

INVESTIGATION ON LEACHATE TRANSPORT ON GROUNDWATER CONTAMINATION USING VERTICAL ELECTRICAL SOUNDING (VES)

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(Received 13th March 2025; revised 05th June 2025; accepted 13th June 2025)

Abstract. The rapid increase in population has heightened concerns regarding waste disposal, with open dumpsites serving as the primary method of waste management in many areas. This study investigates the geo-electric characteristics and groundwater quality of boreholes situated near an open dumpsite in Ogale Community, Eleme Local Government, Rivers State, Nigeria. Vertical Electrical Sounding (VES) using the Schlumberger configuration was employed to examine the effects of leachate on groundwater contamination, subsurface aquifer protective capacity, and lithological corrosivity. Borehole water samples collected from varying depths (50m, 100m, 150m) and a control location (2000m) revealed that pH, temperature, and dissolved oxygen (DO) exceeded the permissible limits set by the World Health Organization (WHO) and the Nigerian Standard for Drinking Water Quality (NSDWQ). Results indicated high chloride concentrations and microbial contamination, particularly in boreholes closest to the dumpsite. The regression model (R^2 values of 0.9743, 0.9962, and 0.9981) demonstrated significant metal corrosivity at varying depths. The Water Quality Index (WQI) values (67.7, 67.6, and 68.1) classified the groundwater as poor and unsuitable for drinking. VES analysis confirmed weak to moderate aquifer protective capacity, correlating with increased acidity and microbial presence. Consequently, metal pipes prone to corrosion are unsuitable for water conveyance, and plastic pipes are recommended. The study underscores the necessity for improved waste management practices, including landfill engineering, solid waste recycling, and enhanced environmental sanitation to mitigate groundwater contamination risks.

Keywords: *investigation, leachate, transport, groundwater, contamination, vertical*

Introduction

Water is the basis of life; about 70% of the human body and about 60-70% of plant cells is made up of water. There is no life without water, and as a result it is one of the major determinants of human settlement, existence and activities on the earth. In addition, water is the most abundant environmental resource on earth. It covers 71% of the earth surface. However, potable water is scarce with about 97.4% of earth's water being salt water (96.5% in oceans and 0.9 in other saline water) and only 2.6% freshwater. About 68.7% of the earth's fresh water is tied up in polar ice caps and glaciers, and a further 30.1% is underground as groundwater, most of which are not available for use, while the remaining 1.2% is in surface and other freshwater including atmospheric moisture, 0.03% and soil moisture 0.05%. Atmospheric water as precipitation falls and forms a part of these surface and ground water sources (Edger and Smith, 2006). Groundwater plays a vital role for urban and agricultural water supply. It accounts for about 0.5% of the total hydrosphere, and is approximately 6.73 x

10km³ in volume. It constitutes a major portion of the earth's water circulatory system known as hydrological cycle and occurs in permeable geologic formation known as aquifers. Aquifers are formations having structure that can store and transmit water at rates fast enough to supply reasonable amounts to wells. Groundwater is less prone to contamination compared to surface water because of the overlaying materials' filtration and adsorption capacity (Ayoade, 2003). When groundwater gets contaminated, diseases may spring up rapidly and spread beyond human expectation because of its flow mechanism and dumpsite leachate is one of the major sources of groundwater contamination (Ayoade, 2003).

Dumpsites have served many years as a disposal site for all types of waste; municipal solid waste, industrial sewage and hazardous waste. Physical, chemical and biological processes interact simultaneously to bring about the overall decomposition of the wastes. One of the by-products of this mechanism is chemically laden leachate. Leachate is produced by the action: of rainwater aiding bacteria in the process of decomposition. Leachate is typically composed of dissolved organic matter, inorganic macro components (such as chlorides, iron, aluminium, zinc and ammonia), heavy metals and xenobiotic organic compounds such as halogenated organics (Aziz et al., 2010). Other chemicals including pesticides and solvents may also be present. Leachates are potentially hazardous waste from dumpsites, if not dealt with properly they can cause pollution to groundwater, health problems to fauna and flora and affect the environment. It is therefore important that leachates are treated and contained to prevent these occurrences (Aziz et al., 2010). Engineered landfills are designed to minimize air, water and soil pollution as well as risks to man and animals. A standardized landfill system involves carefully selected location, placing waste in lined pit or a mound (Sanitary landfills) with appropriate means of leachate and landfill gas collection and control. However, the practice of landfill system as a method of waste disposal in many developing Countries is usually far from standard recommendations as most of these landfills are mere 'holes in the ground' which do not qualify as sanitary means of solid waste disposal (Oyeku and Eludoyin, 2010).

The objectives of this paper are to determine the water quality index of groundwater within the vicinity of an active dumpsite, determine the aquifer protective capacity, lithology and the groundwater flow around active dumpsite using Vertical Electric Sounding (VES) survey, examine the physiochemical and microbial characteristics of groundwater with varying distance from dumpsite and compare the water quality index of the groundwater with relevant standards. Dumpsite leachates are normally composed of organic and inorganic substances that permeates into groundwater systems leading to change in physical and chemical properties of groundwater quality. The greatest contamination threat to groundwater comes from land filled leachates of industrial waste originated with toxic substances (Longe and Omole, 2008). However, it has been widely reported that leachates from landfills for non-hazardous waste could contain biochemical oxygen demand (BOD), chemical oxygen demand (COD), ammonia, chloride, sodium, potassium, hardness, boron, complex organic compound, chlorinated hydrocarbons and metals at concentrations which pose a threat to groundwater quality. Landfill leachates composition and volume vary with time of production and age of landfill. Taylor and Allen (2006), stated that the condition within a landfill often varies over time, from aerobic to anaerobic thus allowing different chemical reactions to take place, and producing different chemical products. Longe and Enekwechi (2007), reported that the volume of leachate depends principally on the area of the landfill, the

meteorological and hydrogeological factors and effectiveness of capping. The volume of leachate generated is therefore expected to be very high in humid regions with high rainfall and high runoff resulting in high tendency of contamination of groundwater with shallow water table in such areas. Consequently, the geology and hydrogeology of any potential landfill site has major bearing on the level of natural protection for groundwater contamination by landfill leachate (Justin and Zupančič, 2009).

Water quality index

There are several methods for evaluating water quality. However, these methods cannot clearly express the water pollutant categories. Yeh et al. (2008), attempted to develop a cost-effective programme for monitoring the quality of groundwater by sampling existing wells in order to obtain useful information. Water Quality Index method (WQI) provides the mechanism for presenting a cumulatively derived numerical expression defining a certain level of water quality. One of the major advantages of WQI is that, it incorporates multiple water quality parameters into a 'mathematical equation that rates the health of water quality with a number. Water Quality Index (WQI) method uses a rating system to determine the influence of individual quality parameters on the overall quality of water. A general WQI approach is based on the under listed factors (Fernández et al., 2004): parameter selection, determination of a quality function curve, and Sub-indices aggregation with mathematical expression. Water Quality Index was developed by Canadian Council of the Ministers of the Environment (CCME) in (2001) using ten (10) of the most commonly measured water quality variables such as dissolved oxygen, pH, coliforms, specific conductance, alkalinity and chloride. Horton's method of water quality assessment is now widely used across Africa, Asia and Europe. Since then several Water quality indices have been formulated by different international organizations. Studies have shown Water Quality Index to be one of the most effective tools to communicate information on the Overall quality status of water to the users and policy makers within each community (Fernández et al., 2004). There are four most common methods for calculating the WQI namely: (1) NSF-WQI (National Sanitation Foundation-Water Quality Index), (2) OWQI (Oregon Water Quality Index), (3) Weighted Arithmetic Water Quality Index Method, (4) The Canadian Council of Ministers of the Environment Index (CCME-WQI).

Arithmetic water quality index method

This method provides information regarding the quality evaluation of a body of water. This method uses the most commonly measured water quality parameters (pH, BOD, COD, DO, P-PO³⁻, N-total, N-NO³⁻, N-NO²⁻, N-NH⁴⁺, SO²⁻⁴, Cl⁻, Cr^{-total}, Pb²⁺, Cd²⁺, Ni²⁺, Fe-total, Mn-total, Zn²⁺As²⁺) and it is calculated according to the following Eq. (1) (Chaterjee and Raziuddin, 2002):

$$WQI = \frac{\sum w_i q_i}{\sum q_i} \quad \text{Eq. (1)}$$

Where; WQI has a value between 0 and 100 which indicates the quality of the water; q_j represents a relative value of the water quality, specific to each parameter; i represents the number of parameters taken into consideration; W_i is a factor which measures the importance of a parameter in the calculation of the WQI index referred to

as relative weight. q_j is calculated by applying the Eq. (2) (Chaterjee and Raziuddin, 2002):

$$q_i = 100 \times \frac{v_i - v_0}{s_i - v_0} \quad \text{Eq. (2)}$$

Where; V_i represents the value experimentally determined for the i th analysed parameter; V_0 represents the ideal value of that parameter; S_i represents the standard, legally accepted, value for the water category in which the analysed water sample was included. W_1 factor is calculated by using the Eq. (3) (Chaterjee and Raziuddin, 2002):

$$W_i = \frac{K}{S_i} \quad \text{Eq. (3)}$$

Where; K is a constant which can result from applying the Eq. (4) (Chaterjee and Raziuddin, 2002):

$$K = \frac{1}{\sum(\frac{1}{S_i})} \quad \text{Eq. (4)}$$

Based on the value obtained for the Weighted Arithmetic WQI method, the water quality status may be determined, as illustrated in *Table 1*.

Table 1. Water quality rating as per weight arithmetic water quality index method.

WQI value	Water quality
0-25	Excellent
26-50	Good
51-75	Poor
76-100	Very poor
>100	Unsuitable for drinking

Source: Chaterjee and Raziuddin (2002).

Materials and Methods

Description of study area

The area of study is located in Ogale community in Eleme local Government Area, Port Harcourt, Rivers State (*Figure 1*). The study area is located on sedimentary basin of the Southern Part of Nigeria with topographic configuration evolving from the sedimentary patterns of the last 75,000 years. It consists of accumulations of cretaceous and tertiary sediments that are influenced by gravitational instability and tectonic forces. The sands are fine, coarse grained, unconsolidated and granular in texture. The depth to the usable aquifer in the area is approximately 30 m to 45 m, which is penetrated by most burials in the vicinity of the dumpsite. The vegetation found is thick mangrove forest, raffia palms, and light rainforest, due to drainage pattern. It also drains both fresh and salt water. The dumpsite is compacted from time to time to reduce the volume of waste in the dumpsite. This is done to prevent the waste from spilling over to the road side.



Figure 1. Map of the study area indicating the location.

Soil samples were collected by drilling holes using a 5m hand auger and consequently collecting samples up to 3-m depth. Soil samples were collected at the dumpsite and at 50m, 100m and 150m from the dumpsite. Groundwater samples were collected from Boreholes located within around the dumpsites. Leachate sample was collected at the dumpsite. Sub-surface water sample was collected at 50m away from the dumpsite. This collection method was done in line with the APHA standard methods in 2005 for the examination of water and waste water. In-situ analysis of water and soil parameters were carried out using all insitu standardized reagents onsite analysis of lithologic variations of drilled hole. Water quality from physical view and dinates were also recorded for data analysis and compares. For offsite analysis, laboratory analysis as stated earlier were carried out using all standard procedures for the analysis, in a bid to compare data with in-situ analysis and to also generate a comprehensive data set for adequate data analysis. Study examined fifteen physical, biological and chemical parameters (including heavy metals) Temperature ($^{\circ}\text{C}$), pH, Nitrate (NO^{-3}), Chloride (Cl), Total Hardness (TH), Total Suspended Solid (TSS), Dissolved Oxygen (DO), Electrical conductivity (EC), Coliform Count, Total Heterotrophic Bacterial Count, Total Heterotrophic Fungal Count, Iron (Fe), Chromium (Cri), Cadmium (Cd), and Lead (Pb). The methods were adopted for analyses of all parameters in potable and waste water was in accordance with APHA standard procedures of 2005. All samples were analysed for the selected physical, chemical, biological and heavy metals parameters.

The pH was measured with a Horiba Multi-Parameter Water Checker. The probe was first calibrated, then dipped into each sample and the displayed reading was allowed to stabilize before recording. The temperature of the sample was measured (in degree Celsius) using a Horiba Multi-Parameter Checker. The water checker was submerged two-thirds below the surface of the sample measurements were taken. The electrical conductivity of samples was determined in situ by APHA 2510 B electrometric method using a calibrated Horiba Multi-Parameter Water Checker. The water checker measured the current that passed through the solution between a pair of electrodes. This method measured the induction current generated by two coils immersed in the solution. Dissolved Oxygen of the water samples was determined electrometrically using Horiba Multi-parameter Water Checker. A sample of solution (50ml) was placed in a 500ml

DO bottle and filled to the mark with previously prepared dilution water. A blank solution of the dilution water similarly prepared and placed in DO bottles. A control solution without dilution water was also prepared and placed in a DO bottle. The bottles were stopped, sealed and incubated for five days at room temperature. The APHA 2540D test method was used for this determination. Deviation from the method was whatman filter paper (0.45um) was used instead of glass fiber filter. Glass fibre filter disk was prepared, sample sizes of the filter were selected. The concentration of the TSS content was calculated as follows (Eq. (5):

$$TSS(mg/l) = \frac{(A-B)100}{\text{sample volume (ml)}} \quad \text{Eq. (5)}$$

Where; A=weight of filter paper+residue (mg); B=weight of filter paper+(mg). APHA 2340 C method was employed in the determination of water hardness in domestic and industrial waste waters. A small amount of Eriochrome Black T dye is added to an aqueous Solution; which solution is then buffered to a pH of about 10.0±0.1. The presence of calcium and magnesium ions in the solution makes it to become wine-red in colour. The solution is then titrated With EDTA to complex the calcium and magnesium ions. As more and more calcium and magnesium ions are complexed from solution, the wine-red colour progressively turns to blue, king the end-point the titration. APHA 45 00-Cl-B test method was used in the titrimetric determination of the concentration of chloride. In a neutral or slightly alkaline solution, with potassium chromate as indicator for the end point of silver nitrate of chloride, silver chloride is precipitated quantitatively before red silver chromate is formed. APHA 4500-NO₃ E was used for determination of Nitrate. Here, nitrate is reduced quantitatively to nitrite in the presence of cadmium treated with copper sulphate and packed in a class column. The NO₃⁻ produced thus is determined by diazotizing with sulphamylamide and coupling with N1²-naphthyl)-ethylenediamine dihydrochloride to form a highly coloured azo dye that is measured calorimetrically. Concentrations of nitrate are obtained directly from standard curve plotted using standard concentration.

Results and Discussion

The investigation on the vertical electric sounding (VES) locations for underground water contamination was considered and the results obtained are presented in Tables and Figures as demonstrated in the paper. *Table 2* shows the lithology of soil with different resistivity, thickness and depth employed by VES locations. VES 1 and 2 were delineated with 5 geo-electric layers while VES 3 was delineated with 4 geo-electric. The 5 geoelectric layered VES points 1 and 2 gave HAK and AAK curve types respectively the 4 geoelectric layered VES point 3 gave an AK curve type.

Table 2. Geoelectric layers with respect to lithology.

VES	Layers	Resistivity (p.Ωm)	Thickness (h,m)	Depth (d, m)	Lithology
1	1	145.2	1.259	1.259	Clay/surface soil
	2	16.56	0.5889	1.848	Clay
	3	360.74	3.2919	5.067	Sand
	4	946.7	21.47	26.53	Coarse sand
	5	444.3	-	-	-
2	1	11.04	1.800	1.800	Clay/surface soil
	2	148.5	0.15658	1.973	Clay
	3	183.8	14.14	16.11	Fine sand
	4	526.6	13.5	29.61	Coarse sand

	5	385.207	-	-	-
3	1	11.08	1.826	1.826	Clay/surface soil
	2	188.6	16.52	8.36	Sand
	3	887	11.26	29.61	Sand
	4	277.2	-	-	-

Ground water potential around VES 1

HAK curves as shown in Figure 1 are generally characterized by a top layer with a higher resistivity than the second layer which is followed by a progressive increase from the third to the fourth layer and then decreases at the fifth layer while the depth of the layers increases progressively from the first to the fourth layer with coefficient of determination $R^2=0.9742$ corresponding to 97.42%. The resistivity and depth sequences are given as $P_1>P_2<P_3<P_4>P_5$ and $d_1>d_2>d_3>d_4$ respectively. Ground water potential increases with the presence of fractured layers as ground water is held between fractured layers (Prabhu and Sivakumar, 2018). The very low resistivity value ($p_2=16.56\Omega m$) of the second layer is an indication of the presence of highly weathered rocks in this layer supported by the lithology and thickness of the layer, which was a thin clay as shown in *Table 2 (Figure 2)*. The sudden jump in resistivity from the third to the fourth layer with 150 percent increase in resistivity between the 3rd and 4th layer explains the presence of fractured rocks in the 4th layer. Hence, around the VES 1 area, the 4th layer was identified as the aquifer layer, the depth from the top soil to the aquifer layer is 8.174m while the aquifer resistivity and thickness were $946.7\Omega m$ and 21.37m, respectively. The decrease in resistivity at the 5th layer ($p_5=444.3\Omega m$) is attributed to the emergence of interconnected fractures capable of yielding moderate groundwater since the resistivity is up to $200 \Omega m$ (Prabhu and Sivakumar, 2018).

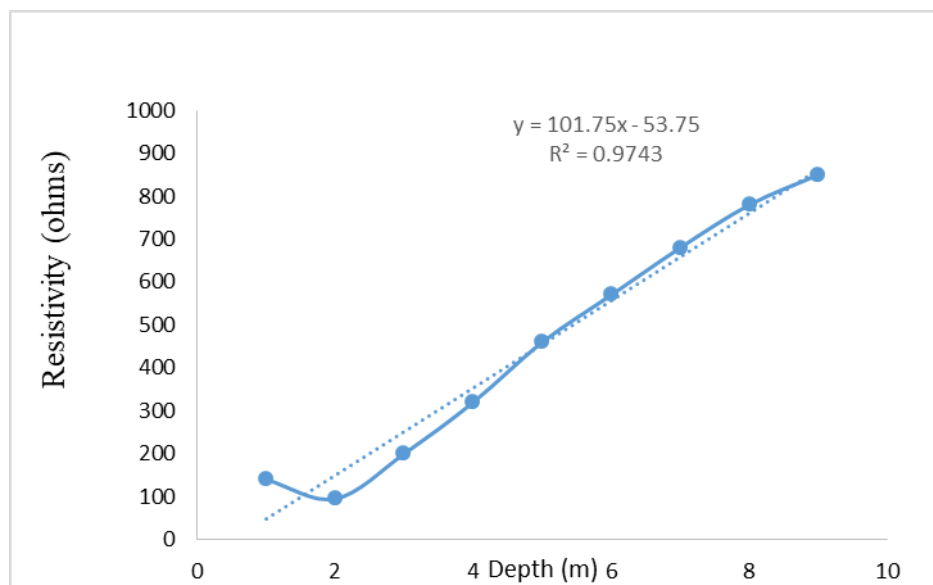


Figure 2. Geo-electric layered VES Point 1.

Ground water potential around VES 2

Figure 3 shows the interpretation of the curve characterized by the resistance and depth sequence of $p_1<P_2<P_3<P_4>P_5$ and $d_1>d_2>d_3>d_4$ respectively. This explains a progressive increase in resistivity with depth among the first four layers followed by a decrease at the fifth layer. The very low resistivity of the top soil ($P_1=11.0411m$) is an

indication of the presence of highly weathered d rock particles with clay contents which is supported by the lithology of the top soil as displayed in *Table 2*. With resistivity less than 200 Ωm for the 2nd and 3'd layers, much less weathered particles would have been encountered. The 4th layer with a sudden jump in resistivity by 186.5 percent from the 3id layer resistivity authenticates the 4th layer as the fractured layer with coarse sand with coefficient of determination $R^2=0.9981$ as shown in *Figure 3*. This implies that the 4th layer is the aquifer layer with the depth from the top soil to the aquifer layer been 19.883m while the aquifer resistivity and thickness are 526.6 and 13.5m respectively. Again the 5th layer with a lower resistivity of 3 85.207 Ωm is attributable to the emergence of interconnected fractures capable of yielding moderate groundwater since the resistivity is up to 200m (Prabhu and Sivakumar, 2018).

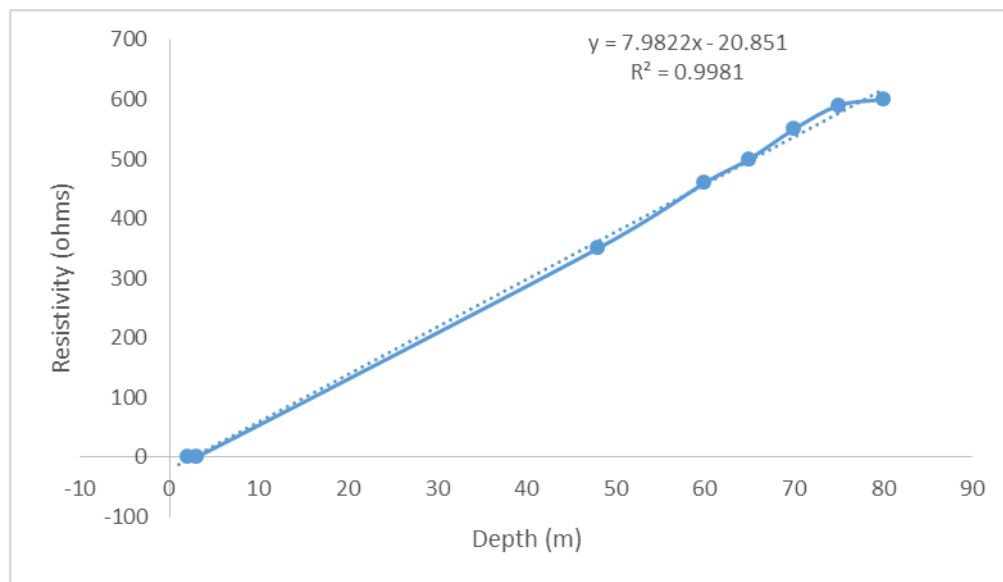


Figure 3. Geo-electric layered VES Point 2.

Ground water potential around VES 3

Figure 4 shows the interpretation of the curve derived from a 4-layer geophysical investigation characterized by a resistivity and depth sequences of $p_1 < p_2 < p_3 < p_4$ and $d_3 > d_2 > d_1$ respectively. Similar to the VES 2, the resistivity sequence explains a progressive increase in resistivity with depth from the first to the third layer which was followed by a decrease at the fourth layer. The very low resistivity of the top soil ($P_i=11.08 \Omega\text{m}$) is an indication of the presence of highly weathered rock particles with clay contents which is supported by the lithology of the top soil as displayed in *Table 2*. With resistivity less than 200 Ωm for the 2nd layer and a sudden jump in resistivity by 407.4 percent from the 2 to 3 layer authenticates the 3rd layer as the fractured layer with coarse sand with coefficient of determination $R^2=0.9962$. This implies that the 3rd layer is the aquifer layer with the depth from the top soil to the aquifer layer been 10.186m while the aquifer resistivity and thickness are 887m and 11.26m respectively. The sequence ending with a layer of lower resistivity of 27.2 Ωm is also attributable to the emergence of interconnected fractures capable of yielding moderate groundwater since the resistivity was up to 200 Ωm (Prabhu and Sivakumar, 2018).

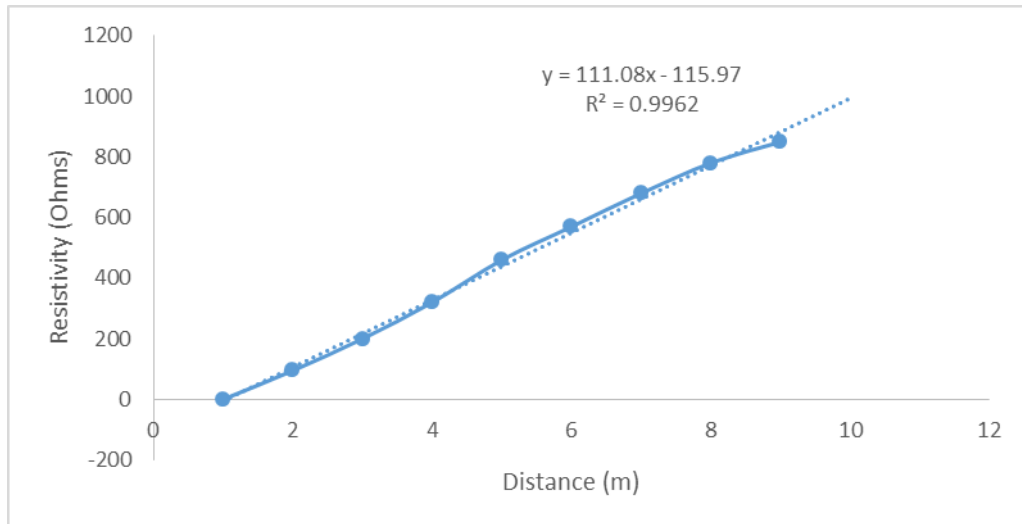


Figure 4. Geo-electric layered VES Point 3.

Electrical anisotropic coefficient (A) and reflection coefficient (K)

The transverse resistivity (Pt), longitudinal resistivity and anisotropic coefficient (A) of subsurface layers in this study are as displayed in *Table 3*. It was observed that $P_t > P_l$ for all VES points. A was within the expected range of most geological conditions. The high λ values ($A > 1$) signify diverse degree of fracturing in the transverse and longitudinal directions of NS-SW for 2 and 3 and N-S direction for VES 1. The high A values also implies that the extensions of fractures in these directions results in high water holding capacity of the entire study area. While the variation between the transverse and longitudinal resistivity explains the non-uniformity medium. It was observed that though the anisotropic coefficient of VES points 2 and 3 the aquifer thickness (H) and the longitudinal conductance (S) for VES points 2 and 3 were similar *Table 3*. The dissimilar values of for VES points 2 and 3 is attributable to the vary number of layers at these VES points. The reflection coefficient (k) values of the 3 VES points were all negative. This means that the last layer of each VES point was dominated by fractured rocks resulting in the high fracture porosity (ϕ) values ranging between 0.692 and 0.901 as shown in *Table 3*. The high porosity suggests that the fractured rocks are likely to possess water with better water holding capacity. It was also observed that VES 3 with the least k value showed the highest value of λ while VES 2 with the highest value of k showed the least value of λ (*Table 3*). This is in agreement with the fact that the degree of fracture is inversely related to the anisotropic coefficient. The fracture porosity in this k did not correlate with the anisotropic values (λ).

Table 3. Anisotropic coefficient of VES points.

Category	H	T	S	Pt	Pi	λ	k	ϕ
VES 1	34.704	27157.95	0.162	782.54	213.78	1.91	-0.36	0.692
VES 2	29.60	9751.16	0.27	329.47	110.99	1.72	-0.16	0.901
VES 3	29.61	13123.52	0.27	443.27	111.68	1.99	-0.52	0.823

Effect of dumpsite leachate on selected physicochemical properties of groundwater contamination

Figure 5 to Figure 8 shows the pH results of ground water samples at various distance 50m, 100m, 150m, including the control which was away from the dumpsite labelled BH D50, BH D100, BH D150 and BH Control respectively showed that water sample were within the acidic range 4.67-4.98. The pH level at various distance were within the threshold of WHO and NSDWQ recommendation. However, there was a decreased with increased distance from the location of dumpsite. Figure 6 shows the dissolved oxygen (DO) for all water samples within 50m to 150m distance from the dumpsite were far higher than the WHO guideline of 2mg/l. DO decreased as the distance from dumpsite increased. Figure 6 shows the temperature ranges across borehole location from dumpsite were within 25.7°C to 25.9°C seen be within the WHO guideline of (25°C). Figure 8 show the level of THB along contaminated groundwater. While coliform count and THF were similar irrespective of distance, THB decreased with distance from dumpsite. The spatial variation of pH, DO and microbial counts suggests the possibility of groundwater contamination by dumpsite leachate. With a description of weak to moderate aquifer protective capacity of the study area as presented by the VES points 1-3, the groundwater around the study area is prone to dumpsite leachate contamination. However, the level of heavy metals such as lead, cadmium, chromium and iron as well as nitrate and chloride being found to be within the safe limits of drinking water by WHO, the dumpsite is suspected to be saturated with high organic waste content. This is attributable to the fact that the dumpsite receives little or no industrial waste and local scavengers pick up plastic, metal, and even electronic wastes that could have been the source of heavy metal contamination.

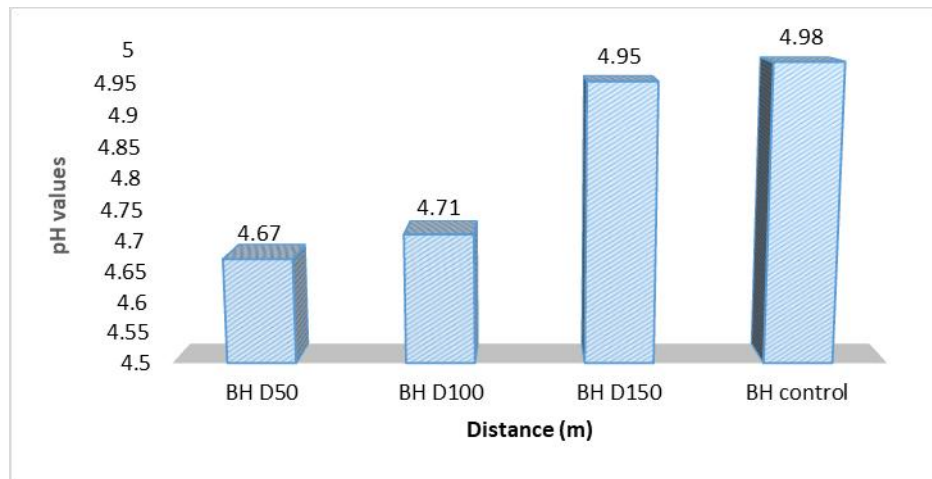


Figure 5. pH values of groundwater at different distance from dumpsite.

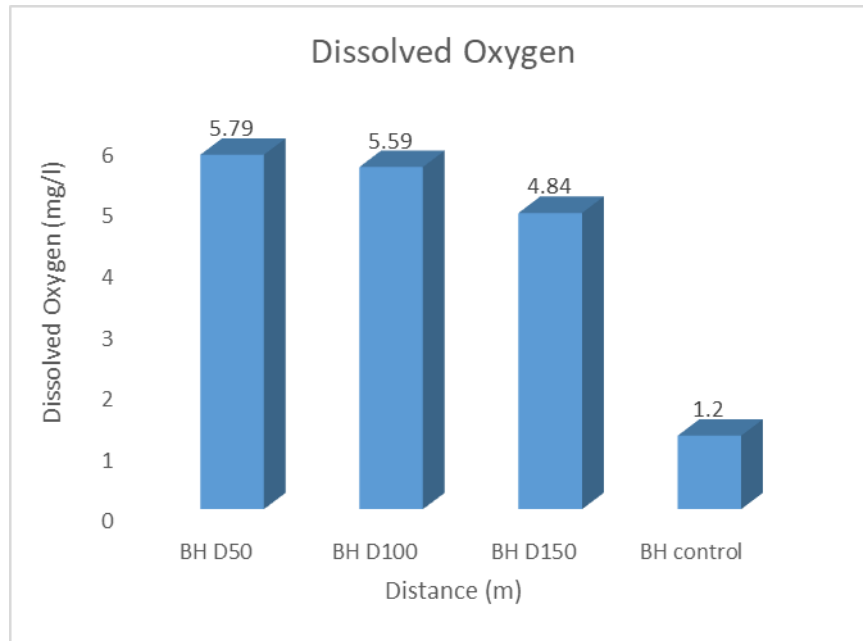


Figure 6. Dissolved Oxygen of groundwater at different distance from dumpsite.

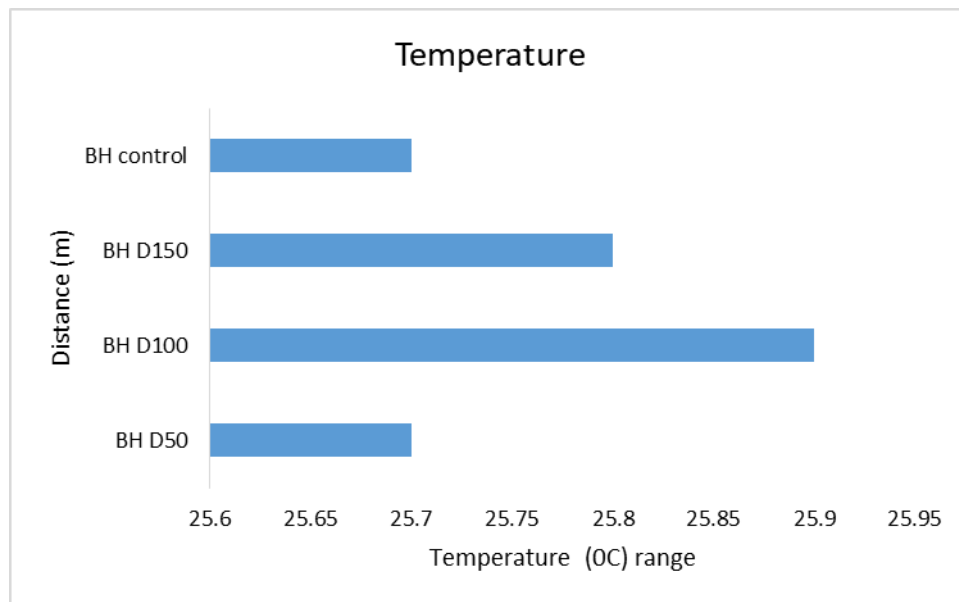


Figure 7. Temperature of groundwater at varying distance from dumpsite.

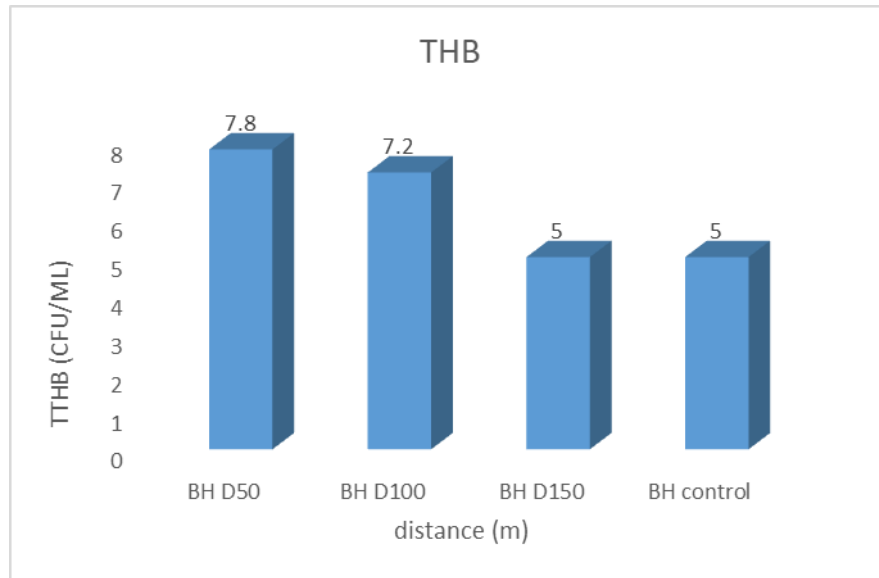


Figure 8. Total Heterotrophic Bacteria of groundwater at distance from dumpsite.

Water quality index

The water quality index from dumpsite suggests the possibility of groundwater contamination with a description of poor water quality presented in *Figure 9*. The groundwater around the study area is prone to dumpsite leachate contamination. The dumpsite is suspected to be saturated with high organic waste content. This is attributable to the fact that the dumpsite receives little or no industrial waste and local scavengers pick up plastic, metal, and even electronic wastes that could have been the source contamination. The water quality of the boreholes within the Ogale, Eleme dumpsite was evaluated using four different boreholes at significant distances from the dumpsite. This was carried out with the motive of establishing the degree of pollutant characteristics of the groundwater attributed to improper management of dumpsites. The weighted arithmetic water quality index served as statistical instrument for rating water quality of sampled boreholes. The study showed that the borehole water within and nearby Ogale, Eleme dumpsite were laden with increased acidity and pathogens. Therefore, with these indices, the study indicated that the borehole water at the dumpsite and the one nearest to it were not fit for consumption until they undergo adequate water treatment.

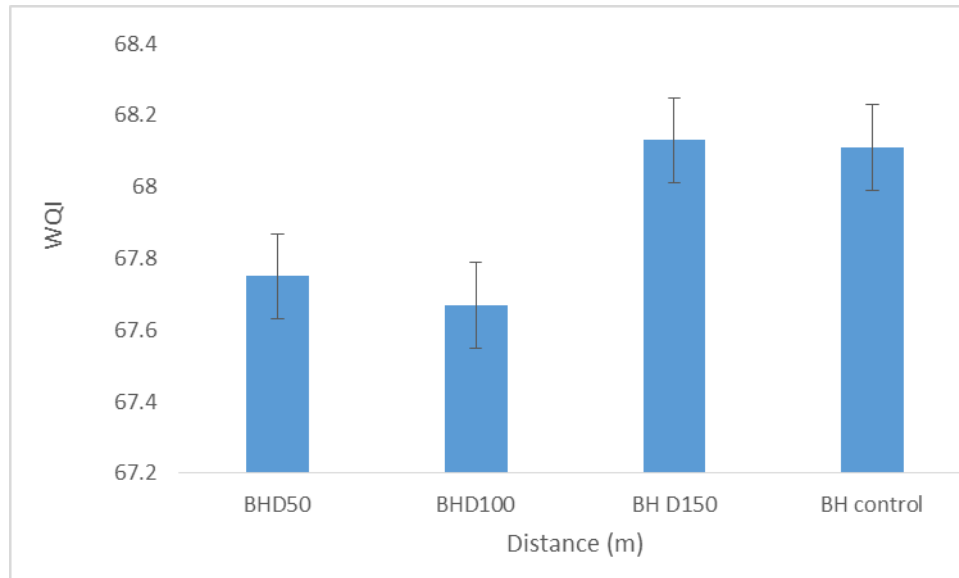


Figure 9. Water Quality Index of Boreholes within Dumpsite Location (Percent error bar).

Effect of dumpsite leachate on soil geochemistry

Since utility pipes for conveying water are buried within the topmost layers of the soil, the resistivity values of these layers were used in the evaluation of the corrosivity potential. From the VES results within a depth of 1.8m, soil resistivity range between 11.06-16.56 L2m which implies that the soil within the study area was moderately corrosive (*Table 4*). The soil pH at the top soil surface ranged from 4.31 to 4.93 while the soil pH 3m from the soil surface ranged from 4.22 to 4.98 (*Figure 10*). From the soil corrosivity classification based on soil pH, the soil pH corroborates the fact that the soil was corrosive. However, the corrosivity decreased with distance from dumpsite as soil pH increased with distance from dumpsite and depth. With respect to depth (3m below the topsoil), soil pH was lower at the dumpsite and at 50m away from dumpsite than at the surface at both depths (soil surface and 3m below the soil surface). The lithological characteristic of the top soil which was clay at the soil surface and sand at 3m depth as shown in *Table 5* explains the phenomenon. Generally, sand has a lower pH than clay, so it is expected that the soil pH at 3m depth should be lower than soil pH at the surface. However, moving further away from the dumpsite, sand pH increased above clay pH. This suggests that the infiltration of the leachate was restricted to less than 3m in the soil because of the less permeable nature of clay. Also, the corrosive nature of the soil suggests that metal pipes that are prone to corrosion should not be used within this area for water conveyance, rather plastic pipes should be used for water conveyance.

Table 4. Soil corrosivity ratings of soil samples.

Soil resistivity (Ω m)	Soil corrosivity	Sample description
<10	Very strongly corrosive (VSC)	NIL
10-60	Moderately corrosive (MC)	VES 1,2 and 3
60-180	Slightly corrosive (SC)	NIL
>180	Practically non corrosive (PNC)	NIL

Table 5. Soil sample lithology.

Sample	Longitude	Latitude	Elevation	Colour	Lithology
D1(surface)	6 ^o 98'E	4 ^o 90'N	23.4M	Black	Clay

1m				Reddish black	Clay
2m				Reddish	Clay
3m				Reddish white	Clay
D50(surface)	6 ^o 57'E	4 ^o 90'N	23.3M	Black	Clay
1m				Reddish black	Clay
2m				Reddish	Clay
3m				Reddish white	Clay
D100(surface)	6 ^o 57'E	4 ^o 54'N	23.2M	Black	Clay
1m				Reddish black	Clay
2m				Reddish	Clay
3m				Reddish white	Clay
D150(surface)	6 ^o 57'E	4 ^o 45'N	23.07M	Black	Clay
1m				Reddish black	Clay
2m				Reddish	Clay
3m				Reddish white	Clay
Control (surf)	6 ^o 99'E	4 ^o 89'N	15.10M	Black	Clay
1m				Reddish black	Clay
2m				Reddish	Clay
3m				Reddish white	Clay

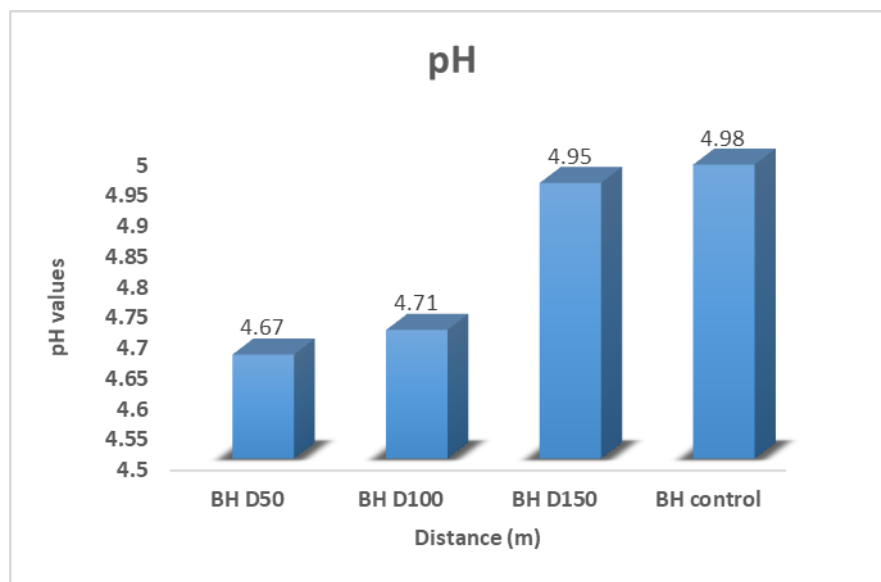


Figure 10. Soil pH at varying depth and distance from dumpsite.

Effect of dumpsite leachate on soil heavy metal content

From the field experiment of dumpsite leachate on Pb from surface to 3m depth ignited concentration of 0.009 to 0.1mg/kg as shown in *Figure 11*. These were observed to be with the permissible limit of lead in loam. These heavy metals include chromium, cadmium, lead and iron. The chromium content in soil in this work ranged between 0.11 to 0.1 7mg/kg was observed to be within the permissible limit of heavy metals in soils 100mg/kg, as shown in *Figure 12*. The chromium content at the surface at 50m away from the dumpsite was the highest but decreased and remained steady from 100m away from the dumpsite till control. This was attributed to the presence of metal scraps of iron found in the area. On the other hand, below the soil surface (at 3m depth) chromium was found to be the highest at the dumpsite but decreased at 50m away from the dumpsite till 100m away and remained steady from 150m away till control (*Figure 12*). This indicates that the soil chromium was affected by the high level of iron content in the soil. On the other hand, soil cadmium and lead were constant irrespective of distance and depth and the contents were insignificant in the soil (*Figure 13*). The at the surface increased at 50m away from the dumpsite but decreased and remained constant an 100m

away from the dumpsite till control. The increase at 50m can be attributed to the presence of metal scraps found in the area. On the other hand, below the soil surface (at 3m depth) iron was found to be the highest at 50m away from the dumpsite but decreased at 100m away from the dumpsite and remained steady till control (*Figure 14*). This indicates that the soil iron was affected by the leachate from the dumpsite as the iron of leachate at dumpsite was observed to be extremely high.

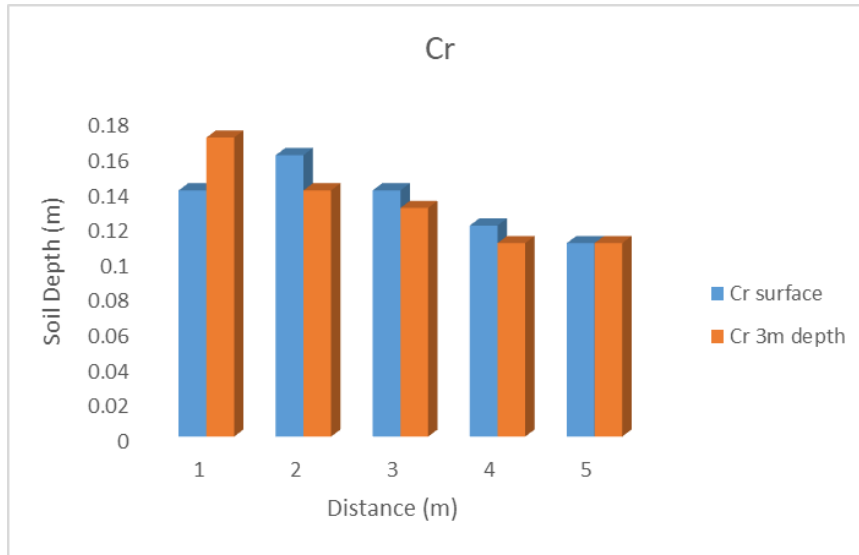


Figure 11. Soil Chromium at varying depth and distance from dumpsite.

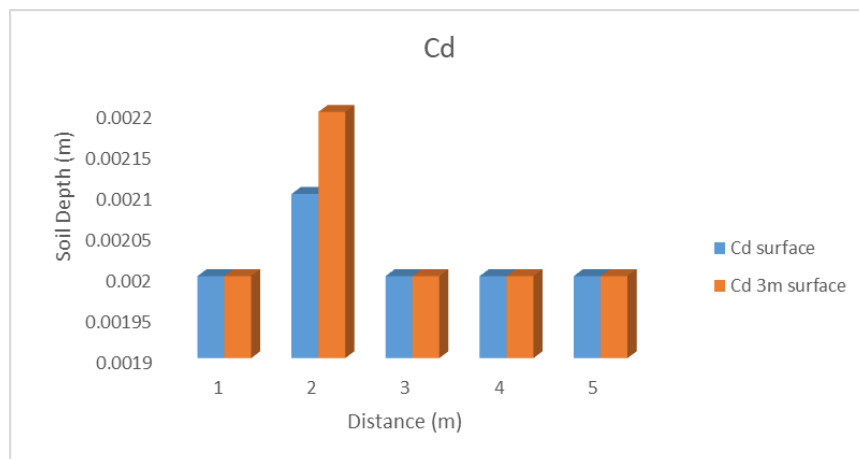


Figure 12. Soil Cadmium and Lead at varying depth and distance from dumpsite.

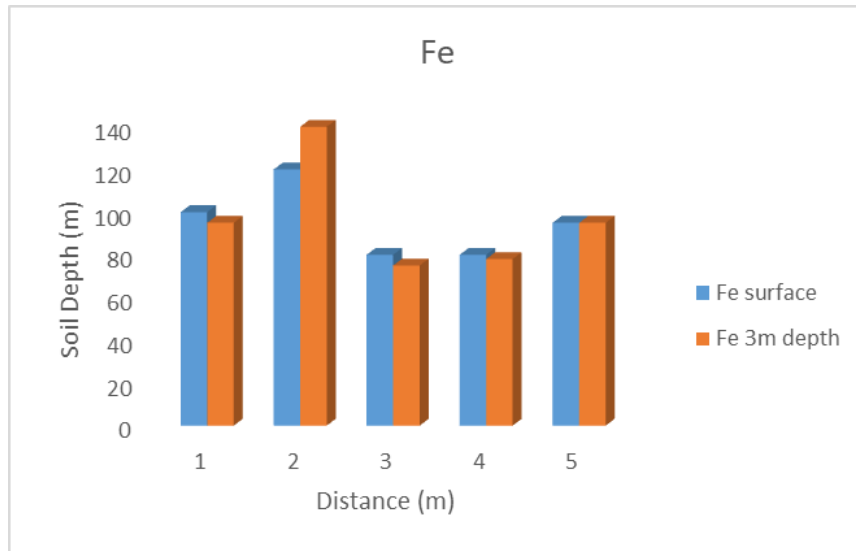


Figure 13. Soil Iron at varying depth and distance from dumpsite.

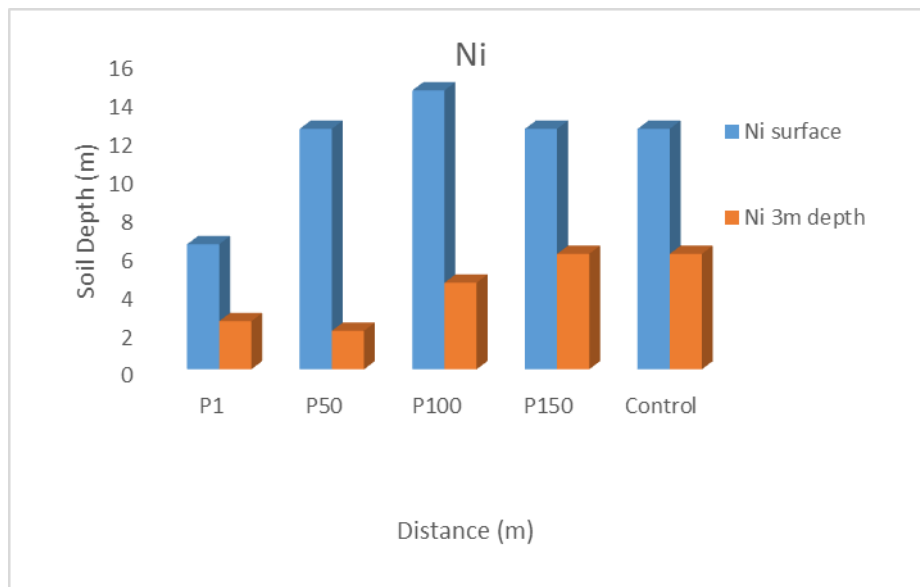


Figure 14. Soil Nitrate at varying depth and distance from dumpsite.

Effect of dumpsite leachate on soil nitrate and chloride

Nitrate and chloride are two common pollutants of ground water sources from dumpsites. It was observed that nitrate at the surface increased at 50m away from the dumpsite and increased further a 100m away from the dumpsite but had a slight decrease at 150m away from the dumpsite till control. On the other hand, below the soil surface (at 3m depth) nitrate decreased at 50m away from the dumpsite but had an increase and remained steady at 100m away till control. This indicates that the soil nitrate was affected by the acidity from the dumpsite as the pH at dumpsite was observed to be highly acidic. Also that nitrate did not percolate to the groundwater table but may have been leached out as surface runoff. Similarly, chloride at the surface decreased at 50m away from the dumpsite but increased and remained steady from 100m away from the dumpsite till control. On the other hand, below the soil surface (at 3m depth) chloride was found to be the highest at 50m away from the dumpsite but eased at 100m away

from the dumpsite and remained steady till control (*Figure 15*). This indicates that the soil chloride was affected by the leachate from the dumpsite as the chloride of leachate at dumpsite was observed to be extremely high. Also chloride percolated downwards.

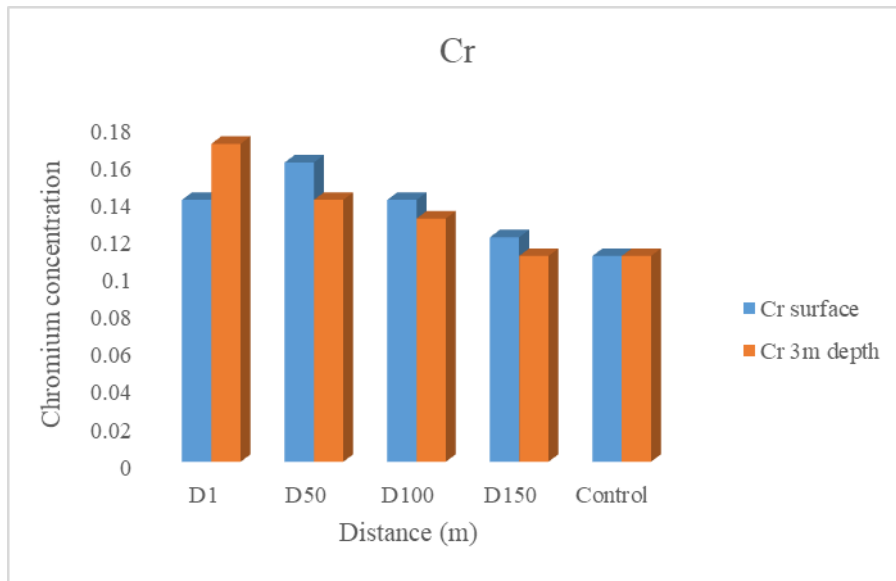


Figure 15. Soil Chloride at varying depth and distance from dumpsite.

Conclusion

The migration of leachate from open dumpsite to ground water table is a point source of ground water pollution. The weighted arithmetic water quality index method and vertical electric sounding served as a statistical instrument for rating bore hole water samples and unravelling the groundwater potential, subsurface aquifer protective capacity and lithology corrosively. The study concluded that the borehole water within the Ogale, Eleme dumpsite was laden with increased acidity and microbial activity, the aquifer protective capacity was weak and the soil was corrosive. Therefore, with these indices, the study concluded that the borehole water at the dumpsite is not fit for consumption until they undergo adequate water treatment. Towards the control of groundwater vulnerability to pollution through dumpsites there is need for: investigation of transport phenomenon of heavy metals through soil strata around the Ogale Eleme dumpsite; investigation of transport phenomenon of microbes around the dumpsite; and due to the corrosive nature of the soil, metal pipes that are prone to corrosion should not be used instead plastic pipes should be used for water conveyance.

Acknowledgement

This research is self-funded.

Conflict of interest

The authors confirm that there is no conflict of interest involve with any parties in this research study.

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