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WITWATERSRAND,
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SIZING LEACHATE DAMS AND TREATMENT INFRASTRUCTURE IN SOUTH
AFRICAN LANDFILLS

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the degree of Master of Science in Engineering.

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Declaration

I, Khutso Nkwane, declare that this research report is my own unaided work. It is being submitted to the University of the Witwatersrand, Johannesburg for the Degree of Master of Science in Engineering. It has not been submitted before for any degree or examination at any other university.

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(Signature of Candidate)

....1.... day of August.... year....2025

ABSTRACT

Adequate leachate management through leachate collection, storage, and treatment is vital for preventing groundwater and surface water pollution during landfilling. The storage of leachate using lined pollution control dams, referred to as leachate dams, before treatment is common in landfills. However, leachate dams are at a high risk of leakage and spillage if they are not designed and sized appropriately. Furthermore, the size of a leachate dam directly informs the sizing of the leachate treatment processes that follow it by establishing a required average treatment rate. The purpose of this research is to provide a framework for the adequate design and sizing of leachate dams and leachate treatment infrastructure in South Africa, by investigating the regulations for the design of leachate dams and leachate treatment; investigating applicable processes for treating leachate; and analysing the suitability of hydrological models and methods to estimate leachate discharge and leachate evaporation for leachate dam size water balance modelling.

A thorough literature review was conducted to study: (1) the regulatory requirements for leachate dam design and leachate treatment; and (2) the different methods and processes used for leachate treatment. Analyses were conducted to verify whether the Hydrologic Evaluation of Landfill Performance model (HELP model) is suitable for estimating leachate generation and discharge; predicted leachate discharge values computed by the HELP model using climatic records of an existing landfill were compared to leachate discharge values observed and recorded at the landfill. Furthermore, a method for estimating leachate evaporation was derived in this research; the method's suitability was analysed by comparing its predicted leachate evaporation values, computed using climatic and leachate composition records at a landfill, to the actual leachate evaporation values observed and recorded at the landfill.

To prevent excessive leachate seepage into groundwater, South African regulations mandate that leachate dams be designed with a liner system that is adequate for the waste disposed in the landfill. Moreover, the regulations mandate that leachate dams must be designed with a storage capacity large enough that they are not likely to spill

more than once every 50 years. Finally, there is a requirement for leachate to be adequately treated for permanent containment or discharge into the environment; the regulations provide minimum requirements for treated effluent's quality, as well as its allowable temperature and discharge rate if it is to be discharged into the environment.

Studies on leachate treatment systems discuss the use of physical and chemical processes, such as flotation and precipitation; biological processes, such as activated sludge processes; and advanced treatment processes, such as reverse osmosis. The studies recommend the use of reverse osmosis as a final leachate treatment step, with a biological treatment and/or chemical treatment step preceding it.

The percent error between the observed average daily leachate discharge rate and the average daily leachate discharge rate predicted by the HELP model was found to be 9.780 %. This proves this research's hypothesis of 10 % was correct. However, this was achieved through the use of an apparent saturated hydraulic conductivity that was significantly higher than what is reported in existing literature for landfilled waste. This variance is considered to be attributable to the methods used in the existing literature for determining saturated hydraulic conductivity; the methods do not take into account large impenetrable pieces of waste, such as drums and containers, creating large flow paths.

The percent error between the observed monthly leachate evaporation and the monthly evaporation predicted by the derived leachate evaporation method was found to be 7.91 %. This slightly surpassed the research's hypothesis of 10 %. The derived method also predicted monthly leachate evaporation more accurately in the wet season than in the dry season.

Both the HELP model and derived leachate evaporation method were found to be suitable for use in leachate dam sizing to ensure that leachate dams fulfil South Africa's regulatory requirements on leakage and spillage. Further research is required using data recorded over longer periods in order to further verify the HELP model's

suitability. Given the significant variability in the chemical and biological composition of leachate, further research is also required using leachates with different compositions in order to further verify the suitability of the derived leachate evaporation method.

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1. INTRODUCTION

1.1 Introduction

Landfills are defined as engineered facilities where waste is systematically disposed of so as to control gas, liquid, and solid emissions from contaminating the surrounding environment (United Nations Office for Disaster Risk Reduction, 2020). In addition to waste disposal areas, landfills may also have roads; stormwater and pollution control infrastructure; laboratories; waste treatment areas; and leachate treatment systems, depending on the size of the landfill, its location's climate, and the type of waste disposed of (Department of Water Affairs and Forestry, 1998). Globally, 2.01 billion metric tonnes of municipal solid waste is generated annually, with 37 % of it being disposed of in some form of a landfill (Kaza, et al., 2018). Furthermore, landfilling is one of the most common forms of solid industrial waste disposal; over 9 billion metric tonnes of solid industrial waste is produced globally (Vignesh, et al., 2021). In South Africa, 98 million metric tonnes of waste is disposed of across the nation's 826 landfill sites (Department of Environmental Affairs , 2012; Engineering News , 2018). This accounts for 60 % of the municipal solid waste and 95 % of the hazardous industrial waste produced in South Africa (Nyika , et al., 2019).

The disposal of waste material by the process of landfilling presents a variety of difficulties for engineers to solve. One such difficulty is potential contamination, at best, and pollution, at worst, of groundwater and surface water by leachate from the landfill. Leachate is defined as liquid (typically rainwater and the initial water content of the disposed of waste) that has percolated through a landfill and has extracted suspended and dissolved contaminants from the landfill due to the physical, chemical, and microbial processes in the waste (Christensen, et al., 2000). Of the several environmental issues that landfills pose, groundwater pollution is considered the most important (Bjerg, et al., 2014). Leachate may contain concentrations of dissolved organic matter and inorganic components that are 1000 to 5000 times higher than in groundwater (Christensen, et al., 2000). Designed landfill's waste disposal areas

typically include a liner system that serves as a barrier between leachate and groundwater and a drainage system, above the liner, to collect and transfer the leachate (Tolaymat & Krause, 2020). However, the inadequate design, construction, and maintenance of these systems may result in the leakage of this leachate, thus contaminating the groundwater (Bjerg, et al., 2014). Sholichin (2012) investigated the groundwater and surface water contamination around a landfill located in Indonesia and found that leachate had a significant impact on the quality of both groundwater and surface water. Ololade et al., (2019) studied the influence of leachate on the groundwater quality around a landfill located in South Africa; the findings indicated that the groundwater was unfit for drinking, domestic use, and irrigation due to being polluted by leachate. The contamination and pollution of groundwater and surface water by leachate can occur through spillages and through leakage of the landfill's liner system. Spillage occurs when leachate builds-up, spills over the waste disposal area's liner system, and either infiltrates into the surrounding soil or runs off into surrounding water bodies. South African regulations require that the liner systems of municipal and hazardous industrial landfills' be composed of a high density polyethylene (HDPE) geomembrane underlain by an impermeable clay layer (Department of Forestry, Fisheries and the Environment, 2024); leakage through these liner systems occur when leachate flows into holes in the geomembrane, caused by damage and manufacture defects, and permeates through the clay layer and then into the ground (Katsumi, et al., 2001; Rowe, 2012).

South African regulations require that landfills have leachate management systems for the collection and treatment of leachate (Department of Water Affairs and Forestry, 1998). A leachate management system is composed of a gravity drainage system, above a waste disposal area's liner system, that collects and discharges leachate into a leachate dam or sump for storage before treatment (Department of Water and Sanitation, 2021; George, 2021). Many landfills opt for the use of leachate dams due to cost implications (George, 2021). Leachate dams typically have leachate discharge and direct rainfall as inflows and evaporation, leakage, and abstraction for treatment as outflows. Thus, leachate dams act as equalization basins for the subsequent leachate treatment processes. Therefore, the size, efficiency, and effectiveness of the treatment

processes rely on the adequate sizing of the upstream leachate dams (Tchobanoglous, et al., 2014). South African regulations require that leachate dams be lined to the same standard as their upstream waste disposal areas (Department of Water and Sanitation, 2021). Leachate dams pose a higher risk of leakage than waste disposal areas due to having higher leachate depths; leakage through a liner system is proportional to the leachate depth above the liner system (Rowe, 2012). Leachate dams are also more susceptible to spillages due to being directly open to the climate, unlike waste disposal areas where rainfall is attenuated by waste.

Leachate management, achieved through collection, storage, and treatment, is one of the most significant contributors to landfill operational and maintenance (O&M) costs, accounting for up to 30 % to 50 % of the O&M expenses for landfills over a life cycle (Gisbert, et al., 2003; Kremen, 2020). Landfill disposal costs in South Africa are approximately R 650 per metric tonne, with 98 million metric tonnes of waste disposed of across 826 landfill sites (ReSource, 2023; Engineering News , 2018). Thus, the average cost of a South African landfill is approximately R 77 million, with the cost of leachate management being approximately between R 23 million and R 39 million. Therefore, there is a substantial cost implication to adequately designing leachate management systems; over-sized leachate collection, storage, and treatment infrastructure adds a high unnecessary cost to the construction and O&M of a landfill.

1.2 Problem Statement

Leachate poses a serious risk to the environment and to human health through the pollution of groundwater and surface water (Dervisevic, et al., 2016). This risk is compounded by the fact that South Africa is a water-scarce country that has several communities obtaining drinking, domestic, and irrigation water directly from the ground and rivers (Odiyo & Makungo, 2012). Groundwater and surface water pollution can result from the inadequate design of a landfill (Bjerg, et al., 2014); this extends to the inadequate design of a landfill's leachate collection, storage, and

treatment systems. Conversely, over-designing a landfill's leachate management systems adds an unnecessary cost to the construction and operation of a landfill.

Several studies have been conducted on the generation of leachate (Schroeder & Peyton, 1988; Fellner & Brunner, 2010), the environmental impacts of leachate (Engelbrecht, 1993; Bjerg, et al., 2014), the treatment of leachate (Renou, et al., 2008; Wiszniowski, et al., 2006), and the design of landfills' waste disposal areas to mitigate leachate leakage (Katsumi, et al., 2001; Rowe, 2012; Giroud & Bonapart, 1989). South African regulations are clear on the liner system design and size requirements of leachate dams, however, very limited information exists on the methods to be used for the sizing of leachate dams in order to achieve the requirements; this is especially relevant given the prevalence of leachate dams and the high risk that they pose to the environment and to human health due to the larger volumes of leachate stored, which lead to a higher probability of leakage and spillages when compared to landfills' waste disposal areas. Furthermore, adequately sized leachate dams serve as effective equalization basins to downstream leachate treatment processes. Thus, it is crucial that factors that inform the appropriate sizing of leachate dams, and hence leachate treatment infrastructure, be investigated.

1.3 Research Questions

This research attempts to address the following questions:

“Is the Hydrologic Evaluation of Landfill Performance (HELP) model suitable for predicting leachate discharge in order to adequately size leachate dams?”

“Is the leachate evaporation equation presented in this research appropriate for adequately sizing leachate dams?”

1.4 Hypothesis

Leachate discharge can be predicted to a percent error of 10 % using the Hydrologic Evaluation of Landfill Performance (HELP) model and leachate evaporation can be predicted to a percent error of 10 % using correlations and equations reported in Valderrama et al. (2015) and Schroeder et al. (1994).

1.5 Aim and Objectives

This research aims to provide a framework for adequately sizing leachate dams and treatment infrastructure in South Africa, in order to minimise environmental impacts and optimise cost. The objectives of this study are to: (1) conduct a review of the South African regulations on leachate dams and leachate treatment; (2) conduct a review of existing leachate treatment processes; (3) analyse the Hydrologic Evaluation of Landfill Performance (HELP) model's suitability for predicting leachate discharge for the purpose of dam sizing; and (4) derive and analyse a method for predicting the leachate evaporation for the purpose of dam sizing, using correlations and equations reported in Valderrama et al. (2015) and Schroeder et al. (1994).

1.6 Assumptions

Two assumptions were made for this study: (1) Water balance modelling can be used for appropriately sizing leachate dams and (2) the only inflows and outflows considered in a leachate dam are leachate discharge, rainfall, evaporation, leakage, and abstraction for treatment.

1.7 Rationale for the study

The annual quantity of solid waste generated in Africa is expected to nearly triple by 2050, with there being a large focus on building sustainable final disposal sites (Kaza, et al., 2018). Furthermore, existing landfills in South Africa have nearly reached their final capacity, with the cities of Johannesburg, Cape Town, and Tshwane reported to have less than ten years of landfill life left (Engineering News, 2021; Engineering

News , 2018). Thus, an increase in the design and construction of new landfills in South Africa is imminent.

Given the risks that landfills pose to the environment and human health, adequate design methods for landfills are vital in preventing pollution due to landfilling. This research attempts to add to the knowledge of adequate landfill design by establishing a framework for the sizing of leachate dams, which are more likely to leak and spill than waste disposal areas. Furthermore, this research attempts to provide a model for predicting leachate evaporation; outside of leachate dam sizing, this model may be used in the water balance modelling of leachate treatment processes that are open to the environment as well, such as lagoons, sequencing batch reactors, and clarifiers.

1.8 Research Report Structure

The structure of the research is as follows:

- Chapter 1: Introduces the research subject, discusses its relevance, and highlights the aim and objectives of the research.
- Chapter 2: Conducts a literature review on the formation of leachate, its treatment, South African regulations surrounding it, and the hydrological models for the estimation of its generation/discharge and evaporation that the research attempts to verify.
- Chapter 3: Details the study area, instrumentation, and methods to be used to attempt to verify the hydrological models that estimate leachate discharge and leachate evaporation.
- Chapter 4: Presents the results and discussion of the research.
- Chapter 5: Concludes the research, highlighting its implications and making recommendations for future research.

2. LITERATURE REVIEW

2.1 Introduction

Landfills present significant environmental impacts, such as greenhouse emissions and air pollution, however, groundwater contamination is considered the most important (Bjerg, et al., 2014). Along with groundwater, landfills also present a substantial environmental impact to surface water and soil. The contamination of groundwater and surface water, as well as the contamination of soil, through landfilling is caused by leachate percolating through a landfill's waste and leaking into the ground, as well as spilling over the landfill's boundaries. Although the chemical and biological composition of leachate varies significantly depending on the waste composition and age of a landfill, as well as the landfilling methods used, leachate typically contains toxic metals, a significant community of pathogenic organisms, and hazardous organic and inorganic matter; these contaminants are severely harmful to living organisms (Siddiqua, et al., 2022; Christensen, et al., 2000; Bjerg, et al., 2014). The pollution of soil, groundwater, and surface water due to landfilling seriously endangers people's health, with children being the most affected due to having weaker immune systems (Dervisevic, et al., 2016; Siddiqua, et al., 2022). These contaminants are also severely harmful to aquatic organisms, causing eutrophication and altering the reproductive rates of marine animals and plants (Siddiqua, et al., 2022).

The adequate design and construction of landfills is vital to the prevention of groundwater, surface water and soil pollution due to leachate (Bjerg, et al., 2014). South African regulations mandate the use of liner systems, to limit the movement of leachate into groundwater, and leachate management systems, to collect, store and treat leachate (le Roux, 2010). The type of liner system used is mandated by South African regulations and depends on the type of waste to be disposed of; however, it typically consists of a combination of HDPE geomembranes and low-permeability soils, such as clays. Leachate collection systems consist of a drainage layer, typically a granular material with perforated pipes embedded, directly above the liner system to drain and discharge leachate into a sump or leachate dam (Department of Water and

Sanitation, 2021). This drainage layer is also required to reduce the hydraulic head on the liner system, thus further limiting leachate movement into groundwater (Department of Water and Sanitation, 2021). From the sump or leachate dam, leachate is treated before release into the environment (Department of Water Affairs and Forestry, 1998).

This chapter reviews: (1) literature on the formation, composition, and treatment of leachate; (2) South African regulations on treated leachate effluent; (3) South African regulations on the design of leachate dams' liner systems; (4) literature on the leakage of leachate through liner systems; (5) the hydrological model that this research expects will be suitable for the prediction of leachate discharge for the purpose of a dam sizing water balance; and (6) literature that the research used to derive a model for the prediction of leachate evaporation.

2.2 Formation and Composition of Leachate

Landfill leachate is generated by rainwater infiltrating and percolating through landfilled waste layers. The physical, chemical, and microbial processes that occur in the waste layers then transfer pollutants from the waste to the percolating water, thus creating leachate (Christensen, et al., 2000). Leachate composition significantly varies according to the waste composition and age of a landfill (Bjerg, et al., 2014). In particular, the age of a landfill is important in determining the composition of leachate due to the fact that there are changes in the physical, chemical, and microbiological processes that occur in the landfill's waste over time. Focusing on typical landfills receiving a mixture of municipal and industrial waste, younger landfills contain large amounts of biodegradable organic matter; thus, rapid anaerobic fermentation occurs resulting in a large release of volatile fatty acids (VFAs) – this phase of a landfill's life is called the acidogenic phase (Wisniewski, et al., 2006). Leachate generated in the acidogenic phase is characterized by average BOD₅/COD ratios of around 0.58, indicating easy biodegradability, lower average pH values of around 6.1, and higher concentrations of inorganic matter and heavy metals (Wisniewski, et al., 2006; Christensen, et al., 2000; Bader, et al., 2022). In contrast, older landfills develop

methanogenic microorganisms that convert VFAs into methane and carbon dioxide – this phase in a landfill’s life is called the methanogenic phase (Wiszniowski, et al., 2006). Leachate generated in this phase is characterized by lower average BOD₅/COD ratios of around 0.06, higher average pH values of around 8, and an organic fraction dominated by non-biodegradable compounds, such as humic acids (Wiszniowski, et al., 2006; Christensen, et al., 2000). Leachate generated in the methanogenic phase nevertheless has lower concentrations of pollutants (Christensen, et al., 2000).

Although the composition of leachate varies from landfill to landfill, according to waste, it can generally be characterised as containing large amounts of four groups of pollutants: (1) dissolved organic matter, expressed in COD, such as methane and VFAs; (2) Inorganic macro components, such as ammonium and sodium; (3) heavy metals, such as lead and zinc; and (4) xenobiotic organic compounds that originate from household and industrial chemicals (Bjerg, et al., 2014; Christensen, et al., 2000). Other compounds, such as lithium and mercury, can also be found in leachate; however, when measured, they are found to exist in low concentrations (Christensen, et al., 2000). **Table 2-1** presents a range of the concentrations of some of the contaminants found in leachate generated in the acidogenic phase and methanogenic phase.

Table 2-1: Contaminant Concentrations Found in Leachate (Christensen, et al., 2000; Wdowczyk & Szymanska-Pulikowska, 2020; Kulikowska, 2012)

Contaminant	Acidogenic phase	Methanogenic Phase
pH	4-5 – 8.9	7.5 – 9.1
Chemical Oxygen Demand (mg/l)	843 – 60 000	500 – 4500
Total Dissolved Solids (mg/l)	3470 – 7830	2140 – 3050

Ammonia nitrogen (mg/l)	66 – 786	0.002 – 786
Calcium (mg/l)	43.9 – 1200	69 – 339
Chloride (mg/l)	11.5 – 2811	0.82 – 765
Iron (mg/l)	2.6 – 2100	0.019 – 280
Zinc (mg/l)	0.1 – 12	0.03 – 4

Table 2-1 was adapted from Christensen et al., (2000) and updated using Wdowczyk and Szymanska-Pulikowska (2020), as well as Kulikowska (2012). Although there is a significant variation in the concentration of contaminants across different landfills, it can be seen that leachates generated in the methanogenic phase generally have smaller concentrations of contaminants than leachates generated in the acidogenic phase.

2.3 Leachate Treatment Systems

Leachate treatment systems reviewed from literature can be broadly classified into four categories: (1) leachate transfer and recycling; (2) biological treatment through aerobic and anaerobic processes; (3) chemical and physical treatment, such as flotation, precipitation, and coagulation; (4) and advanced treatment processes, such as reverse osmosis (Wiszniowski, et al., 2006; Renou, et al., 2008). These categories are often combined in modular or multistage units for the removal of multiple contaminants, which may vary in concentration over time (Wiszniowski, et al., 2006). This section of the literature review investigates the processes used in each category. It will also investigate leachate treatment by leachate absorption and capping, as proposed in the “Minimum Requirements for Waste Disposal by Landfilling” (Department of Water Affairs and Forestry, 1998).

2.3.1 Leachate Transfer and Recycling

Leachate transfer refers to the treatment of leachate together with municipal sewage in a wastewater treatment plant (Renou, et al., 2008). It is a preferred leachate treatment method due to its easy maintenance and low cost of operation, however, it has been criticized because it introduces heavy metals and organic inhibitory compounds with low biodegradability into wastewater treatment plants, reducing the plants' efficiency (Renou, et al., 2008; Wiszniowski, et al., 2006). Multiple studies were published to find the optimal ratio between leachate and sewage that would lead to effective treatment: it was found that when the ratio between sewage and leachate was nine to one, nearly 95% BOD and 50% nitrogen removals were obtained and that a decrease in the ratio would result in a decrease in removal rate (Renou, et al., 2008).

Leachate recycling refers to the recirculation of leachate back into the landfill's waste disposal area (Renou, et al., 2008). This treatment method is widely used due to being one of the least expensive options available. Several studies have shown the benefits of this technique; authors have reported reductions in COD of between 63% to 73% due to leachate recycling (Renou, et al., 2008). Significant decreases in methane production have also been reported when the volume of recirculated leachate was 30% of the initial volume of waste. However, high recirculation rates may cause high concentrations of organic acids, lowering the pH value of leachate and causing acidic conditions in the waste; this has an adverse effect on the anaerobic degradation of the solid wastes due to acidic conditions being toxic to methanogenic microorganisms (Renou, et al., 2008).

2.3.2 Leachate Absorption and Capping

The "Minimum Requirements for Waste Disposal by Landfilling" (1998) states that at the end of a leachate dam's operating life, it must be emptied or filled with an absorbent solid material and sealed by a capping layer. The leachate is stored in the absorbent material, thus preventing leakage through the leachate dam's liner system, while the

capping layer serves to separate the leachate from the environment, thus preventing precipitation from infiltrating into the absorbent material and creating more leachate.

Figure 2-1 presents the capping system required for leachate dams if this method of leachate containment is followed (Department of Water Affairs and Forestry, 1998). The “Minimum Requirements for Waste Disposal by Landfilling” notes, however, that the design of capping systems is highly dependent on on-site conditions and circumstances.

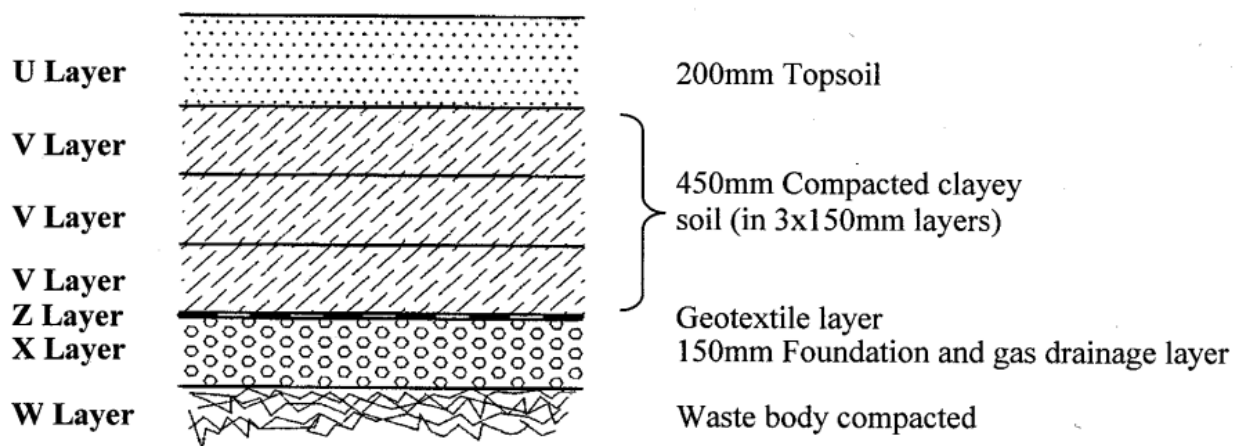


Figure 2-1: Leachate Dam Capping System (Department of Water Affairs and Forestry, 1998)

2.3.3 Microbial Biological treatment

The purpose of biological treatment is the removal of dissolved and particulate organic matter and nutrients using a variety of microorganisms (Tchobanoglous, et al., 2014; Wiszniowski, et al., 2006). Biological treatment can also be accomplished using other living organisms, such as with plant vegetation in constructed wetlands (Bakhshoodeh, et al., 2020). Due to its reliability and cost-effectiveness, microbial biological treatment, in the form of suspended-growth and attached-growth, is commonly used for leachate containing high BOD concentrations. Microbial biological processes have been shown to be very effective in the removal of organic and nitrogenous matter from leachates that have been generated in young landfills, where the BOD/COD ratio is

high (Renou, et al., 2008). However, as the landfill becomes older, the presence of refractory compounds limits the treatment's effectiveness (Renou, et al., 2008).

Due to the effectiveness and cost of treating leachate, the most often used microbial biological treatment process is the activated sludge process (Wang, et al., 2018). Activated sludge reactors consist of a reactor, in which microorganisms responsible for treatment are aerated and kept in suspension, and a downstream liquid-solid separation unit where washed-out microorganisms decant and are returned to the reactor (Tchobanoglous, et al., 2014; Wiszniowski, et al., 2006).

Anaerobic activated sludge processes used for treating leachate include anaerobic membrane bioreactors (MBRs) shown in Figure 2-2(a), and upflow anaerobic sludge blanket reactors (UASBs), shown in Figure 2-2(b).

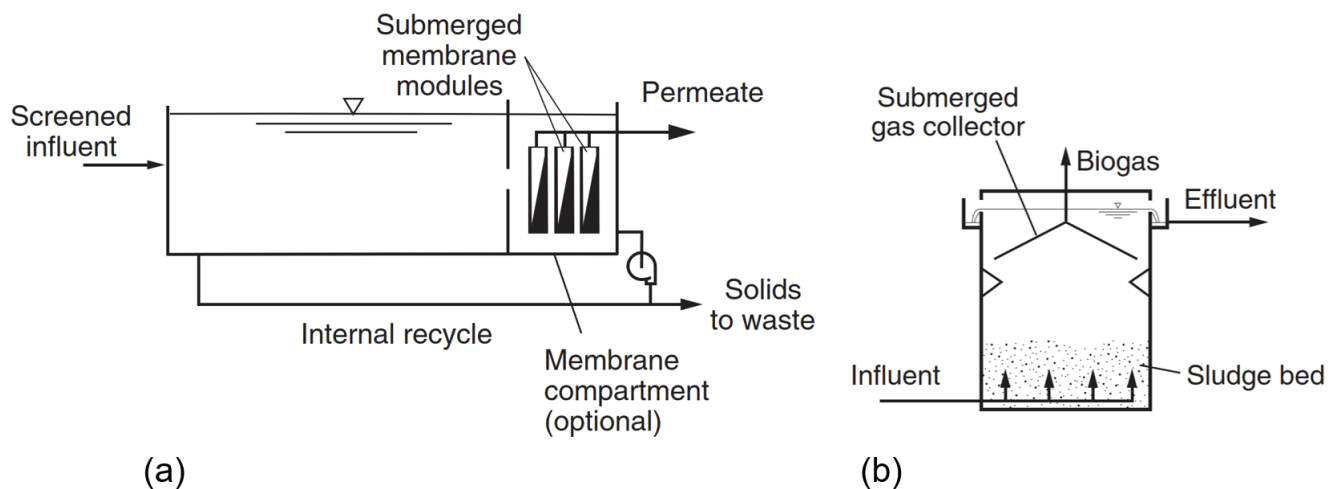


Figure 2-2: Schematics of Anaerobic Activated Sludge Processes: (a) Aerobic Membrane Bioreactor and (b) Upflow Anaerobic Sludge Blanket Reactor (Tchobanoglous, et al., 2014)

MBRs are activated sludge systems where membranes are located at the end of the reactor for liquid-solid separation in lieu of a downstream separation unit where solids have to decant (Tchobanoglous, et al., 2014). UASBs are activated sludge systems that consist of a bottom sludge bed and a dense granular biomass; mixing of the biomass in UASBs is achieved through upflow velocity and biogas generation (Tchobanoglous,

et al., 2014). Studies conducted found that MBRs and UASBs are capable of achieving COD removal rates of 62% and 80%, respectively (Wang, et al., 2018). The UASB process is often used to treat leachate with a high organic content (Wang, et al., 2018).

Aerobic activated sludge processes are typically used to remove ammonium from leachate; however, they have been shown to be effective in reducing COD (Wang, et al., 2018). The most predominant activated sludge systems used in leachate treatment, due to their simple structures and large capacities, are the sequencing batch reactors (SBRs). SBRs consist of single complete-mix reactors in which all of the activated sludge process steps occur, eliminating the need for downstream solid separation units (Tchobanoglous, et al., 2014). Studies conducted have shown that aerobic SBRs are capable of achieving ammoniacal nitrogen removal rates of 99 %. Studies have also shown that aerobic SBRs are capable of achieving COD removal rates between 80 % and 85 % (Wang, et al., 2018).

2.3.4 Chemical and Physical Treatment

Physical and chemical treatment involves the reduction of suspended solids, colloidal particles, toxic substances, and refractory compounds through the application of physical forces and the induction of chemical reactions (Renou, et al., 2008; Wang, et al., 2018; Tchobanoglous, et al., 2014). Due to the high organic matter and ammoniacal nitrogen present in leachate, physical and chemical treatment is typically used before microbial biological treatment processes to reduce toxicity and improve biodegradability (Wang, et al., 2018; Wiszniowski, et al., 2006). Physical and chemical treatment methods include flotation, coagulation/flocculation, precipitation, and chemical oxidation.

Flotation refers to the separation of particles from a liquid through the introduction of fine gas bubbles into the liquid (Tchobanoglous, et al., 2014). Flotation is effective in the reduction of turbidity, COD, and non-biodegradable compounds such as humic acids (Renou, et al., 2008; Palaniandy, et al., 2010). Under optimised conditions,

flotation has been found to achieve turbidity and COD removal rates of 32 % and 36 %, respectively in leachates, as well as a humic acid removal rate of 60 % (Palaniandy, et al., 2010; Renou, et al., 2008). Flotation systems that included coagulant dosages were found to achieve turbidity and COD removal rates of 79 % and 70 %, respectively (Palaniandy, et al., 2010).

Coagulation refers to the process of adding a substance into a solution to chemically destabilise its particles' repelling charges in order to aggregate the particles through microflocculation (Tchobanoglous, et al., 2014). Microflocculation refers to the aggregation of particles smaller than 2 μm through collisions brought about by the random thermal motion of fluid particles (known as Brownian motion). Flocculation refers to the aggregation of particles larger than 2 μm through physical collisions so that they become large enough to settle out of a solution or be filtered out. (Tchobanoglous, et al., 2014). Coagulation and flocculation are widely used in leachate treatment as a pre-treatment, before biological treatment or reverse osmosis, or as a final treatment step in order to remove non-biodegradable organic matter (Renou, et al., 2008). Studies have found coagulation and flocculation to achieve COD removal rates of between 10 % and 38 % for leachates generated in younger landfills and between 50 % and 75% for leachates generated in older landfills (Renou, et al., 2008). Coagulants commonly used include alum, ferrous sulphate, ferric chloride (Wiszniowski, et al., 2006).

Chemical precipitation refers to the addition of chemicals into a solution in order to cause dissolved and suspended solids to settle out of the solution as a solid precipitate (Tchobanoglous, et al., 2014; United States Environmental Protection Agency, 2000). While used in wastewater treatment for the removal of metals, suspended solids, oils, and greases, it has been widely used in leachate treatment as a pre-treatment process for the removal of high strength ammonia and ammonium nitrogen, prior to biological treatment (Renou, et al., 2008; United States Environmental Protection Agency, 2000). Studies have found chemical precipitation to have ammonia removal rates of between 66 % and 86 % (Renou, et al., 2008). Chemicals used in chemical precipitation include

lime, ferrous sulphate, ferric chloride, and alum (United States Environmental Protection Agency, 2000; Tchobanoglous, et al., 2014).

Chemical oxidation refers to the use of oxidizing agents to bring about a change in a substance's chemical composition (Tchobanoglous, et al., 2014). Chemical oxidation is used for the removal of non-biodegradable organic matter and toxic substances (Wiszniowski, et al., 2006; Tchobanoglous, et al., 2014). Studies have found commonly used oxidants, such as chlorine and ozone, to have COD removal rates of between 20 % and 50 % when used for leachate treatment. In particular, advanced oxidation processes (AOPs) have been widely studied as a method for removing non-biodegradable organic matter from leachate. AOPs, such as ozone/UV and ozone/hydrogen peroxide, have been found to have COD removal rates of between 28 % to 95 % (Renou, et al., 2008).

2.3.5 Membrane Processes

Membrane processes involve the removal of particulate and dissolved matter using diffusion and/or sieving through a membrane with minuscule pore sizes; the purpose of the membrane is to act as a selective barrier that allows for selected matter to pass through while preventing and retaining the rest (Tchobanoglous, et al., 2014). The membrane process that is most effective in the removal of particulate and dissolved matter is reverse osmosis (RO). A schematic of the RO process is shown in Figure 2-3.

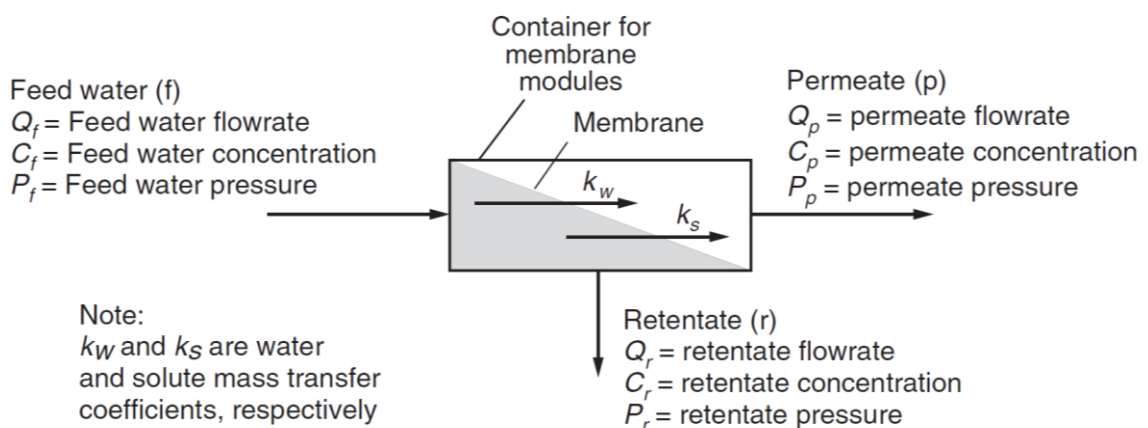


Figure 2-3: Schematic of the RO Process (Tchobanoglous, et al., 2014)

RO has been proven to be the most efficient and indispensable solution in achieving leachate treatment effluent regulations and has become one of the most commonly used methods for leachate treatment (Renou, et al., 2008; Talalaj, et al., 2021). Studies have shown RO to achieve removal rates of between 98 % to 99 % for most contaminants in leachate, such as heavy metals and COD (Wiszniewski, et al., 2006; Renou, et al., 2008; Pure Water Products, LLC, 2023). The most frequent problem related to the use of RO systems is fouling (Talalaj, et al., 2021; Wiszniewski, et al., 2006). Fouling occurs when contaminants in the incoming water (feed water) accumulate on the RO membrane surface or inside its pores and severely decrease flow through the membrane (Tchobanoglous, et al., 2014; Talalaj, et al., 2021). Fouling results in decreased productivity and short membrane lifespans (Renou, et al., 2008). The removal of potential foulants can be achieved through the introduction of biological treatment before the RO process, or through the introduction of chemical and physical treatment, such as coagulation and flocculation (Talalaj, et al., 2021; Wiszniewski, et al., 2006).

2.4 Regulations for Leachate Effluent Quality

Once the leachate is treated, the effluent is generally discharged into the environment, although a portion may be used for landfill operations, such as waste treatment and dust suppression to minimize air pollution (Department of Water Affairs and Forestry, 1998). The general limits of the chemical and physical parameters for water discharged into a surface water resource are regulated by the “Revisions of General Authorisations in terms of Section 39 of the National Water Act, 1998” (2013). The general authorisation replaces the need for users to apply for licenses for the discharge of domestic and industrial wastewater into a surface water resource based on the following conditions: (1) the maximum discharge into a water resource is equal to or less than 2000 m³ on any given day; (2) the discharge does not alter the receiving water body’s normal ambient temperature by greater than 3 °C for general water resources and by more than 2 °C for specific water resources that are listed in the gazette; (3) wastewater complies with the general wastewater limit values set out in **Table 2-2**.

Special limits to specific water bodies that are listed in the gazette are also given in **Table 2-2**.

Table 2-2: Wastewater Limit Values Applicable for Discharge into a Surface Water Resource (Department of Water Affairs, 2013)

Substance/Parameter	General Limit	Special Limit
Faecal Coliforms (per 100 ml)	1 000	0
Chemical Oxygen Demand (COD) (mg/l)	75	30
pH	5.5 – 9.5	5.5 - 7.5
Ammonia (ionised and un-ionised) as Nitrogen (mg/l)	6	2
Nitrate/Nitrite as Nitrogen (mg/l)	15	1.5
Chlorine as Free Chlorine (Cl) (mg/l)	0.25	0
Suspended Solids (SS) (mg/l)	25	10
Electrical Conductivity (EC) (mS/m)	70 mS/m above intake to a maximum of 150 mS/m	50 mS/m above background receiving water, to a maximum of 100 mS/m
Orthophosphate as phosphorous (mg/l)	10	1 (median) and 2.5 (maximum)

Fluoride (F) (mg/l)	1	1
Soap, oil, or grease (mg/l)	2.5	0
Dissolved Arsenic (As) (mg/l)	0.02	0.01
Dissolved Cadmium (Cd) (mg/l)	0.005	0.001
Dissolved Chromium-VI (Cr-VI) (mg/l)	0.05	0.001
Dissolved Copper (Cu) (mg/l)	0.01	0.002
Dissolved Cyanide (CN) (mg/l)	0.02	0.01
Dissolved Iron (Fe) (mg/l)	0.3	0.3
Dissolved Lead (Pb) (mg/l)	0.01	0.006
Dissolved Manganese (Mn) (mg/l)	0.1	0.1
Mercury and its compounds (Hg) (mg/l)	0.005	0.001
Dissolved Selenium (Se) (mg/l)	0.02	0.02
Dissolved Zinc (Zn) (mg/l)	0.1	0.04
Boron (B) (mg/l)	1	0.5

When comparing **Table 2-2** to the typical range of concentrations of contaminants found in leachate, shown in **Table 2-1**, it can be seen that removal rates of up to nearly 100 % may be required for COD and ammonia nitrogen in order to achieve effluent quality standards. Removal rates of up to nearly 100 % may also be required for heavy metals.

2.5 South African Regulations on the Design of Leachate Dams

The design requirements for leachate dams, in order to limit leachate's movement into groundwater and minimise the risk of spillage, are regulated by the "Minimum Requirements for Waste Disposal by Landfill" (1998) and the "Regulations on Use of Water for Mining and Related Activities Aimed at the Protection of Water Resources" (2010). This section discusses the regulations' requirements on: (1) the liner system design of leachate dams; (2) the permissible leakage rates through leachate dams' liner systems; and (3) the acceptable risk of spillage.

2.5.1 Liner System Design of Leachate Dams

the "Minimum Requirements for Waste Disposal by Landfill" (1998) classifies landfills into two categories: (1) landfills where hazardous waste is disposed of and (2) landfills where general waste is disposed of. The regulation states that leachate dams at hazardous waste disposal landfills must have a hazardous waste lagoon liner system, shown in Figure 2-4, while leachate dams at general waste disposal landfills must have the same liner system as the landfills' waste disposal areas, excluding the leachate collection system, and including a 2 mm thick HDPE geomembrane as the uppermost layer.

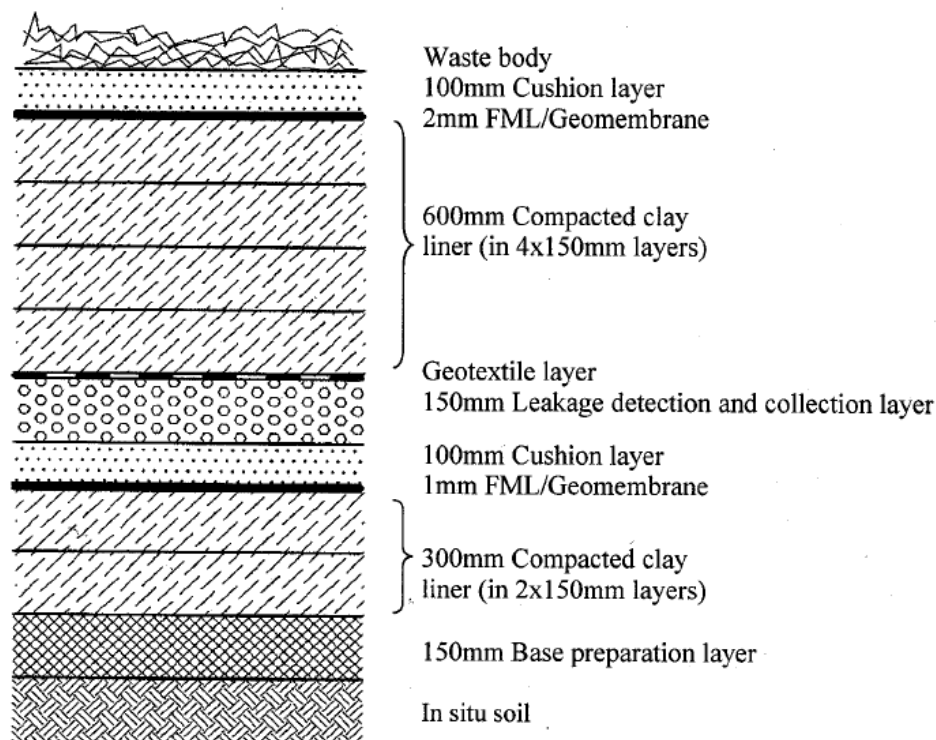


Figure 2-4: Hazardous Waste Lagoon Barrier System (Department of Water Affairs and Forestry, 1998)

Landfill liner systems are discussed in this section as leachate dam liner systems at general waste disposal landfills are adapted from them. In South Africa, the regulatory requirements for the liner system of landfills are set in the “National Norms and Standards for Disposal of Waste to Landfill” (2013) and its amendment (2024). The regulations assign a liner system to a landfill based on the type of waste that can be disposed of in the landfill. The regulation classifies wastes into five categories: Type 0 wastes, which include pesticides and untreated medical waste; Type 1 wastes, which include asbestos and chemicals from analytical or academic laboratories; Type 2 wastes, which include domestic wastes; Type 3 wastes, which include plastics and packaging waste; and Type 4 wastes, which include building rubble, excavated earth, etc. While Type 0 wastes cannot be disposed of in a landfill without going through a treatment and re-classification process, Type 1, 2, 3, and 4 wastes can be disposed of in landfills that have Class A, B, C, and D liner systems, respectively. The different classes of liner systems are shown in Figure 2-5. The Class A liner system is used for wastes that are considered hazardous (Department of Water and Environmental

Affairs, 2013); it consists of a primary 2 mm thick HDPE geomembrane and a 600 mm clay layer to limit leachate leakage. Below the clay layer, a leakage detection system is installed to monitor possible leakage through the primary geomembrane; below the leakage detection system, a secondary 1.5 mm thick HDPE geomembrane and 300 mm clay layer is installed to prevent the possible leachate leakage from seeping into the ground. The Class B, C, and D liner systems are used for the disposal of wastes that are considered general (Department of Water and Environmental Affairs, 2013). Class B and C liner systems consist of a single 1.5 mm thick HDPE geomembrane and clay layer, without a leakage detection system underneath. The Class D liner system has no geomembrane or clay layer. The Class B, C and D liner systems are adapted for their landfills' leachate dam liner systems as follows; for the Class B and C liner systems, the single 1.5 mm thick HDPE geomembrane is replaced with a 2 mm thick HDPE geomembrane, while for the Class D liner system, a 2 mm geomembrane is added.

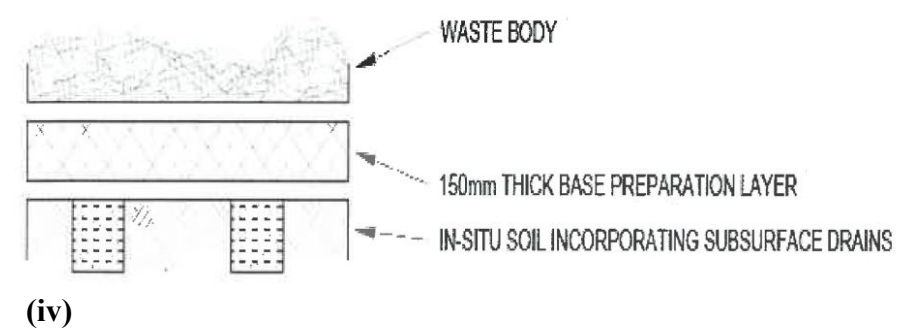
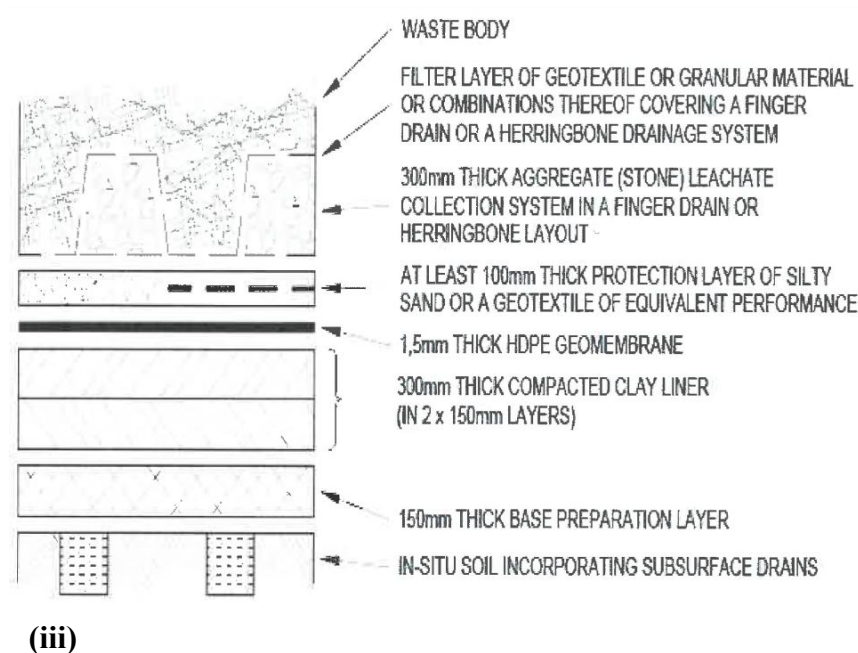
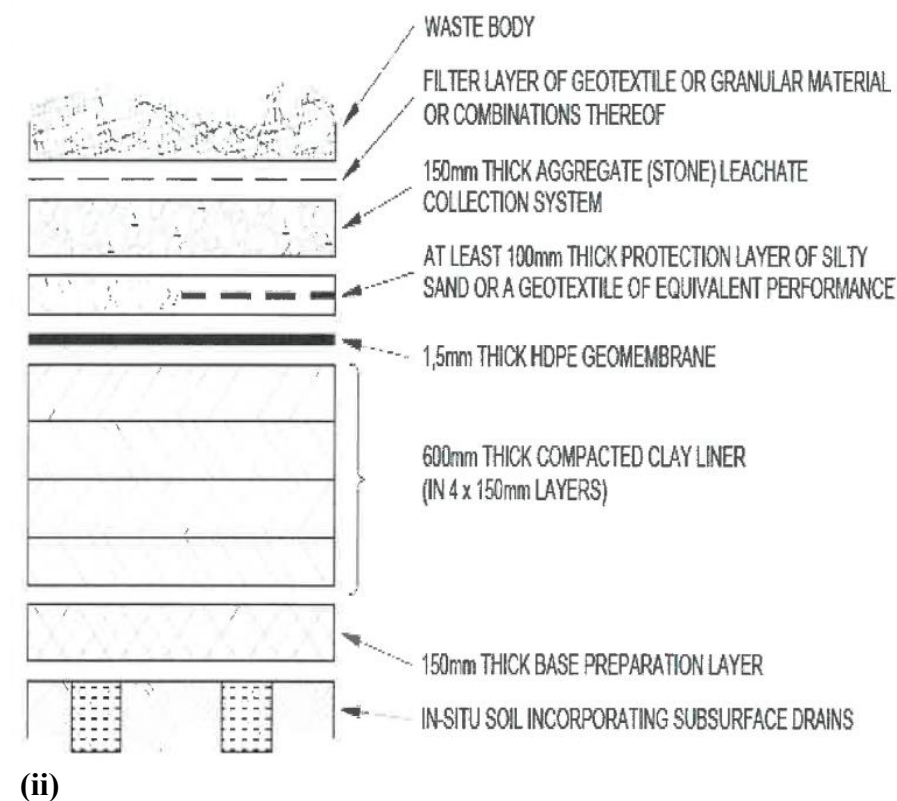
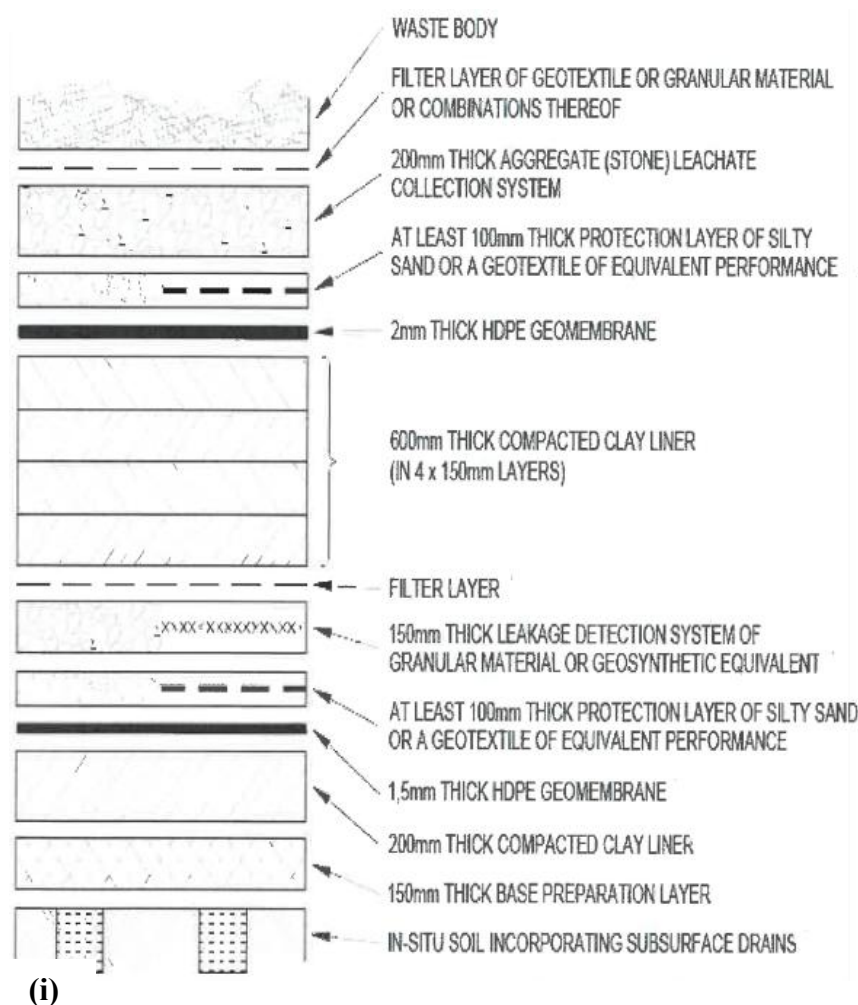


Figure 2-5: Landfill Liner Systems in South Africa: (i) Class A Liner System; (ii) Class B Liner System; (iii) Class C Liner System; (iv) Class D Liner System (Department of Forestry, Fisheries and the Environment, 2024)

Waste types are defined according to the “National Norms and Standards for the Assessment of Waste for Landfill Disposal” (2013). The objective of this regulation is to quantify chemical substances of concern in a waste by obtaining the Total Concentration (TC) of the substances in the waste, as well as the Leachable Concentration (LC) of the substances through the waste. The TCs and LCs are then used to categorise the waste as a Type 0 to a Type 4 waste. The regulation specifies that the TC of the chemical substances must be determined using methods that provide precise, consistent, and repeatable results (Department of Water and Environmental Affairs, 2013). A summary of some of the Total Concentration thresholds (TCTs) is presented in **Table 2-3**.

Table 2-3: Total Concentration Thresholds (Department of Water and Environmental Affairs, 2013)

Elements & Chemical Substances in Waste	TCT 0	TCT 1	TCT 2
Metal Ions (mg/l)			
Arsenic (As)	5.8	500	2 000
Boron (B)	150	15 000	60 000
Cadmium (Cd)	62.5	6 250	1040
Chromium Total (Total Cr)	50	800 000	Not Applicable
Chromium-VI (Cr-VI)	46 000	500	2 000
Copper (Cu)	16	19 500	78 000
Mercury (Hg)	0.93	160	640
Manganese (Mn)	1 000	25 000	100 000

Lead (Pb)	20	1 900	7 600
Selenium (Se)	10	50	200
Zinc (Zn)	240	160 000	640 000
Inorganic Anions (mg/l)			
Fluoride (F)	100	10 000	40 000
Cyanide Total (CN)	14	10 500	42 000
Organics (mg/l)			
Vinyl chloride	-	0.015	0.03
Pesticides (Aldrin and Dieldrin)	0.05	1.2	4.8

The LC analysis of a waste must be conducted using the “Australian Standard Leaching Procedure” (1999). The standard provides an outline for the preliminary assessment of a liquid or waste’s potential to contaminate groundwater (Standards Association of Australia, 1999). A summary of the leachable concentration thresholds (LCTs) used to categorise wastes is presented in **Table 2-4**.

Table 2-4: Leachable Concentration Thresholds (Department of Water and Environmental Affairs, 2013)

Elements & Chemical Substances in Waste	LCT 0	LCT 1	LCT 2	LCT 3
Metal Ions (mg/l)				
Arsenic (As)	0.01	0.5	1	4

Boron (B)	0.5	25	50	200
Cadmium (Cd)	0.003	0.15	0.3	1.2
Total Chromium (Total Cr)	0.1	5	10	40
Chromium—VI (Cr- VI)	0.05	2.5	5	20
Copper (Cu)	2.0	100	200	800
Mercury (Hg)	0.006	0.3	0.6	2.4
Manganese (Mn)	0.5	25	50	200
Lead (Pb)	0.01	0.5	1	4
Selenium (Se)	0.01	0.5	1	4
Zinc (Zn)	5.0	250	500	2000
Inorganic Anions (mg/l)				
Total Dissolved Solids (TDS)	1 000	12 500	25 000	100 000
Chloride	300	15 000	30 000	120 000
Sulphate	250	12 500	25 000	100 000
NO ₃ as N, Nitrate-N	11	550	1 100	4 400
F, Fluoride	1.5	75	150	600
CN ⁻ (total), Cyanide Total	0.07	3.5	7	28

Organics (mg/l)				
Vinyl chloride	0	0.015	0.03	0.12
Pesticides	0	0	0	0

Once the TC and LC of the substances of concern are quantified, the waste can be categorised as a Type 0 to Type 4 waste. **Table 2-5** presents the categorisation system for wastes based on waste types. The TC and LC of all of the substances of concern must relate to their TCT and LCT in the manner described.

Table 2-5: Waste Assessment Classifications (Department of Water and Environmental Affairs, 2013)

Waste Type	Classification
Type 0	$LC > LCT3$ or $TC > TCT2$
Type 1	$LCT2 < LC \leq LCT3$ or $TCT1 < TC \leq TCT2$
Type 2	$LCT1 < LC \leq LCT2$ and $TC \leq TCT1$
Type 3	$LCT0 < LC \leq LCT1$ and $TC \leq TCT1$
Type 4	$LC \leq LCT0$ and $TC \leq TCT0$

Along with Type 0 wastes, as defined in the “National Norms and Standards for the Assessment of Waste for Landfill Disposal” (2013), other types of wastes that are prohibited from disposal by landfill include wastes that are explosive, corrosive, or oxidizing under landfill conditions; wastes with a pH greater than 12 or less than 6; wastes that are flammable or contain compressed gases; untreated mercury; and infectious animal carcasses and animal waste.

2.5.2 Leakage Through a Liner System

The “Minimum Requirements for Waste Disposal by Landfill” (1998) sets the regulatory requirements for the maximum allowable leachate outflow rate through a liner system’s clay layers over a year. It states that the outflow rate through a liner system’s clay layers (i.e., the rate at which leachate will pass through the clay layers, taking into account the hydraulic head of leachate that is likely to accumulate over the liner system) must not exceed 0.3 m/year (1×10^7 cm/s) for hazard waste disposal landfills and 0.1 m/year (1×10^6 cm/s) for general waste disposal landfills (Department of Water Affairs and Forestry, 1998).

For leakage to occur through a leachate dam liner system’s clay layers, it must first travel through the 2 mm HDPE geomembrane placed above the clay layers. There are two primary mechanisms for the transport of leachate through a geomembrane: (1) diffusion through the geomembrane and (2) leakage through holes and defects in the geomembrane. Diffusion occurs when leachate travels through an intact geomembrane at the molecular level, due to a hydraulic head or vapour pressure gradients, (Schroeder, et al., 1994; Katsumi, et al., 2001). Leakage through holes and defects occurs when leachate travels through holes accidentally caused during a geomembrane’s installation or manufacture (Katsumi, et al., 2001; Schroeder, et al., 1994; Rowe, 2012). The level of leakage through diffusion is smaller than the level of leakage through a hole or defect (Schroeder, et al., 1994); thus, leakage through a hole or defect will produce the highest leachate outflow rate through a liner system’s clay layers. Giroud and Bonaparte (1989) investigated the occurrence of holes in installed geomembranes and found that they are unavoidable; 1-2 holes per hectare were found even when construction quality assurance was excellent. Once leachate leaks through a geomembrane hole, it flows through a liner system’s clay layer due to the leachate’s hydraulic head, based on Darcy’s law (Rowe, 2012).

Rowe (2012) shows that leakage outflow rates through composite liner systems (i.e., liner systems that include a geomembrane and clay layer) are higher when a hole sits on or coincides with a wrinkle in the geomembrane, compared to the outflow rates

through a hole that is in direct contact with the clay layer. Wrinkles in HDPE geomembranes occur due to thermal expansion after installation and can cover 3 % to 15 % of the entire installation area (Rowe, 2012). Rowe (1998), as cited in Rowe (2012), presents **Equation 2-1** to predict the leakage outflow rate through clay layers located under a geomembrane hole that has coincided with a geomembrane wrinkle, shown in Figure 2-6.

$$Q = 2L[kb + (kD\theta)^{0.5}]h_d/D$$

Equation 2-1 (Rowe, 2012)

Where,

- Q = Leakage rate through one hole (m^3/s)
- L = Length of a connected wrinkle (m)
- k = Hydraulic conductivity of the compacted clay liner (m/s)
- b = Half of the wrinkle width (m)
- $D = H_L + H_A$ (m)
- $h_d = h_w + H_L + H_A - h_a$ (m)
- θ = Geomembrane-clay liner transmissivity (m^2/s)

$$\text{Log}_{10}\theta = 0.07 + 1.036(\text{log}_{10}k_L) + 0.0180(\text{log}_{10}k_L)^2 - \text{for good contact.}$$

Equation 2-2 (Rowe, 2012)

$$\text{Log}_{10}\theta = 1.15 + 1.092(\text{log}_{10}k_L) + 0.0207(\text{log}_{10}k_L)^2 - \text{for poor contact.}$$

Equation 2-3 (Rowe, 2012)

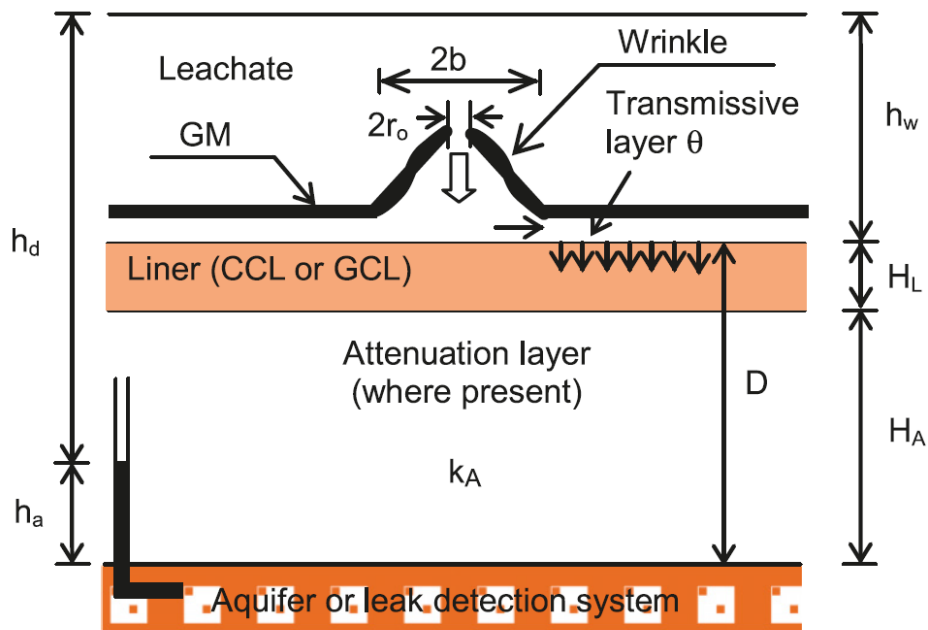


Figure 2-6: Leachate Outflow Rate through a Liner System with a Hole in the Geomembrane (Rowe, 2012)

Rowe (2012) states that the probability of a hole coinciding with a wrinkle is approximately 50 %. Thus, if there is more than one hole in a geomembrane after installation, it is likely that a hole will coincide with a wrinkle. Therefore, given that 1-2 holes per hectare are likely when construction quality assurance is excellent, it can be assumed that one hole will coincide with a wrinkle (Giroud & Bonapart, 1989; Rowe, 2012). The length of a connected wrinkle can vary from 20 m to 1400 m, depending on the solar radiation and temperature on the geomembrane (Rowe, et al., 2012). Although wrinkles occasionally reach 0.2 m in height and 0.5 m in width, the average height of a wrinkle is approximately 0.06 m, while the average width ranges between 0.2 m and 0.25 m (Rowe, 2012). Rowe (2012) states that leachate dams require a suitable ballast layer, such as a thick inorganic granular layer, over their liner systems for **Equation 2-1** to be applicable.

2.5.3 Acceptable Risk of Spillage

The sizing of a leachate dam is regulated by the *Regulations on Use of Water for Mining and Related Activities Aimed at the Protection of Water Resources* (2010). The

regulation requires that dirty water systems be designed, constructed, operated, and maintained in such a manner that they are not likely to spill more than once in 50 years – the regulation defines a “dirty” area as an area that causes, or is likely to cause the pollution of a water resource (Department of Water Affairs and Forestry, 2010); thus, leachate dams are considered dirty water systems. This requirement is accomplished by conducting a site water balance for the leachate dam using climactic data that spans more than 50 years to ensure that it will not spill more than once in the 50 years, however full it is at that point in time (Department of Water Affairs and Forestry, 2006).

2.6 Predicting Leachate Discharge

This section of the literature review gives a background on the Hydrologic Evaluation of Landfill Performance model (HELP Model), details the inputs required for the model, and explains the model’s process for computing daily leachate discharge. This section also discusses the studies conducted on the saturated hydraulic conductivity of landfill wastes.

2.6.1 The HELP Model

The HELP model is a hydrological model that predicts the movement of water into, through, and out of landfills. The model was developed by the “U.S. Army Corps of Engineers’ Waterways Experiment Station” for the “U.S. Environmental Protection Agency (EPA)”. Many United States governmental agencies require the use of the HELP model for landfill design evaluation and regulatory licencing. Additionally, the model is used in more than a dozen countries, with more than 50 universities using the model for training and research (Tolaymat & Krause, 2020). The model is suggested for use in South Africa by the “Minimum Requirements for Waste Disposal by Landfill” (1998).

In order to use the model to compute the leachate discharge out of a landfill’s waste disposal area for use in a daily water balance, it requires: (1) a historical daily rainfall record of the landfill; (2) a historic daily record of the solar radiation at the landfill;

(3) other climactic parameters, including humidity, temperature, and wind speed; (4) hydrological parameters of the waste disposed of at the landfill, including the waste's curve number, total porosity, field capacity, wilting point, and saturated hydraulic conductivity; and (5) the disposed of waste's evaporative zone depth (Tolaymat & Krause, 2020).

The model uses the historical daily rainfall record, along with the curve number of the waste disposed of, to compute the daily infiltration into the waste. Historical daily rainfall records from rainfall stations across South Africa can be acquired from the Daily Rainfall Extraction Utility (DREU) software, developed by the “Institute for Commercial Forestry Research (ICFR)”, in conjunction with the School of Bio-resources, Engineering, and Environmental Hydrology (BEEH) at the University of KwaZulu-Natal (Lynch, 2004). The software can be used to obtain daily rainfall data from all rainfall stations across South Africa, from 1900 to 2000.

The model uses the SCS method to calculate infiltration (Tolaymat & Krause, 2020); a curve number (CN) is an index that is used in the SCS method to express a catchment's run-off response to a rainfall event. Specifically, the HELP model requires an initial curve number (CN_{II}) input; CN_{II} considers the hydrological properties and land cover properties of a catchment, with no regard to moisture conditions of the catchment, before a rainfall event (The South African National Roads Agency SOC Limited, 2013). Using CN_{II}, the curve number for “dry” moisture conditions is calculated using **Equation 2-4**.

$$CN_I = 3.751 \times 10^{-1} CN_{II} + 2.751 \times 10^{-3} CN_{II}^2 - 1.639 \times 10^{-5} CN_{II}^3 + 5.143 \times 10^{-7} CN_{II}^4$$

Equation 2-4 (Schroeder, et al., 1994)

The maximum moisture retention parameter (S_{mx}) is then calculated using **Equation 2-5**. The moisture retention parameter (S) is used to take moisture conditions into account when determining run-off and infiltration – the maximum moisture retention

parameter (S_{mx}) is the moisture retention parameter (S) when moisture conditions are dry.

$$S_{mx} = \frac{1000}{CN_I} - 10$$

Equation 2-5 (Schroeder, et al., 1994)

Since moisture is not evenly spread through the waste layers and the moisture near the waste body's surface has a higher influence on infiltration than moisture located deeper in the waste, the moisture retention parameter (S) is weighted according to depth using the evaporative zone depth (EZD) input. The depth-weighted S is computed using **Equation 2-6** and **Equation 2-7**. The EZD is divided into 7 layers, with the thicknesses and weighting factors presented in **Table 2-6**.

$$S = \sum_{j=1}^7 W_j \times S_j$$

Equation 2-6 (Schroeder, et al., 1994)

$$S_j = \begin{cases} S_{mx} \left[1 - \frac{SM_j - \left[\frac{(FC_j + WP_j)}{2} \right]}{UL_j - \left[\frac{(FC_j + WP_j)}{2} \right]} \right], & \text{For } SM_j > \frac{(FC_j + WP_j)}{2} \\ S_{mx}, & \text{For } SM_j \leq \frac{(FC_j + WP_j)}{2} \end{cases}$$

Equation 2-7 (Schroeder, et al., 1994)

Where,

- W_j = Weighting factor for segment j
- S_j = Moisture retention parameter for segment j
- SM_j = Soil water storage in segment j, meters

- UL_j = Storage at saturation in segment j, meters
- FC_j = Storage at field capacity in segment j, meters
- WP_j = storage at wilting point in segment j, meters

Table 2-6: Moisture Retention Parameter Weighting Factors (Schroeder, et al., 1994)

Layer (relative to the surface)	Thickness	Weighting Factor
1 (top segment)	$\frac{1}{36}EZD$	0.111
2	$\frac{5}{36}EZD$	0.397
3	$\frac{1}{6}EZD$	0.254
4	$\frac{1}{6}EZD$	0.127
5	$\frac{1}{6}EZD$	0.063
6	$\frac{1}{6}EZD$	0.032
7	$\frac{1}{6}EZD$	0.016

Once the moisture retention parameter (S) is obtained, run-off (R) is calculated using **Equation 2-8**. Rainfall (P) is measured in meters.

$$R = \frac{(P - 0.2S)^2}{P + 0.8S}$$

Equation 2-8 (Schroeder, et al., 1994)

Assuming that there is no snow cover and no vegetation (bare ground), the infiltration into the waste ($DR(j)$) is computed using Equation 2-9.

$$DR(j) = P - R$$

Equation 2-9 (Schroeder, et al., 1994)

Once the infiltration is computed, the HELP model computes the daily leachate discharge out of a landfill using Equation 2-10 to Equation 2-14.

$$Q_i = DR_i(j + 1) \cdot A = [\Psi_j(\Pi_i)^{\frac{\lambda_j}{3\lambda_j+2}} + \Upsilon_j + DR_{i-1}(j) - DR_i(j) - ET_{i-1}(j) - ET_i(j)] \cdot A$$

Equation 2-10 (Schroeder, et al., 1994)

$$\Psi_j = -2[UL(j) - RS(j)]$$

Equation 2-11 (Schroeder, et al., 1994)

$$\Pi_i = \frac{DR_i(j + 1)}{K_s(j) \cdot i}$$

Equation 2-12 (Schroeder, et al., 1994)

$$\lambda_j = \frac{\text{Log}[SM(j) - RS(j)] - \text{Log}[UL(j) - RS(j)]}{\text{Log}[\psi_b] - \text{Log}[\psi]}$$

Equation 2-13 (Schroeder, et al., 1994)

$$Y_j = 2[SM_{i-1}(j) - RS(j)]$$

Equation 2-14 (Schroeder, et al., 1994)

Where,

- Q_i = Daily leachate discharging out of the landfill (m³/day)
- A = Landfill footprint/plan Area (m²)
- $DR_i(j + 1)$ = Drainage out of landfill waste body to underlying leachate collection layer (m/day)
- $UL(j)$ = Saturated water content/total porosity of waste body (m)
- $RS(j)$ = Residual water content (m)
- $K_s(j)$ = Saturated hydraulic conductivity of waste body (m/day)
- i = Hydraulic gradient of the waste body – assumed to be 1 when the waste body is not saturated (unitless)
- $SM_{i-1}(j)$ = leachate stored in the waste body on the previous day (m)
- $DR_{i-1}(j)$ = Water into landfill waste body on the previous day – i.e., the portion of the previous day's rainfall that infiltrated into the waste body (m/day)
- $DR_i(j + 1)$ = Water into landfill waste body – i.e., the portion of rainfall that infiltrates into the waste body (m/day)
- $ET_{i-1}(j)$ = Evaporation out of the waste body on the previous day (m/day)
- $ET_i(j)$ = Evaporation out of waste body (m/day)
- ψ_b = Capillary pressure (bars). 0.33 bar when $SM(j)$ = Wilting Point and 15 bar when $SM(j)$ = Field Capacity
- ψ = Bubbling pressure (bars)

The model uses the inputted daily global solar radiation, temperature, relative humidity, and wind speed to compute the evaporation ($ET_i(j)$). ψ and λ_j are computed simultaneously using **Equation 2-13** by equating $SM(j)$ to 0.33 bar and then 15 bar. The waste body's leachate storage ($SM(j)$) is calculated using **Equation 2-15**.

$$SM_i(j) = 0.5 \{ [DR_i(j) + DR_{i-1}(j)] - [DR_i(j+1) + DR_{i-1}(j+1)] - [ET_i(j) + ET_{i-1}(j)] \} + SM_{i-1}(j)$$

Equation 2-15 (Schroeder, et al., 1994)

2.6.2 Saturated Hydraulic Conductivity of Landfilled Waste

Schroeder and Peyton (1988) state that hydraulic conductivity is the most important parameter for determining leachate discharge. Experiments to estimate the saturated hydraulic conductivity of municipal waste have been conducted across several landfill facilities around the world. Machado et al. (2010) conducted borehole infiltration testing in two Brazilian landfills and sampled the waste to test for its saturated hydraulic conductivity using triaxial testing; the saturated hydraulic conductivity was found to range between 1×10^{-5} cm/s and 1×10^{-3} cm/s. Reddy et al. (2009) used auger drilling to collect municipal waste samples from an American landfill to test for the waste's saturated hydraulic conductivity using a large-scale rigid wall permeameter, a small-scale rigid wall permeameter, and a small-scale triaxial permeameter, which had diameters of 30 cm, 6.3 cm, and 5 cm, respectively. The saturated hydraulic conductivities were found to range between 0.2 to 7.8×10^{-5} cm/s, 2.8×10^{-3} to 11.8×10^{-3} cm/s, and 10^{-4} to 10^{-6} cm/s for the large-scale rigid wall permeameter, small-scale rigid wall permeameter, and small-scale triaxial permeameter, respectively. Reddy et al. (2009) attributed the large variation in the large-scale rigid wall permeameter's saturated hydraulic conductivity results to increasing the vertical pressure during testing, which lead to a lower void ratio. It is noted that the saturated hydraulic conductivity increases with the size of the testing apparatus; the larger apparatus tests a more representative sample. Zeis and Majors (1993) sampled municipal waste from a Canadian landfill to test for its porosity, field capacity, residual water content, and unsaturated hydraulic conductivity; the saturated hydraulic conductivity was then computed from these values. The testing apparatus used to compute the saturated hydraulic conductivity had a diameter of 59 cm. They reported an apparent saturated hydraulic conductivity that ranged between 200 cm/s and 2×10^2 cm/s.

2.6.3 Alternative Method for Predicting Leachate Discharge

A verified analytical method of predicting leachate discharge is the two-dimensional two-domain approach proposed in Fellner and Brunner (2010). In the method, a landfill is modelled as being composed of highly permeable vertical flow paths surrounded by a less permeable waste mass (Fellner & Brunner, 2010). Fellner and Brunner (2010) simulated the leachate discharge of an existing landfill using the HYDRUS-2D software and the Rosetta software which calculate water flow using Richard's equation and van Genuchten's equation. Fellner and Brunner (2010) state that the method computes predicted leachate discharge rates that are similar to the leachate discharge rates observed; the predicted cumulative leachate discharge over their analysis period was 2394 m³ compared to an observed cumulative leachate discharge of 2251 m³. This results in a percent error of 5.97 %. The advantage of using the HELP model over the method proposed in Fellner and Brunner (2010) is that the HELP model is freeware based in Microsoft Excel thus making it more accessible.

2.7 Predicting Evaporation from the Leachate Dam

Valderrama et al. (2015) presented **Equation 2-16** for determining a saline solution's latent heat of vaporization based on the salinity of the solution and the ambient temperature at the location of the solution. Latent heat of vaporization is defined as the heat required to change a unit mass of a substance from a liquid to a gas.

$$L_s = [a_1 + a_2(T - 273.15) + a_3(T - 273.15)^2 + a_4(T - 273.15)^3 + a_5(T - 273.15)^4] \\ \times \left(1 - \frac{S}{1000}\right)$$

Equation 2-16 (Valderrama, et al., 2015)

Where,

- L_s = Latent heat of vaporisation (kJ/kg)
- T = Temperature (K)
- S = Salinity (g/kg)

- $a_1 = \text{constant} = 2.501 \times 10^6$
- $a_2 = \text{constant} = -2.369 \times 10^3$
- $a_3 = \text{constant} = 2.678 \times 10^{-1}$
- $a_4 = \text{constant} = -8.103 \times 10^{-3}$
- $a_5 = \text{constant} = -2.079 \times 10^{-5}$

Equation 2-16 is suitable for salinities that range between 0 and 120 g/kg, at temperatures between 0 and 200 °C. The equation has a reported accuracy of $\pm 1\%$ for seawater (Generous, et al., 2020).

Ritchie (1972), as cited in Schroeder et al. (1994), states that the latent heat of vaporization is related to potential evaporation (E_o) according to **Equation 2-17**.

$$E_o = \frac{LE}{25.4L_s}$$

Equation 2-17 (Schroeder, et al., 1994)

Where,

E_o = Potential evaporation (inches)

LE = Energy available for evaporation (langelys)

L_s = Latent heat of vaporisation (langelys per mm)

The subsequent chapters of this research will discuss the use **Equation 2-16** and **Equation 2-17** in developing a method for predicting leachate evaporation for the purpose of leachate dam sizing.

3. METHODOLOGY

This chapter presents: (1) the study area, field data, instruments, and procedures that were used to measure and record leachate discharge and leachate evaporation; (2) the procedures and input parameters that were used in the HELP Model to compute predicted leachate discharge; and (3) the derivation that was used to compute predicted leachate evaporation.

3.1 Description of study area

The data used for verifying the HELP model and the derived leachate evaporation method was recorded from an existing landfill that is located in the South African inland. The landfill cannot be named in this research due to its owners' request that it remain confidential. The landfill is located in a region with a generally dry climate. Summers are warm to hot, with the average daily high temperature being approximately 26 °C, while winters are mild to cold, with the average daily temperature being approximately 18 °C. The mean annual precipitation (MAP) at the landfill facility is approximately 709 mm, with 85% of the annual precipitation falling between the months of October and March (Lynch, 2004). The mean annual evaporation (MAE) of open-water surfaces in the region is between 2000 and 2200 mm (Schulze, et al., 2008).

Geotechnical investigations carried out at the landfill facility show that the site is typically composed of a medium dense silty sand hillwash from the surface to a depth of 1.2 m; a clayey silt nodular ferricrete from a depth of 1.2 m to 2.6 m; and a firm to stiff clayey silt reworked/residual siltstone from a depth of 5.2 m to 25 m. Ground water seepage is encountered at a depth of 4.0 m below ground level.

The landfill in question consists of multiple waste disposal areas (cells) and leachate dams. A single cell and leachate dam was used for this research report; the cell and leachate dam used are hereby referred to as Cell A and Leachate Dam A, respectively.

Cell A and Leachate Dam A were constructed in the early 2010s, as extensions to the landfill. Cell A reached its final landform (FLF) in 2019, remaining significantly unchanged over the course of the collection of the leachate discharge data, while Dam A is currently still in use.

Cell A's liner system consists of a 2 mm thick HDPE geomembrane installed on top of a 600 mm thick clay layer that has been compacted in 150 mm thicknesses. Below the 600 mm clay layer, a 150 mm thick stone layer, embedded with 110 mm perforated HDPE pipes, has been installed to serve as a leakage detection layer. Below the 150 mm thick stone layer, a 300 mm clay layer compacted in 150 mm thick layers has been constructed to prevent potential leakage from seeping into the ground. A subsoil drainage system, embedded with 110 mm perforated HDPE pipes, has been constructed below the 300 mm thick clay layer to prevent groundwater from pushing up against the liner system. Above the liner system, a leachate collection system has been constructed. The collection system consists of a 300 mm thick stone layer with 315 mm diameter perforated HDPE pipes embedded in the stone, spaced at horizontal lengths of between 44 m and 66 m. The stone and HDPE pipes slope at approximately 4 % to 4.5 % towards a low point. Two HDPE solid pipes (315 mm diameter) are connected to the perforated pipes at the low point. The solid outlet pipes penetrate through the 2 mm HDPE geomembrane and gravitationally discharge into a lined concrete sump; the penetration through the geomembrane was designed and constructed to be watertight. From the lined concrete sump, leachate is continuously pumped to Leachate Dam A. A layer of protection sand is placed below the 300 mm stone layer to protect the geomembrane from damage. Separation geotextiles are installed between the different soil and stone layers to act as filters. A cross-section of the liner system and leachate collection system is shown in **Figure 3-1**.

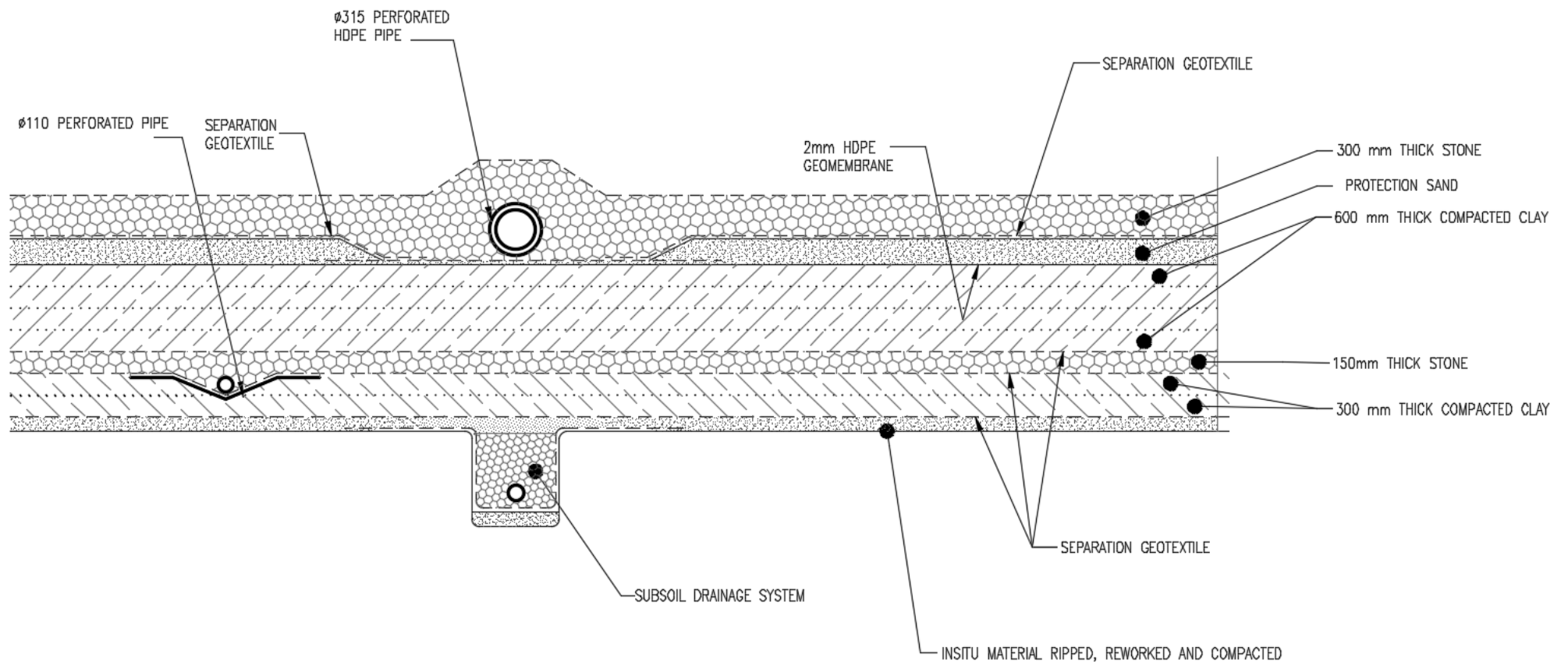


Figure 3-1: Cell A's Liner System

3.2 Instruments and Procedures for Data Gathering

This section describes the instruments and procedures that were used to record field data that was essential as inputs into the HELP model and the derived leachate evaporation method. The data gathered was also used for verification of the model and method; the recorded leachate discharge and leachate evaporation were compared to the predicted values from the model and method.

3.2.1 Observed Leachate Discharge and Leachate Evaporation

Leachate discharge was recorded manually at the landfill every afternoon (excluding public holidays) from 2019 to 2023. The discharge was measured using electronic flow metres connected to Cell A's two 315 mm solid outlet pipes, as they gravitationally drained into Cell A's concrete lined sump. 3.8 years of continuous cumulative daily leachate discharge was recorded over the period that Cell A remained unchanged.

Leachate evaporation was measured and recorded every afternoon (excluding public holidays) at the landfill using A-pans, under conditions recommended by the World Meteorological Organization (2008). If the A-pan's level was found to have dropped by more than 20 mm from its reference level (50 mm), it would be refilled back to the reference level using leachate from Leachate Dam A. Rainfall was also measured at the landfill so that it would be taken into account when computing the evaporation. 2.7 years of continuous daily leachate evaporation was recorded.

3.2.2 Rainfall, Evaporation, and Other Climatic Data

Rainfall and water evaporation were also measured and recorded at the landfill. These records are essential inputs to the HELP model and derived leachate evaporation method. Rainfall was measured daily using conical rain gauges, while water evaporation was measured using A-pans, under conditions recommended by World Meteorological Organization (2008). The MAP of the measured rainfall was 814.7 mm; the percent error between the measured MAP and the MAP reported by Lynch

(2004) is 14.9%. The measured MAE falls with the range reported in Schulze et al. (2008) when converted to lake evaporation using Bosman (1990) and Midgely et al. (1994).

Other climatic data required, such as wind speed and temperature, was provided by the website, *Time and Date AS*. The website accuracy was accessed and confirmed by comparing its average daily temperatures provided to the daily temperature records of a weather station that is located approximately 28 km from the landfill site and lies within the same quaternary catchment as the landfill, according to Midgley et al. (1994). The weather station's daily temperature records were taken at irregular intervals from 1973 to 2020, with every month having around 1 300 days of recorded data (the minimum being February, with 1 278 days recorded). The average per cent difference between the two data sets was found to be 5.06 %.

3.2.3 Composition of Leachate

Nine leachate samples from Dam A were collected and tested for total dissolved solids (TDS) and electrical conductivity (EC), over nine years; this data is to be used in computing predicted leachate evaporation. The samples were tested by a third-party laboratory. The TDS and EC results are shown in **Table 3-1**. The TDS to EC ratio (f-value) was calculated and is also shown **Table 3-1**. The f-values ranged from 0.683 to 1.146. This range falls within the range that is reported by Hubert and Wolkersdorfer (2015), which is between 0.25 and 1.34 for 45 South African mine-water samples; thus, the TDS and EC results are deemed to be realistic.

Table 3-1: TDS and EC Sampled from Leachate Dam A

Sample No.	Date Sampled	TDS (ppm)	EC (µS/cm)	f-value
1	16/07/2014	99 579	102 200	0.974
2	13/07/2015	87 425	115 800	0.755

3	17/07/2016	79 890	117 130	0.682
4	10/07/2017	92 241	135 130	0.682
5	05/07/2018	121 260	120 750	1.004
6	10/07/2019	159 600	139 250	1.146
7	06/07/2020	89 700	135 330	0.663
8	06/07/2021	139 360	123 020	1.133
9	05/07/2022	87 925	118 780	0.740

The nine samples collected from Leachate Dam A were also tested for pH and COD; the results are presented in **Table 3-2**. These results are presented in this research in order to compare the typical range of concentrations of contaminants found in leachate, presented in **Table 2-2**, to Leachate Dam A's leachate; this comparison is important when discussing the derived leachate evaporation method's suitability when applied to different landfills.

Table 3-2: pH and COD of Leachate Dam A

Sample No.	Date Sampled	pH	COD as O ₂ (mg/l)
1	16/07/2014	8.1	16 252
2	13/07/2015	4.9	15 677
3	17/07/2016	7.5	15 571
4	10/07/2017	7.5	18 115
5	05/07/2018	8.4	17 030

6	10/07/2019	8.1	25 819
7	06/07/2020	8.2	25 247
8	06/07/2021	8.2	18 000
9	05/07/2022	8.3	14 331

3.3 Predicting Leachate Discharge Using HELP Model

The cumulative predicted leachate discharge computed on the HELP 4.0 software was compared to the cumulative recorded leachate discharge. The predicted leachate discharge was computed using rainfall and evaporation that was recorded over the same period as the observed leachate discharge. The observed leachate discharge records had 119 non-consecutive days missing due to errors in recording and public holidays – this amounts to a reliability of 91.6 %. The days on which records had been missing were excluded from both the cumulative observed and predicted discharge. The cumulative predicted leachate discharge was computed for all of HELP 4.0's standard waste types to find the waste type that computed predicted leachate discharge values that most closely matched the recorded leachate discharge; each of the model's waste types has a unique saturated hydraulic conductivity, porosity, field capacity, residual water content (expressed, as wilting point). Hydraulic conductivity is the most important parameter for determining leachate discharge that is to be used for the purposes of a water balance (Schroeder & Peyton, 1988); thus, an analysis was carried out on the closest matching waste type to determine the apparent saturated hydraulic conductivity (K_s) that led to an even closer match. The match between the observed and predicted leachate discharge was evaluated by graphical comparison, as well as through the use of the coefficient of determination (r^2).

The input parameters used in the HELP model computations are presented in **Appendix A**. However, two parameters, solar radiation and latitude, are crucial to this research and are highlighted in this section.

3.3.1 Solar Radiation and Latitude

In HELP 4.0, the daily solar radiation input required is the daily global horizontal radiation (R_{si}) at the landfill (Tolaymat & Krause, 2020). The R_{si} is defined as the radiation of the sun on a surface area that is horizontal to the surface of the earth. HELP uses this input to simulate evaporation through the landfill (The World Bank Group, 2023). Daily A-pan evaporation was measured at the landfill, in lieu of solar radiation. The R_{si} was then manually computed in Microsoft Excel using the following formulae from Ritchie (1972), as cited in Schroeder et al. (1994):

$$E_{oi} = \frac{LE_i}{L_v}$$

Equation 3-1 (Schroeder, et al., 1994)

$$L_v = 59.7 - T_{ci}$$

Equation 3-2 (Schroeder, et al., 1994)

$$LE_i = PENR_i + PENA_i$$

Equation 3-3 (Schroeder, et al., 1994)

$$PENR_i = \frac{\Delta_i}{\Delta_i + \gamma} R_{ni}$$

Equation 3-4 (Schroeder, et al., 1994)

$$PENA_i = \frac{15.36\gamma}{\Delta_i + \gamma} (1 + 0.1488^u)(e_{oi} - e_{ai})$$

Equation 3-5 (Schroeder, et al., 1994)

$$\Delta_i = 1.9993[(0.00738T_{ci} + 0.8072)^7 - 0.0005793]$$

Equation 3-6 (Schroeder, et al., 1994)

$$e_{oi} = 33.8639[(0.00738T_{ci} + 0.8072)^8 - (0.000019)|1.8T_{ci} + 48| + 0.001316]$$

Equation 3-7 (Schroeder, et al., 1994)

$$e_{ai} = 33.8639 \times RH[(0.00738T_{ci} + 0.8072)^8 - (0.000019)|1.8T_{ci} + 48| + 0.001316]$$

Equation 3-8 (Schroeder, et al., 1994)

$$R_{ni} = (1 - \alpha)R_{si} - R_{bi}$$

Equation 3-9 (Schroeder, et al., 1994)

$$R_{bi} = R_{boi} \left[a_i \left(\frac{R_{si}}{R_{soi}} \right) - b_i \right]$$

Equation 3-10 (Schroeder, et al., 1994)

$$R_{boi} = 1.171 \times 10^{-7} (T_{ci} + 273)^4 (0.39 - 0.05\sqrt{e_{ai}})$$

Equation 3-11 (Schroeder, et al., 1994)

$$R_{soi} = 711.38DD_i[(H_i \times \sin|LAT| \times \sin SD_i)(\sin H_i \times \cos|LAT| \times \cos SD_i)]$$

Equation 3-12 (Schroeder, et al., 1994)

$$DD_i = 1 + \{0.0335 \sin[0.0172(J_i + 88.2)]\}$$

Equation 3-13 (Schroeder, et al., 1994)

$$SD_i = 0.4102 \sin[0.0172(J_i - 80.25)]$$

Equation 3-14 (Schroeder, et al., 1994)

$$H_i = \arccos[(-\tan|LAT|)(\tan SD_i)]$$

Equation 3-15 (Schroeder, et al., 1994)

Where,

- E_{oi} = Potential evaporation on day i (mm)
- LE_i = Energy available on day i for potential evaporation
- L_v = Latent heat of vaporization for evaporation water (langleys/mm)
- T_{ci} = mean air temperature on day i ($^{\circ}\text{C}$)
- $PENR_i$ = Radiative component of potential evaporation on day i (langleys)
- $PENA_i$ = Aerodynamic component of the potential evaporation on day i (langleys)
- Δ_i = The saturation vapour pressure slope curve at mean air temperature on day i ($\text{mm}/^{\circ}\text{C}$)
- γ = Wet and dry bulb psychrometer equation constant – assumed to be 0.68 mbar/ $^{\circ}\text{C}$ by HELP 4.0
- u = wind speed at a height of 2 metres – assumed to be annual wind speed by HELP 4.0 (km/h)
- e_{oi} = Saturation vapour pressure, on day i, at mean air temperature (mbar)
- e_{ai} = Mean vapour pressure, on day i, of the atmosphere (mbar)
- RH = quarterly average relative humidity on days without rain and equal to one on days with rain.
- R_{ni} = net solar radiation received by the surface on day i (langleys)
- α = albedo (reflectivity coefficient of the surface) – HELP 4.0 uses a value of 0.23 if there is no snow present.
- R_{si} = global horizontal radiation on day i (langleys)
- R_{bi} = long-wave radiation flux from soil on day i (langleys)
- R_{boi} = maximum outgoing long-wave radiation on day i (langleys) – assuming a clear day
- R_{soi} = maximum potential global solar radiation on day i (langleys)

- a_i and b_i = constants that are dependent on humidity on day i (unitless). For $RH < 50\%$, $a_i = 1.2$ and $b_i = 0.2$. For $50\% \leq RH < 75\%$, $a_i = 1.1$ and $b_i = 0.1$. For $RH \geq 75\%$, $a_i = 1.0$ and $b_i = 0$
- J_i = The Julian date for day i when in the northern hemisphere and the Julian date minus 182.5 when in the southern hemisphere
- LAT = Latitude of location (radians)

HELP 4.0 uses an albedo (α) value of 0.23 – this value falls within the range of the albedo for grasslands (Bralower & Bice, 2024). Therefore, the recorded A-pan evaporation readings were converted to “false grassland” readings by applying the monthly factors presented in Midgley et al. (1994). A table of the conversion is presented in **Appendix B**.

The latitude of location input proved to be a challenge for this research. South Africa stretches from -22° to -35° latitude and is therefore out of the bounds of HELP 4.0, which can only accept a latitude input that is between -14.32° to 71.25° . To use HELP 4.0, the latitude where the landfill is located was changed to a positive value for input. The latitude value only affects HELP 4.0’s calculation of the evaporation through the landfill’s surface; outside of frozen soil conditions, the only times that latitude is used in the “HELP Model Engineering Documentation” (1994), which details the computation procedures of the HELP model, is in **Equation 3-12** and **Equation 3-15**. The HELP model considers a soil to have entered frozen soil conditions when the average temperature over the previous 30 days is less than 0°C (Schroeder, et al., 1994). For Cell A, the temperature never drops to less than 0°C ; thus, it is considered never to have frozen soil conditions. Therefore, the latitude input’s sign change was insignificant due to the daily solar radiation (R_{si}) being manually calculated; the manual calculation counter-acted the sign change by not subtracting 182.5 from the Julian date (j_i) in **Equation 3-13** and **Equation 3-14** (see j_i ’s definition above).

To access and confirm the accuracy of the solar radiation manual calculations, and hence the latitude input modification, the computed average daily R_{si} was compared

to the average daily R_{si} from Global Solar Atlas, a web-based solar radiation, air temperature, and PV power simulation model developed by SolarGIS on behalf of the “World Bank Group” and the “International Finance Corporation”. Global Solar Atlas calculates solar resource parameters, such as R_{si} , using data inputs obtained from geostationary satellites and meteorological models (The World Bank Group, 2023). When comparing the computed average daily R_{si} to the average daily R_{si} from the Global Solar Atlas, the percentage difference was found to be 2.78 %.

3.4 Predicting Leachate Evaporation

The cumulative predicted leachate evaporation was computed using the derived method presented in this section. The predicted leachate evaporation was then compared to the cumulative observed leachate evaporation that was recorded over the same period that the predicted leachate was computed. Over the recorded 968 days, the observed evaporation had 66 non-consecutive days missing due to errors in recording, public holidays, and extreme weather events – this amounts to a reliability of 90.5 %. The days on which records had been missing were excluded from both the cumulative observed and predicted evaporation. The comparison between the observed and predicted leachate evaporation was evaluated by graphical comparison, as well as through the use of the coefficient of determination (r^2).

3.4.1 Derived Method for Predicting Leachate Evaporation

This method uses water evaporation typically recorded at a landfill, or from a nearby weather station, and the salinity of the leachate at that landfill in order to predict leachate evaporation. For this research, water evaporation was measured at the landfill. Ritchie (1972), as cited in Schroeder et al. (1994) proposed **Equation 2-17** to calculate evaporation. When the water evaporation is known, the energy available for evaporation (LE) required for **Equation 2-17** can be calculated using **Equation 3-1** by multiplying the observed water evaporation by the latent heat of vaporization of water. The latent heat of vaporization of water is presented in **Equation 3-16**.

$$L_v = 59.7 - 0.0564T_{ci}$$

Where,

- T_{ci} = Mean air temperature on day i (°C)

Equation 3-16 (Schroeder, et al., 1994)

Valderrama et al. (2015) proposed **Equation 2-16** to determine the latent heat of vaporisation of a saline solution, based on its salinity and the ambient temperature. **Equation 2-17** and **Equation 2-16** were used to develop this research's equation for predicting leachate evaporation, presented as **Equation 3-17**.

$$E_{oi} = \frac{41966 \times LE_i}{[a_1 + a_2T_i + a_3T_i^2 + a_4T_i^3 + a_5T_i^4] \times \left(1 - \frac{S}{1000}\right)}$$

Equation 3-17

Where,

- E_{oi} = Potential evaporation (mm) on day i (mm)
- LE_i = Energy available for evaporation for day i (mm)
- T_i = Temperature on day i (°C)
- S = Salinity (g/kg)
- a_1 = constant = 2.501×10^6
- a_2 = constant = -2.369×10^3
- a_3 = constant = 2.678×10^{-1}
- a_4 = constant = -8.103×10^{-3}
- a_5 = constant = -2.079×10^{-5}

Four of the nine leachate samples, discussed in **Section 3.2.3**, have f-values that are close to one – these samples have f-values that range between 0.974 and 1.132. For

these samples, it is assumed that the difference between TDS and salinity (S) is negligible. Thus, these samples were used to predict the leachate evaporation over the analysis period using **Equation 3-17**. The equation given by Valderrama et al. (2015) is only applicable to solutions with salinities between 0 g/kg and 120 g/kg. The average and median TDS for the four leachate samples used are 129.95 g/kg and 130.31 g/kg, respectively; thus, the results of the predicted leachate evaporation may be negatively affected. The average TDS was used to compute the predicted leachate evaporation.

4. RESULTS AND DISCUSSIONS

4.1 Introduction

Amongst the objectives of this research was the analysis of the suitability of the Hydrologic Evaluation of Landfill Performance (HELP) model and the derived leachate evaporation method in predicting leachate discharge and leachate evaporation for the purpose of water balance modelling in order to meet South African regulations on leachate dam design and sizing. This was achieved by comparing the leachate discharge and leachate evaporation computed by the model and method with the leachate discharge and leachate evaporation recorded at an existing landfill over more than 3 years and 2 years, respectively. In this chapter, the results of these comparisons are presented and discussed. Details of the methodology are presented in **Chapter 3**. The computed data of the model and method are presented in **Appendix C**.

4.2 Predicting Leachate Discharge

Figure 4-1 presents the observed monthly precipitation and leachate discharge depth over the period of analysis. The observed monthly precipitation was computed by summing the daily precipitation within a month, excluding days when leachate discharge was not recorded, as discussed in **Section 3.3**. The observed monthly leachate discharge depth was computed by summing the daily leachate discharge depths within a month. The observed leachate discharge depth (mm/day) was calculated from the observed leachate discharge (m³/d) using **Equation 4-1**.

$$O_i = \frac{1000 \times Q_i}{LA}$$

Equation 4-1

Where,

- O_i = Observed leachate discharge depth on day i (mm/day)
- Q_i = Observed leachate discharge on day i (m³/day)
- LA = Landfill Footprint Area (m²)

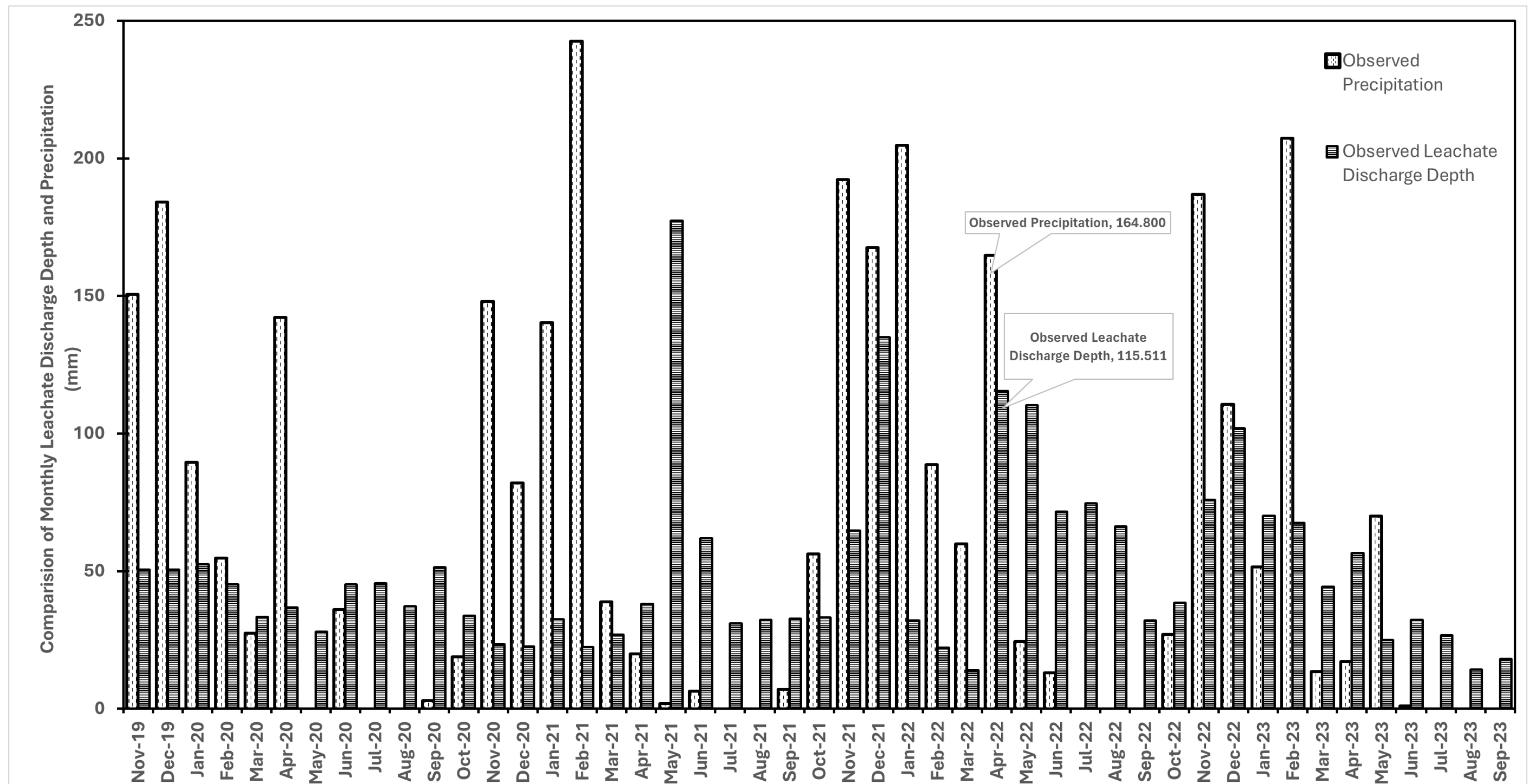


Figure 4-1: Monthly Precipitation and Leachate Discharge Depth Recorded at the Landfill Facility

A visual analysis of **Figure 4-1** shows that the monthly precipitation varies significantly between wet seasons and dry seasons, while the monthly leachate discharge depth is generally constant. Over the analysis period, the average monthly precipitation during the wet and dry seasons was calculated to be 110.6 mm and 21.1 mm, respectively. The average monthly leachate discharge depths during the wet and dry seasons were calculated to be 47.5 mm and 52.6 mm. Thus, the average monthly leachate discharge depth is 42.9 % of the average precipitation during the wet season and 2.5 times greater than the average precipitation during the dry season.

Figure 4-2 presents a comparison of the observed leachate discharge depth to what was predicted by the HELP 4.0 software, using the standard waste types found in the software; the figure shows the first seven months of the analysis period. In the analyses presented in the figure, the initial moisture content of the wastes was input as each waste type's field capacity as this is the most likely state of Cell A after six years of operation and exposure to weather.

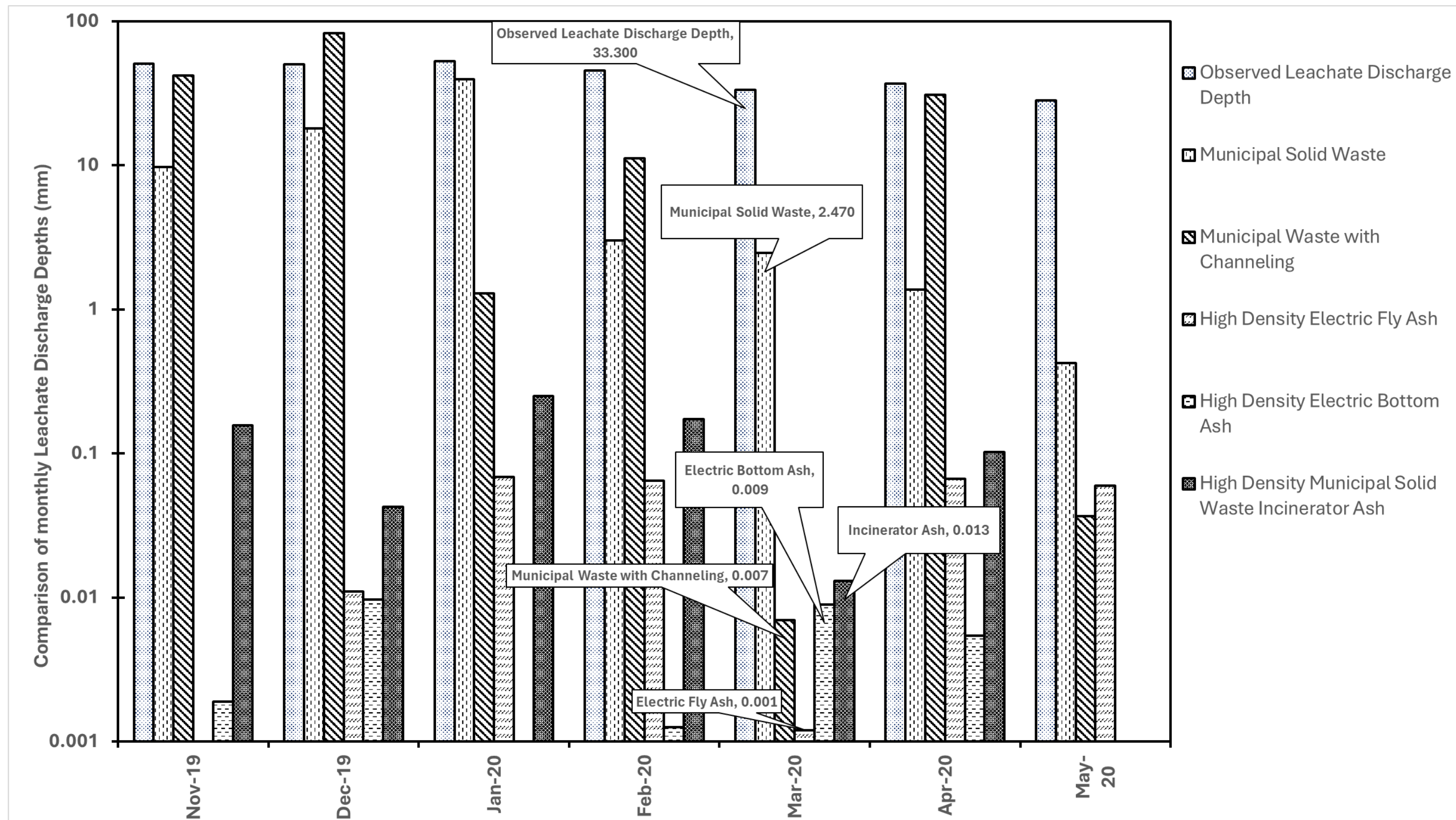


Figure 4-2: Monthly Observed and Predicted Leachate Discharge Depth Over First Seven Months

The monthly average of the observed leachate discharge depth for the first seven months of the analysis period was computed to be 42.433 mm/month. Over the first seven months, the monthly averages of the predicted leachate discharge depths using “municipal waste”, “municipal waste with channelling”, “electric fly ash”, “electric bottom ash”, and “incinerator ash” were 10.627 mm/month, 23.976 mm/month, 0.039 mm/month, 0.039 mm/month, and 0.105 mm/month, respectively. “Municipal waste and “municipal waste with channelling” had the closest averages to the observed leachate discharge depth. Thus, **Figure 4-3** presents a comparison of the cumulative observed leachate discharge depth and the predicted leachate discharge depths from the “municipal waste” and “municipal waste with channelling” standards waste types. The cumulative comparisons are reported because the Rippl mass curve analysis is commonly used to size dams and reservoirs; thus, cumulative comparisons are used to assess the suitability of the HELP model.

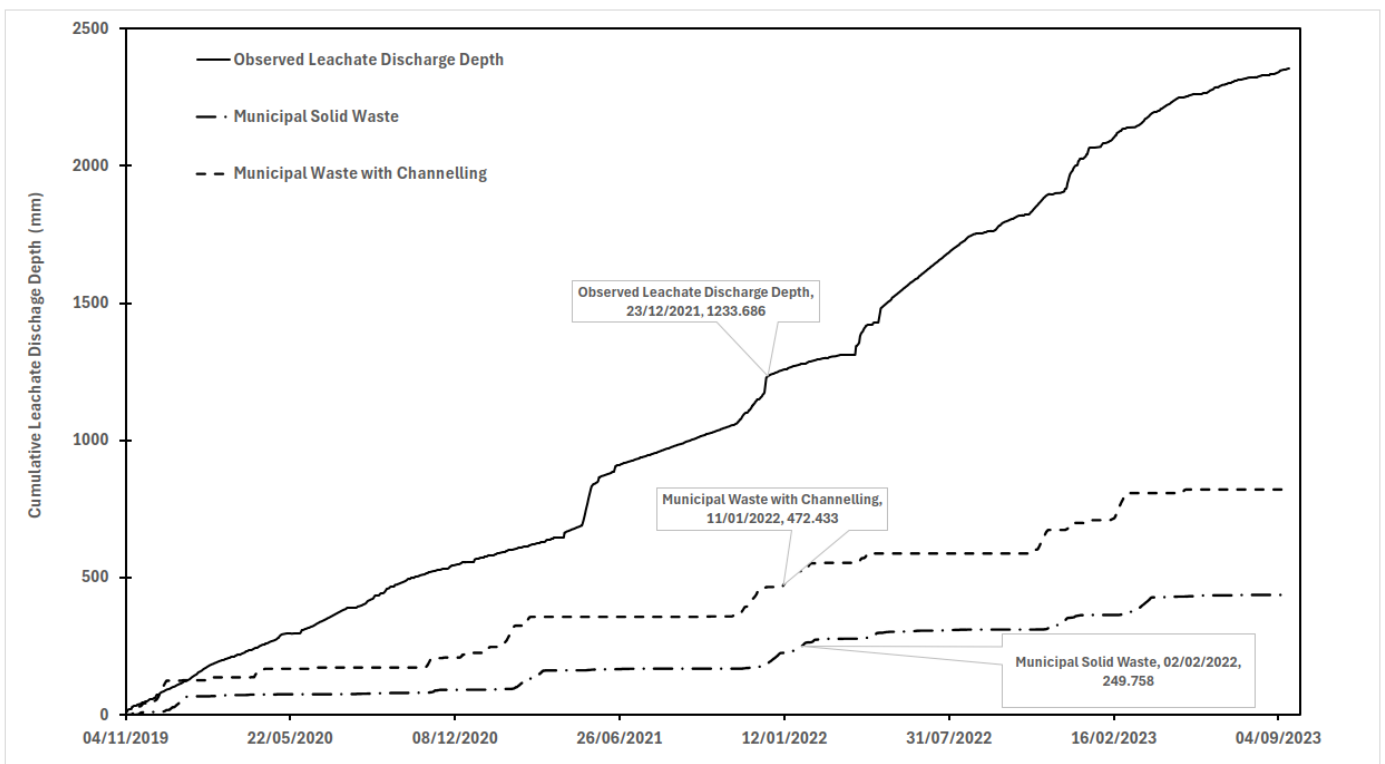


Figure 4-3: Observed Cumulative Leachate Discharge Depth Compared to the Predicted Cumulative Leachate Discharge Depths of Municipal Waste and Municipal Waste with Channelling

A visual analysis of **Figure 4-3** shows that “municipal waste” and “municipal waste with channelling” underestimate the leachate discharge depth. However, it is also seen that the two standard waste types have cumulative leachate discharge depth graphs that are similar in shape to the observed cumulative leachate discharge depth graph. The match between the observed and predicted leachate discharge depth was evaluated using the coefficient of determination (r^2), as well as the gradient (b) of the line of best fit along which the coefficient of determination is based. The range of r^2 falls between zero and one; for a strong correlation between the observed and predicted leachate discharge depths, r^2 must be close to one. For a strong agreement, b must also be close to one. A stronger agreement indicates a greater match between the observed and predicted leachate discharge. Microsoft Excel was used to obtain r^2 and b. The line of best fit, r^2 , and b between the observed cumulative discharge depth and the predicted cumulative discharge depth for the two standard waste types are shown **Figure 4-4**.

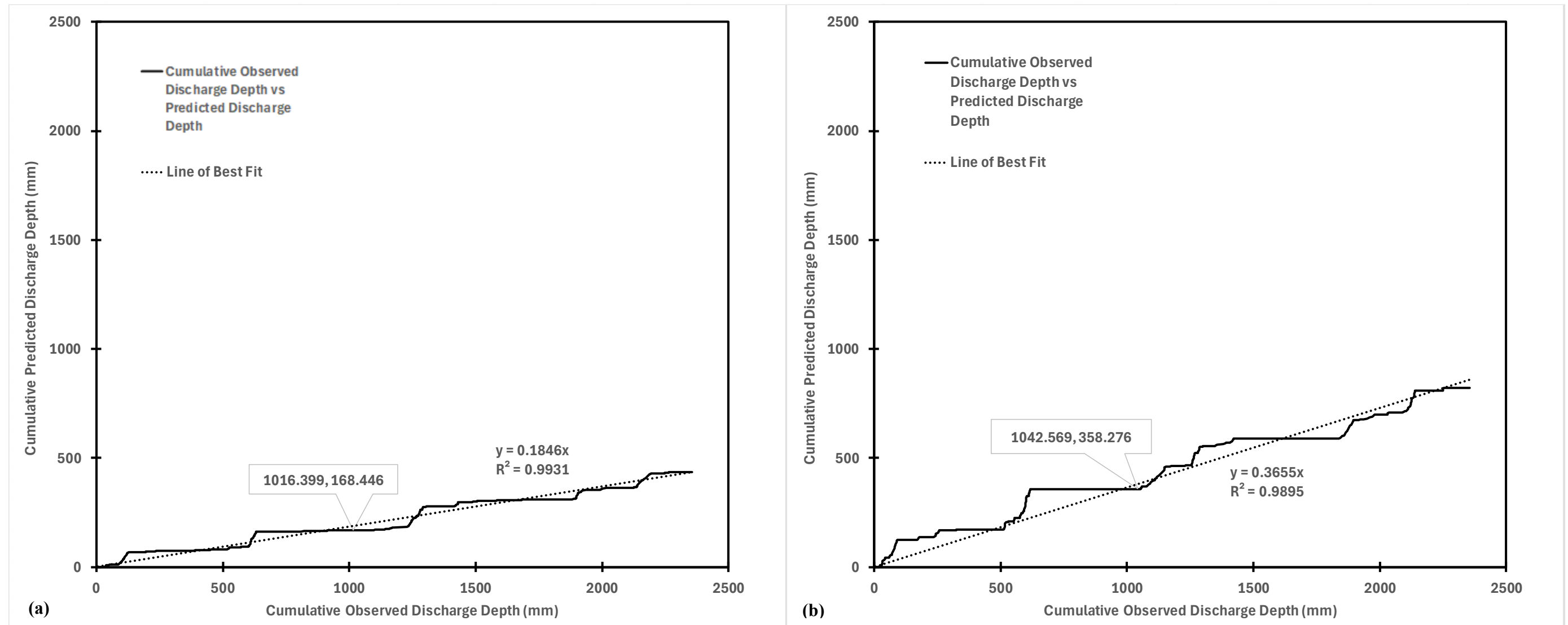


Figure 4-4: Cumulative Observed Discharge Depth Vs Predicted Discharge Depth for: (a) Municipal Waste; (b) Municipal Waste with Channelling

The intercept of the lines of best fit was set to zero for both of the graphs shown in **Figure 4-4**. The graphs show that the two standard waste types compute predicted cumulative discharge depths that have strong correlations with the observed cumulative discharge depths, with r^2 values of 0.993 and 0.990 for “municipal waste” and “municipal waste with channelling”, respectively. “Municipal waste with channelling”, with $b = 0.366$, has a stronger agreement than “municipal waste”, with $b = 0.185$; however, both still have a weak agreement and significantly underestimate the leachate discharge depth.

Given its stronger agreement, “municipal waste with channelling” was used to carry out the analysis for determining the K_s that would lead to the closest match between predicted leachate discharge and the observed leachate discharge; all of the standard waste type’s parameters were used, apart from K_s . The minimum K_s that leads to a strong agreement was found to be 5×10^4 cm/s; The line of best fit, r^2 , and b between the observed cumulative discharge depth and the predicted cumulative discharge depth is shown in **Figure 4-5**.

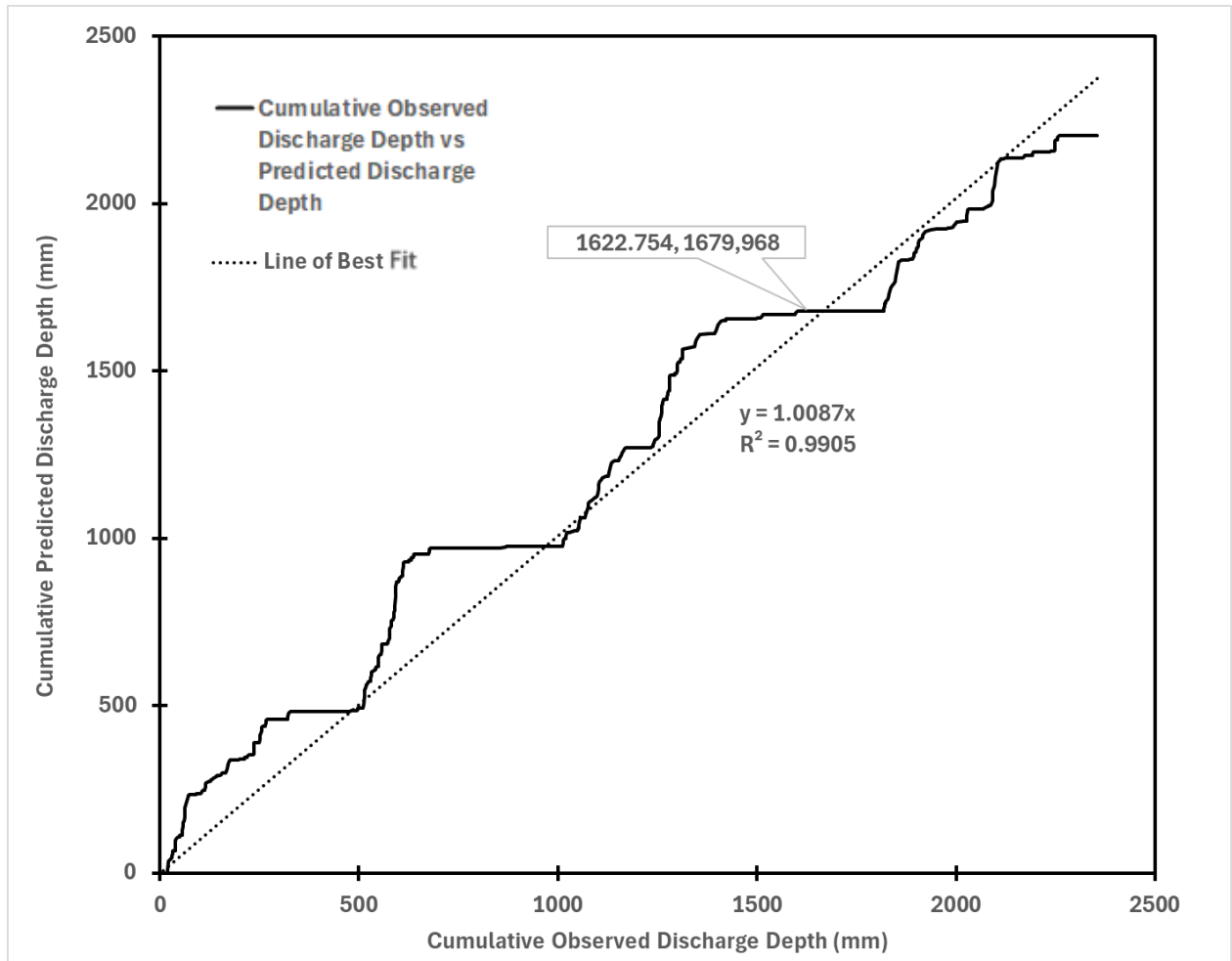


Figure 4-5: Cumulative Observed Discharge Vs Cumulative Predicted Discharge Using Municipal Waste with Channelling with an Adjusted $K_s = 5 \times 10^4$ cm/s

The line of best fit of the graph in **Figure 4-5** has $r^2 = 0.991$ and $b = 1.009$. This indicates that there is a strong agreement between the cumulative observed and predicted discharge depths. The minimum K_s required is 10^2 times greater than the highest K_s stated in the literature. **Figure 4-6** presents the predicted monthly leachate discharge depth (when $K_s = 5 \times 10^4$ cm/s) in comparison to the observed monthly leachate discharge depth over the entire analysis period.

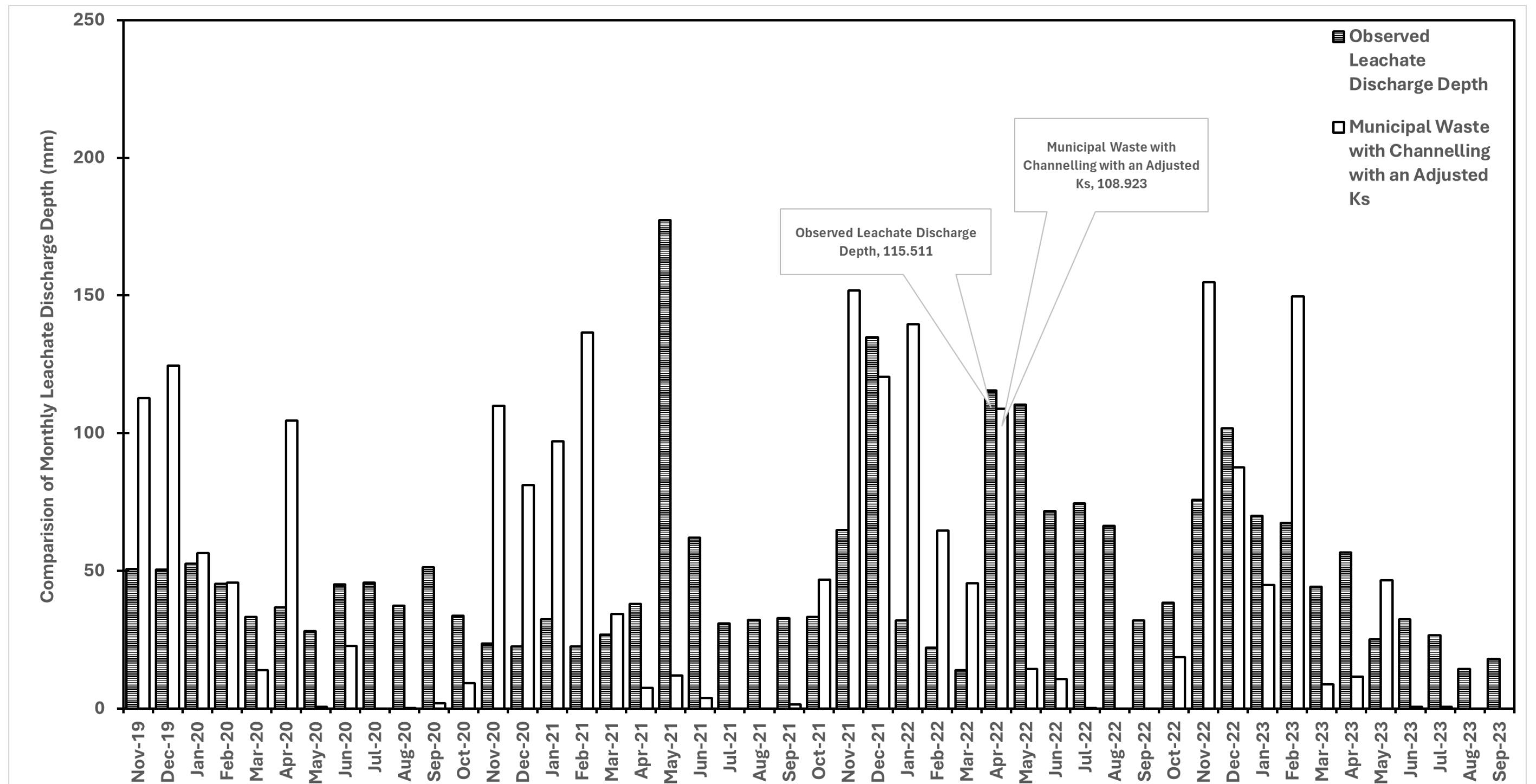


Figure 4-6: Monthly Predicted Leachate Discharge Depth and Observed Leachate Discharge Depth

The average monthly observed and predicted leachate discharge depths were very similar, calculated to be 50.1 mm/month and 46.8 mm/month, respectively. This indicates its suitability for sizing leachate dams. The predicted cumulative leachate discharge depth over the entire analysis period was 2203.6 mm compared to an observed cumulative leachate discharge depth of 2251 mm. This results in a percent error of 6.4 %. This percent error is slightly higher than reported in Fellner and Brunner (2010), which was 5.97%. **Figure 4-7** presents the monthly absolute percent error between the predicted leachate depth and the observed leachate depth for the first seven months of the analysis period.

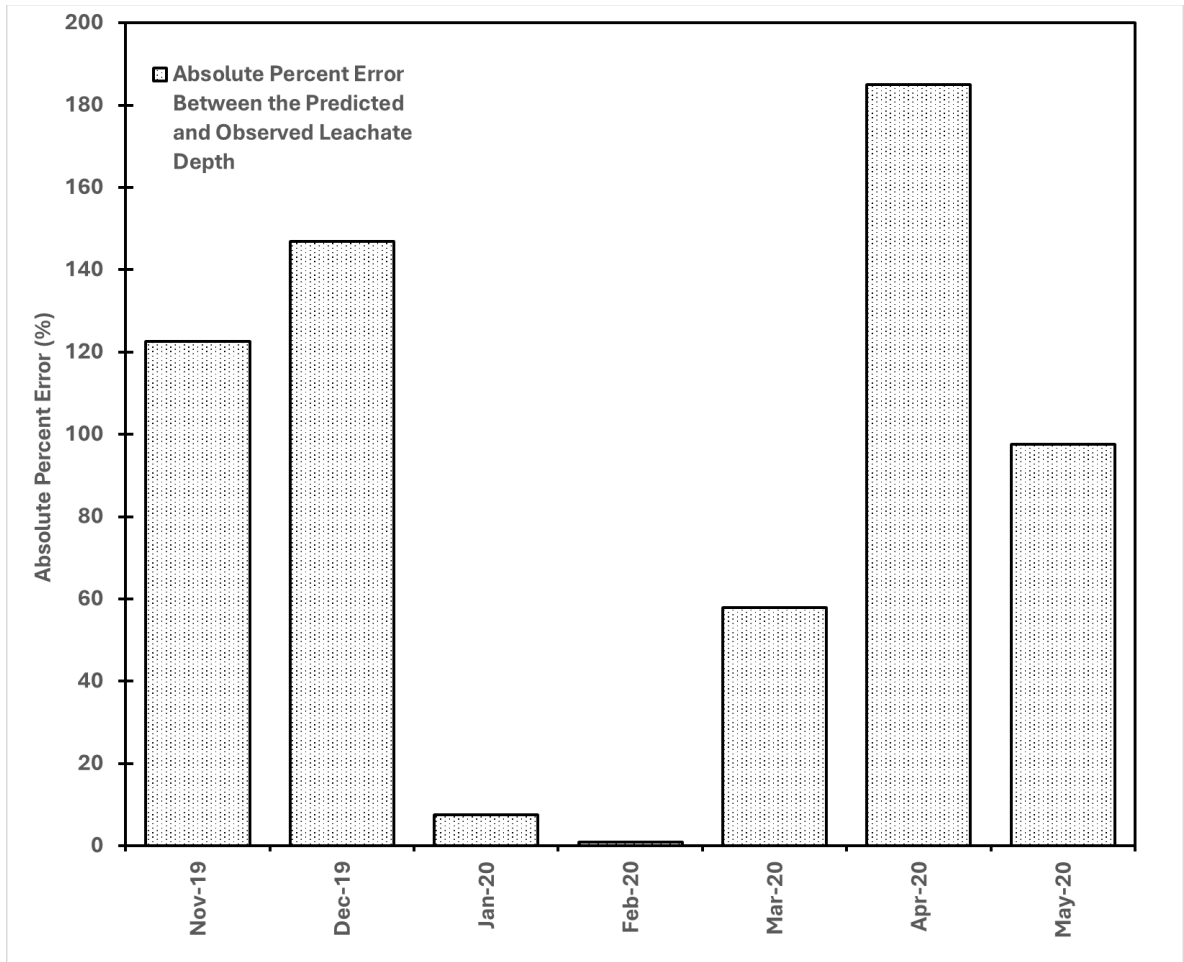
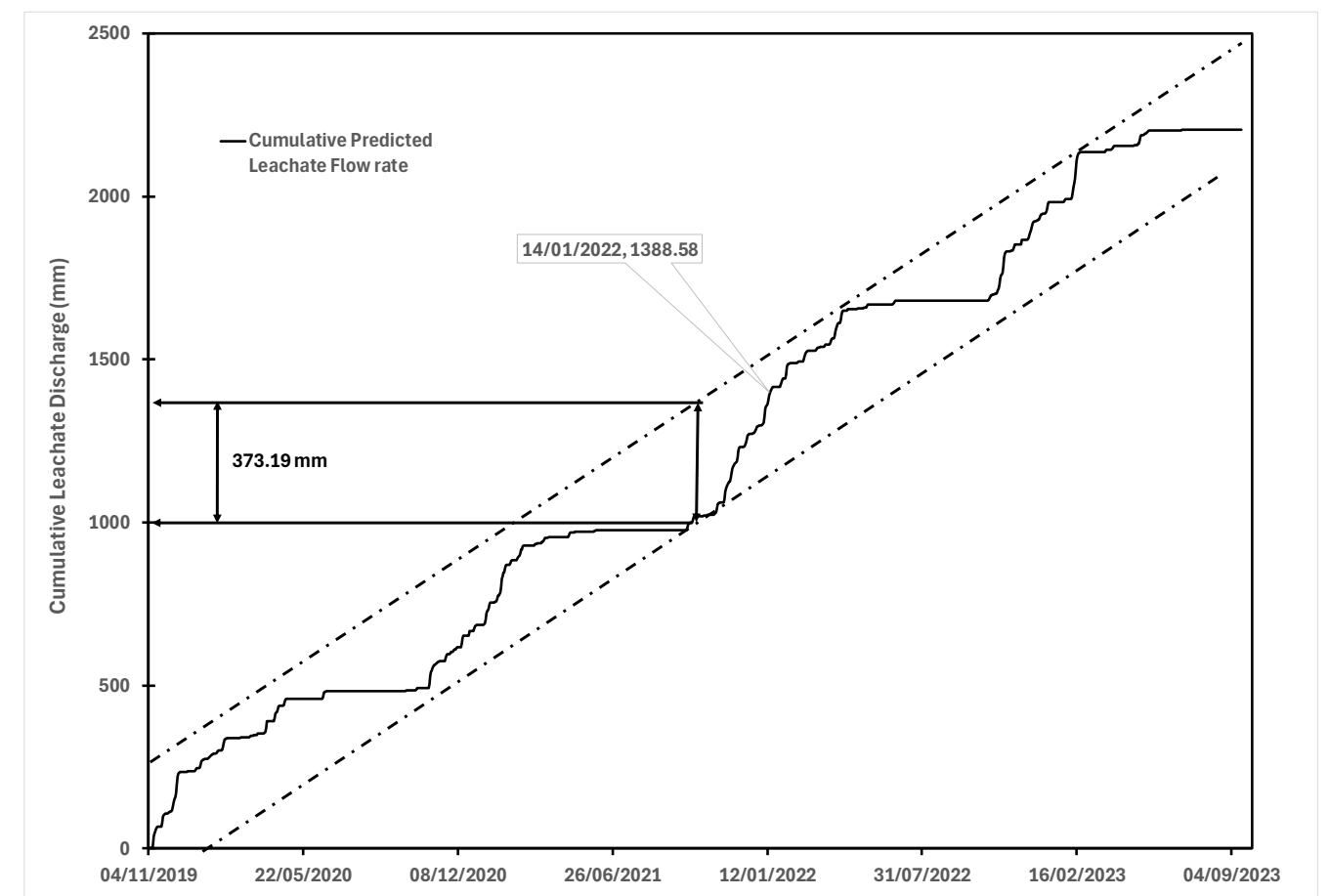
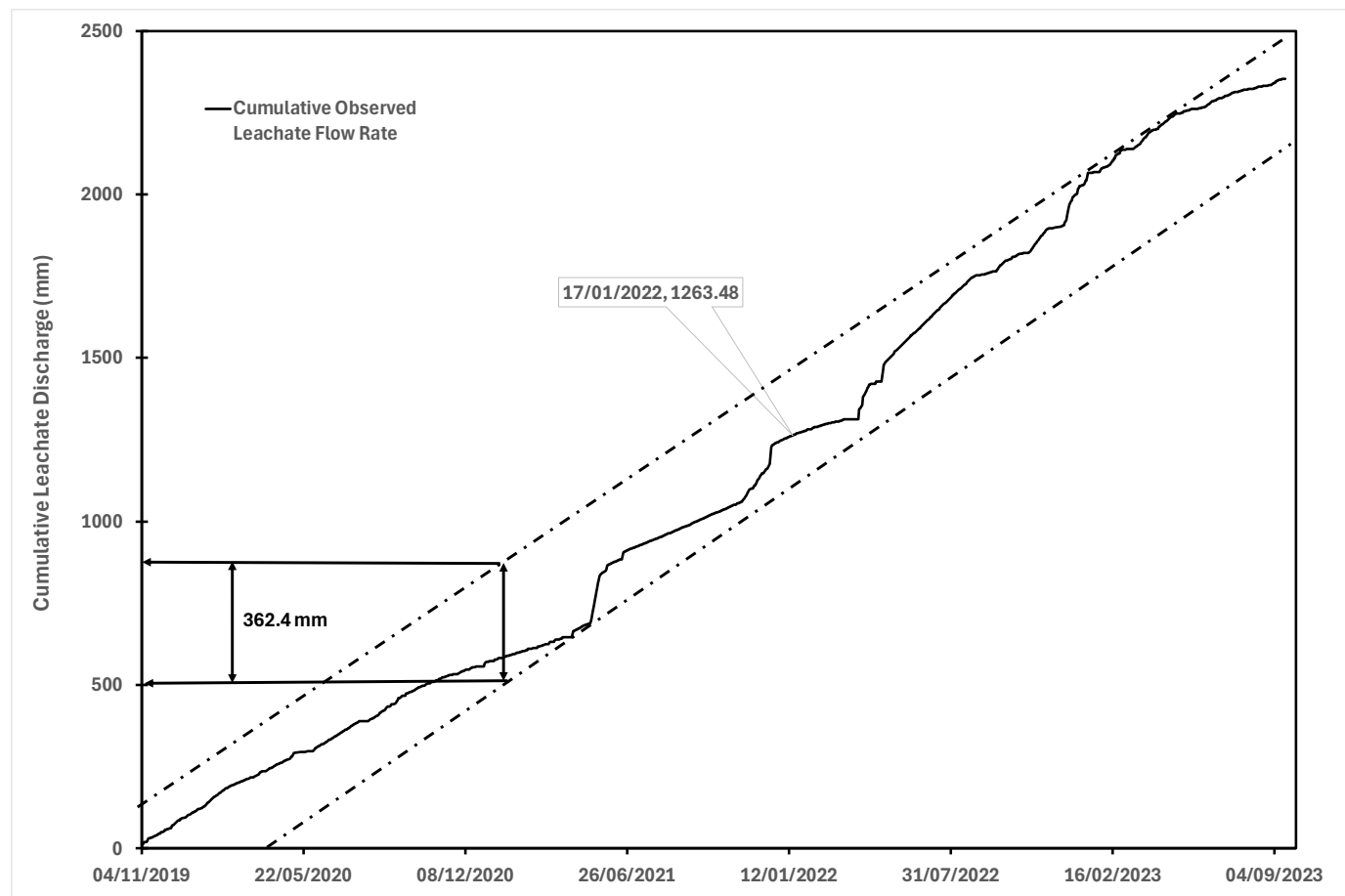


Figure 4-7: Absolute Percent Error Between Predicted and Observed Leachate Discharge.

Figure 4-7 shows that although the average monthly and total cumulative observed and predicted leachate discharges are similar, the month-to-month observed and predicted leachate discharges differ substantially. This indicates that the HELP model may not be suitable for computing monthly leachate discharges. However, given that the purpose of this research is investigate the HELP model's suitability for dam sizing, the Rippl mass curve method was applied. The Rippl mass curve method was used to compare the storage capacity required for the observed and predicted leachate discharge (when $K_s = 5 \times 10^4$ cm/s); this assumes that all other water balance components, such as evaporation and treatment rate, are equal for both. The Rippl mass curve diagrams for the observed leachate discharge depth and the predicted leachate discharge depth are presented in **Figure 4-8**.



(a)

(b)

Figure 4-8: Rippl Mass Curve Diagram for: (a) Observed Leachate Discharge and (b) Predicted Leachate Discharge

The storage depth required for the observed discharge depth was found to be 362.4 mm; this results in a required storage volume of 10 607.6 m³. The storage depth required for the predicted discharge depth was found to be 373.2 mm; this results in a storage volume of 10 923.3 m³. Thus, the percent error between the volume required for the observed discharge and the predicted discharge is 2.976 %. Furthermore, the average daily observed and predicted leachate discharge depths, obtained from each graph's line of best fit, are 1.769 mm/day and 1.596 mm/day, respectively. This results in average daily observed and predicted leachate discharge rates of 51.8 m³/day and 46.7 m³/day. Thus, the percent error between the observed average daily discharge rate and the predicted average daily discharge rate is 9.780 %. The actual leachate dam storage capacity at the landfill is approximately 99 000 m³, which caters for multiple cells that have combined footprint of approximately 240 000 m². Applying the predicted leachate discharge depth to the combined footprint results in a required storage volume of 89 568 m³. This indicates that the existing leachate dams have been oversized by 9 432 m³. It should be noted, however, that this result is based on an analysis period that is too short to compare to the capacity of existing dams since South African regulations require that dams be sized using an analysis period of over 50 years, as discussed in **Section 2.5.3**.

Sensitivity analyses were conducted to determine the influence of the curve number and initial moisture content of the waste on the correlation of predicted discharge depths, computed for “municipal waste with channelling”. The concepts of curve number and initial moisture content are discussed in **Section 2.6.1**. The results of the curve number sensitivity analysis are presented in **Table 4-1**.

Table 4-1: Curve Number Sensitivity Analysis using Municipal Waste with Channelling

Curve Number	Line of Best Fit	
	Coefficient of Determination (r^2)	Gradient (b)
75 (curve number used for this report)	0.990	0.366
20	0.990	0.370
40	0.990	0.370
60	0.990	0.369
80	0.990	0.348
95	0.984	0.100

From **Table 4-1**, it can be seen that decreasing the curve number, while keeping all other parameters the same, has a negligible effect on the predicted discharge depth. Although increasing the curve number would lead to an r^2 that is closer to one, it would also lead to a b that further from one, thus leading to a worse agreement. The results of the sensitivity analysis conducted for the initial moisture content of the waste are presented in **Table 4-2**.

Table 4-2: Waste Initial Moisture Content Sensitivity Analysis using Municipal Waste with Channelling

Initial Moisture Content in the Waste	Line of Best Fit	
	Coefficient of Determination (r^2)	Gradient (b)
0.073 (Initial Moisture Content used for this report)	0.990	0.366
0.092	0.922	0.601
0.111	0.871	0.840
0.130	1.096	0.836
0.149	0.812	1.360
0.168	0.795	0.795

It was found that increasing the initial moisture content significantly distorts the relationship between the cumulative observed and predicted discharge depths – **Figure 4-9** This is illustrated when the initial moisture content is 0.130.

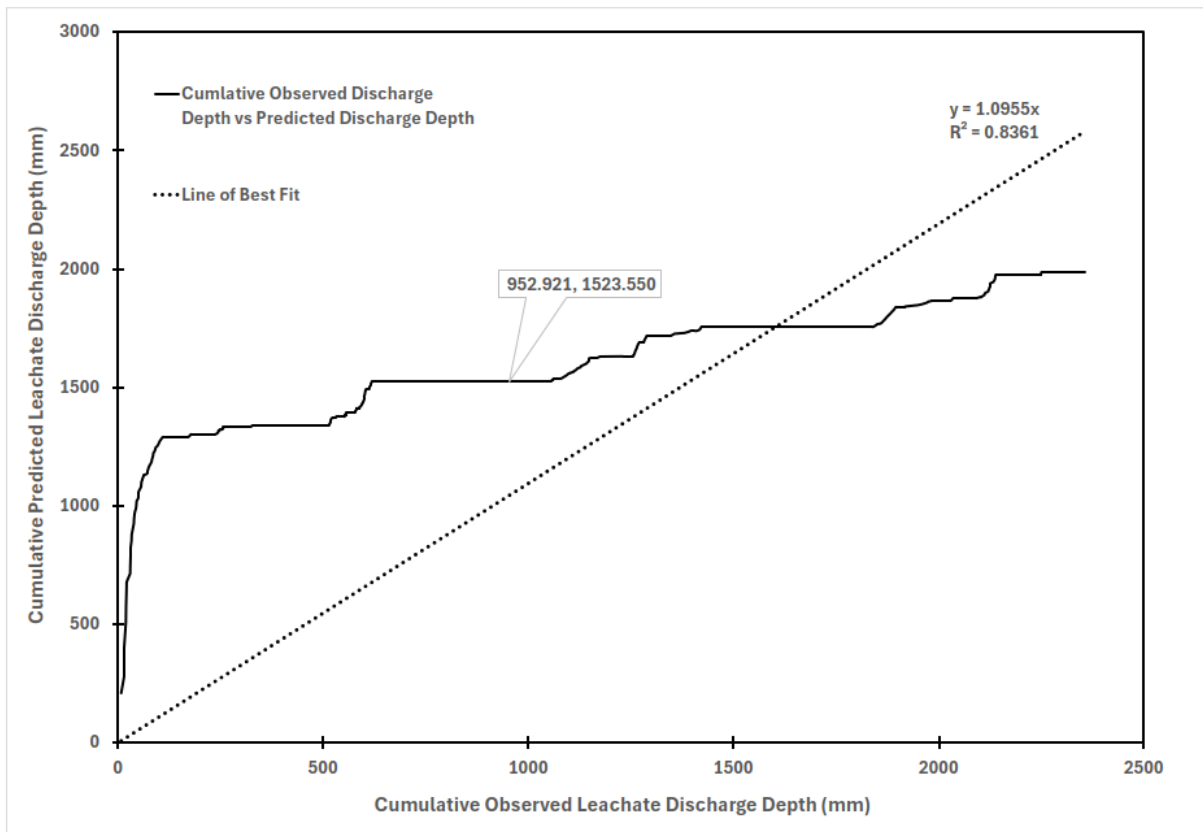


Figure 4-9: Cumulative Observed Discharge Depth Vs Predicted Discharge Depth using Municipal Waste with Channelling with an Adjusted Initial Moisture Content = 0.130

4.3 Predicting Leachate Evaporation

Water evaporation at the landfill facility, along with Dam A's leachate evaporation, TDS, and EC, were recorded and used to analyse the accuracy of the method that was derived in this research for predicting leachate evaporation, as seen in **Section 3.4**. The method was analysed by comparing the predicted leachate evaporation to the observed leachate evaporation from Dam A. **Figure 4-10** presents the observed monthly water and observed leachate evaporation, measured from two separate A-pans.

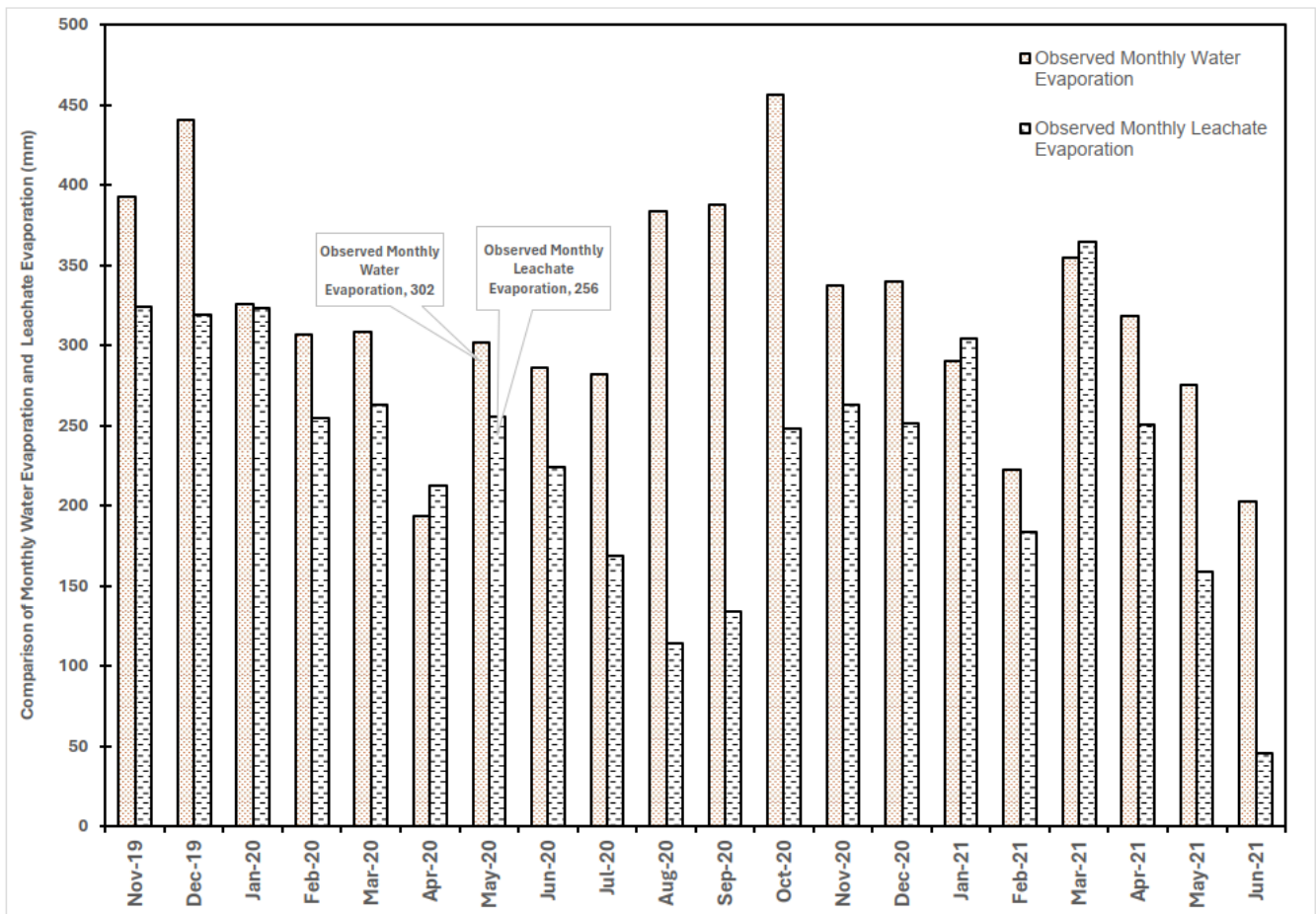


Figure 4-10: Monthly Water and Leachate Evaporation Recorded at the Landfill Facility

Figure 4-10 shows that water evaporation is generally higher than leachate evaporation, although anomalies to this trend are seen in April 2020, January 2021, and March 2022, where leachate evaporation is very similar to water evaporation albeit slightly higher. These anomalies are unusual because leachate generally evaporates at a slower rate than water due to the TDS in the leachate; this leachate in particular as an average EC of 123 000 $\mu\text{S}/\text{cm}$, as presented in **Table 3-1**, compared SANS 241's required EC of less than 12 000 $\mu\text{S}/\text{cm}$ for potable water. However, since the anomalies occurred in months that are in the wet season, it is possible that excessive rainfall could have fallen into the evaporation pans and diluted the leachate to a TDS similar to water – the evaporation pans' leachate/water depth is generally kept at around 50 mm; rainfall data shows that the average monthly rainfall in January, March, and April at the landfill is 117.9 mm, 84.3 mm, and 42.3 mm, respectively. The relationship between the cumulative water evaporation and the cumulative leachate evaporation over time is presented in **Figure 4-11**.

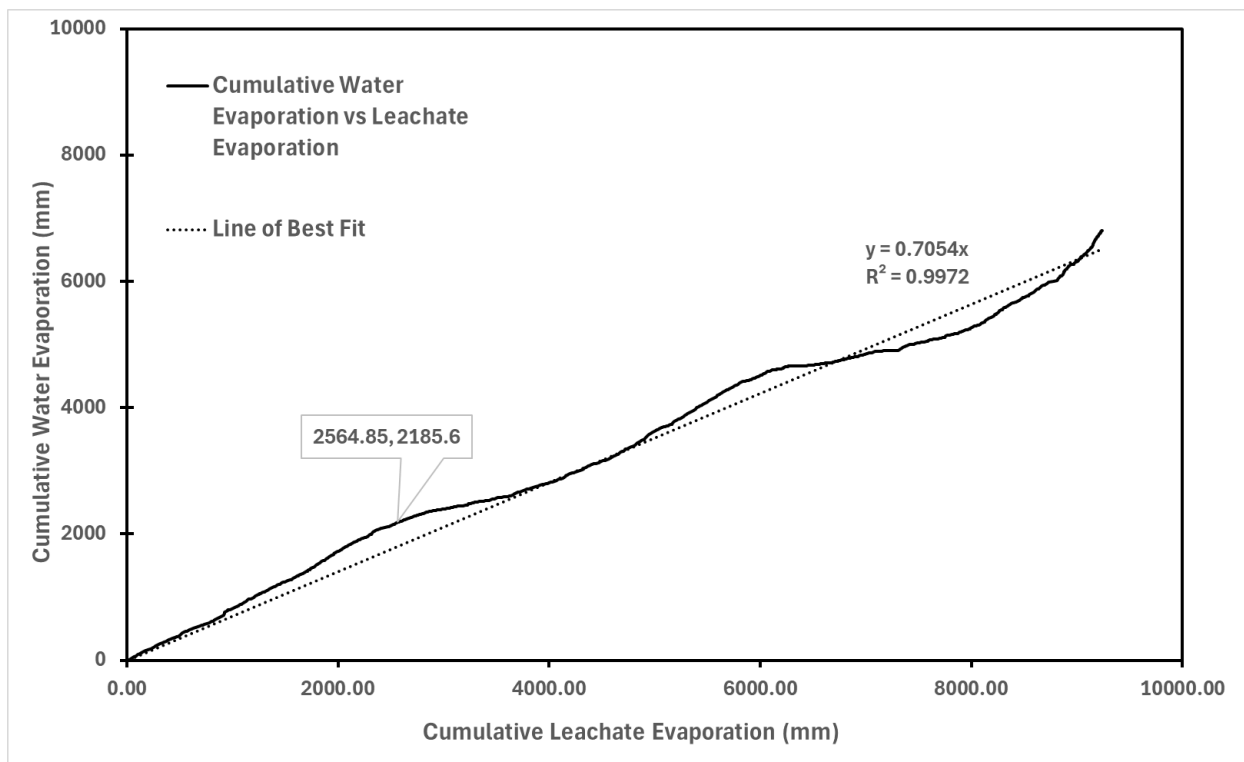


Figure 4-11: Cumulative Water Evaporation vs. Leachate Evaporation

In **Figure 4-11**, the line of best fit has an r^2 that is close to one ($r^2 = 0.997$) – this indicates a very strong correlation between water evaporation and leachate evaporation. The line of best fit of the graph has a function of $f(x) = 0.705x$, which indicates that leachate evaporation is generally equal to 70.5 % of water evaporation – i.e., leachate evaporation is generally 29.5 % less than water evaporation.

In order to assess the suitability of the derived method for predicting leachate evaporation, the cumulative comparisons are investigated because the Rippl mass curve analysis is commonly used to size dams and reservoirs. Thus, **Figure 4-12** presents a comparison of the cumulative observed leachate evaporation to what was predicted.

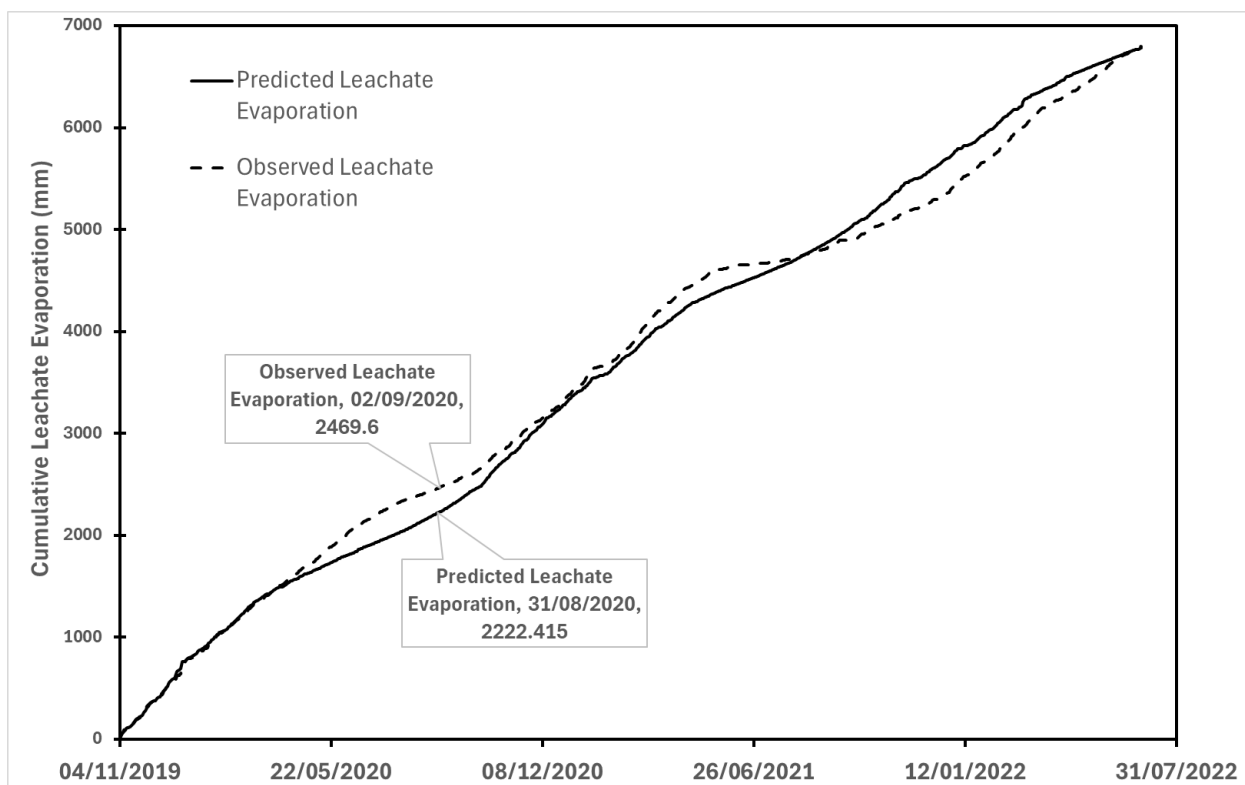


Figure 4-12: Observed Cumulative Leachate Evaporation Compared to the Predicted Cumulative Leachate Evaporation

A visual analysis of **Figure 4-12** shows that the predicted leachate evaporation is very similar to the observed leachate evaporation. The line of best fit, r^2 , and b between the

observed cumulative leachate evaporation and the predicted cumulative leachate evaporation were computed using Microsoft Excel and are shown in **Figure 4-13**.

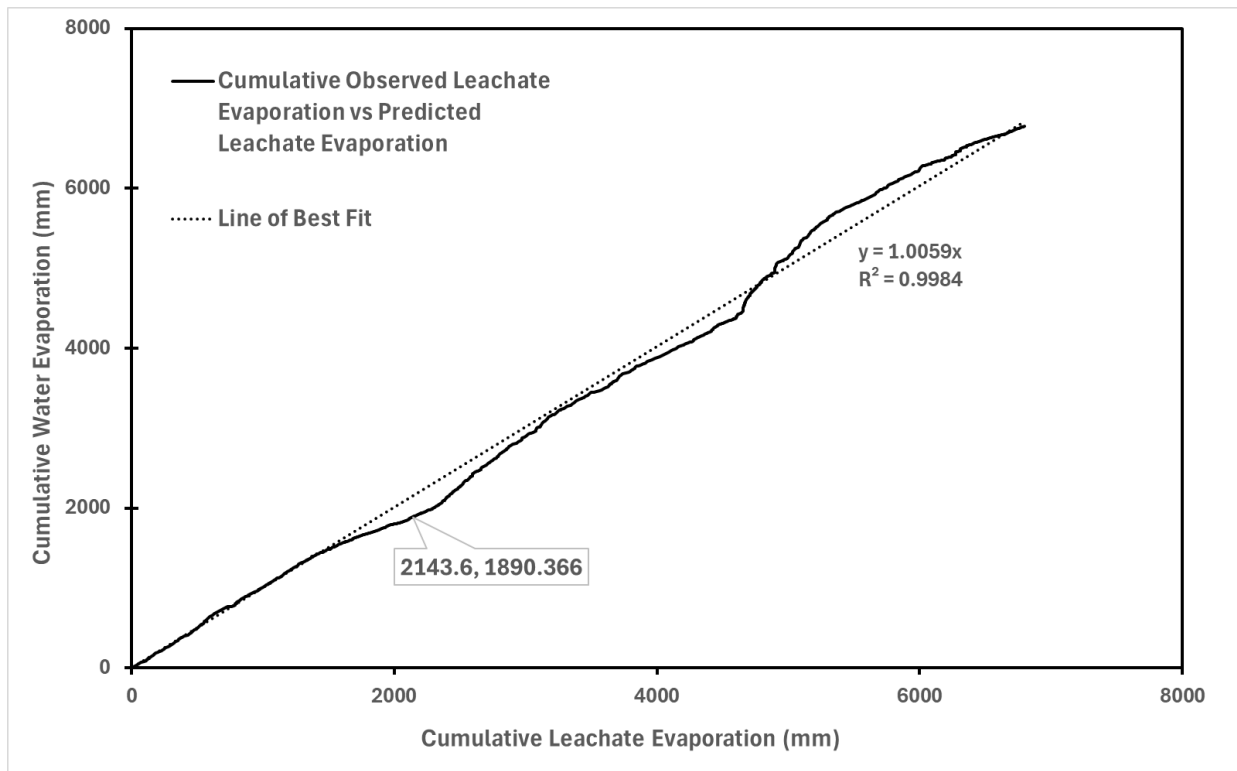


Figure 4-13: Relationship Between the Observed and Predicted Leachate Evaporation Over Time

In **Figure 4-13**, the line of best fit of the graph has an r^2 and a b that are close to one – $r^2 = 0.998$ and $b = 1.006$. This indicates a very strong agreement between the cumulative observed leachate evaporation and the predicted leachate evaporation. **Figure 4-14** presents the monthly predicted monthly leachate evaporation, in comparison to the observed monthly water and leachate evaporation.

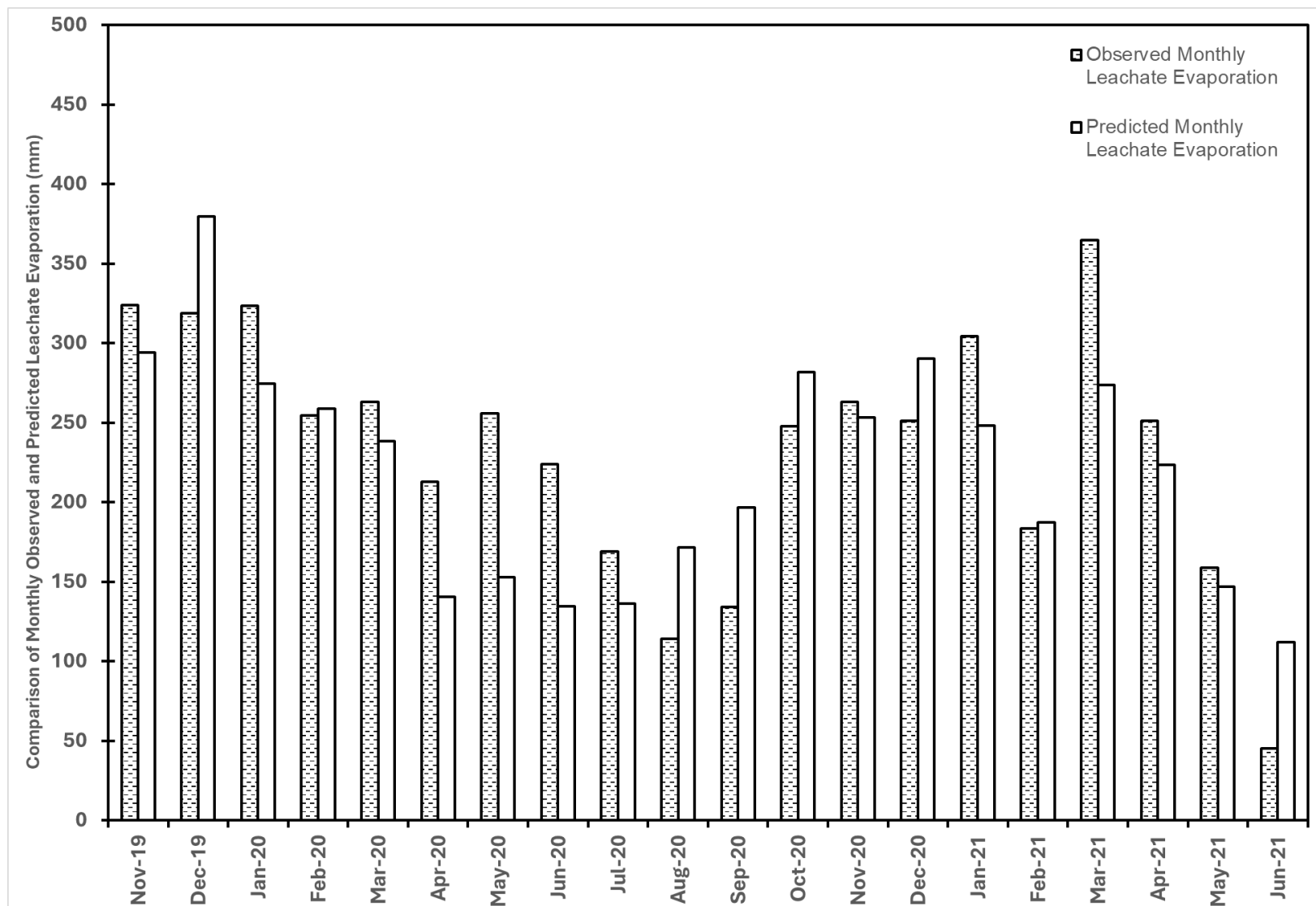


Figure 4-14: Monthly Predicted Leachate Evaporation and Observed Water and Leachate Evaporation

Figure 4-14 shows a general agreement between the predicted and observed leachate evaporation, with a slight tendency of the predicted leachate evaporation to underestimate the leachate evaporation – the percent error between the predicted leachate evaporation and the observed leachate evaporation was calculated to be 7.91 %. The figure shows that the predicted leachate evaporation is more accurate in the wet season, between October and March than in the dry season – the percent error between the predicted leachate evaporation and the observed leachate evaporation during the wet season was calculated to be 4.81 %, as opposed to a percent error of 11.70 % during the dry season.

4.4 Discussion

This section discusses the suitability of predicting leachate discharge and leachate evaporation using the HELP model and derived leachate evaporation method, respectively, for the purpose of leachate dam sizing water balances, based on the comparisons made between predicted and observed leachate discharge and evaporation values for Cell A and Dam A.

4.4.1 Suitability of the HELP Model

The HELP model's predicted leachate discharge tends to strongly correlate to the observed leachate discharge when using standard waste types available in the HELP 4.0 software. However, the model underestimated the leachate discharge when using the standard waste types.

The model's predicted leachate discharge is very sensitive to the waste's saturated hydraulic conductivity. Thus, it was found that increasing the waste's apparent saturated hydraulic conductivity led to the predicted leachate discharge correctly estimating the leachate discharge, while still having a very strong correlation to the observed leachate discharge. The waste's apparent saturated hydraulic conductivity was increased to a value 10^2 times greater than the highest K_s stated in literature, in order to correctly estimate the leachate discharge. An apparent saturated hydraulic conductivity that is so much higher can be explained by the fact that, in the literature,

particle sizes larger than the size of the testing apparatus were discarded – larger particles create larger void ratios and flow paths in the waste, which generally means having a higher apparent saturated hydraulic conductivity. As an example, the particle size of the waste tested by Zeiss and Majors (1993), who recorded the highest apparent saturated hydraulic conductivity in the literature, ranged between 2.9 and 15.3 cm – this is much smaller than the particle sizes of the waste in Cell A (which comprises steel drums, metal containers, etc.). Therefore, it is reasonable to conclude that the K_s for Cell A's waste is higher than the K_s reported by Zeiss and Majors (1993). Hydraulic conductivity laboratory tests were not conducted in this research because the results would not have been representative; large objects, as steel drums and metal containers, would be too large for standard laboratory test apparatus. Further studies are required to determine the apparent saturated hydraulic conductivity applicable to landfills in which large pieces of solid waste are disposed.

After adjusting the saturated hydraulic conductivity, The HELP model's suitability in sizing leachate dams was based on the Rippl mass curve method; the percent error between the volume required for the observed leachate discharge and the predicted leachate discharge is 2.976 %. Furthermore, the percent error between the observed and predicted average daily leachate discharge is 9.780 %. Based on these results it can be concluded that, if an appropriate apparent saturated hydraulic conductivity is selected, the HELP model is suitable for use in leachate dam water balance modelling in order to design for South Africa's regulatory requirements – i.e., to ensure that yearly leachate leakage rates through a dam's liner system do not go beyond the maximum leakage rates stated in the "Minimum requirements for waste Disposal By Landfill" (1998) and to ensure that the dam is not probable to spill more than once every 50 years (Department of Water Affairs and Forestry, 2010). The latter requirement may be achieved using the HELP model when the continuous daily rainfall and evaporation/solar radiation inputs span greater than 50 years. However, the period over which the observed leachate, rainfall, and evaporation were recorded for this research was relatively short. Therefore, future studies using longer periods of analysis should be conducted in order to verify this conclusion.

4.4.2 Suitability of the Derived method for Leachate Evaporation

The derived method for predicting leachate evaporation computed values that have a very strong agreement with the observed leachate evaporation. The derived method also correctly estimated the leachate to a reasonable degree; the percent error between the predicted and observed leachate evaporation was found to be 7.91 %. The accuracy of the derived method also increased during the wet season, with the percent error between the predicted and observed leachate evaporation decreasing to 4.81 %. This means that the derived method was found to be more accurate during the months when leakage and spillages are more likely to occur. Furthermore, the leachate used for this research had higher TDS and COD values than typical values reported in literature (Christensen, et al., 2000; Wdowczyk & Szymanska-Pulikowska, 2020; Kulikowska, 2012). Therefore, based on these results, it can be concluded that this research's derived method for predicting leachate evaporation is suitable for use in dam sizing water balance modelling in order to design for South Africa's regulatory requirements. It is noted, however, that future studies using leachates with different chemical and biological compositions should be conducted to verify this conclusion.

5. CONCLUSION

The aim of this research was to provide a framework for sizing leachate dams and treatment infrastructure by reviewing applicable South African regulations and leachate treatment methods; and analysing methods for predicting leachate discharge and leachate evaporation, which are critical components in dam and treatment infrastructure sizing water balances.

A thorough literature review was conducted to determine the regulatory requirements that are applicable to leachate dams and leachate treatment. The South African regulatory requirements state that a leachate dam be designed with a liner system, according to the hazardousness of the waste disposed of in the landfill that is upstream of the dam, so as to prevent excessive leachate leakage into groundwater. Furthermore, leachate dams must be designed with a storage capacity that is large enough so that it is not probable that the leachate dam spills more than once every 50 years. The regulatory requirements also state that leachate must be adequately treated for permanent containment or discharge into the environment. The regulations also provide minimum requirements for treated leachate's quality, as well as its allowable discharge flow and temperature, if it is to be discharged into the environment.

A thorough literature review was conducted to determine the functions, mechanisms, and efficiencies of different treatment processes that may be used to produce leachate effluent that meets the required regulatory requirements. The literature generally recommends the use of the RO process, with a biological treatment and/or chemical treatment step preceding it in order to minimize fouling.

An analysis was conducted to verify whether the Hydrologic Evaluation of Landfill Performance model (HELP model) was suitable for use in sizing leachate dams. This was done by using the model to determine leachate discharge based on climatic parameters recorded at an existing landfill and reasonable landfill waste parameters and then comparing the leachate discharge given by the model (the predicted leachate

discharge) to the leachate discharge recorded at the landfill over the same period (the observed leachate discharge). The percent error between the observed average daily leachate discharge rate and the predicted average daily leachate discharge rate is 9.780 %, thus proving this research's hypothesis of 10 % correct. For this reason, it was found that the HELP model is suitable for use in leachate dam sizing, to ensure that leachate dams fulfil South Africa's regulatory requirements on leakage and spillage. However, this conclusion is based on a relatively short period of analysis.

A method for predicting leachate evaporation in a leachate dam was derived in this research; its suitability was analysed by comparing its predicted leachate evaporation to the leachate evaporation recorded at an existing landfill. It was found that the derived method effectively predicted the leachate evaporation over a 2.7 year period when no significant change to the leachate's salinity was observed. The percent error between the predicted and observed leachate evaporation was found to be 7.91 %, thus surpassing this research's hypothesis of 10 %. Furthermore, the method was found to have a percent error of 4.81 % in the wet season (i.e., in the months between October and March at the existing landfill in this report); thus, the method is more accurate in the wet season, the season when leakage and spillages are more likely to occur given the higher volumes of precipitation. For these reasons, the derived method was found to be suitable for use in leachate dam sizing to ensure that leachate dams fulfil South Africa's regulatory requirements on leakage and spillage.

Further research is required using data recorded over longer periods in order to further verify the HELP model's suitability for leachate dam sizing according to South African regulations. The HELP model should also be verified using mono landfills, such as ash and slag disposal facilities – this would mean that the waste would be uniform and consist of smaller particles; samples can be taken to more effectively test the drainage and storage parameters of the waste for input into the HELP model. Further research is also required using leachates with different compositions in order to further verify the derived method for predicting leachate evaporation.

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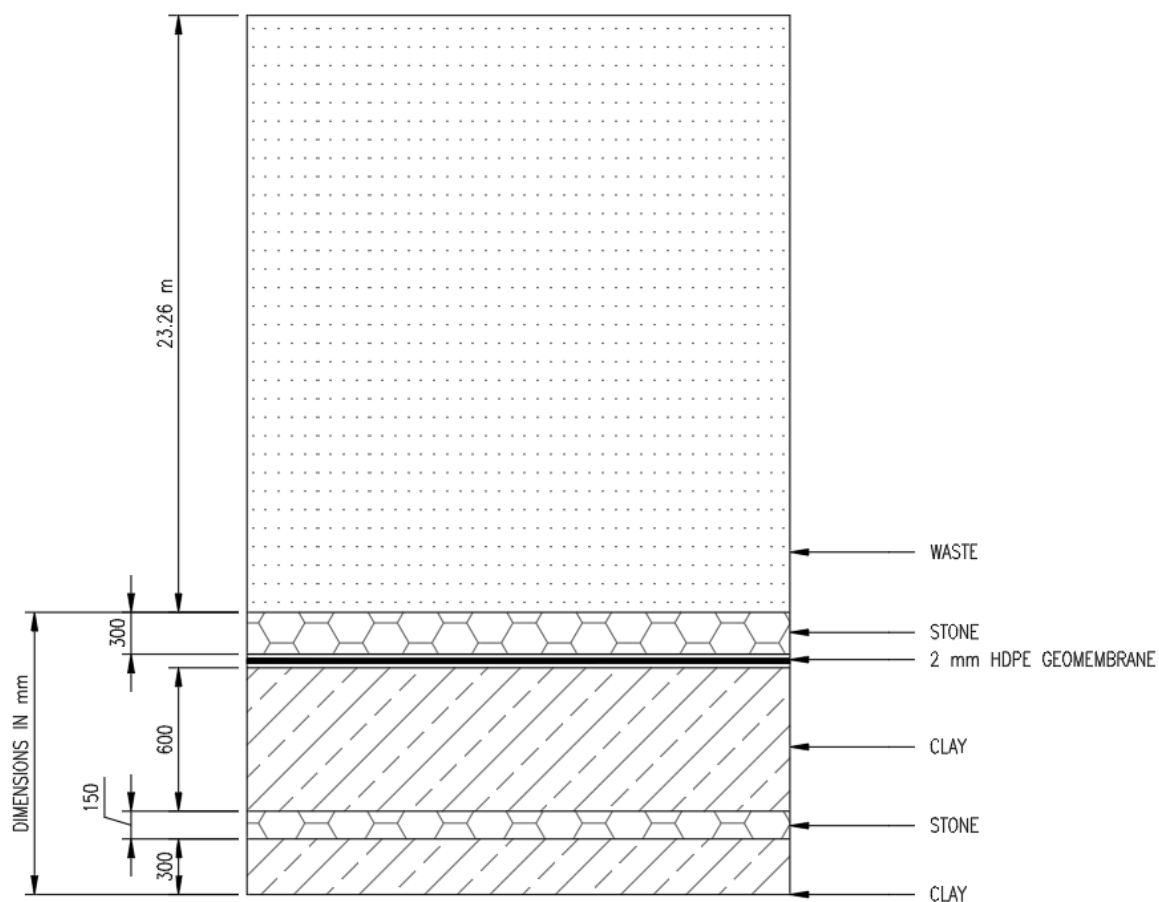
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APPENDIX A – HELP MODEL INPUT PARAMETERS

Input	Valid Values	Values Used
General Information		
Latitude (Lat)	-14.32° to 71.25°	Confidential
Longitude (Long)	-180° to 180 °	Confidential
Years of Simulation	1 to 100 years	4
Landfill Area	> 0 ha	2.927
Percentage of Landfill Area Subject to Runoff	0% to 100%	91.838
Specification of Initial Moisture	Yes/No	Yes
Water/Snow Storage	>= 0 mm	0
Weather		
Daily Precipitation Over Years of Simulation		1 414 days of data
Daily Temperature Over Years of Simulation		Used average daily temperature over each month
Daily Solar Radiation Over Years of Simulation		Manually calculated from 1 414 days of water evaporation data
Average Annual Wind Speed	0 km/h to 32.2 km/h	25
Average Relative Humidity for the First Quarter of a Year	0% to 100%	62.667
Average Relative Humidity for the Second Quarter of a Year	0% to 100%	57.667

Average Relative Humidity for the Third Quarter of a Year	0% to 100%	46.333
Average Relative Humidity for the Fourth Quarter of a Year	0% to 100%	57.333
Start of Growing Season	0 to 367	0
End of Growing Season	0 to 367	1 (Ground is completely bare over historic data)
Maximum Leaf Area Index (LAI)	≥ 0	0
Evaporative Zone Depth	0 cm to the depth of a geomembrane	47.498
Curve Number		
Curve Number Method	Dropdown list 1 to 3	User-specified (1)
Initial Curve Number (CN _{II})	> 0	75
Waste and Barrier System Parameters and Profile		
Layer Category	• Final Cover Soil (Topmost Layer) • Vertical Percolation Layer (Soil) • Lateral Drainage Layer (Soil) • Barrier Soil Liner • Waste • Geomembrane Liner • Geosynthetic Drainage Net	See figure below
Layer Description		See figure below
Layer Thickness	> 0 cm	See figure below
Total Porosity	Determined from layer description or 0 vol/vol to 1 vol/vol if manually inputted	See table below

Field Capacity	Determined from layer description or 0 vol/vol to total porosity if manually inputted	See table below
Wilting Point	Determined from layer description or 0 vol/vol to Field Capacity if manually inputted	See table below
Saturated Hydraulic Conductivity	Determined from layer description or > 0 cm/s if manually inputted	See table below
Initial Moisture	>= wilting point and <= total porosity	See table below
Drainage Length	> 0 m	See table below
Drainage Slope	0 % to 50 %	See table below
Leachate Recirculation	0 % to 100 %	0
Recirculation to Layer	To a vertical percolation or lateral drainage layer only	0
Subsurface Inflow	>= 0 mm/year	0
Geomembrane Pinhole Density	#/hectare	0
Geomembrane Installation Defects	#/hectare	0



Layer	Design Parameters
Waste	Varies with standard waste type being analysed. Shown in the table below.
Stone	- Total Porosity (vol/vol) = 0.397 - Field Capacity (vol/vol) = 0.032 - Wilting Point (vol/vol) = 0.013 - Saturated Hydraulic Conductivity (cm/s) = 3×10^{-1} - Drainage Length (m) = 55 - Drainage Slope (%) = 4.25 - Initial Moisture (vol/vol) = 0.032
2 mm HDPE Geomembrane	- Saturated Hydraulic Conductivity (cm/s) = 2.00×10^{-13} - Geomembrane Pinhole Density (#/hectare) = 0

	• Geomembrane Installation Defects (#/hectare) = 0 • Geomembrane Placement Quality = “Good Field Placement”
Clay	• Total Porosity (vol/vol) = 0.451 • Field Capacity (vol/vol) = 0.419 • Wilting Point (vol/vol) = 0.332 • Saturated Hydraulic Conductivity (cm/s) = 6.8×10^{-7}

Waste Types	Total Porosity (vol/vol)	Field Capacity (vol/vol)	Wilting Point (vol/vol)	Saturated Hydraulic Conductivity (cm/sec)
Municipal Waste	0.671	0.292	0.077	1.0×10^{-3}
Municipal Waste with Channelling	0.168	0.073	0.019	1.0×10^{-3}
High-Density Electric Plant Coal Fly Ash	0.541	0.187	0.047	5.0×10^{-5}
High-Density Electric Plant Coal Bottom Ash	0.578	0.076	0.025	4.1×10^{-3}
High-Density Municipal Solid Waste Incinerator Fly Ash	0.450	0.116	0.049	1.0×10^{-2}

APPENDIX B – A PAN CONVERSION TO FALSE GRASSLAND

Month	Average Monthly A-Pan Evaporation at the Landfill (mm)	A-Pan to False Grass Land Evaporation Factor (Midgley, et al., 1994)	Average Monthly False Grass Land Evaporation at the Landfill (mm)
January	276.938	0.73	254.938
February	250.526	0.73	245.198
March	294.113	0.67	291.488
April	248.525	0.61	233.525
May	217.586	0.43	217.588
June	194.300	0.24	194.300
July	205.750	0.20	205.750
August	239.000	0.27	239.000
September	309.953	0.41	309.953
October	333.733	0.53	324.400
November	312.538	0.65	292.577
December	276.950	0.73	266.900

APPENDIX C – LEACHATE EVAPORATION AND LEACHATE DISCHARGE COMPUTED DATA

Leachate Evaporation Computed Data						
Month	Year	Date	Observed Monthly Water Evaporation	Observed Monthly Leachate Evaporation	Predicted Leachate Evaporation	Percentage Error (%)
11	2019	Nov-19	393.1	324.1	294.2019139	-10.16243766
12	2019	Dec-19	440.5	319	379.5199518	15.94644801
1	2020	Jan-20	325.95	323.7	274.6741441	-17.84873346
2	2020	Feb-20	306.8	254.8	258.7797335	1.537884549
3	2020	Mar-20	309	263	238.3753584	-10.33019594
4	2020	Apr-20	193.5	213	140.3792016	-51.73187879
5	2020	May-20	302	256	152.9248218	-67.40251647
6	2020	Jun-20	286	224	134.6602075	-66.34461219
7	2020	Jul-20	282.5	169	136.1157465	-24.15903694
8	2020	Aug-20	384	114	171.6177451	33.57330275
9	2020	Sep-20	387.5	134	196.7232474	31.88400367
10	2020	Oct-20	456.9	247.9	281.9506079	12.07679889
11	2020	Nov-20	337	263	253.1859646	-3.876216209
12	2020	Dec-20	340.3	251.3	290.2559259	13.42123361
1	2021	Jan-21	290.7	304.2	248.2705478	-22.52762269
2	2021	Feb-21	223	183.5	187.4376438	2.100775361
3	2021	Mar-21	354.9	364.9	273.7845136	-33.2800001
4	2021	Apr-21	318.5	251	223.6997841	-12.20395274
5	2021	May-21	275.5	159	146.7612104	-8.339253688
6	2021	Jun-21	202.95	45.45	112.0335383	59.43179095
Total Average						-7.911710954

[illegible]

[illegible]