumulation of hydrogen was found to be an indication of an imbalance in the microbial population (Barlaz et al., 1989). Researchers have established that utilizing H_2 concentrations to monitor landfill status in high-temperature landfills. They found that samples with higher rates of methanogenesis had lower hydrogen concentrations and more neutral pH values than those with lower rates of methanogenesis because methanogens or other organisms rapidly consumed H_2, and an apparent steady-state was achieved (Mormile et al., 1996). Other important intermediates are organic acids. It has been found that high methane-producing samples have low organic acid concentrations, while low methane-producing samples accumulate high concentrations of organic acids during the initial fermentation reactions. The study proves

that moisture addition to the waste would be a positive effect on methane production even at the acidic phase. R3 showed improved methane production than in R1 (operated without rainfall). More methane has been produced than R1. This confirmed that water added for flushing and diluting organic strength, particularly VFA concentration, more quickly provided a favorable environment for microorganisms during hydrolysis and the acidogenesis stage. However, the enhancement of methane production was also dependent on providing favorable environmental conditions for methanogen. The results indicated that high gas production due to rainfall addition to the landfill. The increase in CH_4: CO_2 production ratio following periods of heavy rainfall can be attributed to the dissolution of CO_2 and downward movement with the leachate. It has been speculated that CO_2 acts as a methanogenesis inhibitor through the raising of the redox potential (Wreford et al., 2000). The CH_4: CO_2 ratio was increased from 0.02 to 0.53 in the reactor (operated with rainfall).

$$C_6H_{12}O_6 + O_2 \rightarrow CO_2 + H_2O + \text{Biomass} + \text{Heat} \quad (6.8)$$

$$NH_4^+ + NO_2^- \rightarrow N_2 + 2H_2O \quad (6.9)$$

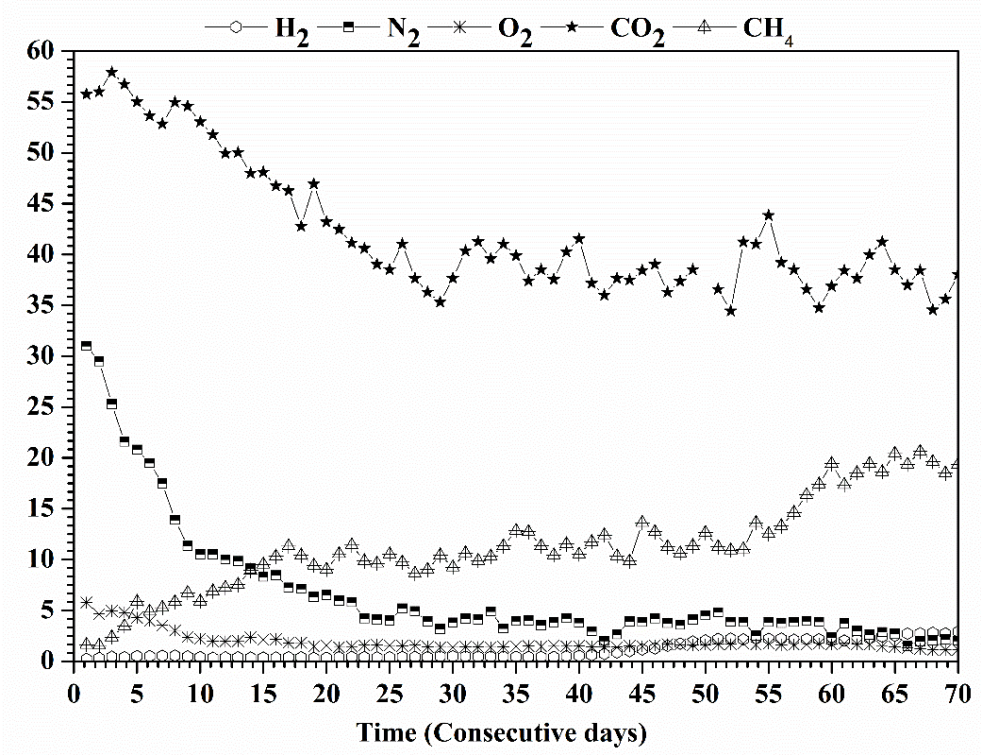


Fig. 6.28. Distribution pattern of LFG emissions in R3 (operated without rainfall)

6.2.4.1 Statistical analysis

- ***Descriptive analysis of gas components, ambient and landfill temperature***

The reactor has a higher inside (waste) temperature than ambient temperature because exothermic reactions increase the landfill temperature during the degradation process. Table 6.11 shows the descriptive analysis of gas compositions and external factors. The high amount of CO_2 produces during the waste degradation process and methane concentration was less due to acidic pH in the reactor. It was confirmed that the decrease in CO_2 and methane concentration ow methane concentrations were considered to be another proof of washout, indicating the inability of the system to develop an active methanogenic population and enhance waste stabilization.

Table 6.11. Descriptive analysis of gas components, ambient, and landfill temperature (^oC)

Parameters	Max	Min	Mean	Stdev
Landfill Temperature	45.67	32	35.80	2.52
Ambient temperature	30.4	17.3	21.79	3.17
H_2	2.92	0.23	1.09	0.88
N_2	31	1.53	6.66	6.37
O_2	5.76	1.10	1.91	0.96
CO_2	59.85	35.6	43.57	6.70
CH_4	20.63	1.56	11.33	4.52

- ***Correlation coefficients matrix between gas components, ambient and landfill temperature (^oC) in R3***

Correlation analysis is a quantitative analytical method used to assess the degree of correlation between the variables involved. Correlation is a statistical technique that tells the linear association between variables. The magnitude of the correlation determines the correlation coefficient r, which signifies how closely two variables. The coefficient represents -1 to +1, value can vary +1 (high positive correlation) and -1 (high negative correlation) (Swinscow and Campbell, 2002). Correlation coefficients of gas components, ambient, waste temperature, and represented in table 6.12. The effect of ambient waste/landfill temperature on gas components and among the parameters were examined. The positive correlation (r=0.41) between waste and ambient temperature, while other parameters like H_2 (r=0.14), N2 (r=0.45) has a weak correlation because N_2 generates from waste pockets and hydrogen evolves through biodegradation. CO_2 has a moderate

correlation with waste temperature because the temperature increase through the degradation process and evolves CO_2. CH_4 had a strong negative correlation with waste temperature. The waste temperature had a good correlation with CO_2 since the waste temperature is a function of biochemical reactions, and CO_2 may increase the temperature inside the landfill. Hydrogen has a negative correlation with CO_2 (r=-0.46) and N_2 (r=-0.43), respectively. Hydrogen has a moderate correlation with methane (r =0.71). Nitrogen has a good positive correlation with CO_2 (r=0.87) and negative correlating with CH_4 (r =-0.75). O_2 was significantly correlated with CO_2 (r=0.82) and a negative correlation with CH_4 (r=-0.67) since methanogenic bacteria can survive without oxygen inside the landfill. The reaction in the anaerobic decomposition releases a minimal amount of heat, and the product gas mainly contains about 54% methane and 46% carbon dioxide (Themelis and Ulloa, 2007), which could reasonably explain the strong correlations observed above. The significant correlations between oxygen and major components (CH_4 and CO_2) are obvious during the anaerobic reaction. Landfilled organic waste generates landfill gas (LFG) when it degrades in an anaerobic environment (Niskanen et al., 2013).

Table 6.12. Correlation coefficients matrix between gas components, ambient, and landfill temperature

Parameters	**Landfill temp**	**Ambient temp**	**H_2**	**N_2**	**O_2**	**CO_2**	**CH_4**
Landfill temp	1						
Ambient temp	0.53	1					
H_2	0.14	-0.29	1				
N_2	0.45	0.77	-0.43	1			
O_2	0.47	0.74	-0.32	0.97	1		
CO_2	0.85	0.78	-0.49	0.87	0.82	1	
CH_4	-0.82	-0.49	0.78	-0.75	-0.67	-0.73	1

6.3 SUMMARY OF PHASE-II (OBJECTIVE-II (WITHOUT RAINFALL ADDITION) AND OBJECTIVE-III (WITH RAINFALL ADDITION))

- There is no leachate production in the landfill. It contains 38% of wet waste and 62% of dry waste due to the high amount of dry waste filled may not reach field capacity and signifies that developed countries have less problem handling the leachate issue.

- In this study, landfill operated without rainfall have waste stabilization rates of BOD_5 (72.2%) and COD (79.3%) within 54 weeks. Therefore, low leaching actions during the dry season resulted in a smaller volume but high-strength leachate quality.
- H_2 was found to a maximum of 2.12% during the early transition stage, and its concentration was decreased after methane production was slowly increased from 0.23% to 13.68%.
- In this study, landfill operated with rainfall has a positive effect on the acceleration of biological activity, as proven by a significant reduction in peak BOD_5 concentrations, which ranged from 7,041 to 39,310 mg/L and peak COD concentrations which ranged from 15,692 to 71,630mg/L, but leachate pH increase is lower as a result of carbon dioxide gas dissolution in leachate/water to form carbonic acid, due to higher partial pressure exerted by CO_2.
- Significant degradation of both BOD_5 and COD was observed in the reactor, i.e., 82% and 78%, respectively, within the period of 39weeks. It was observed that the waste stabilization rates of BOD_5 and COD followed similar trends, which indicated both pronounced biological and chemical degradation. The average BOD/COD ratio of the landfill was 0.53. The results showed that the landfill was in the acidogenic phase.
- First-order decay models were developed to predict the leachate parameters such as $kBOD_5$, kCOD, and TS. The correlation coefficient of the models showed acceptable goodness of fit of the curve, i.e., BOD_5 (R^2=0.83), COD (R^2=0.92), and total solids (R^2 =0.96), and also rate constants found were for $kBOD_5$ (0.033), kCOD (0.036), and ktotal solids (0.04).
- H_2 was found to have a maximum of 2.92% during the early transition stage, and its concentration was decreased after methane production was slowly increased from 1.56% to 20.63%.

CHAPTER

7

Fresh Landfill Leachate Treatment

This chapter enlightens the feasibility of treating fresh leachate from the high amount of organic waste in MSW landfills using a combined treatment system, i.e., the coagulation-flocculation process followed by an upflow anaerobic filter. Comparing the different coagulant treatment efficiency and treatability performance of high organic load with an upflow anaerobic filter having support media of plastic rings. In addition, metagenomics analysis was carried out to identify the microbial diversity in raw and treated landfill leachate. Finally, a biomethane potential test was done with different ratios of digested cow dung to obtain the best ratio for microbial addition to the landfill leachate treatment.

7.1 COAGULATION-FLOCCULATION TREATMENT OF FRESH LANDFILL LEACHATE

7.1.1 Characteristics of young/fresh leachate

Initially, R1 leachate was used to obtain the effective coagulant in terms of coagulant dose and pH using different conventional coagulants, i.e., potash alum $KAI(SO_4)_2 \cdot 12H_2O$, ferric chloride ($FeCl_3$), ferrous sulphate heptahydrate ($FeSO_4.7H_2O$), calcium hydroxide ($Ca(OH)_2$) and bentonite clay. The significance of selecting these coagulants is because of aluminum and iron coagulants' effectiveness, primarily from their ability to form multi-charged polynuclear complexes with enhanced adsorption properties. Polymer-like bentonite clay can destabilize the flocculation process. Initial characterization of leachate (L1) was carried out to obtain the landfill leachate's pollution load, as illustrated in Table 7.1. The average BOD/COD ratio was 0.5, which indicates a high amount of organic compounds that are partially degraded, and the value is higher in young leachate (Swati et al., 2011).

Table 7.1. Initial characteristics of stored landfill leachate (R1) based on one-year samplings

Parameters	Average (n=2)	Std. deviation
*pH	5.2	0.01
COD	47,861	4,759
BOD_5	22,058	2,344
TSS	6,110	134
*Turbidity (NTU)	750-874	12

*All are in mg/L except pH and turbidity

7.1.2 Optimization of coagulant dosage and pH for stored landfill leachate (L1)

7.1.2.1 Potash alum ($KAI(SO_4)_2 \cdot 12H_2O$)

Experiments were first conducted with optimum dosage while measuring the maximum reduction in turbidity concerning the various coagulant dosages, as shown in Fig. 7.1. The sample's initial turbidity was 750 NTU, and the minimum post-treatment turbidity obtained was 45 NTU. However, a maximum of 96.7% removal efficiency was achieved at 4 g/L of alum dosage. Experiments were performed at various pH levels by plotting the values between various pH levels and the percentage of reduction in turbidity, as shown in Fig. 7.1. The better findings were acquired at a pH of 8, obtaining the maximum reduction percentage. Potash alum reacts with calcium bicarbonate in leachate and forms a white gelatinous $Al(OH)_3$, as shown in Eq.7.1.

$$2KAI(SO_4)_2 + 3Ca(HCO_3)_2 \rightarrow 3CaSO_4 + K_2SO_4 + 2Al(OH)_3 + 6CO_2 \quad (7.1)$$

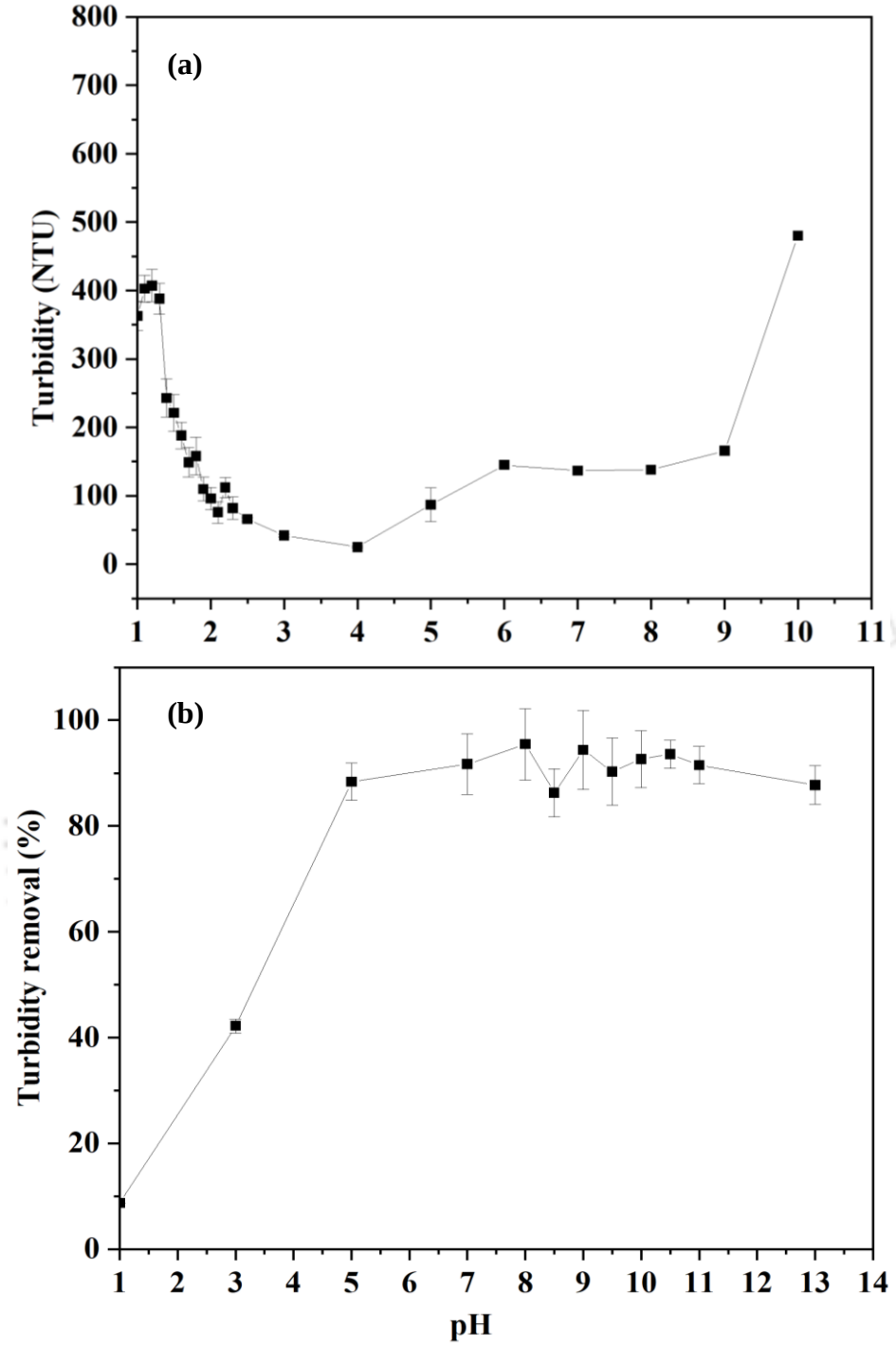
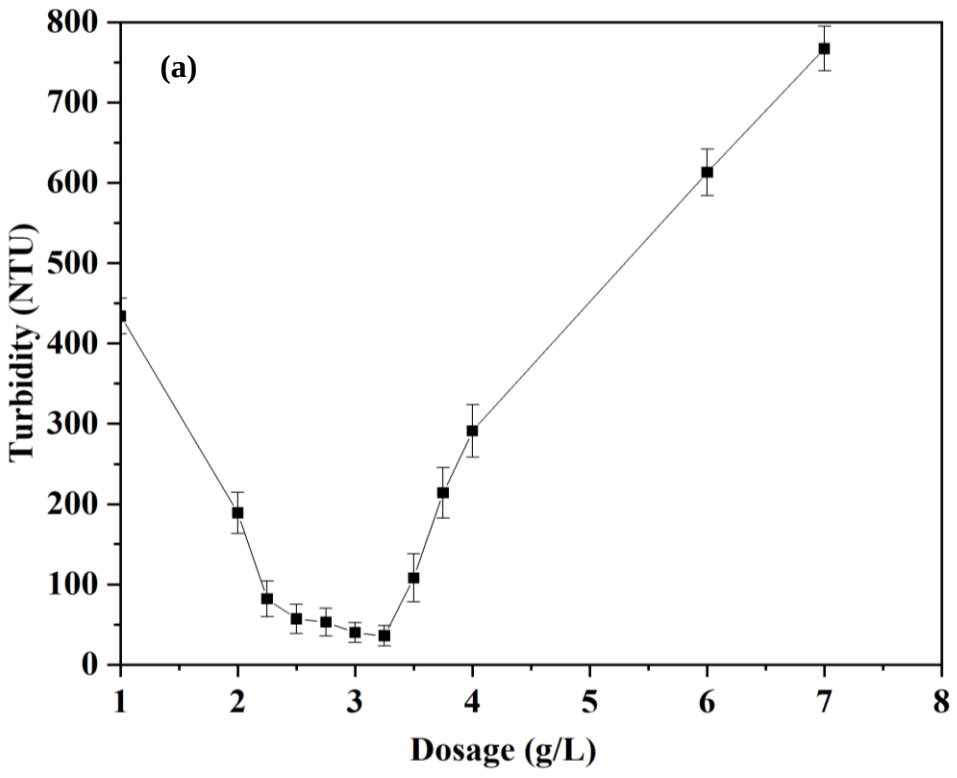


Fig. 7.1. (a) Effect of dosage on the turbidity (NTU) (b) Effect of pH range on turbidity removal (%) of leachate as a function of Potash alum dose

7.1.2.2 Ferric chloride ($FeCl_3$)

The mode of operation for ferric chloride was similar to the potash alum. Experiments were first conducted with optimum dosage while measuring the maximum reduction in turbidity concerning the various coagulant dosages, as shown in Fig. 7.2. The sample's initial turbidity is 874 NTU, and a maximum reduction in turbidity was obtained with final turbidity of 40 NTU. However, a maximum of 95.4% removal efficiency was achieved at 3.25 g/L of alum dosage. Experiments were performed at various pH levels by plotting the values between various pH levels and the percentage of reduction in turbidity, as shown in Fig. 7.2. The better findings were acquired at a pH of 5, 9-11, obtaining the maximum turbidity reduction. $FeCl_3$ reacts with calcium bicarbonate and forms the $Fe(OH)_2$, a brown precipitate, as shown in Eq. 7.2.

$$2FeCl_3 + 3Ca(HCO_3)_2 \rightarrow 2Fe(OH)_3 \downarrow + 3CaCl_3 + 6CO_2 \quad (7.2)$$



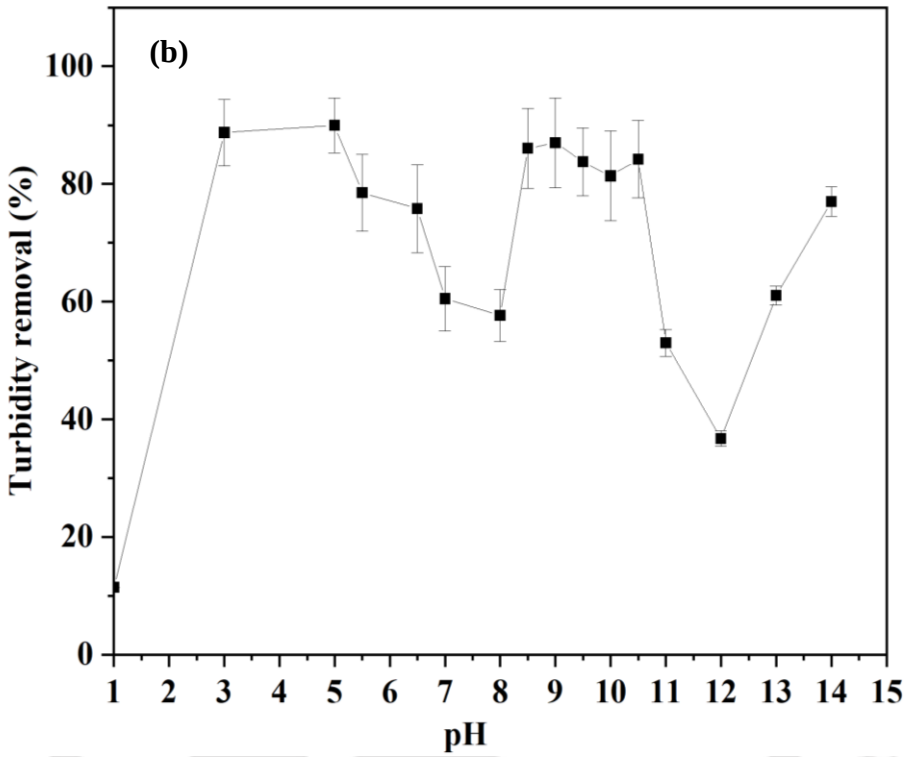


Fig.7.2. (a) Effect of dosage on the turbidity (NTU) (b) Effect of pH range on turbidity removal (%) of leachate as a function of ferric chloride

7.1.2.3 Ferrous sulphate heptahydrate ($FeSO_4.7H_2O$)

Optimum dosage aims to measure the maximum reduction in turbidity concerning the various coagulant dosages, as shown in Fig. 7.3. The sample's initial turbidity is 830 NTU, and the minimum post-treatment turbidity of 16 NTU was obtained. However, a maximum of 98% removal efficiency was achieved at 13 g/L of alum dosage. Experiments were performed at various pH levels by plotting the values between various pH levels and the percentage of reduction in turbidity, as shown in Fig. 7.3. The optimum pH was found to be 11, obtaining the maximum turbidity reduction. $FeSO_4.7H_2O$ reacts with calcium bicarbonate and forms the $Fe(OH)_3$, a brown precipitate, as shown in equation 7.3.

$$FeSO_4.7H_2O + Ca(HCO_3)_2 \rightarrow Fe(HCO_3)_2 + CaSO_4 + 7H_2O \quad (7.3)$$

$$4Fe(HCO_3)_2 + O_2 + 2H_2O \rightarrow 4Fe(OH)_2$$

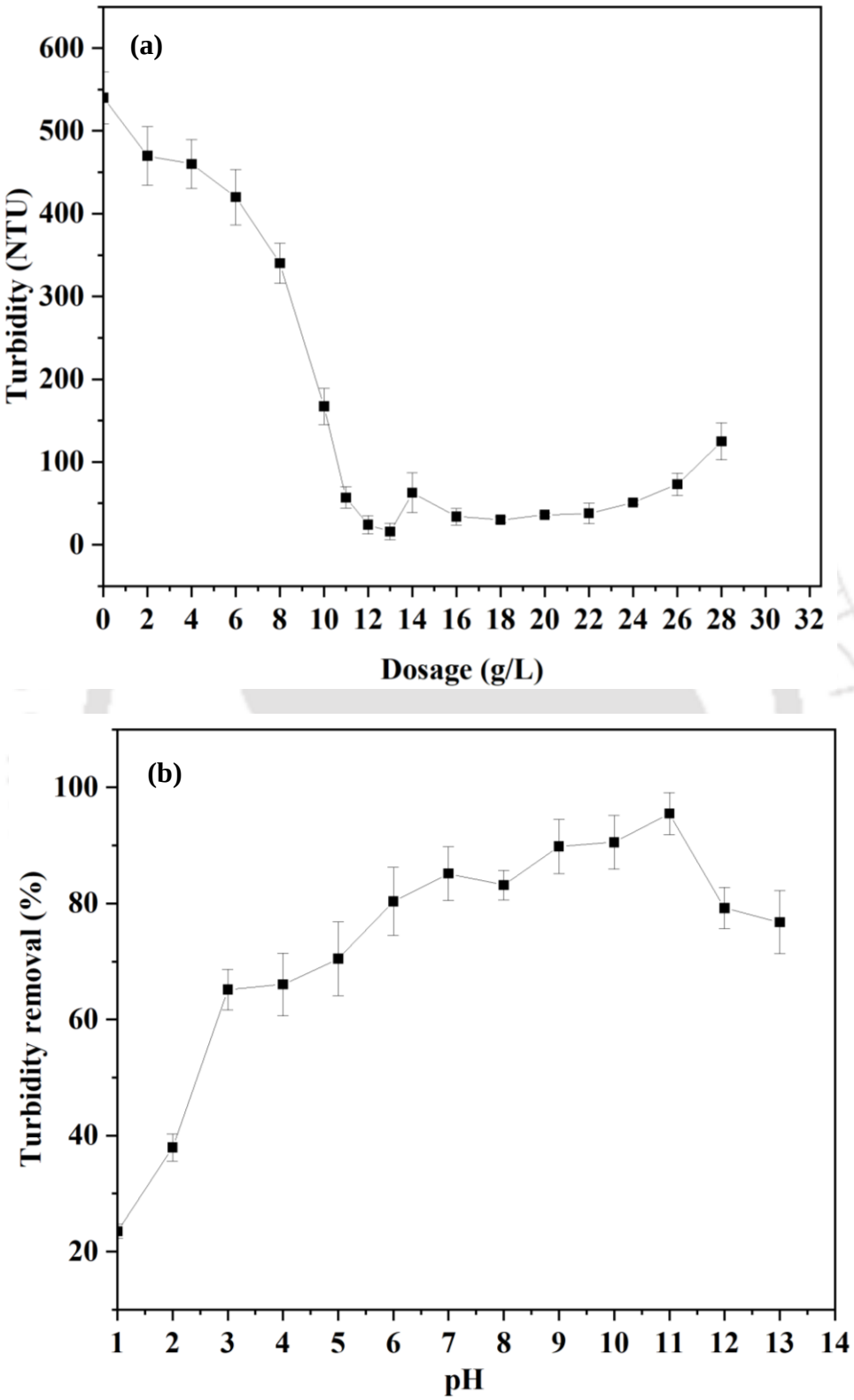
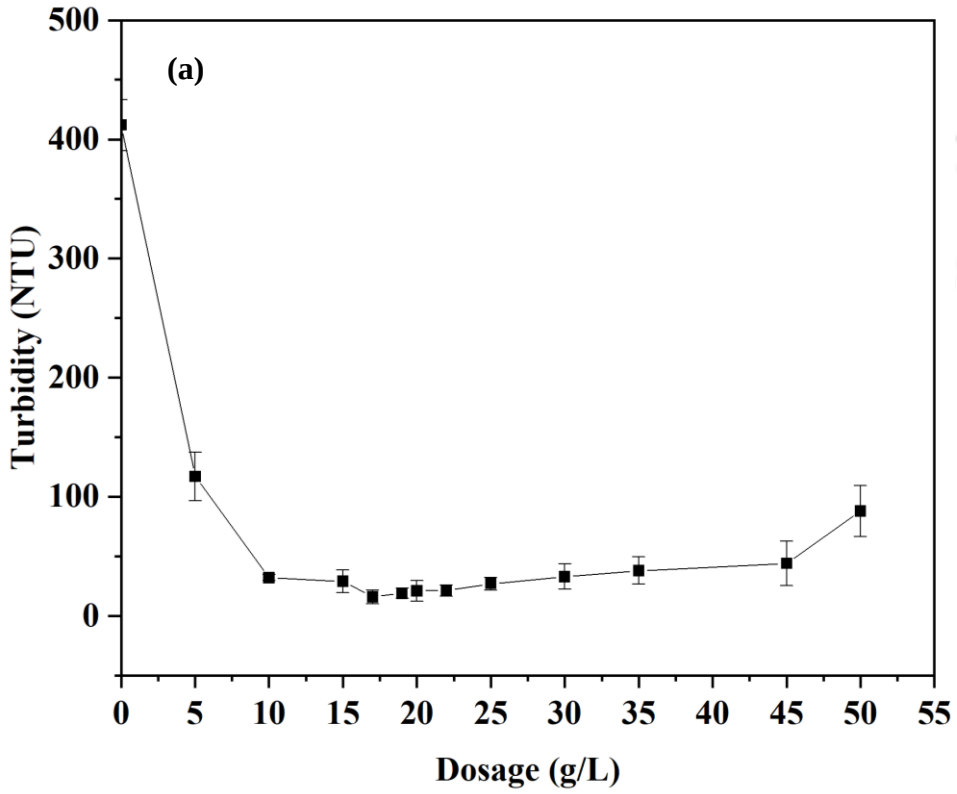


Fig. 7.3. (a) Effect of dosage on the turbidity (NTU) (b) Effect of pH range on turbidity removal (%) of leachate as a function of ferrous sulphate heptahydrate

7.1.2.4 Calcium hydroxide ($Ca(OH)_2$)

Experiments were subsequently conducted to obtain the maximum reduction in turbidity with the various coagulant dosages, as shown in Fig. 7.4. The sample's initial turbidity is 830 NTU, and a maximum reduction in turbidity was obtained with the final turbidity as 17 NTU. However, a maximum of 98% removal efficiency was achieved at 17 g/L of alum dosage. Experiments were performed at various pH levels by plotting the values between various pH levels and the percentage of reduction in turbidity, as shown in Fig. 7.4. The optimum pH was found to be 5 and 11, obtaining the maximum turbidity reduction. $Ca(OH)_2$ reacts with $Ca(HCO_3)_2$ in leachate and forms a white gelatinous $CaCO_3$, as shown in equation 7. 4.

$$Ca(HCO_3)_2 + Ca(OH)_2 \rightarrow 2CaCO_3 + 2H_2O \quad (7.4)$$



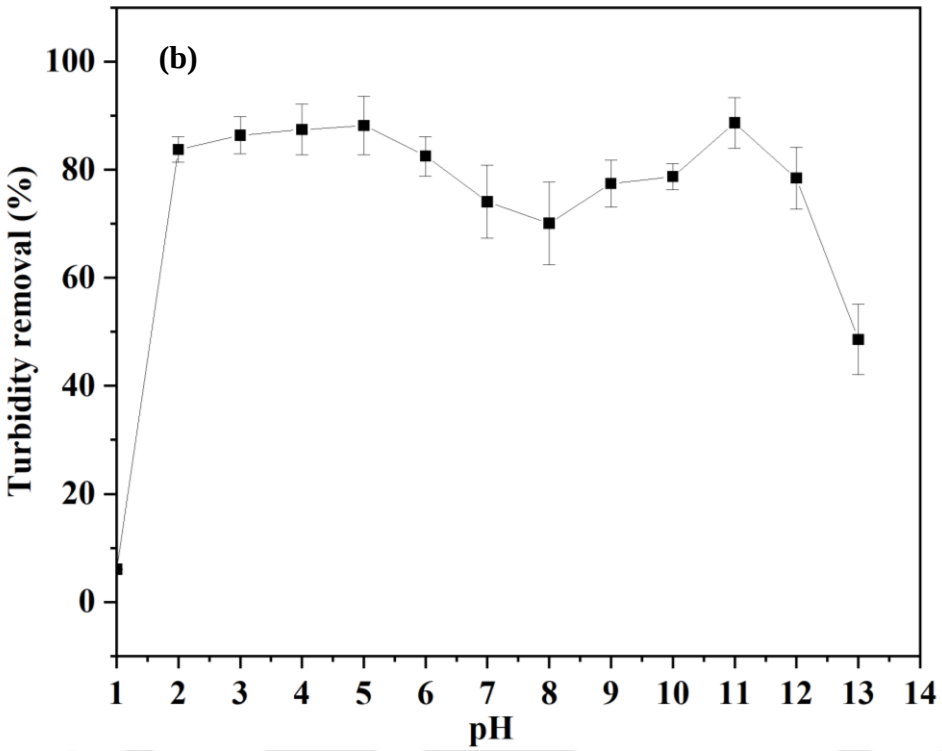


Fig. 7.4. (a) Effect of dosage on the turbidity (NTU) (b) Effect of pH range on turbidity removal (%) of leachate as a function of calcium hydroxide

7.1.2.5 Bentonite clay

Experiments were subsequently conducted to obtain the maximum reduction in turbidity with the various coagulant dosages, as shown in Fig. 7.5. The sample's initial turbidity is 874 NTU, and the maximum reduction in turbidity was 17 NTU was obtained. However, a maximum of 98% removal efficiency was achieved at 15 g/L of alum dosage. Experiments were performed at various pH levels by plotting the values between various pH levels and the percentage of reduction in turbidity, as shown in Fig. 7.5. The optimum pH was found to be 7, obtaining the maximum turbidity reduction.

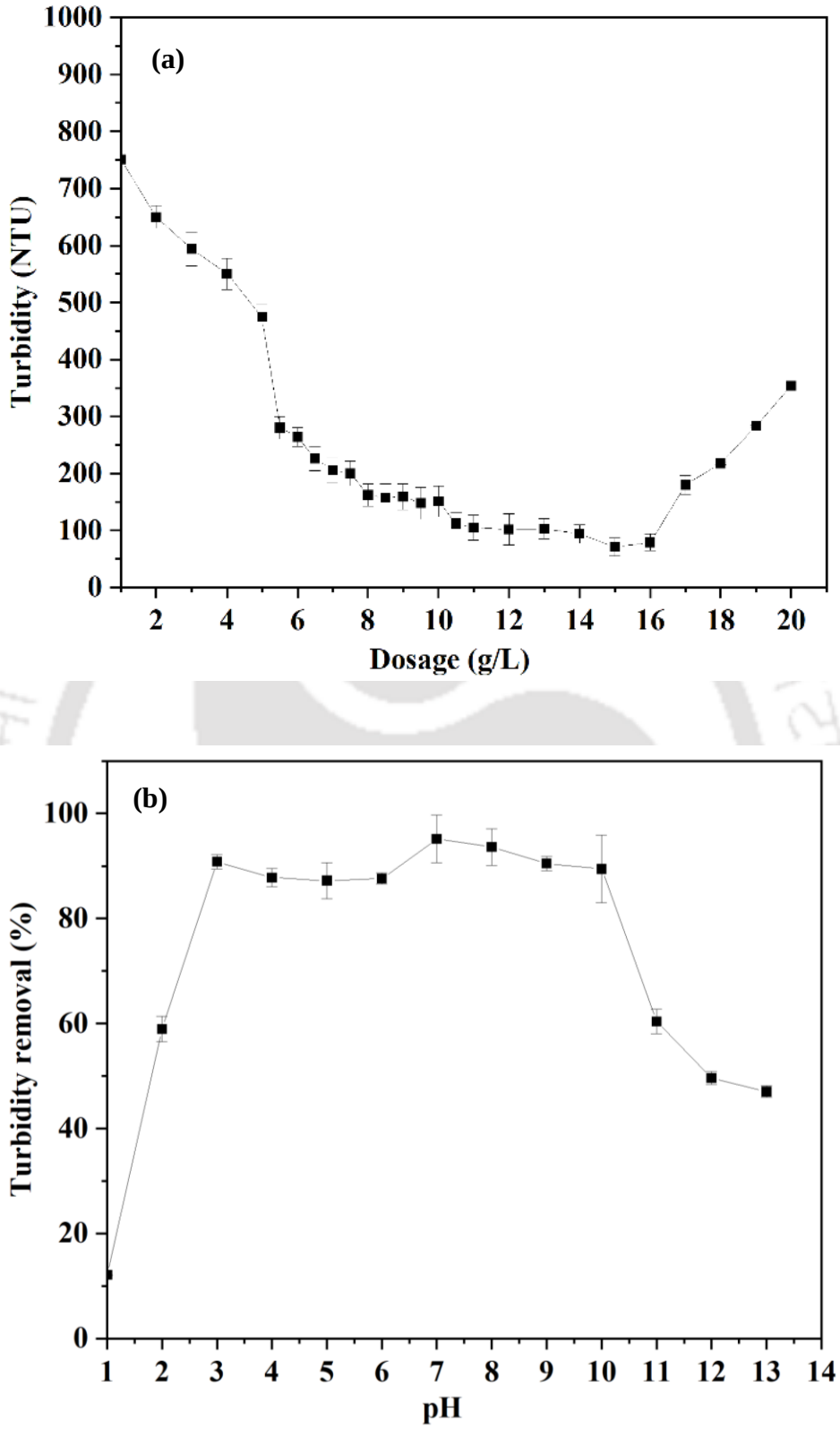


Fig. 7.5. (a) Effect of dosage on the turbidity (NTU) (b) Effect of pH range on turbidity removal (%) of leachate as a function of bentonite clay

7.1.2.6 Comparison of the characterization of leachate with various coagulants

The optimal coagulant dosage and pH were calculated in the analysis, and stored leachate is clariflocculated with a detention period of 2-2.5hrs (Committee et al., 1999). The percentage of COD and TOC removal from coagulation-flocculation was normally 10-25% for young leachates but is highest (50-65%) for low BOD_5/COD ratio leachates (stabilized leachates) (Amokrane et al., 1997). The obtained optimum dosage and pH were used to attain COD, BOD_5, and TSS removal efficiency. The fresh leachate concentrations of suspended solids were relatively high. The 94.1% reduction in TSS as colloidal particles have been destabilized due to the adsorption of heavily charged partially hydrolyzed metallic ions (Duan and Gregory, 2003). over 97% of TSS had been removed with $FeCl_3$ at pH 5, whereas other iron coagulants showed less removal than $FeCl_3$. Bentonite clay also had 96% removal efficiency at a pH of 7, whereas $Ca(OH)_2$ showed 92.47% removal at a pH of 11. Comparatively greater removal of suspended solids at lower and higher pH values was observed. (Aziz et al., 2007) reported that 95% of suspended solids have been testified at lower and higher pH values (4 and 12). Over 55.4% of COD removal was obtained at low coagulant dosage 3.25g/L and pH 5 than alum (49.1% and a dose of 4 g/L at pH 8, respectively). $Ca(OH)_2$ removed only 48.1% of COD and 27% of BOD_5 at pH 11 and dosage of 17g/L, which indicates a higher amount of dosage could perform poorly in removing all contaminants. Others have reported 48.1% of COD removal at 17g/L $Fe(SO)_4.7H_2O$ dose. The removal proportions of 36.2 and less than 30% of BOD_5 were attained using $Fe(SO)_4.7H_2O$ and $FeCl_3$. The maximum removal of COD 55.4% was observed at a dose of 3.25g/L at a pH of 5. As the coagulant dosages varied, leachate has adjusted to pH from 4 to 15 showed good results of suspended solids, COD removed reasonably at lower dosage and pH. Overall, $FeCl_3$ was found to be higher than other coagulants to remove all contaminants. At pH 8.5, the removal of contaminants was fairly low. The obtained outcomes are valid with other reports of $FeCl_3$, and potash alum was applied to wastewater and leachate (Kim et al., 2003; Aziz et al., 2007; Maranon et al., 2010; Verma and Kumar, 2016). The precipitation of soluble organics with the coagulant is usually due to COD via these coagulants (Tebbutt, 1997). They have reported that better performance was achieved at pH 5 for $FeCl_3$ and 5.5 with alum. Turbidity removal is less effective in the alkaline region than in the low pH region. (Amokrane et al., 1997). This is in agreement with the results obtained by other authors, and similar results were obtained in their experiments with leachate CFs (Stegmann and Ehrig, 1981; Ehrig, 1984). The

significant removal of organic colloids was obtained through different coagulants, and the coagulation-flocculation process effectively removes particulate COD in leachate. The study proves that the high amount of COD present in the leachate is particulates (colloid, residual matter) based on the COD removal by the CFs. 97% of SS removal was reported in the study when ferric chloride was used as a coagulant, and a further load of the pollutant was decreased to the biological treatment. Comparison between different coagulants and their removal efficiency among organic parameters were shown in Table 7.2.

Table 7.2 Summary of coagulation-flocculation process treatment results of all coagulants

Initial and final values of leachate parameters (Reduction in (%))								
Parameters	Initial values	$KAI(SO_4)_2 \cdot 12H_2O$ @ pH-8 4g/L	$Fe(Cl)_3$ @ pH-5 3.25g/L	$Fe(Cl)_3$ @ pH-8.5 3.25g/L	Bentonite Clay @ pH-7 15g/L	$Fe(SO)_4.7H_2O$ @ pH- 11 13g/L	$Ca(OH)_2$ @ pH-11 17g/L	$Ca(OH)_2$ @ pH-5 17g/L
TSS (mg/L)	6,110	360 (94.1)	160 (97.3)	380 (93.7)	240 (96)	400 (93.4)	460 (92.47)	440 (92.7)
COD (mg/L)	4,7861	24,840 (49.1)	21,320 (55.4)	27,040 (43.5)	21,600 (54)	25,200 (47.3)	24,800 (48.1)	22,800 (52.3)
BOD_5 (mg/L)	22,058	14,430 (34.5)	14,070 (36.21)	17,085 (22.5)	15,000 (31)	14,670 (33.4)	15,945 (27.7)	15,540 (29.5)

7.2 UPFLOW ANAEROBIC SYSTEM STARTUP AND ACCLIMATIZATION

The total startup time was 36 weeks at a room temperature of 15-40^{0}C. The upflow anaerobic filter setup was a single reactor attached biofilm system. Plastic pall rings were used as a surface for the growth of biofilm were chosen as they have a specific area and large porosity to attach a large number of microorganisms. As a result, it allows high-strength leachate treatment with a high concentration load with significant efficiency in production. Above a minimum of about 2m, reactor height seems to have little effect on performance (Song and Young, 1986). This study's reactor has a height of 6 feet (1.897 meters) designed to investigate the filter's performance. The relative height of the medium within the reactor is a more significant parameter. Too little media appears to allow the excessive loss of solids and reduce COD removal. (Young and Yang, 1989). Plastic media

is more efficient than other media in the clarification of wastewater, allowing good microbial growth due to high specific surface area and low molecular weight.

7.2.1 pH variation and COD reduction during the acclimatization process of upflow anaerobic filter packed with plastic media

After pretreatment, the initial pH of high-strength leachate was 5.1, highly acidic, the leachate COD is 21,320 mg/L, and the BOD is 14,070 mg/L. The BOD/COD ratio had 0.64. After pretreatment, leachate's high strength has been given as continuous feed to the anaerobic filter for 60 days in a conventional manner. The COD reduction and pH are shown in Fig. 7.6. The filter performance packed with plastic media is 23300 mg/L and operating HRT at 5 days when treating influent COD. However, the COD reduction from its initial value of 23,300 to 17,856 mg/L was observed at an HRT of 5 days. The overall efficiency for COD removal was 23.3%. Only 23.3% COD removal was obtained at a high loading rate of 5 kg COD/m^3 d. A related study indicated that 27.7% was achieved with an OLR of 6 kg COD/m^3 d. (Henry and Prasad, 2000). Throughout the startup phase, pH was not controlled. Influent and effluent pH variations were shown in fig.8 during the startup period.

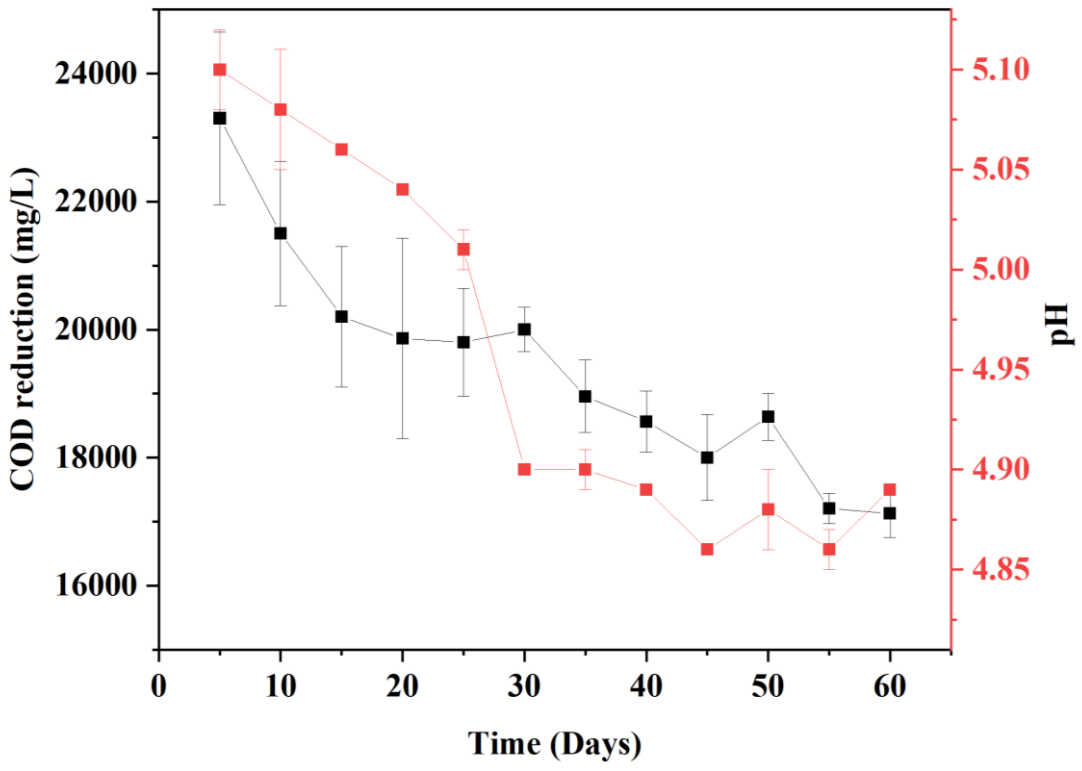


Fig.7.6. pH variation and COD reduction during the acclimatization process of anaerobic filter

The results of pH effluent changes from 5.1 to 4.90 for studied OLR, indicating an accumulation of volatile fatty acids. The study results show that anaerobic filters are capable of treating acidic waste with a pH of 5. In agreement with another report, leachate's anaerobic treatment with low pH was performed, but raw leachate was diluted with water (Wu et al., 1988). The specific surface area, random packing of plastic media, provided better performance even though filter performance was not high, which could be attributed to the high leachate concentration and low pH. A long retention time of 5 days was used to allow the biomass to populate and attach to the medium while preventing biomass washout.

7.2.2 Pretreatment process- determination of optimum condition for L3 landfill leachate using $FeCl_3$

Optimum dosage was first analyzed by measuring the maximum reduction in turbidity concerning the various coagulant dosages. The initial turbidity of the sample was1,150 NTU, as shown in Table 7.3. The maximum reduced turbidity was 140 NTU. However, a maximum of 88% removal efficiency was achieved at 0.5 g/L of $FeCl_3$ dosage. Optimum pH was determined by plotting the values between various pH and percentage of reduction in turbidity, and at pH 8.2, the maximum percentage of reduction was obtained, as shown in Fig.7.7.

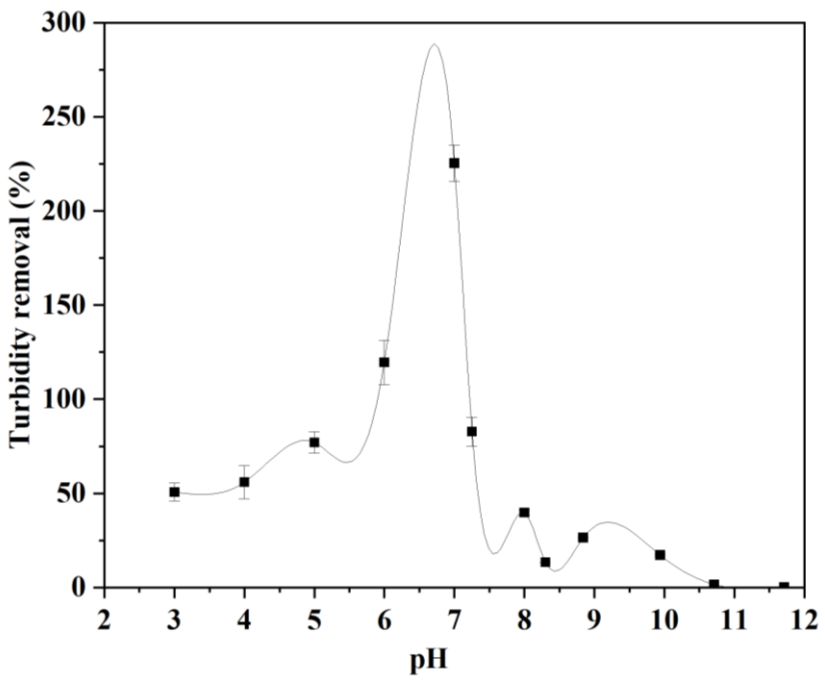


Fig. 7.7. Effect of pH range on turbidity removal (%) of leachate as a function of $FeCl_3$

Pretreatment has been given to stored L3 leachate, simulated with volumetric precipitation. $FeCl_3$ was selected as the better coagulant to treat landfill leachate based on section 7.12 results to obtain the optimum dosage and pH. The initial and final treatment of landfill leachate, as shown in Table 7.3.

Table 7.3. The removal efficiency of targeted landfill leachate parameters (L3) by coagulation process

S.No.	Parameter	Before	After	Percent removal (%)
1	pH	5.15	8.2	-
2	TSS	7,371	1,105	85
3	COD	47,820	31,000	35
4	Turbidity	1,150	140	88

7.2.3 Treatment phase of high strength pretreated leachate

The high-strength leachate from reactor 3 has a wet (73%) and dry (27%) waste composition to which simulated rainfall was added. The pretreatment has given the leachate, obtained an optimum dosage of 50 g/L, and at pH 8.2. The initial COD of L3 ranged from 44,000 mg/L to 48,000 mg/L. Optimum dosage and pH were used to treat leachate then fed to the anaerobic filter packed with plastic media. The anaerobic filter effluent characteristics with different hydraulic loading rates (HRT), as shown in Table 7.4. Different HRTs have been examined to check the performance of the anaerobic filter packed with plastic media. The initial pH was 6.59, and the influent COD was 32,651 mg/L and decreased to 28,632 mg/L. The efficiency was 12.31% and gradually increased to 20.03% as pH increased to 8.02. It indicates that pH and HRT are also significant factors in anaerobic filter performance. At 5 days HRT, the average influent COD concentrations were 32651 to 28975 mg/L, and effluent COD concentrations ranged from 28632 -23170 mg/L. Removal efficiencies were ranged from 12.31 to 20.03%. At 4 days, HRT COD concentrations were ranged from 30765 mg/L to 29,100, and removal efficiency was 13.7 to 15.46%. Therefore, a 20% reduction in COD removal efficiency was observed at 5 days

HRT compared with 4 days HRT, and only a 15.46% reduction in COD removal was observed.

Table 7.4. Upflow anaerobic filter effluent characteristics in the treatment process and their effects on pH and HRT

HRT (days)	Initial pH	Final pH	Average influent COD (mg/L) ± Stdev	Average final effluent COD (mg/L) ± Stdev	Average COD removal (%)
5	6.59	6.32	32651 ± 134.38	28632 ± 263.53	12.31
5	7.02	6.36	31848 ± 198.53	26380 ±159.43	17.17
5	8.02	6.62	28975 ± 235.45	23170 ± 269.64	20.03
4	8.00	6.81	30765 ± 358.34	26550 ± 344.63	13.7
4	8.04	6.72	29100 ± 25.01	24600 ±134.63	15.46

7.3 METAGENOMIC STUDY

The landfill leachate microbiome plays a crucial role in organic degradation, methane production, and nitrogen cycling. In this study, to identify large-scale genomes of the landfill leachate microbiota by the metagenome technique. The effect of a coagulation-flocculation study of a microbial community in landfill leachate was the hypothesis. Two samples were collected from the simulated landfill reactor and pretreated landfill leachate to be analysed microbiologically. Sample 1 (S1) corresponds to raw leachate, and Sample 2 (S2) corresponds to leachate incoming from the biological treatment after coagulation treatment. The efficiency of the biological removal of carbon from leachate was determined by the type of microbial populations present in biological reactors. By comparing the results of raw and anaerobic filter treatment of landfill leachate samples against the microbial genomes with their possible functions in the landfill ecosystem.

7.3.1 Microbial diversity analysis in landfill leachate

7.3.1.1 Comparative analysis between the samples at the phylum level

16S rRNA sequencing high-throughput sequencing revealed the diversity of the microbial community in two different samples at the phylum level. The microbial

community's diversity and abundance in stored raw leachate and anaerobic filter effluent leachate are shown in Fig. 7.8 & Table 7.5. In raw leachate, the phyla Euryarchaeota (45%), Bacteroidetes (21.7%), Firmicutes (12.5%), Spirochaetes (2.3%), Actinobacteria (1.4) were dominant in the landfill, while in anaerobic filter effluent leachate, the dominant phyla were Proteobacteria (50.2%) and Firmicutes (41%).

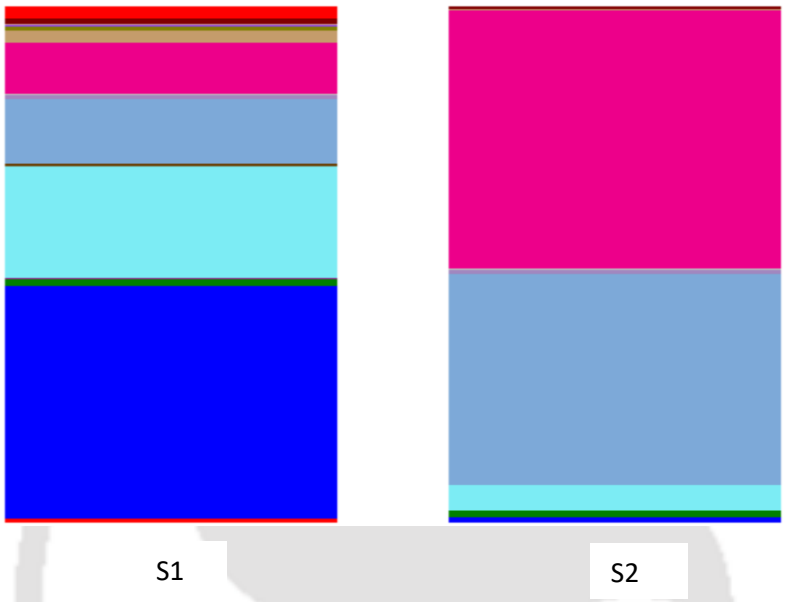


Fig. 7.8. Stacked bar chart showing the relative abundance of each phylum within each sample

Table 7.5. Relative abundance of each phylum within each sample

Legend	Taxonomy	S1	S2
	k__Archaea;p__Euryarchaeota	45	1.1
	k__Bacteria;p__Bacteroidetes	21.7	4.8
	k__Bacteria;p__Firmicutes	12.5	41
	k__Bacteria;p__Proteobacteria	9.9	50.2
	k__Bacteria;p__Spirochaetes	2.3	0.2
	k__Bacteria;p__WWE1	2.2	0
	k__Bacteria;p__Actinobacteria	1.4	1.3
	k__Bacteria;p__Synergistetes	1	0.1
	k__Bacteria;p__Verrucomicrobia	1	0.4

	k__Archaea;p__Crenarchaeota	0.8	0
	k__Bacteria;p__Lentisphaerae	0.7	0.7
	k__Bacteria;p__Chloroflexi	0.5	0
	k__Bacteria;p__Tenericutes	0.5	0

7.3.1.2 Comparative analysis between the samples at class level

The abundance of microbial diversity at the class level is shown in Fig. 7.9 & Table 7.6. in raw leachate, Euryarchaeota_Methanomicrobia (27.4%), Euryarchaeota_Methanobacteria (16.8%), Bacteroidetes_Bacteroidia (21%), Firmicutes_Clostridia (11%), Bacteria__Proteobacteria__Gammaproteobacteria (4.6%), while in anaerobic filter effluent leachate, Bacteria__Firmicutes;c__Clostridia (11.6%), proteobacteria_Gammaproteobacteria (14.5), Firmicutes__Bacilli (28.9), Proteobacteria;c__Alphaproteobacteria (35.1%) were dominant class in landfill leachate.

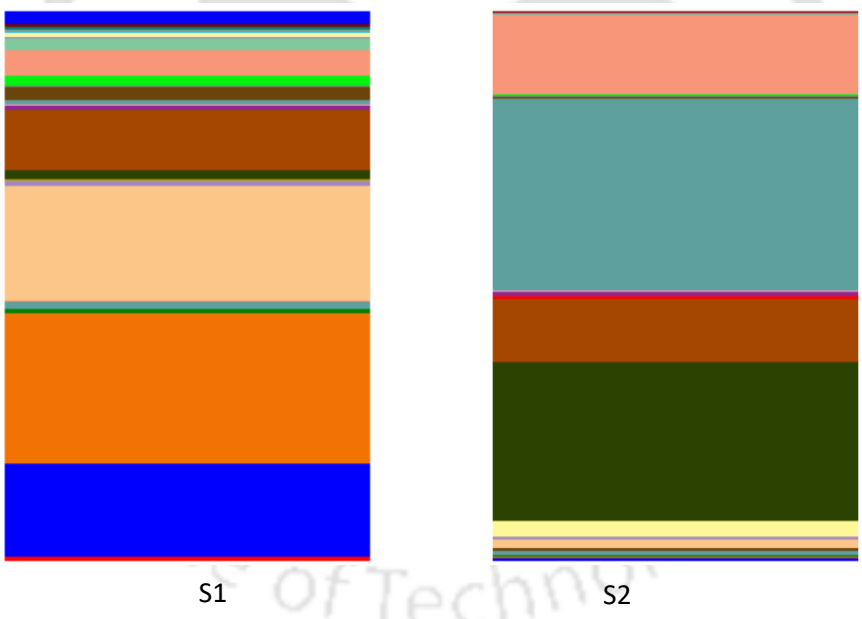


Fig. 7.9. Stacked bar chart showing the relative abundance of each class within each sample

Firmicutes are the most dominant class in landfills (Krishnamurthi and Chakrabarti, 2013; Van Dyke and McCarthy, 2002). Firmicutes are cellulose-degrading bacteria and are believed to play an important role in the anaerobic and methanogenic phases of refuse decomposition in landfills (Song et al., 2015). Bacteroidetes, Firmicutes, and Proteobacteria. The three phyla were also found to be abundant in landfills or landfill

leachate in previous studies, irrespective of the geographical location (Köchling et al., 2015; Wang et al., 2017).

Table 7.6. Relative abundance of each class within each sample

Legend	Taxonomy	S1	S2
	k__Archaea;p__Euryarchaeota;c__Methanomicrobia	27.4	0.3
	k__Bacteria;p__Bacteroidetes;c__Bacteroidia	21	1.5
	k__Archaea;p__Euryarchaeota;c__Methanobacteria	16.8	0.6
	k__Bacteria;p__Firmicutes;c__Clostridia	11	11.6
	k__Bacteria;p__Proteobacteria;c__Gammaproteobacteria	4.6	14.5
	k__Bacteria;p__Proteobacteria;c__Betaproteobacteria	2.4	0.3
	k__Bacteria;p__WWE1;c__[Cloacamonae]	2.2	0
	k__Bacteria;p__Spirochaetes;c__Spirochaetes	2.1	0.2
	k__Bacteria;p__Proteobacteria;c__Epsilonproteobacteria	1.7	0.3
	k__Bacteria;p__Firmicutes;c__Bacilli	1.3	28.9
	k__Bacteria;p__Actinobacteria;c__Actinobacteria	1.2	0.6
	k__Bacteria;p__Synergistetes;c__Synergistia	1	0.1
	k__Archaea;p__Crenarchaeota;c__MCG	0.8	0
	k__Archaea;p__Euryarchaeota;c__Thermoplasmata	0.8	0.3
	k__Bacteria;p__Proteobacteria;c__Alphaproteobacteria	0.8	35.1
	k__Bacteria;p__Lentisphaerae;c__[Lentisphaeria]	0.7	0.7
	k__Bacteria;p__Bacteroidetes;c__Flavobacteriia	0.6	0.5
	k__Bacteria;p__Verrucomicrobia;c__[Pedosphaerae]	0.6	0.3
	k__Bacteria;p__Firmicutes;c__Erysipelotrichi	0.2	0.5
	k__Bacteria;p__Actinobacteria;c__Coriobacteriia	0.1	0.7
	k__Bacteria;p__Bacteroidetes;c__Sphingobacteriia	0	2.9

7.3.1.3 Comparative analysis between the samples at order level

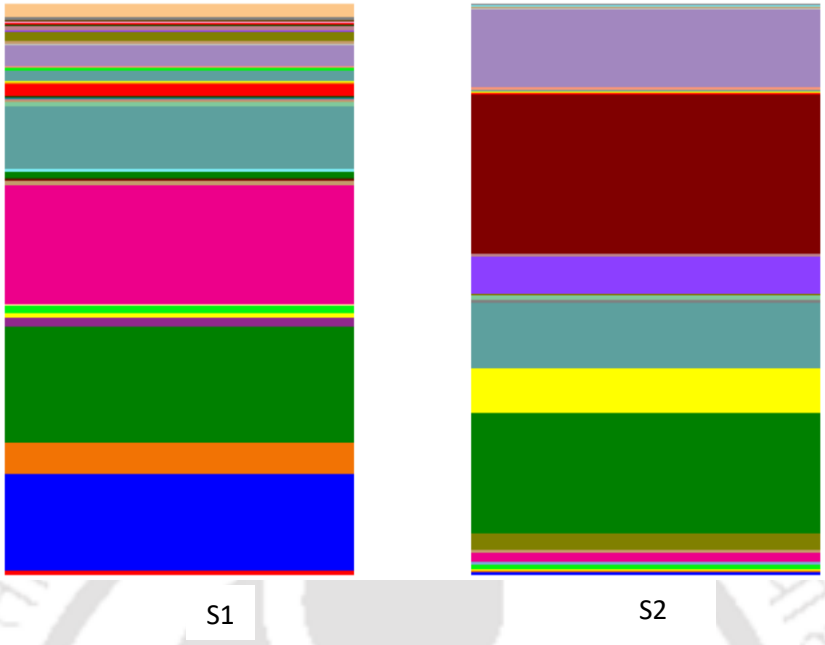


Fig. 7.10. Stacked bar chart showing the relative abundance of each order within each sample

Table 7.7. Relative abundance of each order within each sample

Legend	Taxonomy	Raw leachate (%)	Anaerobic filter effluent leachate (%)
	k__Bacteria;p__Bacteroidetes;c__Bacteroidia;o__Bacteroidales	21	1.5
	k__Archaea;p__Euryarchaeota;c__Methanomicrobia;o__Methanosarc i nales	20.2	0.1
	k__Archaea;p__Euryarchaeota;c__Methanobacteria;o__Methanobacte riales	16.8	0.6
	k__Bacteria;p__Firmicutes;c__Clostridia;o__Clostridiales	11	11.6
	k__Archaea;p__Euryarchaeota;c__Methanomicrobia;o__Methanomi crobiales	5.5	0.2
	k__Bacteria;p__Proteobacteria;c__Gammaproteobacteria;o__Pseudom onadales	3.8	13.4
	k__Bacteria;p__WWE1;c__[Cloacamonae];o__[Cloacamonales]	2.2	0

	k__Bacteria;p__Proteobacteria;c__Betaproteobacteria;o__Burkholderi ales	2	0.3
	k__Bacteria;p__Proteobacteria;c__Epsilonproteobacteria;o__Campylobacterales	1.7	0.3
	k__Bacteria;p__Spirochaetes;c__Spirochaetes;o__Spirochaetales	1.7	0
	k__Archaea;p__Euryarchaeota;c__Methanomicrobia;o__YC-E6	1.6	0
	k__Bacteria;p__Actinobacteria;c__Actinobacteria;o__Actinomycetale	1.2	0.6
	k__Bacteria;p__Synergistetes;c__Synergistia;o__Synergistales	1	0.1
	k__Bacteria;p__Firmicutes;c__Bacilli;o__Bacillales	0.9	21.1
	k__Archaea;p__Crenarchaeota;c__MCG;o__pGrfC26	0.8	0
	k__Archaea;p__Euryarchaeota;c__Thermoplasmata;o__E2	0.8	0.3

7.3.1.4 Comparative analysis between the samples at species level

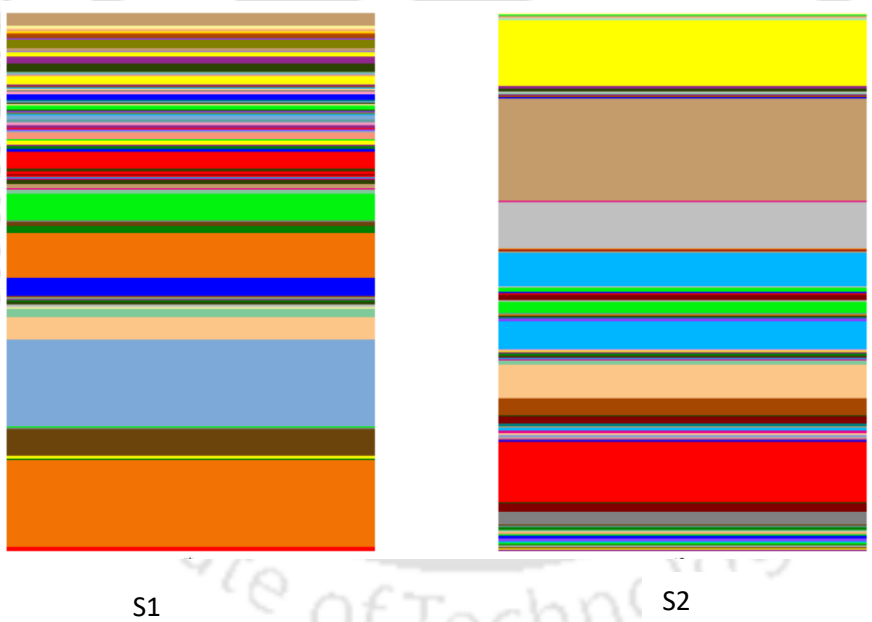


Fig. 7.11. Stacked bar chart showing the relative abundance of each species within each sample

Table 7.8. Relative abundance of each species within each sample (part-1)

Legend	Taxonomy	S1	S2
	k__Archaea;p__Euryarchaeota;c__Methanobacteria;o__Methanobacteriales;f__Methanobacteriaceae;g__Methanobacterium;s__	16.2	0.1
	k__Archaea;p__Euryarchaeota;c__Methanomicrobia;o__Methanosarcinales;f__Methanosarcinaceae;g__Methanosarcina;s__	16.2	0
	k__Bacteria;p__Bacteroidetes;c__Bacteroidia;o__Bacteroidales;f__BA008;g__;s__	8.3	0
	k__Archaea;p__Euryarchaeota;c__Methanomicrobia;o__Methanomicrobiales;f_Methanocorpusculaceae;g__Methanocorpusculum;s__	5	0.2
	k__Bacteria;p__Bacteroidetes;c__Bacteroidia;o__Bacteroidales;f__Porphyromonadaceae;g__;s__	5	0.1
	k__Archaea;p__Euryarchaeota;c__Methanomicrobia;o__Methanosarcinales;f__Methanosarcinaceae;g__Methanosarcina;s__mazei	4	0
	k__Bacteria;p__Bacteroidetes;c__Bacteroidia;o__Bacteroidales;f__;g__;s__	3.5	0.4
	k__Bacteria;p__Firmicutes;c__Clostridia;o__Clostridiales;f__;g__;s__	3.2	0.1
	k__Bacteria;p__WWE1;c__[Cloacamonae];o__[Cloacamonales];f__[Cloacamonaceae];g__;s__	2.2	0
	k__Bacteria;p__Spirochaetes;c__Spirochaetes;o__Spirochaetales;f__Spirochaetaceae;g__Treponema;s__	1.7	0
	k__Archaea;p__Euryarchaeota;c__Methanomicrobia;o__YC-E6;f__;g__;s__	1.6	0

Table 7.9. Relative abundance of each species within each sample (part-2)

Legend	Taxonomy	S1	S2
	k__Bacteria;p__Proteobacteria;c__Gammaproteobacteria;o__Pseudomonadales;f__Moraxellaceae;g__Acinetobacter;s__	1.5	0.4
	k__Bacteria;p__Proteobacteria;c__Epsilonproteobacteria;o__Campylobacterales;f__Campylobacteraceae;g__Arcobacter;s__cryaerophilus	1.4	0
	k__Bacteria;p__Firmicutes;c__Clostridia;o__Clostridiales;f__Clostridiaceae;g__SMB53;s__	1.3	0
	k__Bacteria;p__Proteobacteria;c__Gammaproteobacteria;o__Pseudomonadales;f__Pseudomonadaceae;g__;s__	1.3	0.4
	k__Bacteria;p__Bacteroidetes;c__Bacteroidia;o__Bacteroidales;f__Bacteroidaceae;g__;s__	1.2	0
	k__Bacteria;p__Proteobacteria;c__Betaproteobacteria;o__Burkholderiales;f__Alcaligenaceae;g__;s__	1.1	0.1
	k__Bacteria;p__Bacteroidetes;c__Bacteroidia;o__Bacteroidales;f__SB-1;g__;s__	0.9	0
	k__Bacteria;p__Firmicutes;c__Clostridia;o__Clostridiales;f__Syntrophomonadaceae;g__Syntrophomonas;s__	0.9	0.1
	k__Bacteria;p__Synergistetes;c__Synergistia;o__Synergistales;f__Synergistaceae;g__vadinCA02;s__	0.9	0.1
	k__Archaea;p__Crenarchaeota;c__MCG;o__pGrfC26;f__;g__;s__	0.8	0
	k__Bacteria;p__Bacteroidetes;c__Bacteroidia;o__Bacteroidales;f__Marinilabiaceae;g__;s__	0.8	0
	k__Bacteria;p__Firmicutes;c__Clostridia;o__Clostridiales;f__Clostridiaceae;g__Clostridium;s__	0.8	0.2
	k__Bacteria;p__Bacteroidetes;c__Bacteroidia;o__Bacteroidales;f__Prevotellaceae;g__Prevotella;s__	0.7	0.3
	k__Bacteria;p__Lentisphaerae;c__[Lentisphaeria];o__Z20;f__R4-45B;g__;s__	0.7	0.7
	k__Bacteria;p__Proteobacteria;c__Gammaproteobacteria;o__Pseudomonadales;f__Pseudomonadaceae;g__Pseudomonas;s__	0.7	12.4

	k__Bacteria;p__Bacteroidetes;c__Flavobacteriia;o__Flavobacteriales;f__Flavobacteriaceae;g__Flavobacterium;s__	0.6	0.1
	k__Bacteria;p__Firmicutes;c__Clostridia;o__Clostridiales;f__Christensenellaceae;g__;s__	0.6	0
	k__Bacteria;p__Verrucomicrobia;c__[Pedosphaerae];o__[Pedosphaerales];f__R4-41B;g__;s__	0.6	0.3
	k__Bacteria;p__Actinobacteria;c__Actinobacteria;o__Actinomycetales;f__Corynebacteriaceae;g__Corynebacterium;s__	0.5	0
	k__Bacteria;p__Firmicutes;c__Bacilli;o__Bacillales;f__Planococcaceae;g__Solibacillus;s__	0.5	0
	k__Bacteria;p__Firmicutes;c__Clostridia;o__Clostridiales;f__Clostridiaceae;g__;s__	0.5	0.4
	k__Bacteria;p__Firmicutes;c__Clostridia;o__Clostridiales;f__Peptococcaceae;g__Pelotomaculum;s__	0.5	0
	k__Bacteria;p__Firmicutes;c__Clostridia;o__Clostridiales;f__Ruminococcaceae;g__;s__	0.5	0.3
	k__Bacteria;p__Proteobacteria;c__Betaproteobacteria;o__Burkholderiales;f__Comamonadaceae;g__;s__	0.5	0
	k__Bacteria;p__Firmicutes;c__Bacilli;o__Bacillales;f__Bacillaceae;g__Bacillus;	0.2	11.3

Table 7.10. Relative abundance of each species within each sample (part-3)

Legend	Taxonomy	S1	S2
	s__k__Bacteria;p__Firmicutes;c__Clostridia;o__Clostridiales;f__[Tissierellaceae];g__Sedimentibacter;s__	0.2	1
	k__Bacteria;p__Actinobacteria;c__Coriobacteriia;o__Coriobacteriales;f__Coriobacteriaceae;g__;s__	0.1	0.7
	k__Bacteria;p__Firmicutes;c__Bacilli;o__Bacillales;f__Planococcaceae;g__;s__	0.1	0.6

k__Bacteria;p__Firmicutes;c__Bacilli;o__Bacillales;f__Planococcaceae;g__Lysinibacillus;s__	0.1	1.1
k__Bacteria;p__Firmicutes;c__Clostridia;o__Clostridiales;f__Lachnospiraceae;g__;s__	0.1	5.1
k__Bacteria;p__Bacteroidetes;c__Sphingobacteriia;o__Sphingobacteriales;f__Sphingobacteriaceae;g__Sphingobacterium;s__multivorum	0	2.3
k__Bacteria;p__Firmicutes;c__Bacilli;o__Bacillales;f__;g__;s__	0	1.7
k__Bacteria;p__Firmicutes;c__Bacilli;o__Bacillales;f__Paenibacillaceae;g__Brevibacillus;s__reuszeri	0	0.7
k__Bacteria;p__Firmicutes;c__Bacilli;o__Bacillales;f__Planococcaceae;g__Rummeliibacillus;s__	0	3.1
k__Bacteria;p__Firmicutes;c__Bacilli;o__Lactobacillales;f__Lactobacillaceae;g__Lactobacillus;s__	0	6.2
k__Bacteria;p__Firmicutes;c__Bacilli;o__Lactobacillales;f__Lactobacillaceae;g__Lactobacillus;s__acidipiscis	0	0.8
k__Bacteria;p__Firmicutes;c__Clostridia;o__Clostridiales;f__Lachnospiraceae;g_Butyrivibrio;s__	0	0.6
k__Bacteria;p__Firmicutes;c__Clostridia;o__Clostridiales;f__Ruminococcaceae;g__Ruminococcus;s__	0	1.9
k__Bacteria;p__Proteobacteria;c__Alphaproteobacteria;o__Caulobacterales;f__Caulobacteraceae;g__Brevundimonas;s__diminuta	0	6.3
k__Bacteria;p__Proteobacteria;c__Alphaproteobacteria;o__Rhodospirillales;f__Acetobacteraceae;g__;s__	0	8.6
k__Bacteria;p__Proteobacteria;c__Alphaproteobacteria;o__Rhodospirillales;f__Acetobacteraceae;g__Gluconacetobacter;s__rhaeticus	0	19.1

Table 7.11. Most abundant taxonomy identified in the samples at different taxonomic levels

Taxonomy/ Samples	Raw leachate (%)	Anaerobic filter effluent leachate (%)
Phylum	Euryarchaeota (44.9)	Proteobacteria (50.16)
Class	Methanomicrobia (27.36)	Alphaproteobacteria (35.09)
Order	Bacteroidales (21)	Rhodospirillales (27.91)
Family	Methanosarcinaceae (20.19)	Acetobacteraceae (27.91)
Genus	*Methanosarcina (20.19)*	*Gluconacetobacter (19.20)*
Species	*Unclassified Species from Methanobacteriu m Genus (16.22)*	*Rhaeticus (19.11)*

7.3.1.5 Bacterial community composition

The Microbial communities in raw leachates of different ages reported that Proteobacteria, mainly Pseudomonadales order, were more abundant in rainy seasons, whereas Firmicutes were related to the secretion of extracellular enzymes as cellulases, lipases, and proteases. So, their main function in landfills consists in degrading complex polysaccharides, such as starch and cellulose (Kochling et al., 2015). The Bacteroidetes phylum accounted for 21% of total microorganisms in S1. Microorganisms within this phylum have been described as expert bacteria for the degradation of high molecular weight organic matter to acetic and propionic acid. Microorganisms belonging to Firmicutes, Bacteroidetes, and Firmicutes hydrolysis-acidification bacteria commonly found in anaerobic environments can degrade many macromolecules in landfills, such as carbohydrates, proteins, and lipids (Semrau, 2011). Gammaproteobacteria, Alphaproteobacteria, and Betaproteobacteria were detected in two samples, as shown in Table 7.9. As physiologically diverse bacteria, Proteobacteria were more likely to be enriched in the leachate because they may be involved in the fermentation of soluble substrates (Bareither et al., 2013). Actinobacteria are believed to play a major role in organic matter degradation and play a vital part in carbon turnover (Newton et al., 2011).

In this study, the abundance of Actinobacteria was 1.2% in raw leachate, and 0.6 in anaerobic filter effluent leachate in species level is shown in Table 7.9. Pseudomonas has been widely detected in various environments, especially in organic matter degradation and denitrification. Pseudomonas are well-known pollutant degrading bacteria that utilize a wide range of polycyclic aromatic hydrocarbons as their sole carbon source (Loick et al., 2009). In order level, raw leachate has 3.8% of Pseudomonas, while in the anaerobic filter, effluent leachate has 13.4%, as shown in Table 7.7. Bacillus is frequently found in landfills, degrading cellulose (Westlake et al., 1995) and oxidizing polycyclic aromatic hydrocarbons (Zeng et al., 2016) and chromium. In order level, raw leachate has 0.9% of Bacillus, while in the anaerobic filter, effluent has 21.1%, as shown in Table 7.7. Both universal and archaea-specific primer pairs, the archaeal community demonstrated that Euryarchaeota was the most dominant archaeal phylum in all landfill leachate samples. In previous studies, Euryarchaeota has been consistently identified as the predominant phylum in municipal landfills and other anaerobic environments (Zainun and Simarani, 2018). Methanomicrobia (27.36%) at class level, Methanosarcinaceae (20.19%) at the family level, *Methanosarcina* (20.19%) at the genus level are found in the raw leachate is illustrated in Table 7.11. These three types of methanogens account for methane production in anaerobic systems and have been previously reported in landfills (Chen et al., 2003).

7.4 EFFECT OF INOCULUM ADDITION TO LANDFILL LEACHATE

7.4.1 Initial characterization of landfill leachate and digested cow dung (DCD)

Table 7.1 summarises the Initial characterization of landfill leachate and digested cow dung (DCD). The initial average COD was 4160 mg/L in DCD while in landfill leachate, 26800 mg/L. Thus, the initial nearly neutral pH of inoculum and landfill leachate supports the degradation of organic matter in landfill leachate with DCD because lower degradable organics are present in DCD.

Table 7.12 characterization of the landfill leachate and inoculum

S.No	Parameters	Digested Cow dung (Mean ± Stdev)	Landfill leachate (Mean ± Stdev)
1.	pH	7.76 ± 0.05	6.15 ± 0.2
2.	VS (%TS)	35.62 ± 1.02	48.7 ± 2.13

3.	COD (mg/L)	4160 ± 210	26800 ± 1354

7.4.2 Biomethane potential setup

To evaluate the feasibility of DCD with landfill leachate. Five different ratios varying from 10% to 80% of DCD were tested for 90 days. Inoculum plays a key role in initiating the fermentation by balancing the microbial population of methanogens. The inoculum also sustains the syntrophic metabolism for thermodynamics of the feasibility of the reactor. Initially, the hydrolytic bacteria break down the complex organics into a more soluble form by releasing extracellular enzymes by the process called hydrolysis. This happens to be the rate-limiting step in the overall digestion process. The hydrolysis product acts as a substrate for acidogenic bacteria. Subsequently, the intermediate products are transformed into simpler compounds such as acetate, carbon dioxide, hydrogen, and formate by a secondary fermentation with a final decrease in the pH of the system. The production of acid can be visualized by increased VFA production. Finally, methanogenic archaea, which are highly specific to these substrates, converts them to biogas. The breakdown or degradation of dissolved organics leads to an increase in the COD.

7.4.3 Biomethane production

The fundamental goal of the BMP study is to break down complex organic compound complexes into smaller components, which are then transformed into biomethane. Fig. 7.12 & 7.13 represents daily methane production and cumulative methane (mL). In the early days, gas production data was very uncertain; it was difficult to predict which ratio produces higher biogas. In the current experiment, it was observed that biogas production was initially the same with all the F/M ratios. It can, therefore, be perceived that the initial methane yield may be due to readily available organics present within the inoculums. The maximum cumulative methane achieved for 10% of CDC added to landfill leachate in a 1-liter aspirator bottle was found to be 1792.8 ml, 20% (5500.55 mL), 40% (8933.25 mL), 50% (9897.25 mL), and 80 % (8189 mL), respectively. It is interesting to note that landfill leachate can produce maximum methane with the addition of DCD.

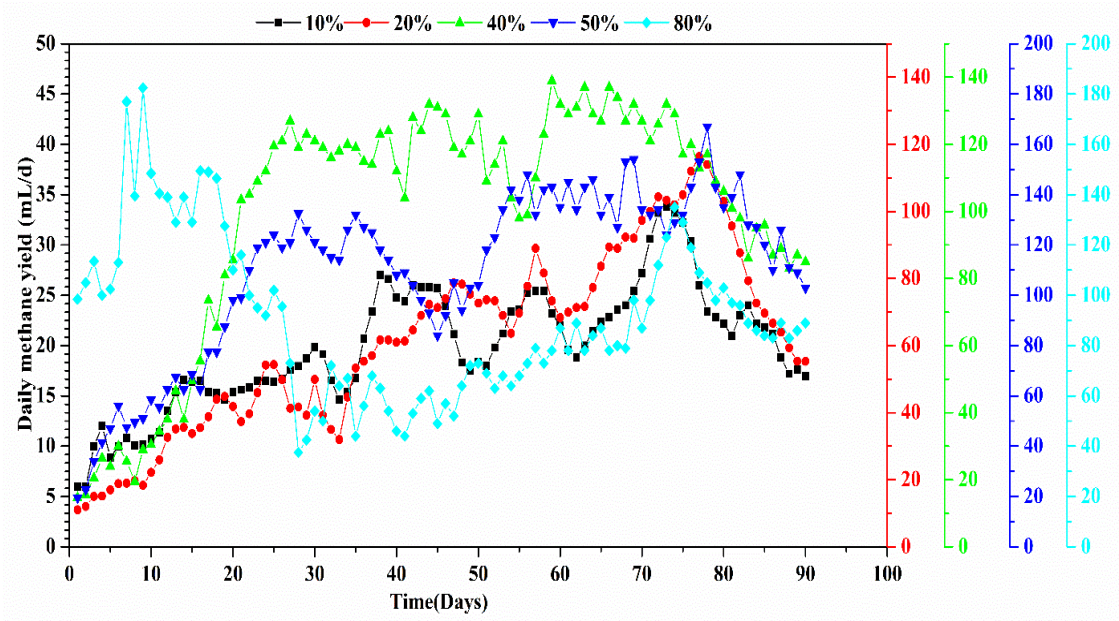


Fig. 7.12. Daily methane production (mL/d)

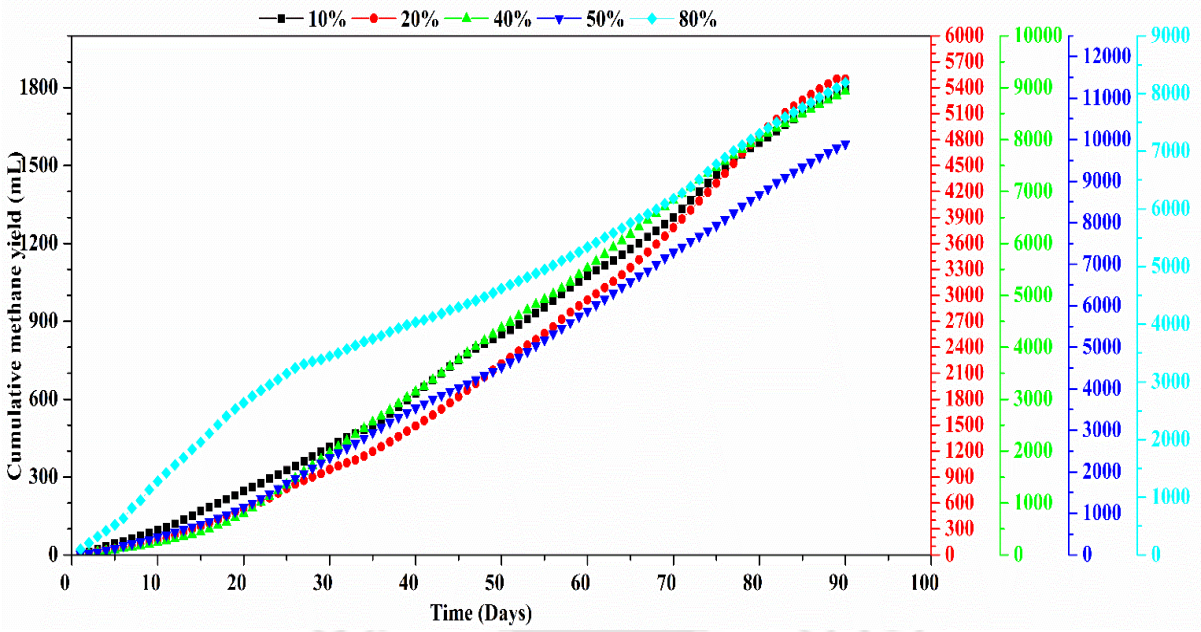


Fig. 7.13. Cumulative methane production (mL)

Therefore, 40 and 50% produce the maximum amount of methane, while 80% also produced maximum methane yield due to the presence of the maximum amount of DCD in the digester.

7.4.4 Effects of degradation on change in pH

In this process, solubilization or digestion of organics is an indication of the change in pH. For a reliable anaerobic process, appropriate pH values in the range of 6.8-7.2 are

required. The methanogens are most sensitive to changes in pH. Respectively. Effects of degradation on change in pH in Fig. 7.14. The VFA produced by acetogenic bacteria reduces the pH, which affects the overall activity of methanogens and subsequently decreases the biogas yield. Therefore, maintaining pH to the optimum range is crucial. During the entire process, no pH addition to the reactor. Maximum pH was observed in 40, 50, and 80% digesters.

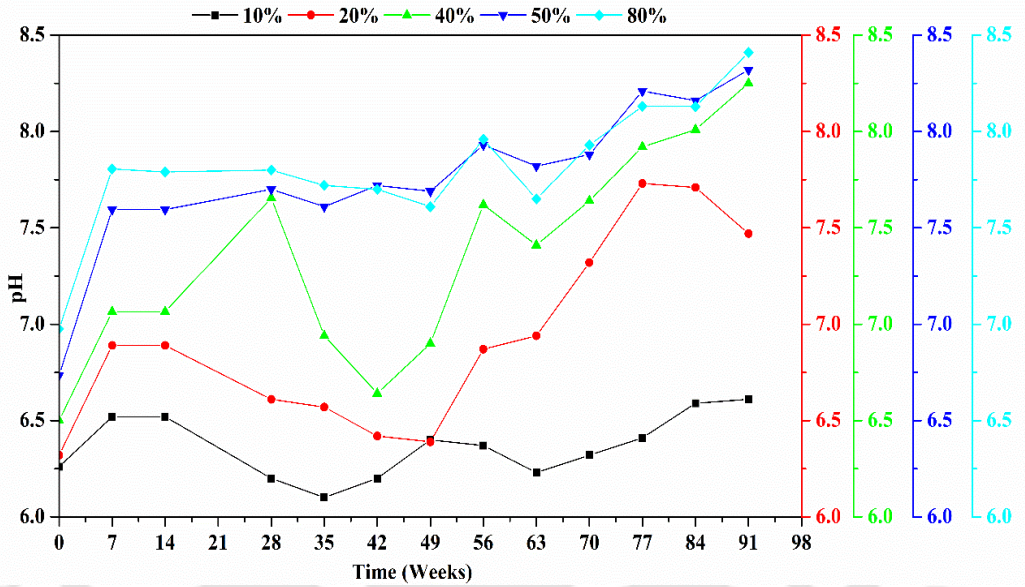


Fig. 7.14. Effects of degradation on change in pH

7.4.5 Optimal reduction of COD in landfill leachate

The information on the COD can be used to determine the hydrolysis and solubilization of the substrate. Hydrolytic bacteria present in the inoculum reduce complex chemical compounds to simpler monomers by releasing extracellular enzymes, increasing COD. Later on, the simpler monomers are consumed by another group of microbes called homoacetogens and methanogens, which simultaneously convert them to biogas under anaerobic conditions, decrease the COD in the digester (Menzel et al., 2020). Fig. 7.15 & 7.16 shows the reduction and removal efficiency of COD from landfill leachate with the addition of DCD. The maximum removal efficiency of COD of 10, 20, 40, 50, and 60% (v/v) of digesters was 17.2, 24.38, 57.56, 78.64, and 27.5%, respectively. Remarkably, the most significant reduction in COD removal was observed in 50% digester, while 80% digester showed a low reduction of COD removal. The reason could be the high C/N ratio provided to the 80% digester, resulting in effective COD removal. Higher inoculum ratio

influenced enhancing the biogas production rate, but a negative effect on COD removal. Therefore, the DCD effect is mainly attributed to more balanced nutrients and increased buffering capacity. The results showed that the addition of DCD in landfill leachate could be one of the options for efficient biogas production and better removal efficiency of high COD concentration.

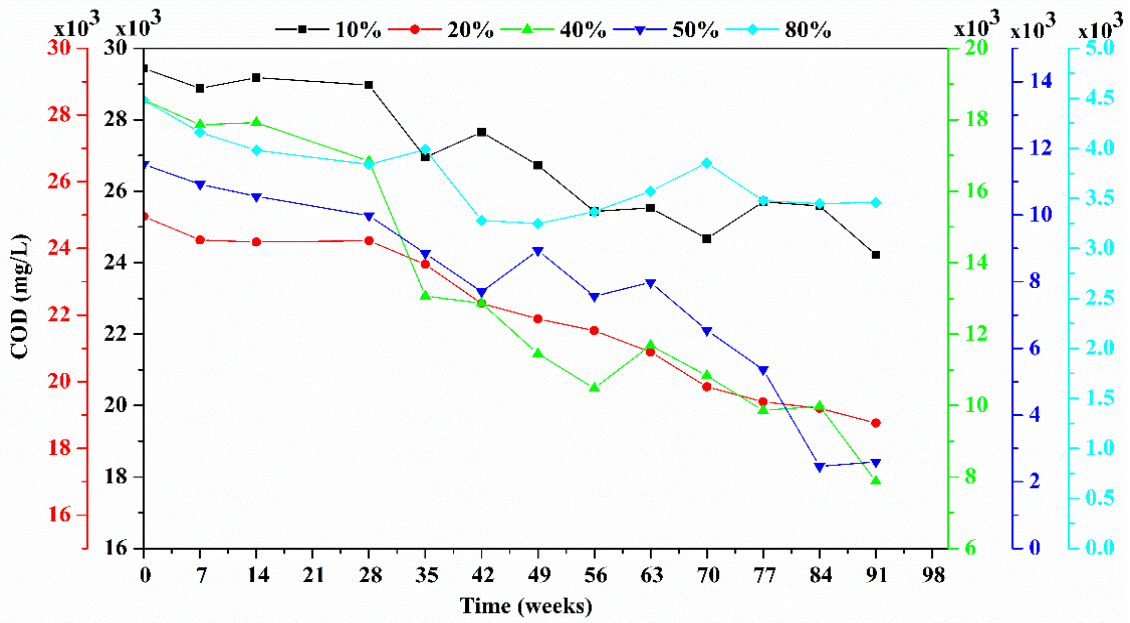


Fig. 7.15. Reduction of COD at different ratios of DCD (mg/L)

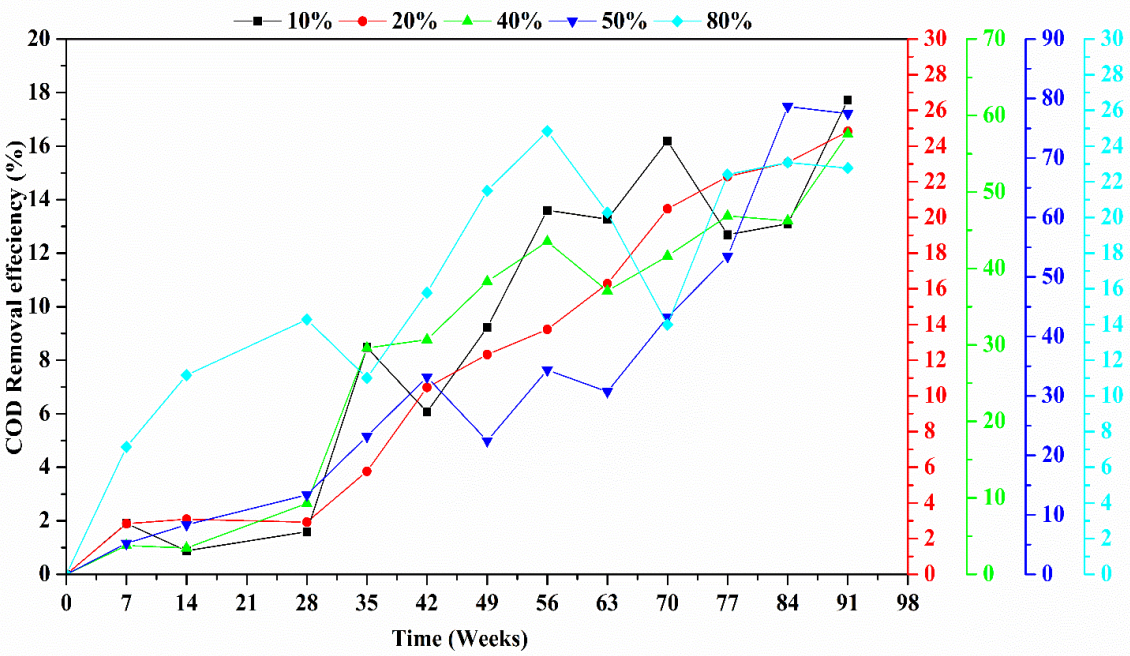


Fig. 7.16. The removal efficiency of COD at different ratios of DCD (mg/L)

7.5 SUMMARY OF PHASE-III (OBJECTIVE-IV (TREATMENT OF FRESH LANDFILL LEACHATE) & OBJECTIVE-V (BMP STUDY)

- Coagulation-flocculation studies conducted on high strength landfill leachate from simulated landfill reactor indicated that $FeCl_3$ could remove suspended solids (97.3%) and COD (55.4%) at a low dosage at pH 5, which is a better performance as compared to other coagulants like alum, $Fe(SO)_4.7H_2O$, Bentonite clay.
- Through physicochemical treatment, the COD value reached 32100 mg/L. Finally, after anaerobic filter treatment, COD of 23,170 mg/L was obtained. COD removal was 35% in coagulation treatment, 20.03% in the anaerobic filter, and overall removal was 51.52%, including both phases.
- Methanomicrobia (27.36%) at class level, Methanosarcinaceae (20.19%) at the family level, Methanosarcina (20.19%) at the genus level are found in the raw leachate. Hence, the study proved that methanogens could grow in an acidic environment.
- The maximum removal efficiency of COD in fresh landfill leachate of 10, 20, 40, 50, and 60% (v/v) of digesters was 17.2, 24.38, 57.56, 78.64, and 27.5%, respectively.
- Remarkably, the most significant reduction in COD removal was observed in 50% digester, while 80% digester showed a low reduction of COD removal. The reason could be the high C/N ratio provided to the 80% digester, resulting in effective COD removal. Therefore, it is interesting to note that landfill leachate can produce maximum methane with the addition of microbial consortium.

CHAPTER

8

Overall conclusions & future recommendations

This chapter has mainly drawn significant conclusions from the life cycle assessment on solid waste disposal strategies, the influence of tropical seasonal variation of leachate parameters, the treatment efficiency of physicochemical and biological technologies. Furthermore, the recommendations for future work are also presented here.

8.1 OVERALL CONCLUSIONS

- **In phase-I,** S4 (manual sorting + windrow composting + sanitary landfill with gas utilization) is the best choice from the perspective of the environment because of its minimum environmental impacts due to methane emissions, SO_X, NO_X, and heavy metal pollution can be avoided by separating waste materials and found to be suitable for the Guwahati municipal solid waste management. Economical perspective, S1 (windrow composting + open dumping) performs better because of less investment in equipment, maintenance cost, labor cost, and other expenses. The present study provides crucial quantitative insights and informative results to the decision-makers for the optimal investment in waste treatment technologies in India. Moreover, the findings of this study support the Indian policies to mandate the segregation of waste to implement the practices of the circular economy.
- **In phase-II (part-I)**, it was observed that there is no leachate production in the landfill which contains 38% of wet waste and 62% of dry waste due to the high amount of dry waste filled may not reach field capacity and signifies that developed countries have less problem handling the leachate issue. Landfill simulator operated without rainfall had higher concentrations of BOD_5, COD, VFA. The correlation coefficients of the

model presented acceptable goodness of fit for all leachate parameters. Therefore, this model can be used to estimate the amount of leachate generated by waste deposition in landfills at any given time. A low amount of methane was generated in the landfill system due to a lack of moisture and a high amount of acid production.

- **In phase-II (part-2)**, it is concluded that the landfill operated with rainfall has a positive effect on the acceleration of biological activity as proven that by a significant reduction in peak BOD_5 and COD concentrations. Leachate pH was lower due to carbon dioxide gas dissolution in leachate/water to form carbonic acid due to higher partial pressure exerted by CO_2. The methane production was the less initial phase of degradation due to pH value not being favorable for methanogenesis and a high VFA toxic to methanogens.
- A weak correlation was found between ambient and inside temperature assessed by the correlation coefficient (R^2=0.04), which means that landfill cell temperature depends significantly on the biological action taking place within the cell. On the other hand, the intercorrelations between leachate parameters were good.
- This study is expected to be a good simulation for cities with the waste composition of high wet waste (>70%) as the estimations of important design parameters such as BOD_5, COD, and VFA are studied in this research.
- As the importance of moisture (precipitation) is established in this study, it is suggestive through this study that some moisture additions can be designed in areas with low rainfall, such as arid zones.
- Result reveals that low leaching actions during dry season resulted in a smaller volume but high-strength leachate quality. Moreover, the seasonal variation of the leachate generation pattern indicates that the highest degradation of MSW occurred during the rainy season and the lowest degradation of MSW during the dry season due to lack of moisture in the landfill system.
- **In phase-III**, coagulation-flocculation studies conducted on high strength landfill leachate from simulated landfill reactor indicated that $FeCl_3$ could remove suspended solids and COD at a low pH which is a better performance than other coagulants like alum Bentonite clay. In addition, $FeCl_3$ proved as the best coagulant among other coagulants for treating fresh leachates where a high amount of inorganic and organic colloids was present. Therefore, the research discussed the important factors affecting

the relationship between the two stages of coagulation-flocculation and anaerobic filter treatment.

- Anaerobic filters have the potential to treat acidic landfill leachates. The coagulation-flocculation and the anaerobic filter technique can thus treat municipal solid waste leachate with the least alteration in the treatment method.
- It is interesting to note that methanogens can grow acidic pH 5. So precipitation had a negative effect on the leaching of potential bacteria from the landfills because of dilution and flushing out from the landfills that actually would be a better representative of landfill cells in open dumpsite conditions, which are mostly in confined conditions except those in top layers or just below the final cover.
- The highest COD removal rate was observed in leachates that were amended with bacteria. Thus, digested cow dung (DCD) showed a high potential to enhance biological treatment given its maximum, and COD removal efficiencies were 78% after three months.
- Therefore, this microbial seeding was found to have potential low-cost applications in the treatment of leachate with high COD. Results verified that applying a microbial seed to landfill leachate, which contains many kinds of pollutants, is a fast and efficient method to remove high levels of COD.
- This entire research outcome provides useful information for the design and management of landfill leachate for the realistic prediction of future trends.

8.2 FUTURE RECOMMENDATIONS

- A similar experimental study can be conducted on different ratios of waste composition by changing the operating conditions of the landfill reactor.
- A detailed investigation should be carried out to landfill leachate treatment process with the aid of microbial inoculum to improve the efficacy of anaerobic filters.
- The attention should be given to improve the degradation of unshredded fresh MSW landfill conditions by adding different microbial inoculum (co-disposal studies).
- Conduct different instrumental analyses on fresh landfill leachate to identify the organic compounds during the process of degradation.
- Potential bacteria can be isolated from the landfill leachate and return to the waste landfills to reduce the length of time of the waste degradation.

Appendix

PUBLICATIONS

Published article

1. Venkatesh Reddy, C., Shekhar Rao, D., & Kalamdhad, A.S. (2020). Statistical modeling and assessment of landfill leachate emission from fresh municipal solid waste: A laboratory-scale anaerobic landfill simulation reactor study. *Waste Management & Research* 38(10), 1161-117**5.** DOI: 10.1177/0734242X20954280.

Articles (Submitted)

1. Venkatesh Reddy, C., Shekhar Rao, D., & Kalamdhad, A.S. (2021). Multistage treatment of fresh/young leachate from MSW simulated landfill using coagulation-flocculation and fixed bed up-flow anaerobic filter, submitted to *Water Processing Engineering* **(Revision submitted).**

2. Venkatesh Reddy, C., Nagendra S., Kalamdhad, A.S. (2021). Modeling of leachate quality generated from fresh municipal solid waste using the first-order exponential model: a simulated anaerobic landfill study under tropical condition, submitted to Environmental Science and Process Safety (**Under review).**

3. Venkatesh Reddy, C., Kalamdhad, A.S. (2021). Life cycle assessment of waste treatment strategies for municipal solid waste in the Guwahati city, North-east India: perspectives on economic analysis of the environmental impacts submitted to *Journal of Material Cycle and Waste Management* **(Under review).**

Book Chapters

1. Reddy C.V., Rao S.D., Kalamdhad A.S. (2021) Impact of Precipitation on Biodegradation of Fresh Municipal Solid Waste in Anaerobic Simulated Reactor. In: Kalamdhad A.S. (eds) *Integrated Approaches Towards Solid Waste Management*. Springer, Cham. DOI: org/10.1007/978-3-030-70463-6_29.

International Conferences

1. Chejarla, Venkatesh., Chakka, Nagendra., Kalamdhad, A.S., (2019) "Treatment of unstabilized leachate by Coagulation- flocculation using different coagulants" Poster presentation on "**Research Conclave'19**, held on March 14-17 at **Indian Institute of Technology, Guwahati.**

2. Chejarla, Venkatesh., Chakka, Nagendra., Kalamdhad, A.S., (2019) "Monitoring and operation of the waste degradation process and leachate characteristics of an anaerobic simulated landfill reactor" oral presentation in the **International Conference on Solid Waste 2019 (ICSW 2019)** held on **Nov13-16, 2019, Hangzhou, China.**

3. Chejarla, Venkatesh., Kalamdhad, A.S., (2020) "Impact of precipitation on biodegradation of fresh municipal solid waste in simulated anaerobic reactor" oral presentation in the international conference **Recycle 2020**, held at **Indian Institute of Technology, Guwahati.**

National conferences

1. Chejarla, Venkatesh., Chakka, Nagendra., Kalamdhad, A.S., 2019. "Impact of MSW on stabilization, temporal leachate characteristics: a laboratory-scale landfill simulation reactor study" Oral presentation on "**National Environmental Conference 2019**" held on 31st January- 2nd February at **Indian Institute of Technology, Bombay.**

Awards/Achievements

1. SERB-DST travel grant for attending the **International Conference on Solid Waste 2019 (ICSW 2019)** held on **Nov13-16, 2019, Hangzhou, China.**

2. Won a partial scholarship in ISWA-SWIS Winter School 2020 held on January 13th – 24th, 2020, at the **University of Texas at Arlington, Arlington, Texas, USA.**

3. IIT Guwahati Student Travel assistance fund for attending ISWA-SWIS Winter School 2020 held on January 13th-24th, 2020, at the **University of Texas at Arlington, Arlington, Texas, USA.**

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