CHAPTER ONE

INTRODUCTION

1.1 Background to the Study

Open dump system of waste disposal is the predominant system of solid waste disposal in Lagos, and Nigeria in general. This system of waste disposal, just like the landfilling system of waste management is very common in low income and developing countries of the world due to the associated low costs. According to Longe and Balogun (2010), almost 100 percent of solid wastes generated in low and medium income developing countries of the world are disposed at landfills, due mainly to the availability of land and the relatively low costs of operations.

There are however, a number of environmental and public health implications associated with landfills and dumpsites (Moreno, 2011). This is because "incorrectly secured and improperly operated municipal solid waste (MSW) landfills pose a serous threat to the environment, mainly surface and ground waters" (Slomczynska & Slomczynska, 2004). Joseph (2002) described the characteristics of open dumps to include indiscriminate disposal of waste, absence of liners and leachate treatment and treatment facilities, inadequate compaction, poor site design, presence of scavengers and limited operational control measures.

Around the various dumpsites located in Lagos, groundwater constitutes a major source of water supply due to the limited, and in some instances, total absence of Lagos State Water Corporation (LSWC) network coverage in these areas. In light of the increasing reliance on groundwater and the fact that little or no treatment is applied to the water before consumption in most cases, the current waste disposal operations in Lagos poses a threat to the quality of groundwater proximate to dumpsites. This is because there are no real sanitary landfills in existence in Lagos and the

current waste disposal sites can at best be categorised as controlled dumps. According to Johannessen and Boyer (1999, p.4) "controlled dumps operate with some form of inspection and recording of incoming wastes, practice extensive compaction of waste, and control the tipping front and application of soil cover". In contrast to controlled dump, a sanitary landfill is properly sited and is outfitted with liner and leachate collection systems (Figure 1.1), which prevents the infiltration of leachate into the groundwater.

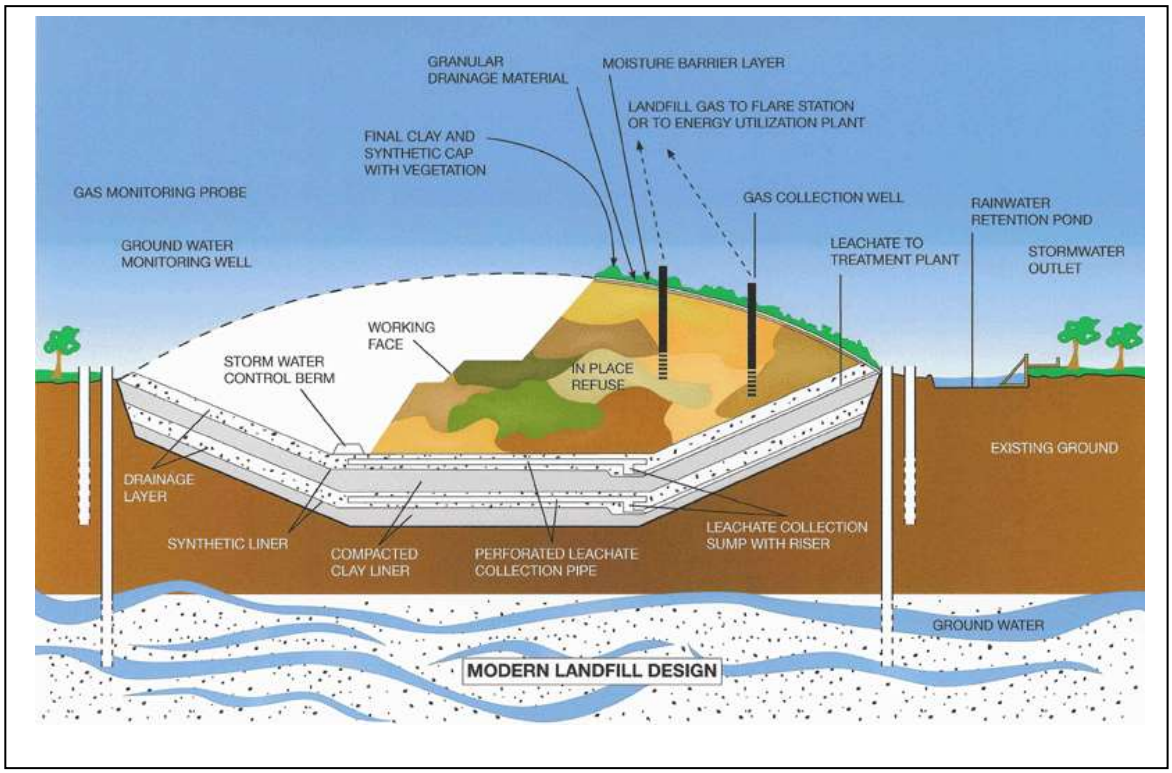


Fig 1.1: Conceptual Diagram of a Modern Engineered Landfill.

Source:http://runcoenv.com/images/landfill.gif&imgrefurl=http://runcoenv.com/landfill.htm&h=510&w=799&sz=182&tbnid=kC3zA-LCBemFBM:&tbnh=78

Most of the waste collected within the Lagos metropolis is deposited at the Lagos State Waste Management Authority's (LAWMA) operated waste dumpsites. These dumpsites (Figure 1.2) include Olusosun Dumpsite located in the Ojota area of Lagos, the Abule-Egba Dumpsite, along

the Lagos-Abeokuta Expressway, and the Solous 1, 2 and 3 Dumpsites, located along the LASU-Isheri Expressway.

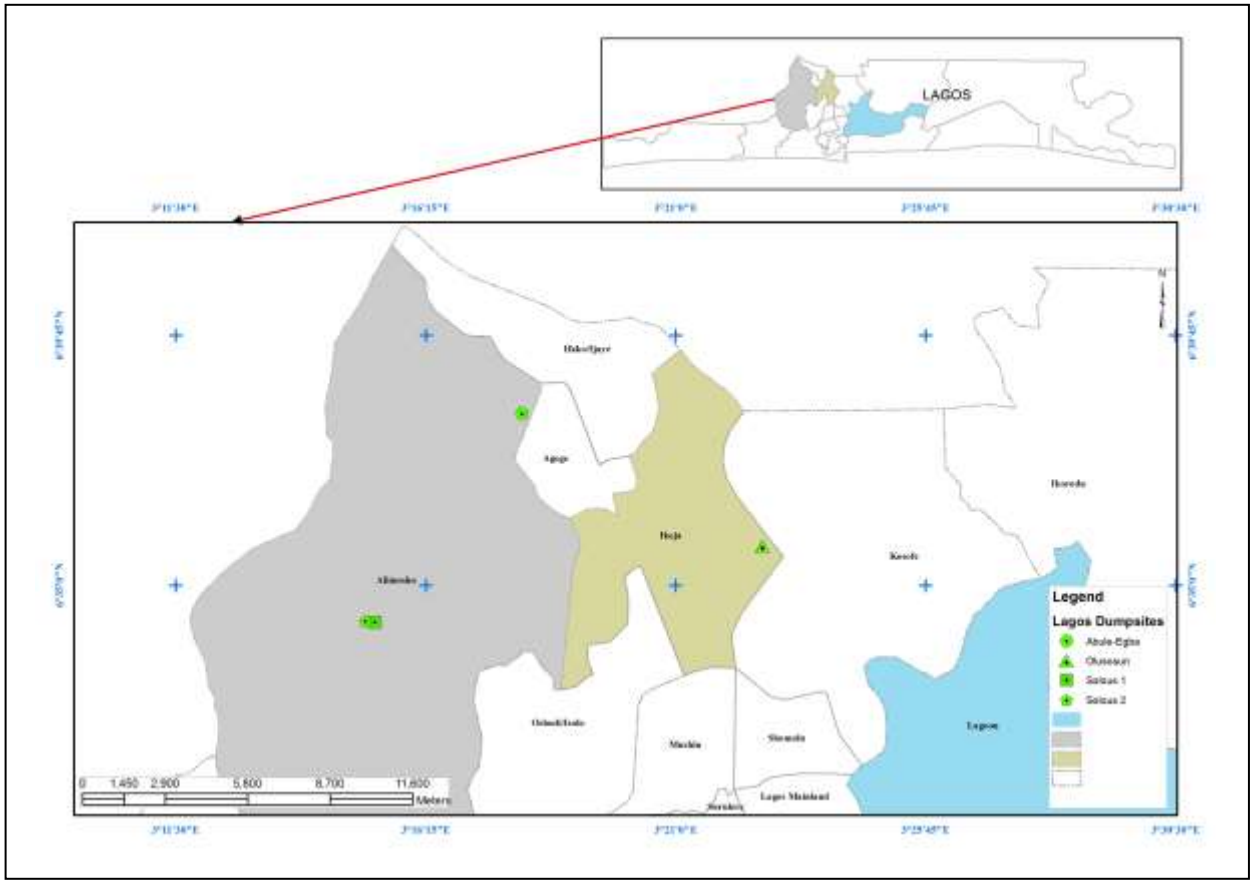


Fig 1.2: Location of Olusosun, Abule-Egba, Solous 1 and Solous 2 Dumpsites within Lagos

Due to the absence of the necessary facilities at these dumpsites, there exists a great potential for the underlying groundwater resources to be contaminated by leachate generated at these dumpsites especially, during the wet season (LAWMA, 2008). Factors identified by Johannessen (1999), as influencing the pollution potential of landfill leachate include the concentration and flux of the leachate, the landfill setting in terms of the hydrogeological setting and the degree of protection it offers, and the quality, volume, and sensitivity of the receiving groundwater.

The term groundwater sensitivity refers to the time required for a contaminant to move vertically from the land surface into an aquifer. It also involves the horizontal movement of groundwater and contaminants to and in the saturated zone (Grub & Alexander, 2009). According to Grub and Alexander (2009), one aquifer/landfill pair is more sensitive than a second aquifer/landfill pair if contaminants reaching the aquifer underlying the landfill at one location will migrate faster to the landfill boundary or well than at the other location.

The pollution potential of leachate and the vulnerability of groundwater to contamination raise a lot of public health concerns. This is due to the myriad of chemical compounds that could be in landfill leachate. Lee and Jones-Lee (1993) noted that the contamination of groundwater by leachate could render large volumes of groundwater unfit for domestic uses. As stated by Jones-Lee and Lee (1994), apart from the potential carcinogens and other highly toxic materials or matter that may be found in dumpsite leachate, municipal solid waste (MSW) leachate also contains many conventional pollutants that may render leachate- contaminated groundwater not only injurious to human health, but also reduce the service life of household appliances.

The recognition of the contamination potential of leachate especially with regards to the current waste disposal practices at various LAWMA operated dumpsites has led to a number studies aimed at investigating the impact of these dumpsites on groundwater quality within their vicinities. These include studies conducted on the impact of Olusosun Dumpsite on groundwater quality (Longe & Enekwechi, 2007; Adeyemi, et al., 2007; Oyeku & Eludoyin, 2010; Odukoya & Abimbola, 2010) and studies on the impact of Solous Dumpsite on groundwater quality (Longe & Balogun, 2010; Aderemi, et al., 2011).

In spite of the number of investigations that have been conducted on groundwater quality around municipal waste dumpsites in Lagos however, much has not been done to characterise groundwater flow around these dumpsites, whereas groundwater flow characterisation is an integral aspect of groundwater quality management around landfills.

Groundwater flow characterisation is pivotal to groundwater quality management around waste disposal facilities due to the complexity of the groundwater flow systems and the role of groundwater in the dissolution, mobilisation and transportation of contaminants within the subsurface (International Atomic Energy Agency [IAEA], 2001). Hence the opinion the National Academy of Sciences (1984, p.12) that the "effective utilisation of the subsurface as a repository for wastes depends on information as to how the wastes are transported". This is because once in the aquifer, the leachate plume migrating downgradient of a landfill or dumpsite is subject to physical and chemical processes in the groundwater depending on whether the contaminants in the leachate are conservative or reactive. Mechanisms which influence the movement of contaminants in groundwater are the physical processes of hydrodynamic dispersion for conservative contaminants, and chemical and biochemical processes or reactions such as acid-base reactions, oxidation-reduction reactions, solution-precipitation reactions, ion-pairing or complexation, microbiological processes and radioactive decay for reactive contaminants (Hiscock, 2005).

It is against this background that numerical groundwater models have increasingly been used by water resources managers and agencies responsible for groundwater quality protection for assessment and management of existing and future quality of groundwater around waste disposal sites. This situation is warranted by the increasing and widespread detection of contaminated groundwater systems, and facilitated by the available enhanced scientific capability in modelling

groundwater contamination and the advances in, and reduced cost of computer technology (van der Heijde, et al., 1988). This has translated into the provision of site specific predictions of the behaviour of specific contaminants, with the aim of predicting the spatial distribution of the contaminant's concentration or flux or both in a porous media.

In their discussion on groundwater modelling and management, van der Heijde et al. (1988) noted that groundwater modelling not only provides the basis and analytical framework for a better insight into the groundwater systems, it is also provides an understanding of the variety of processes that influence their quality, particularly those processes that are subject to anthropogenic intervention in the hydrogeologic system. Furthermore, groundwater modelling plays a significant role in policy making, development of environmental regulations, and the establishment of remedial actions for regulatory agencies responsible for groundwater (Davidson & Rao, 1988). Highlighting the roles of groundwater modelling in the planning and decision making involved in groundwater management, van der Heijde et al. (1988, p.12) stated that:

> Models can provide water resources managers with necessary support for planning and screening of alternative policies, making management decisions, and reviewing technical designs for groundwater remediation based on risk analysis of benefits and costs.

Groundwater modelling has been used in the development of policies and environmental regulations on standards for well setbacks, with respect to landfills and dumpsites. It has provided technical justification for the restriction of land disposal of hazardous wastes at specific waste disposal sites, and has also been used in evaluating the effectiveness of groundwater monitoring programs.

Beyond these, groundwater models are being adopted by operators of hazardous waste facilities to show compliance with permit requirements, while on the part of the regulatory agencies, they are being used to validate information provided for permitting purposes. Furthermore, groundwater models are being used to evaluate municipal and hazardous waste disposal site characteristics in order to determine the best locations for monitoring wells, and the estimation of transport and fate of contaminants at the subsurface of the waste disposal sites. It also helps in establishing the need for remedial actions and aquifer restoration in situations of groundwater contamination.

Other examples in which groundwater modelling has been utilised in groundwater management and protection programs include the development of aquifer or well head protection zones, assessment of exposure, hazard, damage and health risks associated with impaired groundwater quality, and the assessment of liabilities relating to the post closure phase of waste disposal sites.

It is in recognition of the aforementioned, and the need to protect the integrity of groundwater that numerical groundwater modelling has been employed in characterising and simulating groundwater flow and leachate transport within and around landfills and dumpsites in different parts of the world (Franz & Rowe, 1993; Kim, et al., 1999; Blumberga, 2001; Dong, et al., 2008; Nwachukwu, et al., 2010).

1.2 Statement of Problem

Landfills and dumpsites the world over, remain a source of environmental and public health concern. This is due to their adverse implications for public health, surface and groundwater quality, air quality, global warming and climate change. In the vicinities of LAWMA operated

dumpsites, a number of uncertainties surround the short and long-term impact of these dumpsites on the environment and its resources.

Firstly, the sole dependence of the residents on untreated groundwater, mostly from shallow wells and boreholes means that residents are at risk of consuming whatever deleterious matter that may be present in the water as a result of leachate contamination. With regards to human health, conditions such as damage to the central nervous system, irritation of the eye and respiratory tract, palpitations and dizziness have been associated with landfills and dumpsites (Ohio Department of Health [ODH], 2010). According to ODH (2010), the inhalation of Hydrogen Sulphide (H_2S) and Volatile Organic Compounds (VOCs) such as Benzene over a long period of time could lead to the development of certain types of cancer. Liver dysfunction has been attributed to leachate contamination of groundwater in a community in the United States (Health Protection Agency, 2011; Meyer, 1983).

Secondly, groundwater resources around dumpsites are rendered vulnerable to contamination especially in the light of observed leachate springs (Figure 1.3) which usually form at dumpsites during the rainy season. According to Taylor and Allen (2006), the mounding of water percolating through unlined landfill or dumpsites located above an aquifer results in the formation of leachate springs, as the increased hydraulic head developed leads to downward and outward flow of leachate from the dumpsite or landfill. Taylor and Allen further pointed out that the observation of leachate springs, which are the resultant effect of outward flow, is a pointer to the fact that leachate is not only being generated, but it is also migrating within the subsurface, posing a public health threat. Figure 1.4 is a representation of water table mounding and the resultant leachate spring at the periphery of a dumpsite.

The threat to groundwater integrity around LAWMA dumpsites is further exacerbated by the absence of environmental safeguard measures such as artificial sealing liners, upstream diversion channels and leachate collection ponds/control facilities (LAWMA, 2008). Furthermore, the siting of these dumpsites was not based on any hydrogeological considerations. Prior to the commencement of waste disposal operations, no hydrogeological investigations were carried out to determine the potential relationship between the groundwater flow regime and the dumpsites, and the possible effects this relationship might have on the quality of groundwater both within and around the dumpsites. These dumpsites were former laterite excavation sites or burrow pits in which wastes were deposited to gradually reclaim the land. At the commencement of waste disposal, the depth of the excavation site at Abule-Egba (Abule-Egba Dumpsite) was 12m, while the depth at Solous 1 was 9m (Aboyade, 2004).

Thirdly, landfill gases generated from these dumpsites pose significant threat to the environment and human health. From the perspective of the environment, landfill gases are a source of air quality degradation with characteristic unpleasant odour from hydrogen sulphide and other gases. The gases, particularly methane and carbon-dioxide which are green house gases are also a source of global warming. In addition, emitted nitrogen-sulphide enhances acid rain formation. Furthermore the diffusive transport of VOCs emitted alongside other landfill gases has the potential to contaminate groundwater in the dumpsites vicinities with carcinogens. The New Jersey Health Department (1994), identified Benzene, a known human carcinogen to be one of the VOCs discovered in private wells contaminated by leachate around Combe Fill South Landfill. Besides these, landfill gases are known inhibitors of vegetation growth, as well as being sources of explosions (methane) and asphyxiation (carbon-dioxide) hazards in confined

spaces (Ankeny & Stromberg, n.d; Lee & Jones-Lee 1993; Agency for Toxic Substances and Disease Registry [ATSDR], 2000).

Fig 1.3: Leachate Spring at Abule-Egba Dumpsite , 19th November, 2010

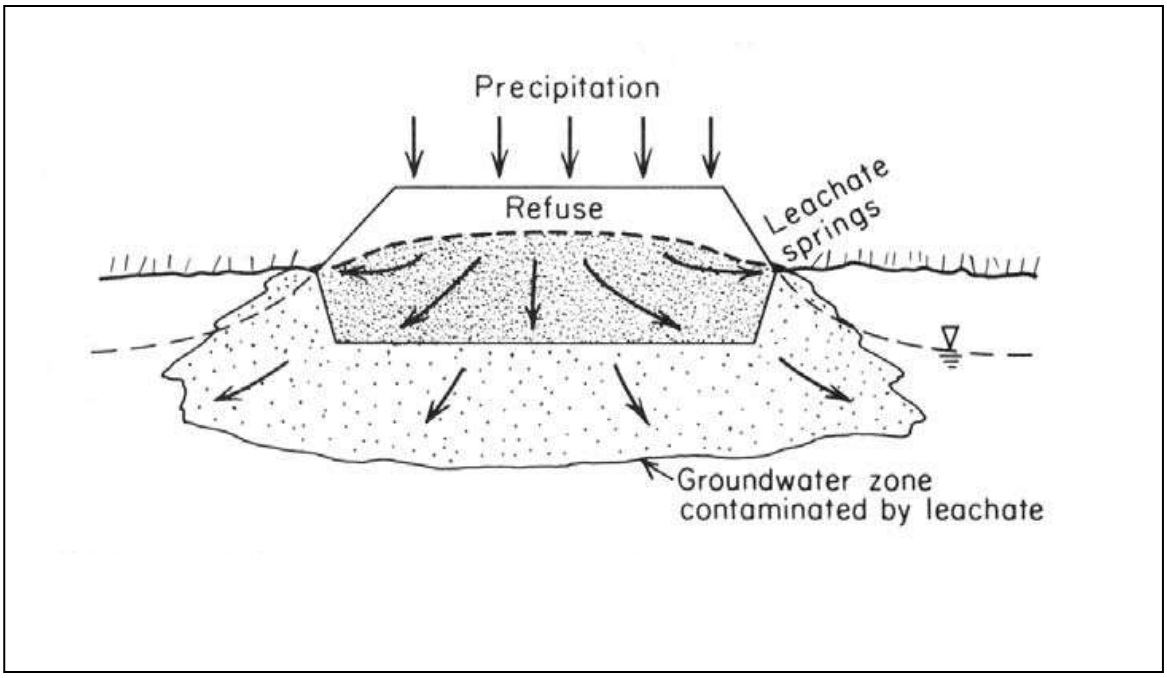


Fig 1.4: Water Table Mounding and Leachate Spring at the Periphery of Waste Deposited at a Dumpsite. (Source: Freeze and Cherry, 1979).

In view of the aforementioned issues, a good understanding of the groundwater conditions within and around LAWMA dumpsites is required. Reliable data and information on the groundwater flow dynamics of the dumpsites would not only aid the quantification of the interaction of the dumpsites with groundwater flow, but will also assist in identifying and modelling the potential pathways of leachate migration. One of the ways in which this can be achieved is through numerical groundwater modelling.

This study therefore seeks to provide real time data and information on basic groundwater movement pattern around some of LAWMA's dumpsites using numerical groundwater modelling. Such information enables the understanding of the groundwater flow system around the dumpsites and helps to forestall the contamination of water supply wells in the areas by preventing inappropriate location of such wells within the environment.

The total absence of LSWC's water supply distribution network at Abule-Egba, and the inadequate coverage of the pipe borne water supply network, coupled with the shortfall in production as in the case of the Igando mini water works, which has an installed capacity of 1.0 million litres/day, but a production capacity of 0.4 million litres/day, has led to dependence of the populace on groundwater.

In the vicinities of the dumpsites, majority of the houses have a well or a borehole, most of which were installed by well drillers with no formal training, and little or no knowledge of the geology and flow pattern of groundwater within the area. The implication of this is that most of the wells and boreholes were drilled without considering their locations relative to the dumpsites. The risk of groundwater contamination around the dumpsites is likely to be exacerbated due the rate of groundwater abstraction. Groundwater abstraction records are however not available. Over-abstraction of groundwater in these areas could lead to changes in flow pattern and the induction of leachate into the aquifers. This by implication means that residents are at an increasing risk of consuming low quality and contaminated water.

1.3 Aim and Objectives

The aim of this study is to assess the possible linkage between the groundwater flow pattern, the leachate generated at the dumpsites and the consequences for groundwater quality within the study locations. In line with the aim of the study, the specific objectives are as follows:

1. To establish a groundwater flow pattern for the vicinities of the dumpsites through numerical groundwater modelling.
2. To characterise and compute the Leachate Pollution Index of the leachates generated at the dumpsites in order to evaluate their contamination potential.

3. To determine the hydrochemical parameters responsible for most of the variability in the quality of groundwater around the dumpsites and establish their spatial patterns.

4. To compute the groundwater Contamination Index (C_d) in order to assess the degree of groundwater contamination around the dumpsites.

5. To establish a possible link between the groundwater flow patterns, the leachate generated at the dumpsites and the quality of groundwater.

6. To examine the implications of results obtained for waste dumpsites management.

1.4 Research Questions

To better articulate the purpose of the study, the following research questions are asked:

1. What is the groundwater flow pattern within the study locations?
2. What is the quality of leachate generated at the three dumpsites?
3. Which hydrochemical parameters accounts for most of the variability in groundwater quality around the dumpsites?
4. What are the implications of the flow pattern for contaminant migration?
5. Is there groundwater contamination? If yes, is the leachate generated the source of the contamination?
6. What are the implications of the results obtained for waste dumpsites management?

1.5 Significance of the Study

The interest in this study is multi-faceted. Firstly, since there is paucity of hydrogeological data and information in the study locations, this study provides pertinent information on groundwater flow pattern in the study locations. Furthermore, the study provides information on the characteristics of leachate generated at the dumpsites, the quality of groundwater around the dumpsites and probable migration pathways for contaminants. Information on the groundwater

flow patterns could be used as a measure to curtailing the likely contamination of water supply wells through proper siting and location of such wells.

Currently, there are no programmes put in place by LAWMA to monitor the extent of leachate infiltration and movement into groundwater at their dumpsites and environs. (Adeboye, personal communication, August 18, 2011). This study thus provides the basis for the putting in place of result-oriented monitoring programmes to enhance environmentally-friendly LAWMA activities.

Thirdly, residents around the dumpsites depend on groundwater for drinking and other domestic purposes, and their knowledge of the quality of groundwater they consume is limited. This study will provide useful information to all stakeholders on the quality of groundwater consumed by people residing near the dumpsites. Finally, the study fills some of the gaps in literature relating to hydrogeology and groundwater flow system around the study locations.

1.6 Scope and Limitation of the Study

The focus of the study is three-fold; characterisation of groundwater flow around the study locations, the potential of the leachate generated at the dumpsites to contaminate groundwater, and the variation of groundwater quality around the dumpsites. Analysis of leachate and groundwater quality spanned the wet and dry seasons, while groundwater flow characterisation is based on 2-Dimensional steady state groundwater conditions within the study locations and environs. The analyses were limited to selected chemical and heavy metal parameters that are commonly found in dumpsite leachates and which affect groundwater. Geological and hydrogeological input was based on accessible data. The scope of this study did not cover the health conditions of residents around the dumpsites.

1.7 Operational Definition of Terms

Acidogenic Leachate – Leachate formed during the acid phase of anaerobic biodegradation of waste, characterised by acidic pH values typically around 5 (Ghosh & Hasan, n.d).

Adsorption – The bonding of an organic chemical to the soil mineral surfaces (clay) or to the organic matter surfaces (Hamidi, 1999).

Complex Ion Formation – Reaction process by which compounds are formed in which molecules or ions form coordinate bonds to a metal atom or ion (Rivett et.al, 2006).

Confining Layer- The geologic strata that are of a lower hydraulic conductivity than the underlying strata under saturated conditions (Poehls & Smith, 2009). A relatively impermeable material stratigraphically adjacent to one or more aquifers (Todd & Mays, 2005).

Co-Precipitation- Involves the incorporation of a soluble compound into crystal structure of a forming mineral through substitution or into an existing mineral by diffusion (MacGregor, 2000).

Conservation of Mass- A natural law stating that matter cannot be be created or destroyed but can be changed to a different state (Poehls & Smith, 2009).

Contamination Potential – Susceptibility of groundwater to contamination by a specific contaminant or source (Zaporozec, 2002).

Desorption – A geochemical process that entails the removal of an adsorbed or absorbed substance, which is partially responsible for controlling the concentration of solutes in natural waters (Poehls & Smith, 2009).

Diffusion- Spread of a solute by molecular movement from zones of high concentration to zones of low concentration (Sharp, 2007).

Equipotential Lines – A contour line or isopleth joining points of equal hydraulic heads (Poehls & Smith, 2009).

Freundlich Isotherm- A description of sorption at a specified temperature with portioning of organic chemicals between the dissolved phase and the sorbed phase (Poehls & Smith, 2009).

Hydraulic Head- The height above a datum plane (such as sea level) of the column of water that can be supported by the hydraulic pressure at a given point in a groundwater system (Mandle, 2002).

Hydrodynamic Dispersion – A process that describes the macroscopic phenomenon whereby dissolved constituents, as well as suspended particulates and colloids are transported as a result of inhomogeneities in porous media (International Atomic Energy Agency, 2006).

Hydrophobic - Having aversion for water. It refers to colloids that do not readily hydrate yet do coagulate easily (Poehls & Smith, 2009).

Ion-Exchange – Replacement of ions within the mineral lattice of a solid with ions in the aqueous solution (Hiscock, 2005).

Langmuir Isotherm- A mathematical expression that assumes a non-linear in relating the mass of solute sorbed to the dissolved solute concentration (Poehls & Smith, 2009).

Leachate – Liquid that has percolated through solid waste and extracted dissolved or suspended materials that may include potentially harmful materials (Wraith, 2003).

Lithology- Driller's description of geologic characteristics, depth and intervals of the formation drilled through (Poehls & Smith, 2009).

Oxidation-Reduction – A chemical or biological reaction where an electron is transferred from an electron donor to an electron acceptor and results in a change in the valency state of the ion (Carey, et al., 2000).

Methanogenic Leachate– Leachate formed during the fourth phase of waste degradation in which intermediate acids are consumed by methane forming consortia (methanogenic bacteria)

and converted to into methane and carbon-dioxide. pH range is typically between 6 and 8 (Taylor & Allen, 2006).

Model- A representation of a real system or process (Konikow, n.d).

Precipitation – Removal of ions from solution by formation of insoluble compound (Rivett et.al, 2006).

Simulation- One complete execution of a groundwater modelling computer program, including input and output (Mandle, 2002).

Sorption- General process that solutes, ions and colloids become attached to solid matter in a porous medium (Sharp, 2007).

Velocity Vectors- Representation of groundwater flow directions at specific locations throughout the model domain (Scientific Software Group, 2012).

1.8 Study Area

1.8.1 Location

Three dumpsites within Lagos were selected as the study locations. These are the Abule –Egba Dumpsite and the Solous 1 and Solous 2 Dumpsites. The Abule-Egba Dumpsite is located within a residential area and lies between 6^0 38′15.55′′N; 3^0 18′04.57′′E and 6^0 38′26.79′′N; 3^0 18′ 09.97′′E; on 20 hectares of land at Oke-Odo, along Lagos-Abeokuta Expressway in Alimosho Local Government Area of Lagos. To the north of the dumpsite is the popular Katangwa Market, and to the south is the Oke-Odo Market. To the east of the dumpsite is the Lagos-Abeokuta expressway, while to the west are the mechanic village and a Celestial Church. Shown in Figure 1.5 are the Abule-Egba Dumpsite and its environs.

The Solous 1 and Solous 2 Dumpsites are located between 6^0 34′17.47′′N; 3^0 15′16.93′′E and 6^0 34′ 13.94′′N; 3^0 15′ 08.94′′E and between 6^0 34′19.37′′N; 3^0 15′05.96′′E and 6^0 34′ 15.68′′N; 3^0 15′ 01.31′′E, respectively. Both dumpsites are located opposite each other along LASU-Isheri Road in Igando Town, in the Alimosho Local Government Area of Lagos State. Solous 1 Dumpsite occupies approximately 3 hectares of land while Solous 2 Dumpsite is located on 7.8 hectares of land. Figure 1.6 shows the Solous 1 and Solous 2 Dumpsites and their environs.

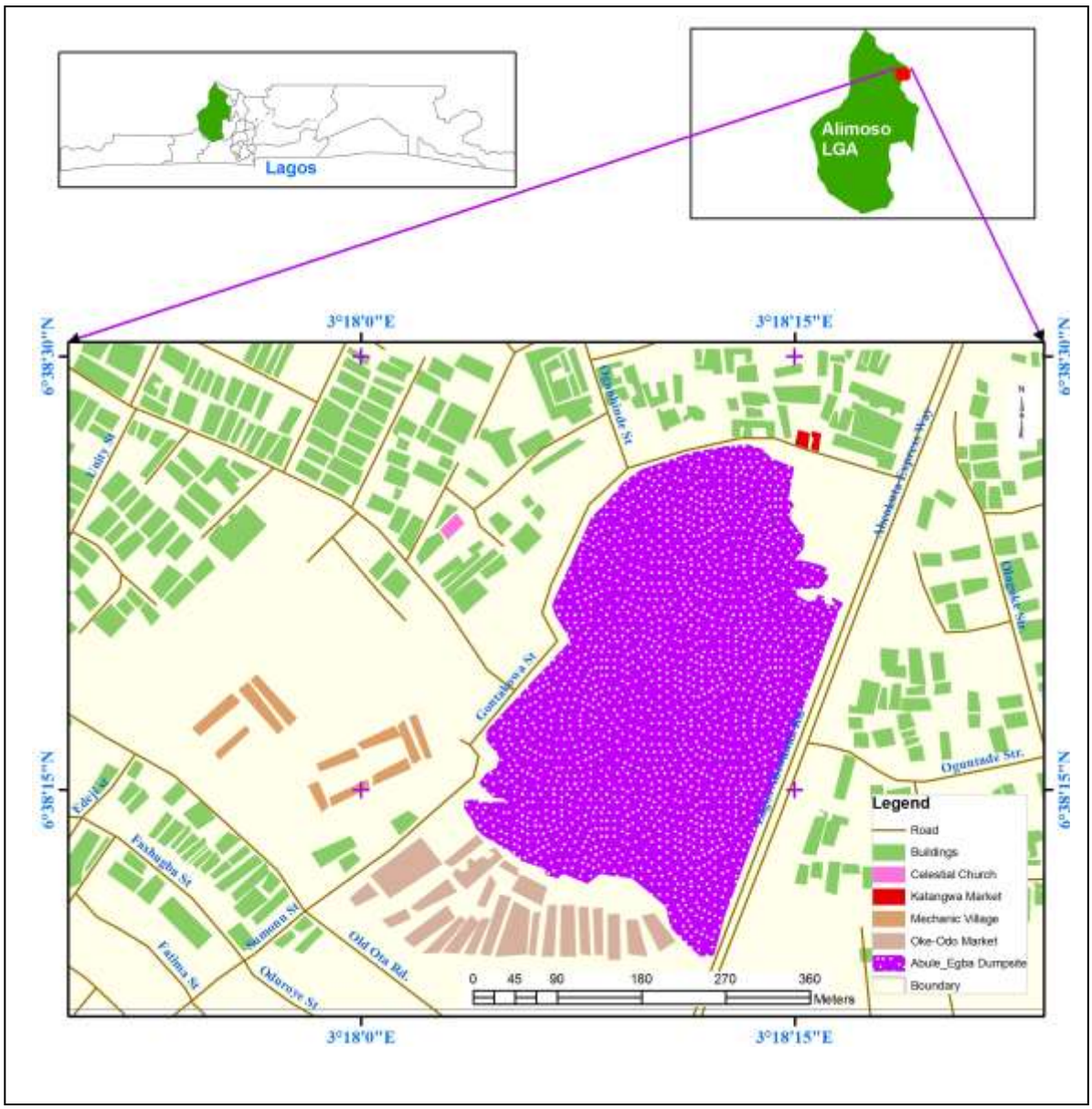


Fig 1.5 Abule-Egba Dumpsite and Environs

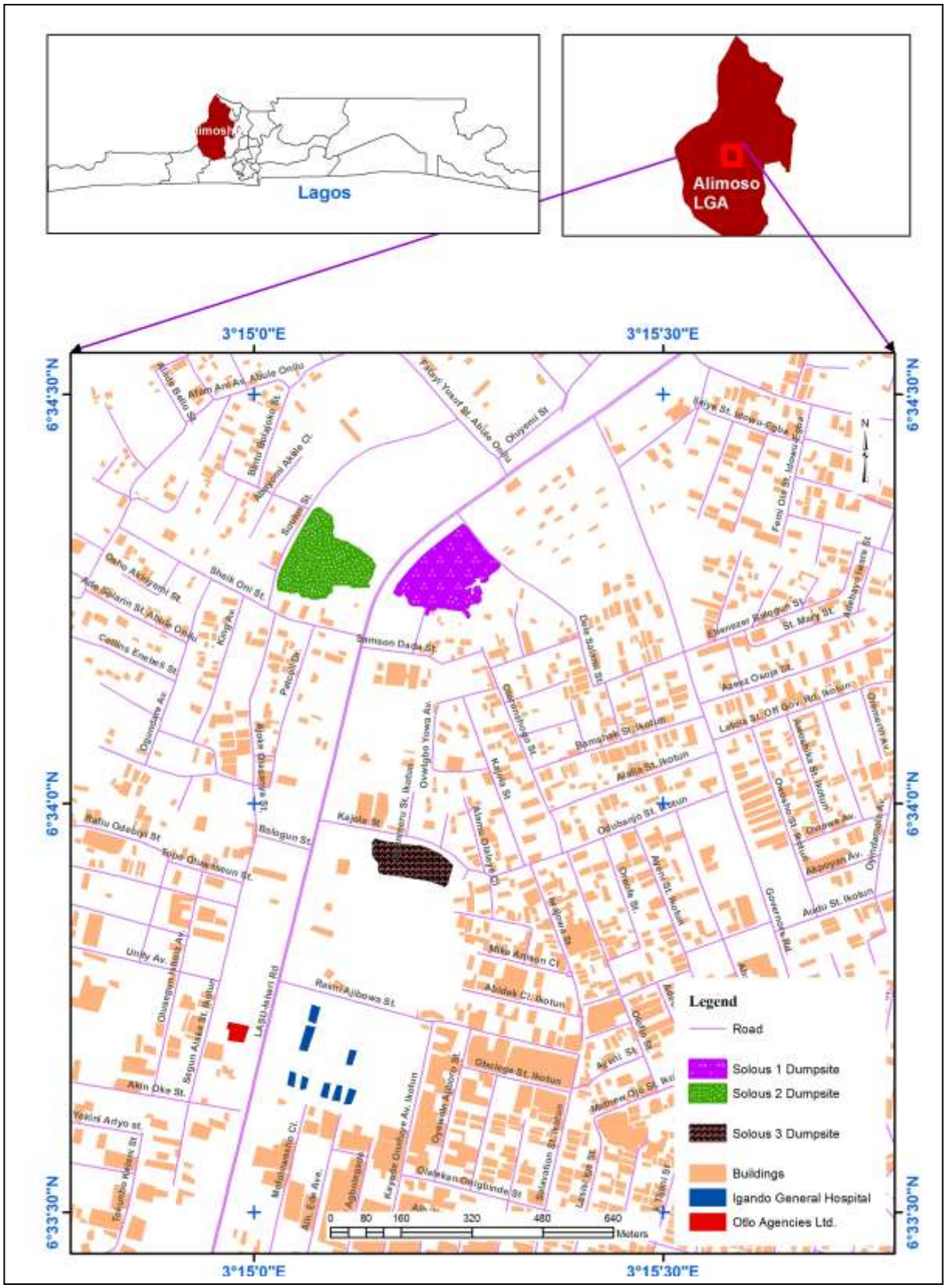


Fig 1.6 Solous Dumpsites and Environs

1.8.2 Geological and Hydrogeological Setting

The three dumpsites are located within the Coastal Plains Sands sedimentary rocks (Benin Formation) of southwestern Nigeria. The area is part of the Dahomey Basin which spans much of the continental margin of the Gulf of Guinea, extending from the Volta- Delta of Ghana in the west to the Okitipupa Ridge of Nigeria in the east (Nton, Ikane & Tijani, 2009). The coastal Plains Sands stratigraphic formation is the youngest of the Dahomey Basin. "It consists of soft, very poorly sorted, clayey sands, pebbly sands, sandy clays and rare, thin lignites" (Jones & Hockey, 1964, p.74).

Records available from the 1997 hydrogeological investigation of Lagos State (Kennard & Lapworth 1997) put the transmissivity of the Coastal Plains Sands formation in Lagos at $100m^2$/day for the Upper Coastal Plains Sands, and between a range of 95-$3700m^2$/day for the lower Coastal Plains Sands. Figure 1.7 is a representation of the stratigraphy of the Eastern Dahomey Basin.

The dumpsites are underlain by sedimentary rocks (Benin Formation). The lithostratigraphy is such that the sub-surface of the dumpsites are characterised by intercalations of laterite, clay and sand as revealed by the records of borehole lithologs from the dumpsites (LAWMA, 2008; Environquest, 2008). This geological sequence is consistent with the results of the studies conducted by Longe, et.al (1987), who found that the lithostratigraphy in Lagos area is characterised by an alternating sequence of sand and clay. Tables 1.1 to 1.3 and Tables 1.4 to 1.6 represent the lithological profile at the Abule-Egba and Solous 1 Dumpsite, respectively.

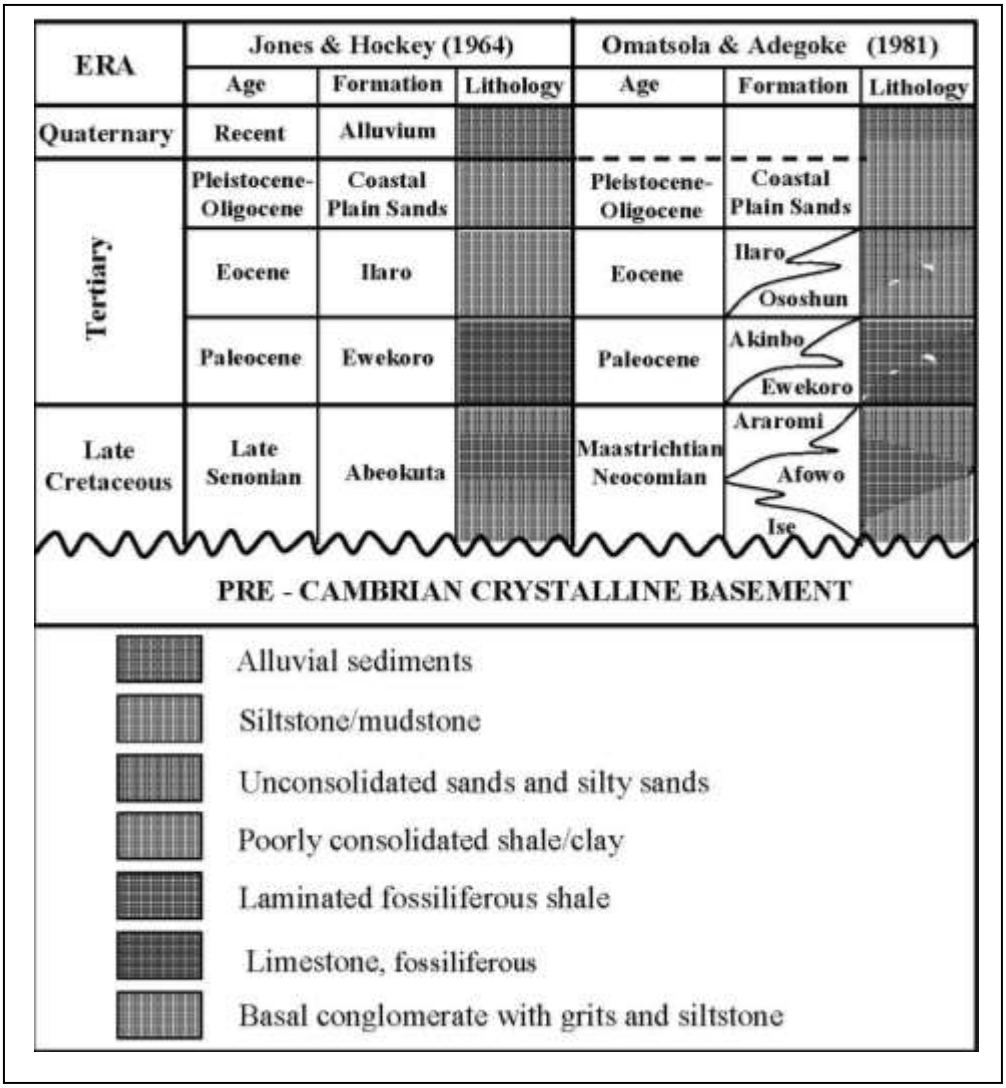


Fig 1.7: Stratigraphy of Eastern Dahomey Basin. (Source Nton et al., 2009).

1.8.2.1 Hydrogeological Profile

The hydrogeological profiles of the monitoring boreholes drilled at Abule-Egba dumpsite is as presented in Tables 1.1 to 1.3, while Tables 1.4 to 1.6 represents the hydrogeological profiles of the monitoring boreholes drilled at Solous 1 Dumpsite. Detailed information on the hydrogeological profile of Abule-Egba Dumpsite and the Solous Dumpsites is presented in Appendix 1 and Appendix 2 respectively.

Table 1.1: Lithological Profile of Borehole 1 at Abule-Egba Dumpsite

Depth		Lithology
From (m)	To (m)	
0	4	Decomposed refuse, Darkish Topsoil
4	7	Darkish Clay
7	12	Reddish Brown Clay
12	20	Very fine grained Silty Sand
20	22	Whitish Clayey Silty Sand
22	26	Very Fine Grained Silty Sand
26	30	Fine-Medium Grained Sand

Source: Environquest (2008)

Table 1.2: Lithological Profile of Borehole 2 at Abule-Egba Dumpsite

Depth		Lithology
From (m)	To (m)	
0	3	Reddish Clay
3	6	Medium-Coarse Grained Reddish Sand
6	9	Reddish Clay Sandy,
9	12	Medium Grained Sand
12	15	Medium-Coarse grained Sand
15	18	Clay
18	21	Sandy Clay
21	24	Clayey Sand
24	27	Fine Grained Sand
27	30	Fine-Medium Grained Sand
30	33	Fine Grained Sand
33	36	Coarse Grained Sand
36	39	Fine-Medium Grained Sand
39	42	Fine Grained Clayey-Sand

Source: Environquest (2008)

Table 1.3: Lithological Profile of Borehole 3 at Abule-Egba Dumpsite

Depth		Lithology
From (m)	To (m)	
0	3	Reddish Clay
3	6	Reddish Sandy Clay
6	9	Clay
9	12	Sandy Clay
12	15	Fine Grained Sand
15	18	Sandy-Clay
18	21	Clayey Sand
21	24	Sandy Clay
24	27	Clay
27	30	Clayey Sand
30	33	Fine-Medium Sand
33	36	Medium-Coarse Sand
36	39	Coarse Sand
39	42	Clayey Sand
42	45	Clay

Source: Environquest (2008)

Table 1.4: Lithological Profile of Borehole at Solous1 Dumpsite

Depth		Lithology
From (m)	To (m)	
0	1	Topsoil, Clay, Brownish
1	10	Laterite
10	12	Whitish Clay
12	15	Clayey Sand
15	19	Fine Grained Clayey Sand
19	20	Fine Grained Sand
20	21	Whitish Clay
21	26	Fine Grained Sand
26	27	Clayey Sand
27	28	Whitish Clay
28	29	Clayey Sand
29	31	Whitish Clay

Source: Environquest (2008)

Table 1.5: Lithological Profile of Borehole 2 at Solous 1 Dumpsite

Depth		Lithology
From (m)	To (m)	
0	1	Clay Topsoil
1	8	Laterite
8	10	Whitish Silty Clay
10	11	Sandy Clay
11	12	Fine Grained, Clayey Sand
12	14	Fine-Medium Grained Sand
14	16	Fine-Medium Grained Sand
16	18	Fine Grained Clayey Sand
18	30	Fine Grained Sand

Source: Environquest (2008)

Table 1.6: Lithological Profile of Borehole 3 at Solous 1 Dumpsite

Depth		Lithology
From (m)	To (m)	
0	1	Clay Topsoil
1	9	Laterite
9	12	Clay
12	13	Brownish Fine Grained Sand
13	14	Medium Grained Sand
14	15	Clayey Sand
15	16	Brownish Clay
16	18	Whitish Clay
18	19	Whitish Sandy Clay
19	21	Brownish Fine Grained Sand
21	22	Medium Grained
22	24	Silty Sand
24	26	Fine-Medium Grained Sand
26	29	Whitish Clay
29	30	Brownish Clayey Sand

Source: Environquest (2008)

1.8.3 Geomorphological Setting

Rivers Illo, Abesan and Opomu (Fig 1.8) are the main surface hydrological features around Abule-Egba and Igando respectively. These rivers are tributaries of River Owo. The source of River Owo is in a town called Toto-Owo in Ogun State, where Rivers Ore and Illo form a confluence with River Opomu. Discharge from River Owo constitutes a major source of water for the Ologe Lagoon (Onuoha, et al., 2010).

River Illo traverses 24 km of land along the Lagos and Ogun State boundary (Omole & Longe, 2008). River Abesan has a maximum depth of about 3metres and the valley is flanked on both sides by aquatic macrophytes and other rain forest vegetation. It traverses various housing estates on its course into the Ologe Lagoon (Agboola & Denloye, 2011).

In the Oke-Odo area where the Abule-Egba Dumpsite is located, v-shaped valleys and depressions which is a common feature along the different tributaries of the Owo River is noticeable (Alimosho Model City Plan, 2010). Elevation around the dumpsite ranges between 40 and 42 metres. Around the Igando area, the topography is mainly undulating with elevation ranging between 28 and 30m around the Solous Dumpsites.

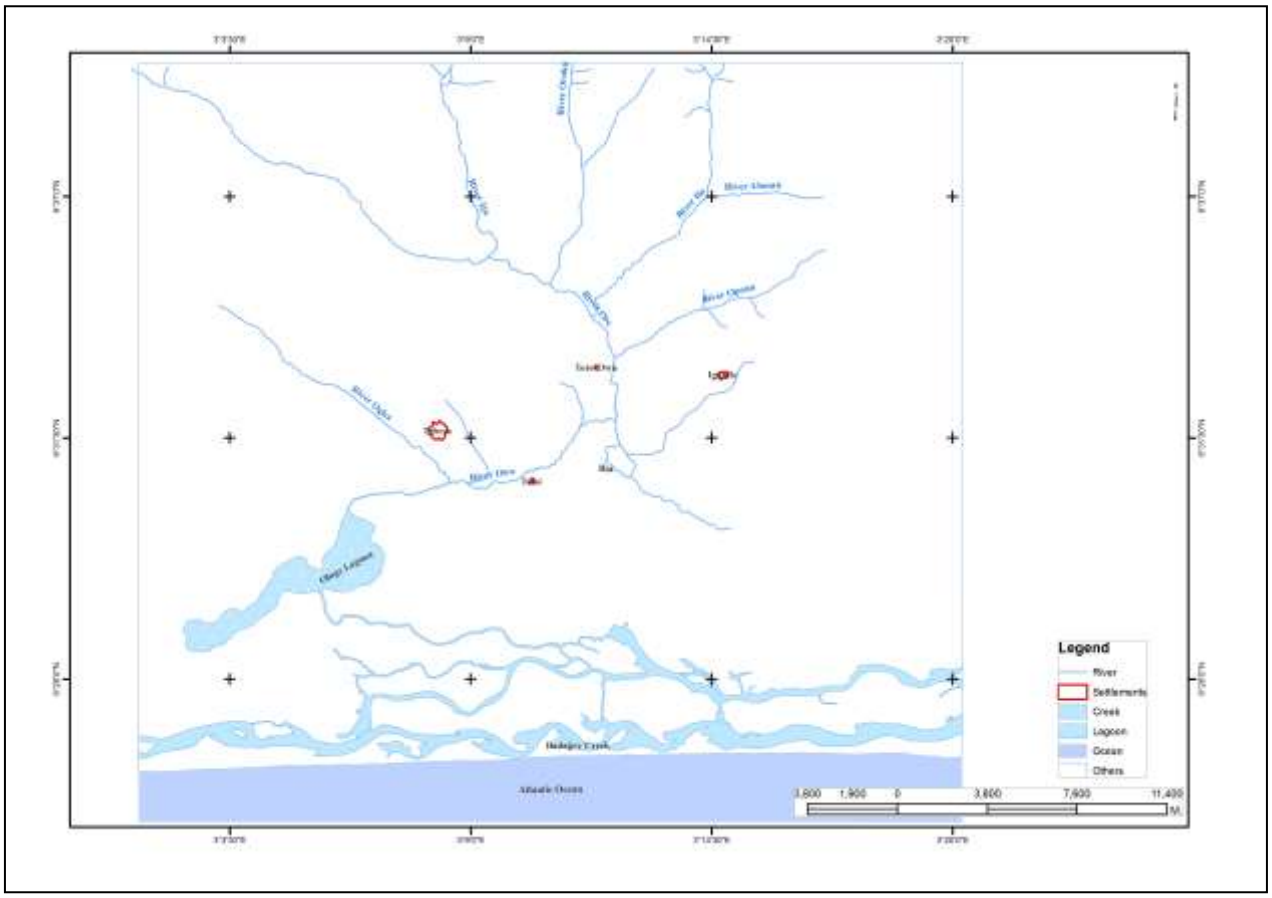


Fig 1.8: Hydrological Features around Abule-Egba and Solous Dumpsites

1.8.3 Climate

The climate of Abule-Egba and Igando is consistent with that of the Lagos Metropolis. It is influenced by tropical air masses characterised by the moisture laden Southwesterlies and the continental Northeasterlies. The movement of these two air masses is responsible for the dry and the wet seasons experienced in Lagos. The dry season in Lagos is usually between the months of November and March, while the wet season usually lasts from April to October. Annual rainfall around the study area is about 1,500mm (Odumosu, 1999). The rainy season is characterised by two peak periods, the first peak when the heaviest rains of the season is received lasts from May to July, while the second peak is between the months of September and October (Building Nigeria's Response to Climate Change [BNRCC] Project, 2012). Maximum temperature around

the study locations ranges between 28 and 33^0C, and minimum between 24 and 26^0C, with narrow diurnal and annual ranges. Relative humidity varies between 60 and 80% all year round, with a steady vapour pressure (Odumosu, 1999).

1.8.4 Vegetation

Vegetation cover at the dumpsites comprises of uncultivated pawpaw and cassava, weeds such Ipomoea mauritania, Solanum torvum, Bull grass and Amaranthus spinosus; creepers such as Luffa aegyptiaca; shrub such as Sodom apple, Coral nut and Golden touch; and trees such as Indian almond, Mango, Neem tree, Dormilon and Coconut (LAWMA, 2008).

1.8.5 Landuse

Abule-Egba and Igando are predominantly residential, with the dumpsites serving as one of prominent features of the communities. According to the Alimosho Model City Plan (2010), the Abule-Egba, Solous 1 and Solous 2 Dumpsites account for 0.15% of the total landmass of the Alimosho Local Government Area. There is also the commercial landuse class consisting mainly of Ala-Gatankowa Market, Oke-Odo Market, Ile-Epo Market and Super Market, all within the Abule-Egba Dumpsite vicinity and the Igando Market, Igando Iron Market and Isheri Night Market in Igando.

Major roads around the Abule-Egba Dumpsite include the Lagos-Abeokuta Expressway, Old Otta Road, Hammed Mohammed Road, and Katangua Road, while the LASU-Isheri Road is the major road at Igando. At Abule-Egba educational landuse comprises of two public primary and two public secondary schools. Other facilities that make up the landuse pattern around Abule-Egba and Igando include health care facilities such as the Orile-Agege General Hospital for the Abule-Egba residents and the Igando General Hospital for the residents of Igando.

1.8.6 Human Setting

The population of Abule-Egba and Igando is multi-ethnic, though the indigenous Yoruba people are the predominant. The population has been on the increase over the years due to influx of people from other parts of Lagos and beyond. The population is made up of mostly traders and artisans, many of which fall within low and middle income categories (LAWMA, 2008). At Abule-Egba the commonest type of houses is the rooming or multiple room buildings known as "face to face". These buildings are occupied by an average of six families, while at the Solous area of Igando the single family bungalow is the predominant type of building.

1.8.7 Operational Practices

Abule- Egba Dump site was commissioned in 1983, and decommissioned in 2009 after an expected lifespan of 26 years. During its operational life the dumpsite received an average of 72 trucks daily from six areas of Lagos state; Agege, Alakuko, Ajegunle, Orile Agege, Pandora and Ojokoro. The site also received waste from cart pushers (LAWMA, 2008).

The Solous 1 Dumpsite was opened in 1981 (Aboyade, 2004). Prior to its official closure in 2008, waste collected from Okota, Isolo, Ire-Akari, Ajao Estate, Iba, Makoko and Akowonjo were deposited at the dumpsite (LAWMA, 2008). Active tipping of waste commenced at the Solous 2 Dumpsite in 2008.

Scavenging of recyclable waste are some of the activities at the dumpsites. Items scavenged include metals, glass and plastics. In figures 1.9 to 1.12, the prevailing environmental conditions, as well as operational and human activities at the dumpsites are shown.

Fig 1.9: Mound of Refuse and Scavenging Activities at Abule-Egba Dumpsite, 19th November, 2010

Fig 1.10: Leachate Runoff at Abule-Egba Dumpsite, 19th November, 2010

Fig 1.11: Waste deposited at Solous 1Dumpsite, 20th November, 2010

Fig 1.12: Scavenging Activities at Solous 2 Dumpsite, 20th November, 2010

1.8.8 Characterisation and Volume of Waste Tipped

Waste tipped at the dumpsites include, household, commercial, industrial, and biomedical waste (LAWMA, 2008). These wastes are received at the dumpsites through the Local Government Authorities trucks, LAWMA trucks, licensed Private Sector Participation (P.S.P) operators, Cart pushers and other sources such as waste from residences and businesses close to the dumpsites. According to LAWMA (2008), 75% of the waste deposited at Abule-Egba Dumpsite, and 82% of the waste deposited at Solous 1 Dumpsite are combustible wastes made up of textiles, wood, plastics, rubber and leather. The Non-combustible wastes which include metals and glasses

account for 25% and 18% of the waste deposited at Abule-Egba and Solous1 Dumpsites respectively.

The analysis of LAWMA data on the volume of waste deposited at Abule-Egba Dumpsite showed that in 2007, a total of 617,657.50 cubic metres of waste was deposited, while in 2008, a total of 688,617.95 cubic metres of waste was deposited. Between January and June 2009, a total of 327, 375 metric tonnes of waste was received at the site. Figures 1.13 and 1.14 depict the volumes of waste from different sources deposited at the dumpsite in 2007and 2008 and Figure 1.15 represents the volume of waste deposited between January and March and April to June 2009.

At Solous 1 Dumpsite, a total of 526,322.50 and 605,141.50 cubic metres of waste was deposited in 2007 and 2008 respectively, while between January and June 2009, a total of 428,729 metric tonnes of waste was received at Solous 2 Dumpsite. Figures 1.16 and 1.17 show the volume of waste received from various sources at Solous 1 Dumpsite for 2007 and 2008, while Figure 1.18 shows the volume of waste received at Solous 2 Dumpsite between January to March, and Apr to June 2009.

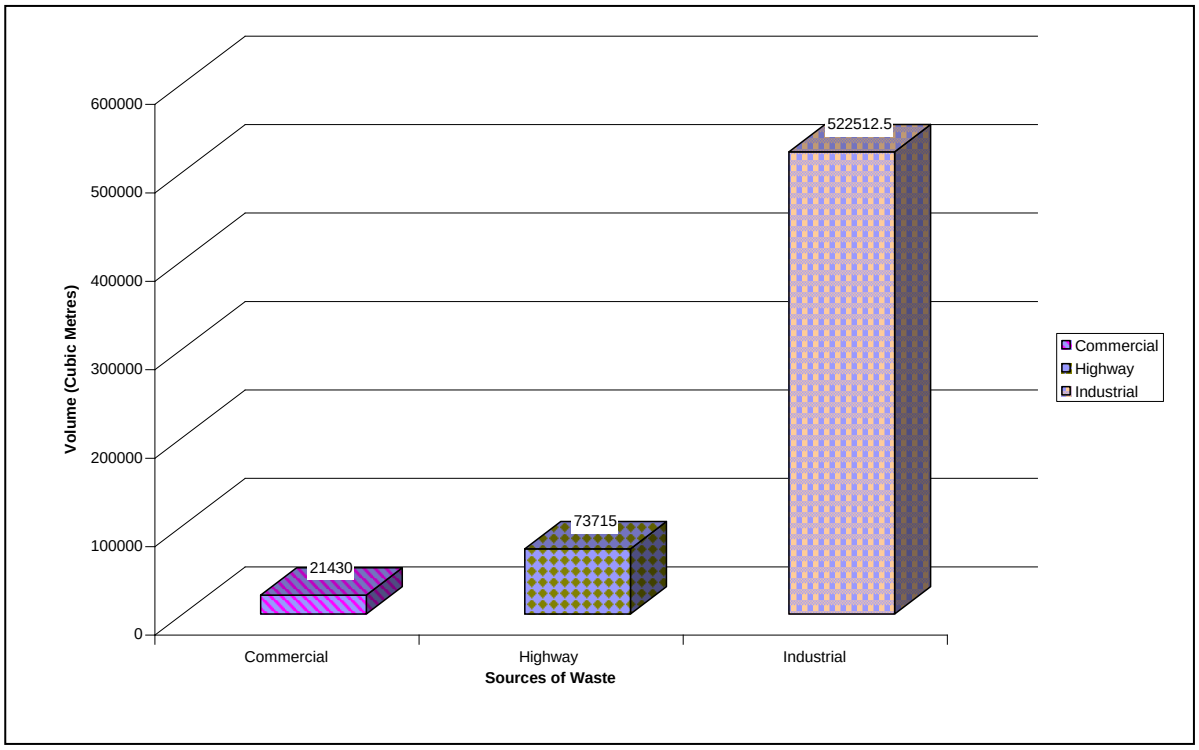


Fig 1.13: Total Volume of Waste from Different Sources Deposited at Abule-Egba Dumpsite in 2007

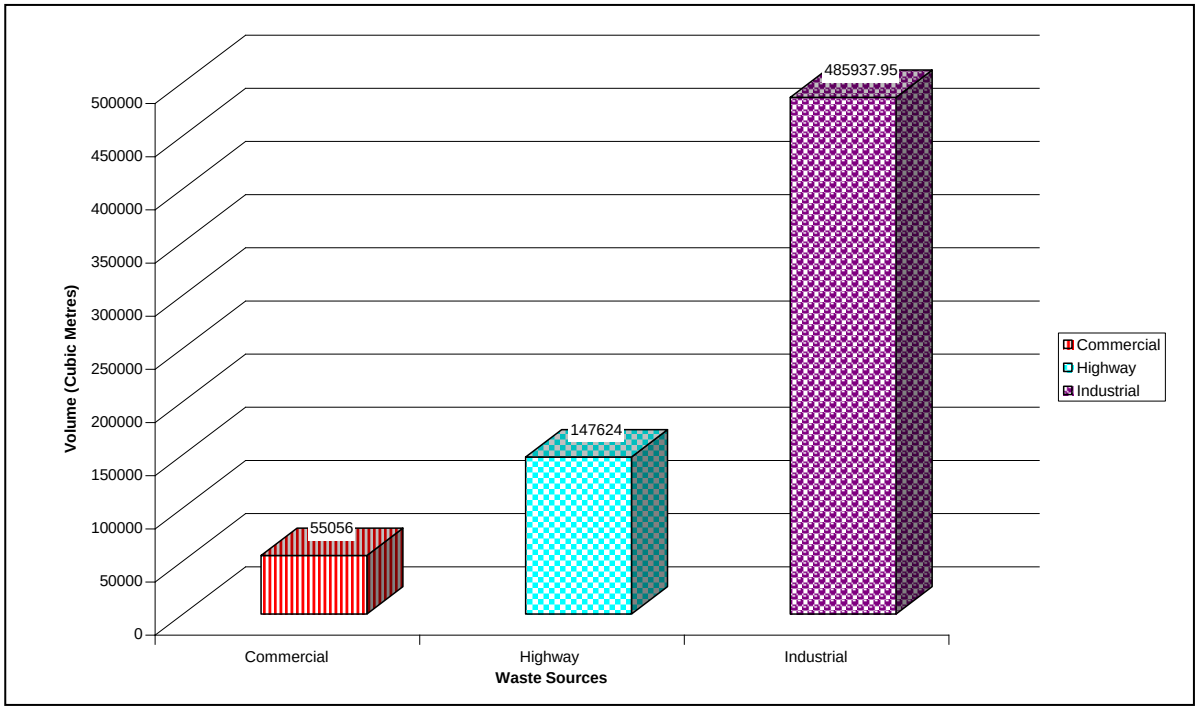


Fig 1.14: Total Volume of Waste from Different Sources Deposited at Abule-Egba Dumpsite in 2008

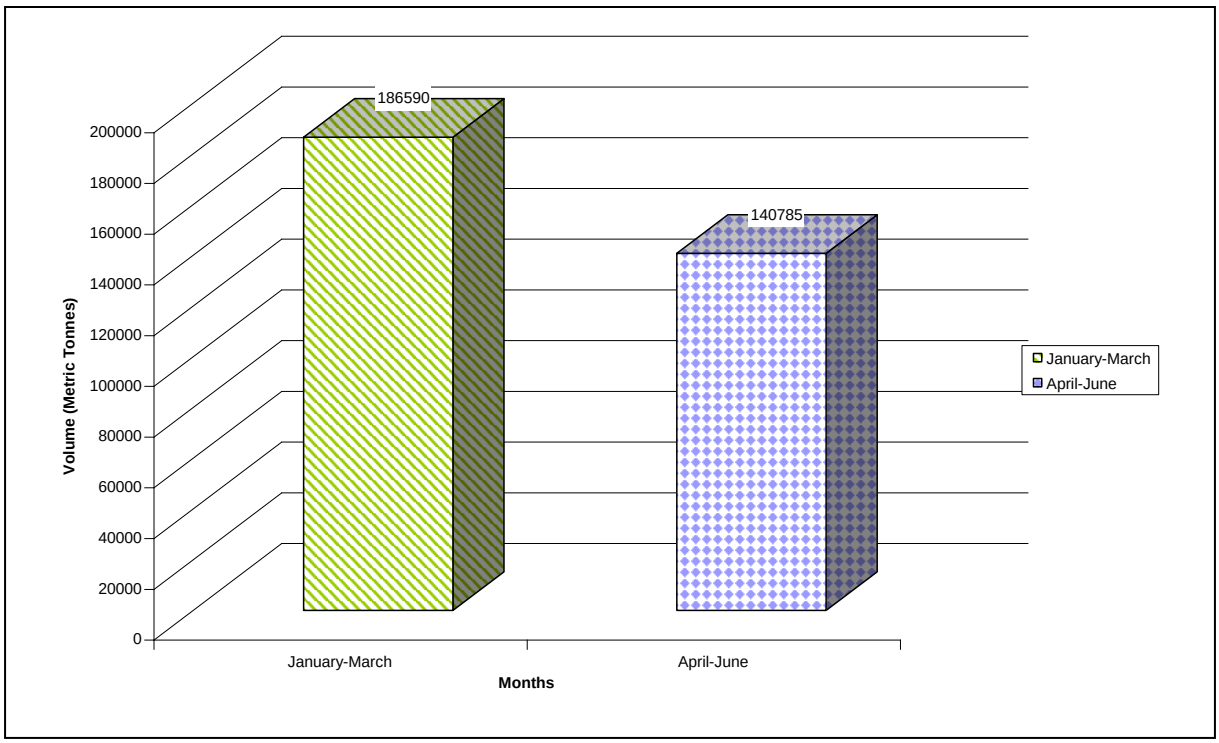


Fig 1.15: Total Volume of Waste Deposited at Abule-Egba Dumpsite from January to June 2009

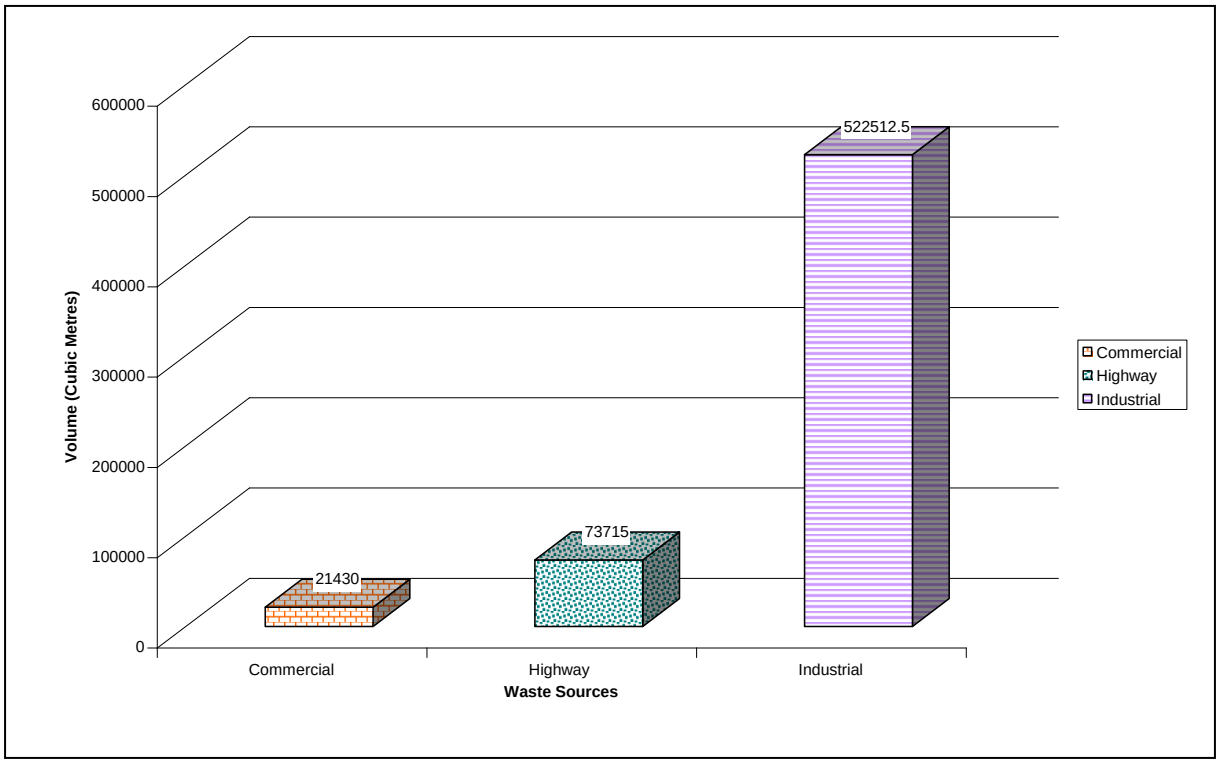


Fig 1.16: Total Volume of Waste from Different Sources Deposited at Solous 1Dumpsite in 2007

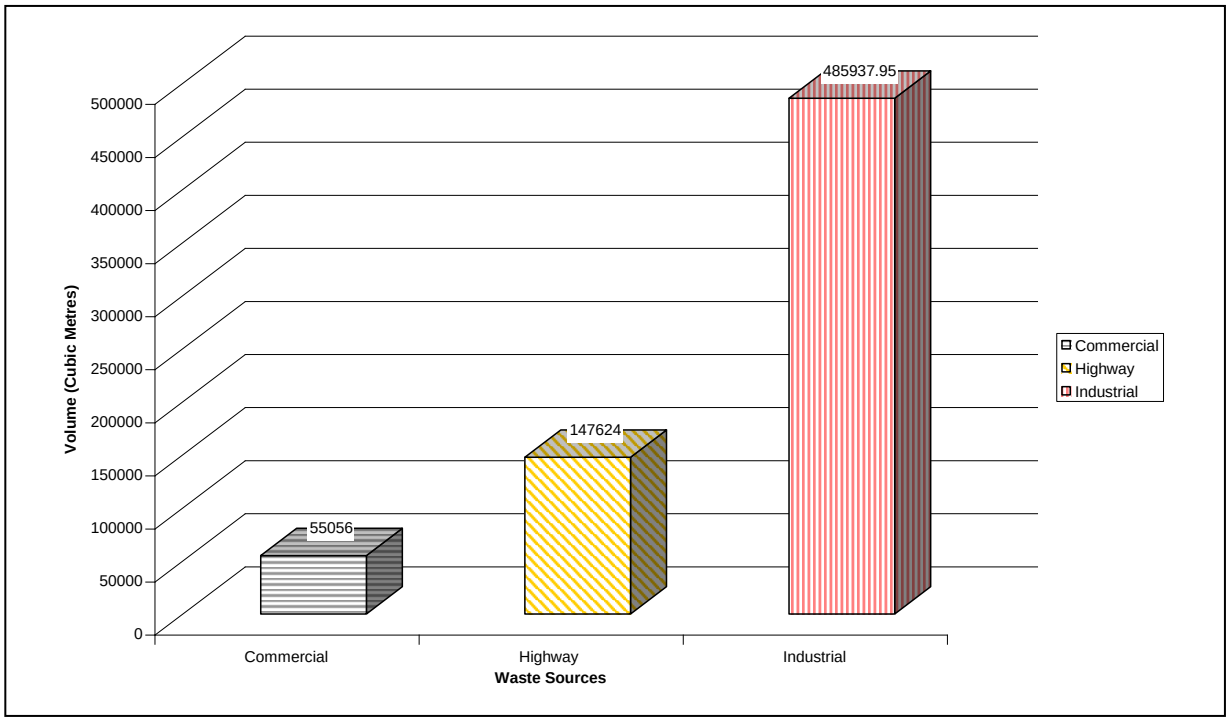


Fig 1.17: Total Volume of Waste from Different Sources Deposited at Solous 1Dumpsite in 2008

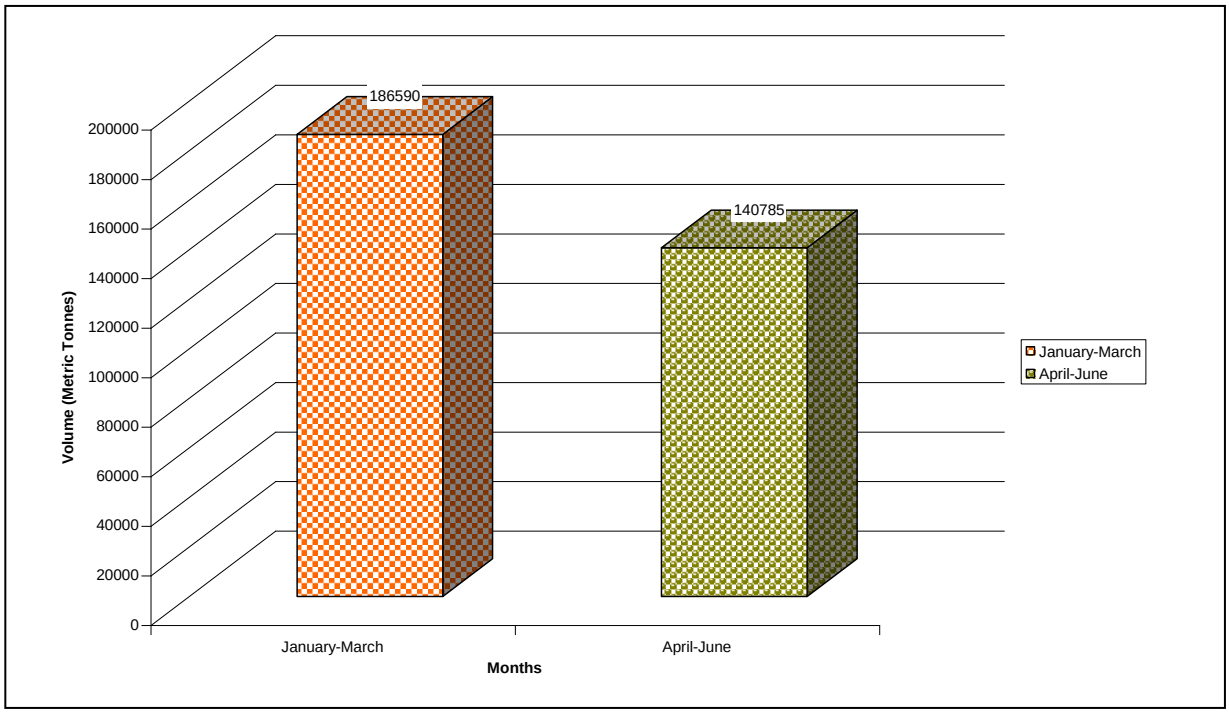


Fig 1.18: Total Volume of Waste Deposited at Solous 2 Dumpsite from January to June, 2009

CHAPTER TWO

LITERATURE REVIEW AND CONCEPTUAL FRAMEWORK

2.1 Literature Review

Literatures were reviewed based on the aim and objectives of this study. A number of literatures were reviewed on the following subject matter; groundwater flow modelling, processes operating in landfills and dumpsites, landfill leachate characterisation and contamination potential, leachate transport through a landfill or dumpsite, approaches used in the assessment of leachate migration from waste disposal sites, impact of landfill leachate on groundwater quality, use of index in the assessment of groundwater quality and attenuation of landfill leachate.

2.1.1 Groundwater Flow Modelling

In recent times the use of groundwater flow models has gained prominence in the description and characterisation of the groundwater flow system and in the study of wide range of hydrogeologic conditions and environments. The predictive capabilities of these groundwater flow models has greatly increased its utilisation in water quality and contamination studies, such as in the prediction of contaminant transport and fate, identifying areas of potential environmental risk, identification of contamination sources, and assessment of possible remedial actions (Van der Heijde, et al.,1988).

The increased usage of groundwater flow models has over the years resulted in the development of a wide variety of groundwater flow models. Kumar (2006) identified fifty of the most commonly used groundwater models and their features. Some of the models identified include 3DFEMAT, AQUA3D, POLLUTE, MOC, PEST, MODPATH, MT3D, UCODE, ChemFlux, PMPATH, FLOWPATH, MODFLOW and Processing Modflow (PMWIM), a graphical interface

for MODFLOW, MODPATH, PMPATH, MT3D, PEST UCODE, SUTRA and SOLUTRANS. Among all the models however, MODFLOW, developed by McDonald and Harbaugh (1984) is the most widely used.

"MODFLOW is a computer program that numerically solves the three dimensional groundwater flow equation for a porous medium by using a finite-difference method" (Harbaugh, et al., 2000, p.1). The model simulates flow in aquifer systems using the finite difference method (United States Geological Survey [USGS], 1997). The finite difference method approximates the first derivates of the partial differential equations which are based on the conservation of mass and Darcy's Law. The partial differential equations are solved for unknown heads at discrete nodes collected within an orthogonal network of nodes. In MODFLOW, the finite difference method utilises the block-centred node approach, in which each node is placed in the centre of each block (Konikow & Reilly, 1999; Fitts, 2002).

The aquifer system or model domain in MODFLOW is discretised or sub-divided into rectangular blocks (or cell) of grids comprising of rows, columns and layers. Within each block or cell, the physical properties of the aquifer are assumed to be homogeneous. Heterogeneity of the aquifer system can also be established by assigning varying properties to different blocks (Fitts, 2002).

According to the USGS (1997, p.1), "MODFLOW can simulate a wide variety of hydrologic features or processes". These include the simulation of steady state and transient flow in unconfined and confined aquifer, as well as confining units. Features and processes that can also be simulated by MODFLOW include streams, rivers, springs, wells, reservoirs and drains, and evapotranspiration and recharge from precipitation and irrigation (USGS, 1997).

For any of these features and processes to be simulated in MODFLOW, the following conditions which are also related to the simulation capabilities of MODFLOW must be met by the aquifer system. These include the existence of saturated flow conditions, constancy of groundwater density, invariability of the principal directions of horizontal hydraulic conductivity or transmissivity within the aquifer system and flow conditions being subject to Darcy's Law (USGS, 1997).

A number of reasons exist for the wide usage of MODFLOW. Kumar (2006,p.158) identified some of the advantages as "numerous facilities for data preparation, easy exchange of data in standard form, extended worldwide experience, continuous development, availability of source code and relatively low price". Others are its ability to manage large data sets required when running large problem, and its ability to allow the user to examine specific hydrologic features of the model independently due to its sub-division into independent sub-routines called modules. Other advantages and reasons for the wide usage of MODFLOW as identified by the USGS (1997) include the following:

1. The relative ease and applicability of the finite difference method of MODFLOW to a wide variety of real conditions.
2. Its ability to work on different types of computer systems with minimal changes.
3. Its capabilities for simulating groundwater in one dimension, two dimensions, quasi-three dimensions and three dimensions.
4. The rigorous test and peer review of each simulation feature.
5. Detailed documentation of input instructions and theories.
6. The relative ease with which new features are allowed to be added due to its modular program design.

7. Its compatibility with input data from other computer programs written by the USGS and other United States Federal agencies.
8. The flexibility of MODFLOW design in allowing other computer programs to read and present the model results in easily understandable formats.

The disadvantages associated with MODFLOW include its inability to simulate multiphase flow, flow in the unsaturated zone and flow in fractured media unless it can be assumed to be an equivalent of a porous media. Other disadvantages of MODFLOW include its inability to simulate density-dependent flow, or flow in an aquifer with varying anisotropy conditions (Fetter, 2001).

Apart from MODFLOW other groundwater models that have been developed and designed for groundwater flow and contaminant transport studies around landfills and dumpsites include MODPATH, VISUAL MODFLOW, FLOWPATH, USGS-MOC and POLLUTE.

MODPATH is a particle tracking post-processing package for MODFLOW. It uses information on aquifer potentials estimated by MODFLOW, and estimates of aquifer porosity to estimate flow paths and compute groundwater velocities or travel times in groundwater systems (USGS, 1997; Kumar, 2006). MODPATH is capable of simulating steady state and transient flow and can be applied in a number of studies. These include studies on paths and time of travel of contaminant movement, studies on recharge areas for hydrologic features such such as springs, rivers, streams, and wells, and studies on capture zone analysis (USGS, 1997).

MOC is an acronym for the Method of Characteristics Model. It is a two-dimensional solute transport model developed by Konikow and Bredehoeft (1978). The model is based on the finite difference approximation of the groundwater flow equation and method of characteristics to solve

the solute transport equation. MOC can be executed for one or two-dimensional problems involving steady state or transient flow. The model utilises a particle tracking procedure to represent advective transport and an explicit finite difference method to compute concentration changes due to hydrodynamic dispersion (Konikow & Reilly, 1999).

MOC is executed on the assumption that temperature, viscosity and gradients of fluid density do not affect velocity distribution. The model computes temporal changes in solute concentration occasioned by processes of advective transport, hydrodynamic dispersion, and mixing or dilution from other fluid sources. It computes temporal changes in first-order irreversible rate reactions such as radioactive decay, reversible equilibrium-controlled sorption with linear Freuidlich, or Langmuir isotherms, and reversible equilibrium controlled ion exchange for monovalent or divalent ions (Konikow & Reilly, 1999; Kumar, 2006). The model can also "be used to simulate groundwater age transport and the effects of double porosity and zero-order growth/ loss" (Todd & Mays, 2005, p.456).

FLOWPATH is a two-dimensional groundwater flow and contaminant transport model that simulates flow in confined, unconfined and leaky aquifers with heterogeneous characteristics, multiple pumping wells and complex boundary conditions. The model is mainly used for determining the capture zones of remediation wells, delineating well head protection zones, determining contaminant fate and exposure pathways for risk assessment, and designing and optimisation of pumping well locations for dewatering projects (Kumar, 2006).

VISUAL MODFLOW is a three dimensional groundwater flow and contaminant transport model. It simulates groundwater flow in three dimensions using finite difference method. The model is a combination of models, namely MODFLOW, MT3DMS and RT3D (Kumar, 2006; Boyraz &

Kazezilmaz-Alhan, 2012). MT3DMS (Modular 3-D-Transport Model Multiple Species) is a new version of MT3D (Modular 3-D-Transport Model). It is a modular three dimensional finite difference groundwater flow model. MT3DMS utilises the same modular structure of the original MT3D, and the modular structure of the model makes it possible to simulate processes like advection, dispersion/diffusion, source/sink mixing, and chemical reactions separately without reserving computer memory space for unused options. The MT3DMS can also be used in simulating different types of boundary conditions, and external sources and sinks (Zheng & Wang, 1999).

RT3D (Reactive Transport in 3 Dimensions) is a computer code developed to solve "the coupled partial differential equations that describe reactive-flow and transport of multiple and/or immobile species in three dimensional saturated groundwater systems" (Clement, 1997, P.6). The RT3D code was originally developed for the purpose of contaminant transport modelling at natural attenuation demonstration sites. It can be used to simulate different types of reactive contaminants such as BTEX (Benezene-Toulene-Xylene), and Chlorinated solvents such as tetrachloroethene (PCE) and Trichloroethene (TCES) (USDE, 1997).

POLLUTE is a finite layer contaminant transport model (Franz & Rowe, 1993). "It can be used for fast, accurate, and comprehensive contaminant migration analysis. It implements a $1^{1/2}$ dimensional solution to the advection-dispersion equation" (Kumar, 2006, p.17). The finite layer technique of contaminant simulation in POLLUTE is applicable to landfills whose hydrostratigraphy can be idealised as being horizontally layered. The model simulates contaminant transport for different categories of landfills, ranging from landfills underlain by natural clays, to those underlain by composite and multiple barriers, as well as those underlain by multiple aquifers. Apart from advective-dispersive transport, POLLUTE can simulate

contaminant transport through fractures and other processes such as biodegradation, adsorption and radioactive decay. (Franz & Rowe; 1993; Kumar, 2006).

The finite layer technique of the model has several advantages, such as easy usage and minimal data input requirement. Because of the semi-analytical nature of the technique used in POLLUTE, contaminant concentrations and total mass flux into the subsurface can be accurately computed at any specified times and locations of interest without determining the entire solution field (Franz & Rowe, 1993).

Due to its wide acceptance and numerous advantages, a number of studies exist in which MODFLOW or MODFLOW in combination with other models have been used. Some case studies in which MODFLOW, MODFLOW in combination with other models, and other case studies involving the use of other models for the simulating groundwater flow and leachate transport in and around municipal waste disposal sites in different parts of the world are reviewed below.

Kim, et al., (1999) modelled the hydraulic head and the pollutant concentration distribution around the Gyunggi-do landfill in Korea using MODFLOW and MT3D respectively. The verification of the model involved the comparison of the simulated hydraulic heads with measured heads from 78 observation holes installed around the landfill. The result of the pollutant concentration distribution was verified by comparison with measured tritium concentrations from 6 sampling points. Tritium was used to compare the simulated pollutant concentration with observed concentration in groundwater. The comparison of the hydraulic heads yielded an error margin of less than 1m while the pollutant concentration distribution ratio revealed a difference 0.02. The results of the study showed that observed tritium concentration

was higher than the simulated concentration. The comparison of the observed and simulated concentration showed that at two points where the original tritium concentration ratio was 0.15, the simulated concentration ratio at the same points was 0.13.

Fatta, et al., (2000) simulated the groundwater flow and solute transport characteristics of Ano Liosia Landfill Site in Athens, Greece, under 2-dimensional and 3- dimensional flow conditions. The results of the groundwater modelling showed that flow direction was downstream towards the northwestern part of the model domain, with hydraulic heads fluctuating between 7 and 9 metres within the landfill site. Hydraulic heads around the landfill varied between 0 and 10 metres. The results of the solute transport simulation of the landfill under a 30-year period of leachate input into the underlying aquifer showed a leachate plume migration extent of 1,843m and 92m, at a mean concentration velocity of 6.5×10^{-2}m/day.

Dong, et.al., (2008) developed a groundwater flow and leachate transport model to assess the impact of leachate generation on groundwater quality at the municipal landfill of Jianxing and environs, in southern China. The results of the groundwater flow modelling indicated that the migration of leachate was mainly to the North and the East of the landfill. This is in correlation with the locations of the surrounding rivers. The results of the solute transport modelling for the Chloride concentration of leachate for a sixteen year period (1996-2012) suggested that by 2005 the leachate would have migrated 300m downgradient of the landfill, while by 2008 and 2012, chloride concentration in the well located 300m downgradient of the landfill would have reached 8 and 20mg/L respectively.

Nwachukwu, et.al., (2010) utilised transient MODFLOW and MODPATH to model the flow regime of groundwater and the migration of contaminants from 4 contaminated sites, among

which is an abandoned solid waste dumpsite in Obinze, Owerri, Nigeria. The results of the simulated 3-dimensional groundwater flow direction at the dumpsite was compared with the result of a previous study by Ibe and Njoku (1999), in which a non-numeric method was used to determine groundwater flow directions at the dumpsite. The results of the 3-dimensional groundwater flow model displayed a southward flow for a distance of 5km before flow changed direction to the south-east, while the results of the non-numeric method by Ibe and Njoku observed flow to be in three directions, North-East, North-South and East-West. The change in flow direction was attributed to the presence of anticline structures and strike-slip faults in the area.

Shashidhar and Ajit- Kumar (2011) simulated the migration of leachate generated at the Warangal Municipal Waste Dumpsite in Urusgutta, India. The dumpsite is a former quarry site, and waste disposal at the site is carried out by the Warangal Municipal Corporation. Open dumping is practised at the dumpsite, and the daily volume of waste received at the dumpsite is estimated at an average of 300 tons. A solute transport model was used to predict the migration of chloride, which is a conservative contaminant from the dumpsite, for the years, 2009, 2013 and 2018. The results of the solute transport model revealed a 200m spread around the dumpsite, with varying levels of chloride concentration.

For the Seri Petaling landfill in Selangor, Malaysia, Saghravani and Mustapha (2011) utilised VISUAL MODFLOW to predict the surface and sub-surface migration of the phosphorous constituent of the leachate generated at the landfill under a steady state flow condition for a period of 10 years. The results showed that phosphorus had migrated into the groundwater and downgradient surface water, as well as other locations. Furthermore the comparison of the 2.38mg/L of the phosphorous concentration obtained from the landfill with the Malaysian

National Standard of 0.1mg/L showed that the unconfined aquifer underlying the landfill was highly contaminated with phosphorous.

A number of case studies exist in literature, in which other groundwater modelling softwares have been utilised in simulating groundwater flow and solute transport in and around municipal waste disposal sites. Some of the studies involving the use of other models include the studies of Franz and Rowe (1993), in which FLOWPATH, USGS-MOC and POLLUTE software packages were used to simulate groundwater flow and contaminant transport at Innisfill landfill site in Ontario, Canada.

The Insisfill landfill is an unlined landfill site. Prior the waste disposal, the site served as a sand and gravel extraction pit. The regional geology of the landfill was characterised as belonging to the predominantly sandy till drumlins. The upper 40 to 50 meters of the lithostratigraphic units of the dumpsite which is made up sand and clay was characterised as glaciofluvial water lain deposit. The upper unit was further classified into three; the upper sand unit, upper silt/clay unit, and intermediate sand unit.

The purpose of the study was to simulate groundwater flow and contaminant transport within the landfill. Groundwater flow modelling was carried in order to simulate the existing groundwater flow in the upper aquifer and to develop an understanding of the interaction between different parameters that influence groundwater flow conditions. The contaminant transport modelling involved the simulation of lateral contaminant transport within the upper aquifer at the landfill site. For the simulation of the regional groundwater flow, FLOWPATH was utilised, while a combination of USGS-MOC and FLOWPATH was adopted in simulating lateral contaminant

migration within the upper aquifer unit. To investigate the vertical transportation of contaminants below the landfill, the POLLUTE software package was used.

The results of the regional groundwater flow modelling indicated that the hitherto assumed leachate mound within the landfill which was based on the high water level as suggested by the field results, was actually a perched condition brought on by the recirculation of leachate through the recharge pits constructed at the landfill when a shallow leachate collection system was installed at the landfill a few years earlier. For the results of the lateral contaminant transport modelling, the longitudinal and transverse dispersivity obtained from the MOC model fitted relatively well with the observed Chloride plume at the landfill. For the vertical contaminant transport modelling of the landfill site, the results of the vertical transport modelling was compared with the observed depth of leachate percolation. The results of the model corroborated the field results which revealed that contaminants like Benzene, Toluene, Phenols and Chloride had migrated to some extent within the silt/clay hydrostratigraphic unit, but is yet to impact the intermediate sand unit beneath the landfill.

Bougiokou, et al., (2005) developed a groundwater flow and contaminant mass transport model for the municipal landfill of Patras city down to Patraikos Gulf in Greece using the Princeton Transport Code. Contaminant plume from the landfill was simulated on a seasonal basis (wet and dry seasons) for a 5 year period to determine the impact that the leachate from the landfill would have on groundwater quality after the simulation period of 5 years. The aquifers used for the study are the Xerolaka aquifer and the Sihaina-Terpsithea aquifer.

The Xerolaka and the Sihaina-Terpsithea aquifers are located in the western part of Greece. The Xerolaka aquifer spans the Northeast end of the municipal landfill of Patras to the confluence of

Xerolaka Stream with Voudani stream. The aquifer depth is between 8 to 10 metres, with a vadose zone of 0.3 to 0.5 metres. The Sihaina-Terpsithea aquifer is located along the Gulf of Patraikos. It is located in the alluvial deposits formed at the confluence of Xerolaka Stream with Voudani stream. The aquifer depth ranges between 10 to 15 metres at Sihaina and more than 100 near the coast, while the depth of the vadose zone ranges between 5 metres at the Sihaina area and 15 to 20 meters near the sea.

The results the study revealed that in case the liner at the landfill ruptures or fails, contaminant plume from the landfill will migrate to locations of municipal drinking water wells in the Sihaina-Terpsithea aquifer in less than a year. Furthermore, the results of the of the simulation indicated seasonally induced fluctuations in the hydraulic head fields, and a continuous groundwater flow from the landfill to the sea due to hydraulic connection between the Xerolaka aquifer and the Sihaina-Terpsithea aquifer.

Tuahid-Ur-Raham (2009) utilised Femlab, a multi-physics based numerical modelling package to simulate the transport of contaminants from waste disposal sites in the Masindi district of Uganda. The results of the solute transport model indicated that contaminants from the waste disposal sites will infiltrate the vadose zone after an hour of being released from the waste disposal sites. The results also indicated that after a 40 hour period the contaminants will get into the phreatic zone, and by 183 hours after its release from the waste disposal sites the contaminants will get to the exit boundary of the model domain.

2.1.2 Waste Degradation Processes Operating in Landfills and Dumpsites

Five different processes have been identified as operating within a landfill or a dumpsite. Williams (1998) described the processes as hydrolysis/aerobic degradation, hydrolysis and

fermentation, acetogenesis, methanogenesis and oxidation. Ritzkowski, et al., (n.d) described the processes as aerobic processes, acid fermentation processes, intermediate anaerobic processes, methanogenic processes and second aerobic processes.

Kjelsen et al., (2002) were of the opinion that decomposition progressed in landfills and dumpsites in four phases, with the fifth phase or process being hypothesized as an aerobic ecosystem and described as an additional or humic phase of decomposition. These phases were identified as an initial aerobic phase, an anaerobic phase, an initial methanogenic phase and a stable methanogenic phase. These same processes were described by Allen and Taylor (2006) as aerobic decomposition, acetogenic fermentation, intermediate anaerobiosis and methanogenic fermentation. Figures 2.1 and 2.2 represents the various processes or stages of landfill or dumpsite waste degradation and their degradation products.

Hydrolysis/ Aerobic Degradation

Williams (1998) described the hydrolysis/aerobic degradation stage as occurring during the deposition of waste and for a period spanning a few days or weeks, depending on the availability of oxygen within the waste matrix. According to McBean et al., and Waste Management Paper 26b as cited in Williams (1998), the available oxygen and a fraction of the organic matter is metabolised to produce simple hydrocarbons, carbon dioxide, water and heat. The depletion of oxygen at the aerobic phase of degradation is attributed to the inhibition of air circulation occasioned by the compaction of waste at the dumpsite or landfill.

Other factors that may be responsible for the inhibition of air circulation within the waste mass include the volume of air trapped within the waste, the extent of waste compaction and how

quickly the waste is covered after deposition (Taylor and Allen, 2006). At this stage of waste degradation, no significant amount of leachate is generated (Ritzkowski et al., n.d).

Hydrolysis and Fermentation

Waste management Paper 26 and Waste Management Paper 26B, as cited in Williams (1998) describe the hydrolysis and fermentation stage as involving a change from the aerobic to the anaerobic phase of waste degradation. This phase is facilitated mostly by facultative anaerobes which are able to operate in the ambient oxygen deficient environment of the waste matrix. The carbohydrates, proteins and lipids contained in the waste are hydrolysed to sugars before further decomposition results in carbon dioxide, hydrogen, ammonia, and organic acids (acetic, formic, butyric, propionic and lactic acids). The decomposition of the proteins through deaminisation results in ammonia, carboxylic acids and carbon dioxide, while the resultant leachate contains high concentration of ammoniacal nitrogen.

Acetogenesis

Acetogenesis is described by Christensen et al., as cited in Williams (1998), as the process in which the organic acids formed during the hydrolysis and fermentation stage of waste degradation is converted to acetic acid and its derivatives, carbon dioxide and hydrogen by acetogen microbes. Furthermore, the hydrogen levels occasioned by the conversion of carbohydrates to acetic acid, encourages the production of methanogens which are responsible for the production of methane and carbon dioxide from the organic acids.

In addition, leaching and solubilisation of metals is facilitated at this stage due to the prevalent acidic conditions and the formation of complexes with metals by organic acids, phosphate, chloride, and ammonium ions. The characteristic high concentration of metal ions in the leachate

generated at this stage is attributed to the acidic conditions prevalent at this stage of waste degradation (Williams, 1998).

Methanogenesis

"The methanogenesis stage is the main landfill gas generation stage, with the gas composition of typical landfill gas approximately 60% methane and 40% carbon dioxide" (Williams, 1998, p.201). "This phase is characterised by methanogenic fermentation elicited by methanogenic bacteria" (Ritzkowski, et al., n.d, p.12). Williams (1998) classified the methanogenic bacteria into two groups; the mesophilic bacteria operative within a temperature range of 30 and 35^0C, and the thermophilic bacteria which is active between 45 and 65^0C.

As explained by Williams (1998), methanogenesis is the longest stage of waste degradation, commencing between 6 months and several years after waste emplacement, and subject to the amount and circulation of water within the waste. At this stage, significant volume of methane gas is generated between 3 and 12 months depending on the production of anaerobes and the derivative products. Depending on the type of waste deposited and the conditions at the landfill or dumpsite, the emission of landfill gas will continue for a period spanning between 15 and 30 years. As further explained by Williams (1998), anaerobic conditions tend to develop at landfill sites with depths exceeding 5 metres and this results in the production of higher volume of landfill gas. At shallower sites lesser volume of landfill gas is produced due to higher level of air exchange and lower anaerobic activities.

The leachate produced at this stage of waste degradation is characteristically low in Total Dissolved Solids and volatile acids, with an almost neutral pH, which is an indication of the almost complete solubilisation of most of the organic constituents of the leachate. This is

reflected in the low ratios of the Biological Oxygen Demand (BOD) to Chemical Oxygen Demand (Taylor and Allen, 2006).

Oxidation

The oxidation/second aerobic/humic phase of decomposition, in the opinion of Kjelsen et al., (2002) is basically speculative and theoretical, as they maintained that they are unfamiliar with any landfill that has progressed beyond the methanogenic phase of waste degradation, although it is being theorised that the landfill or dumpsite will eventually become aerobic at the cessation of waste degradation at the site. The counter opinion of Williams (1998) on this process, however described the oxidation process as the final stage of waste degradation, in which new aerobes gradually replace the anaerobes that have been consumed in the production of landfill gas. According to Williams (1998), aerobic conditions are re-established in the landfill through the conversion of the residual methane to carbon dioxide

and water by newly produced aerobes.

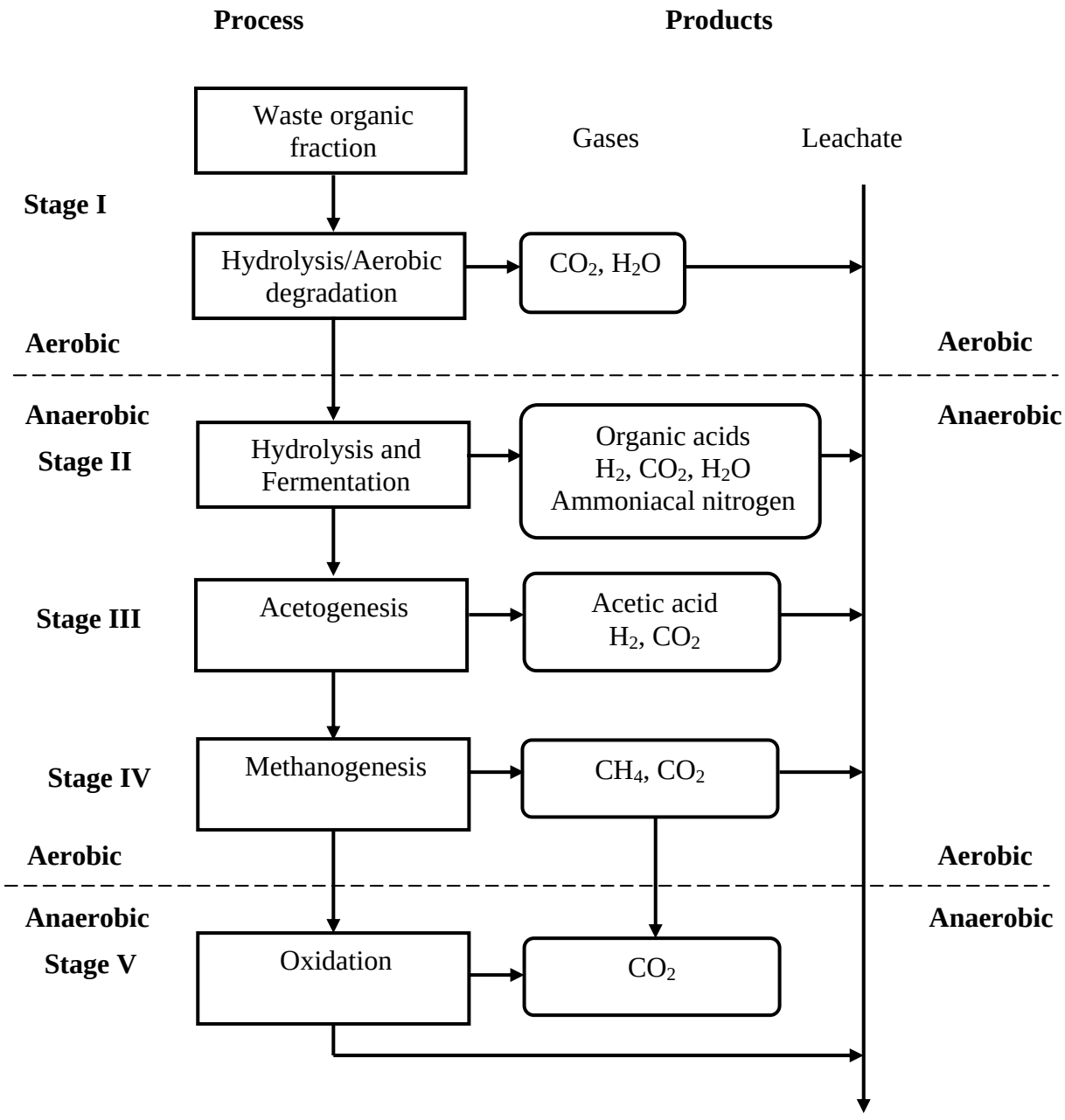


Fig 2.1 Major Stages of Waste Degradation in Landfills and Dumpsites (Modified after Williams, 1998)

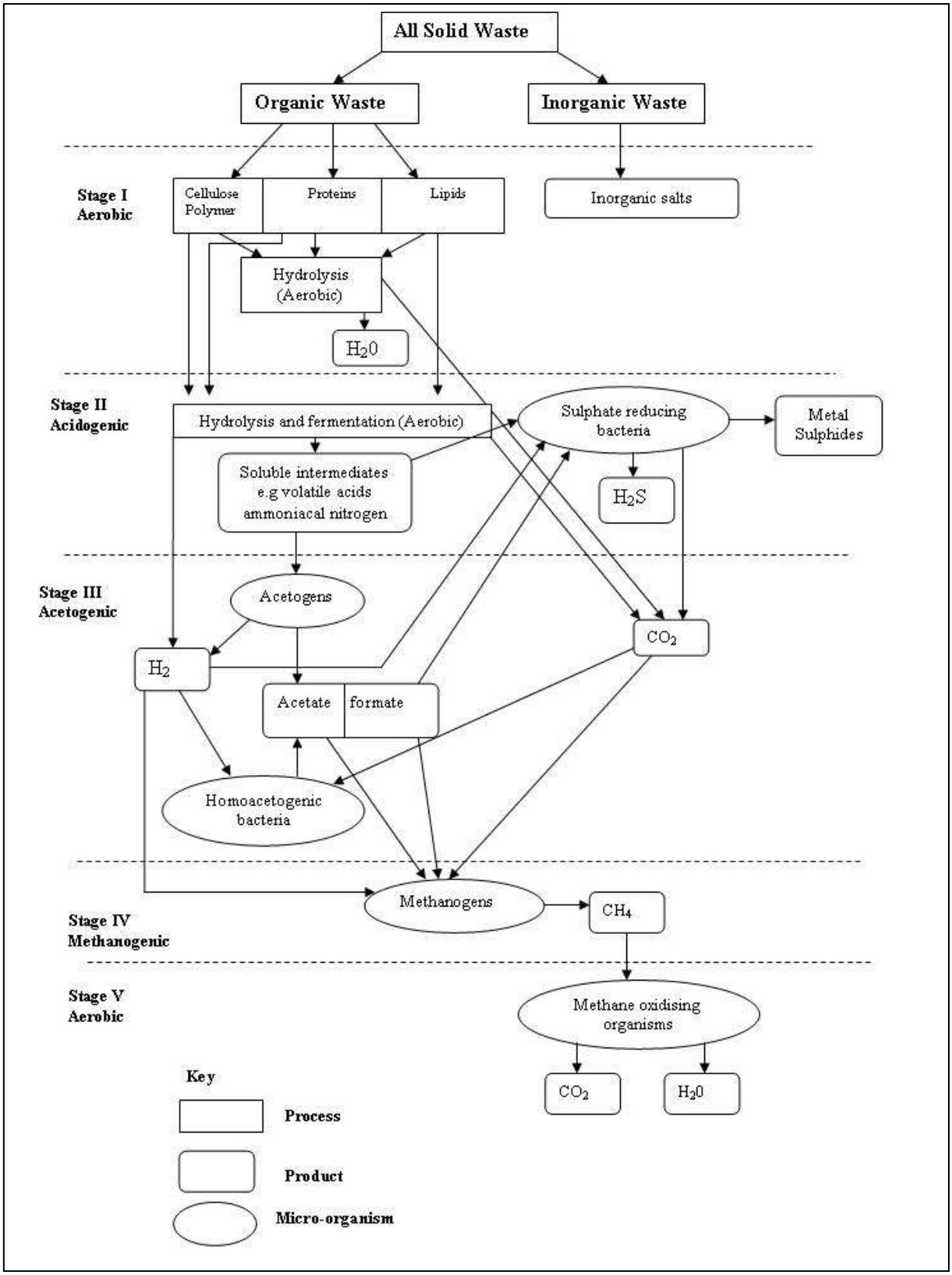


Fig 2.2 Details of Stages involved in the Degradation of Solid Waste (Modified after Williams, 1998)

2.1.3 Landfill Leachate Characterisation and Contamination Potential

Due to the potential of leachate generated at various waste disposal facilities to contaminate soil, surface water and groundwater, a number of studies have been conducted to investigate the composition of landfill leachate.

Presenting a review on the present and long term composition of municipal solid waste (MSW) landfill leachate, Kjelsen, et al., (2002) categorised the contaminants of such landfill leachate into four groups namely, Dissolved Organic Matter such as Chemical Oxygen Demand (COD), Total Organic Carbon (TOC) and Volatile Fatty Acids (VFA), Inorganic Macro-Components such as Chloride, Iron and Sulphate, Heavy Metals such as Cd^{2+}, Zn^{2+} and Ni^{2+} and Xenobiotic Organic Compounds (XOCs) such as Hydrocarbons, Phenols and Pesticides.

The composition of leachate is subject to several factors which include seasonal variation, landfill age, waste composition, geological characteristics and landfilling techniques. The combination of these factors has the potential to influence the quality of leachate generated invariably having an integrated effect on its composition (Asian Institute of Technology & Tongji University, 2004).

Robinson (2007) conducted a review on the quality of leachate generated at large landfills sites in different climates in different parts of the world, and noted that the transition from acetogenic to methanogenic characteristics in landfill leachate at landfills in temperate climates usually takes up to three years. He further noted that the same transition took between twelve and eighteen months in warmer climates, although the precise prediction of the transition from acetogenic phase leachate to methanogenic leachate is difficult due to short term weather patterns. From the examination of data collected for the review, Robinson (2007) observed that although there are great similarities in the quality of leachate generated at the

investigated landfills, methanogenic leachates were very consistent and similar in composition regardless of the climatic conditions prevailing where the landfills were located.

Leachate generated at landfills in the same climate, and also leachate generated within the same landfill could also vary as a result of waste age or depth of burial. The Biological Oxygen Demand /Chemical Oxygen Demand (BOD/COD) ratio of leachate serves as an indicator of waste and leachate age. This is because the BOD/COD ratio of leachate tends to increase with increasing landfill age (Zou-Lianhua as cited in Asian Regional Research Programme [ARRPET] n.d). According to ARRPET (n.d, p.8) "decrease in this ratio can be attributed to the presence of some oxidised organic carbon, which becomes less readily available energy source for microbial growth". The maximum possible BOD/COD ratio in the young leachates or leachate generated at young landfills is about 0.6, while for old leachates and leachates generated at old landfills, the ratio could be as low as 0.04 (Zou-Lianhua as cited in ARRPET n.d). Other indicators of leachate age also include the COD/TOC ratio,Volatile Solids/Fixed Solids(VS/FS) ratio, and VFA/TOC ratio. VS/FS ratio decreases with age of a landfill due to the biodegradable nature of organic compounds which hastens its decrease with the increasing age of leachate generated.

According to the Asian Institute of Technology and Tongji University (2004), fresh leachate is characterised by high organic matter content, low pH, higher VS/FS ratio and an unpleasant smell, while a decline in the VS/FS ratio is indicative of stabilised mature leachate. The mature or old leachate is characterised by increased pH and redox potential and a change in colour from light yellow to dark brown or black due to oxidation from ferrous (Fe^{2+}) to the ferric state (Fe^{3+}).

Lee, et al., (2010) monitored the influence of waste age on the concentration and quality of the leachate generated at a sanitary landfill site in Toronto, Canada, over a six-year period. Utilising

data on young and old leachate (leachate samples not exceeding five years and fifteen years respectively), the results of the study revealed a decreasing trend in the concentration of Ca^{2+}, Cl^{-}, Fe^{2+}, Mg^{2+}, SO_4^{2-}, and Phenol (C_6H_5OH) over time.

The result of the leachate monitoring programme to determine the seasonal variations in the leachate quality from dumpsites in India and Sri-Lanka by Essaku et al (2007), showed that the concentration of contaminants were higher in the leachate generated at Indian dumpsites during the summer. The study showed that leachate generated at the Indian dumpsites get diluted during the monsoon and the contaminants concentration gradually increases after the monsoon due to increased leaching of contaminants from the newly emplaced waste dumped at the top layers during the initial precipitation. At the Sri-Lankan dumpsites it was observed that seasonal variation in leachate quality was higher between dumpsites than variations in leachate samples collected on different days. Furthermore, differences were observed in the COD levels of leachate collected from the dumpsites. This was attributed to variation in the biodegradable fraction of the waste matrix at the point of leachate sample collection.

From the study carried out by the United States Environmental Protection Agency (USEPA) at five municipal landfill sites in Washington, Pennsylvania, Indiana, Tennessee and California, James (1997) reported low levels of Cu^{2+} in the landfill leachates at concentrations below the USEPA standards. It was also shown in the study that Zn^{2+} concentration in the leachate generated declined with age of the landfill, and were also below the USEPA standard. Further revealed by the study is the variability in the concentration of Cd^{2+} and Cr^{3+}, and the high concentrations of Pb^{2+}, Se, Fe^{2+}, Hg^{2+} at concentrations above the USEPA standards, irrespective of the landfill sites.

Adewuyi and Opasina (2010) assessed the physico-chemical and heavy metals constituents of leachate generated at an abandoned dumpsite in Ibadan. Their result showed that the leachate generated at the dumpsite was alkaline and other physico-chemical and heavy metals parameters of the leachate exceeded the WHO guideline and FEPA standards for drinking water quality. Similarly, Ogundiran and Afolabi (2008) investigated the toxicity of heavy metals in the leachate generated at Olusosun dumpsite in Lagos. The results of the study indicated that Zn^{2+} and Cd^{2+} had the highest and the lowest concentrations respectively. The results also showed that the concentration of Pb^{2+}, Cd^{2+} and Zn^{2+} in the leachate collected from active part of the dumpsite was higher than those collected from the dormant and abandoned parts.

The recognition of the contamination potential of leachate has also resulted in the development of a leachate assessment and pollution quantifying tool known as Leachate Pollution Index (LPI). The LPI developed by Kumar and Alappat (2003), is a Delphi technique developed for the purpose of comparing the leachate pollution potential of leachate generated at different landfills within a particular geographical area. For the derivation of the LPI, eighteen of the most commonly reported and significant leachate parameters were selected, and each apportioned a weight based on the significance of its contribution to overall leachate pollution.

The next stage was the establishment of a relationship between leachate pollution and concentration of each parameter. This was achieved with the aid of the averaged sub-index curve developed for the estimation of the LPI. The final stage was the aggregation of the sub-index. For leachate data that is comprised of the 18 parameters, the LPI aggregation was based on the summation of the sub-index value of the i^{th} leachate variable (p_i) multiplied by the respective weight of each parameter. For leachate data with less than 18 parameters, the sum of the sub-index value of the i^{th} leachate variable (p_i) is multiplied by the weight of each parameter and

divided by the sum of the weights of the parameters to derive the LPI value. The overall contamination potential expressed by the LPI is based on a single number ranging from 5-100, with a higher value indicative of poor environmental conditions (Kumar & Alappat, 2003; Kumar & Alappat, 2005). The LPI has since been utilised to determine the contamination potential of leachate generated at different landfills.

Kumar and Alappat (2003) computed a significantly high LPI value for the leachate generated at Okhla landfill in New Delhi, India, and concluded that the leachate generated at the landfills should be treated before being discharged into the environment. Utilising the LPI to evaluate the leachate contamination potential of the leachate generated at two closed and two active landfills in Hong Kong, Kumar and Alappat (2006), reported that the passage of time, particularly the time of closure of a landfill did not necessarily reduce the potential of the leachate to contaminate groundwater.

The computation of seasonal variability of the contamination potential of leachate from Pune city landfill in India by Kale, Kadam, Kumar and Pawar (2010) showed that the leachate generated during the pre-monsoon period had greater potential for contamination than the leachate generated during the post-monsoon period. This may be due to the dilution of leachate by rainfall during the monsoon period.

2.1.4 Leachate Transport through a Landfill or Dumpsite

In describing the migration of leachate from a landfill or dumpsite, USEPA (1985) stated that:

> The many complex factors that control the movement of leachate and the overall behaviour of contaminant plumes are difficult to assess because the final effect represents several factors integrated collectively (p.27).

Taylor and Allen (2006) however stated that inspite of complexity associated with leachate transport through landfills and dumpsites alike, the basic principles of subsurface contaminant transport can be used in explaining the movement of leachate-derived contaminants.

The migration of leachate as described by USEPA (1985) begins with the slow migration of leachate through the unsaturated zone, where it is subjected to various physical (e.g filtration) chemical (e.g ion exchange) and biological (e.g biodegradation) processes. Once it migrates into the saturated zone, flow essentially becomes lateral as defined by the hydraulic gradient. Leachate also becomes diluted due to a number reason which includes filtration of suspended particles, sorption, various chemical processes, microbial degradation, dispersion, time and distance of travel.

Other factors of contaminant transport identified by Taylor and Allen (2006) as being applicable to the movement of leachates within the subsurface include the thickness of the unsaturated zone, permeability and moisture content of the earth materials within the unsaturated zone, hydraulic conductivity and the local hydraulic gradient in the saturated zone.

Furthermore, Taylor and Allen (2006) also identified the manner of waste disposal as one of the factors that influences the migration of leachate from a landfill or dumpsite. Thus, preferential flow paths may develop within a landfill or dumpsite depending on the manner of waste disposal. If waste is compacted prior to disposal permeability of waste is reduced, and if topsoil is regularly applied, layering is induced. According to Taylor and Allen (2006) reduced permeability occasioned by waste compaction could lead to preferential flow paths.

According to USEPA (1985), due to the varying macroscopic velocities and flow path lengths within a porous media such as an aquifer, leachate movement at a higher velocity or shorter flow

path would result in an earlier arrival at an end point than a leachate plume moving at a lower velocity or through a longer path, which culminates in hydrodynamic dispersion.

USEPA (1985) also noted that since dispersion which can both longitudinal and transverse, results in a cone shape downstream of a continuous contamination source, the concentration of leachate is lesser at the margins of the cone and increases towards the source. It was also noted that a plume such as that of leachate increases in size with higher flow rate within the same period due to direct relationship between dispersion and groundwater velocity.

Despite of the similarities between the flow of leachate and groundwater however, a number of differences also exist. Bjerg, et al., (2003) stated that the flow of leachate differed from the flow of groundwater in three respects. These are in relation to the water table gradients, the viscosity of the leachate and the density of the leachate.

With regards to the local water table gradient, Bjerg et al. (2003) explained that the local water table gradients below and around a landfill will most likely be

different from the general water table gradient because the hydrology and hydrogeology of landfills in most cases differ from that of its locality. This difference in hydrology and hydrogeology is sometimes manifested in the form of water table mounds at the landfills. According to Kjelsen et al., (1998a) water table mounding could arise due to higher infiltration in the landfill area as a result of inadequate or poor top cover or different vegetation, lower hydraulic conductivity beneath the landfill, and higher infiltration at the borders of the mounding due to inflow in the waste or surface runoff from sloping cover. The conclusion from the studies conducted by Kjelsen et al. (1998a) over a 9 month period at the Grindsted Landfill, and the studies conducted at the same landfill by Kjelsen et al. (1998b) over a $2^1/2$ year period, was that

although the reason for groundwater mounding are not fully understood, the three reasons highlighted above were held responsible for this phenomenon.

As explained by Christensen et al., (2000), the effects of water table mounding at a landfill include enhanced lateral spreading of leachate, and downward directed hydraulic gradients in the groundwater zone beneath the landfill. The latter is explained as being responsible for unexpected spreading pattern of leachate plume inspite of homogeneity of aquifer conditions, and the limited density difference between leachate and groundwater.

Christensen et al., (2000) further explained that although the enhanced spreading of leachate plume may increase the volume of contaminated groundwater, it also provides opportunities for increased dilution of contaminants present in the leachate. Christensen et al. (2000) and Bjerg et al., (2003) identified some studies in which local water table mounds have been observed at landfills, to include studies by MarFarlene et al. (1993) at the Bordern Landfill in Canada, studies of Kjelsen (1993) at the Vejen Landfill in Denmark, studies of Van Duijvenbooden and Kooper (1981) at the Noordwijk Landfill in Netherlands, and the studies of Kjelsen et al., (1998b) at the Grindsted Landfill site in Denmark. Kjelsen et al., (1998a) also observed water table mounding at the Grindsted Landfill during the period of investigation.

In respect of the viscosity of leachate differentiating its flow from the flow of groundwater, Christensen, et al., (2000) explained that the viscosity of leachate may differ from the viscosity of groundwater based on the theory that higher viscosity lowers flow velocities, and as such the reduced velocity of leachate could allow for dilution. Christensen et al. (2000) reported that the effect of leachate viscosity from 13 Danish Landfills was marginal, with an increase of 1 to 15 percent when compared with water at the same temperature.

The explanation of Bjerg et al., (2003) on the difference in density between leachate and groundwater was that the difference between both may affect the vertical positioning of leachate plume beneath the landfill. Bjerg, et al., further explained that vertical spreading of leachate in aquifers may be due to the effects of density since normal vertical dispersion is usually limited.

2.1.5 Approaches used in the Assessment of Leachate Migration from Waste Disposal Sites

Over the years, a number of techniques have been developed to detect and study the migration of leachate from the waste matrix into the underlying aquifers and groundwater resources. Kjelsen et al., (1995) assessed the spatial variability in the migration of leachate from Grindsted landfill in Denmark utilising historical information (old maps and aerial photographs), groundwater sampling analysis and hydrological investigations (installation of piezometers, wells and leachate wells). The results of the study showed that significant spatial variation existed in the concentration of leachate both within and at the downstream borders of the landfill, with approximately two-thirds of the landfill area having low concentrations of almost all of the constituents of leachate.

Ehirim, et al., (2009) investigated the migration of leachate from Choba landfill in Port-Harcourt by integrating a two-dimensional resistivity survey with microbial and physico-chemical analysis of groundwater. The study revealed a vertical migration of leachate in excess of 31.3m below the aquifers, and a lateral migration of over 40m south of the landfill site. The results of the study also showed that the excessive amount of micro-organisms in the groundwater from the boreholes installed between 18 and 37m from the landfill was due to leachate contamination.

Similarly, Enikanselu (2008) used the electrical resistivity method to analyse the extent of leachate migration from Giwa-Okearo Dumpsite, in Ifo Local Government Area of Ogun State between 1995 and 2005. The results showed that the area of polluted groundwater had increased from 4,275m^2 in 1995 to 10,800m^2 in 2005.

The use of environmental isotopes such as Deuterium (^2H), common Hydrogen (^1H), Carbon 14 (^{14}C), Oxygen 18 (^{18}O) and Titrium (^3H) has gained prominence not only as a guide to groundwater dating and indicators of groundwater source areas, they have also come under increasing use in groundwater pollution and contamination investigations. Syafalni, et al., (2010) used the environmental isotope technique to investigate the migration of leachate from a waste disposal site at Bekasi city, near Jarkata in Indonesia. Utilising a combination of Carbon 14 (^{14}C) and Iodine 131 (^{131}I), with the hydrochemical analysis of groundwater, the study revealed the migration of leachate into the shallow groundwater in areas adjacent to the waste disposal site. The hydrochemical analysis also showed an elevated level of nitrate in the groundwater.

2.1.6 Impact of Landfill Leachate on Groundwater Quality

The contamination of groundwater due to the leachate generated at landfills and dumpsites has been a focus of many studies in different parts of the world for several decades. In a study to assess the deterioration of groundwater quality in the vicinity of Ampar Tenang Dumpsite, in West Malaysia, Bahaa-eldin, et al., (2009) reported that the leachate generated at the dumpsite had contaminated the groundwater underlying the dumpsite, and downgradient of the dumpsite. The results of the study showed elevated levels of Cl^-, NO_3^-, NO_2^-, NH_3-N, Na, Fe^{2+}, Cu^{2+}, Pb^{2+} and Cd^{2+} at concentrations that exceeded the WHO and FAO standards for drinking and irrigation water quality.

Niinien and Kalliokoski (1994) investigated the effects of organic contaminants of landfill leachates on groundwater quality in the vicinities of four landfill sites in Finland, and reported evidences of organic contamination of groundwater in the vicinities of the landfills. The results of the study revealed high levels of COD and low levels of BOD. The high levels of COD and BOD in groundwater was attributed to contamination by organic compounds contained in the waste deposited at the landfills. The study also revealed that while the extent of leachate plume migration remained minimal due to the prevailing hydrogeological conditions and small sizes of the aquifers in Finland, the groundwater was toxic and mutagenic to water fleas. Soil types at the landfills include, sand grid, marsh peat, peat, silt and clay.

In a study conducted at the Novi-Sad Landfill in Serbia, Dvornic, et al., (2011) reported that the values of BOD and COD in the leachate contaminated groundwater at the landfill was lower at the onset of summer due to the dilution of the groundwater by melting snow and rain.

At the Vejen Landfill site in Denmark, the study conducted by Christensen, et al., (1998) showed that the groundwater contaminated by leachate at the landfill was characterised by an almost neutral pH, high levels of DOC, and high concentrations of Ca^{2+}, Na and $NH4^{-}$. Douglas and Borden (1992) assessed the impact of sanitary landfill on surface and groundwater quality in Wake County, North Carolina. Their study determined a broad spectrum of organic and inorganic contaminants, and it was shown that groundwater immediately downgradient of the landfill had been contaminated with variety of contaminants. These include Fe^{2+}, Mn^{2+}, TOC (Total Organic Carbon) and SOCs (Synthetic Organic Chemicals) which were detected at high concentrations. The study also further revealed that the concentration of heavy metals in groundwater samples collected within the landfill was low. Average water table interval of the monitoring wells varied

between 176.65 and 196.60 feet at upgradient locations, and between 179.27 and 180.50 downgradient of the landfill.

Amadi, et al., (2010) investigated the impact of two major dumpsites within the Markudi Metropolis on groundwater quality. Groundwater samples were collected both in the vicinities of the dumpsites and 30 metres from the dumpsites. The results of the study showed that groundwater quality near the dumpsites had been degraded to a certain extent, with low pH and high concentrations of Fe^{2+}, Mn^{2+}, Ca^{2+} and coliform, while the quality of groundwater farther away from the dumpsites were within National and International standards for drinking water quality.

In a study conducted by Longe and Balogun (2010) at Solous1Dumpsite and environs in Igando, appreciable concentrations of Cr^{3+} was detected in fifty percent of the groundwater sampling wells which are located 10, 40 and 375 metres downgradient of the dumpsite. The concentrations of the physico-chemical and heavy metals constituents of the groundwater samples also revealed that groundwater downgradient of the dumpsite had been moderately impacted by leachate from the dumpsite.

Odukoya and Abimbola (2010) assessed the quality of surface and groundwater around the abandoned Isolo Dumpsite (abandoned around 1996), and the active Olusosun Dumpsite. A variety of physico-chemical, trace elements and total coliform were analysed, the results of which revealed that majority of the trace elements analysed in the surface and groundwater around dumpsites were below detection. However, the detected trace elements which include Thalium (Ti), Molybdenum (Mo^{2-}), Lead (Pb^{2+}) and Tungsten (W) were only detected in the surface water and groundwater very close to the dumpsites. The results also showed that pH, NO_3^-, Mg^{2+} and

Na concentration, as well as total coliform in most of the groundwater samples exceeded the USEPA standard for drinking water quality.

Ikem, et al., (2002) assessed the quality of groundwater around a dumpsite in the Oworonshoki area of Lagos, and a dumpsite in the Orita-Aperin area of Ibadan. The quality of groundwater was monitored and assessed for the rainy and dry seasons over a 2-year period. The results of the studies revealed that the concentration of groundwater contaminants was higher during the wet season. The results of the analysis of variance between the wet and dry season quality of groundwater for the vicinity of the Lagos dumpsite showed significant levels of variations at 95 percent confidence level for SO_4^{2-}, NH_3^-, COD, Al, Mn, Ca, Cu, HCO_3^-, total alkalinity and total hardness. For the Ibadan dumpsite significant levels of variations was established for pH, Cl^-, SO_4^{2}-, Ni, Cu, Mg, K, Na, Mn, COD, PO_4^{3-}, $NH3^-$ and NO_3^-.

2.1.7 Use of Index in the Assessment of Groundwater Quality

In the assessment of groundwater quality and its suitability for drinking purposes a number of methods exists which include the Fuzzy Mathematics Method, Membership Degree Method, Gray Modelling Method, Analytical Hierarchy Process Method, Factor Analysis Method and Water Quality Index Method (Pei-Yue, Hui & Jian-Hua, 2010). Groundwater Contamination Index developed by Backman, et al., (1997) is one of the many variants of the water quality index. The Groundwater Contamination Index has since been utilised in Slovania and Finland by Backman et al (1998), in Greece by Tziritis, et al., (2008) and Majidano, et al., (2010).

Adopting the Groundwater Contamination Index for the assessment of groundwater contamination in the Taluka, Daur District of Pakistan, Majidano, et al., (2010) rated 23 out of the 38 groundwater samples collected as having a high level contamination. Similarly, Majidano

and Yar Khuhawar (2009) utilised the Groundwater Contamination Index to assess the level of heavy metal contamination of groundwater and surface water in the Taluka, Daur District. Utilising a total of 38 samples made up of 36 groundwater samples and 2 surface water samples, Contamination Index was computed based on Cd^{2+}, Pb^{2+}, Zn^{2+}, Cu^{2+}, Ni^{2+}, Fe^{2+}, Co and Mn. The results of the computation rated 11 percent of the water samples as having a low level of contamination, another 11 percent as having a medium level of contamination, while 78 percent of the samples were categorised as having a high level of contamination. Groundwater samples with an index value less than 1 is categorised as having low level of contamination, while values between 1 and 3 signifies medium level of contamination. Groundwater is rated as highly contaminated if the index value is greater than 3.

Tziritis et al (2008) computed the Groundwater Contamination Index for the South-Eastern part of the Samos Island in Greece, based on four hydrochemical parameters, As, Cl^-, Fe^{2+} and SO_4^{2-}. Out of the 21 groundwater samples for which the Contamination Index was computed, 13 were adjudged as being uncontaminated, 4 were categorised as having a low level of contamination, 2 were rated as having a medium level of contamination, while another 2 was categorised as having a high level of contamination.

For the Sulaimaniya Governorate of Northeast Iraq, Mustafa (2008) computed the Contamination Index for Grdazubair stream, and the groundwater of Waraz village based on Cd^{2+} and Cu^{2+}. The result of the study showed both the surface and the groundwater fell within the high contamination level category. The result of the study also revealed that the level of contamination increases from upstream locations to downstream locations.

2.1.8 Attenuation of Landfill Leachate

In the 8-year monitoring period of groundwater quality at the Army Creek Landfill in Delaware, Beadecker and Apgar (1984) observed that most of the groundwater contaminants near and further downgradient of the landfill had decreased in concentration. The study attributed the attenuation of the contaminants to a number of factors which included the dilution of leachate with native groundwater and the mixing of the anaerobic leachate with oxygenated water which facilitated the decomposition and precipitation of metals. The attenuation of the groundwater contaminants was also adduced to the interaction of the contaminants with aquifer materials and the operation of the installed recovery well system which removed contaminated groundwater from and reversed the flow of groundwater.

Van Vossen, et al., (2001) studied the influence of natural attenuation on the risk and aftercare of eighty abandoned landfills in Netherlands. The results of the study revealed that only fifteen percent of the eighty analysed landfill contaminants exceeded the threshold value which corresponded to unacceptable groundwater contamination. This signified the natural attenuation of leachate plume downgradient of the landfills. The determination of natural attenuation at the landfills was based on six conditions; the presence or absence of leachate plume, the presence or absence of contaminants in the leachate plume, whether the detected leachate contaminants are in line with expected contaminants in a natural attenuation landfill, whether measured redox conditions are in line with natural attenuation standard landfill, whether detected contaminants not corresponding to the natural attenuation landfill can further be degraded or precipitated due to conditions prevailing outside the landfill and whether natural attenuation can be applied as an aftercare measure.

Investigating the influence of local hydrogeology on the natural attenuation of leachate generated at the Olusosun Dumpsite in Lagos, Longe and Enekwechi (2007), attributed the absence of lead in the groundwater, when it was detected in the leachate generated at the dumpsite to its natural attenuation by the clay that makes up part of the sub-soil stratigraphy of the area. The study also identified the depth of the unsaturated zone as a contributory factor in the natural attenuation of leachate in groundwater downgradient of the dumpsite. The attenuation of lead in clay soil is achieved through the process of ion exchange or sorption reactions such as complexation or surface precipitation (Olive, 2006). Furthermore, lead as a cation has a great potential to be attenuated by clay. This is due to the mineral-specific net negative surface charges of clay which electrostatically attracts and binds cations, particularly metals on to its surface (Yong et al., as cited in MacGregor, 2000).

In a study to assess the impact of a landfill operated on a 'dilute and disperse' principle on groundwater quality in North Yorkshire, Derval Devany (n.d), reported that the leachate generated at the landfill site attenuated. The attenuation of the leachate in a geological environment that is made up of a combination of peat, glaciolacustrine clays, and glacial sands and gravel was adduced to dilution, with the reduction in the concentration of Ammonia and Dissolved Organic Carbon (DOC) of leachate attributed to cation exchange and oxidation.

Christensen, et al., (2002), assessed the level of attenuation that had taken place in the leachate plume migrating downgradient of Vejen Landfill within a ten-year interval. The results of their study showed that within the strongly anaerobic part of the leachate plume, the concentrations of Chloride, Ammonium, Non-Volatile Organic Compounds and several other organic chemicals had reduced when compared with the results obtained ten years earlier. The results of the study also revealed that while Toulene was degraded in the leachate plume as evidenced by the

detection of its degradation product (Benzl-succinic acid) in the groundwater, Benzene and Mecropop were not attenuated in the aquifer that is made up predominantly of Quaternary sand and gravel interspersed with few silt and clay lenses.

At the Burnstump landfill, situated on the Sherwood sandstone in Nottinghamshire, England, the report of the landfill monitoring investigation conducted by the Department of Environment (1995), concluded that dilution and microbial degradation were the two prominent mechanisms responsible for the attenuation of leachate at the landfill site during the period of investigation. Their findings revealed that between 1978 and 1991, the maximum Chloride concentration of the pore water had reduced from 10,000mg/L in 1978 to 1,500mg/L in 1991. The results of their investigations also showed that the ammonicial nitrogen profiles of the leachate also exhibited a rapid attenuation of ammonia within a few meters of the base of the landfill when compared with the Chloride profiles.

Christenson and Cozzarelli (n.d) reported the results of a field experiment on natural attenuation carried out at the old unlined Norman landfill in Oklahoma, USA. Discussing the results of the push-pull or single well-injection-withdrawal test carried out by Istok, et al., (1997), it was revealed that the formate and lactate constituents of the leachate plume in the groundwater disappeared between seven to nine days, although the degradation patterns differed. The results also revealed that there were variations in the rates of natural attenuation of the two contaminants in areas of differing aquifer permeability. The variable degradation rates as evidenced by the natural attenuation was attributed to three factors; the groundwater flow regime, sediment chemistry and microbial community structure.

2.2 Laws and Concepts on the Flow and Quality of Groundwater

To ensure cohesion between the research aim and objectives of this present study, concepts, theories and laws that have proven useful in the fields of groundwater hydrology, contaminant hydrogeology and environmental management were adopted. These include Darcy's Law, Fick's Law, the Natural Attenuation Concept, and the Water Quality Index Concept.

2.2.1 Darcy's Law

Darcy's Law credited to Henry Darcy (1856) constitute the fulcrum upon which the description of groundwater flow in saturated material is predicated (Hiscock, 2005). Furthermore it serves as the basis upon which the development of new concepts and methods for the quantification of processes related to flow through a porous media is founded (Alfaro & Emilio, 2011). The empirical derivation of Darcy's Law otherwise known as Darcy's Equation is based on the result of his study of the flow of water through a porous medium contained in a column. He discovered that the total flow, Q, is proportional to both the difference in the water level $h_1 - h_2$ measured in the manometer tubes at each end of the column and the cross-sectional area of flow A, and inversely proportional to the column Length L. The combination of this, with the constant of proportionality, K, translates to Darcy's Equation expressed as:

$$Q = KA\left(\frac{h_1 - h_2}{L}\right) \quad \text{……..} \quad \text{……2.1}$$

Or alternately written as

$$Q = -KA\left(\frac{dh}{dl}\right) \quad \text{……..2.2}$$

Where

K= hydraulic conductivity of the porous material

$\frac{dh}{dl}$ = hydraulic gradient with the negative sign indicating flow in the direction of decreasing hydraulic head

A= cross sectional area at right angle to flow direction, through which flow occurs

Darcy's law is held valid for certain conditions which include saturated and unsaturated flow, steady-state and transient flow, flow in aquifers and aquitards, flow in homogenous and heterogeneous system, flow in isotropic or anisotropic media, and flow in rocks and granular media.

Darcy's law is of relevance to this present study. This is because the derivation of the governing flow equation for the two-dimensional steady state groundwater flow modelling for this study is mainly subsumed in Darcy's law. As such, the law is held valid for the steady state flow of groundwater which forms the basis for the numerical groundwater flow modelling for this study. Furthermore, the contaminant transport processes of advection and hydrodynamic dispersion which are responsible for the transport of solute (for example leachate) by flowing water and the spread of contaminant plume over a large volume of an aquifer respectively, are linked to Darcy's Law.

2.2.2 Fick's Law

"Fick's law describes the diffusion of a solute for a steady- state condition" (Todd & Mays, 2005, p.380). This is expressed in the first law of Fick as:

$$F = -D\left(\frac{dc}{dx}\right) \quad \text{.....2.3}$$

Where F = mass flux of solute per unit area per unit time ($M/L^2/T$)

D = diffusion coefficient (L^2/T)

C= solute concentration (M/L^3)

$\frac{dc}{dx}$ = concentration gradient ($M/L^3/L$)

The negative sign indicates the movement from greater to lesser concentration. "Fick's second law second describes the change of concentration over time inside the element subject to diffusion flux" (Todd & Mays, 2005, p.381). This is expressed in Equation 2.4 as:

$$\frac{\partial C}{\partial t} = D^* \frac{\partial^2 C}{\partial x^2} \quad \text{.....2.4}$$

The process of dispersıon wnich is responsible for the spreading of contaminant plume as it flows through the subsurface is the result of the process of molecular diffusion and mechanical mixing. According to Hicock (2005), molecular diffusion although not of great importance where mechanical dispersion is dominant, is of great significance in stable geological environments of low hydraulic conductivity and low hydraulic gradient where very long term deep disposal of waste is practised.

Although when Fick's Law is applied to a porous media, the diffusion coefficient is smaller due to the longer paths that ions follow between solid particles and because of adsorption. As such, it is expected that some of the constituents of the leachate generated at a point source of contamination such as the dumpsites would move from regions of higher concentration to regions of lower concentration in the direction of their concentration gradient.

2.2.3 Natural Attenuation Concept

“ Natural attenuation process include a variety of physical, chemical or biological processes that under favourable conditions, act without human intervention to reduce the mass, toxicity, mobility, volume or concentrations of contaminants in soil or groundwater” (USEPA, as cited in Krupa and Martin, 2001, p.1.1). American Society of Testing and Materials cited in National Academy of Sciences (2000, p.23), defines natural attenuation as the “reduction in mass or concentration of a compound in water over time or distance from the source of constituents of concern due to naturally occurring physical, chemical, and biological processes. According to Chapelle & Bradley (n.d) and with regards to the groundwater system, the concept of natural attenuation relates to the ability of the system to reduce contaminant concentrations along groundwater flow paths.

Natural attenuation processes include biodegradation, dispersion, dilution, sorption (physical and chemical), volatilisation, radioactive decay, chemical or biological stabilisation, and other means of transformation or destruction of contaminants.

It is dependent on subsurface hydrogeology and geochemistry, and also on the type of contaminant. This is because the processes are contaminant specific, with different processes determining the fate of individual contaminants differently. The underlying principle of natural attenuation is that prevailing natural processes in the subsurface is able to significantly reduce the concentration of critical pollutants to the extent that they no longer pose a risk to downgradient groundwater users or discharge areas (Christensen, et al., 2000). The evidence that a contaminant plume has undergone a subsurface mass reduction occasioned by natural attenuation can be derived from several indicators. These indicators include historical trends, occurrence of favourable geochemical processes or conditions, breakdown or daughter products,

and laboratory or microcosm studies. Laboratory microcosm studies can be used to simulate aquifer conditions, and show that resident bacteria can create the required biological and chemical reactions to destroy the contaminants of concern (Williams, 1998).

Historical trends throw more light on whether or not there is a decrease, stability or retreat in contaminant plume. Retreat of contaminant plume is indicative of contaminant destruction as soon as they dissolve into groundwater, signifying the attenuation of dissolved contaminants. Favourable geochemical processes can also serve as indicators that natural attenuation has taken place over time. Aerobic and anaerobic biodegradation for instance are two geochemical processes that play an important role in the natural attenuation of contaminant plume. As an indicator of natural attenuation for example, carbon dioxide and organic acids, the reaction products of aerobic biodegradation can lower the pH and increase the alkalinity within a contaminant plume. Anaerobic biodegradation on the other hand may increase pH of the contaminant plume thereby causing increased solubility of metals. Increase in the level of methane produced is also an indication of anaerobic biodegradation.

The discovery of breakdown products of contaminants in groundwater signifies the process of biochemical destruction of original or parent compounds. In order for the contaminant to be completely attenuated, the process of biodegradation must be complete because some breakdown or daughter products are more toxic than the parent compound. For example Taylor and Allen (2006), identified the dehalogenation of Perchloroethene (PCE) to Trichloroethene (TCE) as one of attenuation processes in which the breakdown product of biodegradation becomes more toxic than the parent compound. Evidence of natural attenuation can also be derived from the simulation of aquifer conditions and native micro-organisms in a laboratory environment.

Although natural attenuation landfill is prohibited in European Union countries, United States of America, and other developed countries of the world, supporters of this waste management strategy notably Allen (2000), argue that apart from the low cost of maintenance associated with a natural attenuation landfill, natural geological or hydrogeological barriers do not encapsulate waste and inhibit its degradation, and neither does it disrupt the natural infiltration and percolation characteristics of the subsurface.

In respect of natural attenuation landfill, the physical, chemical and biological attenuation processes operating within the waste body and outside in the surrounding environment depend on the type and quantity of waste tipped in the landfill and also on the geological, geochemical and hydrogeological conditions. The physical attenuation processes operating within a natural attenuation landfill (or disperse and attenuate landfill) includes absorption, adsorption, filtration, dilution and dispersion. The chemical attenuation processes include acid-base interaction, oxidation- reduction, precipitation, co-precipitation, ion-exchange, and complex ion formation, while the biological processes are aerobic and anaerobic micro-organism degradation (Williams, 1998). A natural attenuation or attenuate and disperse landfill is one in which uncontrolled release of leachate into surrounding geological and hydrogeological environment is allowed, relying on prevailing natural geological and hydrological characteristics of the subsurface to significantly reduce leachate contaminants (Williams, 1998; Allen, 2000).

With regards to this study and based on information on the lithostratigraphic profile of the study area, the clay that form part of the area's subsurface is expected at serve as an attenuating layer to the leachate generated at the dumpsites due to its low hydraulic conductivity and the filtration, sorption and cation exchange processes which is expected to take place between the leachate and the clay minerals. With regards to the groundwater flow system, concentrations of reactive and

conservative contaminants contained in the leachate are expected to decrease with distance along the groundwater flow path.

The hydraulic conductivity of clay is very crucial to the attenuation of landfill leachate due to its low permeability and water transmitting capacity. Allen (2000) noted that clay rich geologic units with a hydraulic conductivity less than $10^{-7}ms^{-1}$ serve as a good natural attenuation barrier, due to its characteristic low hydraulic conductivity, and its ability to allow the processes of filtration, sorption and ion exchange to take place simultaneously within the clay unit. In respect of filtration as a natural attenuation process, the small grain size of clay minerals ensures that the particulate matter contained in the leachate generated at landfills and dumpsites alike are prevented from migrating into the groundwater. According to Rivett et.al., (2006), particulate contaminants such as microbes, as well as organic and inorganic colloids are attenuated within the subsurface through the process of filtration, which is achieved by preventing the passage of particulate matter larger than the pore diameter.

Sorption takes when certain leachate contaminants adhere to the surface of individual clay particles. It is the temporary attachment of chemicals or organisms to soil or other geologic formation. It is a process in which incoming cations are sorbed and held in an exchangeable state by clay or any other geological materials (Suhari, 1998; Rivett et.al., 2006; Ramakrishna, et al., 2011). Similarly, leachate contaminants, precisely the cations in the leachate are subjected to attenuation by cation exchange, if the geologic materials underlying a landfill or dumpsite consist of clay and clayey soils. In the course of leachate migration into the subsurface, cations present in leachate may be exchanged for cations present on the surfaces of the clay particles. According to Suhari (1998, p.2), cation exchange is “a natural process in which the cation from an incoming

contaminant replaces the original non-hazardous cation of the geologic material as the pollutant passes through and it retains in an exchangeable state".

The expected reduction in the concentration of reactive and conservative constituents of leachate along the groundwater flow path is due to the mixing and dilution of leachate plume with uncontaminated groundwater downgradient as a result of dispersion along the groundwater flow direction. For the conservative constituents of leachate, dispersion and dilution are the only processes that can reduce the concentrations of the contaminants. Both processes are dependent on the magnitude of leachate and groundwater flow, and the relative concentrations of contaminants in the leachate and in the natural groundwater of aquifers upstream of the landfill or dumpsite (Taylor & Allen, 2006). For the reactive contaminants of leachate, natural attenuation is achieved through a combination of biochemical processes such as precipitation, and the interaction of the contaminants with aquifer materials through such processes as adsorption and cation exchange (Taylor & Allen, 2006).

2.2.4 Water Quality Index Concept

"Water Quality Index concept was first introduced more than 150 years ago, in 1848 in Germany, where the presence or absence of certain organisms in water was used as indicator of the fitness or otherwise of a water source" (Abbasi as cited in Elshemy & Meon, 2011, unpaged). The various water quality indices in existence were developed as tools to categorize the composite impact of water quality parameters, as well as provide information on the degree of purity and quality of the water source under consideration.

Water quality indices are of two broad classifications; objective and the subjective indices (Ott as cited in Bhatti, 2009). According to Bharti & Katyal (2011), the objective indices otherwise

known as statistical indices are based on statistical methods without any recourse to subjective inferences or personal opinions. The subjective indices on the other hand are those derived through the utilisation of weights and rating functions. Both the weights and the rating functions are subjectively derived and are based on answers from experts to which the questionnaires are administered (Ott as cited in Bhatti, 2009).

Some of the water quality indices that have over the years been developed include the Arithmetic Water Quality Index (WQI_A) developed by Horton (1965), the multiplicative Water Quality Index (WQI_M) developed by Brown et al (1970), the Un-Weighted Arithmetic Water Quality Index (WQI_{UA}), the Un-Weighted Multiplicative Water Quality Index (WQI_{UM}), Harkins' (1974) Water Quality Index (WQI_H), the Delphi Approach to water quality index calculation, and the British Columbia Index developed by the Environment Protection Department of British Columbia (Bhatti, 2009).

The computation of Horton's Arithmetic Water Quality Index is based on the following water quality parameters; Dissolved Oxygen (DO), pH, Coliforms, Chloride, Alkalinity and Specific Conductance, which serves as an approximation for Total Dissolved Solids. Other variables included are Carbon Chloroform Extract (CCE) which was included to reflect the influence of organic matter, and sewage treatment to show the effectiveness of pollution abatement measures. The levels of water purity are represented on a graduated scale, while the index score is computed by dividing the sub-index indices by the sum of the weights, and multiplied by the coefficients of temperature and pollution (Abassi, 2002). The weighted arithmetic mean is expressed as:

$$WQI_A = \frac{\sum_{i=1}^{n} w_i q_i}{\quad} \qquad \text{.....2.5}$$

Where n = number of variables

Wi = relative weight of the i^{th} parameter such that $\sum_{i=1}^{n} w_i = 1$

and q_i is the quality of rating of the i^{th} parameter

The Multiplicative Water Quality Index of Brown (1970), also known as the National Sanitation Foundation Water Quality Index is a subjective index based on Delphi technique. The index was developed based on nine parameters; Dissolved Oxygen, Faecal Coliform Density, pH, Biological Oxygen Demand, Nitrates, Total Phosphates, temperature change from equilibrium, Turbidity and Total Solids. The quality of water is categorised into five, with values ranging between 0 to 25, 26 to 50, 51 to 70, 71 to 90 and 91to 100. The ranges denote water of very bad, bad, medium, good and excellent quality (Wills & Irvine, 1996). The index is expressed as:

$$WQI_M = \prod_{i=1}^{n} qi \qquad \text{.....2.6}$$

Where Π = sub-index of each parameter

and q_i is the quality of rating of the i^{th} parameter

With regards to the Un-weighted Arithmetic Water Quality Index and the Un-weighted Multiplicative Water Quality Index the possibility that weights may not be necessary in differentiating the qualities of water led to their development (Landwehr as cited in Bhatti, 2009). The Un-weighted Arithmetic Water Quality Index (WQI_{UA}) and the Un-weighted Multiplicative Water Quality Index (WQI_{UM}) is as defined in equations 2.7 and 2.8 respectively.

$$WQI_{UA} = (1/n)\sum_{i=1}^{n} q_i \qquad \text{.....2.7}$$

Where q_i is the quality of rating of the i^{th} parameter

$$WQI_{uM} = \frac{\left[\prod_{i=1}^{n} qi\right]^{1/n}}{\qquad} \quad \text{.....2.8}$$

Where Π = sub-index of each parameter

and q_i is the quality of rating of the i^{th} parameter

Harkin's Water Quality Index is an objective water quality index in which numerical rankings are apportioned to the values of the hydrochemical parameters relative to selected control values for each parameter. The index is not dependent on rating curves and weightings. The information obtained from the ranking procedure is subsequently used to compute the standardised distance from the control values for each parameter to derive an index of water quality (Sharifi, 1990).

The Delphi approach to water quality index computation is based on ratings of water quality parameter by water quality experts. The ratings obtained from the survey are thereafter subjected to statistical analysis such as regression. For the formulation of the weighted water quality indices, the experts provide the relative weighting factors for all the analytes (Bhatti, 2009).

The British Columbia Water Quality Index was developed by the Canadian Ministry of Environment in 1995 for different purposes namely, drinking, recreation, irrigation, livestock watering, wildlife and aquatic life with each use having a separate ranking. The index is based on the comparison of concentration of water quality parameters with a defined benchmark (Bharti & Katyal, 2011; Bhatti, 2009).

In their review of the different water quality indices in existence, Sohbani as cited in Bharti and Katyal (2011) divided water quality indices into five major groups, namely the public indices, which does not take cognizance of the type of water consumption in the evaluation process, the specific consumption indices which is based on the kind of consumption and application, such as drinking, industrial and ecosystem preservation, the statistical indices which are based on statistical methods only, and the designing indices which are basically aimed at planning and decision making in water quality management projects.

A summary of some of the water quality indices developed nationally or internationally according to Bharti and Katyal (2011) include the Scatter Score Index (Kim & Cardone, 2005), Index of River Water Quality (Liou et al., 2004), the Overall Index of Pollution (Sargaonkar & Deshpande, 2003), the Chemical Water Quality Index (Tsegaye et al., 2006), and the Water Quality Index for Freshwater Life (Canadian Council of Ministers of the Environment [CCME], 2001).

The Scatter Score Index was developed basically to detect positive or negative changes in the quality of water around mining sites in the United States. The index identifies spatio-temporal increases or decreases in the quality of water. The number of hydrochemical parameters that can be included in the index is unlimited, and is not dependent on any water quality standards or guidelines (United Nations Environmental Programme Global Environmental Monitoring System (GEMS)/Water Programme, 2007).

The Index of River Water Quality by Liou et al., (2004) was developed to assess the conditions of rivers in Taiwan. The index is a multiplicative aggregate function of standard scores of selected water quality parameters; temperature, pH, toxic compounds, Dissolved Oxygen, Biological

Oxygen Demand, Ammonia, Suspended Solids, Turbidity, and microorganisms such as Faecal Coliforms. The Index is dependent on the geometric means of the standardised scores for each hydrochemical parameter which has been predetermined on the rating curves for the index. The quality rating of water is from 0 to 100, with 0 representing water of poor quality and 100, water of excellent quality (GEMS, 2007).

The Overall Index of Pollution was developed by Sargaonkar and Deshpande (2003) to assess the quality of Indian Rivers. The index is based on nine analytes; pH, Dissolved Oxygen, Biological Oxygen Demand, Hardness, Total Dissolved Solids, Total Coliforms, Arsenic and Flouride. Each analyte is rated as excellent, acceptable, slightly polluted and heavily polluted, based on Indian or World Health Organisation standards. Each observation or sample is subsequently assigned a pollution index value and the Overall Index of Pollution is calculated as the average of each index value (United Nations Environmental Programme Global Environmental Monitoring System/ Water Programme, 2007).

The Chemical Water Quality Index developed by Tesgaye (2006) is based on seven water quality parameters, namely Total Nitrogen, Dissolved Lead, Dissolved Oxygen, pH, Total Particulate and Dissolved Phosphorous. The index involves the standardisation of each observation to the maximum concentration for each analyte before the concentration of each parameter is summed up (GEMS, 2007).

The Water Quality Index for Freshwater Life developed by CCME (2001) is a computer based model fashioned after the British Columbia Water Quality Index. The Index is based on three factors each of which is scaled from 0 to 100, and combined to create a vector to represent the index. The factor represents the number of variables not meeting the water quality objectives,

while the second factor represents the amplitude and number of times the objectives were not met. The third factor is the amount by which the objectives were not met (Bhatti, 2009). The comparison of observations to a water quality standard or a site specific background concentration rather than normalising the observed values to subjective rating curves, constitute a major advantage of the Water Quality Index for Freshwater Life over other indices (Bharti & Katyal, 2011).

CHAPTER THREE

RESEARCH METHODOLOGY

3.1 Methodological Framework

Research according to Singh (2006, p.1) "is simply the process of arriving as [*sic*] dependable solution to a problem through planned and systematic collection, analysis and interpretation of data". In achieving the aim of this study which is to assess the potential linkage between the groundwater flow pattern, the leachate generated at the dumpsites and the consequences for groundwater quality within the study locations, both primary and secondary data sources were utilised. Information on the types and sources of data used are presented in Table 3.1, while the methodological framework for the study is as shown in Figure 3.1.

Data and information of relevance to the study were obtained from different literatures and reports. These include the Hydrogeology of Lagos Metropolis (Longe et al., 1987), Hydrogeological Investigation of Lagos State (Kennard & Lapworth, 1997), Report of the Lagos State Waste Management Authority Environmental and Social Assessment for Solid Waste Management (2008) and the Report of the Monitoring Boreholes at Landfill Sites in Lagos Metropolis- Abule-Egba and Igando (2008).

The Ikonos imageries of Lagos were obtained to delineate the model boundary. Boundaries of groundwater flow models are geologic and hydrologic features that influence groundwater flow patterns. They include features such as watershed boundaries, surface water features, water table divide, faults and outcrops (Karadas, 2009). The model boundary delineation involved the selection of rivers around the study locations to serve as boundaries for the model domain. The

extent of model domain was subsequently delineated in ArcGIS 9.3 using the clip tool, after which the areal extent of the domain was calculated in ArcGIS using the calculate geometry tool. Information on static water level measurements was obtained from the report of the Hydrogeological Investigation of Lagos State (Kennard & Lapworth, 1997). This served as hydraulic head input into the MODFLOW software.

Leachate samples were collected from the leachate springs formed near the emplaced waste matrix at each dumpsite, while groundwater samples were collected from wells and boreholes in the vicinity of the dumpsites. For the collection of leachate and groundwater samples, one of the field equipment utilised include portable pH/EC/TDS meter, which was used in carrying out the in-situ measurement of pH, Electrical Conductivity (EC) and Total Dissolved Solids (TDS). The pH of the groundwater was measured because pH gives an insight into the total chemical composition of water. According to Sundaram et al., (2009, p.38), pH of water "largely controls the amount and chemical form of many organic and inorganic substances dissolved in groundwater". As further explained by Sundaram et al., (2009), the suitability of water for various purposes and its ability to transport potentially harmful chemicals are controlled by pH. pH was determined in-situ because pH is temperature dependent and subject to the influence of natural factors such as the dissolution of carbon-dioxide. For leachate, pH was measured to determine the type of leachate produced at the dumpsites, as well as probable stage of waste degradation.

A handheld Global Position System (GPS) was utilised to obtain the geographical coordinates of sampling locations for later geoferencing of the combined Ikonos imageries and the digital copy of Lagos Street maps which were used to create the sampling location maps. Georeferencing is the establishment of the relationship between the base maps and coordinates obtained from the

field. It is establishment of the correct position of coordinates on maps. Data from the leachate collected at the dumpsites were utilised for the characterisation of the leachate and the computation of the Leachate Pollution Index (LPI), while the results of the collected groundwater samples were used for the determination of groundwater quality, the computation of Principal Component Analysis, establishment of spatial pattern of groundwater quality through spatial interpolation and the computation of Groundwater Contamination Index (C_d) of the groundwater in the vicinities of the dumpsites and their control sites. Principal Component Analysis was carried out to identify the hydrochemical parameters and the chemical processes responsible for most of the variance in the groundwater quality around the dumpsites.

During the course of the study some challenges were encountered. These include reluctance and some times refusal by residents to have their wells sampled, reluctance of LAWMA to release the report on their dumpsites, and inadequate information in LSWC data bank concerning the study locations. With regards to the reluctance and refusal of some of the residents, they were persuaded and assured that the purpose of sampling was for academic reasons. The benefits of knowing the quality of water they consume was also explained to them in order to get them to change their minds, which they eventually did. The reluctance on the part of LAWMA to release the reports on their dumpsites was overcome by using the goodwill of a faculty member to get them to release the reports.

The inadequacy of information in the LSWC data bank was overcome by complementing the data, with data and information obtained from the Hydrogeology of Lagos Metropolis (Longe et al., 1987) and information obtained on the hydrogeological profile of the study locations from the report of the monitoring boreholes drilled at the dumpsites (Hydromarine-Environ quest 2008; and LAWMA 2008).

Table 3.1: Data Types and Sources

Data	Data Type	Source (s)	Purpose
Geographical coordinates of sampled wells and boreholes	Primary	Fieldwork	Mapping
Results of in-situ and laboratory analyses of selected chemical and heavy metals constituents of leachate	Primary	Fieldwork and Chemistry Laboratory, Unilag	Leachate characterisation and LPI computation
Results of in-situ and laboratory analyses of selected chemical and heavy metals constituents of groundwater	Primary	Fieldwork and Chemistry Laboratory, Unilag	Groundwater quality determination, PCA computation and C_d computation
Hydrogeology of Lagos Metropolis	Secondary	Longe et al., (1987)	Groundwater flow Modelling,
Hydrogeological Investigation of Lagos State	Secondary	Kennard & Lapworth (1997)	Groundwater flow Modelling,
Report of the Lagos State Waste Management Authority Environmental and Social Assessment for Solid Waste Management	Secondary	LAWMA (2008)	Volume of Waste Tipped at the Dumpsites
Report of the Monitoring Boreholes at Landfill Sites in Lagos Metropolis-Abule-Egba and Igando	Secondary	Hydro Marine Environ Quest LAWMA(2008)	Information on the Hydrogeological Profile of the Dumpsites
Ikonos Imagery of Lagos	Secondary	LASPEDA (2001, 2002)	Mapping
Google Earth Imagery	Secondary	Google Earth (2009)	Mapping
Lagos Street Map	Secondary	LASEPDA (2009)	Mapping
Hydraulic Conductivity and Porosity	Secondary	Todd & Mays (2005)	Groundwater flow modelling

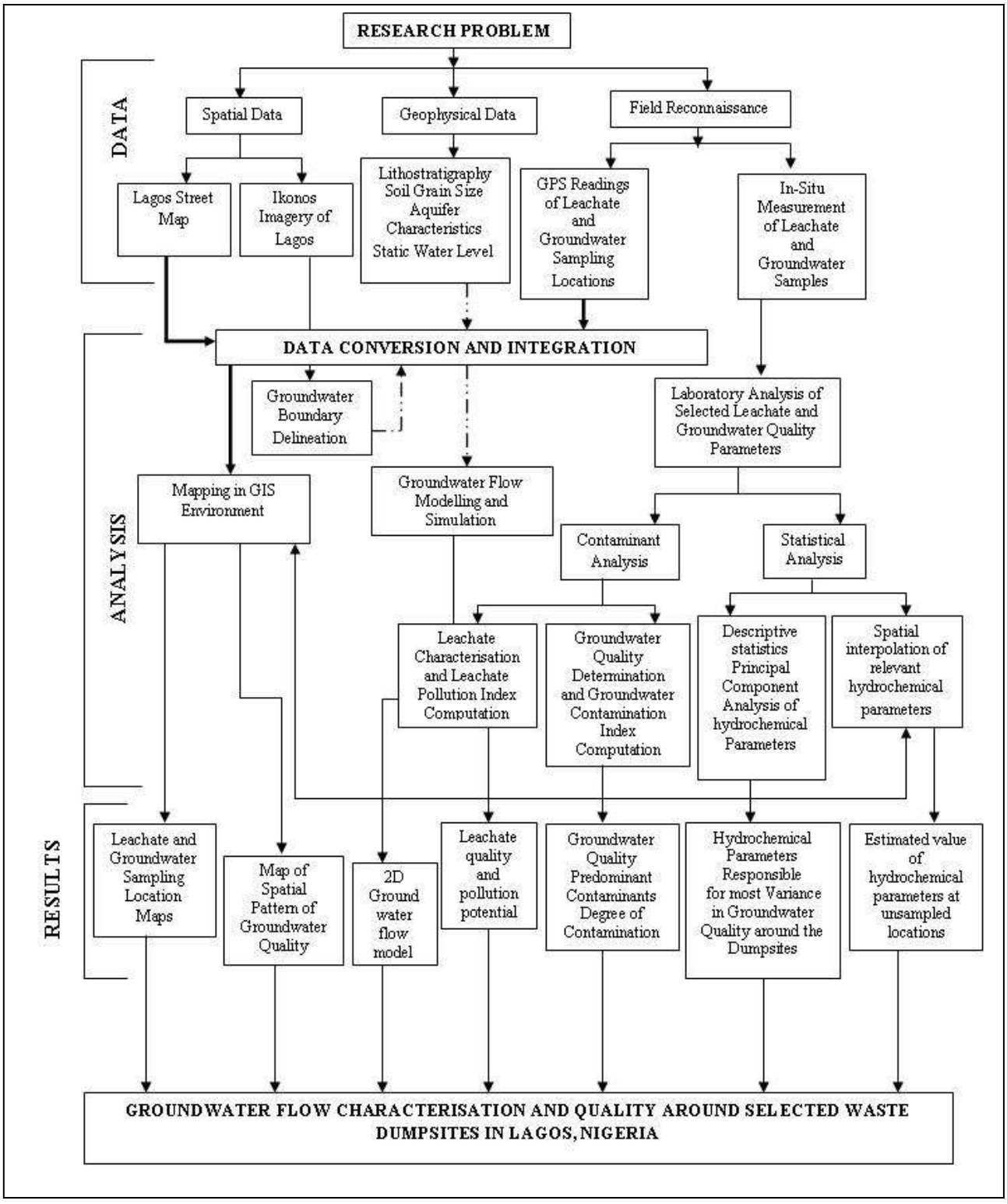


Fig 3.1 Methodological Framework for the Study

3.2 Methodologies Adopted

3.2.1 Characterisation of Groundwater flow around Abule-Egba and Solous 1 and Solous 2 Dumpsites

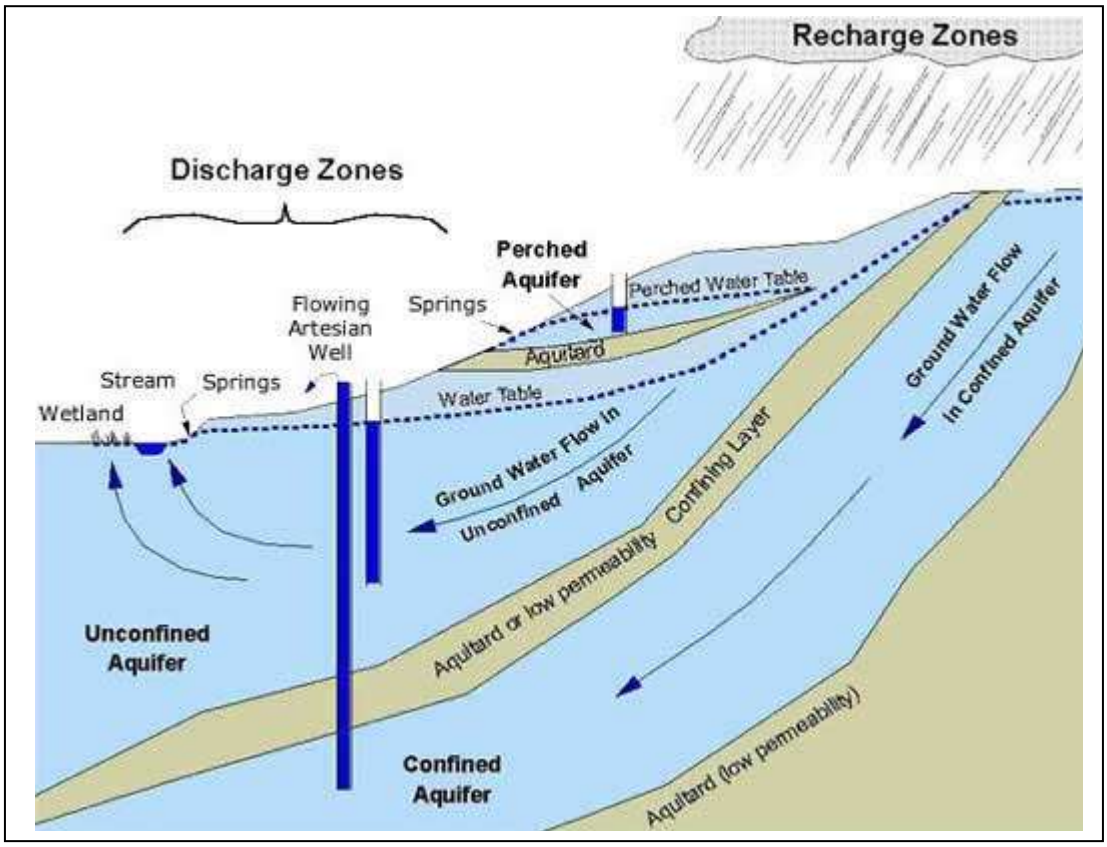
Groundwater flow modelling was adopted as an investigative tool to understand the groundwater flow system around the dumpsites and to determine the possible migration pathways of leachates generated from the dumpsites. Findings should help to forestall the inappropriate siting of wells which are the sole source of domestic water supply within the study locations. For the characterisation of groundwater flow around Abule-Egba and Solous 1 and Solous 2 Dumpsites, the following modelling processes/ protocol was followed:

3.2.1.1 Development of a conceptual model

Conceptual models constitute a means of systematically organising required information about the hydrogeological characteristics of groundwater flow system based on available data. The process of conceptualisation makes field problems easier to analyse because the simulated system simplifies the complex physical reality of the hydrogeological system (IAEA, 2001). In the development of conceptual models, the principle of simplicity is adopted. According to Levins as cited in Barnett et al., (2012), the development of conceptual models require a trade-off between realism, generality and precision because it is impossible to maximise the three simultaneously. Data utilised for this study was obtained from the Hydrogeology of Lagos Metropolis (Longe et al., 1987) and the Hydrogeological Investigation of Lagos State (Kennard & Lapworth, 1997). Hydrogeological data on Agege and Shasha (Longe et al., 1987) was used for simulating groundwater flow at Abule-Egba and Igando respectively due to their close proximity. Data on the static water level utilised in this study was obtained from the borehole static water level data of the Hydrogeological Investigation of Lagos State (Kennard &

Lapworth, 1997). The conceptual model of a groundwater system shown in Fig 3.2 describes the movement and direction of groundwater flow within different classes of geological formations.

The construction of a conceptual model for the dumpsites and their vicinities required information on the physiography (Relief, Topography, and Drainage), Geology (Local and Regional), Hydraulic Properties and Hydrostratigraphic Units (Stratigraphy and Structural Geology) of the area. These information are presented in sections 1.8.2 and 1.8.3.



Source: http//imnh.isu.edu/digitalatlas/hydr/concepts/gwater/aquifer.htm
Fig 3.2 Conceptual model of a Groundwater Flow System

3.2.1.2 General Simplifying Assumptions

"In nature, groundwater flow patterns are complex, and continuously change with time, but for the purposes of modelling, simplifications are required" (Barnett, 2012, p.8). Due to this reason the principle of simplicity is usually invoked in the specification of general simplifying assumptions for modelling of groundwater systems.

In order to simulate groundwater flow in the vicinities of the dumpsites the following simplifying assumptions were made. These include the following:

i. Steady state conditions were assumed to hold for aquifer and groundwater flow system. "Steady state flow occurs when at any point in a flow field, the magnitude and direction of flow velocity are constant with time" (Freeze and Cherry, 1979, p.49).

   The steady state flow theory is based on the assumption that inflows into a system is equal to outflow from the system. This signifies a condition in which heads and water table elevation remain invariant with time (Barnett et al., 2012). Steady state conditions were assumed for the study locations for a number of reasons. Firstly, it is conventional to assume steady state flow conditions for groundwater modelling. According to Barnett et al., (2012), one of the common simplifications required for model design is the assumption of steady state. Furthermore, the assumption of steady state flow conditions are allowed for groundwater flow modelling due to some of the uncertainties and impossibilities associated with simulating and accounting for all spatial and temporal variability associated with the groundwater flow system.

   According to Hashemi et al., (2012), it is common to use steady state conditions to estimate and define aquifer and boundary conditions due to the fact that hydraulic conductivity, recharge and other aquifer parameters cannot be directly and adequately

measured. Lastly, groundwater flow was simulated under steady state conditions due to the non-availability of continuous temporal data regarding groundwater flow system of the study locations.

ii. The aquifer is assumed to be single layer, and thus vertical flow is considered negligible.

iii. Flow is assumed to be 2-dimensional and horizontal on the assumption that the aquifer is a regional aquifer, and the aquifer is thin in comparison to its lateral extent. Groundwater flow may be assumed to be areally 2-dimensional if the aquifer is relatively thin compared to its lateral extent.

This assumption allows the 3-dimensional flow equation to be reduced to the case of 2-dimensional flow for which additional simplifications can be effected (Konikow, n.d). Riegger (2010) also highlighted the conditions for which 2-dimensional groundwater flow modelling can be adopted to include, when the aquifer thickness is much smaller than the horizontal extent of the layer, and when only one layer of the aquifer is being considered or modelled.

The term “regional refers to a relatively large aquifer domain in which the horizontal dimension may range between tens to hundreds of kilometres, while the thickness may be tens to hundreds of metres (Bear et al., 1992).

iv. The aquifer is assumed to be homogenous and isotropic. This assumption is allowed in groundwater modelling due to limited amount of information that is available on the variation and magnitude of hydraulic conductivity (Hemker & Bakker, 2006).

v. Different boundary conditions are assumed in the model; at Abule-Egba, a constant head boundary was assumed for the Western and Eastern boundaries of the model domain while the no boundaries conditions were specified for Northern and Southern boundaries. At Igando the Northern and Western boundaries of the model domain were assumed to be a constant head boundary while no boundary conditions were specified for the Eastern and Southern boundaries of the model domain.

3.2.1.3 Governing Flow Equations

The assumption that groundwater flow is areally 2-dimensional based on the relative thinness of the aquifer when compared to its lateral extent allows the 3-dimensional finite-difference groundwater flow equation to be reduced to a 2-dimensional groundwater flow equation.

The 2-dimensional steady state groundwater flow equation that combines Darcy's equation with the continuity equation was used for the computer simulation. The mathematical model simulates flow indirectly by means of a governing equation thought to represent the physical processes that occur in the groundwater system. The two dimensional steady state flow equation is expressed as:

$$\frac{\partial}{\partial x}\left[K_x \frac{\partial h}{\partial x}\right] + \frac{\partial}{\partial y}\left[K_x \frac{\partial h}{\partial x}\right] = 0 \quad \ldots\ldots\ldots\ldots\ldots\ldots (3.1)$$

Where

h = Hydraulic head

Kx and Ky = Hydraulic conductivities in their present coordinate directions

3.2.1.4 Modelling Approach

A numerical groundwater modelling software known as Processing MODFLOW which was originally developed for a remedial project of landfill site in the coastal region of Germany by Chiang and Kinzelbach in 1989 was used for the construction of the groundwater models for the study locations. For the construction of the groundwater models of the study locations, the Processing MODFLOW Software (Pmwin Version 5.3) was utilised.

In order to employ the finite difference technique of groundwater modelling, a grid was overlaid on an Ikonos Imagery of the study locations and the hydraulic and aquifer parameters necessary to solve the groundwater flow equation was imputed.

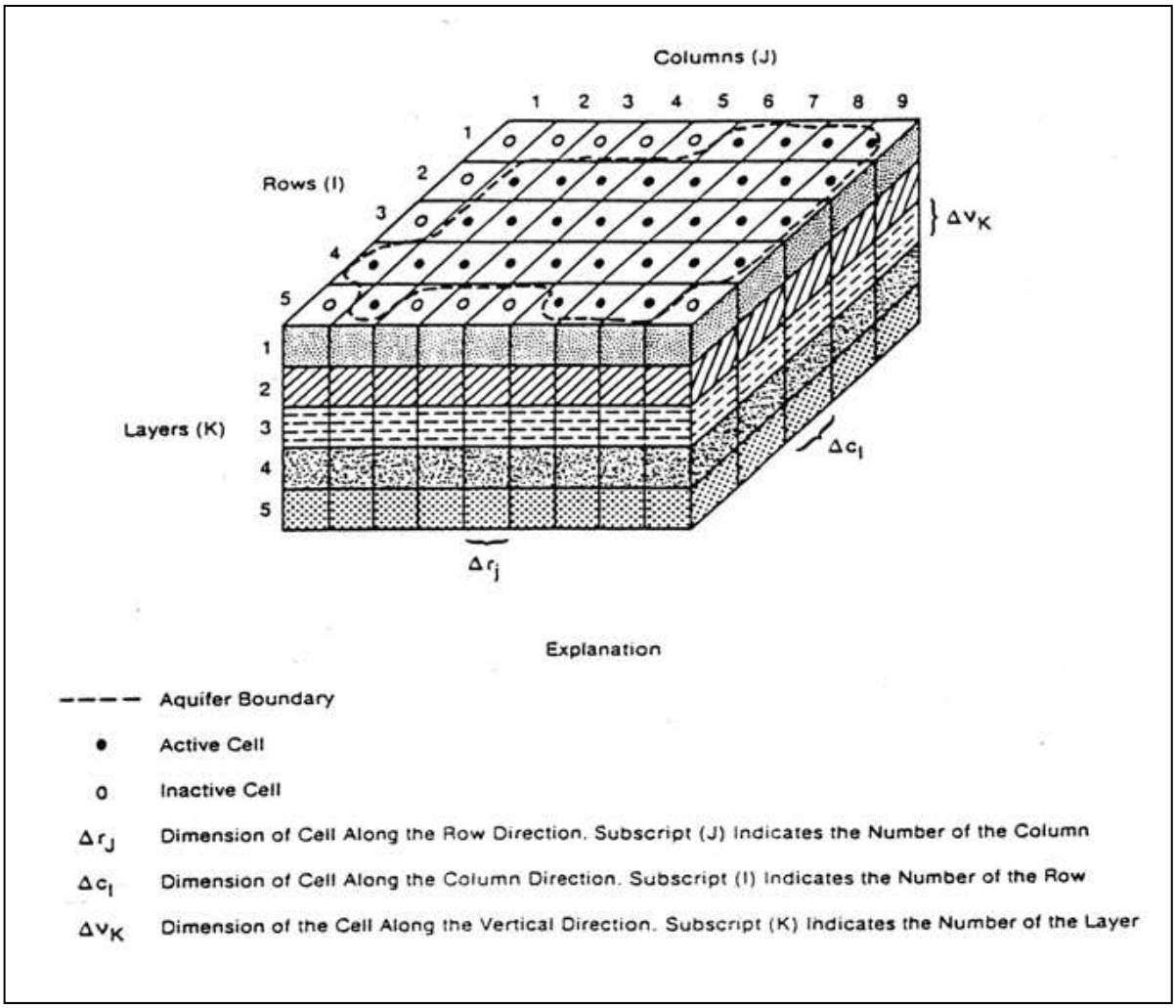
MODFLOW was utilised for this study for a number of reasons. Firstly, it is suitable for landfill studies and it has been used in several researches to study groundwater and contaminant flow around landfills. Secondly, depending on the level of detail provided by the available data, MODFLOW can be used for simulating groundwater in one dimension, two dimensions, quasi-three dimensions and three dimensions. Thirdly, it is much easier to understand when compared with other models, and lastly it requires relatively fewer data to model groundwater flow.

In order to employ the finite difference technique of groundwater flow modelling, a grid was overlaid on the Ikonos imageries of the study locations. This was done to discretise the aquifer system represented in the model domain into an array of nodes and associated finite difference blocks or mesh of cells described in terms of columns, rows and layers as illustrated in figure 3.3 The number of rows and columns (which is referred to in MODFLOW as the model grid) obtained from the superimposition of the grid over the Ikonos imageries, and the number of aquifer layers to be simulated is specified or imputed into MODFLOW.

This is followed by the specification of boundary conditions, which is subsequently followed the specification of aquifer geometry. This stage is followed by the specification and imputing of the hydraulic parameters necessary to solve the groundwater flow equation. These parameters include the initial hydraulic head, hydraulic conductivity, transmissivity and effective porosity. This is subsequently followed by the performance of flow simulation (model run) in MODFLOW. All units used during the modelling process were consistent, as this is one of the requirements of using the Pmwin software. Consistent units here refer to the fact that parameters imputed or specified in the model were expressed in metric units.

3.2.1.5 Spatial Discretization of Model Grid

The boundary delineation and spatial discretisation for Abule-Egba resulted in a model domain of approximately 4.92km^2. The model grid consisted of 51 columns and 38 rows, with each cell having dimension of 58.38m by 44.57m. For Igando, the model domain covered an area of approximately 10.72km^2, with the model grid consisting of 74 columns and 39 rows. The dimension of individual cells contained within the grid was 84.55m by 43.32m. The calculate geometry tool of ArcGIS 9.3 was used to compute the areal extent of the model domain and area of each cell making up the model grid.



Source: Thangarajan (2007).
Fig 3.3 Illustration of the Spatial Discretisation of an Aquifer System and the Cell Indices

3.2.1.6 Physical Parameters of the System

A groundwater modelling process involves the specification of initial values for all necessary parameters. The spatial input variables were initial hydraulic heads, horizontal hydraulic conductivity, transmissivity and effective porosity. Values were interpolated for each cell based on each parameter and in relation to the nature of the boundary.

3.2.1.6.1 Initial Hydraulic Heads

The initial hydraulic heads refer to the hydraulic head distribution in the groundwater system at the onset of the simulation. For groundwater flow modelling at Abule-Egba, the initial hydraulic heads were obtained from the borehole static water level measurement of Lagos (Kennard & Lapworth, 1997). The static water level of 22.02m and 30.02m were obtained from the static water level of the boreholes of the Ministry of Horticulture, Agege, and Agege Mini Water Works. For groundwater flow modelling at Igando, the initial hydraulic heads of 44.41m and 46.50m obtained from the static water level measurement of Shasha Mini water works was used. These values were imputed and interpolated in Pmwin as the prescribed hydraulic heads in each cell within the model domain.

3.2.1.6.2 Hydraulic Conductivity and Transmissivity

Hydraulic conductivity is a coefficient of proportionality describing the rate at which water is transmitted through a unit width of aquifer under a unit hydraulic gradient. This water transmitting characteristic in turn controls the rate of groundwater movement in the saturated zone, thereby controlling the migration and fate of contaminants. “One of the important hydraulic parameters required for a steady state groundwater flow model is either transmissivity or hydraulic conductivity in a distributed fashion cell by cell across the model grid” (Tesfaye, 2009, p.42).

According to Longe et al (1987), the lithology of the first aquifer horizon at Agege is characterised by fine to medium poorly sorted sands. For Shasha, the poor to fairly well-sorted medium to coarse sands of the second aquifer is assumed to prevail in the first aquifer. The lithological characteristics of Agege and Shasha were adopted for Abule-Egba and Igando respectively. Due to non-availability of data on the hydraulic conductivities of the first aquifer

horizon at both locations, representative values of hydraulic conductivities of geologic materials was used. The representative value of hydraulic conductivity of medium sand was used for the simulation. According to Morris and Johnson as cited in Todd and Mays (2005), the representative value of hydraulic conductivity for medium sand is 12m/day.

The transmissivity values utilised for the model was computed from the value of the hydraulic conductivity. The transmissivity values were computed based on the formula:

$T = K_t b$ …………………………………………………………………… (3.2)

Where T= Transmissivity

K_t = Hydraulic Conductivity of the aquifer

b = Aquifer thickness

The value of 216m^2/day was computed for Abule-Egba while a value of 144m^2/day was computed for Igando.

In the construction of two dimensional groundwater flow models, vertical hydraulic conductivity or leakance is not required for two reasons. Firstly, the assumption of a single layer aquifer suggests a predominantly horizontal flow and secondly, MODFLOW assumes that the bottom of the aquifer is underlain by impermeable materials. Leakance "is the ratio of the vertical hydraulic conductivity of a confining unit divided by its thickness" (Mandle, 2002, p.42). It can also be described as "the rate of flow across a unit (Horizontal) area of a semi-pervious layer into (or out of) an aquifer under one unit of head difference across this layer" (Mandle, 2002, p.42).

3.2.1.6.3 Aquifer Geometry

The demarcation of the aquifer extent for the model was based on information on the lithological profile of boreholes at Agege and Shasha (Longe et al, 1987). The first aquifer horizon at Agege

and by inference Abule-Egba is between 20 and 38m above sea level, while for Shasha and by inference Igando, the first aquifer horizon is between 27 and 12m.

3.2.1.6.4 Aquifer Thickness

Aquifer thickness refers to the vertical extent of a geological formation capable of transmitting water. For a confined aquifer, it is equal to the formation thickness while for an unconfined aquifer it is equal to the saturated thickness. The aquifer thickness was specified as 18 and 15m for Abule-Egba and Solous respectively. This was based on the difference between the top and bottom elevation of the first aquifer horizon.

3.2.1.6.5 Effective Porosity

The Representative value of porosity for medium sand which constitutes the aquifer matrix was used for the groundwater flow modelling. According to Morris & Johnson as cited in Todd & Mays (2005), the representative value of the porosity of medium sand is 39 percent.

3.2.1.6.6 Model Boundary Conditions

" Boundary conditions are mathematical statements specifying the dependent variable (head) or the derivatives of the dependent variables (flux) at the boundaries of the problem domain" (Elango, 2005, p.82). Precise and accurate definition of boundary conditions is an important element in the conceptualisation and modelling of groundwater flow systems. This is because boundary conditions are essential in specifying or defining how an aquifer, and indeed the site specific model interact with the environment outside the model domain and the entire groundwater flow system (Sinha, 2005; ASTM, 2008). In a steady state groundwater flow modelling, such as this study, boundaries largely determine the flow patterns.

For Abule-Egba Dumpsite and its vicinity, a constant head boundary was assumed for the Western and Eastern boundaries of the model domain. River Illo which forms the Western and

River Abesan and the adjoining canal which forms the Eastern boundary of the model domain is taken as a constant head boundary. No boundary conditions were specified for the Northern and Southern boundaries of the model domain. The head was allowed to be automatically calculated because no visible natural features exist within the specified model domain that could serve as a boundary.

For the boundary conditions at Igando (Solous area and beyond), the Northern and Western boundaries of the model domain were assumed to be a constant head boundary. No boundary conditions were specified for the Eastern and Southern boundaries of the model domain, and flow was automatically calculated by MODFLOW because no natural boundary features were near enough to the specified model domain.

The rivers which were used as boundaries of the model domain were taken as constant head boundaries because the assumption of constant head is consistent with the steady state flow conditions of the groundwater flow modelling.

3.2.1.6.7 Model Run or Flow Simulation

The model run involved imputing all parameters required for the generation of a groundwater flow model such as model grid size and spacing, layer elevations, boundary conditions, hydraulic conductivity, transmissivity and effective porosity. The hydraulic heads used for the flow simulation in MODFLOW was used in generating the velocity vectors and equipotential lines. This was executed in the environment option of PMPATH, a menu of MODFLOW that generates pathlines and contours.

3.2.2 Leachate and Groundwater Quality Analyses

Leachate and groundwater samples were collected for the wet and dry seasons at the dumpsites and environs. Prior to samples collection, a reconnaissance survey of the study area was carried out based on the groundwater flow model and Technical reports on the dumpsites. All the sampling locations (Figures 3.4 and 3.5) except the control sites were, less than 500m from the dumpsites. The sampling points were confined to less than 500m because a distance of 500m is considered a safe distance from a landfill for water abstraction points such as wells and boreholes (Taylor and Allen, 2006).

The choice of the control sites for Abule-Egba Dumpsite was based on the results of the groundwater flow model and the distance of the control sites from the dumpsite. Both control sites are more than 600m away from the dumpsite. For the Solous 1 and 2 Dumpsites, the results of the groundwater flow model and the presence of Solous 3 Dumpsite within a 600m distance from the two dumpsites necessitated the choice of the control sites both of which are located more than 2km away from the two dumpsites. Furthermore, the natural attenuation capacity of the clay that forms part of the area's geology and its low hydraulic conductivity would ensure that leachate from the dumpsite has no influence on groundwater at the control sites at more than 2km distance from the dumpsites.

Procedures for sample collection, in-situ and laboratory analyses of the hydrochemical parameters were in conformity with the American Public Health Association Standardised method for the examination of water and wastewater (APHA, 1992). In collecting water from wells and boreholes with installed pumps, the pump was allowed to run for 15 minutes before samples were collected. For water collected directly from wells through the use of a bailer, the water in such wells was agitated before being drawn out. These were done in order to obtain a

homogeneous and representative sample of groundwater from each of the sampling locations. As part of the quality assurance procedure, the polyethylene sample bottles were thoroughly washed with detergent and rinsed with de-ionised water. Polyethylene bottles were used for the collection of samples due to the lack of interaction between polyethylene and analytical parameters (EPA Victoria, 2009). At the point of sample collection each sample bottle was rinsed thrice with groundwater from the point of collection before being filled with the water. Sample bottles were filled to the brim and tightly capped in order to exclude air from the bottles.

Leachate samples were collected from the leachate springs at the dumpsites. Leachate samples for laboratory analysis were collected with a plastic container which was slowly submerged into the spring to prevent any disturbance. The leachate collected with the plastic container was also slowly poured into decontaminated Polyethylene bottles, and put in an ice box to retard chemical reactions. Furthermore parameters with short holding time such as pH and EC, as well temperature and TDS were measured in-situ.

The process of laboratory analysis of leachate heavy metals involved the filtration of 50ml leachate samples with a 0.45μm filter into a digestion vessel. After filtration, Nitric acid (HNO_3) was added to the leachate samples, and placed inside a microwave digester, where it was heated under pressure. After the digestion process was completed, the clear colourless samples which were the products of the digestion process were allowed to cool, and later aspirated into 200 Perkin-Elmer Atomic Absorption Spectrophotometer (AAS) for the analysis of heavy metals.

The selection of the leachate and groundwater quality parameters was informed by past studies which have shown that these parameters are among those which are frequently found in landfill

leachate and also because of their contamination potential (Kjelsen et al., 2002). In Table 3.2 the analytical devices used in the determination of the groundwater quality parameters are presented.

For the in-situ measurement of pH, TDS, EC and Temperature, Hanna Combo pH and EC waterproof, H198129 meter was used. The portable pH and EC meter was calibrated in a buffer solution of pH 7 for the purpose of pH measurement, while the portable meter was calibrated in H17031 (1413µS/cm) solution for the EC and TDS measurements, 6 hours before the commencement of sampling. According to Weight (2008), calibrating a pH meter 6 hours to the commencement of sampling ensures that the best result is obtained. In order to prevent cross contamination the electrode of the portable meter was rinsed with de-ionised water in between measurements. The leachate and groundwater samples collected on the field were transported in ice to the laboratory.

Table 3.2: Leachate and Groundwater Parameters and Analytical Devices

Leachate and Groundwater Parameters	Analytical Devices/Procedure
pH	Hanna Combo pH & EC, waterproof, H198129 meter.
Cl^-	Silver Nitrate Titration with Potassium Chromate as an indicator
SO_4^{2-} & NO_3^-	Spectronic 20D+
Zn^{2+}, Fe^{2+}, Pb^{2+}, Ni^{2+} & Cd^{2+}	200 Perkin Elmer
TDS, EC & Temp	Hanna Combo pH & EC, waterproof, H198129 meter.

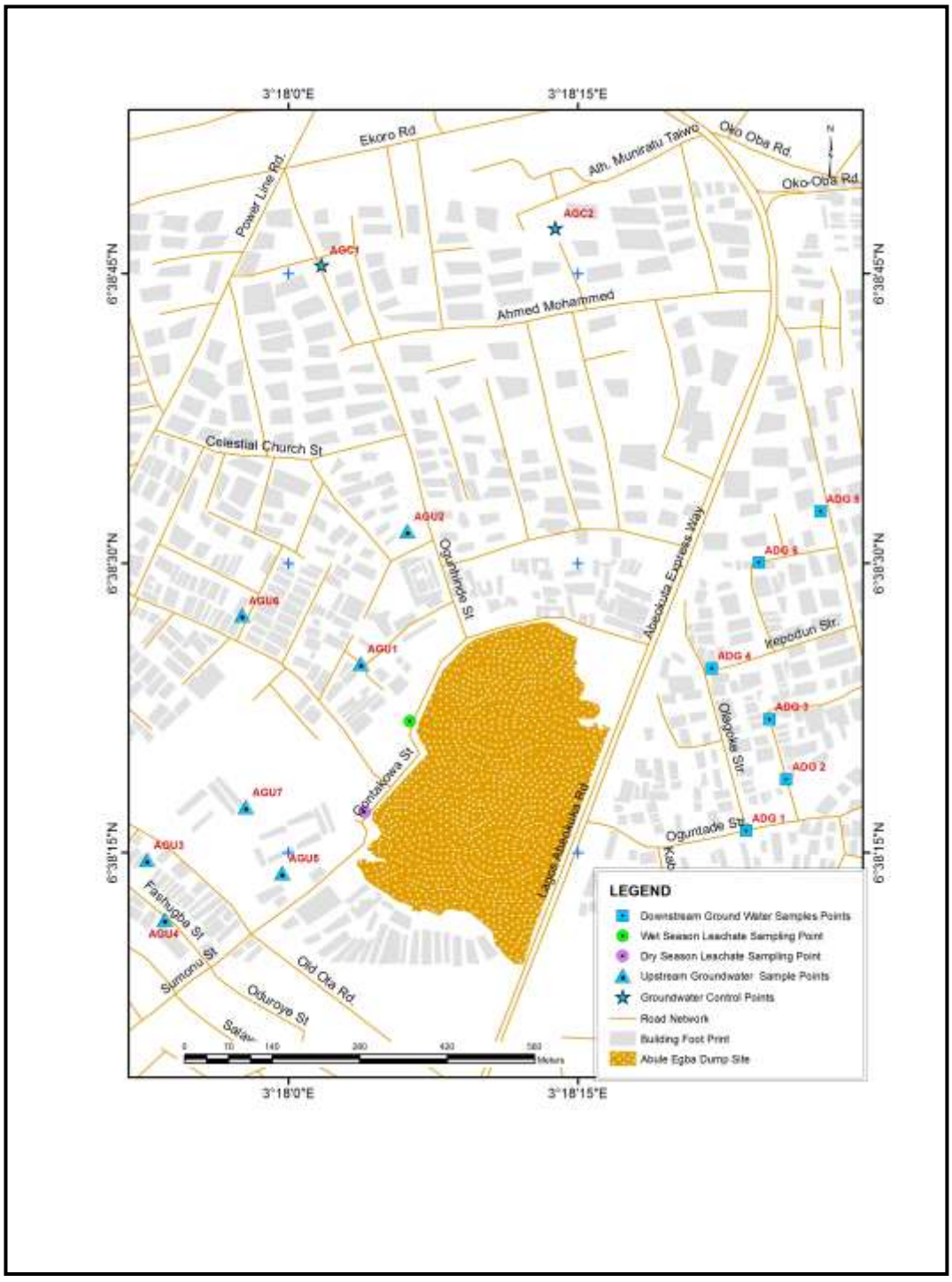


Fig 3.4: Leachate and Groundwater Sampling Locations at Abule-Egba

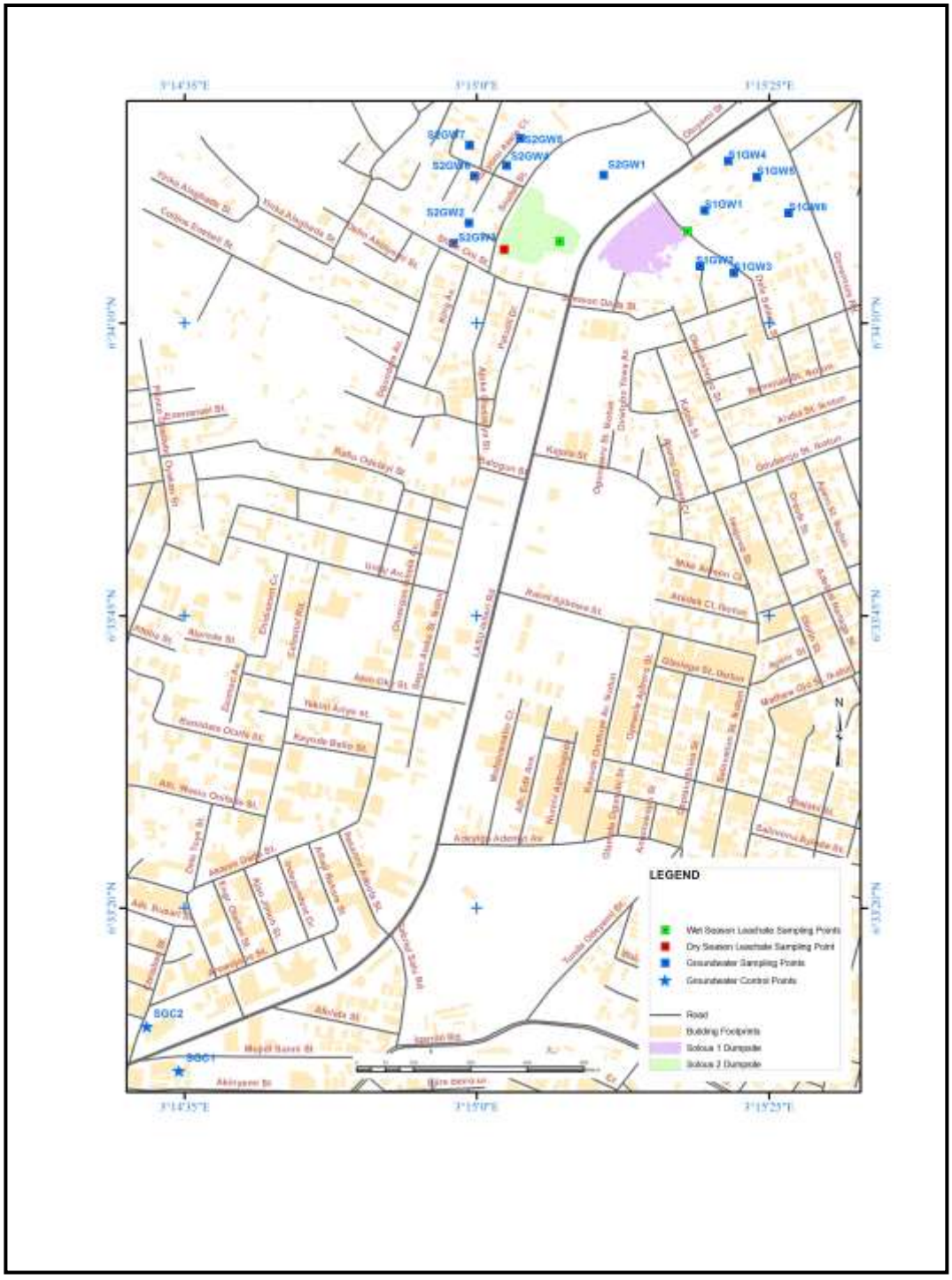


Fig 3.5: Leachate and Groundwater Sampling Locations at Solous, Igando

3.2.3 Leachate Characteristics and Pollution Index

The characterisation of the leachate generated at the dumpsites involved the interpretation of the results of the laboratory analyses of the chemical and heavy metals constituents. The characterisation of the leachate was based on literature. The computation of the Leachate Pollution Index (LPI) for the leachate generated at the dumpsites involved the adoption of the LPI model developed by Kumar and Alappat (2003).The LPI was developed to serve as a tool for evaluating the contamination potential of landfill leachate, and also to serve as policy making and public awareness tool about the threat of pollution from landfills and dumpsites (Kale, et al., 2010).

The LPI was computed using the equation (Kumar & Alappat 2005):

$$LPI = \frac{\sum_{i=1}^{m} WiPi}{\sum_{i=1}^{m} Wi} \quad \ldots\ldots (3.3)$$

As opposed to

$$LPI = \sum_{i=1}^{n} WiPi \quad \ldots\ldots (3.4)$$

Where LPI = the weighted additive leachate pollution index

Wi = the weight of the i^{th} pollutant variable

pi= the sub-index score of the i^{th} leachate pollutant variable

m= the number of leachate pollutant parameters for which data is available, which in this case, m <18, and $\sum^{m} Wi < 1$, because for the computation of LPI with all the 18 parameters available $\sum_{i=}^{n} Wi = 1$

For the computation of LPI Equation 3.3 can be used in lieu of equation 3.4 when the data for all 18 leachate pollutant variables included in the computation of the LPI is unavailable.

3.2.4 Variability and Spatial Patterns of Groundwater Quality

3.2.4.1 Variability of Groundwater Quality

Principal Component Analysis (PCA) was utilised for the determination of the groundwater quality parameters responsible for most of the variability in groundwater quality around the dumpsites and their control sites, PCA is a multivariate statistical tool. It is pattern recognition technique utilised in the explanation of variance of a large set of inter-correlated variables and their transformation into a smaller set of independent variables known as principal components (Lehman et al., 2005; Mohd- Nasir et. al 2011).

The "cumulative percent of variance accounted for" criterion was used as the basis for selecting the number of principal components to be retained for interpretation, with 80% adopted as the critical value. This criterion was used in order to ensure that group of components whose combination account for a minority of the variance in the data set is avoided.

The component scores was obtained through the use of the Statistical Package for the Social Sciences, Version 17 (SPSS Inc, 2008), based on the formula:

$S_{ik} = \Sigma D_{ij} L_{jk}$ …… ……..(3.5)

Where D_{ij} = Standardised value for observation i on variable j

L_{jk} = Loadings of variable j on component K ……..……. (3.6)

S_{ik} = Score of observation i on component K ……... (3.7)

3.2.4.2 Spatial Patterns of Groundwater Quality

Spatial interpolation was adopted for the establishment of the spatial patterns of the hydrochemical parameters responsible for most of the variance in groundwater quality around the dumpsites, and to enable the prediction and estimation of values of the concerned hydrochemical parameters at unsampled locations. Spatial interpolation is a procedure or technique utilised for the estimation of values of an environmental variable at unsampled locations using data from sampled locations within the same environment (Burrough & McDonell as cited in Li & Heap, 2005). The inverse distance weighting (IDW) method was used for the estimation of the values of the concerned hydrochemical parameters at the unsampled locations. Li and Heap (2008, p.23) noted that:

> The inverse distance weighting or inverse distance weighted (IDW) method estimates values of an attribute at unsampled points using a linear combination of values at sampled points weighted by an inverse function of distance from point of interest to the sampled points.

The weights can be derived using the formula:

$$\lambda i = \frac{1/d_i^p}{\sum_{i=1}^{n} 1/d_i^p} \quad \ldots\ldots\ldots\ldots\ldots\ldots \text{(3.8)}$$

Where d_i = distance between x_0 and x_i,

p= Power Parameter

n = No of sampled points used for estimation

The spatial interpolation of the relevant hydrochemical parameters was performed using the spatial analyst and 3D analyst tools of ArcGis 9.3.

3.2.5 Groundwater Contamination Index

For the assessment of the degree of groundwater quality contamination and the parameters responsible, the Contamination Index (C_d) developed by Bodis and Rapant (1995) was employed. The index was developed as a tool to examine the quality of water by computing the degree contamination. According to Majidamo et al., (2010), the C_d which is computed separately for each analysed groundwater sample is based on the sum of the factors derived from parameters whose concentration exceed the upper permissible level of WHO recommended value for drinking water quality.

The Groundwater Contamination Index was adopted in order to get a better insight into the quality and general hydrochemical conditions of groundwater in the vicinities of the dumpsites. The index was basically developed as a tool to examine the quality of water by computing the degree contamination.

The Contamination Index according to Backman et al., (1997) is defined as:

$$C_d = \sum_{i=1}^{n} C_{fi} \quad \text{..... (3.9)}$$

Where

$$C_{fi} = (C_{Ai}/C_{Ni}) \quad \text{..... (3.10)}$$

and

C_d = Contamination Index

C_{fi} = Contamination factor of the i-th component

C_{Ai} = Analytical value of the i-th component

C_{Ni} = Upper permissible concentration of the i-th component

The contamination Index is grouped into 3 categories, with $C_d < 1$ signifying low contamination, $1 < C_d < 3$ meaning medium contamination and $C_d > 3$ signifying high contamination. It should be noted however, that all groundwater samples that had no parameters exceeding the upper permissible limit of the World Health Organisation (WHO) Guideline for Drinking Water Quality were adjudged as having low contamination and therefore their C_d values were specified as 0.

CHAPTER FOUR

RESULTS AND DISCUSSION

4.1 GROUNDWATER FLOW MODELLING FOR ABULE-EGBA AND SOLOUS DUMPSITES ENVIRONS

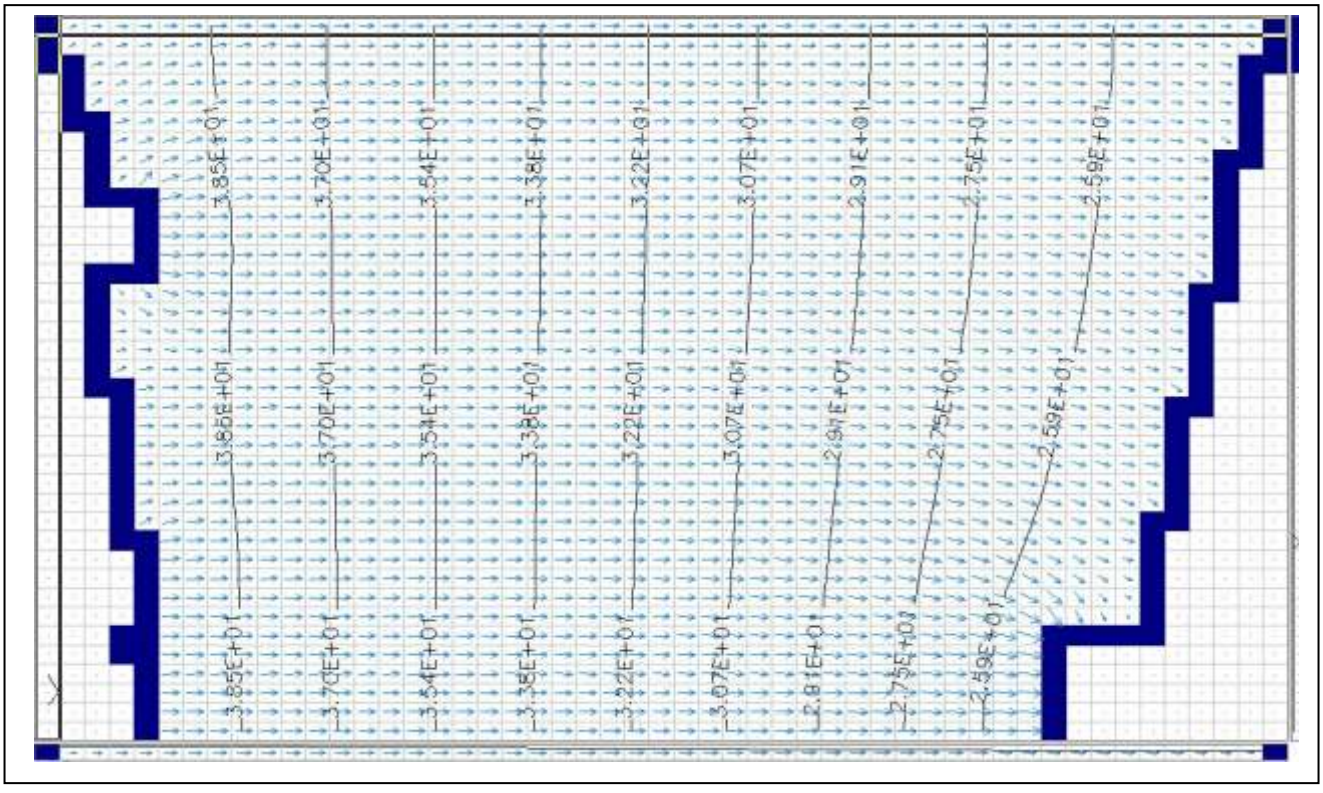
The results of the 2-Dimensional steady state groundwater flow modelling for the vicinities of the dumpsites are presented in Figures 4.1 and 4.2.

4.1.1 Abule-Egba Dumpsite Vicinity

As shown by the equipotential lines (isohypses) in Figure 4.1, the groundwater level elevation is higher in the western part of the model domain, indicating that groundwater flows from the west to the east. The direction of the velocity vectors which is perpendicular to the equipotential lines also follows the same west-east direction.

As shown by the velocity vectors, flows within the model domain is predominantly horizontal with minor indications of refraction (bending) at the upper and middle parts of the western boundary of the model, and the eastern boundary of the model. Flow lines (ie the velocity vectors) refract when they come in contact with geologic materials with differing hydraulic conductivities, making the prediction of the direction in which the flow lines will follow difficult. According to Weight (2008), the issue of refraction becomes a problem when the siting of a landfill is being considered or when an attempt is being made to trace the source of contamination. This is because depending on the geology and the groundwater flow system, a closer contamination source may erroneously be considered to be source of contamination, when in actual fact the contamination may be from a distant source.

Furthermore, the predominantly horizontal flow and the almost vertical equipotential lines also indicates that the model domain, and by extension the Abule-Egba Dumpsite vicinity, is neither a recharge nor a discharge area. The location of Abule-Egba Dumpsite along the horizontal flow path suggests that wells downgradient of the dumpsite are susceptible to contamination by the leachate generated at the dumpsite. However, some of the contaminants in the leachate are likely to be attenuated due to the depth of the unsaturated zone, which at the dumpsite ranges between 22 and 30 metres and the clays that make up part of the area's lithostratigraphy and general geology.



Scale: 1:1.019

Fig 4.1: Equipotential Lines and Velocity Vectors Depicting Groundwater Flow Direction and Pattern in the Vicinity of Abule-Egba Dumpsite

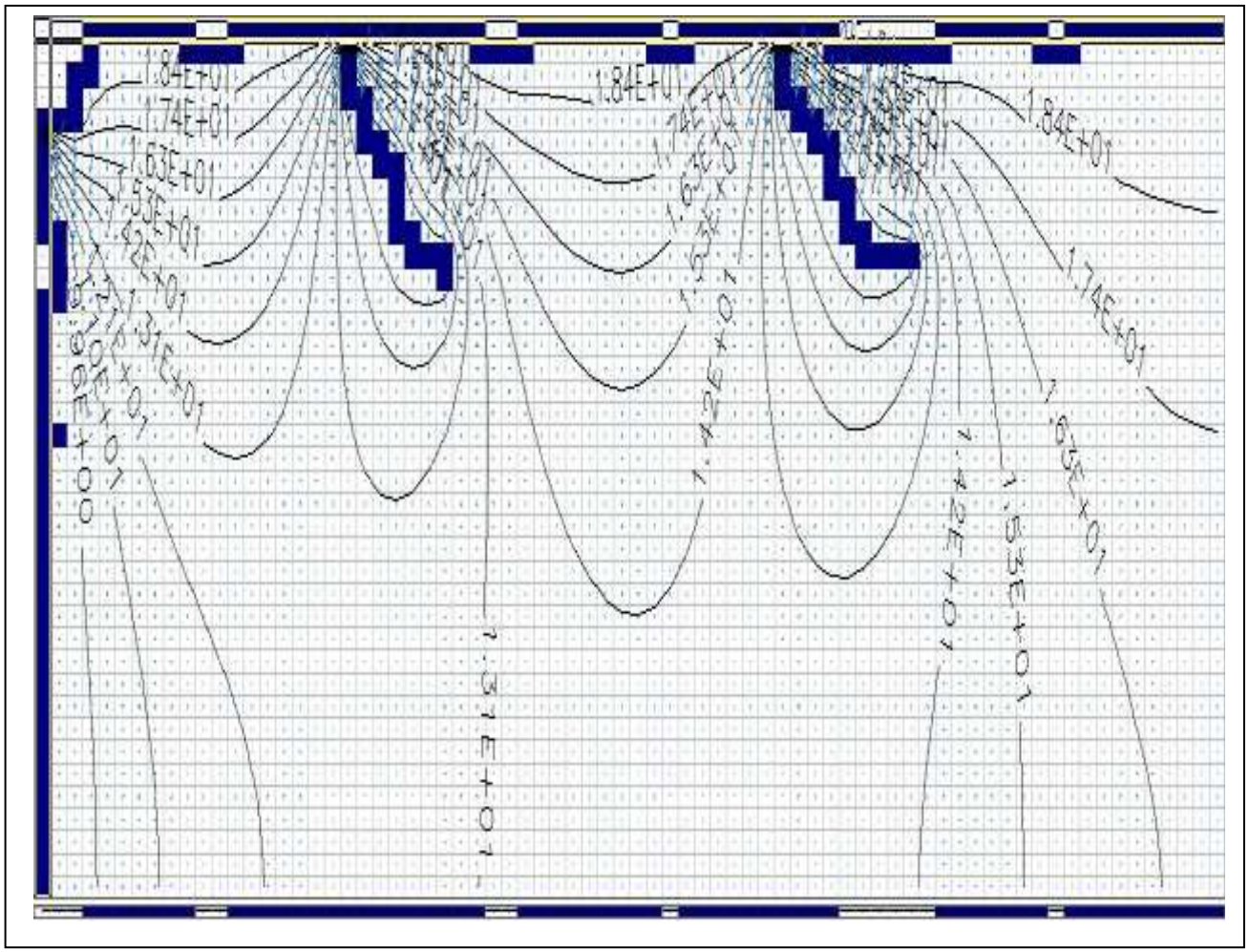
4.1.2 Solous Dumpsites

As shown in Figure 4.2, the results of the groundwater flow modelling shows that flow around the 10.72km^2 model domain (which from the extent of the model domain represents an intermediate groundwater flow system) is from the north to the south as indicated by the equipotential lines. The results also show that variation exists between the flow pattern in the north-western and the north-eastern extremes of the model domain. The flow pattern exhibited between the two tributaries is also different.

At the north-western part of the model domain the flow pattern is mostly radial and convergent, with minor indications of downward movement. Conversely flow is characterised by horizontal and vertical motions at the north-eastern part of the model domain. In the immediate vicinity of the tributaries of River Opomu, the radial and convergent flow pattern is characterised by both upward and downward motions, indicating an interaction between both surface and groundwater around the tributaries and the main river channel.

Between the two tributaries which are individual sub-catchments of the Opomu River and which are where the Solous 1 and Solous 2 Dumpsites are located, a local groundwater flow system is established. This is in view of the location of the dumpsites between the tributaries, and the fact that groundwater in the dumpsites vicinity is expected to travel a relatively short distance before discharging into either of the tributaries, which are located 1.74km apart. As explained by the United States Geological Survey [USGS] (2011), local groundwater flow systems characteristically involves short flow paths, usually less than 5km, with groundwater discharging into the nearest stream, wetland or lake. The existence of these two tributaries therefore seems to be responsible for the multiple flow directions around the dumpsites.

The implication of this groundwater flow pattern is that choosing suitable locations for the siting of water supply wells around Igando may be difficult, if contaminants are to be prevented from migrating into the wells. In addition, land uses that have high contamination potential should be avoided. As can also be seen in Figure 4.2 the remaining part of the model domain exhibits a divergent flow pattern, characterised by horizontal motions as depicted by the velocity vectors.



Scale: 1:1.014

Fig 4.2: Equipotential Lines and Velocity Vectors Depicting Groundwater Flow Direction and Pattern in the Vicinity of the Solous Dumpsites

4.2 LEACHATE CHARACTERISATION AND LEACHATE POLLUTION INDEX

4.2.1 Leachate Characterisation

The characteristics of leachate generated at landfills and dumpsites, are dependent on several factors which include the type of waste deposited (Organic-Inorganic, Degradable-Non degradable, Soluble-Insoluble), the degree of waste compaction, Landfill/Dumpsite conditions (pH, Temperature, Redox Potential, Moisture and age), climate, season, site hydrogeology, interaction of leachate with the environment, composition of soil cover, Landfill design and operation (Chen & Bowerman, 1974; Reinhart & Grosh, 1998).

In Table 4.1 the results of the in-situ measurement and Laboratory analyses of the leachate generated at the three dumpsites for the wet season and the Abule-Egba and Solous 2 dumpsites for the dry season is shown, while in Table 4.2 the concentration of the ions and heavy metal constituents of the leachates in unit of milliequivalent are shown.

During the wet season leachate effervescence was observed at Solous 2 Dumpsite at the point of emergence at the ground surface (Figure 4.3). The effervescence of landfill leachate is attributed to the presence of dissolved and entrained gases within the leachate. At Solous 2 Dumpsite therefore, there is clear indication that the waste deposited at that part of the Dumpsite is undergoing the methanogenic phase of waste degradation. Methanogenesis or the methanogenic phase of solid waste degradation is the main stage of landfill gas generation.

Fig 4.3: Emergence of Effervescent Leachate from the Solid Waste Matrix at Solous 2 Dumpsite during the Wet Season, 20th November, 2010

Table 4.1: Constituents and Concentration of Leachate Collected at Abule-Egba, Solous 1 and Solous 2 Dumpsites

Dumpsites						
Parameter	Abule-Egba		Solous 1		Solous 2	
pH	Wet Season	Dry Season	Wet Season	Dry Season	Wet Season	Dry Season
	7.61	5.16	7.36	NG	6.73	5.71
EC (μS/cm)	4960.0	206.0	435.0	NG	10020	850.0
SO_4^{2-} (mg/L)	0.79	25.1	0.08	NG	1.22	43.5
NO_3^- (mg/L)	5.7	16.2	1.33	NG	7.3	28.4
Cl^- (mg/L)	94.0	404.0	50.0	NG	167.0	1576.0
Zn^{2+} (mg/L)	4.4	0.24	3.2	NG	7.6	0.34
Fe^{2+} (mg/L)	4.9	0.11	4.4	NG	5.2	0.12
Pb^{2+} (mg/L)	0.02	ND	0.01	NG	0.03	ND
Ni^{2+} (mg/L)	0.001	0.102	0.002	NG	0.04	0.098
Cd^{2+} (mg/L)	0.1	ND	0.02	NG	0.11	0.07

Source: Fieldwork, November 2010; March 2011

***ND represents not detected**
*** NG represents leachate not generated**

Table 4.2: Constituents and Concentration of Leachate Collected at Abule-Egba, Solous 1 and Solous 2 Dumpsites in Milliequivalent Units

	Dumpsites					
Parameter	Abule-Egba		Solous 1		Solous 2	
	Wet Season (meq/l)	Dry Season (meq/l)	Wet Season (meq/l)	Dry Season	Wet Season (meq/l)	Dry Season (meq/l)
SO_4^{2-}	0.016	0.523	0.00167	NA	0.025	0.90567
NO_3^-	0.092	0.261	0.00215	NA	0.118	0.458
Cl^-	2.652	11.397	1.411	NA	4.711	44.459
Zn^{2+}	0.135	0.00734	0.098	NA	0.233	0.0104
Fe^{2+}	0.175	0.00394	0.158	NA	0.186	0.00429
Pb^{2+}	0.000193	ND	0.0000965	NA	0.000289	ND
Ni^{2+}	0.0000341	0.00347	0.0000681	NA	0.00136	0.003348
Cd^{2+}	0.000178	ND	0.000356	NA	0.00196	0.00124

***ND represents not detected**
*** NG represents leachate not generated**

As shown in Table 4.1 the wet season chemical constituents of the leachate generated at Abule-Egba Dumpsite was of the order Cl^-> NO_3^-> Fe^{2+} > SO_4^{2-} while the heavy metals was of the order Zn^{2+}> Cd^{2+}> Pb^{2+}> Ni^{2+}. For the dry season, the order was Cl^->SO_4^{2-} > NO_3^- > Fe^{2+}, for the chemical constituents and Zn^{2+} > Ni^{2+} for the heavy metals. Pb^{2+} and Cd^{2+} were undetected in the leachate. For Solous 1 Dumpsite the chemical and heavy metal constituents for the wet season leachate was of the order Cl^-> Fe^{2+} >NO_3^-> SO_4^{2-} for the chemical constituents and Zn^{2+}> Cd^{2+}> Pb^{2+}> Ni^{2+} for the heavy metals. During the dry season leachate was not generated at Solous 1 Dumpsite. The reason for this, apart from the characteristic dry conditions of the season, is that whatever moisture might have been available for the generation of leachate had been lost to evaporation and transpiration from the vegetal cover which had mostly taken over the dumpsite.

For Solous 2 Dumpsite, the order of the leachate constituents was of the order Cl^-> NO_3^-> Fe^{2+} > SO_4^{2-}, and Cl^-> SO_4^{2-}> NO_3^-> Fe^{2+} for the chemical parameters, for the wet and the dry season respectively, while for the heavy metals the order was Zn^{2+}> Cd^{2+}> Ni^{2+} >Pb^{2+} for the wet season, and Zn^{2+}> Ni^{2+} > Cd^{2+} for the dry season. Pb^{2+} was not detected in the dry season leachate.

The pH of the wet season leachate for the three dumpsites is indicative of methanogenic fermentation of waste at their points of collection while the dry season leachate for Abule-Egba and Solous 2 Dumpsite is indicative of acidogenic degradation of waste at their points of collection. As an indicator of the quantum of dissolved organic species or total ions concentration present in the leachate samples, the results of the electrical conductivity shows that the leachate generated at Solous 2 Dumpsite during the wet season is the most mineralised, while the leachate generated at Abule-Egba Dumpsite during the wet dry season is the least mineralised.

The SO_4^{2-} concentrations of the leachate generated at the dumpsites were low, with the wet season concentration being lower than the dry season. The relatively lower wet season concentration further suggests that the leachate is methanogenic. As stated by Kjelsen et al. (2002, p.308) "Sulfate concentrations are lower in the methanogenic phase due to microbial reduction of sulfate to sulfide." The generally low concentration of NO_3^- in the leachates generated at the three dumpsites is attributed to denitrification and anaerobic conditions at the dumpsites. According to Clement as cited in Akerman (2005), low concentration of NO_3^- in landfill leachate is attributed to the denitrification of leachates in the young phase, and the prevailing anaerobic conditions of landfills which does not allow oxidation of Ammonium, one of the compounds of Nitrate. The concentration of Cl^- in the leachates generated at the dumpsites may be due to some of the domestic wastes deposited at the dumpsites. According to Kale et al. (2008), some of the sources of Cl^- in landfill leachate include kitchen wastes.

The generally low concentration of heavy of metals in the leachate is attributed to the attenuation process of precipitation and sorption. These processes according to Bozkurt, et al., (1999), are responsible for the immobilisation and eventual low concentration of heavy metals in leachate. The Sorption process according to Bozkurt et al. (1999) is further enhanced by the presence of soil and organic matter in the deposited waste, especially at neutral to high pH values of methanogenic leachate such as those collected at Abule-Egba, Solous1 and Solous 2 Dumpsites during the wet season.

4.2.2 Leachate Pollution Index

The results of the computed LPI are presented in Figure 4.4. The LPI was computed using 7 parameters; pH, TDS, Cl^-, Zn^{2+}, Fe^{2+}, Pb^{2+} and Ni^{2+}. The LPI was computed for the wet and dry seasons for Abule-Egba and Solous 2 Dumpsites in order to determine if seasonal variation exists in the contamination potential of the leachates generated at these dumpsites. For Solous 1 LPI was not computed for the dry season due to the non-generation of leachate at the site.

As shown in Figure 4.4, the LPI value of the leachate generated at Abule-Egba Dumpsite was higher in the wet season. This is an indication that the leachate generated in the wet season has a higher potential for contamination. This assertion is supported by the relatively higher concentrations of TDS, Zn^{2+}, Fe^{2+}, and Pb^{2+} in the leachate generated during the wet season. Two factors are however held responsible for the low LPI values of the leachate generated at the dumpsite for both seasons. First is the low concentrations of heavy metals in the leachate (Kumar & Alappat, 2005), and second is the reduced volume of waste deposited at the dumpsite since its official closure in 2009. The exceedance of the WHO guideline values for drinking water quality by Zn^{2+}, Fe^{2+}, and Pb^{2+} in the wet season leachate sample suggests the need to monitor the type of waste deposited in order to protect the groundwater from heavy metal contamination.

For Solous 1 Dumpsite the low LPI value computed for the wet season is presumed to be due to the length of time that had elapsed since active tipping of waste stopped at the dumpsite. For Solous 2 the 20% decrease in the LPI value from the wet to the dry season is assumed to be due to the reduced availability of moisture required to leach out the soluble fraction of the waste deposited at the dumpsite.

In spite of the low LPI values however, individual contaminants like Zn^{2+}, Fe^{2+}, and Pb^{2+} for Solous 1 leachate, and TDS, Zn^{2+}, Fe^{2+}, and Pb^{2+} for Solous 2 leachate exceeded their stipulated WHO guideline values in water meant for drinking purposes.

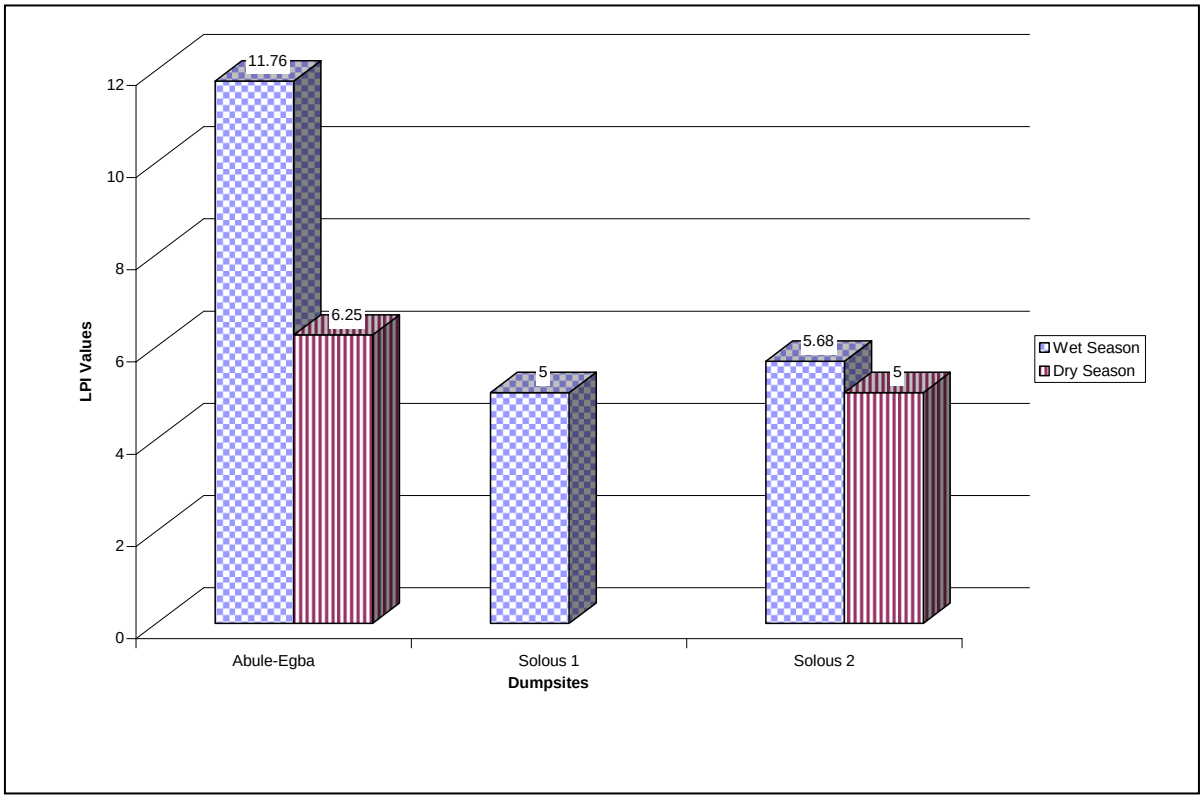


Fig 4.4: Leachate Pollution Index of Abule-Egba, Solous 1and Solous 2 Dumpsites for the Wet and Dry Seasons

4.3 VARIABILITY AND SPATIAL PATTERNS OF GROUNDWATER QUALITY

4.3.1 Statistical Summary of Groundwater Quality

The statistical summary of the chemical and heavy metal constituents of groundwater in the vicinity of Abule-Egba dumpsite, and the Solous 1 and 2 Dumpsites for the wet and the dry season are presented in Tables 4.3 and 4.4, and Tables 4.5 and 4.6 respectively.

Table 4.3: Statistical Summary of the Chemical and Heavy Metals Analyses of Groundwater Quality in the Vicinity of Abule-Egba Dumpsite for the Wet and Dry seasons

		Wet Season							Dry Season						
Upgradient															
Physico-Chemical Parameters															
Constituent	Unit	Min	Max	Mean	Std Deviation	Std Error	Variance	Range	Min	Max	Mean	Std Deviation	Std Error	Variance	Range
pH		4.93	5.53	5.12	0.21	0.80	0.05	0.60	4.55	5.20	4.89	0.25	0.95	0.63	0.65
EC	µS/cm	27.40	609.00	182.57	201.56	76.18	40627.91	581.60	46.20	367.00	140.97	126.12	47.66	1509.73	320.80
TDS	mg/L	21.30	450.00	124.26	148.40	56.08	22013.34	428.70	22.90	255.00	81.96	85.29	32.24	7275.23	232.10
SO_4^{2-}	mg/L	-	-	-	-	-	-	-	1.00	15.00	4.13	5.10	1.93	26.00	14.00
NO^{3-}	mg/L	0.40	1.94	0.93	0.54	0.20	0.29	1.54	0.07	6.10	1.55	2.34	0.89	5.49	6.03
Cl^-	mg/L	30	50	38.57	6.90	2.61	47.62	20	16.00	148.00	42.29	46.92	17.74	2201.91	132.00
Fe^{2+}	mg/L	0.09	1.10	0.50	0.36	0.14	0.13	1.01	0.23	0.82	0.44	0.20	0.07	0.39	0.59
Heavy Metals															
Constituent	Unit	Min	Max	Mean	Std Deviation	Std Error	Variance	Range	Min	Max	Mean	Std Deviation	Std Error	Variance	Range
Zn^{2+}	mg/L	1.22	3.90	1.70	0.97	0.40	0.95	2.68	0.12	2.11	0.47	0.72	0.27	0.52	1.99
Pb^{2+}	mg/L	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Ni^{2+}	mg/L	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Cd^{2+}	mg/L	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Downgradient															
Physico-Chemical Parameters															
Constituent	Unit	Min	Max	Mean	Std Deviation	Std Error	Variance	Range	Min	Max	Mean	Std Deviation	Std Error	Variance	Range
pH		5.73	7.22	6.22	0.57	0.23	0.32	1.49	3.81	6.67	5.02	1.07	0.44	1.15	2.86
EC	µS/cm	122.00	606.00	389.83	188.76	77.06	35631.37	484.00	132.00	1039.00	449.67	339.83	138.73	115483.87	907.00
TDS	mg/L	81.30	301.00	197.55	90.75	37.05	8235.54	219.70	64.00	516.00	223.33	169.14	69.05	28608.67	452.00
SO_4^{2-}	mg/L	0.49	0.82	0.63	0.14	0.06	0.02	0.33	1.00	19.00	6.33	6.86	2.80	47.07	18.00
NO^{3-}	mg/L	1.00	4.87	3.10	1.50	0.61	2.23	3.87	1.10	2.30	1.49	0.42	0.17	0.17	1.20
Cl^-	mg/L	52.00	120.00	82.00	29.37	11.99	862.40	68.00	28.00	188.00	80.67	57.43	23.45	3298.67	160.00
Fe^{2+}	mg/L	0.09	0.99	0.70	0.32	0.13	0.10	0.90	0.08	0.18	0.13	0.04	0.01	0.001	0.10
Heavy Metal Parameter															
Constituent	Unit	Min	Max	Mean	Std Deviation	Std Error	Variance	Range	Min	Max	Mean	Std Deviation	Std Error	Variance	Range
Zn^{2+}	mg/L	0.42	0.82	0.61	0.14	0.06	0.02	0.40	0.00	0.40	0.13	0.17	0.07	0.03	0.40
Pb^{2+}	mg/L	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Ni^{2+}	mg/L	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Cd^{2+}	mg/L	-	-	-	-	-	-	-	-	-	-	-	-	-	-

Source: Fieldwork, November 2010; March 2011

Table 4.4: Statistical Summary of the Chemical and Heavy Metals Analyses of Groundwater Quality of the Control Sites of the Abule-Egba Dumpsite Vicinity for the Wet and Dry seasons

		Wet Season							Dry Season						
Physico-Chemical Parameters															
Constituent	Unit	Min	Max	Mean	Std Deviation	Std Error	Variance	Range	Min	Max	Mean	Std Deviation	Std Error	Variance	Range
pH		5.12	5.15	5.14	0.02	0.15	0.00	0.30	5.13	5.66	5.40	0.40	0.27	0.14	0.53
EC	µS/cm	41.10	280.00	160.55	168.93	119.45	28536.61	238.90	41.60	364.00	202.80	227.97	161.20	51970.88	322.40
TDS	mg/L	26.30	140.00	83.15	80.40	56.85	6463.85	113.70	20.90	167.00	93.95	103.31	73.05	10672.61	146.10
SO_4^{2-}	mg/L	-	-	-	-	-	-	-	2.00	7.10	4.55	3.61	2.55	13.01	5.10
NO^{3-}	mg/L	0.50	2.50	1.50	1.41	1.00	2.00	2.00	0.20	3.80	2.00	2.55	1.80	6.48	3.60
Cl^-	mg/L	30.00	70.00	50.00	28.28	20.00	800.00	40.00	16.00	64.00	40.00	33.94	24.00	1152.00	48.00
Fe^{2+}	mg/L	0.27	0.66	0.47	0.26	0.20	0.76	0.39	0.11	0.29	0.20	0.13	0.09	0.02	0.18
Heavy Metal Parameters															
Constituent	Unit	Min	Max	Mean	Std Deviation	Std Error	Variance	Range	Min	Max	Mean	Std Deviation	Std Error	Variance	Range
Zn^{2+}	mg/L	1.20	4.12	2.66	2.06	1.46	4.26	2.92	0.24	2.01	1.13	1.25	0.89	1.57	1.77
Pb^{2+}	mg/L	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Ni^{2+}	mg/L	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Cd^{2+}	mg/L	-	-	-	-	-	-	-	-	-	-	-	-	-	-

Source: Fieldwork, November 2010; March 2011

Table 4.5: Statistical Summary of the Chemical and Heavy Metals Analyses of Groundwater Quality in the Vicinity of Solous1 and Solous 2 Dumpsites for the Wet and Dry seasons

	Wet Season							Dry Season						
Solous 1														
Physico-Chemical Parameters														
Constituent	Unit	Min	Max	Mean	Std Deviation	Std Error	Variance	Range	Min	Max	Mean	Std Deviation	Std Error	Variance	Range
pH		6.33	7.24	6.82	0.35	0.14	0.13	0.91	5.37	7.91	6.71	0.10	0.40	0.97	2.54
EC	µS/cm	26.0	172.00	67.60	56.52	23.07	3194.13	146.0	26.40	196.00	95.28	71.17	29.05	5064.91	169.60
TDS	mg/L	13.0	88.1	34.30	29.30	858.24	717.72	75.10	12.80	96.40	47.22	35.18	14.40	1237.42	83.60
SO_4^{2-}	mg/L	-	-	-	-	-	-	-	2.00	55.2	13.8	21.45	8.76	459.99	53.2
NO^{3-}	mg/L	0.05	1.31	0.81	0.48	0.20	0.23	1.26	0.04	33.70	9.06	13.60	5.55	185.06	33.66
Cl^-	mg/L	30.0	60.0	41.67	11.55	4.77	136.67	30.00	16.00	100.0	38.70	31.20	12.86	996.27	84.00
Fe^{2+}	mg/L	0.05	1.30	0.62	0.41	0.17	0.17	1.25	0.12	0.72	0.46	0.21	0.09	0.38	0.60
Heavy Metals														
Constituent	Unit	Min	Max	Mean	Std Deviation	Std Error	Variance	Range	Min	Max	Mean	Std Deviation	Std Error	Variance	Range
Zn^{2+}	mg/L	1.10	4.50	2.09	1.40	0.57	1.97	3.40	0.22	0.39	0.30	0.06	0.30	0.004	0.17
Pb^{2+}	mg/L	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Ni^{2+}	mg/L	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Cd^{2+}	mg/L	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Solous 2														
Physico-Chemical Parameters														
Constituent	Unit	Min	Max	Mean	Std Deviation	Std Error	Variance	Range	Min	Max	Mean	Std Deviation	Std Error	Variance	Range
pH		6.37	7.55	6.90	0.40	0.15	0.16	1.18	5.72	7.44	6.60	0.77	0.29	0.59	1.72
EC	µS/cm	82.50	713.0	352.07	221.04	83.55	48858.70	630.50	201.00	690.00	375.43	184.15	69.60	33910.62	489.00
TDS	mg/L	41.10	488.0	207.30	148.55	56.14	22065.70	446.90	101.00	382.00	209.30	107.01	40.44	11450.57	281.00
SO_4^{2-}	mg/L	0.01	0.52	0.11	0.20	0.07	0.04	0.51	2.50	105.00	30.50	35.70	13.50	1274.44	102.50
NO^{3-}	mg/L	0.53	9.09	4.41	3.56	1.34	12.64	8.56	0.36	86.40	24.26	29.90	11.30	894.12	86.04
Cl^-	mg/L	30.00	470.0	178.43	171.51	64.82	29414.62	440.0	28.00	548.00	293.14	170.09	64.29	28931.81	520.00
Fe^{2+}	mg/L	0.21	2.20	0.87	0.70	0.26	0.49	1.99	0.14	0.48	0.22	0.11	0.45	0.14	0.34
Heavy Metal Parameter														
Constituent	Unit	Min	Max	Mean	Std Deviation	Std Error	Variance	Range	Min	Max	Mean	Std Deviation	Std Error	Variance	Range
Zn^{2+}	mg/L	1.10	10.30	4.42	3.23	1.22	10.45	9.20	0.18	0.38	0.30	0.76	0.29	0.01	0.20
Pb^{2+}	mg/L	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Ni^{2+}	mg/L	-	-	-	-	-	-	-	0.001	0.002	-	-	-	-	0.001
Cd^{2+}	mg/L	-	-	-	-	-	-	-	-	-	-	-	-	-	-

Source: Fieldwork, November 2010; March 2011

Table 4.6: Statistical Summary of the Chemical and Heavy Metals Analyses of Groundwater Quality of the Control Sites of the Solous 1 and Solous 2 Dumpsites Vicinity for the Wet and Dry seasons

		Wet Season							Dry Season						
Physico-Chemical Parameters															
Constituent	Unit	Min	Max	Mean	Std Deviation	Std Error	Variance	Range	Min	Max	Mean	Std Deviation	Std Error	Variance	Range
pH		6.40	6.44	6.42	0.03	0.02	0.001	0.04	5.33	5.38	5.36	0.35	0.03	0.001	0.05
EC	µS/cm	59.00	102.00	80.50	30.41	21.50	924.50	43.00	62.10	97.50	79.80	25.03	17.70	626.58	35.40
TDS	mg/L	28.00	66.50	52.50	19.80	14.00	392.00	28.00	37.70	48.10	42.90	7.35	5.20	54.08	10.40
SO_4^{2-}	mg/L	-	-	-	-	-	-	-	3.40	8.10	5.75	3.32	2.35	11.05	4.70
NO^{3-}	mg/L	0.43	3.90	2.17	2.45	1.74	6.02	3.47	0.56	2.14	1.35	1.18	0.79	1.25	1.58
Cl^-	mg/L	30.00	290.00	160.00	183.85	130.00	3380.00	260.00	24.00	112.00	68.00	3.32	44.00	3872.00	88.00
Fe^{2+}	mg/L	0.30	0.53	0.47	0.16	0.12	0.30	0.23	0.42	0.47	0.45	0.36	0.25	0.001	0.05
Heavy Metal Parameters															
Constituent	Unit	Min	Max	Mean	Std Deviation	Std Error	Variance	Range	Min	Max	Mean	Std Deviation	Std Error	Variance	Range
Zn^{2+}	mg/L	1.32	6.60	3.96	3.73	2.64	13.94	5.28	0.17	0.19	0.18	0.01	0.01	0.00	0.02
Pb^{2+}	mg/L	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Ni^{2+}	mg/L	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Cd^{2+}	mg/L	-	-	-	-	-	-	-	-	-	-	-	-	-	-

Source: Fieldwork, November 2010; March 2011

4.3.2 Variability and Spatial pattern of Groundwater Quality around the Dumpsites

4.3.2.1 Variability of Groundwater Quality

The result of the PCA used to determine the hydrochemical parameters responsible for most of the variability in groundwater quality in the vicinity of Abule-Egba Dumpsite and the Solous Dumpsites for the wet and dry seasons are presented in Tables 4.7, 4.8, 4.9 and 4.10 respectively. On the Tables the components with the diamond symbol (♦) represents the three components retained for interpretation after applying the "cumulative percent of variance accounted for" criterion (80 percent critical value) as a yardstick. The asterisked component loadings in the three retained components represent hydrochemical variables with high loadings and significance within each component.

4.3.2.1.1 Abule-Egba Dumpsite Vicinity

For the wet season (Table 4.7), the first three components accounted for 84.91% of the total variance in groundwater quality. Most of the variance is contained in Component 1 (52.98%), with high loadings of NO_3^- and Cl^-. Component 2 accounted for 21.20% of the total variance and has a loading of Zn^{2+}, while component 3 which explained 10.73% of the total variance had a high loading of TDS. The high loading of NO_3^- and Cl^- in the first component is associated with the influence of the dumpsite on groundwater quality. The high loading of Zn^{2+} in the second component may be due to the mineral dissolution or the disposal of waste from which Zn^{2+} may be leached, while the high loading of TDS in the component is attributed to a combination of the leaching of contaminants from the waste deposited at the dumpsite and the dissolution of minerals that make up the underlying aquifer.

The high loadings of these four hydrochemical variables suggest that they are responsible for a greater portion of the variability recorded in the quality of groundwater around the dumpsite during the wet season. Figures 4.5, 4.6, 4.7 and 4.8 represent the spatial patterns of the variability in the concentration of NO_3^- , Cl^-, TDS and Zn^{2+} in the groundwater of the Abule-Egba Dumpsite vicinity.

For the dry season (Table 4.8), 81.43% of the total variance in the groundwater quality was accounted for by the first three components. Component 1 with high loadings of TDS and Cl^- accounted for the highest proportion of the total variance at 44.29%. The second and third components with high loadings of NO_3^- and Fe^{2+} accounted for 20.36% and 16.78% of the total variance respectively.

The high loadings of TDS and Cl^- are attributed to the dissolution of aquifer minerals and leachate from the dumpsite. The high loading of NO_3^- in the second component suggests the influence of wastewater such as leachate, while the high loading of Fe^{2+} in the third component is indicative of the dissolution of some of the Fe^{2+} minerals present in the aquifers, and which comprises part of the area's geology. Figures 4.9, 4.10, 4.11 and 4.12 represent the spatial patterns of the variability in the concentration of TDS, NO_3^- , Cl^-, and Fe^{2+} in the groundwater of the Abule-Egba Dumpsite vicinity for the dry season.

Table 4.7: Component Loadings after Varimax Rotation of Hydrochemical Parameters for Abule-Egba Dumpsite Vicinity for the Wet Season

Variable	Component Loadings						
	Comp 1♦	Comp 2♦	Comp 3♦	Comp 4	Comp 5	Comp 6	Comp 7
pH	0.488	-0.210	0.105	-0.02	0.808	0.231	0.003
TDS	0.352	-0.064	*0.929	-0.052	0.071	0.037	-0.003
SO_4^{2-}	0.369	-0.401	0.078	0.243	0.411	0.685	0.003
NO_3^-	*0.842	-0.063	0.417	-0.020	0.233	0.210	0.124
Cl^-	*0.920	-0.045	0.233	0.058	0.276	0.106	-0.083
Zn^{2+}	-0.043	*0.968	-0.058	-0.142	-0.135	-0.141	-0.001
Fe^{2+}	0.019	-0.138	-0.045	0.985	0.001	0.090	-0.001
Eigenvalues	3.709	1.484	0.755	0.645	0.229	0.157	0.020
Percent of Total Variance Explained	52.98	21.20	10.78	9.22	3.28	2.24	0.29

♦represents components retained for interpretation
***represents high component loadings**

Table 4.8: Component Loadings after Varimax Rotation of Hydrochemical Parameters for Abule-Egba Dumpsite Vicinity for the Dry Season

Variable	Component Loadings						
	Comp 1♦	Comp 2♦	Comp 3♦	Comp 4	Comp 5	Comp 6	Comp 7
pH	0.296	-0.076	0.047	0.937	0.137	0.082	0.001
TDS	*0.895	0.072	-0.235	0.172	-0.167	-0.243	-0.149
SO_4^{2-}	0.488	0.489	0.015	0.160	-0.037	0.704	0.002
NO^{3-}	0.125	*0.967	0.021	-0.087	-0.132	0.155	-0.002
Cl^-	*0.927	0.169	-0.097	0.259	-0.109	0.089	-0.126
Zn^{2+}	-0.160	-0.125	0.070	0.121	0.969	-0.023	-0.002
Fe^{2+}	-0.179	0.022	*0.980	0.041	0.067	0.002	-0.002
Eigenvalues	3.101	1.425	1.175	0.708	0.352	0.206	0.033
Percent of Total Variance Explained	44.29	20.36	16.78	10.11	5.03	2.95	0.48

♦represents components retained for interpretation
***represents high component loadings**

4.3.2.1.2 Solous Dumpsites Vicinity

For the wet season (Table 4.9), the first three components accounted for 84.86% of the total variance in the quality of groundwater in the Solous Dumpsites vicinity. The highest proportion of the total variance is accounted for by component 1 with 53.91% of the total variance and high loadings of NO_3^-, Cl^- and Zn^{2+}. Component 2 with a high loading of pH accounted for 17.60% of the total variance while component 3 with a high loading of Fe^{2+} accounted for 13.35% of the total variance.

In component 1 the high loading of NO_3^- and Cl^- is suggestive of the influence of leachate while the loading of Zn^{2+} may be due to the leaching of Zinc based waste deposits. The high loading of pH in component 2 is an indication that leachate from the dumpsite is likely to contribute to the variability recorded in the quality of groundwater around the dumpsite. The high loading of Fe^{2+} in component 3 is a reflection of the mineral composition of the sub-surface geology of the area which is made up of an appreciable proportion of clay and laterite. In Figures 4.13, 4.14, 4.15, 4.16 and 4.17 the spatial patterns of the variability in the concentration of NO_3^-, Cl^-, Zn^{2+}, pH and Fe^{2+} in the groundwater of the Solous Dumpsites vicinity for the wet season is shown.

For the dry season (Table 4.10), 81.82% of the total variance in the groundwater quality was accounted for by the first three components. Component 1 with high loadings of $SO_4^{2-}$and NO_3^- accounted for 39.35% of the total variance, while component 2 with a high loading of TDS accounted for 29.46% of the total variance. Component 3 accounted for 13.01% of the total variance, with a high loading of Zn^{2+}. In Figures 4.18, 4.19, 4.20 and 4.21 the spatial patterns of the variability in the concentration of SO_4^{2-}, NO_3^-, TDS and Zn^{2+} for the dry season is presented. The high loading of SO_4^{2-} and NO_3^- in component 1 is associated with anthropogenic activities

such as leachate arising from the disposal of municipal solid waste, while the high loading of Fe^{2+} is associated with the influence of the area's geology on the quality of groundwater. The high loading of TDS in component 2 is ascribed to the influence of leachate and the mineral dissolution of the materials making up the aquifer and area's geology. The high loading of Zn^{2+} in the third component may also be due to the mineral dissolution of the earth's crust and waste disposal activities.

Table 4.9: Component Loadings after Varimax Rotation of Hydrochemical Parameters for the Solous Dumpsites Vicinity for the Wet Season

Variable	Component Loadings						
	Comp 1♦	Comp 2♦	Comp 3♦	Comp 4	Comp 5	Comp 6	Comp 7
pH	0.48	*0.974	0.212	0.029	0.049	0.041	-0.006
TDS	0.120	0.027	-0.025	0.988	0.095	0.012	0.000
SO_4^{2-}	0.443	0.064	0.225	0.143	0.851	0.073	0.002
NO_3^-	*0.953	0.117	0.129	0.135	0.160	0.002	-0.129
Cl^-	*0.919	-0.096	0.188	0.076	0.285	0.027	0.155
Zn^{2+}	*0.742	0.219	0.191	0.053	0.359	0.482	0.002
Fe^{2+}	0.234	0.255	*0.918	-0.033	0.187	0.046	0.002
Eigenvalues	3.774	1.232	0.934	0.512	0.368	0.142	0.038
Percent of Total Variance Explained	53.91	17.60	13.35	7.32	5.26	2.02	0.54

♦represents components retained for interpretation
***represents high component loadings**

Table 4.10: Component Loadings after Varimax Rotation of Hydrochemical Parameters for the Solous Dumpsites Vicinity for the Dry Season

Variable	Component Loadings						
	Comp 1♦	Comp 2♦	Comp 3♦	Comp 4	Comp 5	Comp 6	Comp 7
pH	0.227	-0.050	0.237	0.915	0.221	0.064	0.000
TDS	-0.020	*0.921	-0.112	-0.055	-0.235	0.284	0.000
SO_4^{2-}	*0.977	0.010	0.059	0.115	-0.108	0.120	-0.043
NO_3^-	*0.956	0.001	0.125	0.149	-0.113	0.180	-0.047
Cl^-	0.294	0.356	0.006	0.075	-0.192	0.863	0.000
Zn^{2+}	0.124	-0.094	*0.968	0.199	0.011	0.002	0.001
Fe^{2+}	-0.201	-0.258	0.012	0.235	0.898	-0.178	0.000
Eigenvalues	2.755	2.062	0.911	0.737	0.288	0.244	0.004
Percent of Total Variance Explained	39.35	29.46	13.01	10.53	4.11	3.48	0.06

♦represents components retained for interpretation
***represents high component loadings**

4.3.2.2 Spatial Pattern of Groundwater Quality

The results of the spatial patterns established for hydrochemical parameters with high loadings from the Principal Component Analysis for the wet and dry seasons for Abule-Egba Dumpsite vicinity are presented in Figures 4.5 to 4.12, while the results for both seasons for the Solous Dumpsites environs are presented in Figures 4.13 to 4.21.

4.3.2.2.1 Spatial Pattern of Groundwater Quality around Abule-Egba Dumpsite for the Wet Season

4.3.2.2.1.1 Nitrate

Figure 4.5 represents the spatial distribution and pattern of NO_3^- in groundwater in the vicinity of Abule-Egba Dumpsite. For location AGU1 (123.50m upgradient of the dumpsite) and an area of

7,209.41m^2 around it, NO_3^- concentration is estimated to vary between 1.89 and 2.39mg/L. For location AGU2 (203.36m upgradient of the dumpsite) and an area of 30,762.07m^2 around it, the concentration is estimated to range between 0.40 and 0.90mg/L.

For locations AGU3 (335.47m upgradient of the dumpsite), AGU6 (266.48m upgradient) and an areal extent of 481,562.77m^2, NO_3^- is estimated to range between 0.90 and 1.39mg/L. For locations AGU4 (319.14m upgradient of the dumpsite), AGU5 (120.91m upgradient of the dumpsite) and an area of 99,132.97m around them, NO_3^- concentration is estimated to vary between 0.40 and 0.90mg/L. For location AGU7 (198.17m upgradient of the dumpsite) and an area of 672.07m^2 around it, the concentration is estimated to vary between 0.90 and 1.39mg/L.

For the downgradient locations, locations ADG1 (212.87m downgradient of the dumpsite), ADG2 (218.31m downgradient of the dumpsite) and an area of 105,789.4207m^2 around them is estimated to have a NO_3^- concentration ranging between 3.38 and 3.88mg/L. For location ADG3 (196.54m downgradient of the dumpsite) and an area of 3,207.59m^2 around it, and location ADG4 (150.14m downgradient of the dumpsite) and an area of 6,720.64m^2 around it, the range of NO_3^- concentration is estimated to be the highest at values between 4.37 and 4.87mg/L.

For location ADG5 (395.44m downgradient of the dumpsite) and an areal extent of 260, 967.66m^2 downgradient of the dumpsite, the concentration is estimated to be between 2.39 and 2.88mg/L. For location ADG6 (266.48m downgradient of the dumpsite) and an area of 11, 516.70m^2 around it, NO_3^- concentration of groundwater is estimated to vary between 1.39 and 1.89mg/L.

At the control sites, location AGC1 (643.05m away from the dumpsite) and an area of 90,483.21m^2 around it, NO_3^- concentration is estimated to range between 0.40 and 0.90mg/L. For

location AGC2 (626.74m away from the dumpsite) and an area of 34, 580.37m^2 around it, the concentration of NO_3^- is estimated to vary between 2.39 and 2.88mg/L.

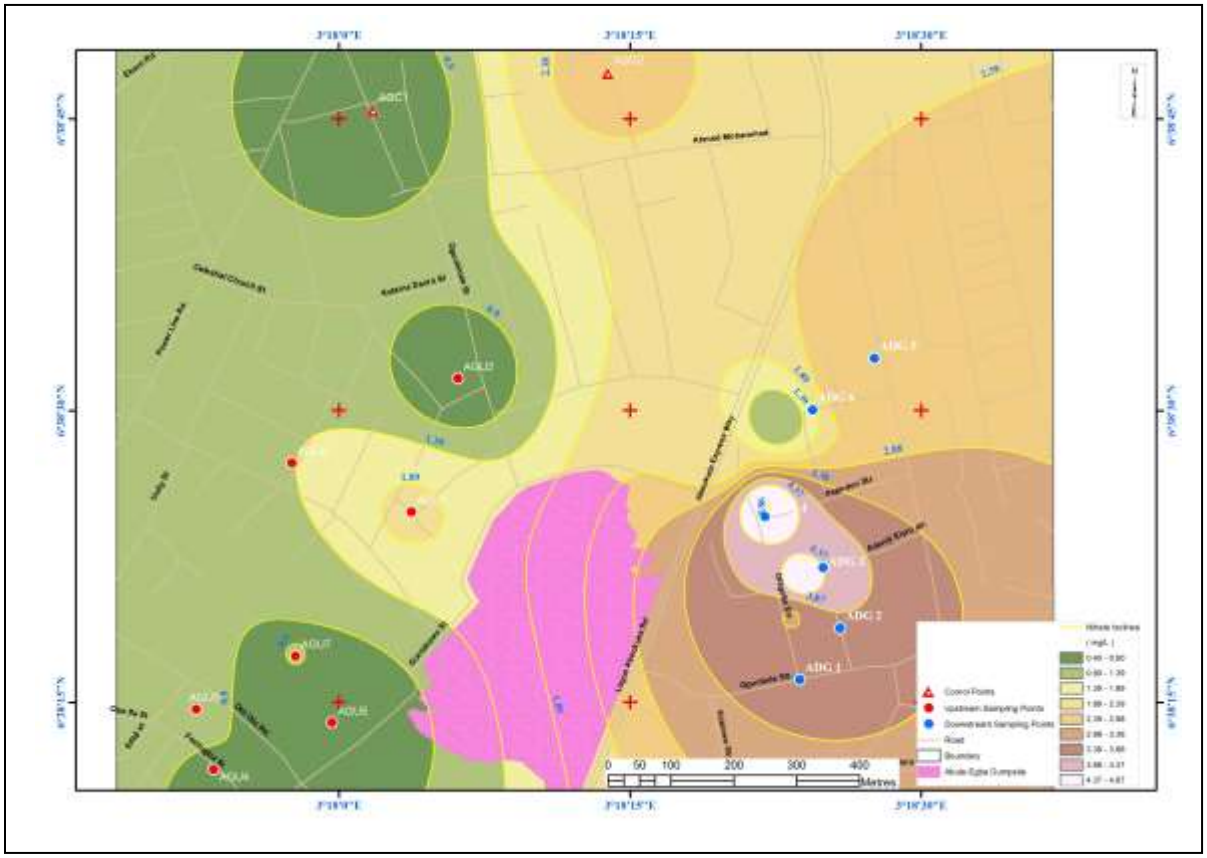


Fig 4.5: Spatial Pattern of Variability in Nitrate Concentration of Groundwater around Abule-Egba Dumpsite for the Wet Season

4.3.2.2.1.2 Chloride

As shown in figure 4.6, the Cl^- concentration of groundwater downgradient of the dumpsite is generally higher compared with upgradient locations. For locations AGU1, AGU3, AGU4, AGU6 and an areal extent of 505,188.10m^2 upgradient of the dumpsite, chloride concentration is estimated to range between 40 and 50mg/L. For locations AGU7, AGU5 and an area of 77,560.09m^2 around them, Cl^- is estimated to vary between 30 and 40mg/L. For locations AGU2,

AGC1 (one of the control sites) and an area of 230,688.40m^2 around them, the concentration is estimated to range 30 and 40mg/L.

For location ADG1 and an area of 5,255.17m^2 around it, and location ADG4 and an area of 4, 542.36m^2 around it, Cl^- concentration is estimated to vary between 110 and 120mg/L. For locations ADG5, ADG6, AGC2 (one of the control sites) and

areal extent of 511,774.86m^2, Cl^- concentration of groundwater is estimated to range between 60 and 70mg/L.

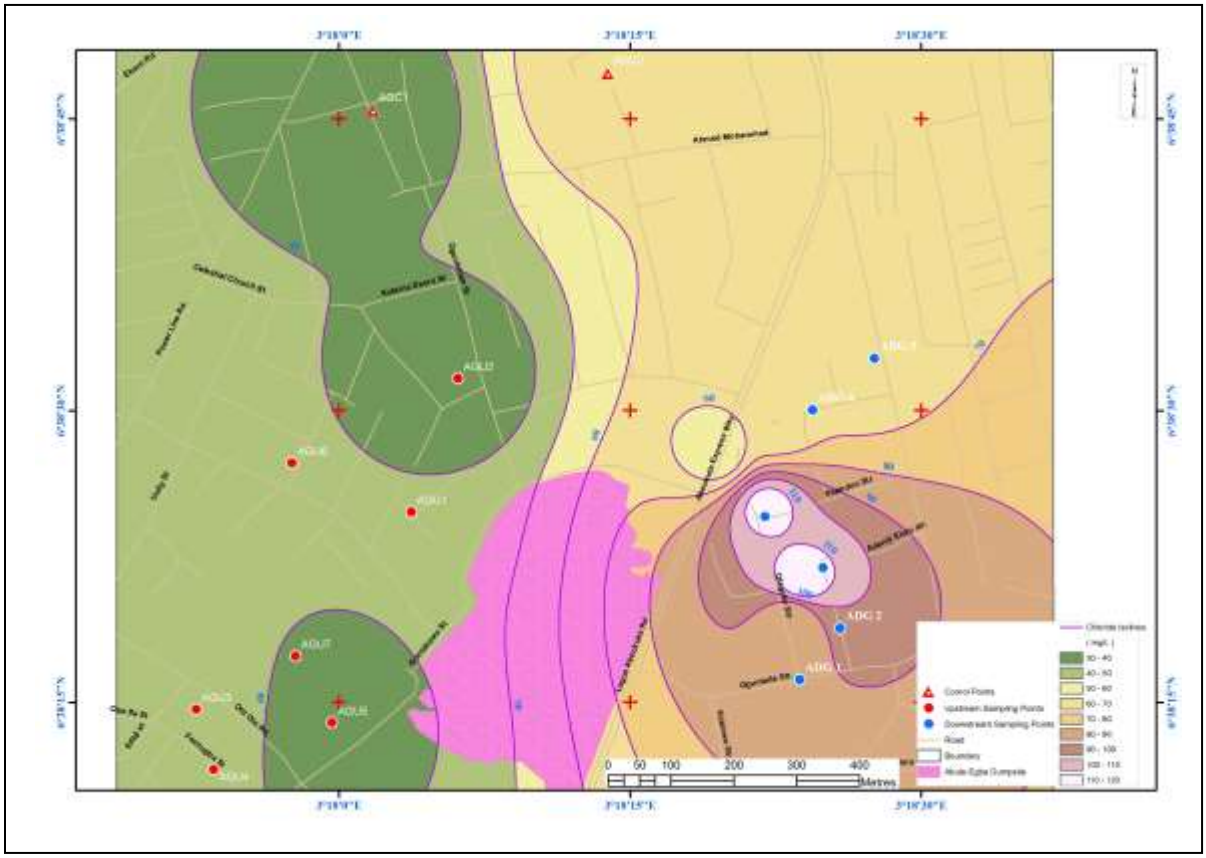


Fig 4.6: Spatial Pattern of Variability in Chloride Concentration of Groundwater around Abule-Egba Dumpsite for the Wet Season

4.3.2.2.1.3 Zinc

As shown in Figure 4.7, 71 percent of the upgradient sample locations, locations AGU1, AGU4, AGU5, AGU6 and AGU7, and an area spanning 692,392.26 m^2 is estimated to have a Zn^{2+} concentration ranging between 1.24 and 1.65mg/L. For location AGU3 and an area of 2,858.01m^2 around it, Zn^{2+} concentration is estimated to vary between 3.71 and 4.12mg/L. For location AGU2 and an areal extent of 309,692.46m^2, the concentration is estimated to range between 0.83 and 1.24mg/L.

For the downgradient sample locations and an area of 535,374.20m^2 around them, Zn^{2+} is estimated to be between 0.42 and 0.83mg/L. For AGC1 and an area of 22, 100.25m^2 around it, Zn^{2+} concentration is estimated to be between 0.83 and 1.24mg/L, while for AGC2 and an areal extent of 40,011.50m^2 around it, the concentration is estimated to vary between 3.71 and 4.12mg/L.

4.3.2.2.1.4 Total Dissolved Solids (TDS)

For AGU1 (Figure 4.8) and an area of 13, 329.16m^2 around it, the level of TDS in the groundwater is estimated to be between 402.35 and 449.98mg/L. For AGU2 and an area of 280,52.89m^2 around it, TDS is estimated to be between 21.31 and 68.94mg/L, while for locations AGU3, AGU4, AGU7, and an area of 170, 912m^2 around them, the TDS level is estimated to vary between 68.94 and 116.57mg/L. For location AGU5 and an area of 22,421.63m^2 around it, the TDS level is estimated to range between 21.31 and 68.94mg/L, while for location AGU6 and an area of 23,822.41m^2 around it, the level is estimated to vary between 211.83 and 259.46mg/L.

For locations ADG1, ADG2 and an area of 194, 251.90m^2 around them, the level of TDS in the groundwater is estimated to vary between 211.83 and 259.46mg/L, while for ADG3, ADG4, and

an area of 23,822.41m^2 around them, TDS is estimated to range between 259.46 and 307.09mg/L. For ADG5 and an area spanning 438, 210.71m^2 both upgradient and downgradient of the dumpsite, TDS is estimated to be between 164.20 and 211.83mg/L. For location ADG6 and an area of 575,143.28m^2, spanning parts of the downgradient, upgradient and control site (AGC2) and its vicinity, TDS is estimated to be between 116.57 and 164.20mg/L. For location AGC1 and an area of 101,863.76m^2 around it, TDS is estimated to be between 21.31 and 68.94mg/L.

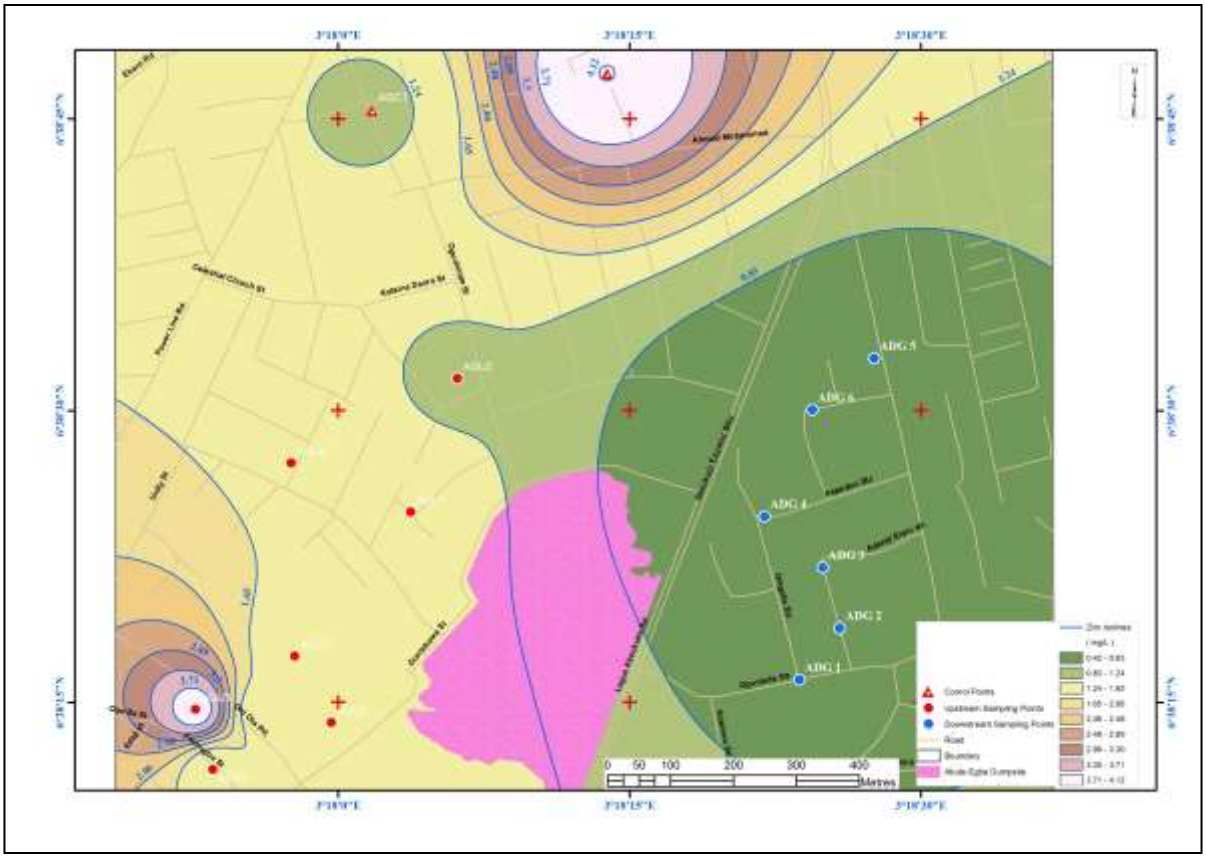


Fig 4.7: Spatial Pattern of Zinc Concentration in Groundwater around Abule-Egba Dumpsite for the Wet Season

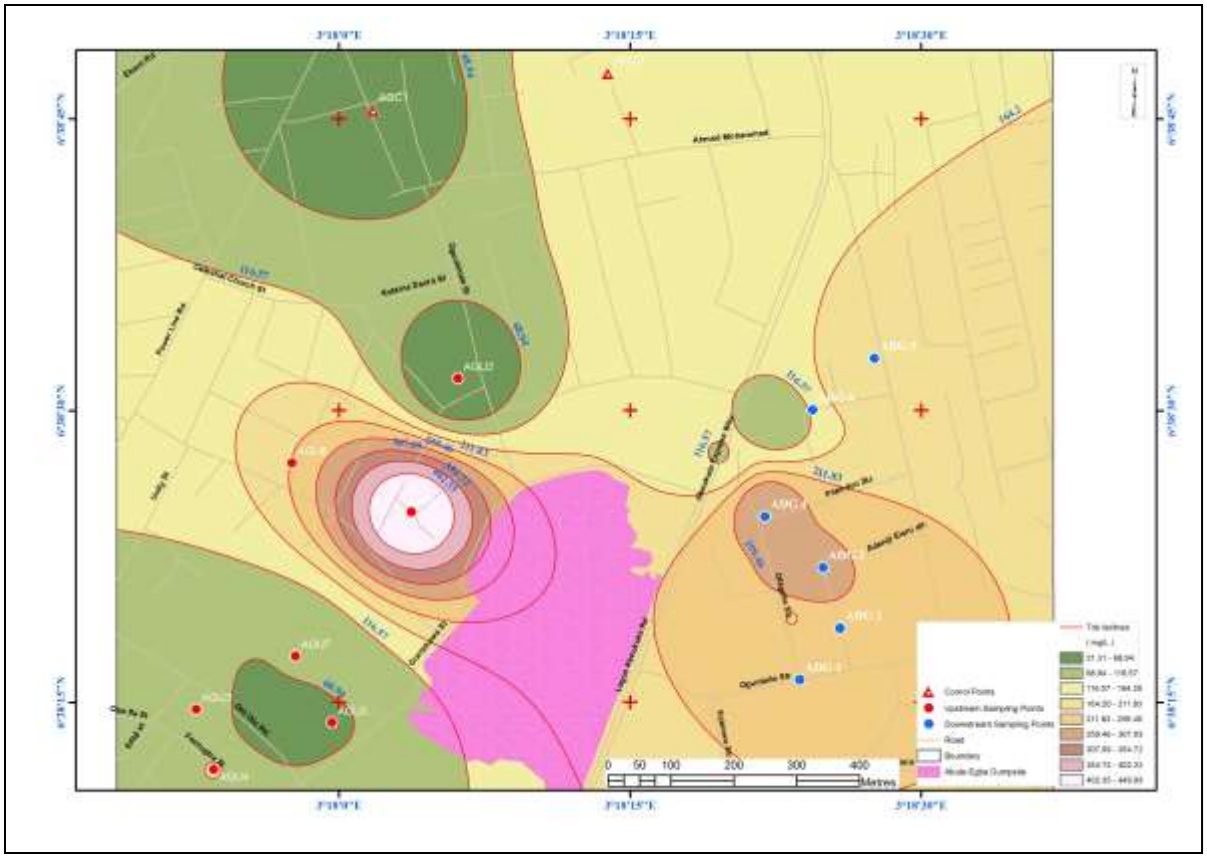


Fig 4.8: Spatial Pattern of Variability in Total Dissolved Solids Level of Groundwater around Abule-Egba Dumpsite for the Wet Season

4.3.2.2.2 Spatial Pattern of Groundwater Quality around Abule-Egba Dumpsite for the Dry Season

4.3.2.2.2.1 Total Dissolved Solids (TDS)

As shown in figure 4.9, for location AGU1 and an area of 3, 049.31m^2 around it, TDS level in the groundwater is estimated to range between 240.89 and 295.88mg/L. For locations AGU2, AGC1 and an area of 217, 267.22m^2 around them, TDS is estimated to vary between 20.90 and 75.90mg/L. Similarly, for locations AGU3, AGU5, AGU7 and an area 147,259.14m^2 around

them, TDS is estimated to vary between 20.90 and 75.90mg/L. For AGU4 and an area of 934.25m^2 around it, and AGU6, and an extent of 533,087.59m^2 spanning parts of the upgradient, downgradient and control sites (AGC2 and environs) locations, TDS is estimated to vary between 130.89 and 185.89mg/L.

For locations ADG1, ADG2 and an area of 178,232.66m^2 around them, TDS is estimated to range between 240.89 and 295.88mg/L, while for ADG3 and an area of 4, 348.55m^2 around it, the level of TDS in the groundwater is estimated to vary between 405.88 and 460.87mg/L. For location ADG5 and an area of 238,996.53m^2 around it, TDS is estimated to be between 185.89 and 240.89mg/L, while for location ADG6, its immediate vicinity, and locations spanning a large extent upgradient of the dumpsite, all of which amounts to a total area of 453,550.75m^2, TDS is estimated to vary between 75.90 and 130.89mg/L.

4.3.2.2.2.2 Chloride

As shown by the surfaces generated to represent the spatial pattern and distribution of Cl^- (Figure 4.10), the concentration of Cl^- in the groundwater at location AGU1 and an area of 15,554.89m^2 around it, is estimated to be between 130.64 and 149.74mg/L. For AGU2 and an area of 25,917.65m^2 around it, Cl^- is estimated to be between 16 and 35.11mg/L. For location AGU6 and an area extending downgradient to cover to location ADG4 and environs, which amount to an areal extent of 271,158.68m^2, Cl^- is estimated to be between 73.32 and 92.42mg/L.

For locations ADG1, ADG2 and an area of 140, 538.38m^2 around them, the concentration of Cl^- is estimated to range between 92.42 and 11.53mg/L, while for location ADG3 and an area of 3,796.79m^2 around it, the concentration is estimated to vary between 149.74 and 168.85mg/L. For location ADG5, an area of 592,456.87m^2 which includes AGC2 and environs, Cl^- is

concentration of groundwater is estimated to range between 54.21 and 73.32mg/L. For location ADG6, its immediate vicinity and an areal extent spanning a significant portion upgradient of the dumpsite, which in total amounts to an area of 444,941.75m^2, Cl^- is estimated to range between 35.11 and 54.21mg/L.

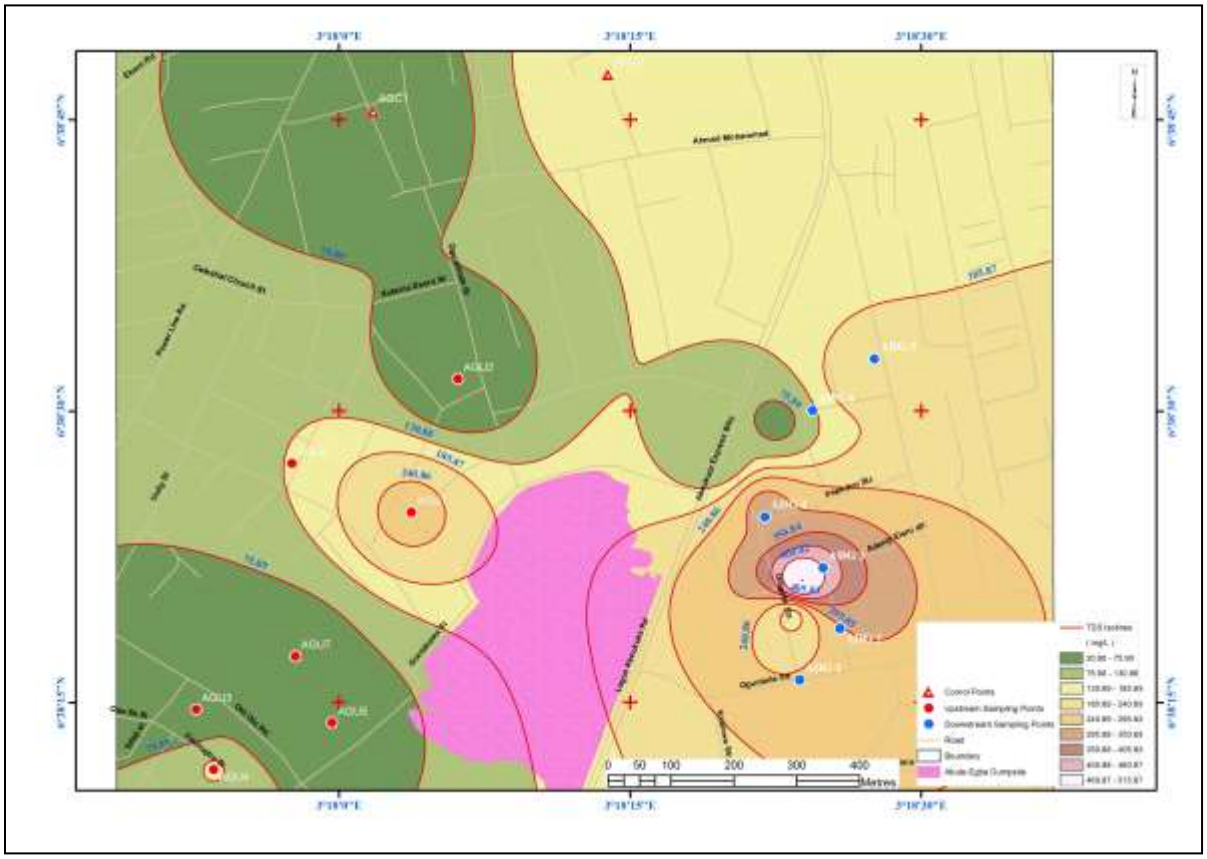


Fig 4.9: Spatial Pattern of Variability in Total Dissolved Solids Level of Groundwater around Abule-Egba Dumpsite for the Dry Season

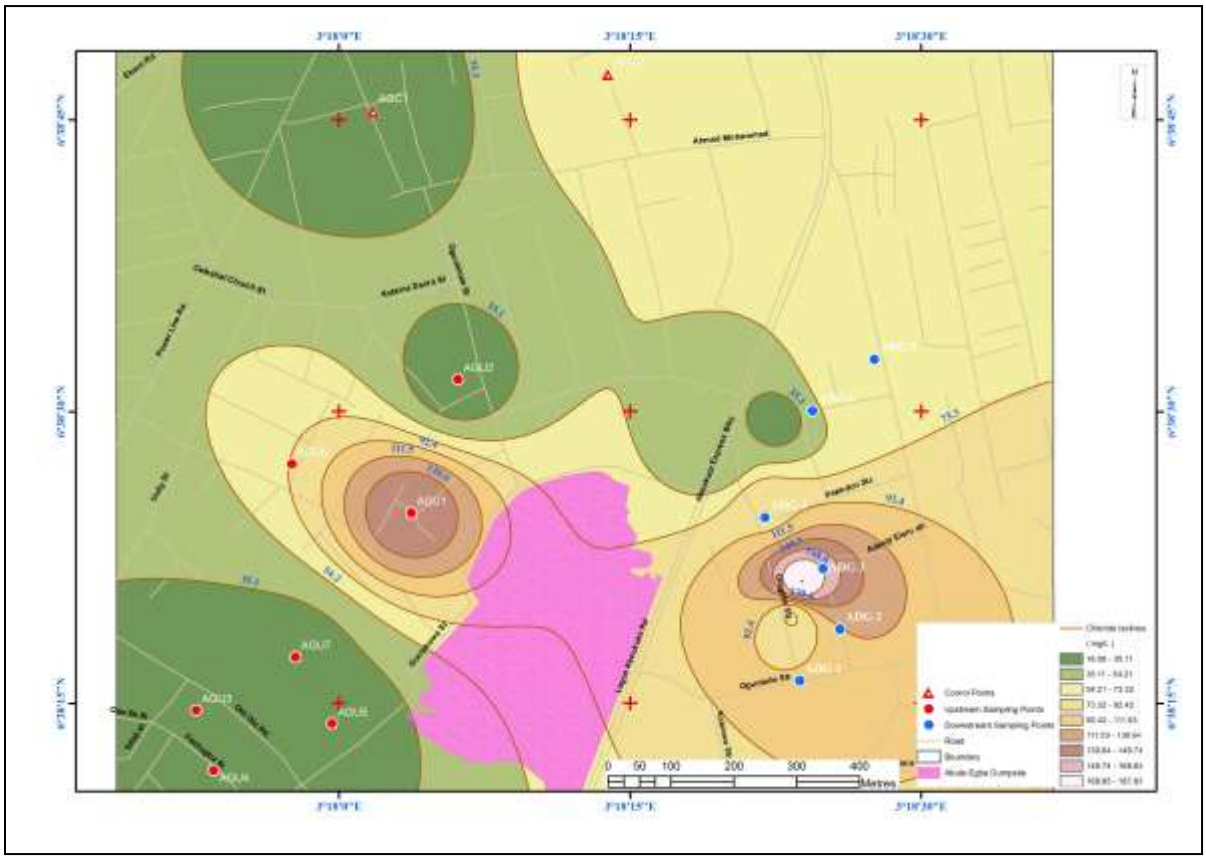


Fig 4.10: Spatial Pattern of Variability in Chloride Concentration in Groundwater around Abule-Egba Dumpsite for the Dry Season

4.3.2.2.2.3 Nitrate

The spatial pattern and distribution of NO_3^- as depicted in figure 4.11, shows that for location AGU1 and an area of 6,884.37m^2 around it, the concentration of NO_3^- is estimated to range between 3.42 and 4.09mg/L. For location AGU2 and an area of 46,567.72m^2 around it, the concentration is estimated to vary between 0.07 and 0.74mg/L.

Similarly, for locations AGU3, AGU4, AGU7 and an area of 89,819.66m^2 around them, NO_3^- is estimated to vary between 0.07 and 0.74mg/L, while for AGU5 and an area of 5,596.70m^2 around it, the concentration is estimated to range between 5.43 and 6.10mg/L. For locations

AGU6, ADG3 and an areal extent of 829,876.34m^2, NO_3^- is estimated to vary between 1.41 and 2.08mg/L.

For locations ADG1, ADG2 and an area of 24, 089.97m^2 around them, and ADG5, ADG6 and an area of 83,749.72m^2 around them, the NO_3^- concentration is estimated to range between 0.74 and 1.41mg/L. For location ADG4 and an area of 7,325.86m^2 around it, NO_3^- concentration is estimated to be between 2.08 and 2.75mg/L. For the control sites, location AGC1 and an area of 11,255.17m^2 around it, NO_3^- is estimated to range between 0.07 and 0.74mg/L, while for location AGC2 and an area of 44,225.96m^2 around it, NO_3^- is estimated to range between 3.42 and 4.09mg/L.

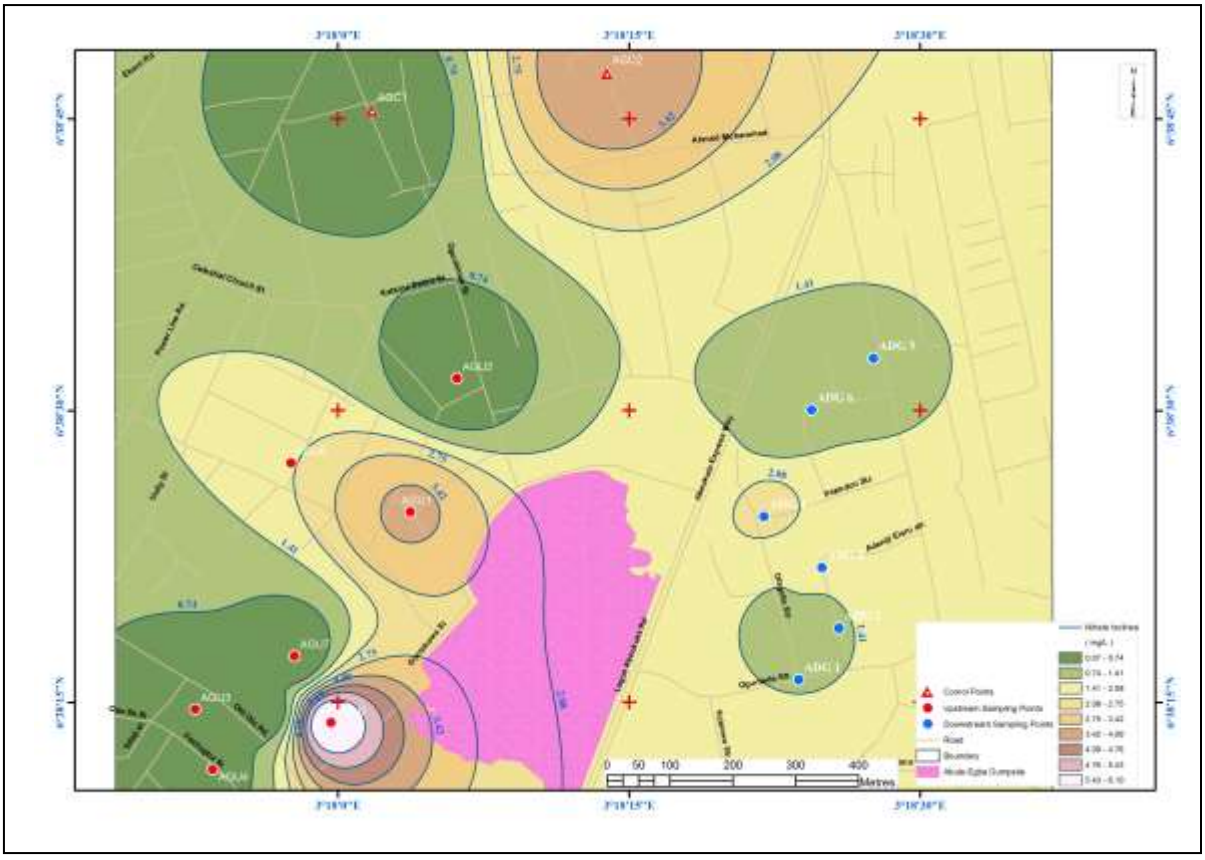


Fig 4.11: Spatial Pattern of Variability in Nitrate Concentration in Groundwater around Abule-Egba Dumpsite for the Dry Season

4.3.2.2.2.4 Iron

As shown in figure 4.12, locations AGU1, AGU3 and an area of 168,323.74m^2 around them, is estimated to have a Fe^{2+} concentration in the range of 0.41 and 0.49mg/L. For locations AGU2, AGU6 and an area of 347,528.05m^2 around them, Fe^{2+} is estimated to range between 0.33 and 0.41mg/L. For locations AGU4 and an area of 31,285.95m^2 around it, the concentration is estimated to range between 0.16 and 0.24mg/L, while for AGU5 and an area of 31,285.95m^2 around it, the concentration is estimated to range between 0.49 and 0.57mg/L. For AGU7 and an area of 11,190.23m^2 around it, Fe^{2+} is estimated to vary between 0.24 and 0.33mg/L.

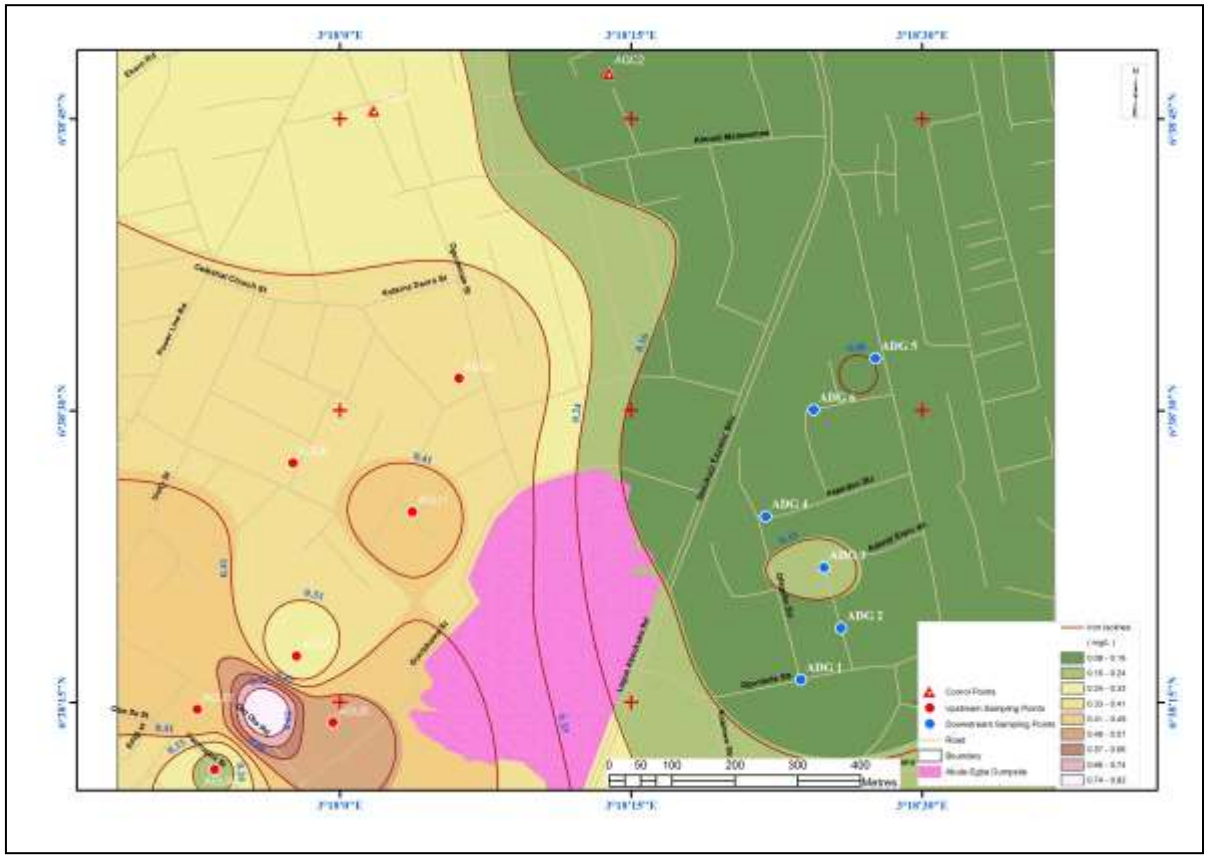


Fig 4.12: Spatial Pattern of Variability in Iron Concentration in Groundwater around Abule-Egba Dumpsite for the Dry Season

For an areal extent covering downgradient locations (except for ADG3) to the location of one of the control sites (AGC2) and its vicinity, which amounts to a total area of 736, 312.23m^2, Fe^{2+} concentration is estimated to vary between 0.08 and 0.16mg/L.For AGC1 and an areal extent of 220,266.35m^2, Fe^{2+} is estimated to range between 0.24 and 0.33mg/L.

4.3.2.2.3 Spatial Pattern of Groundwater Quality around Solous Dumpsites for the Wet Season

4.3.2.2.3.1 Nitrate

As shown in the surfaces generated for nitrate concentration of groundwater around the Solous Dumpsites (figure 4.13), for the wet season, the spatial distribution of nitrate around Solous 2 Dumpsite is generally of a higher concentration than around Solous 1 Dumpsite. As shown in figure 6.9, location S2GW1 (156.25m northwest of Solous 2 Dumpsite) and an area of 84,798m^2 around it, nitrate concentration is estimated to range between 7.08 and 8.09mg/L. For Location S2GW2 (93.40m Southwest of Solous 2 Dumpsite) and an area of 9,873.46m^2 around it, the nitrate concentration is estimated to range between 6.08 and 7.08mg/L. For location S2GW3 (81.65m north of Solous 2 Dumpsite) and an area of 12,586.17 m^2 around it, the range of nitrate concentration is estimated to be the highest at 8.09 and 9.09mg/L. For location S2GW4 (116.65m southwest of Solous 2 Dumpsite) and an area of 995,242.01 m^2 around it, the range of nitrate concentration is estimated to be between 4.07 and 5.07mg/L. For locations S2GW5 and S2GW7 (located 136.99m and 190.62m northwest of Solous 2 Dumpsite respectively), and an area of 57,938m^2 around them, the nitrate concentration is estimated to be the least, with concentrations ranging between 0.05 and 1.05mg/L. For location S2GW6 (130.62m north of Solous 2 Dumpsite) and an area of 95,941.38 m^2 around it, the nitrate concentration is estimated to vary between 1.05 and 2.06mg/L.

For Solous 1 Dumpsite, groundwater at locations S1GW1, S1GW3,S1GW4, S1GW5 and S1GW6, located 74.67m, 154.99m, 208.15m, 235.01m and 271.65m northwest of Solous 1 Dumpsite and an area of 25,876.71m^2 around them is estimated to have a nitrate concentration ranging between 0.05 and 1.05mg/L. For location S1GW2 (75.26m southeast of Solous 1 Dumpsite) and an extent spanning an area of 1,906,429.73m^2, nitrate concentration is estimated to vary between 1.05 and 2.06mg/L. For the control sites, nitrate is estimated to range between 3.06 and 4.07mg/L at location SGC2 and an area of 278,628.25m^2 around it, while at location SGC1 and an area of 153,767.42m^2 it, the concentration of nitrate is estimated to range between 0.05 and 1.05mg/L.

4.3.2.2.3.2 Chloride

The spatial pattern of Cl^- around the Solous Dumpsites as shown in figure 4.14 indicates that for location S2GW1 and an area of 35,199.39m^2 around it, Cl^- concentration of groundwater is estimated to range between 322.33 and 372.22mg/L. For S2GW2, S2GW4 and an extent of 2,580, 436.23m^2 around them the concentration is estimated to vary between127.78 and176.67mg/L. For location S2GW3 and an area of 4940.01m^2 around it, Cl^- is estimated to vary between 30 and 78.89mg/L.

For locations S1GW1, SIGW2, S1GW3, SIGW4, S1GW5, S1GW6 and an area of 888,417.97m^2 around them, Cl^- concentration of groundwater is estimated to be between 30 and 78.89mg/L. For control site SGC1 and an area of 77,441.19m^2 around it, Cl^- concentration is estimated to be between 30 and 78.89mg/L, while for SGC2 and an area of 29,230.26m^2 around it, the concentration is estimated to vary between 274.44 and 323.33mg/L.

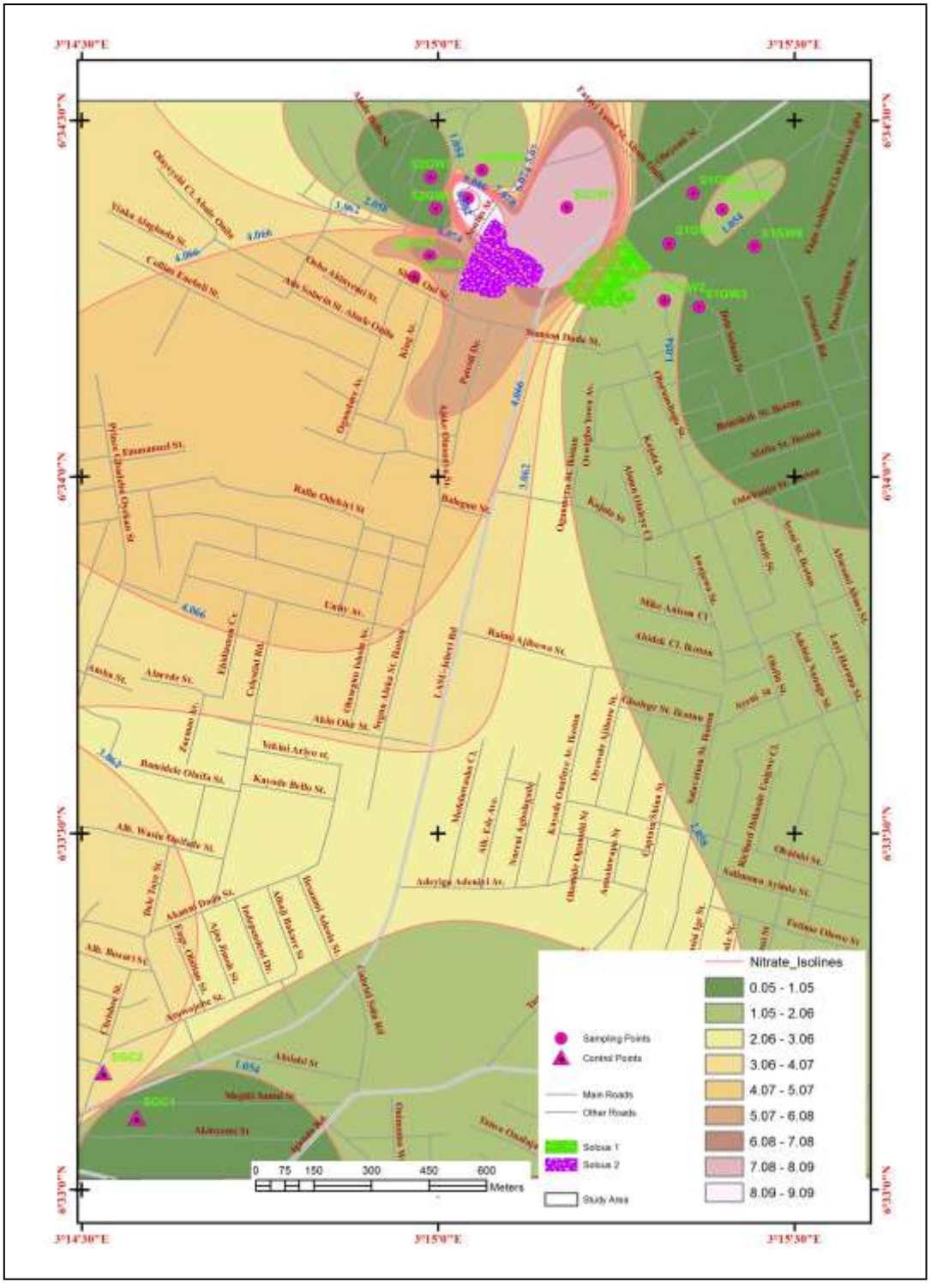


Fig 4.13: Spatial Pattern of Variability in Nitrate Concentration of Groundwater around the Solous Dumpsites for the Wet Season

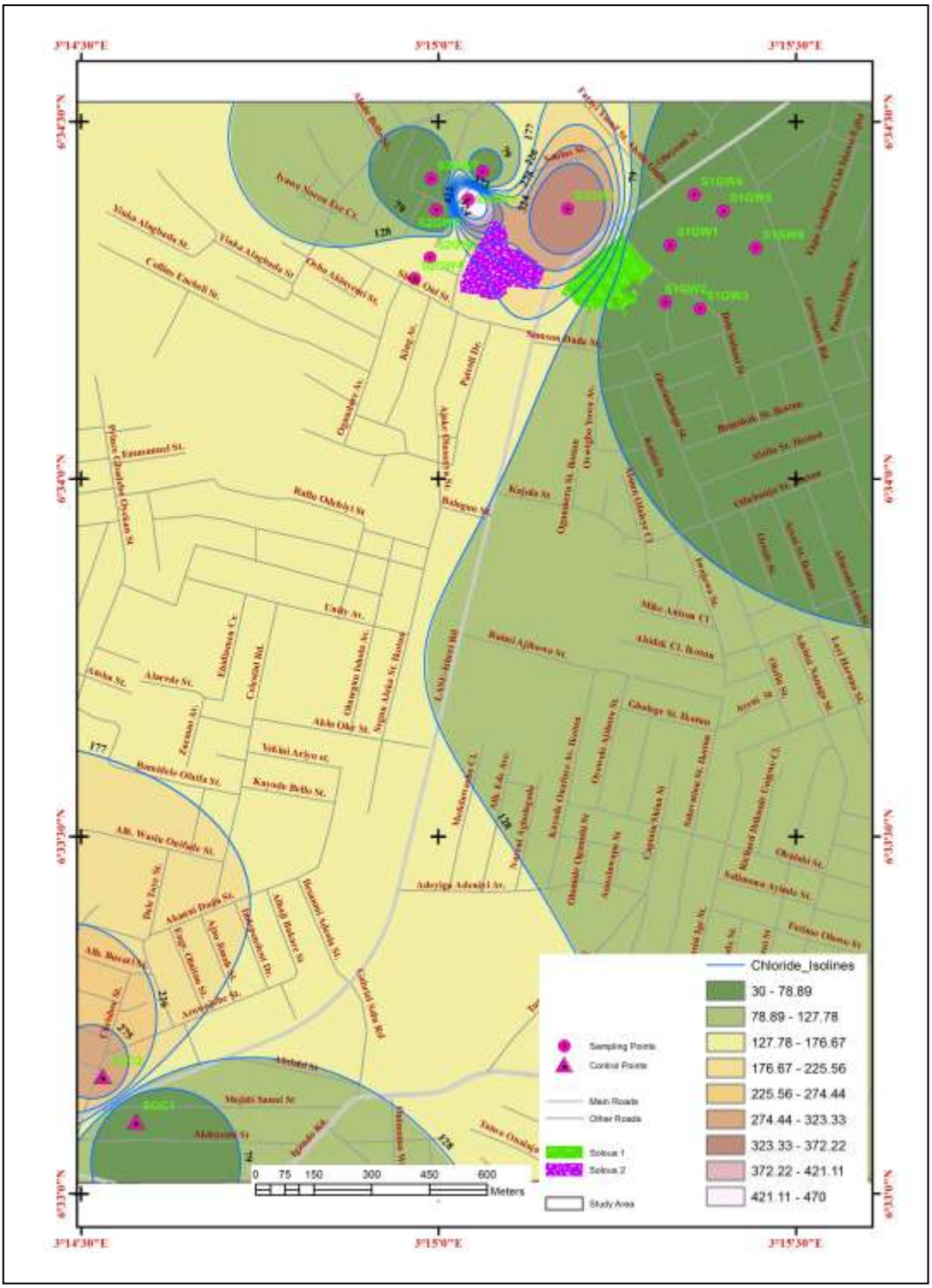


Fig 4.14: Spatial Pattern of Variability in Chloride Concentration of Groundwater around the Solous Dumpsites for the Wet Season

4.3.2.2.3.3 Zinc

For location S2GW1 (Figure 4.15) and an areal extent spanning 1,146,266.52m^2, the Zn^{2+} concentration of groundwater is estimated to range between 3.14 and 4.17mg/L. For locations S2GW2, S2GW4 and an area of 793,112.28m^2 around them, Zn^{2+} is estimated to range between 5.19 and 6.21mg/L. For location S2GW3 and an area of 15681.88m^2 around it, Zn^{2+} is estimated to range between 9.28 and 10.30mg/L, while for locations S2GW5, S2GW7 and an area of 110,812.76m^2 around them, the concentration is estimated to vary between 1.10 and 2.12mg/L. For location S2GW6 and an area of 98095.68m^2 around it, Zn^{2+} is estimated to range between 2.12 and 3.14mg/L.

For locations S1GW1, S1GW3, S1GW4, S1GW6 and an area of 606,283.52m^2 around them, Zn^{2+} concentration is estimated to be between the range of 1.10 and 2.12mg/L. For location S2GW2 and an areal extent of 1,066,941.34m^2, Zn^{2+} is estimated to vary between 4.17 and 5.19mg/L, while for location SIGW5 and an area of 62,018.69m^2 around it, the concentration is estimated to vary between 2.12 and 3.14mg/L.

For the control sites, location SGC1 and an area of 186,581.35m^2 around it, Zn^{2+} is estimated to vary between 1.10 and 2.12mg/L, while for SGC2 and an area of 142,590.90m^2 around it, the concentration is estimated to vary between 6.21 and 7.23mg/L.

4.3.2.2.3.4 pH

As shown in figure 4.16, the level of groundwater pH at location S2GW1 and an area of 89,874.69m^2 around it, is estimated to vary between 6.33 and 6.47. For locations S2GW2, S2GW3 and an area spanning 1,088,053.90m^2, pH is estimated to range between 7.01 and 7.14. For location S2GW4 and an area of 145650.03m^2 around it, pH is estimated to vary between

7.41 and 7.55, while for location S2GW5 and an area of 17,259.30m^2 around it, the level pH is estimated to range between 6.74 and 6.87. For locations S2GW6, S1GW4 and an area of 193391.59m^2 around them, pH is estimated to range between 6.47 and 6.60, while for S2GW7 and an areal extent of 865,144.96m^2, pH is estimated to vary between 6.87 and 7.01.

For locations S1GW1, S1GW6 and an area of 156,396.74m^2 around them, pH is estimated to vary between 6.74 and 6.87, while for location S1GW5 and an area of 35,621.71m^2 around it, the estimated range is between 6.33 and 6.47. For locations S1GW2, S1GW3 and an area of 271,468.62m2 around them, the pH levels is estimated to vary between 7.14 and 7.28, while for the control sites, SGC1, SGC2, and an area of 1,310,726.79m^2 around them, the range is estimated to be between 6.33 and 6.47.

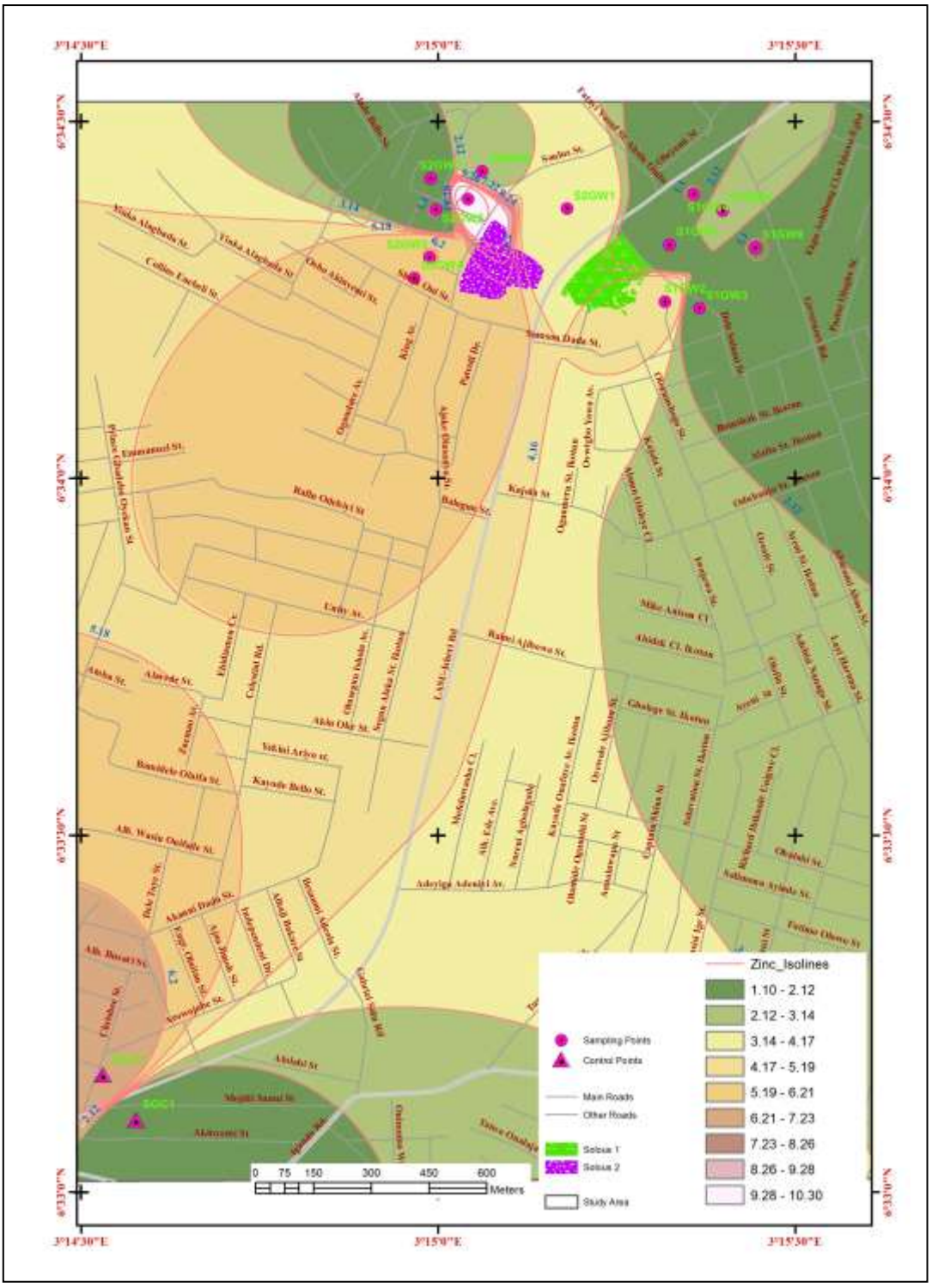


Fig 4.15: Spatial Pattern of Variability in Zinc Concentration of Groundwater around the Solous Dumpsites for the Wet Season

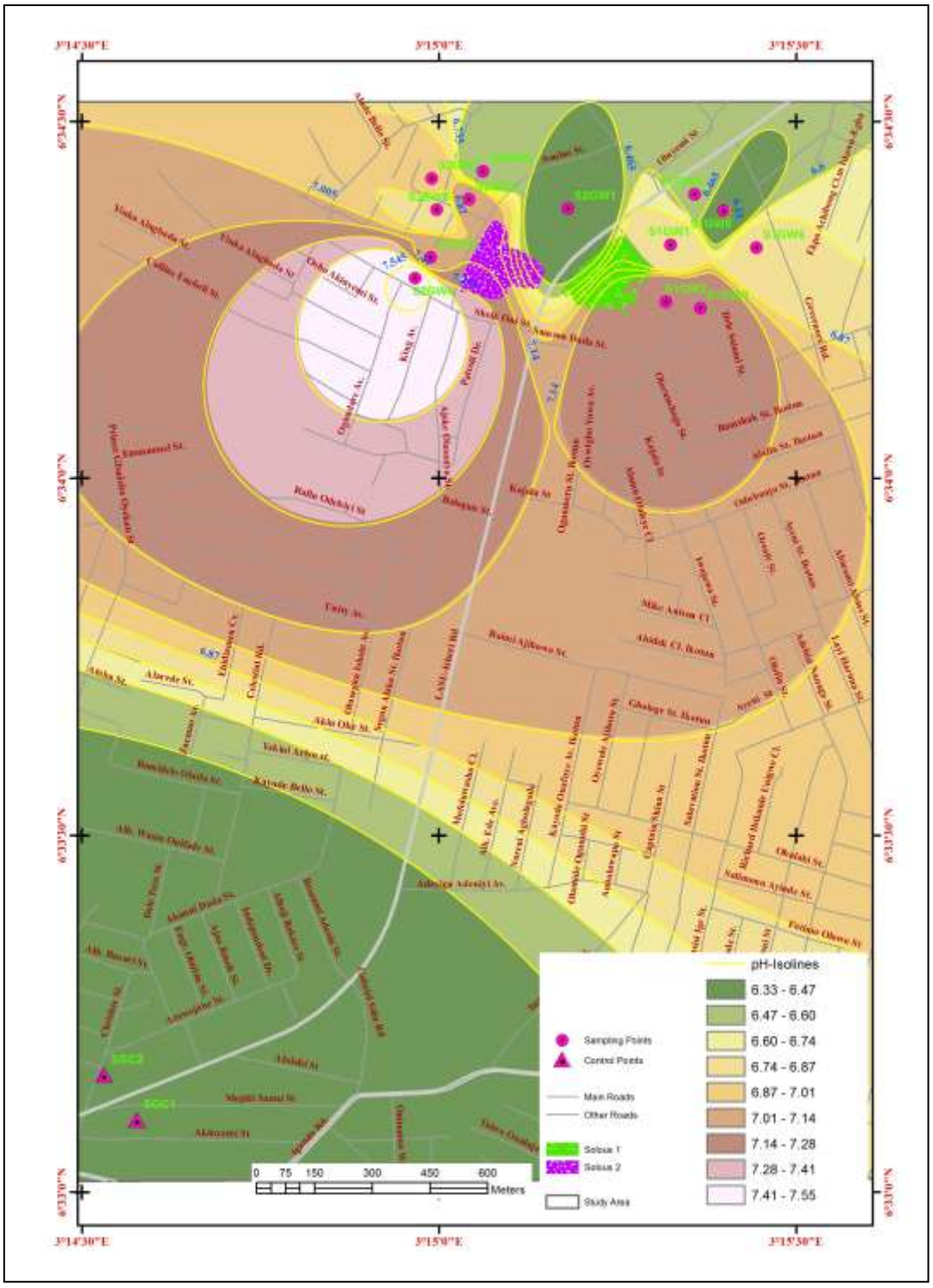


Fig 4.16: Spatial Pattern of Variability in pH Levels of Groundwater around the Solous Dumpsites for the Wet Season

4.3.2.2.3.5 Iron

As shown by the spatial pattern of Fe^{2+} concentration around the Solous Dumpsites represented in figure 4.17, location S2GW1 and an area of 116,975.60m^2 around it, is estimated to have a Fe^{2+} concentration between the range 0.29 and 0.53mg/L. Similarly, for locations S2GW2, S2GW5, and an area of 36,623.66m^2 around them, Fe^{2+} is estimated to range between 0.29 and 0.55mg/L, while for location S2GW3 and an area of 3066.27m^2 around it, the concentration is estimated to vary between 1.96 and 2.20mg/L.

For location S2GW4 and an area of 1,013,182.10m^2 around it, the concentration is estimated to vary between 1.01 and 1.24mg/L, while for location S2GW6 and an area of 49,143.38m^2 around it, Fe^{2+} is estimated to range between 0.05 and 0.29mg/L. For location S2GW7 and an area of 108,201.71m^2 around it, the concentration of Fe^{2+} is estimated to vary between 1.01 and 1.24mg/L.

For locations S1GW1, S1GW2, S1GW3 and an areal extent of 1,735,268.37m^2, Fe^{2+} is estimated to vary between 0.53 and 0.77mg/L. For location S1GW4 and an area of 60,730.91m^2 around it, Fe^{2+} is estimated to vary between 1.24 and 1.48mg/L, while for location S1GW5 and an area of 92,399.54m^2 around it, the concentration is estimated to range between 0.29 and 0.53mg/L. For location S1GW6 and an area of 222, 858.80m^2 around it, Fe^{2+} is estimated to range between 0.05 and 0.29mg/L. For the control sites, it is estimated that for location SGC2 and an areal extent of 2,115,470.81m^2, the concentration of Fe^{2+} is estimated to be within the range of 0.29 and 0.53mg/L.

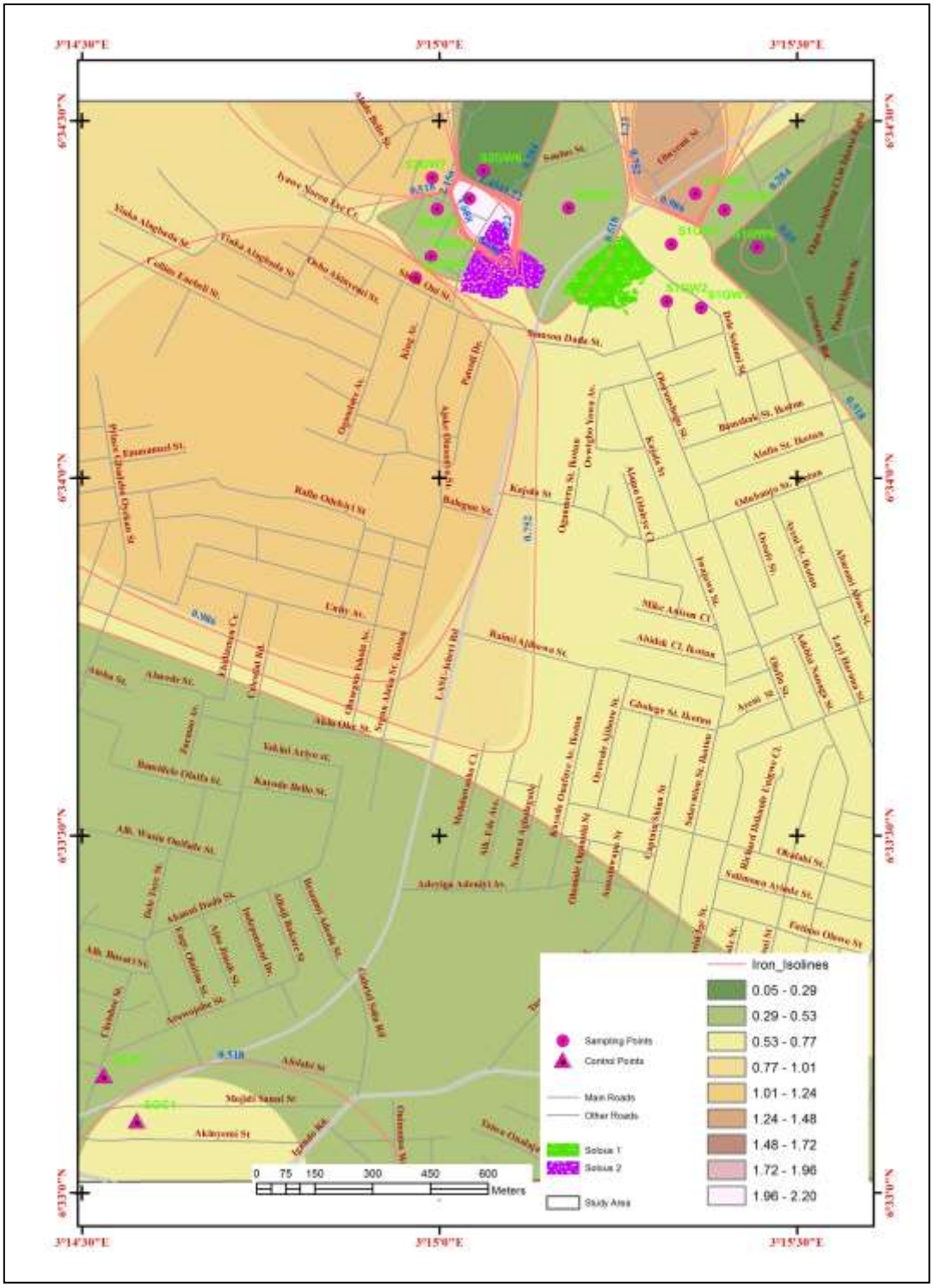


Fig 4.17: Spatial Pattern of Variability of Iron Concentration in Groundwater around the Solous Dumpsites for the Wet Season

4.3.2.2.4 Spatial Pattern of Groundwater Quality around Solous Dumpsites for the Dry Season

4.3.2.2.4.1 Sulphate

As shown in figure 4.18, the SO_4^{2-} concentration of location S2GW1 and an areal extent of 1,330,888.04m^2 is estimated to range between 23.33 and 35mg/L. For location S2GW2 and an area of 271,095.49m^2 around it, SO_4^{2-} is estimated to vary between 35 and 46.67mg/L. For locations S2GW3, S2GW7 and an area of 78,177.06m^2 around them, SO_4^{2-} is estimated to range between 0 and 11.67mg/L, while for location S2GW4 and an area of 41,567.90m^2 around it, the concentration is estimated to range 11.67 and 23.33mg/L.

For location S2GW5 and an area of 10,851.25m^2 around it, SO_4^{2-} is estimated to vary between 93.33 and 105mg/L, while for location S2GW6, through to location S1GW3 and an areal extent of 1,845,889.18m^2, the concentration is estimated to range between 11.67 and 23.33mg/L. For locations S1GW1, S1GW2, S1GW4, S1GW5 and an area of 340,668.58m^2 around them, SO_4^{2-} is estimated to vary between 0 and 11.67mg/L, while for location S1GW6 and an area of 129,544.04m^2 around it, the concentration is estimated to range between 46.67 and 58.33mg/L. For the control sites (SGC1 and SGC2) and an areal extent of 1,902,061.52m^2, SO_4^{2-} is estimated to vary between 0 and 11.67mg/L.

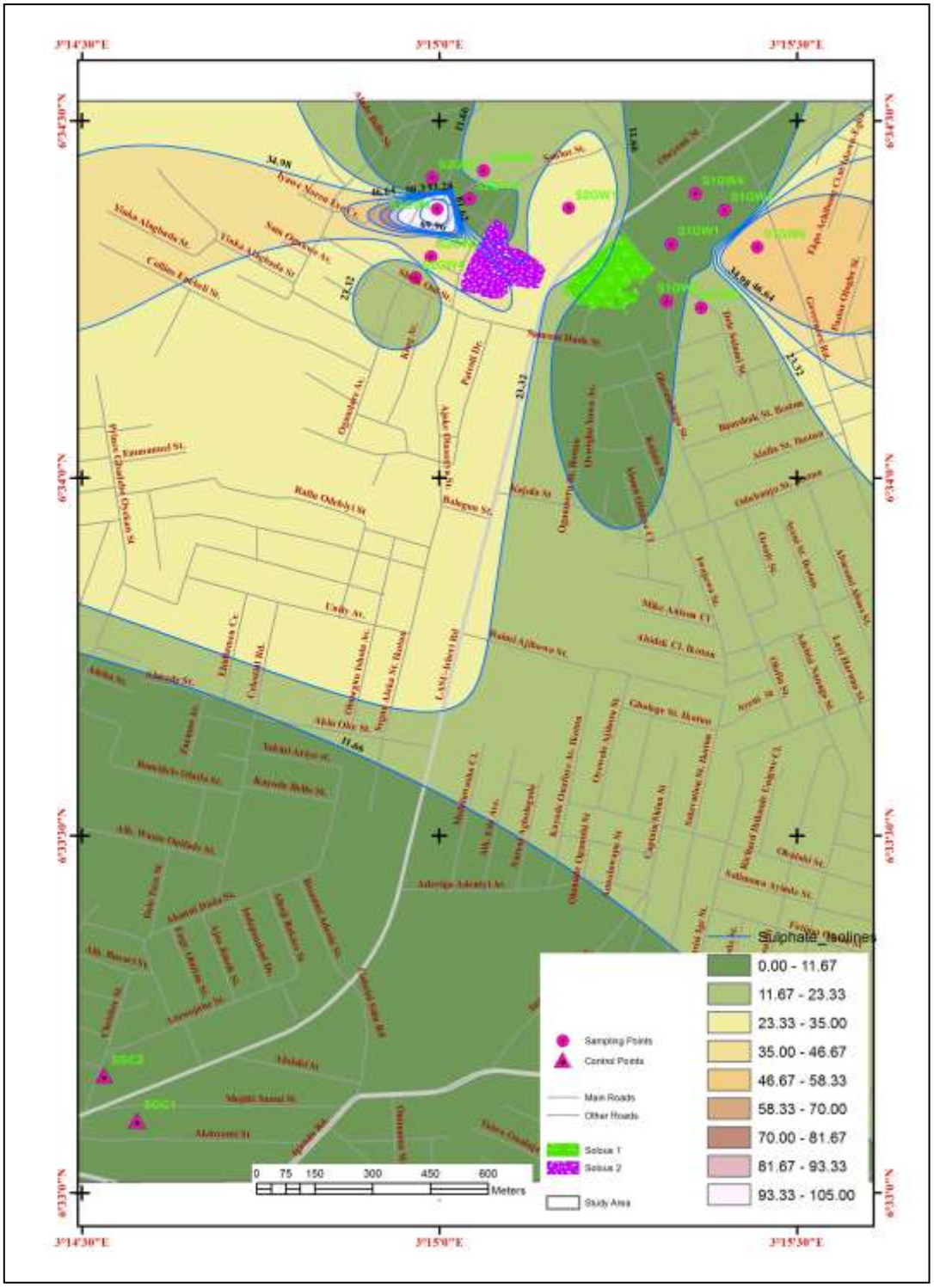


Fig 4.18: Spatial Pattern of Variability in Sulphate Concentration of Groundwater around the Solous Dumpsites for the Dry Season

4.3.2.2.4.2 Nitrate

For the spatial pattern of NO_3^- shown in figure 4.19, location S2GW1 and an extent spanning 1,442,780.83m^2, is estimated to have a NO_3^- concentration between the range of 19.23 and 28.23mg/L. For location S2GW2 and an area of 104,228.36m^2 around it, the concentration is estimated to vary between 28.83 and 38.42mg/L. For locations S2GW3, S2GW6, S2GW7 and an area of 121,731.32m^2 around them, the concentration is estimated to range between 0.04 and 9.64mg/L. For location S2GW2 and an area of 62, 217.36m^2 around it, NO_3^- is estimated to vary between 9.84 and 19.23mg/L, while for location S2GW5 and an area of 107,786.65m^2 around it, the concentration is estimated to be between 76.80 and 86.40mg/L.

For locations S1GW1, S1GW2, S1GW4, S1GW5 and an area of 374,057.24m^2 around them, NO_3^- concentration is estimated to range between 0.04 and 9.64mg/L, while for location S1GW3 and an areal extent spanning 1,671,919.21m^2, the concentration is estimated to range between 9.64 and 19.23mg/L. For location S1GW6 an area of 130,696.55m^2 around it, NO_3^- is estimated to vary between 28.83 and 38.42mg/L, while for the control sites (SGC1 and SGC2) and an areal extent of 2, 090,563.08m^2, NO_3^- is estimated to range between 0.04 and 9.64mg/L.

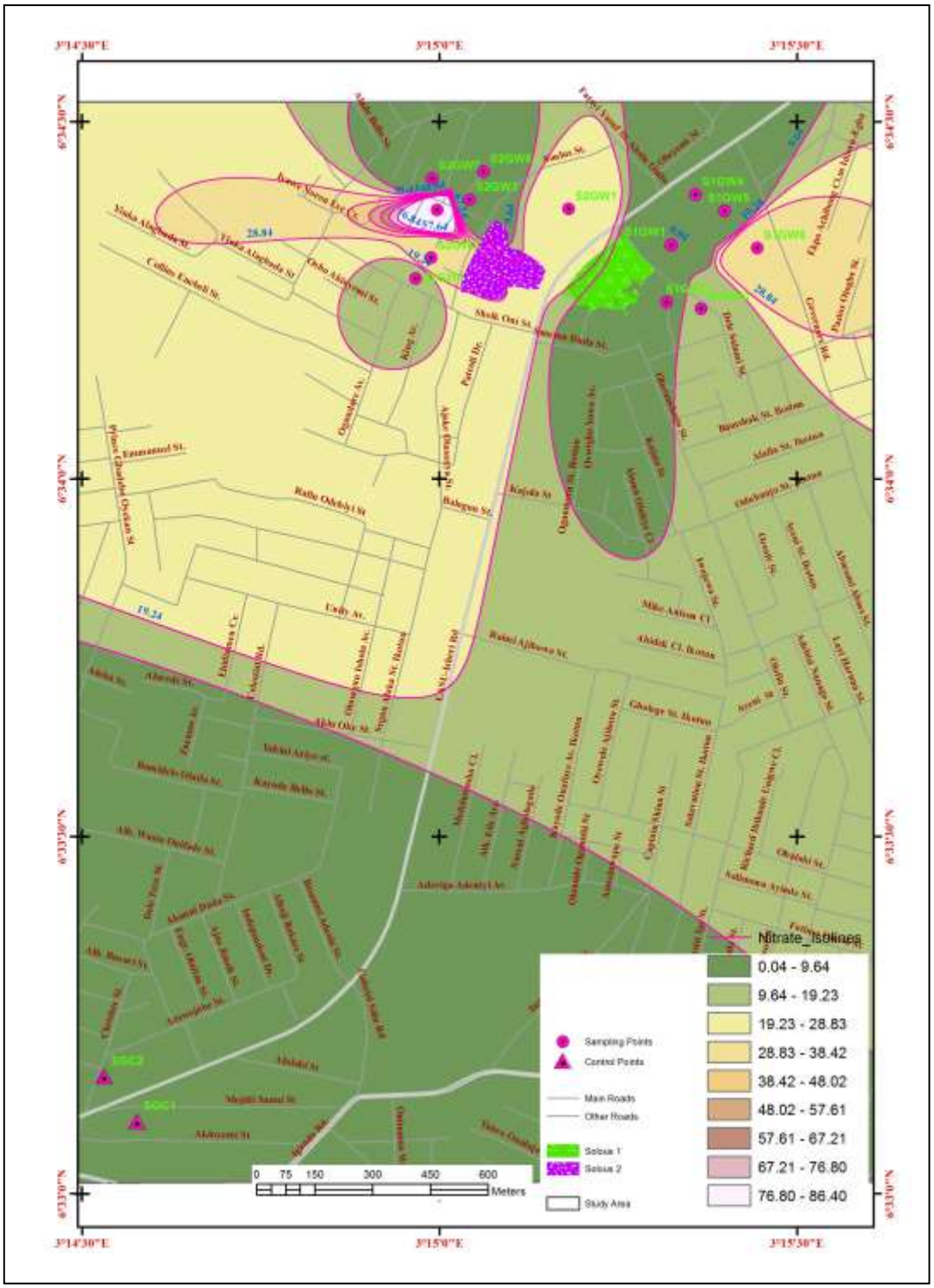


Fig 4.19: Spatial Pattern of Variability in Nitrate Concentration of Groundwater around the Solous Dumpsites for the Dry Season

4.3.2.2.4.3 Total Dissolved Solids (TDS)

As shown by the spatial pattern of TDS (figure 4.20), location S2GW1 and an areal extent of 352,045.26m^2 is estimated to have a TDS level between the range 94.84 and 135.87mg/L. For location S2GW2 and an area of 37,096.01m^2 around it, the TDS level of groundwater is estimated to vary between 176.89 and 217.91mg/L, while for location S2GW3 and an area of 23,292.30m^2 around it, the level is estimated to be between 299.96 and 340.98mg/L. For location S2GW4 and an area of 31,687.25m^2 around it, TDS is estimated to vary between 135.87 and 176.89mg/L. For location S2GW5 and an area of 16,046.12m^2 around it, TDS is estimated to vary between 94.84 and 135.87mg/L, while for location S2GW6 and an area of 48,374.09m^2 around it, the level is estimated to range between 340.98 and 382mg/L. For location S2GW7 and an area of 178,480.57m^2 around it, TDS is estimated to range between 176.89 and 217.91mg/L.

For locations S1GW1, S1GW2, S1GW4, S1GW5 and an area of 245,867.68m^2 around them, the TDS level of groundwater is estimated to be between 12.80 and 53.83mg/L, while for location S1GW3 and an areal extent of 1,908,455.22m^2, TDS is estimated to vary between 53.82 and 94.84mg/L. For locations S1GW6 and an area of 76,636.26m^2 around it, TDS is estimated to range 94.84 and 135.87mg/L. For the control sites (SGC1 and SGC2) and an areal extent of 1,845,489.89m^2, TDS is estimated to vary between 12.80 and 53.82mg/L.

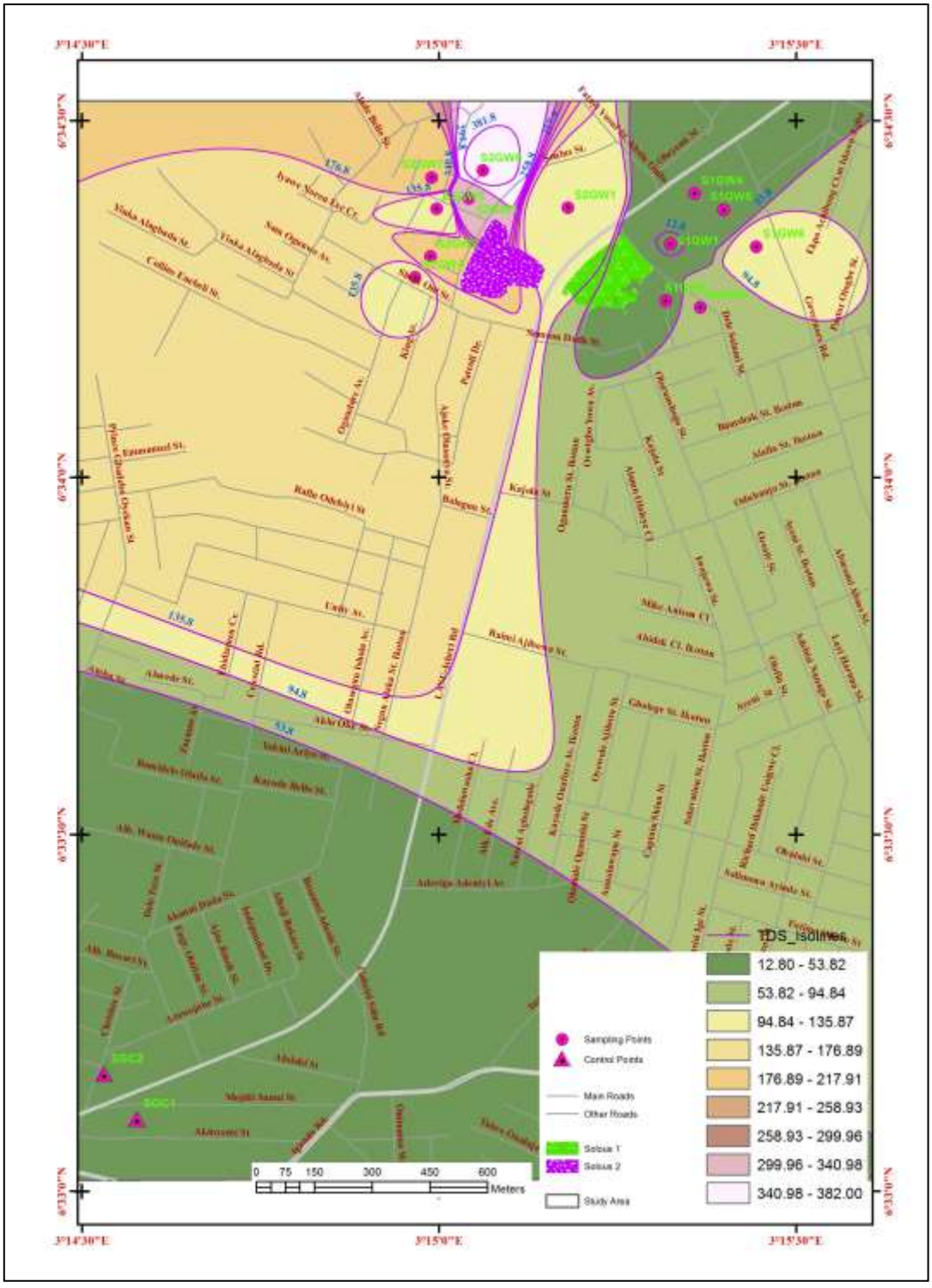


Fig 4.20: Spatial Pattern of Variability Total Dissolved Solids Levels of Groundwater around the Solous Dumpsites for the Dry Season

4.3.2.2.4.4 Zinc

As shown in figure 4.21, the concentration of Zn^{2+} in the groundwater at location S2GW1 and an area of 55,837.13m^2 around it, is estimated to range between 0.32 and 0.34mg/L. For locations S2GW2, S2GW5 and an area of 893,604.86m^2 around them, Zn^{2+} is estimated to vary between 0.32 and 0.34mg/L, while for location S2GW3 and an area of 25,921.60m^2 around it, the concentration is estimated to range between 0.19 and 0.22mg/L. For location S2GW4 and an area of 106,426.33m^2 around it, Zn^{2+} is estimated to vary between 0.37 and 0.40mg/L, while for location S2GW6 and an area 37,048.38m^2 around it, the concentration is estimated to vary between 0.17 and 0.19mg/L. For location S2GW7 and an area of 32,077.67m^2 around it, Zn^{2+} is estimated to vary between 0.34 and 0.37mg/L

For location S1GW1 and an area of 32,077.67m^2 around it, the concentration of Zn^{2+} is estimated to vary between 0.24 and 0.27mg/L, while for location S1GW2 and an area of 36,553.03m^2 around it, the concentration is estimated to vary between 0.22 and 0.24mg/L. For location S1GW3 and an area of 45,892.01m^2 around it, Zn^{2+} is estimated to vary between 0.37 and 0.40mg/L, while for locations S1GW4, S1GW5 and an area of 79,462.32m^2 around them, the concentration is estimated to vary between 0.32 and 0.34mg/L. For location S1GW6 and an area of 106,449.44m^2 around it, the range of Zn^{2+} concentration is estimated to be between 0.22 and 0.24mg/L. For the control sites (SGC1 and SCG2) and an areal extent of 1,400,878.27m^2, Zn^{2+} concentration is estimated to be between 0.17 and 0.19mg/L.

Fig 4.21: Spatial Pattern of Variability in Zinc Concentration of Groundwater around the Solous Dumpsites for the Dry Season

4.4 GROUNDWATER CONTAMINATION INDEX

4.4.1 Abule-Egba Dumpsite Vicinity

The results of the computed Groundwater Contamination Index (C_d) for the wet season at Abule –Egba Dumpsite vicinity (Table 4.11) show that Fe^{2+} is the main contamination parameter. Out of the 7 groundwater samples collected upstream of the dumpsite, 5 had Fe^{2+} as the contamination parameter, while the remaining 2 had no contamination parameter and were therefore categorised as having low level of contamination. Out of the 5 sample locations which had Fe^{2+} as their contamination parameter, 4 were adjudged as having medium level of contamination with C_d values in the range of $1< C_d < 3$, while 1 was categorised as having a high level of contamination with $C_d > 3$.

Downgradient of the dumpsite Fe^{2+} was the sole contamination parameter, except for one location (ADGI) which had no contamination parameter. For the other locations which had Fe^{2+} as their contamination parameter, 4 was categorised as having medium level of contamination with computed C_d values in the range of $1< C_d < 3$, while 1 was adjudged as having a high level of Fe^{2+} contamination at $C_d > 3$.

At the control sites Fe^{2+} and Zn^{2+} were the contamination parameters, with groundwater at location AGC1 having a medium level of Fe^{2+} contamination and groundwater at AGC2 having medium level of Zn^{2+} contamination.

For the dry season, medium level of Fe^{2+} contamination was computed for all but one location (AGU4) upgradient of the dumpsite. For downgradient locations and the control sites $C_d = 0$ due to the fact none of the groundwater quality parameters exceeded the upper permissible limit of

the WHO guidelines for drinking water quality, and were thus adjudged as having low levels of contamination.

The emergence of Fe^{2+} as a contamination parameter suggests that the local geology is a major determinant of Fe^{2+} concentration in the groundwater around the dumpsite, in spite of the 4.9mg/L concentration in the leachate collected at the dumpsite during the wet season. The predominance of clay (known to be made up of a significant proportion Fe^{2+}) in the lithostratigraphic units of the dumpsite and environs (Table 1.1 to 1.3) and the high proportion of clay in the soil of the dumpsite is indicative of the influence of the Fe^{2+} constituent of the area's geology on the groundwater. Information derived from the soil grain size of the Dumpsite (LAWMA, 2008), shows that the top soil (0-50 cm) at the dumpsite had 90.2 and 75.01 percent clay composition for the wet and the dry season respectively, while the clay composition of the subsoil was 13.77 and 74.4 percent for the wet and dry season respectively. Furthermore, past studies have revealed that elevated levels of Fe^{2+} is not uncommon in Lagos, as all the three aquifers mostly used for water supply in Lagos have high concentration of Fe^{2+} (Lagos Ministry of Environment and Physical Planning [MEPP],2002).

Table 4.11: Contamination Index of Groundwater in the Vicinity of Abule-Egba Dumpsite

Location/Sample Code	Contamination Parameter & Contamination Index (Cd)				Level of Contamination	
	Contamination Parameter	Wet Season Index	Contamination Parameter	Dry Season Index	Wet Season	Dry Season
Upgradient Locations						
Location 1/AGU1	Fe^{2+}	1.06	Fe^{2+}	1.4	Medium	Medium
Location 2/AGU2	None	0	Fe^{2+}	1.27	Low	Medium
Location 3/AGU3	Fe^{2+}	1.43	Fe^{2+}	1.5	Medium	Medium
Location 4/AGU4	Fe^{2+}	3.6	Fe^{2+}	0	High	Low
Location 5/AGU5	Fe^{2+}	2.03	Fe^{2+}	1.7	Medium	Medium
Location 6/AGU6	Fe^{2+}	2.4	Fe^{2+}	2.7	Medium	Medium
Location 7/AGU7	None	0	None	0	Low	Low
		Downgradient Locations				
Location 1/ADG1	None	0	None	0	Low	Low
Location 2/ADG2	Fe^{2+}	2.33	None	0	Medium	Low
Location 3/ADG3	Fe^{2+}	2.2	None	0	Medium	Low
Location 4/ADG4	Fe^{2+}	2.96	None	0	Medium	Low
Location 5/ADG5	Fe^{2+}	3.30	None	0	High	Low
Location 6/ADG6	Fe^{2+}	2.73		0	Medium	Low
Control Site						
Location 1/AGC1	Fe^{2+}	2.2	None	0	Medium	Low
Location 2/AGC2	Zn^{2+}	1.37	None	0	Medium	Low

4.4.2 Solous Dumpsites Vicinity

Based on the results of the Contamination Index computed for the wet season (Table 4.12), it was revealed that Fe^{2+} was the sole contamination parameter for majority of the groundwater sampled for Solous 1 Dumpsite. The only exceptions to this were location S1GW5 which had Fe^{2+} and Zn^{2+} as its contamination parameters and location S1GW6 which had no contamination parameter. For the 4 locations whose contamination parameter was Iron only, 3 had C_d values within the $1< C_d < 3$ range and one fell within the $C_d > 3$ range.

For Solous 2 Dumpsite, Cl^-, Zn^{2+} and Fe^{2+} were the contamination parameters for locations S2GW1 and S2GW2, Fe^{2+} and Zn^{2+} for S2GW3 and S2GW4, Fe^{2+} only for S2GW5 and S2GW6 and none for S2GW7. These resulted in $C_d > 3$ for locations S2GW1, S2GW2, S2GW3, S2GW4

and S2GW5, C_d values within the range of $1< C_d < 3$ for location S2GW6 and a C_d value = 0 for sample S2GW7. Worthy of note is the fact that locations S2GW1 and S2GW2 are the only locations with chloride as one of its contamination parameters. This implies the migration and intrusion of leachate into the aquifers.

For the dry season all but one location for Solous 1 Dumpsite was categorised as having medium level of contamination, with Fe^{2+} as the contamination parameter. For Solous 2 Dumpsite, the degree of groundwater contamination at most of the locations fell within the medium level category. The only exceptions were locations S2GW2 and S2GW3 whose C_d values = 0, and location S2GW5 whose level of contamination was rated as high. The contamination parameters for Solous 2 Dumpsite during the dry season were Cl^-, Fe^{2+} and NO_3^-. Observed among all the locations, is the fact that location S2GW5 was the only groundwater sample with NO_3^- as a contamination parameter.

At the control sites Fe^{2+} and Zn^{2+} which were the contamination parameters for the wet season fell within the medium category range, while during the dry season Fe^{2+} was the sole contamination parameter and it also fell within the medium category range.

The emergence of Fe^{2+}, Cl^- , Zn^{2+} and NO_3^- as contamination parameters for both seasons shows that the quality of groundwater around the dumpsites is determined by the combination of the area's geology and the waste disposal activities at the dumpsites. The contamination of the groundwater by Fe^{2+} is attributed to the laterite and clay that forms part of the lithostratigraphy of the dumpsites and environs (Tables 1.4-1.6). The emergence of Cl^- as a contamination parameter is an indication of the migration of leachate plume from the dumpsite into the groundwater. This is because Cl^- is one of the few persistent substances used in the detection and

characterisation of leachate plume migration and therefore indicates the influence that leachate can exert on groundwater quality downgradient of landfills and dumpsites (Taylor and Allen, 2006). The emergence of Zn^{2+} as a contamination parameter may be due to geological influence or the disposal of items such as batteries, fluorescent tubes, tires and galvanised products at the dumpsites.

Table 4.12: Contamination Index of Groundwater in the Vicinity of Solous 1 and Solous 2 Dumpsites

Location/Sample Code	Contamination Parameter & Contamination Index (Cd)				Level of Contamination	
	Contamination Parameter	Wet Season Index	Contamination Parameter	Dry Season Index	Wet Season	Dry Season
Solous 1						
Location 1/S1GW1	Fe^{2+}	2.0	Fe^{2+}	1.57	Medium	Medium
Location 2/S1GW2	Fe^{2+}	2.2	Fe^{2+}	2.13	Medium	Medium
Location 3/S1GW3	Fe^{2+}	2.33	Fe^{2+}	2.4	Medium	Medium
Location 4/S1GW4	Fe^{2+}	4.33	None	0	High	Low
Location 5/S1GW5	Fe^{2+} & Zn^{2+}	2.36	Fe^{2+}	1.43	Medium	Medium
Location 6/S1GW6	None	0	Fe^{2+}	1.17	Low	Medium
		Solous 2				
Location 1/S2GW1	Cl^-, Fe^{2+} & Zn^{2+}	3.94	Cl^-	1.39	High	Medium
Location 2/S2GW2	Cl^-, Fe^{2+} & Zn^{2+}	12.64	None	0	High	Low
Location 3/S2GW3	Fe^{2+} & Zn^{2+}	3.69	None	0	High	Low
Location 4/S2GW4	Fe^{2+} & Zn^{2+}	5.89	Cl^- & Fe^{2+}	2.17	High	Medium
Location 5/S2GW5	Fe^{2+}	3.67	Cl^- & NO_3^-	3.3	High	High
Location 6/S2GW6	Fe^{2+}	1.47	Cl^-	2.19	Medium	Medium
Location 7/S2GW7	None	0	Fe^{2+}	1.17	Low	Medium
Control Sites						
Location 1/SGC1	Fe^{2+}	1.76	Fe^{2+}	1.4	Medium	Medium
Location 2/SGC2	Zn^{2+}	2.2	Fe^{2+}	1.57	Medium	Medium

4.5 POSSIBLE LINK BETWEEN THE GROUNDWATER FLOW PATTERNS, THE LEACHATE GENERATED AT THE DUMPSITES AND THE QUALITY OF GROUNDWATER

For the Abule-Egba dumpsite vicinity the higher mean concentration of EC, TDS and Cl^- in the groundwater downgradient of the dumpsite (Table 4.3), when compared with lower mean concentration of the same parameters upgradient, suggest that some of the inorganic constituents of leachate from the dumpsite are being transported along the flow path into downgradient groundwater. Furthermore, the higher concentration of a highly mobile and conservative contaminant like Cl^- which is unreactive with the mineral components of the aquifer matrix, downgradient of the dumpsite points to the influence of leachate.

Similarly, the 2,490mg/L and 149mg/L TDS concentration of the leachate (Table 4.1) for the wet and dry season when compared with the mean concentration of 124.6mg/L and 197.55mg/L for the upgradient and downgradient locations for the wet season and the mean concentration of 81.96mg/L and 223.33mg/L for the upgradient and downgradient locations for the dry season, is an indication that some the leachate are being transported along the horizontal flow path.

The moderate level of groundwater contamination exhibited downgradient of the dumpsite, when a higher level of contamination would have been expected is attributed to two factors. First is the low contamination potential of the leachate as revealed by the computed LPI, and second is the likely influence hydrodynamic dispersion on the concentration of the leachate contaminants along the horizontal flow path. Hiscock (2005), in explaining the transport of contaminants in groundwater stated that the transportation and attenuation of conservative contaminants like chloride in groundwater is facilitated by hydrodynamic dispersion, which plays a significant role

in the reduction of contaminant concentration with distance away from the surface. The influence and extent of hydrodynamic dispersion in the dumpsite vicinity however calls for further study.

Furthermore, the absence of heavy metals in the downgradient groundwater is attributed to its low concentration in the leachate and its attenuation by the clayey strata of the subsurface during the migration of the contaminants along with the groundwater.

For the Solous Dumpsite vicinity the multi-directional groundwater flow pattern around Igando (Fig 4.2) implies that the migration of leachate from the dumpsites is likely to follow the different flow paths depicted in the model. This in essence suggests that the variations in the concentration of groundwater contaminants in the vicinity of the dumpsites may be partly due to the non-definitive flow pattern. Other processes include the diffusive transport of leachate occasioned by concentration gradient, dispersion due to the mixing of the contaminants at relatively high flow brought about by local variations in groundwater flow, and reactions, absorption or decay of some the leachate contaminants (Jhamnani & Singh, 2009).

CHAPTER FIVE

IMPLICATIONS OF THE RESULTS OF THE STUDY FOR WASTE DUMPSITE MANAGEMENT

The results of the study have a lot of implications for the current and future waste management practices at the LAWMA operated dumpsites. The implications of the results are presented as follows:

5.1 Groundwater Flow Patterns and Waste Dumpsite Management

5.1.1 Abule-Egba Dumpsite and Environs

The predominantly horizontal flow pattern exhibited by the groundwater flow model for Abule-Egba Dumpsite and its environs suggests the need to implement measures that will curtail the migration of leachate into the groundwater. This is regardless of the dumpsite being located in a low permeability hydrogeological environment, and the relative protection offered by the confining beds (clay) which in parts of the dumpsite extends to a depth of 12 metres. This is because coupled with the horizontal flow pattern, the groundwater in the vicinity of the dumpsite may become more susceptible to contamination if the integrity of the confining beds overlying the aquifers in the environment becomes compromised, and discrete pathways are created for groundwater flow and leachate transport (Bradury et al., 2006; LAWMA, 2008).

According to Bradbury et al (2006), these pathways could be inform of windows, fractures and macropores created by tree roots and burrowing animals, or human constructed pathways such as uncased or abandoned wells, or wells constructed with ineffective seal between well casing and borehole wall.

Measures that had been used in curtailing and minimising groundwater contamination around municipal waste dumpsites and landfills include the installation of interception walls and dewatering wells (Kim et al., 1999).

5.1.2 Solous Dumpsites and Environs

The non-definitive groundwater flow pattern around the Solous Dumpsites implies a need for constant groundwater quality monitoring by LAWMA, as there are no preferred flow directions, and the leachate generated at the dumpsites has the potential to migrate in multiple directions. The non-definitive flow pattern exhibited around the dumpsites is an indication of the need for leachate generated at the dumpsites to be minimised and contained. Incorporating this into the waste management practices especially at the active Solous 2 Dumpsite would reduce the risk of leachate contamination of the groundwater.

5.2 Characteristics of the Leachate Generated and Waste Dumpsite Management

The presence of heavy metals in the leachate generated at the three dumpsites and the effervescence of the leachate generated at Solous 2 Dumpsite during the wet season has a lot of implications not only for dumpsite management, but also for the environment and human health.

5.2.1 Presence of Heavy Metals in the Leachate

The non-containment nature of the dumpsites which allows for the leachate generated at the dumpsites to infiltrate the soil and form leachate springs during the wet season, means that the heavy metals contained in the leachate are allowed to contaminate the soil. The implication of this for groundwater quality is that once the heavy metals binding sites of the soil is filled up, and the metals retention capacity of the soil is reached, some of the heavy metals that had previously been immobilised by the clay and organic matter fraction of the soil, through the

process of adsorption and precipitation, would be desorbed and dissolved into the groundwater (Domenico & Schwartz as cited by Jagloo, 2002).

In the light of the current waste management practices and the presence of heavy metals in the leachate, highly mobile and less preferentially bound heavy metals such as Zn^{2+} would easily migrate into the groundwater due to its lesser affinity to be bound to a clay soil especially when compared to a heavy metal like Pb^{2+} which has a higher binding affinity (Lu, 2005). The risk of heavy metal contamination of groundwater posed by the leachate generated at these dumpsites may require a reduction in the disposal of waste from which heavy metals may potentially be leached. Such wastes include batteries, galvanised products, tires and so on.

5.2.2 Effervescence of Leachate

The effervescence of the leachate generated at Solous 2 Dumpsite during the wet season is indication of the emission of landfill gas at the dumpsite. Although the main compositions of landfill gas are Methane (CH_4) and Carbon-dioxide (CO_2), it also contains many trace Volatile Organic Compounds (VOCs) which are toxic and injurious to human and environmental health. The notion of the easy attenuation and destruction of contaminants by soil does not hold for VOCs. This is because in addition to resisting adsorption by fine grained soil, VOCs are highly mobile in soil and sediments, and can easily migrate into groundwater. Furthermore, VOCs are resistant to biological and chemical degradation (Battista & Johnston, 1989). Examples of VOCs contained in landfill gas include toxic chemicals like such as Benzene, Toulene, Chloroform, Vinyl Chloride, Carbon Tetrachloride, and Naphthalene. Some of the likely impacts of landfill gas occasioned by the effervescence of leachate such as observed at the Solous 2 Dumpsite include:

5.2.2.1 Impact on Groundwater Quality

The emission of landfill gas as indicated by the effervescence of the leachate generated at Solous 2 Dumpsite, suggests the probability of the dissolution of landfill gas into the groundwater, and a concomitant contamination by VOCs. According to Morris (n.d), p.587, "VOCs are transported with landfill gas through the subsurface in unsaturated soils away from landfill and are transferred from the vapour to the aqueous phase".

Although low permeability hydrogeological environments such as where the dumpsites are located provide fewer passageways for landfill gas to migrate, the existence of windows or fractures within the subsurface provides greater opportunities for the gases to migrate and dissolve in groundwater. The vulnerability of groundwater in the vicinities of the dumpsites to VOCs contamination is further increased if the top of well screens in with the environment is above the water table. According to Zumdahl as cited by Morris (n.d, p.588):

> In the event that the top of the well screen is above the water table, the gas migrating through the vadose zone would enter the well casing and fill the headspace. With a change in temperature and /or pressure, the VOCs contained within the landfill gas could then condense on the walls of the casing and run into the groundwater.

The concern about VOCs in drinking water stems from its persistence and human toxicity. As stated by Zogorski et al. (2007, p.9):

> The presence of elevated VOC concentrations in drinking water may be a concern to human health because of their potential carcinogenicity. In addition to cancer risk, VOCs may adversely affect the liver, kidney, spleen, stomach and heart, as well as the nervous, circulatory, reproductive, and respiratory systems. Some VOCs may affect cognitive abilities, balance or coordination, and some are eye, skin, and/or throat irritants.

The release and downward migration of Carbon-dioxide contained in the effervescent leachate also has the potential to further acidify the groundwater and increase its hardness, through the process of diffusive transport through the underlying clay. As stated by Tchobanoglous and Kreith (2002, p.14.20), "because carbon dioxide is readily soluble in water, it usually lowers the pH, which in turn can increase the hardness and mineral content of the groundwater through solubilization".

5.2.2.2 Impact on the Environment

Some of the environmental impacts associated with the effervescence of leachate which emits landfill gas are contribution to the green house effect, inhalation of toxic gases by humans, explosion hazard, and vegetation stress.

5.2.2.2.1 Contribution to the Green House Effect

The effervescence of the leachate generated at Solous 2 Dumpsite, and indeed the methanogenic degradation of waste at all the dumpsites is an that indication of the waste management practices at the dumpsites is contributing to global warming. This is because Methane and Carbon-dioxide which are components of landfill gas are also green house gases responsible for the greenhouse effect which prevents the long wave radiation reflected from the earth from escaping into the upper atmosphere.

5.2.2.2.2 Inhalation of Toxic Gases by Humans

Other components of landfill gas such as Hydrogen Sulphide (H_2S) and VOCs such as Benzene (C_6H_6), Perchloroethylene (PCE), Trichloroethylene (TCE) and Vinyl Chloride (H2C:CHCl) can become toxic to humans (especially dumpsite workers and nearby residents) when inhaled at high concentrations. According to the United States Agency for Toxic Substance and Disease Registry [ATSDR], (2001), the afore mentioned gases can become

toxic to humans when inhaled in large concentrations, leading to breathing difficulties, nausea, dizziness, headaches, central nervous system problems and certain types of cancer.

5.2.2.2.3 Explosion Hazard

Buildings adjacent to these dumpsites are exposed to explosion hazard from the build up of Methane which if allowed to migrate through the soil and fractures present within the subsurface could lead to explosion under favourable circumstances.

5.2.2.2.4 Vegetation Stress

"Vegetation is typically adversely affected by landfill gases and low oxygen concentrations on gas-emitting from a landfill" (Ankeny & Stormberg, n.d, p.4). This is because during the upward movement of Methane and Carbon-dioxide from the waste, the oxygen needed for the growth of plant is displaced and consumed by soil microbes especially in the presence of Methane.

In summary the characteristics of the leachate generated at the dumpsites suggests the need for both groundwater quality monitoring and the control and monitoring of the landfill gas emanating from the dumpsites in order to minimise both its environmental and health implications.

5.3 Groundwater Quality, Contamination Index and Waste Dumpsite Management

The higher concentration of TDS in the groundwater in the immediate vicinity of Abule-Egba Dumpsite for the wet and dry seasons, when compared with the levels of TDS at other locations indicates that contaminants leached from the waste deposited at the dumpsite had migrated into the groundwater immediately downgradient. This conclusion is further corroborated by the especially high chloride concentration of the groundwater at the same location during the dry season. This is because an "excess of Cl^- in water is usually taken as

index of pollution and considered as tracer for groundwater contamination" (Loizidou & Kapetanios, as cited by Mor, et al., 2006, P.10).

Downgradient of Solous 2 Dumpsite the elevated levels of TDS and Cl^- (for both seasons) in the groundwater samples collected between 210 and 245m is an indication of leachate migration into groundwater. According to Olaniya and Saxena as cited by Mor et al., (2006), high concentration of TDS is one of the indicators of groundwater contamination around open dumpsites.

With regards to the elevated TDS and Cl^- concentration of groundwater in the immediate vicinities of Abule-Egba and Solous dumpsites, there is a need for the volume of leachate generated at the dumpsites to be controlled in order to limit the volume leachate that could infiltrate the soil and potentially get into the groundwater. This would mean the application of daily cover of impermeable clay in order to limit the infiltration of water (especially during the wet season), and invariably the volume of leachate generated. The daily cover would also serve to control other nuisances, and help protect the health of the dumpsites workers and the general environment.

Managing and reducing the volume of leachate generated at the dumpsites would also require the planting of vegetation at the dumpsites in order to increase the rate of evapotranspiration from the dumpsites.

With reference to the contamination index and the predominant contamination parameter which is Fe^{2+}, reduction in the volume of Iron based waste received to the dumpsites, would be a good waste management strategy that could further reduce the concentration of Fe^{2+} that could potentially get leached and mobilised into the groundwater. Examples of Iron based products that could contribute to increased Fe^{2+} concentration in leachate, and subsequently

the groundwater around the dumpsites include construction materials, paints, pigments, colour compounds, polishing agents and electrical materials (Aziz, et al., 2004).

5.4 Waste Management Policies in Lagos

A number of policy and administrative framework, as well as legal instruments exist to guide waste management practices and operations in Lagos, and the country at large. Some of these include the National Policy on the Environment (1988), National Environmental Standards and Regulations Enforcement Agency Act [NESREA Act] (2007), Environmental Impact Assessment Act (1992), National Environment [Sanitation and Waste Control] Regulations (2009), the Harmful Waste Decree [Special Criminal Provisions] Act (1990), Environmental Protection Agency Decree (1988) and National Environmental Protection Regulations-Management of Solid /Hazardous Waste (1991).

Others include National Effluent Limitations Regulations (1991), Solid and Hazardous Waste Management Regulations (1999), Lagos State Environmental Protection Agency [LASEPA] Edict (1999), Lagos State Waste Management Authority [LAWMA] Edict (2007), Lagos State Environmental Sanitation Law (2000) and National Guidelines and Standards for Environmental Pollution Control in Nigeria (1991).

In spite of the existence of these legal, administrative and policy framework however, not much has been achieved in terms of sustainable waste practices. This is because most of these regulations and standards poorly enforced, and the required pollution abatement measures are either non-existent or grossly inadequate.

In view of the various implications identified in sections 5.1 to 5.3 and the continuous threat that open dumping poses to the environment and groundwater quality in particular, it is advocated that waste management policies in Lagos be geared towards the adoption of

alternative technologies. This is in order to minimise some of the adverse consequences of waste disposal by open dumping.

Some of the available alternative technologies to waste disposal by open dumping include recycling, composting, incineration, anaerobic digestion, gasification and pyrolysis. In light of the cost implications of some of these technologies however, it is advocated that recycling and composting be fully integrated into the policy framework for solid waste management in the state. This is due to the low level of adverse environmental impacts associated with these technologies. With regards to recycling for instance, waste that would otherwise have been deposited at the dumpsites are diverted and converted into other useful products, while the adoption of composting of organic and biodegradable waste greatly reduces the volume of waste deposited at the dumpsites.

CHAPTER SIX

SUMMARY OF FINDINGS, CONCLUSION AND RECOMMENDATIONS

6.1 Summary of Findings

Based on the focus and objectives of this study, the results of the characterisation and groundwater flow modelling of the study locations, established a predominantly horizontal groundwater flow pattern for Abule-Egba, and a multi-directional flow pattern for the Solous Dumpsites environs. The characterisation and assessment of the contamination potential of the leachate generated at the dumpsites adjudged the leachates generated at the dumpsites as having low contamination potential.

The assessment of the quality of groundwater around the dumpsites showed that leachate generated at the dumpsites has had a minimal impact on the quality of groundwater around the study locations. This situation is attributed to the natural attenuation of leachate contaminants by the prevailing hydrogeological environment. In the same vein, the results of the computed Groundwater Contamination Index categorised most of the groundwater samples obtained within the study locations as having a medium level of contamination. The study further revealed that a combination of waste disposal practices and natural weathering of aquifer materials is mostly responsible for the observed variation in the quality of groundwater. The spatial pattern of the hydrochemical parameters responsible for most of the variation in groundwater quality as established by the Inverse Distance Weighting method of spatial interpolation, showed the estimated concentration of the concerned hydrochemical parameters at unsampled locations around the dumpsites. With regards to the possible linkages between groundwater flow pattern, leachate and the quality of groundwater around the dumpsites, the higher mean concentration of some hydrochemical parameters such as

Chloride in downgradient groundwater is attributed to the transportation of some of the contaminants along the groundwater flow path.

Some identified implications of the study include increased vulnerability of groundwater to contamination in the event of compromise of the integrity of confining layers, the increased likelihood of heavy metal contamination of groundwater if the waste management approach remains unchanged. Others include the probability of groundwater contamination by volatile organic compounds emitted from landfill gases generated as a result of leachate effervescence, and the contribution of landfill gases to global warming and vegetation stress.

6.2 Contribution to Knowledge

In terms of contribution to knowledge this study has been able to:

- Provide an understanding of the groundwater flow patterns and the possible migration pathways of the contaminants within and around the dumpsites in order to guide future locations of wells and boreholes around the dumpsites. Information provided on the flow patterns and migration pathways of contaminants would also guide waste management practices at the dumpsites, so that future groundwater contamination can be forestalled.

- Provide information on the characteristics of leachate generated at the dumpsites and their potential to contaminate groundwater resources in the vicinities of the dumpsites through the utilisation of LPI. This information can be used to develop leachate monitoring programmes at the dumpsites, and other dumpsites in Lagos State. Furthermore this information can be used as a tool by LAWMA, the environment ministry and other relevant agencies to develop and enforce leachate discharge standards for all dumpsites in the state. This information can also be used as a

decision making and planning tool for waste disposal and management at the dumpsites.

- Identify the contaminants responsible for most of the variance in the quality of groundwater around the dumpsites, and the factors and processes responsible for such variance. The study has been able to show that a combination of waste disposal and lithogenic inputs from the interaction of the underlying geology with groundwater is responsible for most of the variance in the quality of groundwater.

- Provide data and information concerning the quality status of groundwater around the dumpsites, and the role that the hydrogeology plays in determining the level of contamination and attenuation of some heavy metals.

- Provide information on the possible impacts of the current waste management practices on the quality of groundwater and the environment with a view to evolving more environmentally friendly waste management practices at the various dumpsites. Furthermore, information provided by this study can serve as a guide for future waste management operations at current and yet to be established dumpsites within the state.

6.3 Conclusion

Leachate generated from various dumpsites located in Lagos represents a major threat to groundwater quality within their environs for a number of reasons ranging from the indeterminate composition of waste tipped at the dumpsites, to lack of environmental safeguard measures and general shortcomings in design, construction and maintenance of the dumpsites. Other reasons held responsible for the continued threat include low level, and sometimes non-implementation of existing environmental and groundwater regulations, and absence of monitoring programs geared towards the sustenance of groundwater integrity. The

current state of waste dumpsites in Lagos therefore brings to the fore, the need for the characterisation of groundwater flow and quality around the dumpsites.

The results of the groundwater flow characterisation for Abule-Egba and the Solous Dumpsites environs suggests the need for a better understanding of the hydrogeology and groundwater movement in the area to prevent further inappropriate siting of wells and boreholes in the areas.

In spite of the minimal level of groundwater contamination and the relative protection the underlying geology offers to the quality of groundwater resources around the dumpsites, there is a need for better waste disposal operations and management at the dumpsites. This is because groundwater resources around the dumpsites stand a greater risk of contamination if this business as usual approach continues unabated.

6.4 Recommendations

The following recommendations are made to LAWMA, environmental regulatory bodies, other concerned government institutions and the residents of the dumpsites environs:

- In order to reduce the vulnerability of groundwater to leachate contamination as a result of the current waste management at the dumpsites, groundwater quality monitoring programmes should be instituted and incorporated into the waste management operations of LAWMA, and the environmental monitoring programmes of relevant institutions. Periodic or regular monitoring of the status of groundwater in the vicinities of the dumpsites is needed in order to effectively monitor the impacts of the dumpsites on groundwater resources in the area.
- The existing policies concerning waste disposal and management should be strictly enforced and adhered to in order to ensure environmentally friendly and sustainable waste management operations at the dumpsites. This would also go a

long way in reducing the risk of groundwater contamination around the dumpsites.

- In view of the lack of environmental safeguard measures (e.g bottom liner, leachate collection and treatment system) at the dumpsites, reduction of the volume of leachate generated at the dumpsites, especially during the rainy season should be given priority. Detailed hydrogeological and groundwater flow characterisation should be carried out before the commencement of waste disposal at proposed or future dumpsites in Lagos, in order determine the suitability of such sites for waste disposal activities.

- Groundwater flow and contaminant transport modelling should be incorporated into any groundwater quality monitoring programmes developed for groundwater and solid waste management within Lagos. The incorporation of contaminant and groundwater flow modelling in groundwater and solid waste management has the added advantage of reducing the cost of groundwater quality monitoring in the long run.

- More effort should be geared towards educating the citizenry on the 3-R principle of reduce, reuse and recycle in order to reduce the volume of waste that could potentially get deposited at the dumpsites. This is with a view to reducing the volume of waste from which contaminants may potentially be leached.

- It is recommended that in siting of future dumpsites a buffer zone of at least 3 miles be allowed between the dumpsites and nearest source of groundwater abstraction, in order to reduce the potential for the consumption of contaminated groundwater. According to Lee and Jones-Lee (2005), 3 miles of aquifer buffer is considered an appropriate distance for minimising the risk of groundwater

contamination in aquifers used as a source of domestic water supply in an environment where a landfill is sited.

- Efforts should be intensified on the part of the Lagos State Government, through the LSWC to provide pipe-borne water and ensure adequate coverage of its water distribution network within the study locations, in order reduce the dependence of residents on groundwater.

- New wells around the dumpsites should be drilled deeper in order to be assured of getting a more potable source of water. Based on the hydrogeological profile of the dumpsites, it is suggested that wells be drilled deeper than 42 metres and 31 metres at Abule-Egba and Solous dumpsites vicinity respectively. It is also suggested that the upper sections of the wells be grouted off, in order to prevent contamination from leachate. It is also recommended that residents who can afford the cost of drilling deeper should drill into the third aquifer in order to be further assured of accessing water of better quality.

6.5 Future Research Directions

Presently, it is obvious that waste management practices in Lagos and the whole country is poorly regulated, and falls below accepted best practices both nationally and internationally. The current state of affairs has led to a situation where these dumpsites have become a permanent source of environmental challenge. In view of these and the results of this current study, the following are identified as future research directions:

- In depth study on the groundwater flow regime and transportation of contaminants (leachate plume) within the subsurface.
- In depth investigation into the natural attenuation of different types of leachate contaminants within the subsurface.

- Monitoring of groundwater quality in the dumpsites environs.
- Study on the extent of compliance of waste management practices at the dumpsites with policies and guidelines that addresses the issues of waste management.
- Study on the implications of the use of groundwater in the dumpsites vicinities.

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APPENDICES

APPENDIX 1

HYDROGEOLOGICAL PROFILE OF ABULE-EGBA DUMPSITE

From the hydrogeological profiles of the three monitoring boreholes drilled at the dumpsite, the top of the aquifers are delineated at a depth of 19 to 20 metres in borehole 1, 24 metres in borehole 2, and 30 metres in borehole 3. The dark coloured topsoil at a depth of 0 to 4 metres, and the dark colour of clay between 4 and 7 metres in borehole 1 is an indication of well decomposed organic matter, which is a product of biodegradable waste deposited at the dumpsite. According to Foth (1990), soils with black or nearly black colour are indicative of well decomposed organic matter such as humus. The presence of humus to a depth of 4 metres in borehole 1 suggests the existence of the potential for the immobilisation of some of contaminants present in the leachate generated at the dumpsite. According to Burden and Sims (1999), the presence of many highly reactive surfaces within soil organic matter may help in retaining waste constituents within the soil system. As stated by Burden and sims (1999, p.3), "hydrophobic organic constituents may be sorbed onto soil organic matter and thus become less mobile in the soil system".

The colour and textural classes of the soil making up each lithostratigraphic unit is very important. This is because it provides information on the hydrologic regime of the subsurface. According to Foth (1990, p.38), "subsoil colours are very useful in predicting the likelihood of subsoil saturation with water and poor aeration". Furthermore, the soil textural classes provide information on the occurrence and presence of aquifers and confining beds within the subsurface.

With regards to the presence of aquifers and various types of confining or impermeable layers beneath the dumpsite, the lithological profile of borehole 1 reveals the presence of confining beds/ layers (clay) between the depth of 4 and 12 metres within the subsurface. The confining layer (clay) present between 4 and 12 metres within the subsurface can be characterised as an aquiclude. “An aquiclude is a saturated but relatively impermeable material that does not yield appreciable quantities of water to wells; clay is an example” (Todd & Mays, 2005, p.37). In boreholes 2 and 3, clay was encountered within a depth of 3meters below sea level. The hydrogeologic implication of the presence of the aquiclude is that the percolation of water and the downward movement of contaminants such as leachate will be impeded due to its low transmitting capacity.

Furthermore, the reddish colour of the soil within the upper profiles of the boreholes indicates that the profiles are unhydrated and oxidised. As explained by Burden and Sims (1999), red soil colour is usually associated with oxidised Iron Oxide and unhydrated conditions. The unhydrated and aerobic conditions as suggested by the red and reddish brown colours of the profiles further justifies these sub-strata as the vadose zones within the boreholes and around the dumpsite. Within borehole 1, the whitish colour of the silty-clayey sand between the depth of 20 and 22 metres is an indication of saturated conditions.

The sandy clay profiles encountered at depths 6 to 9 metres, and 18 to 24 metres at borehole 2, and between 3 to 6 metres, 9 to 12 metres, 15 to 18 metres and 21 to 24 metres at borehole 3 confirms these hydrostratigraphic units as aquitards.“An aquitard is a saturated but poorly permeable stratum that impedes groundwater movement and does not yield water freely to wells” (Todd & Mays, 2005, p.37). A good example of an aquitard is sandy clay (Todd & Mays, 2005).

The occurrence of medium grained and medium coarse grained sand between the depth of 9 and 15 metres in borehole 2 is indicative of the presence of a leaky aquifer above the main aquifer. A leaky aquifer occurs "where a permeable stratum is overlain and underlain by a semi-pervious aquitard or semi confining layer" (Todd & Mays, 2005, p.57). The series of impermeable and semi-permeable (confining) layers overlying the aquifer at borehole 3 indicates that the aquifer is a confined aquifer. The locations of the 3 monitoring boreholes drilled at the dumpsite are presented in Figure 1, while 3-Dimensional representations of the lithological profile of the boreholes are presented in Figures 2 to 4.

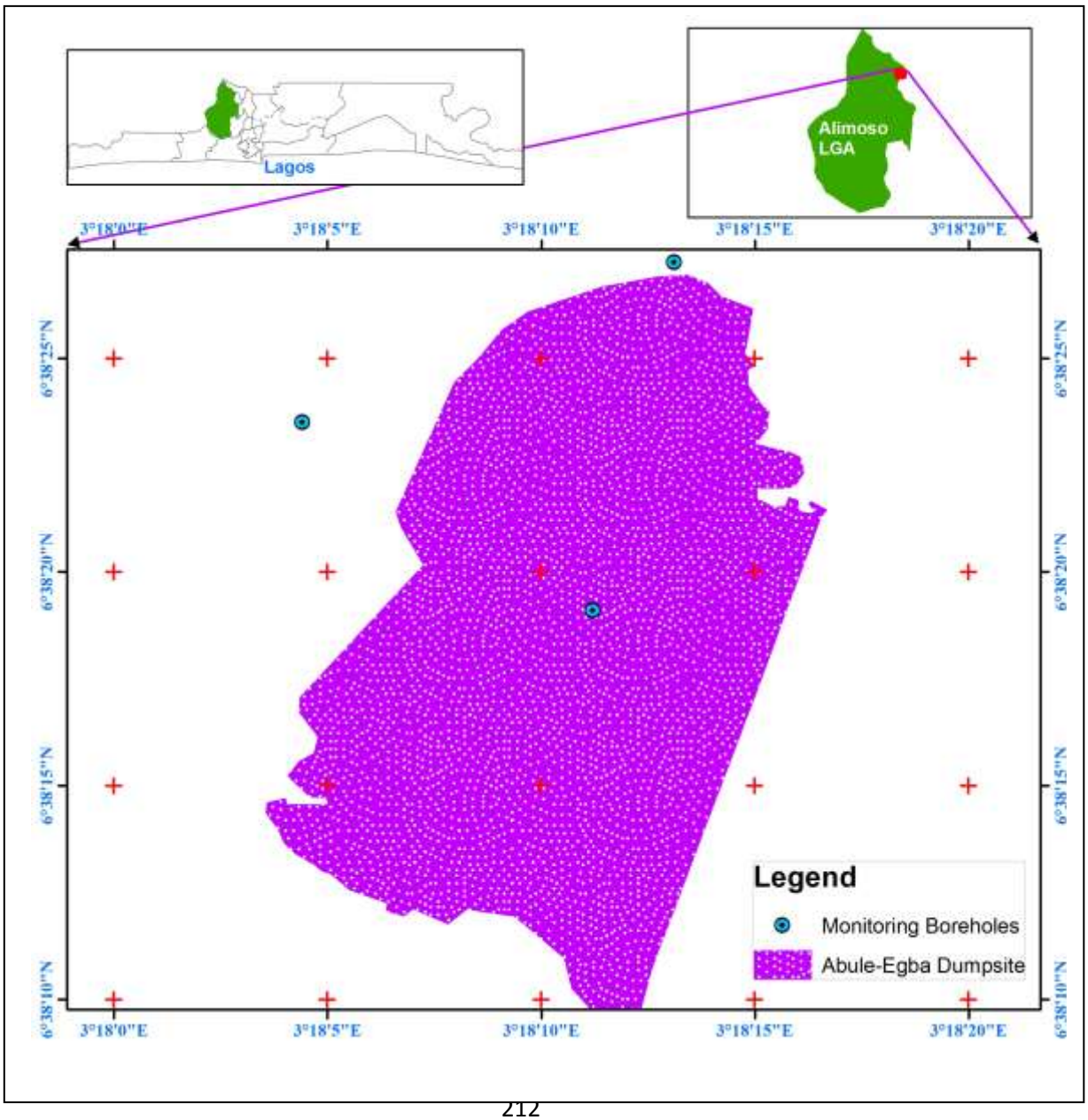


Fig 1: Location of the Monitoring Boreholes at Abule-Egba Dumpsite

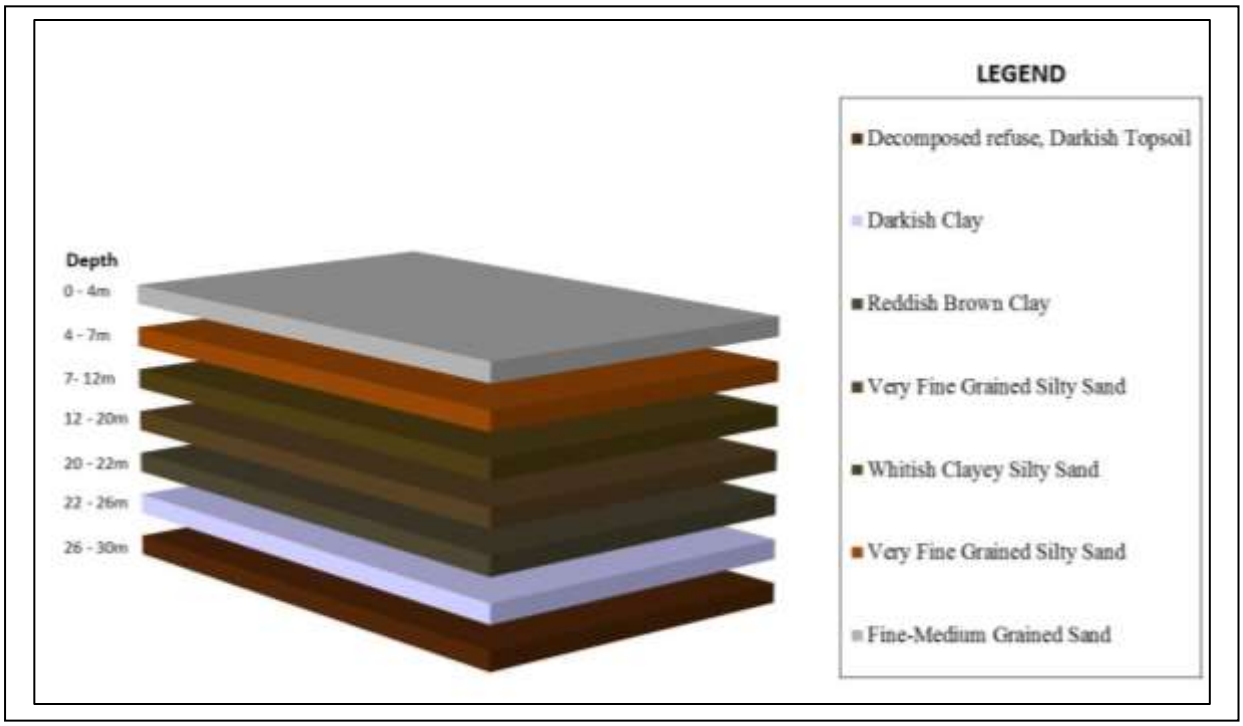


Fig 2: 3D Representation of the lithological profile of Borehole 1 at Abule-Egba Dumpsite

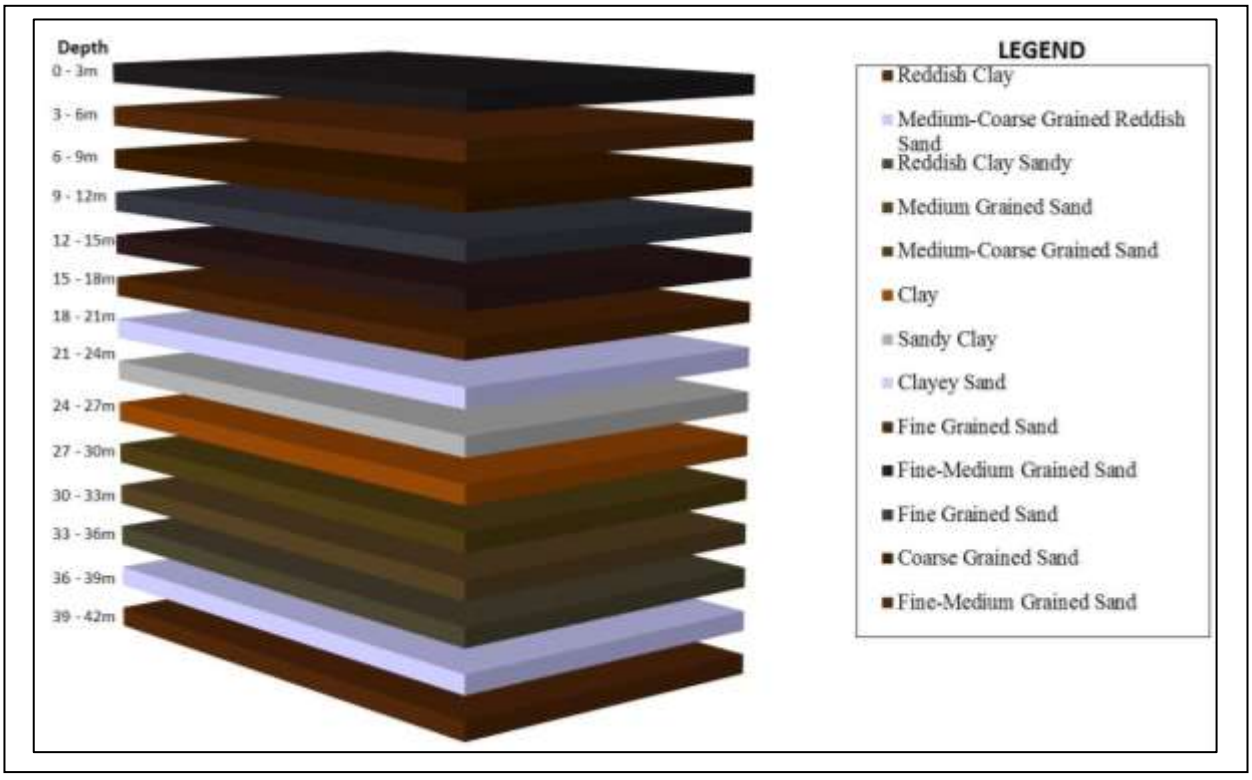


Fig 3: 3D Representation of the lithological profile of Borehole 2 at Abule-Egba Dumpsite

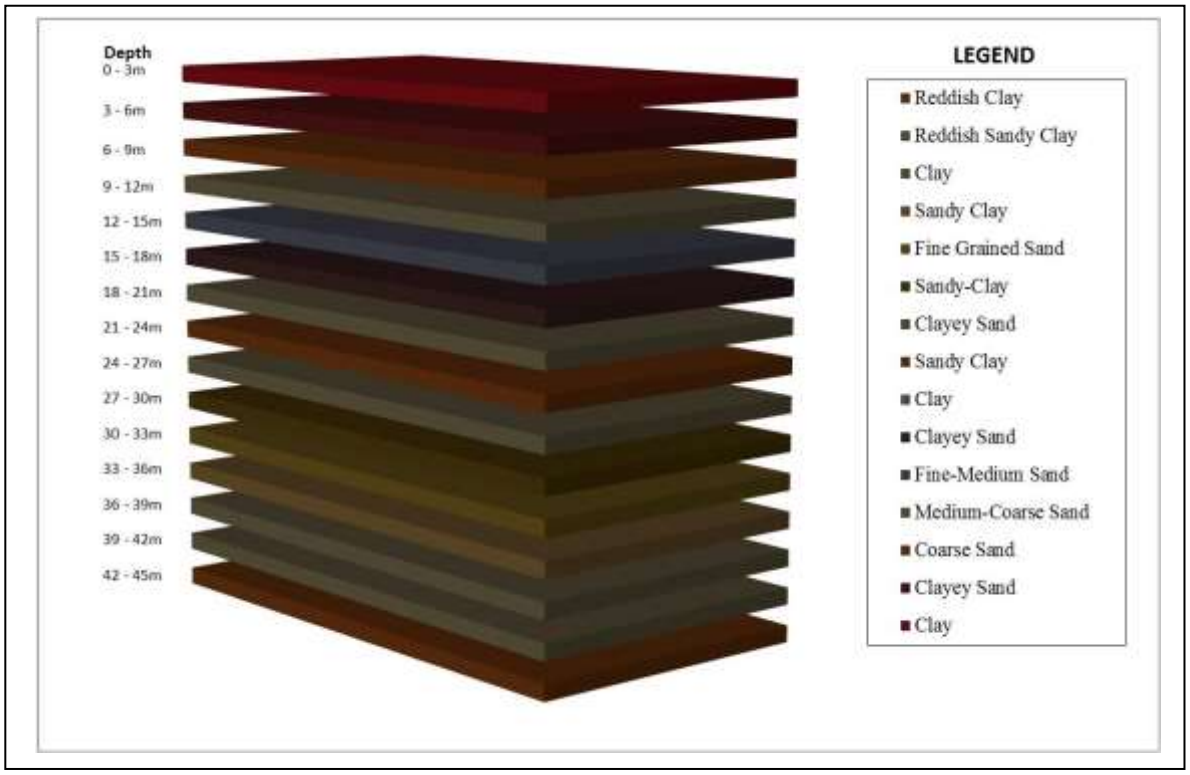


Fig 4: 3D Representation of the lithological profile of Borehole 3 at Abule-Egba Dumpsite

APPENDIX 2

HYDROGEOLOGICAL PROFILE OF SOLOUS DUMPSITES

The hydrogeological profiles of the three monitoring boreholes drilled at Solous 1 dumpsite revealed that the top of the first aquifer was encountered a depth ranging between 10 to 12 metres at borehole 1, 8 to 10 metres at borehole 2, and 16 to 18 metres at borehole 3. This suggest that the vadose zone in boreholes 1 and 2 are within a depth 0 and 10 metres below sea level, while the vadose zone in borehole 3 is between a depth of 0 to 16 metres below sea level. The whitish colour of the clay encountered between 10 to 12 metres, and 27 to 28 metres in borehole 1, between 16 to 19 metres, and 26 to 29 metres in borehole 3 are indicative of the saturation of soil at these depths. The locations of the 3 monitoring boreholes are shown in Figure 5, while the 3-Dimensional representations of their lithological profiles are presented in Figures 6 to 8.

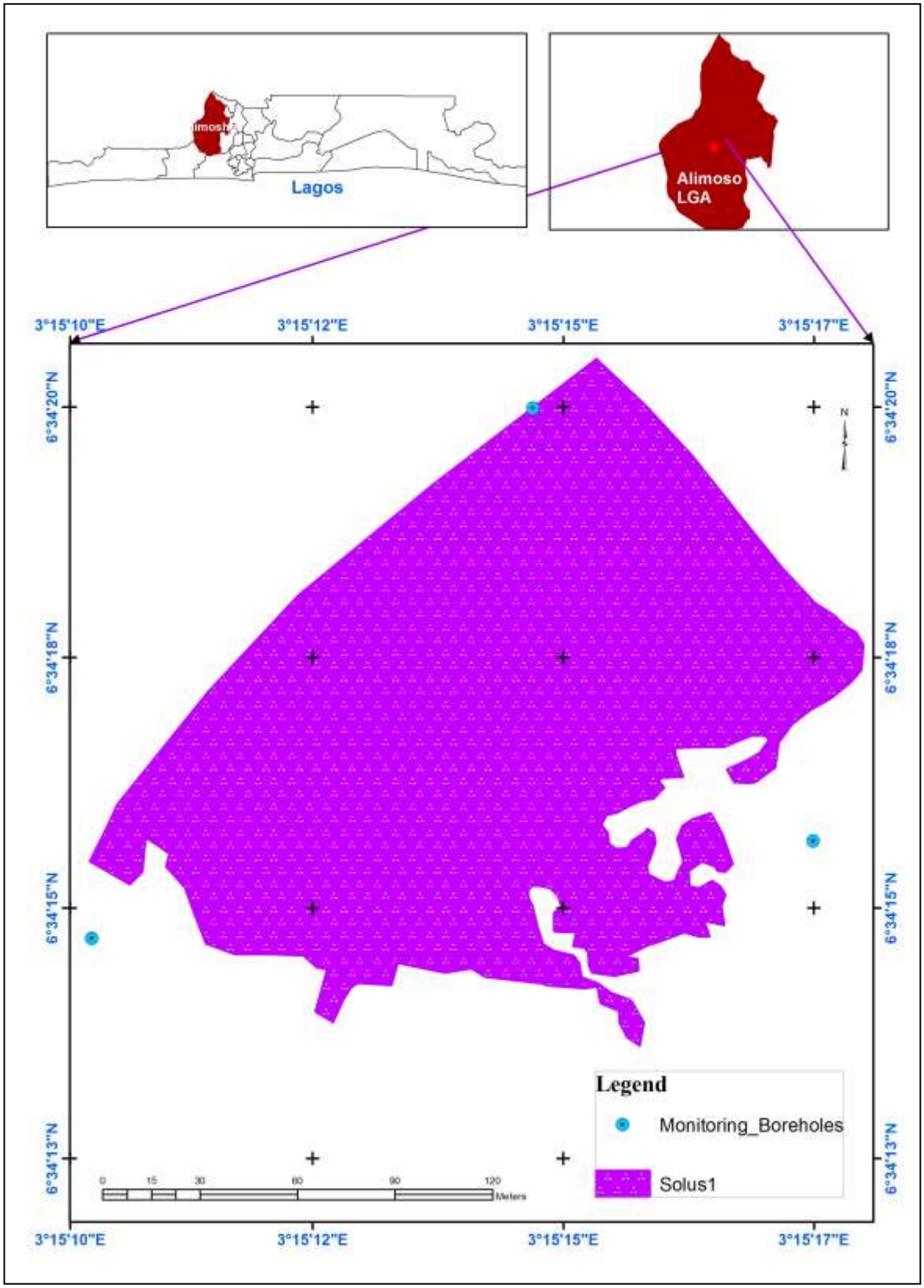


Fig 5: Location of the Monitoring Boreholes at Solous 1 Dumpsite

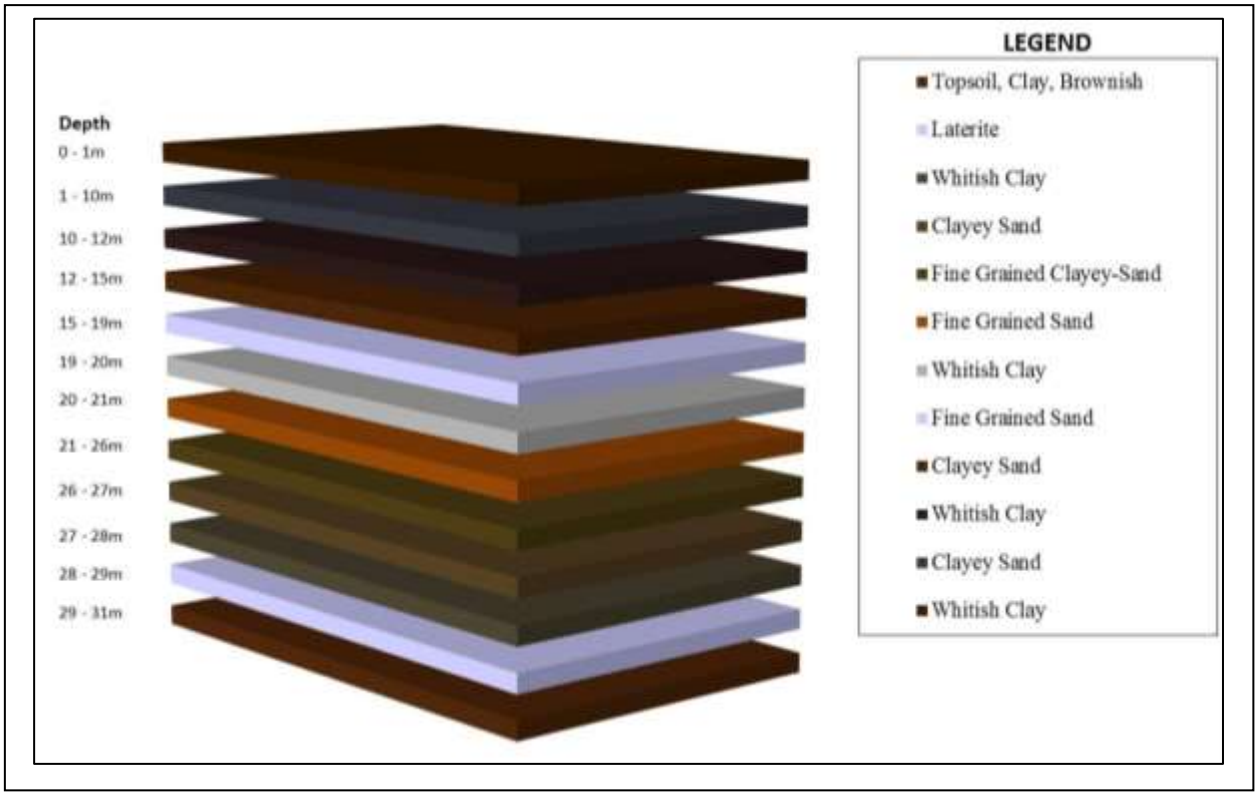
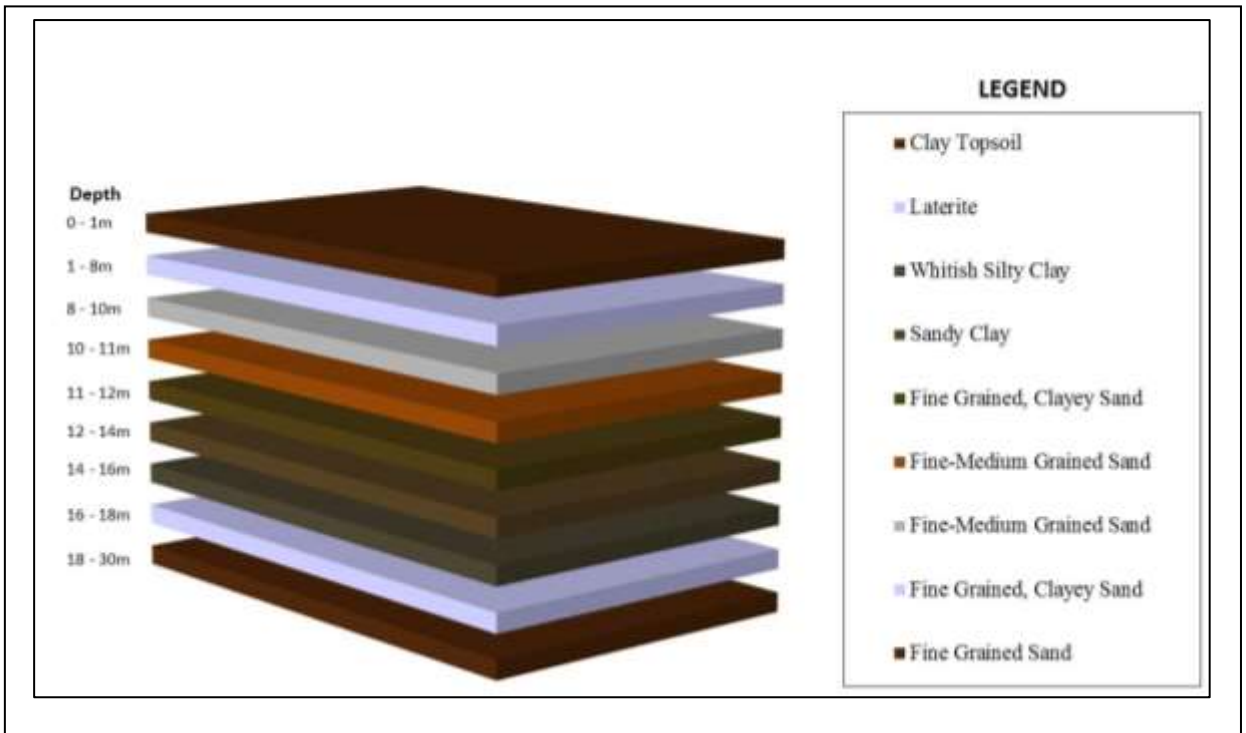


Fig 6: 3D Representation of the lithological profile of Borehole 1 at Solous 1 Dumpsite



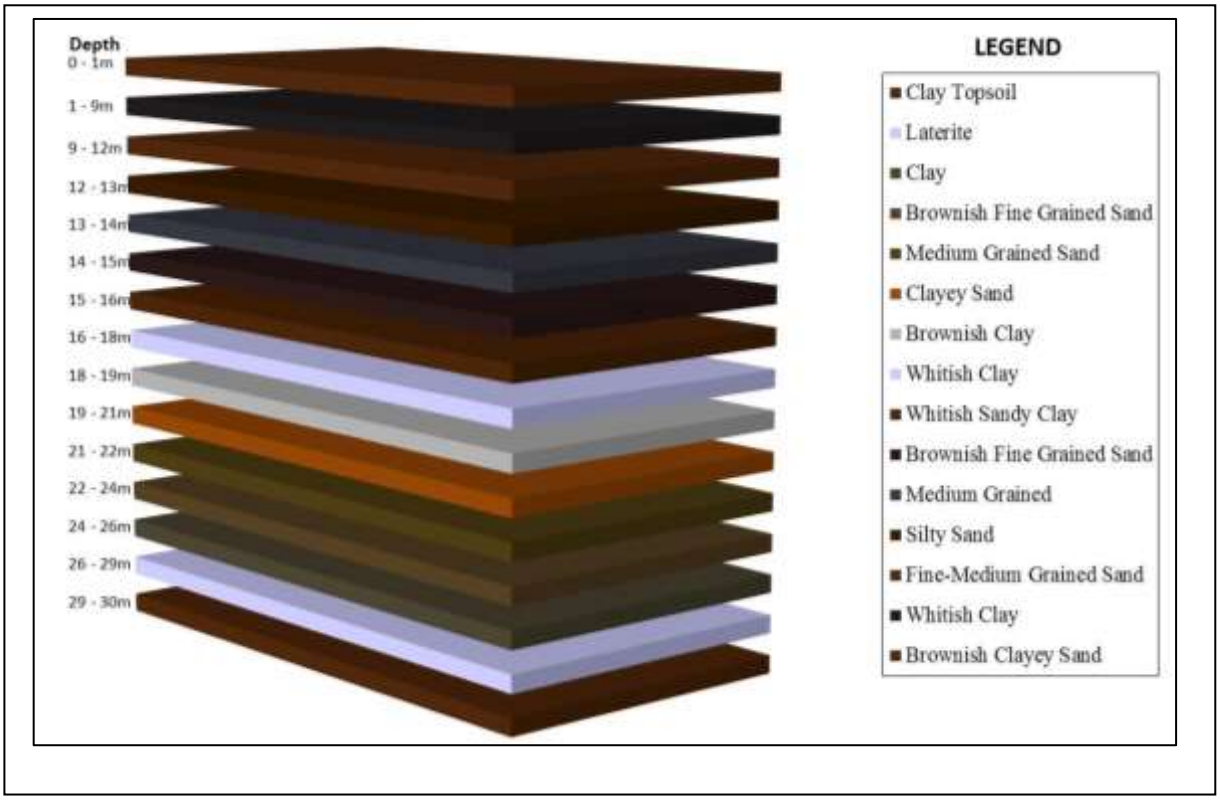


Fig 8: 3D Representation of the lithological profile of Borehole 3 at Solous 1 Dumpsite

APPENDIX 3

Chemical Constituents of Groundwater in the Vicinity of Abule-Egba Dumpsite for the Wet and Dry Seasons

	Wet Season							Dry Season						
Location & Sample Code	pH	EC (µS/cm)	TDS (mg/L)	SO_4^{2-} (mg/L)	NO^{3-} (mg/L)	Cl^- (mg/L)	Fe^{2+} (mg/L)	pH	EC (µS/cm)	TDS (mg/L)	SO_4^{2-} (mg/L)	NO^{3-} (mg/L)	Cl^- (mg/L)	Fe^{2+} (mg/L)
Location 1 /AGU1	5.04	609.0	450	ND	1.94	40.0	0.32	5.11	367.0	255	6.1	3.5	148.0	0.42
Location 2/AGU2	5.03	51.2	26.2	ND	0.6	30.0	0.1	4.83	56.8	27.5	2.1	0.23	24.0	0.38
Location 3/AGU3	5.53	171	112.0	ND	1.34	50.0	0.57	5.11	51.7	25.6	2.0	0.31	32.0	0.45
Location 4/AGU4	5.08	236.0	117.0	ND	0.8	40.0	1.1	4.55	269.0	133.0	1.61	0.45	20.0	0.23
Location 5/AGU5	4.97	68.4	67.9	0.17	0.4	30.0	0.61	4.68	87.1	53.6	15	6.1	28.0	0.51
Location 6/AGU6	5.28	42.7	21.3	ND	0.55	40.0	0.72	5.2	46.2	22.9	1.0	0.22	16.0	0.82
Location 7/AGU7	4.93	115.0	75.3	ND	0.9	40.0	0.09	4.74	109.0	56.1	1.1	0.07	28.0	0.25
Location 8/ADG1	7.22	412.0	211.0	0.62	3.35	80.0	0.09	5.23	347.0	173.0	2.0	1.32	72.0	0.13
Location 9/ADG2	6.57	587.0	296.0	0.55	4.56	120.0	0.70	6.67	1039.0	516.0	19.0	1.42	188.0	0.18
Location 10/ADG3	5.99	606.0	301.0	0.49	4.87	116.0	0.66	3.81	623.0	310.0	9.0	2.3	88.0	0.15
Location 11/ADG4	5.94	250.0	115.0	0.77	1.94	52.0	0.89	4.62	156.0	77.0	5.0	1.44	36.0	0.1
Location 12/ADG5	5.73	362.0	181.0	0.82	2.85	64.0	0.99	5.73	401.0	200.0	2.0	1.35	72.0	0.08
Location 13/ADG6	5.88	122.0	81.3	0.52	1.0	60.0	0.82	4.1	132.0	64.0	1.0	1.1	28.0	0.13
Location 14/AGC1 CLRTL 1	5.15	41.1	26.3	ND	0.5	30.0	0.66	5.66	41.6	20.9	2	0.2	16.0	0.29
Location 15/AGC2 CLRTL 2	5.12	280.0	140.0	ND	2.5	70.0	0.27	5.13	364.0	167.0	7.1	3.8	64.0	0.11

Source: November, 2010; March 2011

*ND represents not detected

APPENDIX 4

Heavy Metal Constituents of Groundwater in the Vicinity of Abule-Egba Dumpsite for the Wet and Dry Seasons

	Wet Season				Dry Season			
Location & Sample Code	Zn^{2+} (mg/L)	Pb^{2+} (mg/L)	Ni^{2+} (mg/L)	Cd^{2+} (mg/L)	Zn^{2+} (mg/L)	Pb^{2+} (mg/L)	Ni^{2+} (mg/L)	Cd^{2+} (mg/L)
Location 1 /AGU1	1.35	ND	ND	ND	0.22	ND	ND	ND
Location 2/AGU2	1.22	ND	ND	ND	2.11	ND	ND	ND
Location 3/AGU3	3.9	ND	ND	ND	0.24	ND	ND	ND
Location 4/AGU4	1.33	ND	ND	ND	0.14	ND	ND	ND
Location 5/AGU5	1.33	ND	ND	ND	0.28	ND	ND	ND
Location 6/AGU6	1.34	ND	ND	ND	0.22	ND	ND	ND
Location 7/AGU7	1.39	ND	ND	ND	0.12	ND	ND	ND
Location 8/ADG1	0.69	ND	ND	ND	0.40	ND	ND	ND
Location 9/ADG2	0.82	ND	ND	ND	0.28	ND	ND	ND
Location 10/ADG3	0.59	ND	ND	ND	0.03	ND	ND	ND
Location 11/ADG4	0.42	ND	ND	ND	0.04	ND	ND	ND
Location 12/ADG5	0.51	ND	ND	ND	0.03	ND	ND	ND
Location 13/ADG6	0.62	ND	ND	ND	ND	ND	ND	ND
Location 14/AGC1 CLRTL 1	1.20	ND	ND	ND	2.01	ND	ND	ND
Location 15/AGC2 CLRTL 2	4.12	ND	ND	ND	0.24	ND	ND	ND

Source: November, 2010; March 2011

*ND represents not detected

APPENDIX 5

Chemical Constituents of Groundwater in the Vicinity of Solous 1 and Solous 2 Dumpsites for the Wet and Dry Season

	Wet Season							Dry Season						
Location Sample Code	pH	EC (µS/cm)	TDS (mg/L)	SO_4^{2-} (mg/L)	NO^{3-} (mg/L)	Cl^- (mg/L)	Fe^{2+} (mg/L)	pH	EC (µS/cm)	TDS (mg/L)	SO_4^{2-} (mg/L)	NO^{3-} (mg/L)	Cl^- (mg/L)	Fe^{2+} (mg/L)
SOLOUS 1														
Location 1 /S1GWI	6.79	26.0	13.0	ND	0.52	30.0	0.60	7.29	26.4	12.8	ND	0.04	16.0	0.47
Location 2/S1GW2	7.24	91.4	47.6	ND	1.30	60.0	0.66	7.91	104.0	51.7	2.0	0.41	20.0	0.64
Location 3/S1GW3	7.21	33.1	16.1	ND	0.90	40.0	0.70	7.33	161.0	80.1	19.20	16.4	36.0	0.72
Location 4/S1GW4	6.60	52.2	25.7	ND	0.05	30.0	1.3	5.37	56.2	29.0	2.10	0.21	20.0	0.12
Location 5/S1GW5	6.33	30.9	15.3	ND	1.31	50.0	0.4	5.78	28.1	13.3	4.3	3.6	100.0	0.43
Location 6/S1GW6	6.75	172.0	88.1	ND	0.8	40.0	0.05	6.55	196.0	96.4	55.2	33.7	40.0	0.35
SOLOUS 2														
Location 1 /S2GWI	6.37	258.0	132.0	ND	8.0	360.0	0.41	7.44	251.0	126.0	24.1	21.6	348.0	0.20
Location 2/S2GW2	7.13	572.0	282.0	0.17	6.2	160.0	0.49	6.08	274.0	198.0	43.5	34.2	232.0	0.22
Location 3/S2GW3	7.10	304.0	233.0	0.52	9.09	470.0	2.20	5.72	690.0	329.0	4.10	1.23	156.0	0.21
Location 4/S2GW4	7.55	82.5	41.1	ND	4.9	130.0	1.21	7.34	266.0	135.0	22.10	17.6	348.0	0.48
Location 5/S2GW5	6.78	713.0	488.0	ND	0.53	30.0	0.44	7.40	201.0	101.0	105.0	86.4	392.0	0.14
Location 6/S2GW6	6.49	358.0	184.0	ND	1.42	66.0	0.21	5.88	566.0	382.0	12.2	8.4	548.0	0.14
Location 7/S2GW7	6.89	177.0	91.0	0.01	0.72	33.0	1.10	6.35	380.0	194.0	2.5	0.36	28	0.17
CONTROL SITES														
Location 1/SGC1	6.4	59.0	38.5	ND	0.43	30	0.53	5.38	62.1	37.7	8.10	2.14	24.0	0.42
Location 2/SCC2	6.44	102.0	66.5	0.17	3.90	290	0.30	5.33	97.5	48.1	3.40	0.56	112.0	0.47

Source: November, 2010; March 2011

*ND represents not detected

APPENDIX 6

Heavy Metal Constituents of Groundwater in the Vicinity of Solous 1 and Solous 2 Dumpsites for the Wet and Dry Seasons

	Wet Season				Dry Season			
Location & Groundwater Sample Code	Zn^{2+} (mg/L)	Pb^{2+} (mg/L)	Ni^{2+} (mg/L)	Cd^{2+} (mg/L)	Zn^{2+} (mg/L)	Pb^{2+} (mg/L)	Ni^{2+} (mg/L)	Cd^{2+} (mg/L)
SOLOUS 1								
Location 1 /S1GWI	1.33	ND	ND	ND	0.27	ND	ND	ND
Location 2/S1GW2	4.50	ND	ND	ND	0.23	ND	ND	ND
Location 3/S1GW3	1.40	ND	ND	ND	0.39	ND	ND	ND
Location 4/S1GW4	1.10	ND	ND	ND	0.32	ND	ND	ND
Location 5/S1GW5	3.10	ND	ND	ND	0.32	ND	ND	ND
Location 6/S1GW6	1.10	0.001	ND	ND	0.22	ND	0.001	ND
SOLOUS 2								
Location 1 /S2GWI	3.40	ND	ND	ND	0.32	ND	ND	ND
Location 2/S2GW2	6.20	0.002	ND	ND	0.32	ND	0.002	ND
Location 3/S2GW3	10.3	0.001	ND	ND	0.20	ND	0.001	ND
Location 4/S2GW4	5.60	ND	0.01	ND	0.38	ND	ND	0.01
Location 5/S2GW5	1.10	ND	ND	ND	0.32	ND	ND	ND
Location 6/S2GW6	3.0	ND	ND	ND	0.18	ND	ND	ND
Location 7/S2GW7	1.34	ND	ND	ND	0.35	ND	ND	ND
CONTROL SITES								
Location 1/SGC1	1.32	ND	ND	ND	0.17	ND	ND	ND
Location 2/SCC2	6.6	0.002	ND	ND	0.19	ND	0.002	ND

Source: November, 2010; March 2011

*ND represents not detected

APPENDIX 7

LPI Computation for the Leachate Generated at Abule-Egba Dumpsite for the Wet Season

Leachate pollutant Variable	Pollutant Concentration	Variable Weight (w_i)	Pollutant Sub-Index Value (p_i)	Aggregation (w_ip_i)
pH	7.61	0.055	5	0.275
TDS	2490	0.05	55	2.75
Cl^-	94	0.049	5	0.245
Zn^{2+}	4.4	0.056	5	0.28
Fe^{2+}	4.9	0.045	5	0.225
Pb^{2+}	0.02	0.063	5	0.315
Ni^{2+}	0.01	0.052	5	0.26
		Σw_i = 0.37		Σw_ip_i = 4.35 LPI = $\Sigma w_ip_i/\Sigma w_i$ = 4.35/0.37 =11.76

APPENDIX 8

LPI Computation for the Leachate Generated at Abule-Egba Dumpsite for the Dry Season

Leachate pollutant Variable	Pollutant Concentration	Variable Weight (w_i)	Pollutant Sub-Index Value (p_i)	Aggregation (w_ip_i)
pH	5.16	0.055	12	0.66
TDS	149	0.05	5	0.25
Cl^-	404	0.049	5	0.245
Zn^{2+}	0.24	0.056	5	0.28
Fe^{2+}	0.11	0.045	5	0.225
Pb^{2+}	ND	-	-	-
Ni^{2+}	0.102	0.052	5	0.26
		Σw_i = 0.307		Σw_ip_i = 1.92 LPI = $\Sigma w_ip_i/\Sigma w_i$ = 1.92/0.307 =6.25

APPENDIX 9

LPI computation for the Leachate Generated at Solous 1 Dumpsite for the Wet Season

Leachate pollutant Variable	Pollutant Concentration	Variable Weight (w_i)	Pollutant Sub-Index Value (p_i)	Aggregation (w_ip_i)
pH	7.36	0.055	5	0.275
TDS	219	0.05	5	0.25
Cl^-	50	0.049	5	0.245
Zn^{2+}	3.2	0.056	5	0.28
Fe^{2+}	4.4	0.045	5	0.225
Pb^{2+}	0.01	0.063	5	0.315
Ni^{2+}	0.002	0.052	5	0.26
		$\Sigma w_i = 0.37$		$\Sigma w_ip_i = 1.85$ $LPI = \Sigma w_ip_i / \Sigma w_i$ $= 1.85/0.37$ $=5.00$

APPENDIX 10

LPI Computation for the Leachate Generated at Solous 2 Dumpsite for the Wet Season

Leachate pollutant Variable	Pollutant Concentration	Variable Weight (w_i)	Pollutant Sub-Index Value (p_i)	Aggregation (w_ip_i)
pH	6.73	0.055	5	0.275
TDS	5011	0.05	10	0.5
Cl^-	167	0.049	5	0.245
Zn^{2+}	7.6	0.056	5	0.28
Fe^{2+}	5.2	0.045	5	0.225
Pb^{2+}	0.03	0.063	5	0.315
Ni^{2+}	0.04	0.052	5	0.26
		$\Sigma w_i =$ 0.37		$\Sigma w_ip_i = 2.10$ $LPI = \Sigma w_ip_i / \Sigma w_i$ $= 2.10/0.37$ $=5.68$

APPENDIX 11

LPI Computation for the Leachate Generated at Solous 2 Dumpsite for the Dry Season

Leachate pollutant Variable	Pollutant Concentration	Variable Weight (w_i)	Pollutant Sub-Index Value (p_i)	Aggregation ($w_i p_i$)
pH	5.71	0.055	5	0.275
TDS	470	0.05	5	0.25
Cl^-	1576	0.049	10	0.49
Zn^{2+}	0.34	0.056	5	0.28
Fe^{2+}	0.12	0.045	5	0.225
Pb^{2+}	ND	-	-	-
Ni^{2+}	0.098	0.052	5	0.26
		$\Sigma w_i = 0.307$		$\Sigma w_i p_i = 1.78$ $LPI = \Sigma w_i p_i / \Sigma w_i$ $= 1.78/0.37$ $=4.81$ ≈ 5.00