UNIVERSITY OF CAPE COAST

GEOHYDROLOGICAL CHARACTERISTICS OF LANDFILLS TO GROUNDWATER SENSITIVITY: A CASE OF OTI LANDFILL SITE-KUMASI

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BY

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Thesis submitted to the Department of Geography and Regional Planning of the Faculty of Social Sciences, College of Humanities and Legal Studies, University of Cape Coast, in partial fulfilment of the requirement for the award of Master of Philosophy degree in Geography and Regional Planning.

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DECLARATION

Candidate’s Declaration

I hereby declare that this thesis is the result of my own original research and that no part of it has been presented for another degree in this university or elsewhere.

Candidate’s Name: Obiri Yeboah

Signature: …………………………………… Date: ……………………

Supervisors’ Declaration

We hereby declare that the preparation and presentation of this thesis were supervised in accordance with the guidelines on supervision of thesis laid down by the University of Cape Coast

Principal Supervisor’s Name: Professor Benjamin Kofi Nyarko

Signature: ………………………………… Date: ………………………

Co- Supervisor’s Name: Professor Edward Akwasi Ampofo

Signature: ………………………………… Date: …………………………

ABSTRACT

Soil plays a crucial role in preservation of groundwater resources from contamination and pollution. In view of this, the study focused on the geophysical and chemical condition of the Oti landfill soil and also assessed the physicochemical characteristics of local groundwater and leachate. The groundwater quality assessment tool was used to assess the impact of the Oti landfill site on the local groundwater. Systematic, grab and quartering sampling techniques were used to sample the soil, leachate and groundwater. On the leachate and groundwater samples, in-situ measurements were made to record the pH, EC and TDS concentration. Three different pits with depth 150 cm each were dug at the landfill site and three different samples were taken from each pit at varying depths. Samples were taken to the laboratory under a temperature below 4°C. Parameters that were analysed included K^+, Ca^{2+}, Mg^{2+}, Na^+, Al^{3+}, BOD, COD, DO and some selected heavy metals including Fe^{3+}, Zn, Cu, Mn, As, Cd, and Pb. Results obtained from the groundwater samples showed that the water was still in a good condition for usage. Hydrogeophysical data of the study area was also collected and the DRASTIC model was used to generate a DRASTIC vulnerability index in assessing how vulnerable the local groundwater was to pollution and also to validate the laboratory results. Both the vulnerability and sensitivity analysis results validated the laboratory results showing that the landfill has had no negative effect on the local groundwater system.

KEY WORDS

Geohydrological

Geophysical

Geomorphological

Biogeochemical

Hydrophysical

Hydrogeological

Hydraulic conductivity

Porosity

Bulk density

Titre

Landfill

Contaminants

Pollutants

Leachate

Hydrogeology

Percolation

Sorption

Absorption

Advection

Dispersion

Physicochemical

Exchangeable

Acidity

Groundwater

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DEDICATION

I dedicate this work to my parents WOII Peter Yeboah and Elizabeth Akua Agyeiwah.

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CHAPTER ONE

INTRODUCTION

Groundwater pollution as described by World Health Organisation (2006) is the introduction of harmful substances in undesirable concentration that is not characterised in groundwater. Even though the pollutants are anthropogenic in nature, some are also lithogenic (Gao, Yu, Luo, Zhou, 2012). According to Adelekan and Abegunde (2011), movement of contaminants and pollutant into soils and groundwater systems is a multi-media problem because there are several pathways and sources from which contaminants and pollutants emanate to contaminate or pollute these natural resources.

In Ghana, one of the identified sources of pollution and contamination of groundwater are poorly managed landfills. Landfills have been categorised into different types depending on the nature of waste they receive but in Ghana, the dominating type is the municipal landfill. (Aderemi, Oriaku, Adewumi & Otitoloju, 2011). Municipal landfills have a high concentration of organic waste that affects the biogeochemical processes within the landfill environment and generate strong anaerobic leachate which has a high content of dissolved organic carbon, ammonium, organic compounds, salts and heavy metals released from the waste (Rodríguez-Eugenio, McLaughlin & Pennock, 2018).

According to Boateng, Opoku and Akoto (2019), the leachate is produced when water mixes with the assorted waste at the landfill site. This leachate can leak into groundwater through geomembrane resulting construction defects and enters into aquatic environment and also through vapor diffusion via the liner used for the construction. This, when gets into groundwater, pollutes and contaminant this this resource used for domestic, industrial and agricultural uses.

Background to the Study

Rapid population growth in urban centres have resulted in the production of various solid waste residues (Tuxen, Palle, Aberechtsen, Reitzel, Perdersen, and Bjerg, 2003). One of the various ways of managing these residues is by dumping them on landfill sites. Globally, landfilling is one of the most convenient strategies for solid waste disposal. This is because it is less expensive for the disposal of solid waste and plays a key role in the integrated solid waste management. Increase in Ghana's urban population has increased the amount of municipal, industrial, e-waste and hazardous wastes that enter the waste trail on a daily basis (Boateng, Asare & Danuor, 2013).

Generally, contamination and pollution of groundwater is mainly influenced by two main factors namely man induced factor and naturally occurring one resulting from the geological formation of the area. A study conducted by Smedley, Edmunds, Pelig-Ba, (1994) showed that groundwater naturally gets contaminated and polluted when it interacts with the local geology due its mineralogical composition. Examples of this situation was seen when samples of groundwater from 93 boreholes in the Tamale Metropolis were taken and analysed by the regional Ghana Water and Sewage Cooperation Laboratory in Tamale. It was found that 63.5% of the boreholes had fluoride concentration above the WHO admitted threshold of 1.0 mg/L. The study revealed that the contamination of the groundwater was due the interaction between the local groundwater, the soil and the rocks found there.

The anthropogenic source of groundwater pollution according mainly results from improper agricultural activities through excessive use of weedicides, pesticide and fertilizers. Between 1969 to 1980 a survey that was conducted in the Upper East region of Ghana revealed a substantial amount of nitrate in the groundwater and the main entry point was from agricultural lands (Boateng, Asare and Danuor, 2013). Other identifies source of groundwater pollution and contamination is through leakage of septic tanks, leakages in buried oil tank, mining activities and places for waste management (Duah, 2006).

According to Boateng, Opoku & Akoto, (2018), one key source of groundwater pollution in Ghana is from landfill site. This occurs when leachate produced from landfill site comes in contact with the local groundwater resource. Leachate is liquid waste from municipal solid waste at landfill sites (Denutsui, 2012). They are produced when waste dumped at landfills comes in contact with water. It undergoes a number of physicochemical, and microbiological changes that lead to the release of this toxic waste which contains countless organic and inorganic compounds. They have highly concentrated effluent inorganic compounds such as calcium, magnesium, ammonium, sodium, and sulphur as well as heavy metals such as lead, zinc, cadmium, chromium, nickel and also xenobiotic substances. Human exposure to constituents of this leachate above the internationally accepted limits can associate one's life to several bio-metal poisoning related diseases like cardiovascular and renal disease, ataxia, cancer, gastrointestinal disorder, skeletal deformities, neurological diseases, hypertension, pneumonitis, diarrhea and anemia (Boateng. Opoku and Akoto, 2019).

The rate of leachate movement within soil to saturated groundwater system and the chemical characteristics of the leachate that is produced from the landfill depends mainly on solid waste composition, the geological and the hydrological characteristics of the site, the compaction of the soil of the landfill, and the temperature conditions available (Belghaza, Piga, Loddo, Messari, & Touhami 2013). The geology of Kumasi is mainly dominated by meta sedimentary rock of Birimian origin which is highly jointed, foliated and featured. Primarily, little amount of water is able to permeate through the fractured joints. However, secondary permeability has developed due to intensive weathering and fracturing that allows substantial flow of water (Ewusi, Annor, Seidu & Gyeabour, 2016). Aquifer within Kumasi Metropolitan Assembly generally ranges between 3 meters and 26.5 meters with an average depth of about 11.38 meters (Ewusi et al., 2016).

According to Kesse, (1985) and cited in Ewusi et al., (2016), the deepest aquifer in Kumasi Metropolis was found between depth 55 m and 61 m while the shallowest was between 6 m and 18 m. The maximum yield was 900 L/m and the minimum yield was 81 L/m. The permeability is high in the phyllites dominated areas than granite dominated zones. This, according to Kesse (1985), is as a result of secondary porosities developed due to intense fracturing and weathering. These secondary porosities have influenced the movement of fluids and defines the hydrogeological nature of the Kumasi Metropolis.

According to Taylor and Allen (2006), the hydrogeology controls the decomposition of waste and the subsequent movement of leachate at landfills into groundwater. The hydrogeology within the geographical settings controls and influence the mobility and diminution of dissolved solids contained in the leachate as caried by the groundwater. To Taylor and Allen, whenever residues are dumped at landfill sites, these residues become part of the local hydrological system. Leachates produced from the landfill site through hydrolysis and solubilisation processes form a complex and series biochemical reactions during decomposition and degradation of organic residues, percolate through the deposit and mobilize other components within the wastes.

As noted by Sulemana, Antwi-Agyei and Horgarh (2015), a properly managed landfills do have any significant effects on the surrounding groundwater whereas others that are poorly managed can seriously degrade groundwater resources and the effect of this degradation may be difficult to overcome. Therefore, before a landfill can be properly designed, the hydrogeological environment of the landfill site must be a key factor.

However, the effects of hydrogeological factors on landfills, the design of landfills under a given geologic and climatic conditions in Ghana are mere concepts rather than specific designs. According to Taylor and Allen (2006), the solution to this problem requires an efficient and effective geophysical approach. This then calls for the integrated use of efficient hydrogeological, geohydrological and managerial methods to manage contaminants that generate from landfills.

Statement of the Problem

Groundwater pollution in recent years has become a critical issue that is catching the attention of the global world. This is as a result of the ever-increasing and the constant demand of groundwater for domestic, industrial usage as well as agricultural purposes (Osei, et al., 2011). Globally, groundwater is considered as one of the safest sources of water for domestic use.

Global estimated shows that about 2 billion of global population uses groundwater, and is the most extracted raw material with an estimation of about 982 km^3/yr. In Ghana, there are more 56,000 groundwater extraction systems which is made up of boreholes and dug-outs wells which are used for domestic, industrial and agricultural purposes (Margat & van der Gun, 2013).

While freshwater usage is on ascendency both globally and locally, the pollution of the aquifer system by human activities particularly from landfill sites is on the rise (Sulemana, et al., 2015). Percolation of leachate from the landfill into the landfill soils can affect local aquifer as well as surface water that surrounds the landfill site. This make is a major concern in Ghana and other developing issues where residential buildings are keeps on springing up closer to landfills. Majority of residential buildings at the Oti landfill site uses groundwater for domestic purposes via individual residential boreholes and are at risk fetching contaminated water (Osei et al., 2011).

Globally, a number of researches has been done landfill sites and groundwater pollution. In Ghana, related researches include Evaluation of Landfills Sites in Accra on the surrounding Environment by Osei et al., 2011; Heavy Metal Contamination Assessment of Groundwater Quality: a case study of Oti landfill site, Kumasi by Thomas Kwame Boateng, Francis Opoku & Osei Akoto 2019 among others. The focus of these researches was on the physicochemical characteristics of the landfill leachate and the local groundwater quality. Research concerning the soil characteristic of these landfills and how it can aid in the migration of leachate produced from the landfill site to the local groundwater system is still missing in literature. However, the geohydrological characteristics of landfills serve as the conduit for the movement of the leachate from the landfill sites to the surrounding groundwater. A study conducted by Boateng et al., (2013) suggested a potential migration of leachate from the Oti landfill site to the local groundwater. It is this reason that this research topic has espoused to research into the geohydrological characteristics of the Oti landfills site in Kumasi.

General objective of the study

The general objective of the study is to evaluate the geohydrological characteristics of the Oti landfill site and how it can cause groundwater pollution.

Specific Objectives of the Study

1. Analyse hydrophysical properties of the landfill soil in relation to groundwater pollution
2. Analyse the soil chemical characteristics of the Oti landfill in relation to groundwater sensitivity.
3. Analyse the physicochemical characteristic of the leachate and surrounding groundwater on how the landfill has impacted it.

Research Questions

The research questions of the study were:

1. What are the hydrophysical properties of the landfill soil and its effects groundwater sensitivity?
2. What are the chemical characteristics of the soil and its relation groundwater sensitivity?

3. What are the physicochemical characteristics of leachate and the surrounding groundwater?

Significance of the Study

Every research is aimed at solving a particular problem or adding knowledge to the existing one. The findings of this research will be useful in understanding how leachate and its inherent pollutants and contaminants behave when migrating within a given soil. The research proposes some solutions that are helpful in curtailing groundwater pollution from leachates. The findings are also relevant to the people and communities surrounding the landfill whose source of water is from the local groundwater through reporting the current state of the groundwater in their community to them. It is also a reference material for academic and future research works.

Delimitation

The study employed experimental design which covers only the selected landfills. The study was limited only to the soil, groundwater, and leachates that were collected from the Oti landfills site during the time of the research. This made the study limited in generalization since any results derived suit and reflects only soil characteristics of the researched landfills and if possible other soils with similar characteristics and under similar environmental conditions. Besides, the study delimited itself to the parameters that were mentioned in the methodology and analysed even though there are several other parameters that could have been looked into.

Limitation of study

The study was limited by a number of factors, this included limited fund to carry on with data collection and laboratory analysis. Another challenge faced had to do with the collection and movement of soil sample to the laboratory for analysis. This was due to the sensitive and fragile nature of soils that are taken to the laboratory for hydraulic conductivity analysis. I was compelled to go back several times to take samples again anytime the laboratory one got spoiled under preparation.

Organization of the study

The remaining chapters of the thesis are organized as follows: chapter two of the study reviews related literature on geo-hydrological characteristics of landfill sites to groundwater sensitivity. The chapter three presents the materials and methods used for the study. The chapter talked about the research design, short account of the study area, the units of analysis, sources of data, sample size and sampling procedure, instrumentation and data collection and data processing and analysis. The fourth chapter is on the results and discussion of the analysed data. The chapter five is on the summary conclusion, recommendations and suggested future research.

CHAPTER TWO

LITERATURE REVIEW

Introduction

This chapter reviews the theoretical framework, conceptual model, and empirical review of porous media and contaminant transport. The theoretical review looked at the theory of contaminants transport and groundwater pollution. The theory employed and thoroughly discussed is based on the theory of dispersion within a porous medium. The conceptual models review various ideas that conceptualised to show landfill-soil-groundwater interactions. The models included the DRASTIC model and S-P-R model which have been widely used to assess groundwater pollution. The empirical review also looked at literature on soil and groundwater pollution, sources of groundwater pollution, landfill and groundwater pollution, movement of contaminants and fates as well as groundwater flow.

Theoretical Framework

The Theory of Dispersion in Porous Media

According to Dullien (1992), dispersion is the spreading or scattering away of substances including but not limited to contaminant and pollutants from it flow path. When pollutants or contaminants are exposed and introduced into the earth’s surface in a fluid form, there is a differential migration and attenuation as some are transported faster and some slower than the average flow rate of the groundwater. Others, according to Dullien (1992), travel horizontal or vertical from the flow path of the groundwater. The number of contaminants and pollutants that are dispersed from the flow path when measured constitute the dispersion coefficient.

Dispersion of flux is governed by multiple factors which include the chemical properties forming the contaminant, the homogeneousness of the aquifer materials, and changes to groundwater flow path or direction over time.

According to Hunt, Skinner and Ghanbarian (2011), pollution of groundwater is governed by dispersive processes and that only in some few instances that it is convective when the velocity is prime to its mixing with the aquifer. Hunt et al. defined dispersion as a flow of flux in a porous media that occur and evolve from one transition to another transition between two domains with different composition of flux phases. According to Dullien (1992), dispersion in porous media theory qualitatively describes and quantitatively estimate how miscible fluxes behave when displacing from one medium to another medium.

Hunt and Ewing (2009), noted that whenever two or more miscible fluxes come in contact, sharp interface develops at the beginning and late ceases into a transition zone when their chemical concentration and physical characteristics levels up with time. According to Hunt and Ewing, the development of the dispersion theory has three steps. The first is laboratory experiment which investigates how the miscible fluids displace in a porous medium. The second step is deriving mathematical processes formulas and equation to describe the movement during the laboratory experiment. The laboratory processes, equation and the mathematical formulas describe the synthetic of the results on how fluids disperse in porous medium. The third step applies the laboratory results to a field problem of real pollution and subsequently modifying the equation. The modification is as a result of the complex nature of the porous soil in the real world.

The difficulties in predicting the transporting properties in both unconsolidated and consolidated porous media result from the complex geometry within the porous matrix (Dullien, 1992). According to Bijeljic, Mostaghimi and Blunt (2001), when solutes are transported throughout in a porous medium, it requires a time sufficiently long before the dispersion coefficient reaches asymptotic value and the dispersion becomes mesokurtic or Gaussian.

When leachate from landfills enters into soils and groundwater, mixing of the solute within the leachate with the groundwater at the initial stage is unidirectional at time zero. At this time, the movement of the leachate is unidirectional and the mixing occurs at a constant rate (Dullien, 1992). If there is no physicochemical interaction of the solutes of the leachate with the soil matrix, the process of adsorption becomes very limited. As the movement continues, the concentration rate of the injected leachate reaches a transition zone and then lengthens with time (Neuman, 1990)

Generally, there are two mechanisms through which leachate from landfill sites disperses within soil and aquifer media. According to the theory of dispersion, it disperses either as mechanical action or physicochemical action (Dullien, 1992). With the mechanical action, the distribution of the flow velocity is not uniform within the entire soil due to the boundary effects resulting from the matrix of the soil. Basically, the boundary effect results from three main reasons:

1. Viscous velocity normally has a velocity of zero on a solid surface
2. Differences in the dimensions of the pore spaces create discrepancies among the maximum velocities along the axes of the pores.
3. There is a fluctuation of streamlines with respect to mean directional flow.

These three mechanical actions occur simultaneously to yield a mechanical dispersion of leachate and other solutes within a soil medium (Hunt & Ewing, 2009). The physicochemical dispersion, on the other hand, is molecular diffusion that is produced from the chemical potential gradient. The chemical potential of the leachate and the groundwater interacts with their chemical concentration. Molecular diffusion within physicochemical dispersion continues to take place even when the fluxes are not in motion (Bijeljic and Blunt, 2006).

Characteristic Parameters of Dispersion

The characteristics that define the parameters of dispersion can be grouped into two main domains namely the numerical value which measures dispersion and those parameters that influence the dispersion process.

Parameters Measuring Dispersion

According to Sahimi (2011), a quantitative measure of the dispersion of leachate within a soil and groundwater aquifer demands a definition of a scale to which the measurements are made. Scale measurements are very useful in physics and play a pivotal role in the context of flux movement in porous media and dominate several fundamental concepts and methodologies which govern groundwater-pollution studies.

Raoof and Hassanizadeh (2010) stated that in measuring the dispersion of pollutants within a porous media, the theory of dispersion is used in three main levels namely the local level, the macroscopic level, and the pore-volume level. The local level parameters define the physical quantity at a given point, a given volume element which is consistent with molecular-physics data. Raoof and Hassanizadeh further stated that the pore volume level parameters are described as "mean" parameters with respect to other averaging processes which correspond to local level parameters over a finite volume of fluid. Within the porous medium, the concept "pore quantities" is based on this description and the mean is taken over a set of pores. The macroscopic as used in the study of flow through a porous media helps to describe a continuum which is equivalent to the fluid set and the solid matrix.

Measurement of dispersion of pollutants in a porous medium according to Bijeljic and Blunt (2006) can be done either through physical quantities that have a direct physical meaning and its numerical value can be obtained in a direct way at least in theory; a set of mathematical quantities which result from mathematical interpretation of the theory of dispersion of solutes in a porous media. The physical quantities include the velocity of the flux, the concentration rate, and the densities. The mathematical densities represent the dispersion coefficient.

Review of Conceptual Model

Groundwater conceptual model is an abstract representation of the physical hydrogeological system and its hydrological processes. According to

Nirmalakhandan (2001), groundwater conceptual models are usually presented mathematically or graphically in a cross-section form with supporting d Net Recharge at explain the essential features of the landfill-groundwater system and the basic fundamentals for geohydrological assessment. It further defines the groundwater and the subsurface hydrogeological processes within the surroundings of the landfill and how the landfill interacts with the groundwater (Egbi, Akiti, Osae, Dampare, Abass & Adomako, 2015).

DRASTIC model

The DRASTIC model is used to assesses the pollution potential of a given area through a systematic evaluation of hydrogeological parameters of that geographical space. The model uses existing hydrogeological data and has two main potions: the description of mappable units termed hydrogeological settings and the superposition of a relative ranking system called DRASTIC. The hydrogeological settings integrate the key hydrogeological factors and inferences are made on the potentials for contaminants and pollutants entering and polluting or contamination the local groundwater.

Table 1: Appraisal of Net-Recharge, Hydraulic Conductivity, and Depth to Groundwater

		Range			Ratin
Ra	Net recharge	Hydraulic conductivity		Depth to groundwater	
0-2	1	1-100	1	0-5	10
2-4	3	100-300	2	5-15	9
4-7	6	300-700	4	15-130	7
7-10	8	700-1000	6	30-50	5
10+	9	1000-2000	8	50-75	3
		2000+	10	75-100	2
				100+	1

Source: Aller et al., (1987).

The DRASTIC model is an acronym of seven parameters. **“D”** represents the depth from the surface to the groundwater which is mainly measured in meters or feet; **“R”** represents the net recharge of the area (total inflow minus total outflow); **“A”** is the aquifer characteristics; **“S”** is the soil media characteristics; **“T”** is the topography of the area under study, **“I”** is the impact of the vadose zone on the groundwater system while the **“C”** is the hydraulic conductivity of the area.

Table 2: Evaluation of the Aquifer Factor in the DRASTIC Model

Type of aquifer media (Range)	Rating
Massive shale	1-3
Metamorphic/Igneous	4-5
Weathered metamorphic/igneous	3-5
Glacial till	4-6
Bedded sandstone, limestone and shale sequence	5-9
Massive sandstone	4-9
Massive limestone	4-9
Sand and gravel	6-9
Basalt	5-10
Karst limestone	9-10

Source: Aller et al., 1987.

This model was developed in the United States purposely to protect groundwater resources. According to Lathamani, Janardhanab, Mahalingam and Sureshad (2015), the DRASTIC model is an empirical groundwater model that can estimate and predict groundwater contamination and vulnerability of aquifer systems based on the hydrogeological setting and processes of that area. The model integrates the geomorphological, geological, hydrogeological and the meteorological characteristics of the area to explain the susceptibility of a groundwater to pollution using overlay and index approach.

Table 3: Evaluation of Soil Factor in the DRASTIC Model

Range	Rating
Thin or absent	10
Gravel	10
Sand	9
Shrinking and or Aggregated clay	7
Sandy Loam	6
Loam	5
Silty Loam	4
Clay Loam	3
Non-shrinking and non-aggregated clay	1

Source: Aller *et al*., 1987.

Each parameter of the model was assigned a weight based on its potential in polluting groundwater. The typical rating ranges from 1-10 and weight from 1-5 to compute for DRASTIC Index (DI). The DI is a measure of pollution potential of each factor and it is computed by the summation of the products or the values of the weight and the rating for each factor. The final result from the computation for each hydrogeological setting is a numerical value called the DRASTIC Index. The DRASTIC model is developed to evaluate and predict the vulnerability of groundwater in an area of not less than 100 acres (Mato, 2012).

$$DI = DrDw + RrRw + ArAw + SrSw + TrTw + IrIw + CrCw \qquad (2\text{-}1)$$

Where: “r” = rating and “w” = weight

Table 4: Evaluation of Vadose Zone Factor in the DRASTIC Model

Range	Rating
Silt/clay	2-6
Shale	2-5
Limestone	2-7
Sandstone	4-8
Bedded limestone, sandstone, and shale	4-8
Sand and gravel with significant silt and clay	4-8
Metamorphic/igneous	2-8
Sand and gravel with significant silt and clay	6-9
Basalt	2-10
Karst limestone	9-10

Source: Aller *et al*., 1987

The subscripts “r” and “w” in Eq. 2-1 is the rating of the factor being considered and its corresponding weight assigned to each factor. The higher the value of the DI the greater the pollution potential. Navulur (1996) as cited in Mato (2012), converted the computed DRASTIC indices into risk categories. It was categorised as low. Moderate, high and very high.

Table 5: Evaluation of topography Factor in the DRASTIC Model

Range (% slope)	Rating
0-2	10
2-6	9
6-12	5
12-18	3

Source: Aller *et al*., 1987

Assumptions of the Model

1. That contamination of groundwater originates from the surface.

2. That contamination reaches the water table when there is enough rainfall at the surface to serve as a transporting agent to move contaminants from the surface into the saturated zone.
3. That the rate of movement of contaminants is the same as the rate of movement of water.
4. That the model is applied to a region not greater than 100 acres.
5. That the pollution occurs in an unconfined zone
6. That the dominating pollutant is not from pesticides even though the method can be modified to include it.

Table 6: DRASTIC Index (DI) Ranges for Qualitative Risk Categories

(DI)	Low	Moderate	High	Very High
	1-100	101-140	141-200	200+

Source: Navulur et al., (1996)

Since the model was construction in the United States, it has been used and lengthily applied in many groundwater quality assessments. Mato (2012) used the DRASTIC model to assess groundwater pollution in Der el Salaam-Tanzania; Lathamani, et al., 2015 used it to assess aquifer vulnerability in Mysore- India. Navulur (1996) recommended the use of the model as a tool when formulating policies regarding groundwater management strategies. Engel (1996) stated that the approach the model uses for forecasting groundwater vulnerability equally does well for both nutrients and pesticides.

Areas the model predicts to have moderate to high very high requires detailed investigation. The DRASTIC model has been used to develop a preventive plan for several watersheds which includes Managua city, Nicaragua. The results of the DRASTIC model were further validated against the local hydrogeological knowledge and found to be useful. Scharp (1994) recommends a continuous update of the rapid assessment whenever new information is obtained.

The DRASTIC model was used and complemented by five overlay maps, which gave information on drainage basins, water sources; groundwater supplies and groundwater resources. Adoption of DRASTIC technique and its associated overlays served as the initial step by the Swiss government during the development of a comprehensive plan in the managing Sweden's groundwater supply (Lathamani et al, 2015). The methodologies applied by the DRASTIC model, according to Mato (2012), enable city planners to identify and recommends site that are hydro-geologically sound for landfills development.

Source-Pathway- Receptor (S-P-R) Model

According to Gouldby, Sayers, Mulet-Marti, Hassan, and Benwell, (2008), the S-P-R conceptual model was first used in the field of environmental engineering in the late 1970s to show and explain the migration of environmental contaminants and pollutants from a particular source propagating through several pathways to possible receptors. Since then the model has been used in several environmental risk assessments (Narayan, Hanson, Nicholls, and Clarke, 2012). This model describes the characteristics of the source of a release of a contaminant and the possible pathway through which the contaminant migrates to reach a receptor.

In the study of the potentials of landfills in the contamination and pollution of groundwater resources, Gouldby, et al., (2008), stated that the model in combination with a system diagram is a powerful way of collating a comprehensive description of the state of landfill and the local groundwater system, its constituent components, and its interrelation. In developing the system diagram, the preliminary attention is the identification of the possible receptor and construction up a link of pathways. A major principle of this model is recognising the 'pathway' and 'receptors'. In its application, the principal all components of the system can simultaneously function as pathways to 'downstream' receptors and as receptors in their own right. Pathways of particular importance can then be identified based on the value of the connected receptors (Narayan, et. al., 2012). Other possibilities of this approach include identification of receptors vulnerable to combinations of sources and identification of weak links and failure routes.

Evaluation of the DRASTIC and SPR Model.

Both the DRASTIC model and the Source-Pathway-Model have been used by several research agencies and individuals in the assessment of groundwater pollution. Mato (2012) used DRASTIC model to assess the groundwater pollution in Dar el Salam in Tanzania (Saidi, Bouri and Dhia, 2010) also used DRASTIC model to assessed groundwater vulnerability and risk- mapping of the Hajeb-jelma aquifer in Central Tunisia. The S-P-R model also used by the European Commission (EC) (2010) for Common Implementation Strategy for the Water Framework Directive (2000/60/EC): Guidance on Risk Assessment and the use of Conceptual Models for Groundwater where it was used in all member states of the EC for groundwater risk assessment. Also, the model was used as the main model in CIEF Seminar Report: March 2004 when presenting on Contaminated Land Risk Assessment in Glasgow.

Within the SPR, the landfill was identified as the main source for the production of pollutants. The soil media, the vadose zone and the local aquifer are the pathways for the carriage of pollutants and contaminants to the surrounding wells. These parameters can help understand landfill-groundwater connections. However, the DRASTIC model integrates all the parts of the S-P-R model and includes in its other parameters like the depth to groundwater, the topography, and the net groundwater recharge of the area of the area under investigation. In view of these other factors, the DRASTIC model was chosen ahead of the Source- Pathway-Receptor-Model.

Empirical Review

Soil and groundwater pollution

FAO, (2017) defined pollution as any alteration in the physical, chemical and or biological properties of a matter thereby restricting its normal and natural usage. According to Rodríguez-Eugenio, et al., (2018), soil pollution is the presence of any chemical which is out of place or present at a very high concentration which poses an adverse effect on any non-targeted organism. Environmental Pollution Centre (2017) also defined soil pollution as any process of introducing toxic chemical or any harmful substance in the soil which is high enough to pose a risk on the ecosystem. Woodford (2019) defined it as an introduction to the human environment substances which have great enough concentrations to render the environment harmful for plants, animals, and humans. Soil pollution can, therefore, be defined as an introduction of any chemically induced substance into the soil which compromises the quality of the soil and its living organisms. Groundwater pollution as defined by Groundwater Foundation (2016) is the introduction to groundwater any harmful man-made products that make it unsafe for man usage. That is groundwater contamination is a reduction of the quality of groundwater resources by harmful anthropogenic or natural substances.

According to Adeolu (2011), the contamination of groundwater can be traced and grouped into four main origins. The pollution origin can be from domestic, industrial, agricultural or any given environmental which happens either accidentally as in the case of sudden break of septic tank intrusion of marine water into groundwater aquifer or the pollution can be continues as in an example of rainwater infiltrating into sanitary landfills and infiltration of pesticide from farmlands into a local aquifer system.

According to findings by Nagarajan, Thirumalaisamy, Elango and Lakshumanan (2012), the impact of industrial hazards and municipal solid waste on soil and groundwater pollution should be of high concern. From their findings, European chemical industries produced 319 million tonnes in 2015 and 117 million tonnes were found hazardous to the environment. As at 2012, the estimated global municipal solid waste was found to be around 1.3 billion tonnes and it is estimated to be around 2.2 billion tonnes by 2025. Nagarajan et al., (2012), noticed that some developing including Rwanda and Ethiopia have increased their pesticide usage by over six times in the past decade, Sudan almost ten times and Bangladesh about four times.

According to Rodríguez-Eugenio et al., (2018), there is a growing awareness on the increasing soil and groundwater pollution around the world as well as an increase in the research on the assessment and remediation of these pollutions. Universally, soil and groundwater contamination and pollution have been used interchangeably with but the Intergovernmental

Technical Panel on Soil (ITPS) under the Global Soil Partnership formalised the distinctions between the two. To ITPS, contamination occurs when the chemical constituent present in soil and groundwater is greater than they will occur naturally but may not necessarily harm the medium. Soil and groundwater pollution, on the other hand, is the introduction of a chemical or substance out of place and/or present at a higher than the normal concentration that has adverse effects on any non-targeted organism ITPS (2015).

Sources of groundwater pollution

Point Source Pollution

It has been identified that soil pollution and contamination can be caused by either some specific event or series of events from where pollutants and contaminants are released into the soil. Places where a source of pollution can easily be identified is point source. Anthropogenic activities that goes on within a given place is known as point source pollution. Example includes landfill sites, places where excessive agrochemical has been exposed to, places with chemical tanks leakages, mining places among other. These places normally carry out activities with poor environmental safety standards thereby realising heavy meatal substances and other environmental hazards in many parts of Ghana and the world at large (Lu et al., 2015; Mackay, Taylor, Munksgaard, Hudson-Edwards, & Burn-Nunes, 2013; Podolský, et al., 2015).

Other identified point-source pollution and contamination are aromatic hydrocarbons and poisonous metals, which are usually present in oil products. The pollution ranges from the leaking of installed tanks from pump stations which cause the release of aromatic hydrocarbons and other heavy metals into the soils and groundwater there by exceeding the WHO allowable threshold limits (Fritt-Rasmussen, Jensen, Christensen, & Dahllöf, 2012). Also are some leakages that occur from oil refinery which occurs accidentally (Bayat, Hashemi, Khoshbakht, & Deihimfard, 2016). Mostly, point-source pollution is predominant in urban centres than in rural areas. Soils and groundwater at urban centres have high heavy metal concentration and also high aromatic hydrocarbons and other forms of pollutants. (Kim et al., 2011; Kumar and Kothiyal, 2016). Abandoned landfills where solid wastes were discharged according to their known toxicity or where they were discharge properly are also identified as point-source pollution to soils and groundwater (Bauman-Kaszubska and Sikorski, 2009).

Diffuse Source Pollution

Diffuse pollution is that pollution that the pollution occurs or accumulate in soils at a wide geographical space and does not have a single or particular identified source of pollution. It mostly occurs when there is an emission, transformation as well as dilution of pollutants or contaminants in other environmental media and later enters into the soil and subsequently gets into groundwater. This pollution equally includes ones that transcend via air-soil- groundwater systems (FAO and ITPS, 2015). In assessing the movement of pollutants through these three environmental compartments, Geissen, et al. (2015) stated that it requires complex analyses to know the movement of the pollutants so that the pollution can properly be addressed. This makes analyses of diffusion pollution very complex and difficult and also challenging when tracking the movement of the pollutants in space.

In a study made by Grathwohl and Halm (2003), many of these contaminants and pollutants that pollutes soils and groundwater in mainly from this source since their fate in the is not always well known and understood. There are several examples of diffuse pollution which include nuclear weapon activities, contaminated effluents released into wetlands and water catchment areas, poorly managed dump sites; uses of fertilizers, weedicides and pesticides for agricultural activities; continues organic pollutants; transportation of excessive nutrients and agrochemicals downstream of a river, by runoff or by flooding; atmospheric transportation and deposition; and soil erosion of pollutants. Diffuse source pollution had adverse effect of both human health and the environment even though its impact and extent are not always unknown.

Landfill as a Source of Groundwater Pollution

According to Postigo & Miglioranza (2018), the pollution of groundwater by human-induced activities started during the Holocene period when man became sedentary and began to farm. From then urbanisation and agricultural activities have intensified and have yielded a substantial amount of nitrate from agricultural activities and pollutants from waste generated from urban and industrial waste. This has released nitrogen from agricultural fields and bacteria and pathogens from sanitary landfill sites to spread into aquifers (Kanmani & Gandhimathi, 2013). According to (Opoku, Boateng & Akoto 2019; Egbi et al., 2015), landfills have been identified as one of the major sources of groundwater pollution. poor construction, poor maintenance and mismanagement (Lee & Jones-Lee, 2005).

According to Kanmani & Gandhimathi (2013), leachate is produced when fluids gets into landfill and mix the contaminant in the waste to produce moisture content which is more sufficient to flow beneath or on the surface of earth. This leachate produced from these municipal solid waste sites according to Boateng et al., (2018) generally has high concentration in both trace metals such as calcium, magnesium and potassium; heavy metals like Manganese, chromium, lead, copper, arsenic and cadmium as well as organic compounds like chloroform, acetone, phenol and Polycyclic Aromatic Hydrocarbons (PAHs). Danutsui (2012); Kanmani and Gandhimathi (2013) further stated that the concentration of these elements in the leachate depends on the waste type, the hydrology of the site, moisture content, temperature, oxygen present, degree of compaction, and sizes of the particle.

In a study on physicochemical characteristics of leachate produced at Oblogo landfill site in Accra by Danutsui (2012) indicated a high concentration of 2,630 S/cm and 8,169S/cm respectively from two different observation points. Beside TDS which represent the general quality or salinity of the water was reported to be 1,611 mg/L and 3,940 mg/L for respective two sections on the Oblogo landfill site. It is an indication that when leachate from the landfill site finds its way into the local groundwater it can compromised the water's quality. Egbi et al., (2015) reported a pH value of range 6.98 to 8.78 at the Abloradjei landfill waste disposal site. These values were assumed to be so due to a possible neutralisation of the low acidic condition by bicarbonates and hydroxides as well as ammonia produced at the site. 1171 mg/L to 3040 mg/L and 1052 to 4720mg/L were recorded respectively for both Sodium and Potassium on a leachate analysis. With cations, Calcium concentration was recorded to be 234.98mg/L and Magnesium recorded from the site was 517.04mg/L. Sulphate concentration was recorded to be 24.24 to

47.9 mg/L. These figures indicate how high contaminants that pollute groundwater at landfill sites.

Movement of Contaminants and Fate

As stated by Mato (2012) movement of groundwater is regarded in its potential of transporting dissolved substances and solutes which can be either a natural chemical or contaminant or anthropogenic. During groundwater movement, solutes move either by advection which is a bulk transport of solutes. When small number of solutes being it contaminants or artificial tracers get out from a polluting source into an aquifer, the contaminants disperse away from the advective flow part, form a plume of diluted solutes and broaden the flow path along and perpendicular to the advective flow direction (Bijeljic and Blunt, 2006).

Generally, there are two dominant processes with which this phenomenon occurs. Firstly, at the initial flow, there is a molecular diffusion of chemical constituent that flow in direction with the concentration gradient which is caused by kinetic energy of the solute constituents. The second process that broadens the flow path of solute is mechanical dispersion. This comes as a result of tortuous nature of the soil pore channel found with aquifer and the fissures present in the fractured aquifer. Again, the flow path of the solute is broadened by differences is the flow rate of the groundwater which moves within fractured aquifer channel which have different widths.

In dual porosity aquifers, groundwater solute concentrations according to UNEP (1996), differ significantly in fissured aquifer than in matrix aquifer. In either diffuse or point source pollution, contaminated water moving within a fissured aquifer has a higher concentration of contaminants than when it moves in matrix which is also known as “the pore-water”. When the source of the contaminant or the pollutant is removed, the concentration of the solute flowing within the fissure reduces in the pore water and diffusion may occur in opposite direction

Sampling groundwater at this stage using conventional method sample water in the fissures and possibly can miss important accumulated pollutants present in the aquifer matrix. Whenever pollutants get into aquifer through the immiscible phase, transportation of the pollutants is regulated by different factor completely from those factors that influences groundwater flow predominantly the viscosity and the density of the immiscible fluid (Dagan & Neuman, 1997). Aromatic hydrocarbon has less density than water causing them to flow faster than the groundwater. At this phase, due to their density, they float on the water table and their lateral migration is dependent of the hydraulic gradient. At here, the pollutants rise when there is a rise in the water level. However, when the water table recesses, the hydrocarbons are at time trapped at the surface of the pore space within the unsaturated zone by surface tension. On the contrary, chlorinated solvents present in the immiscible phase have higher densities than the flowing water. Due to its high densities, it is more viscous and has less velocity than the moving groundwater. These properties of chlorinated solvents result in aquifer up take of chlorinated solvent. These solvents when reach the base of the aquifer accumulated under depression and or they migrate downslope irrespective of the flow path or the direction of the groundwater (Schwille, 1981).

One key factor that controls the continuity of solvents in groundwater is the degree to which immiscible phase displaces water from fine grained porous matrix as exampled in aquifer from fissured limestone. The displacement of the solvent depends on the surface tension properties the immiscible flux possesses relative to the groundwater and rock minerals that forms the aquifer matrix (Lawrence and Foster, 1987). Normally, a high excess of pressure is required before the immiscible phase of most organic fluids can penetrate into the aquifer matrix. When these immiscible dense solvents accumulated within the aquifer, it is able to burry within parts of the aquifer matrix and serve as a source of pollution to groundwater for several number of years. As it accumulates in the water, its rate of dissolution is a function of the water compound, the flow rate of the groundwater as well as the degree at which mixing of the solvent and the groundwater is permissible due to the spreading of the solvent in vis a vis the of the local hydraulic and flow pathways of the groundwater. The complex nature of immiscible contaminats and pollutants creates considerable number of difficulties in designing assessment programs and interpreting monitoring data.

Interaction of contaminants with soil and geologic structures is a very key factor when assessing the fate and movement of contaminants in groundwater flow system. Contaminants like sodium chloride (NaCl) which is very soluble readily moves from the top surface to the saturated materials below the groundwater table mostly during or few times after rainfall. On the other hand, contaminants that are not highly soluble have considerably longer residence time before they reach the saturated zone. Some contaminants adsorb promptly onto soil particles and gradually dissolve during the hours of precipitation and infiltration in the vadose zone resulting in dissolved fractions of concentrations of pollutants or contaminants moving into groundwater (Hunt and Ewing (2009). There are several modes of this movement and it includes Advection, adsorption, dispersion, diffusion, and sorption.

Advective Movement of Groundwater

Advective flow is the most common form of contaminant migration within the subsurface flow. The term advective flow according to (Bijeljic and Blunt 2008) is the transfer of contaminants or solutes by bulk movement of groundwater. The flow velocities are based on the average bulk properties of the aquifer material and the average hydraulic gradient causing the flow. Darcy's law is the basis for quantifying the rate of fluid flow through saturated subsurface material. According to Digmann, (2003), the velocity of the bulk movement of groundwater is the average linear groundwater velocity or the advective velocity when it involves the transport of solutes in the groundwater medium. The advective equation in a one-dimensional flux for the transport of is given by $J= vx.\ Cne$ - equation 2

Where:

J= mass flux per unit area per unit time

Vx= average linear groundwater velocity in the direction of the flow

C= concentration in mass per unit volume of solution

ne= effective porosity of the geological medium.

This equation assumes that the movement is not affected by the pattern of flow.

Contaminant Reaction and Degrading

Some pollutants and contaminants do go under biological reactions. radioactive decay as well as chemical reduction and oxidation. Over all, the contaminant and pollutant concentration decrease with time due to these reactions making them less harmful and toxic than they were before. However, there are some molecules, pollutants and some contaminants that do not degrade and they are known as conservation molecules or contaminants/pollutants. In many instances, pollutants released from landfill sites form a plume within the subsurface of the landfills. Their concentration within the subsurface at a point in time depends on effects of sorption, dispersion, advection among other chemical reactions (Sangodoyin & Agbawhe, 1992). Within aquifer dynamics, aquifer parameters, pumping wells, aquifer flow rate among others give more heterogenous patterns in three physical pattens or directions in time.

In real aquifers heterogeneities in the contaminant characteristics, aquifer flow fields, aquifer parameters, recharge events, pumping wells, etc., produce much more complex patterns in three physical directions and in time.

Groundwater flow

According to Harter (2001), the movement of groundwater is from a higher gradient to a lower gradient and also from a high-pressure location to a low-pressure location. Its general movement is generally slow as compared surface water with a movement usually less than 0.3 meters within 24 hours with few in some extreme cases 3.3 meters in 24 hours. The general flow is governed and understood by Darcy’s Law which states that “the rate of flow is directly proportional to the hydraulic gradient”. According to UNEP (1996), the principal driving force which moves groundwater is the hydraulic head. Its movement is in the downward direction relative to the hydraulic head gradient. Groundwater moves with the syndrome "no hydraulic gradient, no flow". This hydraulic gradient (*dh/dl*) according to Hillel, (2004) is the difference that exist between the potentiometric surface at two given points divided by the horizontal distance between these two points.

As stated by Lawrence and Foster (1987), the geological formation of the aquifer within which groundwater moves has a significant impact on its movement. To them, groundwater moves rapidly and faster in sand and gravel than in clays. The geologic materials define the hydraulic conductivity. According to Wainwright and Mulligan (2004), hydraulic conductivity measures the permeability rate of the geologic material. That is that hydraulic conductivity measures the amount of flux that can pass through a given geologic material within a defined time period. These measurements are done in gallons per day (gpd) per square foot (pft2). High hydraulic conductivity indicates high permeability or flow discharge even if there is a constant hydraulic gradient.

UNEP (1996), stated that discharge rate (hydraulic conductivity) of the sandy aquifer is between 100 to 10000 gpd/pft2 who is almost 10 to 1000 ft/day. The hydraulic conductivity for clay which its particles restrict water movement has a microscopic hydraulic conductivity of 0.001 gpd/ft2 or less as compared to that of sand. For fractured rocks, the hydraulic conductivity depends on the extent of the fracturing. Harter (2001) indicated that the rate of groundwater movement is a product of hydraulic conductivity and hydraulic gradient with adjustment of the void spaces of the soil material ranging from 5 to 20%.

Groundwater velocity = which is mathematically represented as 'Q' = - $K\ i\ A$ where Q is the rate of discharge, '-' is the negative pressure, 'A' is the cross sectional area, 'i' is hydraulic gradient and 'K' is the hydraulic conductivity.

Groundwater Flow within the Unsaturated Zone.

According to Hillel (2004), most of the processes that go on between soil and water including the supply of moisture and nutrients and transporting solutes beyond root zone occur predominantly within the unsaturated zone. The flow processes within the unsaturated zone are very complicated and describing it quantitatively is difficult. The difficulty arises from the change in state it undergoes during the flow period. Some of the changes it undergoes include a change in the soil wetness, suction, and conductivity whose interconnections are extra complicated by spatial variations and hysteresis.

In view of this, the determination of movement of water within the unsaturated zone cannot be the same as the one for the saturated zone. This is because the soil water pressure must be greater than the atmospheric pressure before it can freely move to fill void spaces. Special methods are therefore employed and applied to describe and measure the movement of fluids and contaminants within this zone (Harter, 2001).

As stated by Wainwright and Mulligan (2004) that hydraulic potential gradient is the main driving force that moves flux within the soil medium and that its flow rate is proportional to the potential gradient and the soil material properties through which is flows within the saturated zone, Dingman (2003), stated that similar processes also goes on within the unsaturated zone. Unsaturated zone, unlike the saturated zone where movement is under positive pressure potential, the unsaturated zone, movement of fluids under negative pressure potential or matric suction. The matric suction is caused by the affinity that goes on between the moving fluid and the soil matrix (both soil matrix and capillary pores). Dingman (2003) and Hillel (2004) further explained that there is no driving force when there is a uniform suction within a soil column horizontally. To them when this occurs water is drawn from a thicker zone where hydration engulfs to soil particles to a thinner zone and to a less curved capillary meniscus to a high curved capillary meniscus. This is there is a spontaneous flow of water from a low matric suction to a high matric suction.

According to Hussain and Nabi (2016), there is a partial differentiation between the hydraulic conductivity for the saturated and the unsaturated zone. Due to the partial filled pores within the unsaturated, there is always a continuity with the water phase thereby making conductivity maximal within the saturated zone. With the unsaturated zone, the partial fill of pores with air makes the conductive section of the soil's cross-section area reduce hence not attaining a maximal conductivity. During the period if water reduction within the pores by suction, the large pores which are most conductive are emptied first leaving flow to occur within the smaller pores. In a coarse-textured soil, water is confined within the capillary wedges to form a separate and discontinuous perch of water

CHAPTER THREE

MATERIALS AND METHODS

Introduction

This chapter describes the various methods that were used for the study. Specifically, the chapter provides the research design, brief description or profile of the study area, the unit of analysis, instrumentation and data collection, data processing and analysis.

Research Design

True experimental research design was used for the study due to the mode of the data collection and the analysis that was made. The soil samples were collected from the landfill site and the surrounding community. The soils sample from the landfill site were used as an experimental group and the sample taken from the surrounding community was used as a controlled group for comparison of results of the soil chemical properties from the landfill and the surrounding community. The design according to Creswell (2011); Leedy and Ormord (2001); Kothari (2004); Williman (2011) helps in the establishment of cause and effect that exist between phenomenons.

In the assessment of the effects of the landfill to the soil, experimental design provides the framework within which results of the soil from the landfill and from the surrounding community could be compared and make judgement on the impact of the landfill on the soil. The leachate on other hand was used as an experimental group and the surrounding groundwater also as controlled group. The results of the chemical constituents of the leachate and the surrounding groundwater were made to evaluate the effect of the leachate produced from the landfill has on the surrounding groundwater.

Brief description of the Study Area

The Oti landfill site, according to (Sulemana, et al., 2015) which belongs to Kumasi Metropolitan Assembly even though it is within the Asokwa Municipal Assembly. The Oti township is a suburb of Kumasi and a residential community under the Asokwa Municipal Assembly. The Oti Sanitary Landfill on latitude 6.35°–6.40′N and longitude 1.30°–1.35′W with a digital address of AK-762-2122. The site is within the plateaus of the south-west with an elevation range of about 250 to 300 meters above sea level (Iddrisu 2012). It shares boundaries with Kaase and Atonsu Kuwait in the north and north-eastern respectively, Dompoase in the south-east and Aprabon in the south.

The climatic characteristics of landfill site falls within the wet sub-equatorial belt of Ghana which has an average diurnal temperature of about 26.1°C and an average minimum temperature of 21.5°C and a maximum of 30.7°C and an average humility of about 84.2% during sunshine and 60% during sunset. The study area has a double maxima rainfall regime of about 214.3mm in June and 165.2mm in September.

According to Murray et al., as cited in Ewusi et al., (2016), the geological formation of the study area is primarily dominated by Birimian meta-sedimentary units of middle Precambrian origin. It is of dark grey phyllites, phyllites, tuffaceous phyllites, and greywackes. The rocks are intensely fractured due to intense heat and fresh outcrops of phyllites. Quartz veins intrude the phyllites and occur all over the area is underlain by the Birimian meta-sedimentary units. The major soil types are the Forest Ochrosols with its detailed soil associated with the Bekwai–Oda soil series which is very rich in soil nutrients and for that matter enables the practices of urban agriculture within the area (Iddrisu, 2012). The soil aquifer of the are study area is graded as shallow with a depth range averaging 7- 10 m thickness. Documented soil information from borehole and hand-dug drillings show that the soil strata were covered with rubbish.

According to Owusu-Sekyere, Osumano and Yaro (2013), the landfill facility receives about 1700 metric tonnes of waste daily. The wastes range from households' solid waste through municipal and industrial waste to human faecal matter which are generated within the metropolis. The waste also includes hospital waste and waste from Kaase slaughterhouse. These wastes are not sorted neither at the source and nor the landfill site but dumped together in the landfill. Pre-treatment of the waste is also not practised before dumping. The site has a large volume of buried waste because it has been in operation since 2004. The facility design has a life expectancy of 15 years but according John Stanley-Owusu Group Limited, who manage the day-to-day operations of the facility, said it could go beyond the expected life span.

The facility covers 404.686 m^2 with nine stabilisation ponds for leachate treatment and also has a septage treatment plant which was attached with the landfill (Boateng, Opoku & Akoto, 2018). However, field recognisance survey showed that the treatment ponds and the septage treatment plants all do not function. The non-functioning of the leachate treatment plant and the peaking level of the landfill makes leachate flow through drain channels and join a downstream river which flows southwards and finally joins Oda river. The faecal was is also directly discharges into a drain south of the landfill and this and also runs downwards to join the downstream river. The facility is surrounded with river and wetlands at the southern part where these wastes drain into.

Before the commencement of the facility in 2004, the entire surroundings were with no residential establishment. However, rapid population growth coupled with unplanned residential development has resulted in the encroachment of the surrounding environment (Boateng et al., 2019). Due to encroachment, there is no defined buffer zone from the facility to the surrounding communities as some residential buildings less than 5 metres (Field survey, 2017). Besides, due to the intermittent water supply from the Central Water Supply System from Barekese by Ghana Water Company, the main source of water for many residences in the surrounding communities is the ground water. They tap it either from a mechanised borehole or a hand-dug borehole.

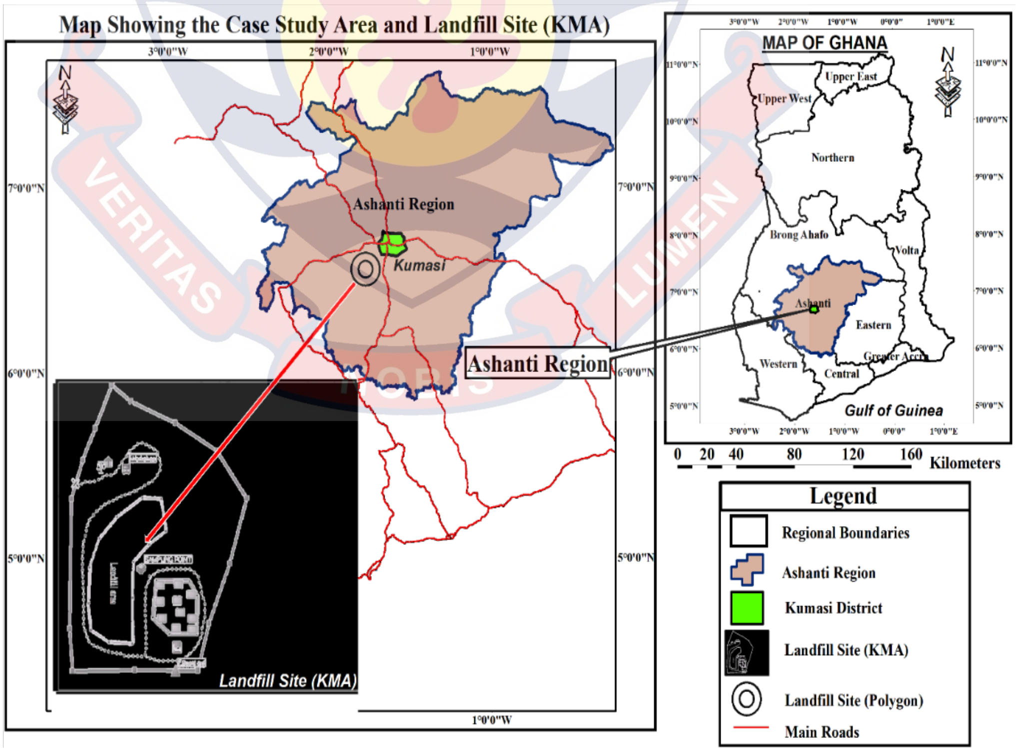

Figure 1: Map of Ghana showing the study area (b) study area in details.

Source: Author construct, 2020.

Units of Analysis and Sampling Procedure

A unit of analysis is the various objects, artefacts or individuals that scientific enquiry looks at Kothari (2004) The units of analysis for the study were soils, leachates and groundwater that were collected from the study area during the study period. The soils included both the topsoil and subsoil with the depth up to 150 cm from three different parts at the landfill site and a control soil sample from the surrounding community. The leachate sample was the liquid that is produced from the landfill and the groundwater unit was the one collected from the hand-dug wells surrounding the landfill site. Sampling procedure according to Kothari is the process through which similar characteristics are selected from a whole to serve as a basis for estimating and predicting the possible outcome of a situation, regarding a bigger group. That is take a portion of a material which is small enough to be carried away but still large enough to represent the whole material when analysed (Kothari, 2004).

Groundwater Sampling Procedure

The surface area of the hand-dug wells and its consideration as a small facility required the research to adopt a grab sampling procedure for sampling the groundwater in the wells while grab and quartering sampling procedures were also used to sample the leachate. During the sampling of the groundwater, a buffer zone with a radius of 200 m away from the boundary of the landfill into the surrounding community was created. Within this buffer zone, a total of 14 houses were identified having hand-dug wells in their homes. Out of the 14 hand-dug wells identified, 5 were randomly selected for the water sample collection. This is because geographically the area represents a small catchment with similar aquifer characteristics hence 5 wells were considered very representative. Hand-dug wells were chosen ahead of mechanised boreholes because of the interest in assessing total dissolved solids

Soil Sampling Procedure

To achieve spatial equity, the landfill was divided into three different parts. These were the northern, middle and the southern parts and a 150 cm PIT was dug in each part, giving PIT 1, PIT 2, and PIT 3 in the northern, middle and southern parts respectively. Each PIT was divided into three different parts (00-50 cm, 50- 100 cm and 100- 150 cm). On the accounts of knowing the varying changes that occur within the soil cross-section in term of physical and chemical characteristics soil samples were taken every 50 cm of descend. At these various depths soil samples for physicochemical analysis, hydraulic conductivity and other soil physical properties.

Leachate Sampling Procedure

The leachate sample was selected using the composite and quartering sampling procedures (Clesceri, Greenberg, and Eaton, 1998). A 1000ml of leachate was collected from eight different parts of the landfill site. These samples were poured into one container stirred giving a total leachate volume of 8000 ml. This volume was divided into four, a quarter of it was taken and further stirred and divided again into four. The process continued until the required sample volume of 500 ml was attained. This method made the leachate sample very representative with regards to leachate produced from all segments of the landfill.

Data Collection

Groundwater and Leachate Samples Collection

The groundwater and leachate samples were collected in January, 2019 (during the dry season) and June, 2019 (during the rainy season). During the data collection, 500ml plastic bottles were used to collect both the leachate and groundwater samples. Plastic bottles were used over aluminium bottles because the absorption rate of some heavy metals is very high and can easily leach in glass bottles Clesceri, Greenberg, and Eaton (1998). The plastic bottles were first cleaned with acid water and was later rinsed with de-ionized water. As quality control of water sampling requires, the bottles were washed at the field with some of the groundwater before filling it. The water in the hand-dug wells was disturbed for some time through fetching some of the water and pouring it back into the well to enable settled solid to mix up before the water was finally fetched. The 500 ml bottles were filled with the groundwater and preserved with a 2 mL concentration of nitrogen acid (HNO_3). They were put in an ice chest at a temperature of 4^0C with ice blocks and transported to the laboratory for analysis. The depth of these sampled wells was taken using a measuring tape. This helped in knowing the depth of the water table from the surrounding surface. Well one designated as HD1 had a depth of 10.2ft (3.11 m), well two (HD2) had a depth of 4.6ft (1.40 m), well three (HD3) also had a depth of 11ft (3.35 m) well four and five had depths of 26.6ft (7.92 m) and 10.9ft (3.32 m) respectively.

Soils Sample Collection

Three different sets of instruments were used to collect the soil samples. The core sampler with a length of 15cm and a diameter of 9cm was used to collect soil samples for the soil physical properties, earth chisel was used for the soil chemical property samples and pressure pipes a length of 30cms and a diameter of 8.5cm were also used for the hydraulic conductivity sample.

During the collection of the soil sample for the physical soil properties (with the exception of the hydraulic conductivity), core sampler was hammered into the ground to take the required soil sample. Both edges of the core sampler were trimmed to have the same dimensions with the core sampler. Each sample was bagged in sampling bag and was well sealed. Also, pressure pipes were drilled vertically along the soil cross-section to get the varying layers in the soil and filled half the length of the pressure pipe as American Society for Testing and Materials (ASTM) rules and regulation requires. The edges were sealed to prevent the surfaces of the sample from breaking off or loosening.

Data Collection for DRASTIC Vulnerability Index

The statistical results of the samples obtained from the analysis of hydrogeological data from the study area were validated using the DRASTIC model. A DRASTIC Vulnerability Index was determined using the following information on: (i) the depth of groundwater, (ii) net annual groundwater recharge (iii) the aquifer characteristics, (iv) soil media, (v) the topography of the area (vi) impact of the vadose zone and (vii) the hydraulic conductivity of the vadose zone of the area.

The depth to groundwater was measured using 30m measuring tape to measure from the ground to groundwater surface. The net annual recharge rating data was obtained using differences in groundwater surface during the dry and the rainy season. On the aquifer characteristics, data was collected from drilled boreholes documented for the area. Data on the rating of the hydraulic conductivity and vadose zone were collected from documented borehole geology of the area. Also, soil media data was collected from the available documented soil information of the area. Lastly, data on the topography was measured using a clinometer

Data Processing and Analysis

The physical and chemical soil analysis was done at the soil science laboratory at Kwame Nkrumah University of Science and Technology (KNUST), Kumasi, while the leachate and groundwater sample analysis were also done at the Biology Laboratory at KNUST. The soil analysis was done for the bulk density sample, the hydraulic conductivity, and some physicochemical properties of the soil.

Determination of Soil Bulk Density

Bulk density measures the weight of the soil solids per unit volume and it is expressed in grams per cubic centimetre (g/cm^{-3}). The core soil samples were weighed as soon as the samples were taken to the laboratory using Mettler PM 400 digital weighing device to get the weight of the wet soil bulk density (W1). The samples were put on metal trails and oven dried at a temperature of 105^{0}C for 24 hours. They were removed from the oven and cooled for 3hrs with natural air. The oven dried soil was weighed to get the weight of the dry soil (W2). The recorded data was inputted into excel statistical software to generate the bulk density graph.

Calculation of the bulk density.

$Pb = (W2)/Vt$

Where:

Pb = *Bulk density*

$W1$= *Wet soil weight (g)*

$W2$= *Dry soil weight (g)*

Vt= *Total Volume of soil* (cm^3) ($\pi r^2 h$)

r= *radius of the core cylinder.*

h= *height of the core cylinder*

***Determination of Total Porosity* (ϕ)**

The total porosity of soils measures the total volume of space within the soil that is not occupied by solid but occupied by air and water relative to the volume of the entire soil. It is calculated as:

ϕ =

Where:

Φ = total porosity

Pb = dry bulk density (g/cm^3)

Ps = particle density (g/cm^3) = 2.65

Determination of soil moisture content

Determination of gravimetric soil moisture content $(W) = (W1-W2)/W2 \times 100.$

The volumetric moisture content (θ) measures the volume of moist in the soil relative to the volume of the total soil. It was measured by dividing the volume of the soil water by the volume of soil. That is

Where:

θ = volumetric soil moisture content

W = gravimetric water content

Pb = dry bulk density (g/cm^3)

Pw = density of water 1 g/cm^3

Degree of Saturation (S)

The degree of saturation is the fraction of the void space that is occupied by water within a given column of soil. It is calculated by dividing the volume of water by the volume of voids.

Calculation

S =.

Where:

S = degree of saturation

Θ = *volumetric soil moisture content*

Φ = total porosity

Determination of Hydraulic Conductivity of the Soil

Saturated hydraulic conductivity (*Ksat*) measures the drainability of saturated soil. The falling head method as described by Stibinger (2014) was used to determine the hydraulic conductivity. The soil samples with a length of 15 cm and diameter of 8.5 cm were placed in water. The base of the soil in the pressure pipe was covered and tied with a rubber band and was soaked in water until it was saturated through capillary rise. A stand was mounted and a graduated cylindrical test tube with open ends and a measuring ruler was tied to it. A rubber tube with a diameter of 0.66 cm was fixed to the base end of the cylindrical test tube and was connected to the pressure pipe. The saturated soil sample in the pressure pipe was placed on a perforated plastic container filled with gravel (<2 cm). The upper part of the pressure pipe (15 cm) was filled with water and with the connected rubber tube, the hydraulic head was read from the cylindrical test tube. Subsequent 1 cm pace readings were done in relation to time.

Calculation

Where:

K = the conductivity of the soil;

a = the cross-sectional area of the tube (m^2)

L = the length of the soil column (m)

A = the cross-sectional area of the soil sample (m^2)

T = the time taken for the change in the two pressure (s)

In = returning into the natural logarithm;

Ho = final pressure head

Hi= initial pressure heads.

Analysis of the DRASTIC Vulnerability Index and Sensitivity analysis

. The DRASTIC vulnerability index was obtained from the data that were taken for the various seven factors that make up the model using equation 1 (DI = DrDw + RrRw + ArAw + SrSw + TrTw + IrIw + CrCw). Sensitivity analysis was done to check the effectiveness of each rating and weight of each parameter using the single- parameter sensitivity analysis. This analysis compares the ratings of each parameter assigned weights of the model. The single- parameter sensitivity analysis is given by the formula *W=* 100*

Where *W*= assigned weight, *Pr* and *Pw* = assigned rating and weight for each parameter respectively and *V* = DI.

Analysis of chemical properties of the leachate, soil and groundwater

On the geochemical characteristics of the soil, leachates and groundwater, the following parameters were analysed for the water samples: pH, total dissolved solids (TDS), dissolved oxygen (DO), biochemical oxygen demand (BOD), chemical oxygen demand (COD), sodium (Na^+), potassium (K^+), calcium (Ca^{2+}), bicarbonate (HCO_3), iron (Fe), Zinc (Zn), copper (Cu), and arsenic (As). These parameters were chosen because of its relevance in water quality analysis. Parameters analysed from the soil sample also included the $_p$H, Cation Exchange Capacity (CEC) Effective Cation Exchangeable Capacity (ECEC) (Ca^{2+}, Mg^{2+}, Na^+, K^+, H^+, Al^{3+}), Iron (Fe), Arsenic (As), Lead (Pb), Manganese (Mn), Copper (Cu) and Cadmium (Cd). These parameters were also chosen base on the various characteristics it exhibits in soil. This includes how soil pH influences the movement of soil trace metals and how some trace metals like iron absorbs soil other heavy metals within soils.

Determination of the soil pH

The soil pH was measured with the electrometric method using the 1:2.5 soil- water ratio. A 10 g of air dried was put in a 50 ml plastic bottle with a crew cap and 10 ml of distilled water was added and stirred vigorously for 20 minutes. The soil- water suspension was allowed to stand for 30 minutes by which most of the suspended clay particles had settled out from the suspension. The pH meter was calibrated with blank at pH of 4 (highly acidic) and 10 (highly alkaline) respectively. The glass electrode of the pH metre was inserted into the partly settled suspension and the pH value of the result was then read from the pH metre.

Determination of Soil Heavy Metals

A 10 g of soil sample was weighed into Kjeldahl digestion tube. Perchloric, nitric and hydrochloric acid was added in the ratio of 1:2:3. The mixture was then digested at a temperature of 450 ° C until digestion was complete, indicated by change in solution colour to whitish. This usually takes 30 to 60 minutes. After digestion was complete the mixture was decanted into a 100 mL volumetric flask and the solution diluted to the 100 mL. The unknown concentrations of this solution were then read using an atomic absorption spectrophotometer for the various metals at a specified wavelength.

Instruments used

The basic setup (air pressure = 50 – 60 psi, acetylene pressure = 10 -15 psi and voltage = 208 – 240V) of the AAS was ensured. The file for the type of analysis and hollow cathode lamps were selected with appropriate wavelengths of; Cu at 324.8 nm, As at 193.7 nm, Pb at 217.0 nm Cd at 228.9nm, Zn at 213.9 nm, Fe at 248.3nm. A calibration curve was plotted for each of the elements to be analyzed from the stock standards (Buck Scientific). The prepared sample solution digest was analysed for the elements. The Y in the calibration equation is absorbance of the element and X is the concentration of the element in the sample. X was calculated after substituting the absorbance reading of the sample into the calibration equation. This gave X in terms of mg/L. The total concentration of the element in the sample solution (100 ml) was calculated by multiplying the concentration in mg/L by 0.1L. This gave the total mass of the element in solution. The percentage amount of the element was found by dividing the mass of the element in solution by initial amount of sample taken followed by a multiplication by 100.

Calculation:

Conc. (Cu, Pb, As, Mn, Fe, Cd) (mg/kg) =

$$\frac{\text{Concentration recorded from AAS} \times \text{Nominal volume}}{\text{Sample weight (g)}}$$

Where,

Nominal volume =100 ml

Sample weight = 1.00g

Exchangeable Cations

Exchangeable cations base are those cations that are replaced by the soil's cations. The exchangeable cations Ca^{2+}, Mg^{2+}, K^{+}, Na were determined in 1 M NH_4OAc extract of soil (FAO, 2008). Approximately 10 g of air- dried soil sample was weighed into 100ml extraction bottle and 20 mL of ammonium acetate solution was added and stirred. The solution was allowed to stand for 24 hours. The suspension was then filtered and the soil was leached with 20ml of ammonium into 100 mL volumetric flask allowing the filtering to drain between each addition. The filtrated was made up of 100 mL mark with ammonium acetate. The Ca^{2+}, Mg^{2+}, K^+ and Na^+ were then determined using the aliquots as follows:

The exchangeable K^+ and Na^+ were determined using flame photometer. The working standards of 0, 2, 4, 6, 8, and 10 µg mL^{-1} of both K^+ and Na^+ were prepared in ammonium acetate, aspirated into the flame photometer to record the emissions. The soil extracts were also aspirated and their emissions were also recorded. After that, calibration curve was drawn using their emissions and standards. The K^+ and Na^+ concentrations in the samples were read from the curve.

The exchangeable Ca^{2+} and Mg^{2+} were also determined using titration methods. The method involved the chelation of cations with ethylene diaminetetra-acatic acid (EDTA). The traditional procedure involved the determination of Ca^{2+} and Mg^{2+} together using solochrome black indicator. In determining the Ca^{2+}, an aliquot of 25 ml of the extract was placed into 250 ml conical flask and the solution was diluted to 150vml with distilled water. Some 10 drops each of KCN, $NH_2OH.HCl$ and triethanolamine (TEA) were added; NaOH was also added to raise the pH to slightly over 12. Five drops of calcon indicator was added and the solution was titrated from red to blue end point with EDTA.

The exchangeable Mg^{2+} was also determined by placing an aliquot of 25 ml of the sample extract in 250 ml and adding distilled water to make a total volume of 100 ml. A 20ml of 20% tungstate solution and enough buffer was added to it to obtain a pH of 10. The solution was then heated and the content was filtered using filter paper. The paper and the precipitate were washed with a solution containing 50 ml of buffer per litre. Some 10 drops each of KCN, $NH_2OH.HCl$, $K\,Fe\,(CN)_6$ and TEA were added and allowed to stand for few minutes for each reaction to take place. Afterwards, some 10 drops of EBT indicator was added and the solution was titrated from red to a permanent blue end point with EDTA.

Exchangeable Acidity

The acidic cations Al^{3+} and H^+ were extracted with KCl solution. Approximately 10 g of soil sample was weighted into a baker; a 30 ml of 1 M KCL was added and allowed to stand for 24 hours. The soil was successively leached with 10 ml of KCl into 100 ml volumetric flask and the solution was made up to the mark. A 50 ml of the KCl extract was pipetted into a 250 ml conical flask and 5 drops of phenolphthalein indicator was added. The solution was titrated to a pink end point with NaOH.

Calculation

$cmol_c\ K^+kg^{-1}$ =

$cmol_c\ Na^+kg^{-1}$ =

$cmol_c\ Ca^{2+}kg^{-1}$ =

$cmol_c\ Mg^{2+}kg^{-1}$ =

$cmol_c\ Al^{3+}kg^{-1}$ =

Where:

C = concentration of extract from standard curve

T = sample titre value

Wt = weight of soil sample used

Exchange Capacity was calculated by summing up exchangeable bases and exchangeable acidity. (ECEC= $Ca^{2+} + Mg^{2+} + K^+ + Al^{3+} + H^+$)

Determination water quality parameters of groundwater and leachate sample.

Measurement of pH, TDS, and the EC

The pH meter (HANNA model 209) was used to determine the pH of water samples. The pH electrode was cleaned with distilled water and then calibrated in order to give a precise measurement using pH4, pH7 and pH10 standard buffer solutions before used to measure the water samples. After calibration of the pH electrode, 50 ml of the water sample was poured into a small beaker. The pH electrode was immersed into the water sample and the pH reading was shown on the LCD of the meter. The pH was recorded after the reading stabilized.

Thee electrical conductivity (EC) and total dissolved solid (TDS) measured with a digital conductivity meter with cell constant of 1.0. The instrument was kept clean and standardized with KCl solution before it was used. Distilled water was used to rinse the electrode and lowered into the water sample contained in a plastic container. The conductivity in µS cm^{-1} of the sample was recorded. The TDS of the samples were also measured by selecting the appropriate TDS key while the electrode remained in the water sample used to measure conductivity and the TDS value was recorded. The conductivity of the water sample was read on the LCD screen of the meter. The probe was rinsed in de-ionized water after final reading was taken. This was repeated for all the water samples at the various sampling sites.

Determination of DO, BOD and COD

` The dissolved oxygen was measured using PCD650 EUTech instrument. The probe was dipped into the sample and the range set to dissolved oxygen. It recorded readings to two decimal places. Magnesium sulphate solution (MgSO4 7H_2O) was used to prepare the BOD 300 ml sample was pipetted into a BOD bottle containing aerated dilution water. Firstly, the DO content was determined and recorded and the bottle was incubated in the dark for five days at 20°C. At the end of five days, the final DO content was determined and the difference between the final DO reading and the initial DO reading is calculated. The decrease in DO after the 5 days' incubation was corrected for sample dilution, and represents the biochemical oxygen demand.

Several boiling stones were placed in a reflux flask with an aliquot dilute of 50 ml and 1 g of $HgSO_4$ 5 ml concentration of H_2SO_4. It was swirled until the mercuric sulphate was dissolved. The reflux flask was placed in an ice bath and slowly added a swirling 25 ml of standard potassium dichromate $K_2Cr_2O_7$ solution (0.250 N). 70 ml concentration of H_2SO_4 containing 23.5 g silver sulfate, (Ag_2SO_4), per 4.09 kg bottle was added and was stirred for about 30 minutes for the silver sulfate to dissolve. The reflux and the flask were heated for two hours and was allowed to cool down and the condenser was washed with 25 ml of distilled water. The solution was diluted with acid solution to about 300 mL with distilled water and allow the solution to cool to about room temperature. After that 10 drops of ferroin indicator was added to the solution and titrate the excess dichromate with 0.25 N ferrous ammonium sulfate solution to the end point changing the colour from a blue-green to a reddish hue

Calculation

BOD, mg/L = [(Initial DO - Final DO) x 300]/mL

Determination of Ca^{2+}, Na^{+} and K^{+}

A 100 ml of the water sample was put into a 250 mL conical flask. 4 ml sodium hydroxide solution was added to the contents of the flask followed by the addition of about 0.2 g murexide indicator. The content in the conical flask was titrated against 0.02 M EDTA to end point. This is indicated by a pink colouration. Titration was repeated until a consistent titre was obtained. The sample aliquot was digested in nitric acid, and diluted appropriately. The aspirate and the absorbance were measured at 248.3 nm using UNICAM 969 SOLAAR 32 Atomic Absorption Spectrophotometer. The readings were compared to identically prepared standard and blank solutions, using an air-acetylene oxidizing flame.

Determination of Zn, As, Cu and Fe

Standard procedure of open digestion set by American Public Health Association (1998) was employed for the digestion of water and leachate samples. 50 ml well-mixed, acid preserved sample was measured and transferred into a 42 beaker. 5 ml of concentrated HNO_3 was added to 50 ml of the water and leachate sample. The mixture was heated slowly to evaporate to a volume of about 15 to 20 ml on a hot plate. Continues heating and adding of concentrated HNO_3 as necessary was employed until digestion was complete as shown by a light- coloured, clear solution. The walls of the beaker were washed down with double distilled water and then filtered with a 0.45 µm pore filter paper. The filtrate was transferred to a 50 mL volumetric flask and topped to the mark. The digested samples were used to measure the individual metal concentrations in the water using an atomic absorption spectrometer (AAS) (Spectra 220). Blanks were also made going through the same procedure but without the samples.

Software Analysis of Data

With the statistical analysis, General Statistics software version 12 was used to analyse the results. Some of the basic statistical parameters observed included the mean value, standard deviation and correlation analysis. The correlation analysis was used to check the associations between variables of the single sample. The results were reported at a significance level of 5 per cent using Fisher's probability.

CHAPTER FOUR

RESULTS AND DISCUSSION

Introduction

This chapter presents and discusses the results that were obtained from Oti landfill site at Kumasi and the hydro- geophysical properties of the soil and its effects on the local groundwater system. This chapter included the physicochemical properties of the soil, the leachate as well as the groundwater. Also presented is the DRASTIC Index (DI) that showed how the depth to groundwater, rainfall of the areas, the aquifer characteristics, the type of soil media, the topography of the area, the groundwater net-recharge as well as the hydraulic conductivity of the soil can contribute to the leachate produced from the landfill polluting the groundwater.

Generally, the chapter is divided into four main sections. The first and second sections are on the results and discussion of the physicochemical properties of the soils sample. The third section is on the physicochemical characteristics of the leachate and groundwater sampled and the fourth section is on the DRASTIC Vulnerability Index (DI).

Physical Properties of the Pit Soil

From Table 7, Pit, 2 50 -100, Pit 2 (50 -100 cm) had the highest bulk density of 1.19 g/cm^3 and followed by 1.09 g/cm^3 of (Pit 1, 00-50 cm3). The least compaction of 0.82 g/cm^3 was observed at (Pit 2, 00-50 cm) followed by 0.92 g/cm^3. The results showed no significant difference (F-pro 0.65) in bulk density at Pit 1. However, differences (F-pro <.001) where observed at both Pit 2 and 3 indicating that the soil physical properties changed with depth. At Pit 3, the difference occurred between depth 00-50 cm and 50-100 cm while at Pit 2, differences in the bulk density occurred at all levels. However, Table 4.1 reveals no significant difference (F-pro 0.24) in the three Pits put together at one unit. To Hillel (2004), bulk density increases with depth but from Table 4.1a, the bulk density results did not show any particular order. The low bulk density of the site indicates that the proportionate part of pore spaces within the soil was high and could contain more air and flux were greater than the space occupied by solids.

The bulk density values from all the three Pits at the Oti landfill were lower than the standard particle density range of 2.5 g/cm^3 – 3.0 g/cm^3 as given by Hillel (2004) and soil compaction Table provided by Keller, (2004). According to Sheard, the degree of compaction of soil is low if the bulk density is less than 1.3 g/cm^3. This suggests that the compaction of the soil at the landfill site generally was low. According to Yuanshi, Qiaohong, and Zongjia (2003) low bulk density in soil samples shows that the mass of solid particles per unit volume of soil was low.

The porosity for the study area was generally high with the least porosity value of 58% occurring at both Pit 2 (50-100 cm) and Pit 3 (100-150 cm) whiles the highest value of 69% occurred at Pit 2 (00-50 cm). The porosity, like the bulk density, did not show any particular order of decrease with depth. Pit one did not show any statistical difference (0.62) in the pores of the soil while Pit 2 and 3 showed a significant difference (F-pro 0.023) and (F-pro 0.002) for Pit 2 and 3 respectively. From Pit 2, the difference in porosity was observed between depth 00- 50 cm and 50- 100 cm while at from Pit 3, the difference in porosity occurred at depth 50 – 100 cm. However, while some of individual pits showed a difference in porosity with depth, the combined results of the 3 Pits did not show any changes (F-pro 0.45) in porosity with depth. This means that the average pores within the soil did not change with depth. This can cause the infiltration of fluids within the soil to move at a relatively same pace.

The greater the porosity value, the lesser the bulk density value and vice versa. This indicates that the porosity of the soil was influenced by the compactness level of the soil. Like the bulk density, the porosity did not follow a specific trend with respect to depth and this can be attributed to the excavation of the area during the construction period. All the values obtained were greater than 39.6 % obtained by Adu-Bitherman (2011) at University of Cape Coast Research farm.

With regards to the volumetric water content as shown in Table 7, Pit 2 had the highest volumetric water content value of 0.28, 0.18 and 0.21 followed by Pit 1 with 0.20, 0.16 and 0.18. Pit 3 had the least water content with 0.10, 0.16 and 0.07. From the Table 7, it was seen that a significant change (<.001) in volumetric water content with depth only occurred at Pit 3. The low volume of water content in the landfill soil can be as a result of the time the samples were taken (January, 2019) when the place had experience low volume of rainfall and high rate of evaporation. The statistical result (F-pro = 0.57) from the combined Pits also showed no significant differences between the water content at the various depths. This indicates that the volume of leachate that infiltrates into the soil do not vary with depth.

Table 7: Analysis of individual Pits' Soil at Oti Landfill Site

PIT	Depth(cm)	Pb(g/cm3)	Φ (%)	θ	S %	K(mm/h)
1	00-50	1.15	0.65	0.26	33.95	0.23
	50-100	1.13	0.67	0.3	35.36	0.23
	100-150	1.08	0.64	0.21	25.48	0.21
	SED	0.08	0.03	0.06	4.04	0.04
	F-pro	0.65	0.62	0.36	0.09	0.89
2	00-50	0.88	0.69	0.34	40.39	0.23
	50-100	1.193	0.583	0.2	32.77	0.58
	100-150	1.07	0.6	1.54	118.5	0.93
	SED	0.04	0.03	0.55	36	0.46
	F-pro	<.001	0.023	0.09	0.1	0.38
3	00-50	0.95	0.61	0.11	16.92	0.21
	50-100	1.07	0.68	0.19	23.8	0.22
	100-150	1.08	0.58	0.07	12.57	0.14
	SED	0.01	0.01	0.01	0.01	0.01
	F-pro	<.001	0.002	<.001	<.001	0.003

Pb= bulk density, ϕ = porosity, θ = volumetric water content, S = degree of saturation, K = hydraulic conductivity, SED = standard deviation, F-pro= Fisher's probability

Source: Fieldwork, 2019

The degree of saturation from Table 4.1a also shows that the wetness level of the soil was also low. The highest saturation was 40.33 % (Pit 2, 00-50cm) followed by 34.17% (Pit 2, 100-150 cm) while Pit 3, (100 – 150 cm) had the least saturation of 12.57 % followed by 16.91 % (Pit 3, 00-50 cm). It results revealed no significant difference (F-pro=0.09) and (F-pro=0.1) for Pit 1 and 2 respectively while at Pit 3 significant difference (F-pro= <.001) was occurred at all levels of depth. The combined result from the three Pits also showed no statistical changes (F-pro=0.69) with depth. In general, the flux saturation at the landfill varied from Pit to Pit with Pit 3 having the least saturation of water in the soil.

The results of the volumetric water content and the degree of saturation shows that the higher the water content the higher the degree of saturation relative to the total pores. This is seen in the definitions for the two parameters that there is a positive relation between soil water content and degree of saturation. The statistical value of (F-pro = 0.86) shows that there are no significant differences between the soil wetness with depth.

Again, on table 7, the hydraulic conductivity calculated from the landfill shows a very slow movement of leachate into the soil. The highest conductance value was recorded at Pit 2 (00 – 50 cm) with a value of 0.93 mm/h and the least hydraulic conductivity of 0. 14 mm/h recorded at Pit 3 (100-150 cm). Pit 1 had a calculated conductivity value of 0.23 mm/h, 0.23 mm/h, and 0.21 mm/h at varying depths. From Pit 2, the hydraulic conductivity was 0.23 mm/h, 0.58 mm/h, and 0.93 mm/h at the various varying depth. Pit 3 also had calculated values of 0.21 mm/h, 0.22 mm/h and 0.14 mm/h at varying descend of depth. The statistical analysis showed no significant difference for Pit 1 and 2 while at Pit 3, there was a significant change (F-pro= 0.003) at depth 100 – 150 cm. This means that during the movement of leachate the rate at which the fluids infiltrate changes along its path.

Comparing the hydraulic conductivity and the bulk density results, it could be seen that the compactness of the soil to some extent influenced the movement of leachate in the soil. The smallest bulk density value of 0.88

g/cm^3 (Pit 2, 00-50 cm) had a conductivity value of 0.23 mm/h while the highest bulk density 1.19 g/cm^3 (Pit 2, 50-100 cm) had a conductivity value of 0.58 mm/h being the least value. This is mainly caused by the pore size distribution within the soil column. The size of the pore which ranges between sub-microscopic pores to bio- macroscopic pores serve as one of the principal determinants of the movement fluxes within a porous media (Dingman 2003).

In summary, the mechanical composition of soil is one of the most important factors that influences the movement and contamination of soils and groundwater by contaminants and pollutants.

Table 8: Vertical Analysis of Soil at Oti Landfill Site

PIT	Depth(cm)	Pb	Φ (%)	θ	S %	K(mm/h)
1,2,3	00-50	1.0	0.65	0.24	30.42	0.22
1,2,3	50-100	1.13	0.64	0.23	30.64	0.34
1,2,3	100-150	1.08	0.61	0.61	52.18	0.43
	SED	0.07	0.04	0.39	28.00	0.23
	F-pro	0.24	0.45	0.57	0.69	0.69

Pb= bulk density, ϕ = porosity, θ = volumetric water content, S = degree of saturation, *K* = hydraulic conductivity SED = standard deviation, F-pro= Fisher's probability

Source: Fieldwork, 2019

Exchangeable Cation Concentration in Soil Samples

The highest exchangeable calcium was 8.73 cmol/kg occurs at Pit 3 (100 -150 cm) followed by 8.17 cmol/kg (Pit 3, 50- 100 cm) and with the least exchangeable calcium concentration being 5.06 cmol/kg at Pit 1 (00-50 cm). It can be seen that all the exchangeable calcium concentration from the three Pits increased with depth. Despite the increase with depth, the results from Pit 1 showed no significant difference (F-pro=0.98) with depth while Pit 2 and 2 both showed a significant difference (F-pro= 0.013) between depth 00-50 cm and 100-150 cm.

Table 9: Analysis of Cations in Soils at Oti Landfill Site (cmols/kg)

PIT	Depth(cm)	Exchangeable cations				Exchangeable Acidity	
		Ca	Mg	Na	K	Al	H
1	00-50	5.06	2.33	0.84	0.64	2.12	1.92
	50-100	5.18	2.50	1.01	0.81	2.29	2.09
	100-150	5.26	2.42	0.93	0.73	2.20	2.01
	SED	0.87	0.91	0.3	0.33	0.24	0.37
	F-pro	0.98	0.98	0.86	0.88	0.79	0.9
2	00-50	6.37	5.58	0.90	0.74	1.82	2.74
	50-100	6.93	1.47	0.29	0.33	1.54	1.87
	100-150	7.267	0.81	0.43	0.27	1.85	2.68
	SED	0.26	0.07	0.08	0.08	0.09	0.08
	F-pro	0.013	<.001	<.001	0.002	0.016	<.001
3	00-50	7.83	3.03	0.42	0.29	2.07	3.24
	50-100	8.17	4.14	0.13	0.20	1.940	1.78
	100-150	8.73	3.00	0.56	0.48	2.200	2.00
	SED	0.21	0.14	0.14	0.14	0.14	0.14
	F-pro	0.013	<.001	0.052	0.201	0.249	<.001

Ca= calcium, Mg = magnesium, Na= sodium, K= potassium, Al= aluminium, H= hydrogen, SED = standard deviation, F-pro= Fisher's probability

Source: Fieldwork, 2019

However, the statistical result from the combined Pits showed no significant difference (F-pro = 0.65) between these concentrations at the various depths. Even though the calcium concentration from individual Pits increased with depth, the difference in change was not statistically significant. This shows that the leaching of this cation does not change with depth.

Table 10: Analysis of Cations in Soils at Oti Landfill Site (cmols/kg)

PIT	Depth (cm)	Exchangeable cations				Exchangeable Acidity	
		Ca	Mg	Na	K	Al	H
1,2,3	00-50	6.42	3.65	0.72a	0.56	2.00	2.63
1,2,3	50-100	6.76	2.73	0.48a	0.45	1.92	1.91
1,2,3	100-150	4.75	1.83	0.52a	0.43	1.49	1.36
	SED	2.23	1.26	0.32	0.24	0.61	0.62
	F-pro	0.65	0.41	0.74	0.86	0.68	0.20

SED = standard deviation, F-pro= Fisher's probability

Source: Fieldwork, 2019

The exchangeable magnesium concentration had the highest concentration of 5.58 cmol/kg (Pit 2, 00-50 cm) followed by 4.14 cmol/kg (Pit 3, 50- 100 cm) while the least concentration was 0.81 cmol/kg (Pit 2, 100-150 cm) followed by 1.47 cmol/kg (Pit 2, 50 – 100 cm). It can also be seen from Table 9 recorded exchangeable magnesium concentration at a decreasing rate with depth. The exchangeable magnesium also recorded no significant difference (F-pro=0.98) at Pit 1. From Pit 2, significant difference (<.001) was observed at all levels while at Pit 3, significant changes (<.001) occurred at depth 50-100 cm. The exchangeable magnesium for the combined Pits showed no significant difference (F-pro 0.41) among the Pits with respect to depth. The highest mean values of 3.33 cmol/kg obtained from the landfill soil was lesser than the least mean values of 17.79 cmol/kg recorded by Muhammed (2015).

The exchangeable sodium and potassium also showed no significant difference (F-pro= 0.86) and (F-pro=0.88) at Pit 1 while at Pit 2, a significant difference (F-pro<.001) and (F-pro 0.002) was observed for sodium and potassium respectively. These significant differences were seen between depth 00-50cm and 50 -100 cm for both exchangeable sodium and potassium. However, at Pit 3, there was not significant change (F-pro 0.201) in potassium at the various levels while a significant change (0.052) was observed between depth 50- 100 cm and 100- 150 cm. Among the three Pits, the highest exchangeable sodium was 1.01 cmol/kg and 0.81 cmol/kg for potassium respectively both occurring at Pit 1 (50-100 cm). Also, the least values were 0.13 cmol/kg Pit 3 (50- 100 cm) for sodium and 0.20 cmol/kg (Pit 3, 50-100 cm) for potassium. The concentration values did not show any particular order of either increasing or decreasing with depth. Also, the concentration at various depths for the combined Pits showed no significant difference (F-pro= 0.74 and 0.86) for sodium and potassium.

The general low levels of the basic cations (K^+, Ca^{2+}, Mg^{2+}, Na^+) at the study area can be attributed to both acidification and leaching losses. According to Bailey, Horsley and Long (2005) acidification usually leads to depletion of base cations because of the exchange between H^+ and the bases. Also, the equal concentration of the basic cation could have been influenced by the less compactness of the soil as recorded from the bulk density and porosity of the landfill soil (Table 7).

The exchangeable acidity (Al^{3+} and H^+) of the soil for Pits 1 did not show any statistical difference (F-pro = 0.79 and 0.90) for aluminium and hydrogen. At Pit 2, significant difference (F-pro 0.016 and <.001) for aluminium and hydrogen respectively were observed at 50- 100 cm. AT Pit 3, a significant change in hydrogen concentration was observed between depth 00-50 cm and 50 to 100 cm while no significant change occurred in the aluminium concentration at varying depths of descend. The highest exchangeable Aluminium (2.29 cmol/kg) was recorded from Pit 1 (00 -5- cm) and the lowest being 1.54 cmol/kg (Pit 2, 50 -100 cm). However, the highest exchangeable hydrogen (3.24 cmol/kg) was recorded from Pit 3 (00-50 cm) while the lowest value was 1.78 cmol/kg (Pit 1, 50 -100 cm). It can be seen that the mean values of the exchangeable acidity were greater that than the exchangeable sodium and potassium. This can explain the reason for the slight acidic nature of the soil. This suggests that the introduction of acidic cations into the landfill's soil by leachate was minimal.

Even though both the exchangeable bases and acidity were low compared to the results of Muhammed (2015), it was however greater than the results from the control soil: 2.02 cmol/kg, 0.98 cmol/kg, 0.22 cmol/kg, 0.07 cmol/kg, for calcium, magnesium, sodium and potassium respectively and 1.20 cmol/kg and 1.36 cmol/kg for aluminium and hydrogen when studied in abandoned waste pit in Obuasi mine. This shows that infiltration and redistribution of leachate from the landfill serve as the major source the increase of these cations' concentration at the landfill soil.

Other Chemical Properties of the Soil at Oti Landfill Site.

Table 11 presents the analysis of the soil pH, Cation Exchange Capacity, the Electrical Conductivity and the Base Saturation and the Effective Cation Exchange Capacity. The results show that the soil was slightly acidic ranging between 5.84 (Pit 1, 00-50 cm) to a near neutral of 6.86 (Pit 2, 100- 150 cm). The pH for Pit 1 showed an increasing value with depth (5.86, 6.42 and 6.59) while Pit 2 and 3 however did not show similar results. From Pit 2, the recorded pH values were 6.35, 6.14 and 6.68 while Pit 3 had pH values of 6.35, 6.80 and 6.69. All the three Pits showed significant difference (F pro=<.001, <.001 and 0.01) for Pit 1, 2, and 3 respectively. However, the results for the combined Pits showed no significant difference (F-pro= 0.27) among them.

The results from the various Pits shows that Pit 3 had the highest pH values followed by Pit 2 and 1. The pH values can be attributed to both the infiltration of leachate into the soil and high rainfall within the area. The high concentration hydrogen ions in the leachate (fig 1) and the high rainfall in the area tend to increase the hydrogen concentration within the soils. However, the pH values (Table 11) were greater than the pH value range of 4.64 – 5.60 recorded by Sarkodie (2017). Soil pH plays an important role in the movement of trace metals and metalloids in soils. Both high and low pH can render some heavy metals highly mobile (Aydinalp & Marinova, 2003). That is in highly acidic soils, the mobility of metallic elements is much higher than soils with neutral and alkaline reaction.

The electrical conductivity from the three Pits had a range between 100 µS/cm – 810 µS/cm at varying depths (Table 11). The highest electrical conductivity was 810 µS/cm (Pit 2, 100-150 cm) followed by 680 µS/cm (Pit1, 50- 100 cm) and 600 µS/cm (Pit 3, 50-100 cm). The least EC was 100 µS/cm (Pit 3, 00-150 cm) followed by 120µS/cm (00 -50 cm). The results showed that the electrical conductance of the soil varied with depth. Both Pit 1 and 3 did not record any particular order while Pit 2 recorded an increasing trend with depth. While all the individual Pits showed a significance difference (<.001), the combined results of the three Pits (Table 12) showed no significant difference (F-pro=0.22) among the Pits.

The concentration of the recorded electrical conductivity in the soil can be attributed to the continuous infiltration of leachate throughout the year and the associate double maxima rainfall characterised in the area. According to Adviento-Borbe, Doran, Drijber, and Dobermann (2006) the continuous inflow of leachate and rainwater tend to dissolve salt ions within soil minerals and increase the electrical conductivity of the soil. The variations in the EC values are predominantly caused by potential amount of leachate that it held at different parts within the soil which dissolve the ion concentration of that part of the soil.

The cation exchange capacity (Table 11) of the clay minerals and organic matter of the landfill soil was low with a range of 8.24 cmol/kg (Pit 1, 100- 150 cm) to 13.59 cmol/kg (Pit 2, 00-50 cm). The highest CEC was 13.59cmol/kg (Pit 2, 50-100 cm) followed by 12.77 cmol/kg (Pit 3, 100-150 cm) while the least CEC was 8.24 cmol/kg (Pit 3, 00-50 cm) followed by 8.77 cmol/kg (Pit 2, 50-100 cm). All the CEC for the three Pits showed a significant difference (F-pro = 0.04, <0.01 and 0.05) for Pit 1, 2 and 3 respectively. The significant differences for the three pits were observed between depths 50 – 100 cm. However, the statistical results (F-pro 0.64) showed no significant difference with depth for the combined three Pits. The range of the CEC (8.24 -13.59 cmol/kg) was within the range provided by Reganold and Harsh (1987) on a Table provided for various clay soils and their cation exchange capacity. According to Rganold and Harsh, clay with CEC range of 3 cmol/kg – 16 cmol/kg are considered as kaolite clay which has a substantial amount of iron.

The highest mean value (13.59 cmol/kg) occurring at Pit 2 (00- 50 cm) shows that within depth 00- 50 cm, there was a high amount of Ca^{2+}, Mg^{2+}, K^{+} and Na^{+} being absorbed and exchanged by the soil anions. The absorption of these cations by the soil's negatively charged ions help reduce the number of contaminants contained in the leachate that can percolate through the soil into the groundwater system. That is during movement of contaminants from the surface of the earth to the groundwater system, the presence of clay minerals and organic matter, as stated by Rengasamy and Churchman (1999), preferentially absorbs high amount of Ca^{2+} and Mg^{2+} to flocculate around the colloidal surface of the clay minerals there by reducing the amount of the Ca^{2+} and Mg^{2+} migrating into the groundwater system. This can be the reason for the low cation values that were present in the groundwater samples as shown in fig 1-12. According to Cornell University Cooperative Extension (2007), low values of CEC correspond with low value of exchangeable bases since there is a positive relationship between the two. This phenomenon can also explain the reason for the low exchangeable base.

The base saturation values as shown in Table 11 was averagely high with a range value of 61.30 % (Pit 3, 00-50 cm) – 77.46 % (Pit 2, 100-150 cm). From Pit 1 the BS was 65.91 %, 73.11 % and 70.63 % for the varying depths respectively. Pit 2 had a BS of 72.46 %, 69.79 % and 77.46 % for depth various depths of descend while Pit 3 also had a BS of 61.30 %, 71.11 % and 72.37 % for depth 00 -50 cm, 50 -100 cm and 100 -150 cm respectively. There was a significant difference in the ECEC for Pit 1

These recorded values from the various Pits showed a significant difference (F-pro=<.001) for all the three Pits. This indicates that the base saturation for of the individual Pits varied significantly at varying depth. Both Pit 1 and 2 did not show any particular order of increase or decrease with depth.

The statistical results for the combined Pits (F- pro = 0.17) showed that there was no significance difference between the base saturation at the Pits. From the BS values also indicates that there was much absorption of Al^{3+} and H^+ (exchangeable acidity). The absorption of these cations by the clay minerals also helps in reducing number of contaminants being transported from the landfill into the local groundwater system.

The effective cation exchange capacity recorded at the landfill, like the CEC was moderately low with the highest value of 18.15 cmol/kg (Pit 2, 00-50 cm) and lowest being 11.97 cmol/kg (Pit 1. 100-150 cm).

Table 11: Analysis of Other Chemical Properties of the Soil at Oti Landfill Site.

PIT	Depth	pH	EC (µS/cm)	CEC	BS	ECEC
1	00-50	5.86	120	9.67	65.91	14.69
	50-100	6.42	680	9.80	73.11	13.67
	100-150	6.590	150	8.243	70.63	11.97
	SED	0.01	0.56	0.49	0.56	0.73

	F-pro	<.001	<.001	0.036	<.001	0.026
	00-50	6.361	160	13.59	72.46	18.15
2	50-100	6.150	190	9.01	69.79	12.42
	100-150	6.868	810	8.77	77.46	13.29
	SED	0.01	0.472	0.374	0.01	0.52
	F-pro	<.001	<.001	<.001	<.001	<.001
	00-50	6.59	100	11.58	61.30	16.90
3	50-100	6.83	600	12.64	71.11	16.36
	100-150	6.70	376	12.77	72.37	16.97
	SED	0.04	0.71	0.4	0.71	0.66
	F-pro	0.01	<.001	0.05	<.001	0.62

Depth (cm), EC (µS/cm), CEC (cmol/kg), BS (%), ECEC (cmol/kg).

Source: Fieldwork, 2019

Pit 1 had a 14.69 cmol/kg, 13.77 cmol/kg and 11.97 cmol/kg for the respective depths of descend. Pit 2 also had ECEC values of 18.15 cmol/kg, 12.42 cmol/kg and 13.29 cmol/kg for the various depths of descend respectively while Pit 3 had ECEC of 16.90 cmol/kg, 16.32 cmol/kg and 16.97 cmol/kg for the various depths of descend. There was a significant difference (F-pro=0.026) for Pit 1 and Pit 2 (F-pro= <.001).

The statistical results for the combined Pits (F-pro=0.34) show no significant difference in the ECEC of the depths of the various Pits. These statistical indifferences can be attributed to the deferential absorption of various cations at different phases within the soil column. Averagely, like CEC, the first 50 cm of the soil at the landfill had the highest ECEC value of 18.15 cmol/kg. This is due to the presence of high organic matter and clay minerals with is able to absorb large concentration of cations found within the soil. The low recorded ECEC values at the study area could easily be attributed to the combine high effect of the basic cations (Ca^{2+} and Mg^{2+}) and also the high values of the exchangeable acidity (Al^{3+} and H^{+}) of the study area. The absorption of these cations by the soil helps prevent advection of contaminant into the groundwater system.

Lastly, when compare the results obtained from the landfill with the results of the control soil sample, it can be concluded that leachate from the landfill has altered the soil chemistry. The highest pH results from the landfill was 6.68 was less compared to the pH of 7.11 recorded from the control soil. Holding rainfall constant, it can be said that the difference in pH can be attributed to the additional introduction of leachates into the soil which is rich in inorganic compounds. The CEC = 3.29 cmol/kg and ECEC = 5.85 cmol/kg were all lesser than the least values of 7.65 cmol/kg and 12.07 cmol/kg for CEC an ECEC respectively recorded from the landfill. However, the base saturation for the control soil (88.87 %) was higher than the highest recorded for the landfill soil (77.45 %). This factor can be caused by low Al^{3+} and H^{+} recorded for the control soil sample. This is because soils with less exchangeable acidity than exchangeable base then to have high base saturation since CEC are more replaced in to soil than the acidic cations.

Table 12: Results of Other Chemical Properties of the Soil at Oti Landfill Site

PIT	Depth(cm)	pH	EC (μS/cm)	CEC	BS	ECEC
1,2,3	00-50	6.27	126.8	11.61	66.56	16.58
1,2,3	50-100	6.47	490.4	10.48	71.34	14.15
1,2,3	100-150	6.72	445.7	9.93	73.49	14.08
	SED	0.25	201.50	1.74	3.23	1.75
	F-pro	0.27	0.22	0.64	0.17	0.34

Source: Fieldwork, 2019

Analysis of Heavy Metal at the Oti Landfill Site

Table 13 shows the heavy metal concentrations that were found in the soil at the Oti landfill site. Generally, the iron concentration found within the landfill soils was very high with the highest value of 18,347 mg/kg (Pit 1, 00-50 cm and 100- 150 cm) while the least concentration was 10,201 mg/kg (Pit 2, 00 – 150 cm). Pit 1 had the highest iron concentration of 18,347 mg/kg, 17,979 mg/kg and 18,347 mg/kg at varying depth of descend. Pit 2 also had a concentration value of 10,201 mg/kg, 10,620 mg/kg and 11,530 mg/kg for depth 00- 50cm, 50- 100cm and 100-150 cm respectively while Pit three also had iron concentration of 14917 mg/kg, 16,133 mg/kg and 11,474 mg/kg for depth 00- 50 cm, 50-100 cm and 100 -150 cm respectively. Iron concentration for Pit 1 and 2 showed a significant difference (F-pro= <.001) for both Pits while no significant difference (F-pro=0.616) in iron concentration was observed at Pit 3. Again, the significant difference (F- pro = 0.21) from the combined pits showed that the iron concentration for the entire landfill did not change with depth.

The high concentration of iron present in the soil samples from the Pits at the landfill compared to the concentration detected in the leachate indicates that the landfill may be contributing very little amount of iron content into the soil. Also, the concentration found in the soil and the groundwater samples suggest that the iron concentration is in a residual form. This agrees with the statement made by Ogundiran and Osibanjo (2009), that fractionalisation of heavy metals in soils exist in six main different forms ranging from water soluble form, exchangeable to carbonate bound form to Fe-Mn oxide bound form to organic matter bound and lastly residual form. The extent of contamination of soil according to Ogundiran and Osibanjo, is dependent on the form in which the metal ions exist. Besides, the concentration of iron which also increased with depth in the soil at the landfill had a great influence on the amount of trace metal that will be available for speciation. This is because within soil, iron exist at an exchangeable cation that absorbs heavy metals present in the soil there by reducing their concentration. This possibly explains the reasons for the low contamination of heavy metals that were present in the groundwater sample.

The manganese also had its highest concentration of 89.50 mg/kg (Pit 2, 00- 50 cm) and the lowest concentration 50.89 mg/kg (Pit 1, 50 – 100 cm). The manganese concentration also did not show any particular order of either increasing or decreasing with depth. Here, manganese concentration found in Pit 1 and 2 showed much significant difference (F-pro=<.001) for both Pits while Pit 3 had no significant concentration difference at various depths of descend. This suggests that manganese concentration within the soil varies.

The concentration of manganese in the soil at the landfill can be said to have been influenced by the soil pH, soil organic content and clay fraction. According to Schulte and Kelling (1999), there is an inverse relationship between soil pH and manganese concentration in soil. To Schulte and Kelling, manganese concentration increases when soil pH decreases and vice versa. Therefore, slightly acidic nature of the soil possibly explains the high values (Table 4.4a) of manganese recorded from the sample. However, since manganese is an essential heavy metal, its presence in soil in a form as exchangeable manganese or Ferric-Magnesian silicate mineral helps in the absorption and restriction of movement of non- essential heavy metals into groundwater. This also helps in protecting the local groundwater system at the landfill from contamination and pollution.

The copper concentration (Table 4.4) showed significant difference (F-pro = <.001) at varying depths for Pit 1 and 2 while Pit 3 showed no significant difference (F-pro= 0.132) at all levels. From Table 13 the first Pit had a copper concentration of 45.36 mg/kg, 20.30 mg/kg and 23.70 mg/kg at various respective depths of descend. Pit 2 also had copper concentration of 51.59 mg/kg, 58.71 mg/kg, and 36.31 mg/kg while Pit 3 also recorded copper concentration of 33.22 mg/kg, 45.36 mg/kg and 34.24 mg/kg for depth 00- 50 cm, 50- 100 cm and 100- 150 cm respectively.

Table 13: Analysis of Heavy Metals in Soils at Oti Landfill Site (mg/kg)

Pit	Depth	As	Cd	Cu	Fe	Mn	Pb
	00-50	0.86	2.42	45.36	18347	79.66	0.12
1	50-100	0.60	1.74	20.30	17979	50.89	0.12
	100-150	0.68	2.23	23.70	18347	71.39	0.13
	SED	0.01	0.02	0.01	7.22	0.01	0.01
	F-pro	<.001	<.001	<.001	<.001	<.001	0.23
	00-50	0.67	1.60	51.59	10201	89.50	0.11
2	50-100	0.72	2.75	58.71	10620	61.00	0.10
	100-150	0.93	1.32	36.31	11530	87.01	0.16
	SED	0.02	0.02	0.02	0.02	0.02	0.02
	F-pro	<.001	<.001	<.001	<.001	<.001	0.085
	00-50	0.46	2.233	33.22	14917	66.05	0.10
3	50-100	0.68	2.277	45.36	16133	80.72	0.15
	100-150	0.68	1.730	23.24	11494	50.93	0.32

SED	0.02	0.43	9.21	4691.9	20.51	0.15
F-pro	<.001	0.421	0.132	0.616	0.405	0.38

SED = standard deviation, F-pro= Fisher's probability

The high concentration of copper within the soil (Table 13) compared to the concentration detected in the leachate and the groundwater samples (fig 8) shows that the concentration of copper in the soil was more than the concentration that was present in the groundwater and the leachate. Like the iron, the low concentration of copper in the leachate but high concentration in the soil suggest a successive accumulation of copper ions in the soil by the leachate or the concentration occurs naturally as stated by Masindi and Muedi (2018) that natural weathering of rock minerals can cause the release of trace metals from their endemic spheres into their environmental compartments or different environmental compartment. However, the low concentration copper found in the groundwater compared to the soil concentration can be explained by the statement made by Fijalkowski and Kacprzak (2012) that cations of both metallic and metalloid elements undergo a strong specific adsorption, forming partially covalent bonds with ligands, mainly on the surface of iron ox hydroxide, Fe^{2+} Al^{3+} and Mn. This helps in reducing the concentration of copper as it migrates in the soil and this helps to protect the local groundwater from contamination and pollution.

Table 14: Analysis of Heavy Metals from Combined Pits at Oti Landfill Site (mg/kg)

PIT	Depth(cm)	As	Cd	Cu	Fe	Mn	Pb
1,2,3	00-50	0.75	2.08	43.39	14488	78.40	0.11
1,2,3	50-100	0.67	2.26	41.46	14911	64.20	0.12
1,2,3	100-150	0.76	1.759	27.75	13790	69.78	0.20
	SED	0.14	0.38	10.78	3230.40	12.44	0.05
	F-pro	0.77	0.46	0.35	0.94	0.55	0.21

SED = standard deviation, F-pro= Fisher's probability

The cadmium concentration in the soil recorded had its highest value from Pit 2 (50-100 cm) with a concentration of 2.75 mg/kg and the least cadmium concentration being 1.32 mg/kg (Pit 2, 50-100 cm). Here, Pit 1 and 2 again showed significant difference (F- pro= <.001) at all levels while Pit 3 did not show any significant difference in cadmium concentration at various depths. Also, the combined results from the three Pits also did not show any significant difference (F-pro=0.46) for the entire study area.

The Cd concentration levels found within the various depths can be attributed to the acidic nature of the soil pH. According to Aydinalp and Marinova (2003), the mobility of cadmium within soil is mainly influenced by the pH of the soil. Cadmium is highly mobile at high soil pH (4.2- 6.6) and less mobile within soil when the soil is alkaline (6.7- 7.8). Also, it can be stated that the presence of clay minerals in soil as well as high organic content absorbs heavy metals like Cd. The absorption of the trace metals by the soil physicochemical properties helps shield groundwater resources from contamination and pollution.

The arsenic concentration on the other hand had the highest value of 0.93 mg/kg from Pit 2 (100-150 cm) and the least recorded value of 0.45 mg/kg was also recorded at Pit 3(00-100 cm). The mean results however showed a fall from the first 00-100 cm (0.82 mg/kg) to 50- 100 cm (0.59 mg/kg) and a rise at depth 100 – 150 cm. The statistical results (F-pro= 0.10) showed a significant difference between the depth 00-50 cm and depth 100-150 cm but no significant difference between 00-50 cm, 100-150 cm and depth 50- 100 cm. The arsenic concentration in the soil was however not less than the allowable threshold of 5 mg/kg as recommended by ISO (1995) stated in Toth, Hermann, Silva and Montanarell (2016). Besides, the values obtained was comparatively less than the 1.3 mg/kg – 2.3 mg/kg recorded by Walsh, Sumnert and Keeney (2007).

Comparing the arsenic concentration of the leachate and the soil, it could be seen that the concentration found in the leachate (4.0 mg/L) was higher than the highest value of 0.93 mg/kg (Pit 2, 00-50 cm and 100- 150 cm) found in the soil while no concentration was detected in the groundwater samples. The results from the three individual Pits showed significant difference (F-pro= <.001) while the combined results did not show any significant difference (0.77) for the entire study area. At Pit one, the significant difference in arsenic concentration occurred at all the levels of depths while at Pit 2 and 3, it occurred between two different levels. It can be seen from Pit 1 and 2 that there was significant decrease in arsenic concentration from depth 00-50 cm. This indicates that during the movement of arsenic within the soil, large amount of its concentration is lost through sorption. This sorption can be attributed to the presence of presence of high organic matter, high manganese and iron concentration that was present in the soil.

According to Toth, et al., (2016) and Walsh, et al., (2007) all soils with high sorption capacity for cations, i.e. land containing a large amount of clay minerals, have the ability to accumulate metallic elements. Binding of metal cations increases with increase of their valence, atomic weight and ionic potential. Also, Pigna, Caporale, Cavalca, Sommella, and Violante (2015) asserted that hydrous ferric oxides are very effective in sorption of arsenate. This suggest that the absorptive rate of ions such as arsenate by soils is a function of its clay content. Also, study conducted by Ursitti et al., (2004) as cited by Toth et al., (2016) stated there is not lateral movement of arsenic and its vertical movement is also very limited. This possibly explains no content detected in the groundwater sample.

The Lead (Pb) concentration in the soils sample was low as compared the other trace metals and did not show any significant difference (F-pro= 0.23, 0.09 and 0.38) for Pit 1, 2, and 3 respectively. The highest Pb concentration was 0.16 mg/kg (Pit 2, 100-150 cm) and the least concentration was 0.10 mg/kg (Pit 2, 50-100 cm and Pit 3, 00- 50 cm). Pit 1 had Pb concentration of 0.12 mg.kg, 0.12 mg/kg and 0.13 mg/kg for the various depth of descend while Pit 2 on the other hand had a recorded Pb of 0.11 mg/kg, 0.10 mg/kg and 0.16 mg/kg. At Pit 3, the Pb recorded values were 0.10 mg/kg, 0.15 mg/kg and 0.32 mg/kg for varying depths of descend. The mean values for the vertical movement showed an increasing value with depth with a significant difference (F-pro = 0.21) which shows that there was no significant difference between the concentration of Pb at various depths of descend.

Comparing the concentration of lead 0.16 mg/kg (highest concentration) to the mean concentration of 29,000 mg/kg (least value) recorded by Ogundiran and Osibanjo (2009) when researched into “mobility and speciation of heavy metals in soils impacted by hazardous waste, chemical speciation & bioavailability”, it shows that the lead concentration found in the soils of Oti Landfill site was very minimal. The results further suggest that the landfill receives very little or no amount of batteries and other lead containing waste probably due to recent recycling of automobile batteries. The low lead concentration detected in the soil can also be attributed to the presence of high concentration of hydrous oxides of Fe, Mn and Al and its special absorptive qualities there by reducing the quantity of lead present in the soil for leaching and pollution of groundwater. This then support the research finding of Fijalkowski, Malgorzata and Kacprzak (2012) that hydroxides of amorphous Fe, Mn and Al influences the binding of heavy metals by the mineral phase by forming coating on crystalline particles in the soil solid phase to absorb metal ions.

Finally, comparing the results obtained from the various Pits to the results from the control soil sample (Table 13), it can be said that the landfill which is the major potential of soil pollution in the area is the principal cause of the high values recorded from the landfill site vis a vis the results from the control soil sample.

Chemical Properties of Local Groundwater and Leachate at the Landfill Site

This section describes the results and discussion of the physicochemical properties of the leachate and the surrounding groundwater that was taken from the landfill site and the surrounding community. The results included the results of pH, EC, TDS, DO, BOD, COD, bicarbonate, calcium, sodium, potassium iron, copper, zinc and arsenic.

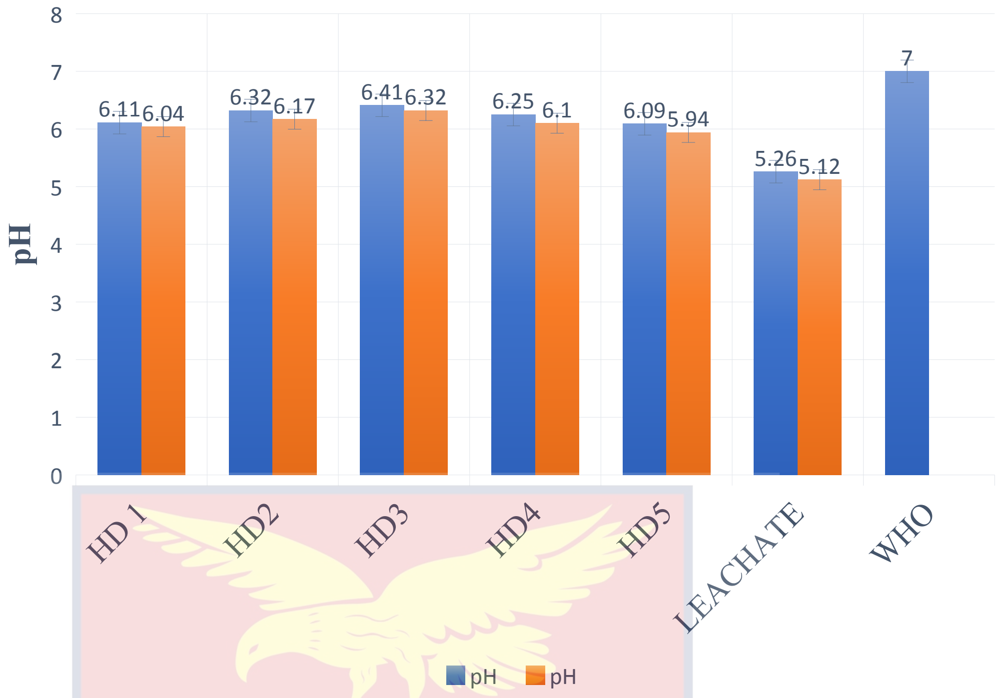

Figure 2: pH in the dry and wet seasons of the local groundwater, leachate and WHO standards.

Source: Fieldwork, 2019

Figure 1 shows the results of the pH of the local groundwater and the leachate from the study area for dry and wet seasons. The pH from the sampled wells for both two seasons were less than the neutral value of 7.0 (WHO standard for drinking water) making it acidic in nature. Also, the range of the pH (6.09 to 6.41) for the dry season and 5.94 to 6.32 for the wet season for the hand dug well fell below the WHO (2018) accepted range of 6.5 to 8.5 for drinking water. It could be seen from the results that the pH values were higher in the dry season than in the rain season. This was possible because the landfill is situated within the industrial hub of Kumasi where acidic rainfall is high and this can account for the reduction of pH for both the groundwater and the leachate during the rainy season. That is there is excessive addition of acid content water into the landfill and also into the local groundwater system.

This means that continues intake of groundwater from these hand-dug wells can cause acidosis due to the acidic nature of the water. This can lead to the development of lung and kidney diseases due to the possible increase of acid content in the blood fluids.

The low pH values obtained from the leachate were no different from the measured values obtained by Sulemana et al., (2015) when researched on the potential migration of leachate from active landfills (Oti landfill). With that study, the measured pHs were 6.09, 5.56 and 5.66 from Aprabon, Atonsu-Kuwait and Dompoase respectively even though the pH for the leachate was 7.4. Again, the pH value from the study was lesser than the recorded value of 8.5 that Danutsui recorded from the Oblogo landfill in Accra 2012. These low values of pH indicate a reduction in hydrogen concentration present in the water and can affect the soluble substances and toxicity of heavy metals like Iron, Manganese, Lead and Zinc and Copper in the water.

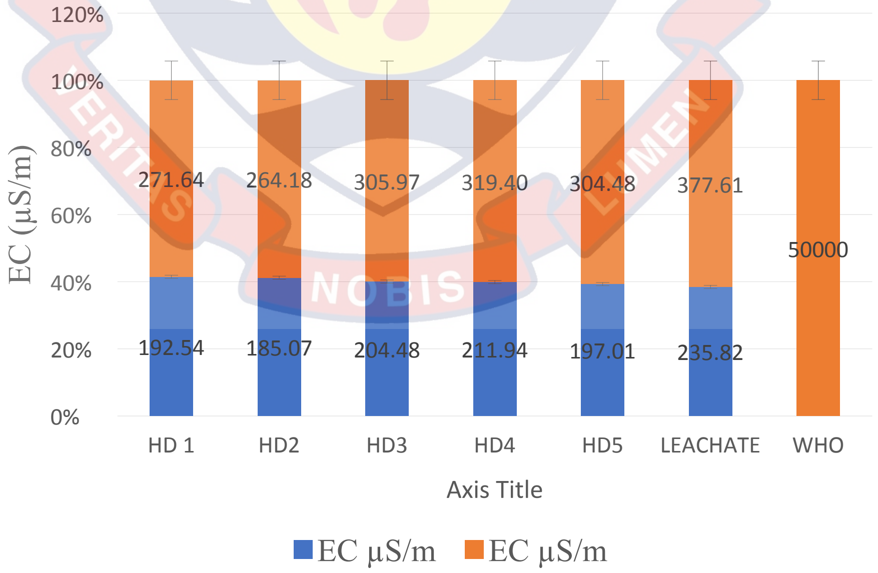

Figure 3: The EC in the dry and wet seasons of the local groundwater, leachate and WHO standards.

Source: Fieldwork, 2019

Figure 2 shows the seasonal variations in the electrical conductivity of the leachate and the local groundwater of the study area. The dry season had an EC range of 185.07 µS/m (HD 2) to 211.94 µS/m (HD 4) and the wet season was 264.18 µS/m (HD 2) to 319.40 µS/m. The higher the EC value, the higher the concentration of dissolved ions. All the measured values of the groundwater for the two seasons showed a low electrical conductivity as compared to WHO standard for drinking water (500 µS/m). The results show that the quantity of dissolved solids, ions and salt that were present in the leachate were more than that of the local groundwater. These values, however, were lesser than the EC value of 829.33 µS/m which was recorded by Sulemana et. al., (2012) on the topic “potential migration of leachate at the Dompoase Engineered Landfill Site”. Besides, the leachate had a recorded EC range value of 235.83 µS/cm to 377.61 µS/cm which indicates that the number of dissolved solids concentration that was in the leachate was greater than the concentration that was found in the local groundwater. This implies that the groundwater at the surrounding community is safe from dissolved ion from the landfill site.

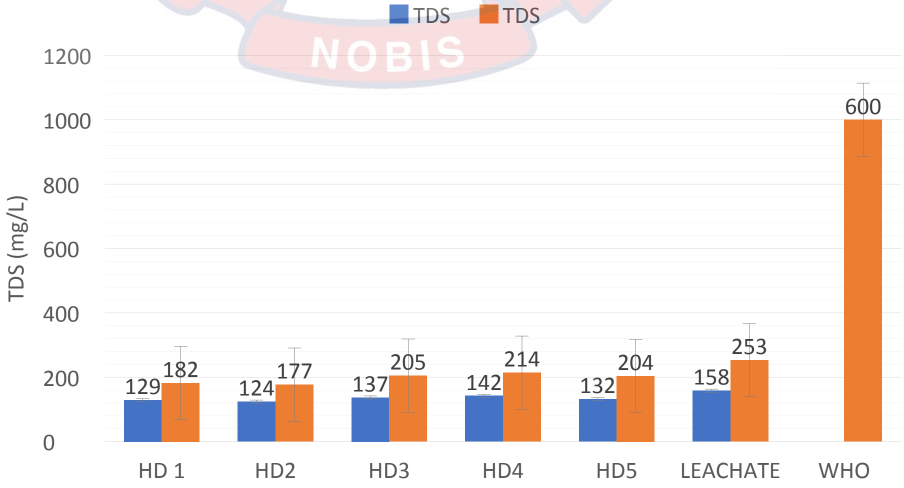

Figure 4: The TDS in the dry and wet seasons of the local groundwater, leachate and WHO standards.
Source: Fieldwork, 2019

The total dissolved solids for the two seasons as shown in figure 3 was low compared to the WHO standards for drinking water. The two seasons had a range of 124 mg/L to 158 mg/L and 177 mg/L to 253 mg/L for dry and wet seasons respectively. There was also an increase in the TDS of the leachate in the rainy season from 158 mg/L to 253 mg/L. This shows that there was an increase in inorganic matter concentration in the leachate during the wet season. The general low level in total dissolved solids in the samples as compared to WHO limit of 600 mg/L is an indication that there was very little amount of dissolved inorganic salt like Sodium, Calcium, Bicarbonate, and Magnesium in the samples even though it increased during the wet season.

From the WHO's (2018) water quality and guidelines, both surface and groundwater with a TDS value less than 600mg/L is considered to be potable for man usage. Also, the Safe Drinking Water Foundation (2018) also categorised water having a TDS value of less than 300mg/L to be excellent for drinking. These two categorisations make the groundwater safe for usage as far as TDS in concern. The leachate value was also low when compared to the one reported by El-Salam, and Abuzaid, (2014) from Borg El-Arab landfill site in Egypt which was 27,452mg/L. It was also less than 300mg/L and 1,500mg/L values as reported by Kanmani & Gandhimathi, (2013) when studied on heavy metals from dumping sites and groundwater contamination at Ariyamangalam, Tiruchirappalli District, Tamil Nadu in India.

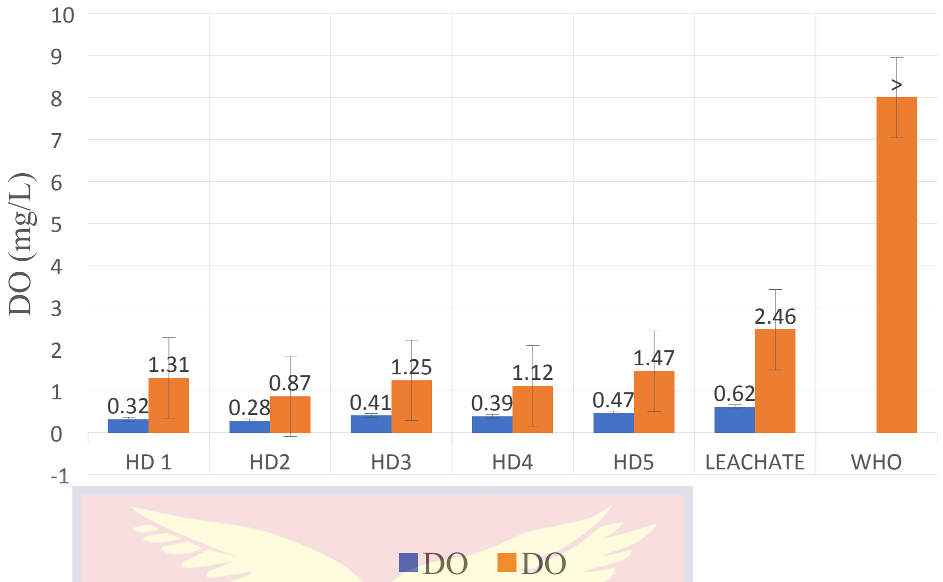

Figure 5: The DO in the dry and wet seasons of the local groundwater, leachate and WHO standards.

Source: Fieldwork, 2019

Figure 4 represents the results of the dissolved oxygen recorded for the two seasons which were lower than the recommended threshold of > 8mg/L. The dry season had a DO range of 0.28 mg/L to 0.47 mg/L while the wet season also recorded a rage of 0.28 mg/L to 0.47 mg/L. Even though there was an increase in DO during the wet season, the values recorded were generally lower than the WHO minimum threshold for quality drinking water.

It can be seen from these values that well 2 had the lowest DO in both seasons while well 5 recorded for the highest DO content. These low amount of DO values in the measured samples might have resulted from several reasons. Firstly, the warmness of groundwater as compared to surface water might have caused these reductions. That is when water becomes warm, there is an increase in molecular activities which pushes oxygen molecules from spaces that are being occupied by these oxygen molecules thereby reducing the level of dissolved oxygen content in the water. Also, the low measured DO values might have been caused by the presence of bacteria in the water which tends to increase the oxygen demand for biological activities in the water. Lastly, these low levels of DO might also be affected by the proximities of these wells to the nearby streams and wetlands where plants at these wetlands tend to demand more oxygen for their photosynthesis.

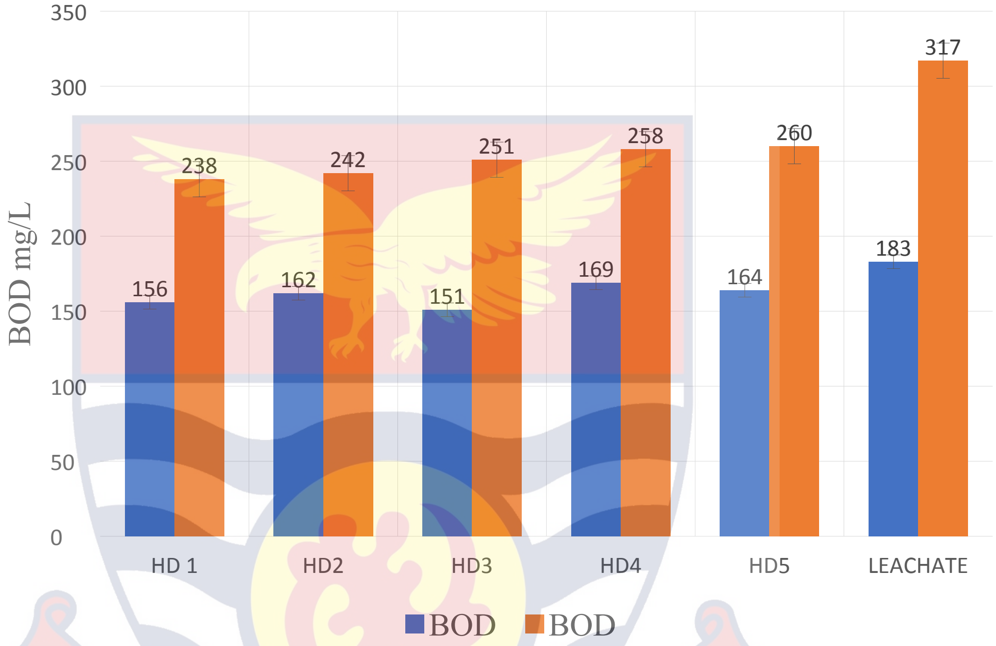

Figure 6: The BOD in the dry and wet seasons of the local groundwater, leachate and WHO standards.

Source: Fieldwork, 2019

The biochemical oxygen demand from both the groundwater and the leachate samples were high. From well 1, BOD level was recorded to be 158mg/L and 162mg/L was measured from well two. The third well had a recorded BOD value of 151mg/L while the fourth, fifth and leachate had a recorded BOD level of 169mg/L and 164mg/L and 183mg/L respectively. The high values of BOD in the wells is an indication of the presence of bacteria which its activities had reduced the total dissolved oxygen and had increased the demand for oxygen for biochemical activities.

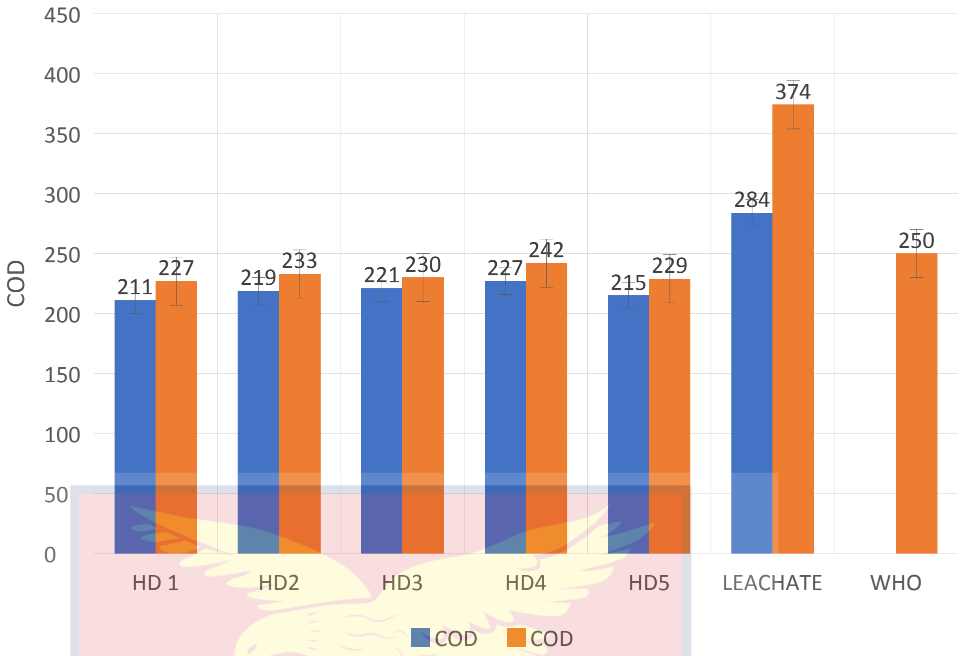

Figure 7: The COD of the selected samples and WHO standards.

Source: Fieldwork, 2019

The chemical oxygen demand values were moderately high even though it was within the WHO accepted limits for drinking water. The COD from the various wells during the dry season were 211mg/L, 219mg/L, 221mg/L, 227mg/L and 215 mg/L and 227 mg/L, 233 mg/L, 230 mg/L, 242 mg/L and 229 mg/L were recorded during the wet season for well 1,2,3,4 and 5 respectively. Also, 284 mg/L and 374 mg/L of COD concentration was found present in the leachate sample in the dry and wet seasons respectively. These measured values indicate the presence of an organic contaminant in the wells even though its values do not reach the WHO thresholds of 250mg/L as cited in Danistui (2012). These values were greater than the values obtained by Danitsui (2012) who recorded a COD values of 155mg/L and 108.67mg/L at Oblogo landfill site in Accra. Comparing the BOD and COD values obtained for this study, it could be seen that the value of the chemical oxygen demand is greater than the biochemical oxygen demand and this is in agreement of the statement made by Manivaskam, (2005) that COD values are always greater than BOD values due to chemicals required by both organic organisms and inorganic substances when undergoing oxidation. Also, the values for the leachate and groundwater implies that the landfill has no or little impact on the local groundwater.

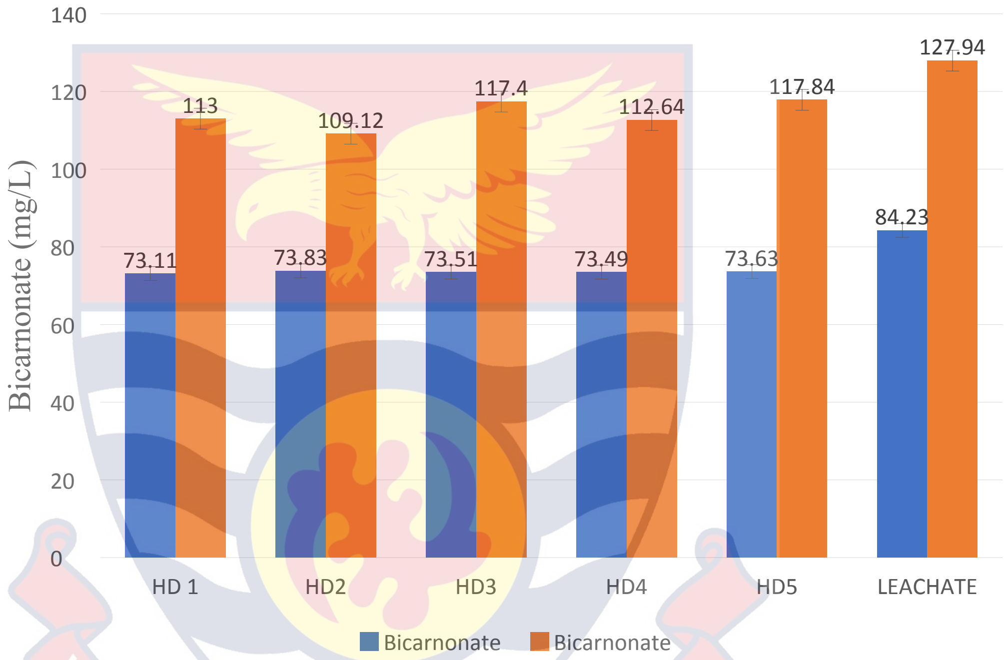

Figure 8: The bicarbonate in the dry and wet seasons of the local groundwater, leachate.

Source: Fieldwork, 2019

The figure 8 above describes the bicarbonate concentrations that were detected in both the groundwater sample and the leachate sample. The bicarbonate content was generally high in the wet season than in the dry season. In the dry season, the recorded concentration for the sampled groundwater was 73.11mg/L for well one; 73.83mg/L for well two and 73.51mg/L, 73.49mg/L and 73.63mg/L respectively and 84.23 mg/L for the leachate. During the wet season, the bicarbonate concentration for the sampled groundwater was 113 mg/L; 109.12 mg/L; 117. 40 mg/L; 112.64 mg/L and 117.84 mg/L and 127.94 mg/L for the leachate sample. The leachate result is an indication that the landfill receives a substantial amount of human excreta and domestic waste like food remains. The values obtained from the groundwater indicate that the ionic processes going on at both the landfill and the groundwater system is more anionic than being cationic. Also, the presence of these bicarbonate content can be attributed to leachate migration from the Oti landfill site since it serves as the major source for producing bicarbonate.

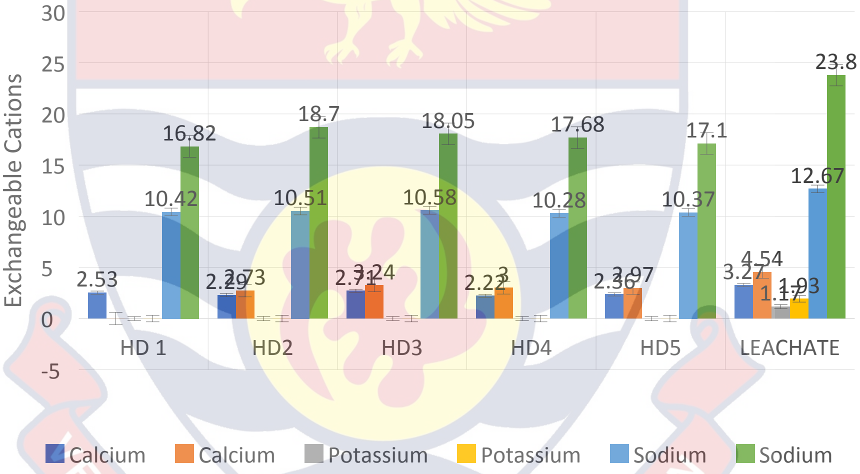

Figure 9: The cations in the dry and wet seasons of the local groundwater, leachate and WHO standards.

Source: Fieldwork, 2019

The figure 9 above shows the concentration values of various cations that were analysed from the groundwater and leachate samples. The analysis showed no potassium concentration in the groundwater samples for both wet and dry season. However, there was a concentration of 1.17 mg/L and 1.93 mg/L found in the leachate during the dry and wet season respectively. The sodium content that was found in the groundwater for the two seasons were generally low as compared to the WHO guideline value of 200mg/L. From fig 9, it can be seen that the sodium concentration recorded in the dry season was 10.42mg/L, 10.51mg/L, 10.58mg/L, 10.28mg/L and 10.37mg/L while 16.82 mg/L, 18.70 mg/L, 18.05 mg/L, 17.68 mg/L, 17.10 mg/L and 23.80 mg/L was recorded in the wet season for well 1 to 5 respectively. The leachate also had a recorded sodium concentration of 12.67mg/L and 23.80 mg/L for the dry and wet season respectively. It can be seen that the concentration of sodium that was present in the groundwater was lesser than the one found from the leachate samples in both seasons.

The calcium content detected from the groundwater samples were also generally low as compared to WHO (2018) standard limit of 200 mg/L for drinking water. From fig 9, the calcium concentration recorded was 2.53mg/L, 2.29mg/L, 2.71mg/L, 2.22mg/L and 2.36mg/L for the dry season and 3.04 mg/L, 2.73 mg/L, 3.24 mg/L, 3.00 mg/L and 2.97 mg/L were recorded in the wet season. The leachate had a calcium concentration of 3.27 mg/L and 4.54 mg/L for the dry and wet seasons. The results suggest that the landfill receives little or no amount of cement content waste from construction sites. The calcium values obtained from the groundwater samples indicate that there is very little positively charged ion solution concentration present in the water. Comparing the calcium and the pH values, it can be said that the groundwater within the surrounding community is soft water since its pH values and calcium content falls within the WHO (2018) drinking water quality range of 6.5 to 7.5 for pH and >60mg/L for calcium.

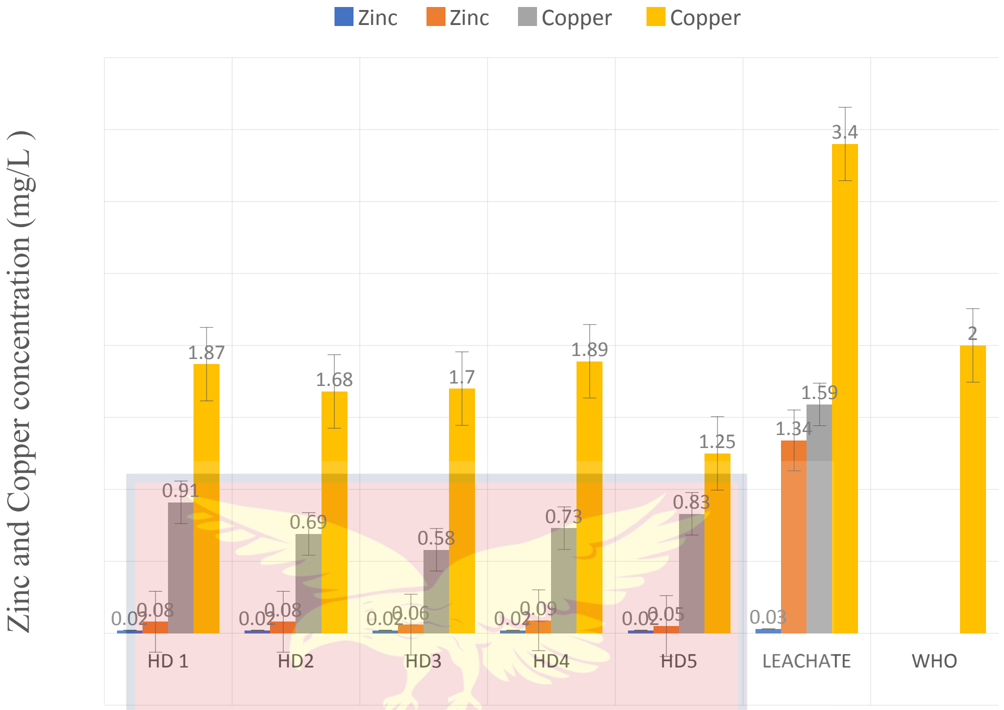

Figure 10: Heavy metals in the dry and wet seasons of the local groundwater, leachate and WHO standards.

Source: Fieldwork, 2019

Figure 10 shows zinc and copper concentration that were present in both the groundwater samples and the leachate during the dry and wet season. The zinc content that was detected from all the samples taken from the selected wells was 0.02 mg/L for the dry season and 0.08 mg/L, 0.08 mg/L, 0.06 mg/L, 0.09 mg/L and 0.05 mg/L for well 1 to 5 respectively while the leachate concentration was 0.03 mg/L and 1.37 mg/L for the dry and wet seasons respectively. These concentrations can possibly be caused by the landfill due to the concentrations levels that were found in both the leachate and the soil samples. The high value of zinc content found in the leachate sample indicates that the landfill receives waste of fluorescent bulbs or lamps and dry cells. The values obtained were lesser than the values that Denutsui

(2012) detected in similar groundwater and leachate analysis which had a detected average value of 3.23mg/L and 6.03 mg/L respectively for both groundwater and leachate at Oblogo landfill site in Accra.

All the copper concentration detected in the sampled wells in the two seasons were than the WHO guideline value of 2.0mg/L. The copper concentration detected in well one sample was 0.91 mg/L and 1.87 mg/L for the dry and wet season. At well two, the concentration reduced to 0.69 mg/L and 1.68 mg/L and a subsequent reduction at well three for the dry season and increase in the wet season with a concentration of 0.55mg/L and 1.70 mg/L. However, there was an increase in the concentration from 0.58mg/L to 0.73 mg/L for the dry season and 1.70 mg/L to 1.89 mg/L for the dry season at well four. The last well (well 5) had a recorded copper concentration of 0.83mg/L for the dry season and 1.25 mg/L for the wet season. However, the concentration from the leachate sample was 1.59mg/L and 3.40 mg/L for dry and wet season respectively. These values indicate that the landfill receives a substantial amount of electrical waste with copper materials, paints, metal pipes, pigment alloy and other waste materials made with copper. The value 3.4mg/L that was concentrated in leachate analysis that was conducted by Kanmani and Gandhimathi (2013) but was less than 13.78mg/L and 8.97mg/L that Denutsui (2012).

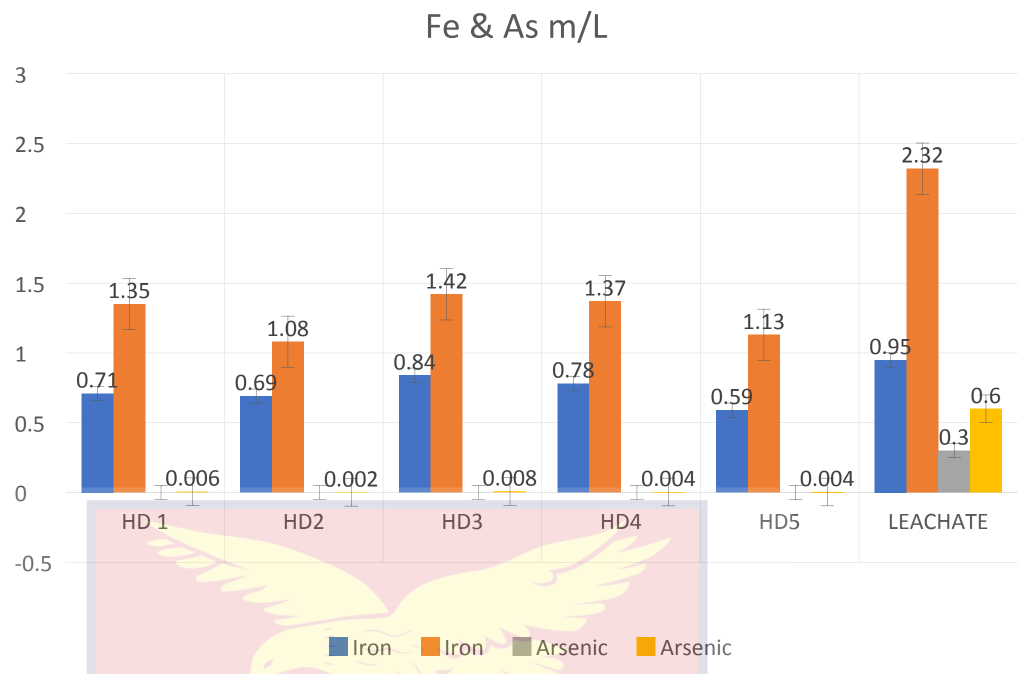

Figure 11: The heavy metals of the selected samples

Source: Fieldwork, 2019

The iron content that was detected from the groundwater samples for both the wet and dry seasons from the selected wells were found to be below the Provisional Maximum Tolerable Daily Intake (PMTDI) value of 10 percent which WHO (2018) converted to be 2 mg/L. Samples taken from well one had a detected iron value of 0.71 mg/L and 1.35mg/L. Well two had iron concentration of 0.69 mg/L dry season and 1.08 mg/L for wet season. Sample from well three had a detected iron content of 0.84 mg/L and 1.42 mg/L which was the highest among the ten groundwater samples that were taken during the study. Well four and five had a recorded iron concentration of 0.78mg/L, 1.37 mg/L and 0.59 mg/L, 1.13 mg/L for the dry and wet seasons respectively. All these values were less than 2 mg/L permissible value prescribed by WHO. The concentration value of iron detected in the leachate was 0.95 mg/L and 2.32 mg/L for wet and dry seasons respectively. This is an indication that the landfill receives iron-containing waste materials like animal remains, food remain, metallic parts among others. The concentration of iron content in the groundwater can be naturally occurring through the release of soluble furious iron or insoluble ferric iron into groundwater during weathering process. With reference to the iron content that was present in the soil, the leachate and the groundwater, it can be said that the high concentration of iron in the landfill soil was as a result of accumulation of ion from the landfill site over the past years. However, low iron content detected in the groundwater sample was can be due to the fact that much of the concentration is fractionallised in residue bound within the soil which will prohibits it from moving into the ground water system.

Arsenic content detected within the groundwater samples were generally low as compared to WHO standards 0.01 mg/L. From the sample, no arsenic concentration was detected during the dry season and the arsenic content detected during the wet season were below the WHO allowable limit of 0.01 mg/L. However, the arsenic concentration of well one was 0.006 mg/L while from well two, the detected arsenic content was 0.002 mg/L. At well three, the detected arsenic concentration was 0.008 mg/L while 0.004 mg/L and 0.004 mg/L were found samples collected from well four and five respectively. However, within the leachate sample, the detected arsenic concentration was 0.03 mg/L for the dry season and 0.6 mg/L in the wet season. This value indicates that the landfill is receiving glass waste, cosmetic products, waste from wood preservatives and possibly waste pesticides products and some sulphate waste minerals. Concerning the concentration found in the groundwater samples, infiltration and percolation of leachate that runs on the surface at the landfill site could be a possible cause for the presence of the concentration in the groundwater even though it cannot be entirely concluded that the landfill is the sole cause of these concentrations in the groundwater since arsenic is also largely present in arsenopyrite rock-forming minerals.

The DRASTIC Vulnerability Index

Table 15 shows the results of the DRASTIC vulnerability index for the study area. The total index obtained from the product of the weight and the rating of the various parameters indicates that the local groundwater was moderately vulnerable to pollution where the laboratory results of the groundwater validates it. According to qualitative classification of vulnerability of groundwater (Table 6) made by Navulur et al., (1996), the potential of the local groundwater present at the Oti landfill site to be vulnerable to contamination and pollution is within the range of low to moderate. The soil media, the impact of the vadose zone, and the aquifer characteristics which serve as a conduct for the movement of fluxes and pollutants within the local hydrogeology is the main parameters that shield the groundwater form contamination and pollution. However, the topography of the landfill is the parameter that had the least potential to contribute to groundwater vulnerability while the depth to groundwater was the most potential parameter render the surrounding groundwater vulnerable to pollution and contamination.

From the sensitivity analysis (Table 16). The contributing potential of each parameter to the vulnerability of the groundwater resources are in the order of: topography < hydraulic conductivity, soil media < aquifer characteristics < impact of the vadose zone < net recharge < depth to groundwater in an increasing order for both Table 15 and 16

Within advective movement, the transporting fluids and pollutants from the landfill to the groundwater system is governed by the transport time which is proportional to the distance travelled. Considering the results from both the laboratory and the DI, it indicates that there are two main possible factors responsible for the preservation of the water quality. These are the conductance of the local geology which controls the discharge rate of the local aquifer and the residence time of the leachate which is governed by the topography of the landfill. This reaffirms the results of the laboratory's hydraulic conductivity results since soil pollution is influenced by the amount of clay content that is present in the soil. The higher the clay content, the higher the absorption potential of the soil. Secondly, the residence time of solute to in a groundwater system is mainly influenced by the discharge rate of the aquifer. The higher the discharge rate the lower the residence time of a contaminant or pollutant in the aquifer system. The aquifer characteristics of the study area had a rating of 4 and a DI of 12 which shows a moderately high discharge. This also accounts for the low values of concentration of pollutants and contaminants found in the sampled water.

Table 15: Analysis of DRASTIC Index

DRASTIC model	Site characteristics	Weight	Rating	DI
Depth to groundwater	least depth (5-15 ft)	5	9	45
Net-recharge	7-10 inches	4	8	32
Aquifer Characteristics	weathered metamorphic	3	4	12
Soil media	clay loam	2	3	6
Topography	slope = 18+	1	1	1
Vadose zone	silt clay	5	4	20
HC	Range (100-300 gpd/ft^2)	3	2	6
Total				122

HC = Hydraulic Conductivity, gpd = gallons per day, DI = DRASTIC vulnerability index.

Source: Fieldwork, 2019.

Table 15: Sensitivity Analysis of the DRASTIC Vulnerability Index.

DRASTIC model	Site characteristics	Pr & Pw		DI
Depth to groundwater	least depth (5-15 ft)	45	122	36.89
Net-recharge	7-10 inches	32	122	26.23
Aquifer Characteristic	weathered metamorphic	12	122	9.84
Soil media	clay loam	6	122	4.92
Topography	slope = 18+	1	122	0.82
Vadose zone	silt clay	20	122	16.39
HC	Range (100-300 gpd/ft^2)	6	122	4.92

Total DI	122	100.00

HC = Hydraulic Conductivity, gpd = gallons per day, DI = DRASTIC vulnerability index

Source: Fieldwork, 2019.

CHAPTER FIVE

SUMMARY, CONCLUSIONS, AND RECOMMENDATIONS

Summary

The study drove towards assessing the susceptibility of geohydrological characteristics of Oti landfill site to groundwater pollution by investigating into the hydrophysical and geochemical characteristics of the landfill site in Kumasi. The research questions were the analyse of hydrophysical properties of the soil and its susceptibility to groundwater pollution, the geochemical characteristics of the soil and its effects on groundwater pollution and lastly, the chemical characteristics of leachate and the surrounding groundwater and how the landfill has affected it using the DRASTIC model. Within the methodology, the principles and methods of experimental research design were used while probability sampling which adheres to the principles of the theory of statistical regularity and the central limit theorem were used to take samples from the various units of analysis for the study. Here, the quartering method, the systematic and the composite sampling methods were used.

Both the collection and analyses of the samples were done in accordance with the WHO, ISO and ASTM quality assurance procedures. During the analyses, the hydrophysical properties of the soils were limited to the bulk density, the porosity, the gravimetric and volumetric water content and the degree of saturation of the soil as well as the hydraulic conductivity of the soil. The physicochemical characteristics of the soil, the groundwater and leachate samples were also analysed. Some of the selected parameters that were analysed included the exchangeable cations of the soil, some selected heavy metals for both the leachate, groundwater, and soils.

The results from the hydrophysical properties of the landfill soil showed some individual variations within the various pits which was anonymous to the normal assumed natural order for undisturbed soil. The general porosity values were found to be high while the bulk density, the volumetric, gravimetric and degree of saturation were found to be low. The hydraulic conductivity for the area was observed to be low. Even though the average values showed assumed natural order with depth, the statistical analysis indicated that there were no significant differences in these variations. With the exception of the hydraulic conductivity which was statistically significant.

The physicochemical analysis of the soils showed that the soil had a pH value within the acidic range. This low pH was is found to be influencing the Cation Exchangeable Capacity of the soil thereby increasing the concentration of calcium and magnesium within the soil. The physicochemical analysis for the soil also showed that there were high concentration and dispersion of heavy metals iron, copper and manganese at the landfill site by pH of the soil.

The leachate and groundwater quality results were found to be within the WHO water quality guidelines limits which means that there is no contamination of the groundwater by the landfill. The physicochemical values measured and obtained from the leachate sample were generally higher than the value that was obtained for the groundwater. This suggests that there is continues infiltration of leachate from the landfill site making the groundwater safe for human use. The parameter that was found to have a high concentration in the groundwater was bicarbonate which had a minimum concentration of 73.11 mg/L but the concentration has no threshold limit in drinking water per the WHO guideline.

Conclusion.

In conclusion, the present groundwater condition for the communities surrounding the Oti landfill site was found neither polluted nor contaminated by the landfill which is the major potential source of groundwater contamination. The results from the DRASTIC model showed that a low to moderate vulnerability. Comparing the concentrations for the various parameters found in the leachate and the groundwater, it just that the landfill was sited at a location where hydraulic conductivity is low (the hydraulic conductivity of from Table 7 proves it) and the facility has been managed well over the past years.

However, the findings of the research also reveal that there are possibilities for the groundwater for the area to be contaminated by major possible trace metals like Manganese, Lead, Copper, Arsenic, Iron and another anionic compound like bicarbonate. Their concentration values were found to be high both in a horizontal and vertical direction. In addition is the potential of hydrogen ions to dissolve heavy meatal and leach them to into the groundwater system. This is because of the low pH that was found in both the leachate, soil, and the groundwater.

Recommendation

Base on the findings of the research, the study seeks to make the following recommendations

1. There should be a continues proper management of the landfill to help safe the surrounding groundwater from pollution.
2. Lastly, the study recommends sorting of waste to help reduced materials that contain high concentration of heavy metals since it was found to be high in the soils sampled compared to the result of the control soil sample.

Suggested future research

The research suggests the following for future research.

1. Future research should look into methods and management approach will be needed to prevent the movement of heavy metals in the soil that has the potentials of polluting the local groundwater.
2. Again, future research should be carried on this same topic by using different techniques like electrical resistance block to find the inverse relations of the moving leachate and the electrical resistance of the soil porous volume. Also using Capacitance and time-domain reflectometry to assess the movement of the leachate.

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APPENDICES

Pressure pipes used for the collection of the hydraulic conductivity soil sample

Taking soil samples at varying depths

Field assistant helping in the collection of soil samples

Collection of soil samples using core bulk density sampler

Air drying of soil chemical samples

Preparation of soil chemical samples

Sieving of soil chemical samples

Weighing of wet bulk density sample