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Assessment of groundwater pollution near Aba-Eku municipal solid waste dumpsite

Olukemi Aromolaran · Obasola E. Fagade · Olawale K. Aromolaran · Emmanuel T. Faleye · Harald Faerber

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Abstract Municipal solid waste (MSW) dumpsite constitutes a major anthropogenic point source of leachate contamination to the ambient groundwater and poses a significant threat to the geo-ecosystem. This study investigated the pollution of groundwater by leachate emanating from Aba-Eku MSW dumpsite in Ibadan, Nigeria, using bacteriological, hydrochemical, and geophysical techniques. There is a diversity of bacteria in the leachate and the dominant phyla being proteobacteria (83%) and firmicutes (17%). The mean concentrations (mg/L) of Mn, Fe, Al, Cu, Mo, and Cr in the leachate samples were above the World Health Organization wastewater discharge limits. The hydrochemical parameters of the groundwater samples around the dumpsite were generally within the permissible limits, except for K and Cl^-; which invariably indicate major inputs from water-rock interaction and minor contributions from the dumpsite. Three geoelectrical layers were indicated from the vertical electrical sounding data, which are the topsoil, the lateritic clay layer, and the weathered basement. Low resistivity values of 5–33 Ωm and 3–24 Ωm were obtained within 2 m and 5.5 m depths for the topsoil and the lateritic layer, respectively; while the 2-D subsurface model reveals leachate plume beyond 5 m. Although the MSW leachate is heterogeneous, the hydrochemical data show that the aquifer around the dumpsite has not been seriously polluted with the leachate, but there is a continuous percolation of leachate into the soil subsurface, based on the geophysical findings. Discontinuing waste dumping and groundwater extraction, which would over time reduce the leachate plume, are measures to enhance the groundwater quality in the area.

Keywords Municipal solid waste · Leachates · Aba-Eku · Soil subsurface · Nigeria

O. Aromolaran (✉)
Department of Biological Sciences, Bowen University, Iwo, Osun State, Nigeria
e-mail: kemiaromolaran@gmail.com

O. E. Fagade
Department of Microbiology, University of Ibadan, Ibadan, Nigeria

O. K. Aromolaran
Department of Geological Sciences, Osun State University, Osogbo, Osun State, Nigeria

E. T. Faleye
Department of Physical and Chemical Sciences, Elizade University, Ilara-Mokin, Ondo State, Nigeria

H. Faerber
Institute of Hygiene and Public Health, University of Bonn, Bonn, Germany

Introduction

In most developing nations, unwanted solid waste materials from domestic, commercial, and industrial activities are commonly disposed off indiscriminately in open dumps (open landfills), where some of the combustible components are generally incinerated and the remnants are invariably exposed to natural biodegradation processes. These waste disposal and waste reduction practices have been reported to have significant environmental and

health implications, including air pollution due to the emission of airborne particulate matter (APM) and the generation of unpleasant odour; global warming effects associated with the production of greenhouse gases (GHGs); fire outbreaks; toxic leachate generation which pollute ambient surface and groundwater; and open dumps being fertile breeding sites for disease-causing vectors (El-Fadel et al. 1997; Christensen et al. 1998; Annepu 2012; Abdel-Salam and Abu-Zuid 2015; Oketola and Akpotu 2015).

Municipal solid wastes (MSW) in open landfills usually contain potentially toxic and hazardous substances, which are constantly subjected to various decaying processes (Ludvigsen et al. 1999; Kehew 2001). When percolating rainwater mixes with these decaying solid wastes, it extracts soluble and suspended materials from the wastes and facilitates chemical and microbial processes that result in the formation of leachates (Christensen et al. 2001; Kjeldsen et al. 2002; Slack et al. 2005; Shabiimam and Dikshit 2012; Fernandez et al. 2014). Leachates from solid waste disposal facilities are the main pollutants of groundwater resources, as they contain varying concentrations of inorganic and xenobiotic organic pollutants (Christensen et al. 2001; Kehew 2001; Kjeldsen et al. 2002; Slack et al. 2005) and some disease-causing organisms, such as *Vibrio cholerae*, *Salmonella typhi*, *Shigella dysenteriae*, *Entamoeba histolytica*, and some enteric viruses (Gerba and Smith 2005; Oguntoke et al. 2009; Gerba et al. 2011; Pandey et al. 2014).

Untreated MSW leachates from waste disposal facilities and untreated waste rock piles (WRPs) from abandoned mine sites are two common anthropogenic point sources of surface and groundwater contamination (Kehew 2001; Pedretti et al. 2017; Blackmore et al. 2018). Several models have been advanced for hydrogeological and geochemical assessment of contaminant pathways from these point sources into pristine groundwater tables and some receiving fluvial channels (Kale et al. 2010; Bailey et al. 2013; Pedretti et al. 2017; Blackmore et al. 2018). While external and internal tracers are commonly utilised to model heavy metal loading from untreated WRPs in abandoned mines (Pedretti et al. 2017; Blackmore et al. 2018); MSW leachate pathways are generally deciphered and elucidated using seismic refraction (Cardarelli and Bernabini 1997; Onu and Ibe 1998), very low–frequency electromagnetic method (Kaya et al. 2007; Onu and Ibe 1998) and electrical geophysical imaging techniques (Benson et al. 1997; Olayinka and Olayiwola 2001; Kaya et al. 2007; Pujari et al. 2007; Oladunjoye et al. 2011; Ariyo et al. 2013; Ganiyu et al. 2015; Bichet et al. 2016; Mosuro et al. 2017; Giang et al. 2018). Assessment of groundwater contamination from MSW facilities in Nigeria has been mostly the use of hydrochemical data to infer spatial extension of leachate contamination (Ikem et al. 2002; Abimbola et al. 2005; Ameloko and Ayolabi 2008), with very little focus on the direction of subsurface leachate migration and hydraulic characteristics of subsurface soil and rock materials (Olayinka and Olayiwola 2001; Ogunseiju et al. 2015). It is particularly relevant to know that adequate evaluation of groundwater contamination from MSW effluents requires the use of hydrochemical and geophysical techniques. Geoelectrical imaging techniques enable the elucidation of soil types, subsurface stratigraphy, the degree of subsurface leachate saturation, delineation of subsurface contamination plume, and ultimately, the determination of depth to aquifers (Benson 1993; Ogunsanwo and Mands 1999; Olayinka and Olayiwola 2001).

Most solid disposal facilities in Nigeria are poorly conceptualised particularly with respect to site selection, design, and maintenance (Olayinka and Olayiwola 2001), with cases of groundwater pollution and the outbreak of epidemics documented in various districts (Oguntoke et al. 2009). Aba-Eku dumpsite, located towards the southeast of Ibadan Metropolis (Fig. 1), is one of the major solid waste disposal facilities receiving domestic, agricultural, and industrial wastes from the metropolis in the past 25 years. Urban population in Ibadan has increased at an unprecedented rate over the past decades with more than 3,565,810 residents (NPC 2006) with a yearly growth rate of 2.8%. Consequently, its waste generation has grown in leaps and bounds, annually indicating more than 996,102 t (Ayuba et al. 2013; Amuda et al. 2014). Aba-Eku MSW dumpsite is sited on a fairly gently dipping landscape with no lining system in place; as such, there is a conspicuous direct ingress of leachate materials into the adjoining stream and, by extension, the shallow groundwater table. Apparently, there is a need to monitor the contamination levels of leachates and assess the hydrochemical status of the adjoining groundwater of the area. Accordingly, the present study essentially seeks to determine the compositional characteristics of Aba-Eku MSW leachate, evaluate its impacts on the groundwater of the area, and assess the extent of leachate infiltration from the waste disposal facilities into the soil subsurface using

vertical electrical sounding (VES) and 2D geoelectrical imaging techniques.

Materials and methods

Description of the study site

Aba-Eku MSW dumpsite is situated in the southeast of Ibadan metropolis, southwest Nigeria, between latitude 07° 19′ 15″ and 07° 19′ 40″ N and longitude 003° 59′ 00″ and 003° 59′ 30″ E (Fig. 1); while the entire study area lies within latitude 07° 18′ 00″ and 07° 21′ 00″ N and longitude 003° 58′ 00″ and 004° 02′ 00″ E. The elevation of the study area ranges between 140 and 160 m above the mean sea level. The solid waste facility was opened in 1994 and sited several kilometres away from residential areas, but rapid population growth and rapid urbanisation have now turned the area to a built-up area. The dumpsite covers an area of about 10 ha, with leachate migration following the local topographic setting into the nearby stream. The wastes deposited in the facility predominantly contain domestic, agricultural, industrial, and medical wastes without any prior segregation, except for few scavengers who partially pick up metals, glass, and plastic materials for reuse and recycling purposes. The Aba-Eku waste disposal facility lacks lining system, while leachate collection system via pipes into a central pond for chemical treatment and natural attenuation processes has been dysfunctional shortly after installation in 1998 (Aluko and Sridhar 2005). The foregoing has resulted in unrestricted interaction of the percolating MSW leachates with the nearby Omi Stream and the ambient groundwater of the area.

The climate of Ibadan district is that of humid tropical rainforest, typified by two alternate seasons; which include a wet season that extends from early March to mid-November and a distinct dry season that stretches from mid-November to February. Peak rainfall is usually from May to August, with average annual precipitation of about 2150 mm. Temperature is fairly constant with mean values of 32 °C and 25 °C commonly reported for day and night periods.

Geological and hydrogeological settings

The study area lies within the Precambrian Basement Complex of southwestern Nigeria, which comprises banded gneiss, augen gneiss, granite gneiss, migmatites, and quartzite as the dominant rock types (Rahaman and Lancelot 1984; Rahaman 1988). The geology of the study area is made up of banded gneiss, quartzite/quartz schist, and biotite granite as the major rock units (Fig. 2), while pegmatites, dolerites, and vein quartz constitute the minor rock types of the area. The strike of the prevalent foliation, which is consistent with the regional trend, is generally NE-SW often with gentle dips to the east. The rocks of the area have generally suffered varying degrees of alteration with few outcrops still preserved in situ.

The Aba-Eku MSW dumpsite is underlain by biotite granite that has been extensively weathered and fractured, showing varying thicknesses of regolith. The aquifer in this area is mainly derived from the weathered regoliths that are marked by thin thicknesses of the vadose zone, which invariably results in the percolation of leachate into the shallow aquifer particularly in the proximity of the dumpsite. The residents of the area are of low socio-economic status and mainly depend on shallow hand-dug wells that show profound proneness to the contamination from the leachate from the solid waste facility. The dumpsite area is mainly drained by Omi Stream (Fig. 1), which flows southeastwards and further carries the leachate downstream.

Sample collection

In order to assess the impact of the leachate from the Aba-Eku MSW dumpsite on the surrounding groundwater, the location of the available hand-dug wells was inventoried. The MSW leachate samples were collected bi-monthly between June 2012 and May 2013 in 1.5-l capacity polyethylene bottles for physicochemical analyses; while sterile glass bottles were used to collect leachate samples for bacteriological analysis. Fourteen groundwater (labelled AB1-AB14) samples were collected in the vicinity of the dumpsite (Fig. 1). The groundwater sampling sites were selected based on the direction of flow of leachate and the availability of hand-dug wells along the groundwater flow direction. A control sample was collected from a hand-dug well at about 950 m upstream away from the dumpsite, where there is no leachate input from any receiving fluvial channel. The samples were immediately taken to the Soil Chemistry Laboratory at the Department of Agronomy, Osun State University, Nigeria, for physicochemical analyses; while the cations in the leachate were

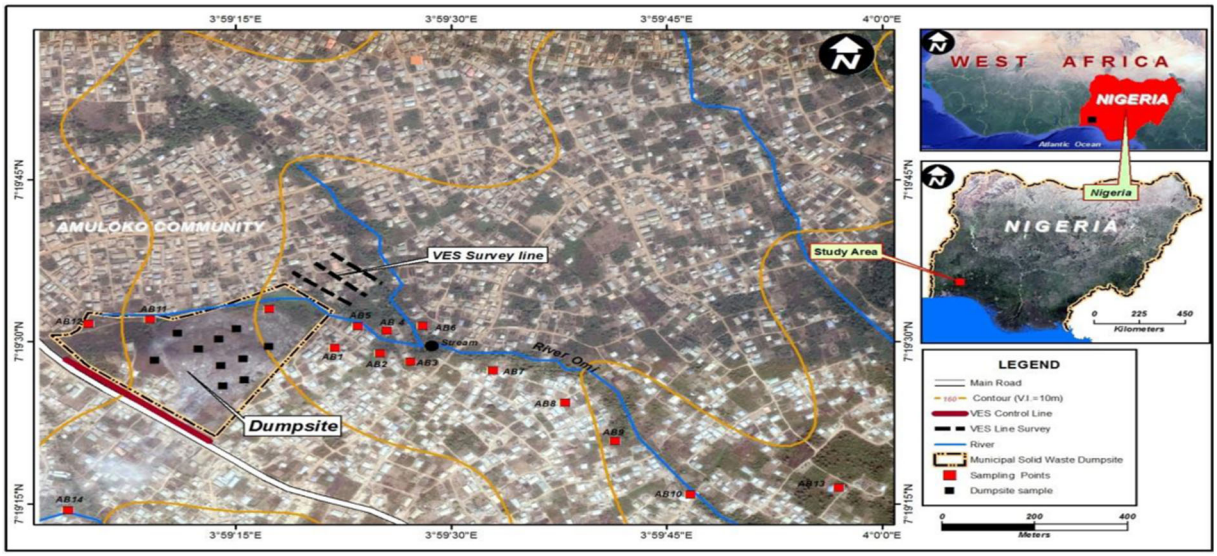


Fig. 1 Map of the study area showing Aba-Eku waste disposal facility, the sampling sites, and the geophysical survey lines

determined at the Institute of Hygiene and Public Health, University of Bonn, Germany.

Bacteriological analysis

Total coliform, faecal coliform, and total heterotrophic bacterial counts present in the leachate were determined by the standard pour plate technique. Bacteria in the leachate samples were isolated by culturable technique and identified by the analysis of their 16S rDNA genes. Pure colonies of each isolate were cultured in Nutrient Broth (NB) (Difco Lab. USA) and incubated overnight at 30 °C. The bacterial cells were centrifuged, washed, and re-suspended in Tris/EDTA (TE) buffer. The DNAs of the isolates were extracted using the Qiagen kit (QIAamp DNA Mini Kit 250, South Africa). The 16S

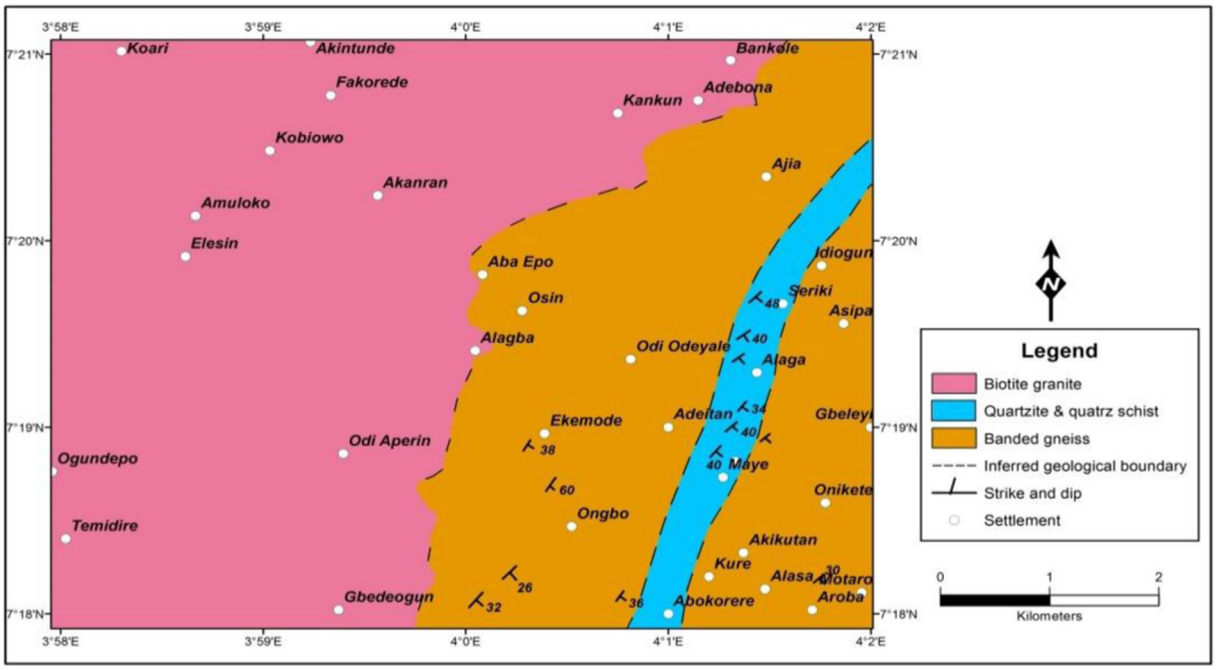


Fig. 2 Geological map of the study area

rDNA fragments were amplified using an Applied Biosystem Thermocycler (Applied Biosystem, Foster City, California, USA) with universal primers F27 and R1492. Amplified DNAs were examined by electrophoresis on 1.5% agarose gel with 2-μL aliquots of PCR products in 1× Tris-acetate-EDTA buffer. The purified PCR products were sequenced, using an ABI 3130 genetic analyser (Applied Biosystem). The DNA extraction, amplification, and sequencing were carried out at the International Institute for Tropical Agriculture (IITA), Ibadan, Nigeria. Nucleotide sequence identification was done using the basic local alignment search tool (BLAST) facility of the National Center for Biotechnology Information (NCBI).

Physicochemical analysis

The leachate, surface water, and groundwater samples were analysed for various physicochemical parameters, using the standard procedure as described by APHA (1998). The parameters considered include temperature, pH, electric conductivity (EC), turbidity, dissolved oxygen (DO), biochemical oxygen demand (BOD), chemical oxygen demand (COD), total suspended solids (TSS), total dissolved solids (TDS), total solids (TS), total alkalinity (TA), total hardness (TH), phosphate (PO_4^{3-}), ammonia (NH_4^+), total chloride (Cl^-), sulphate (SO_4^{2-}), nitrate (NO_3^-), sodium (Na^+), phosphorus (P), magnesium (Mg^{2+}), sulphur (S), potassium (K^+), calcium (Ca^{2+}), aluminium (Al), chromium (Cr), manganese (Mn), iron (Fe), cobalt (Co), zinc (Zn), copper (Cu), arsenic (As), molybdenum (Mo), cadmium (Cd), mercury (Hg), and lead (Pb). Temperature, pH, and EC were determined in situ, using the Hanna multi-parameter digital portable metre (HI 98130); while in situ TDS determination was achieved, using the T 76-TDS waterproof metre. DO and BOD were determined by Winkler's titration method and COD by reflux titrimetry, while TA and TH were determined by titrimetric methods. Sulphate, phosphate, and nitrate were determined by UV-VIS spectroscopy method, while the cations were determined using the inductively coupled plasma-mass spectroscopy (ICP-MS) instrument (Agilent 7700 Series) and the Perkin Elmer atomic absorption spectrophotometer (AAS) model 200A.

Statistical analyses

In order to identify the principal sources of the ionic loadings responsible for the hydrochemical status of the groundwater around the Aba-Eku MSW dumpsite, physicochemical parameters of the groundwater samples were subjected to correlation analysis using Pearson product-moment coefficient of linear correlation (significance levels were drawn at 0.05) and principal component analysis (PCA) (a multivariate statistical tool). The correlation analysis measures the strength of the linear relationship between two parameters in the physicochemical data set, without any suggestion that one of the parameter is dependent on the other (Rollinson 1993); while PCA separates the possible sources of ionic loading in the groundwater samples into specific components. Correlation matrices generated from physicochemical data are very informative in distinguishing key physicochemical parameters that are associated with geological and anthropogenic processes. The correlation analysis was accomplished using Statistical Package for Social Sciences (SPSS 10); while the PCA was computed using Paleontological Statistical Software Package (PAST 3.13). The effectiveness of these statistical methods, in unravelling the principal factors responsible for the ionic enrichment in groundwater, has been indicated in several studies (Olayinka and Olayiwola 2001; Abimbola et al. 2005; Olobaniyi et al. 2007; Han et al. 2014).

Determination of leachate pollution index

Leachate pollution index (LPI) is used to determine the pollution potential of leachates emanating from waste sites. Leachate contamination potential from Aba-Eku waste dumpsite was calculated, using the equation proposed by Kumar and Alappat (2005):

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

where LPI, the weighted additive LPI; W*i*, the weight for the *i*th pollutant variable; P*i*, the sub-index score of the *i*th leachate pollutant variable; and *n*, number of leachate pollutant variables used in calculating LPI.

$$\sum_{i=1}^{n} Wi = 1$$

When all the values of the pollutant variables included in LPI are not available, the LPI can be calculated

using the concentrations of the available leachate pollutants. Therefore, LPI can be calculated using the equation:

$$\mathrm{LPI} = \frac{\sum_{i=1}^{m} WiPi}{\sum_{i=1}^{m} Wi} \quad (2)$$

where m is the number of leachate pollution parameters for which data are available, but in the present study, $m < 18$ and $\sum_{i=1}^{m} Wi < 1$. Therefore, the LPI values for Aba-Eku MSW leachate was calculated using Eq. (2).

Geophysical survey

The geophysical survey was carried out as described by Hamza et al. (2014). Ten vertical electrical soundings (VES) profiles and two resistivity imaging profiles were carried out for this study. Nine VES traverse stations were established at the base of the dumpsite towards the direction of leachate flow, and one was conducted 100 m uphill away from the dumpsite, to serve as the control (Fig. 1). Garmin Global Positioning System (GPS) was used in determining the topography and groundwater flow around the study area. The Schlumberger electrode configuration was adopted with an optimum electrode spread of 65 m for the VES. Two outer electrodes were used to penetrate the electrical current into the ground, and another inner pair of electrodes was used to measure the current potentials resulting from the current flow in the ground. Resistivity was measured by using ABEM (SAS) 1000C (Earth resistivity metre). The 2D subsurface imaging (employing combined horizontal profiling (HP) and VES) was carried out using a dipole-dipole array, with an electrode spacing of 5 m and expansion factor of $n = 1$ to $n = 5$. For the dipole-dipole measurement, four electrodes were involved. At $n = 1$, a current was injected into the subsurface through a pair of an electrode (current), while another pair measures the potential difference. The potential electrodes were moved at intervals of 5 m, and the measurement procedure was repeated up to $n = 5$. Quantitative interpretation using conventional partial curve matching was adopted for the VES; while 2-D inversion of resistivity data was carried out for the combined HP and VES. Interpretation software utilised includes Resist Version 1.0, DIPROWIN and Surfer 12.

Results

Bacteriological characteristics of Aba-Eku municipal solid waste leachate samples

Faecal coliforms, total coliforms, and the total heterotrophic bacteria counts (CFU/mL) of the leachate samples are presented in Table 1.

Table 2 presents the bacteria isolated from Aba-Eku MSW leachate samples and their percentage occurrence. Forty-one bacteria species were isolated and identified from the MSW leachate; out of which, 97.56% of the isolates showed > 90% similarity identity to bacteria. The predominant bacteria species found in the leachate samples were *Pseudomonas* spp. (19.51%), *Bacillus* spp. (14.63%), and *Enterobacter aerogenes* (12.20%). There were 82.93% Gram-negative bacteria and 17.07% Gram-positive bacteria represented in the MSW leachate samples. The results of the analysis showed that the bacteria are grouped within the alpha-proteobacteria (81%), beta-proteobacteria (2%), and firmicutes (17%) (Fig. 3).

Physical and chemical analysis of Aba-Eku MSW leachates

The results of the physicochemical and heavy metal analyses of the leachate samples are presented in Table 3. The leachate samples have alkaline pH values, ranging between 8.80 and 9.70.

The mean BOD_5/COD value of the leachate sample was 0.33. High concentrations of nitrate (84.6–98.16 mg/L), phosphate (46.21–66.21 mg/L), sulphate (820.61–894.61 mg/L), ammonia (98.01–134.01 mg/L), and total chloride (1620–1920 mg/L) were observed in the leachate samples. Potassium and sodium were the most abundant cations in the leachate samples, with concentrations ranging from 791.36 to 1852.89 mg/L (mean value of 1169.71 mg/L) and 307.17 to 795.23 mg/L (mean value of 605.39 mg/L), respectively. Concentrations of Ca and Mg were 90.80–139.59 mg/L and 63.11–218.75 mg/L in the waste leachate, respectively. The concentration of sulphur was moderate, ranging between 35.37 and 78.68 mg/L (mean value of 64.00 mg/L), while phosphorus value was low (0.46–12.41 mg/L). The concentrations of Fe (23.03 mg/L), Mn (6.40 mg/L), Mo (53.90 μg/L), Cr (312.79 μg/L), Cd (9.39 μg/L), Pb (261.23 μg/L), and Cu (1077.56 μg/L) in the leachate samples were higher than those of World

Table 1 Bacterial count in Aba-Eku MSW leachate samples (CFU/mL)

	Faecal coliforms	Total coliforms	Heterotrophic counts
Leachate samples	40×10^4	87×10^4	179×10^6

CFU/mL, colony forming unit per millilitre

Health Organization (WHO) permissible limits of 5 mg/L, 0.2 mg/L, 10 μg/L, 20 μg/L, 19.39 μg/L, 200 μg/L, and 200 μg/L, respectively, for wastewaters allowed to be discharged directly into streams. Zinc (1971.25 μg/L), Hg (1.98 μg/L), Ni (103.77 μg/L), Co (41.09 μg/L), and As (15.40 μg/L) were high in the leachate samples, although still below the standard permissible limits. The concentration of Se (< 7.81 μg/L) was low and below the standard limits of 50 μg/L.

Table 4 presents the descriptions of some of the hand-dug wells around Aba-Eku MSW dumpsite and gives information on the depths and the hygienic conditions of the wells. The depths of the hand-dug wells ranged between ~ 2 and ~ 7 m. The walls of wells AB4, AB8, and AB10 were not lined with concrete to protect them from direct infiltration of subsurface impurities. All the wells are used for domestic purposes except Well AB4, which has been abandoned. Out of the entire wells sampled, only well AB3 had a steel cover; Well AB8 was covered with a plank, while some have iron covers, and some do not have any cover. The anthropogenic activities around the wells directly affect the water quality.

Physicochemical characteristics of groundwater around Aba-Eku MSW dumpsite

The results of the physicochemical parameters and heavy metal concentrations of the groundwater samples around Aba-Eku MSW dumpsite are presented in Table 5. The temperature of the water samples ranged from 25.5 to 28.8 °C; while the pH values varied between 6.0 and 6.7. The EC values of the water samples varied between 0.4 and 2.59 μS/m, while the values of the TDS ranged from 0.1 to 0.6 g/L. High EC and TDS values are particularly marked in Well AB4, which can be ascribed to its shallow depth, absence of concrete lining, and possible inputs from nearby septic tanks. On the other hand, the control groundwater samples (Well AB14) correspondingly show lower pH (6.1), TDS (0.2 g/L), and EC (0.8 μS/m) when compared with other groundwater samples (Table 5).

Table 2 Bacteria isolated from Aba-Eku MSW leachate and their percentage occurrence

S/N	Bacterial isolates	Number of isolates	Percentage similarity to its closest relative (%)	Percentage occurrence (%)
1	*Enterobacter aerogenes*	6	93.0–97.8	12.20
2	*Staphylococcus arlettae*	1	96.0	2.44
3	*Klebsiella aerogenes*	1	98.0	2.44
4	*Escherichia coli*	3	91.5–97.5	7.32
5	*Bacillus* spp.	7	90.2–97.7	14.63
6	*Shewanella decolorationis*	1	97.5	2.44
7	*Providencia* spp.	3	88.7–97.2	7.32
8	*Acinetobacter* spp.	3	95.3–96.0	7.32
9	*Alishewanella soliquinat*	2	97.0–97.6	4.88
10	*Halomonas johnsoniae*	2	97.0–97.6	4.88
11	*Pseudomonas* spp.	9	92.0–98.0	19.51
12	*Aeromonas* spp.	1	91.0	2.44
13	*Alcaligenes aquatilis*	1	91.8	2.44
14	*Acinetobacter schindleri*	1	96.5	2.44
	Total	41		100

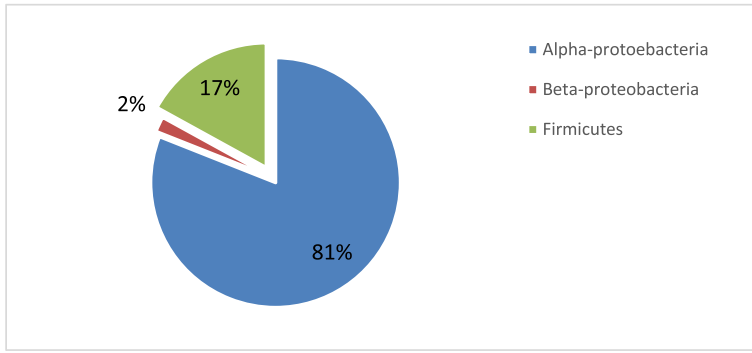


Fig. 3 Taxonomic distribution of bacteria in Aba-Eku MSW leachate samples

The nitrate (NO_3^-) concentration in the groundwater samples ranged between 0.3 and 2.3 mg/L, showing a mean value of 1.1 mg/L. Wells AB5 and AB6 sited some distances away from the waste dumpsite, showed the highest values of nitrate, which were 2.3 and 2.2 mg/L, respectively. The ammonium (< 0.1 mg/L) contents, like nitrate, showed low values in the groundwater samples and fall below the 0.5 mg/L WHO recommended threshold. Sulphate (SO_4^{2-}) concentration levels (58.3–141.7 mg/L, a mean value of 95.1 mg/L) varied greatly in the well water samples. It could be observed that about 27% of the groundwater samples exceeded the Nigeria Standard for Drinking Water Quality (NSDWQ) maximum permissible limit of 100 mg/L.

Elevated phosphate (0.33–4.87 mg/L, with a mean value of 0.73 mg/L) concentrations were noted in the groundwater samples of the area. High chloride ion concentration was observed in Well AB1 (426 mg/L) sample. However, the chloride ion concentration levels in all other well water samples in the study area were below the WHO and NSDWQ permissible limits for drinking water (Table 5). Bicarbonate (HCO_3^-) and carbonate (CO_3^-) ions mainly account for the total alkalinity (TA) of the water samples. The TA values in the well water samples of the study area were in the range of 22.5 to 95 mg/L, which were well below the WHO and NSDWQ desirable permissible limits of 310–500 and 500–1000 mg/L, respectively. The total hardness (TH) of the groundwater samples ranged between 130.55 and 649.51 mg/L, with a mean value of 340.39 mg/L. It could be observed that all the water samples indicated TH values above the 100 mg/L WHO recommended level but generally fall below the 500 mg/L maximum permissible limit. The well water sample AB1 shows noticeably high TH value (649.51 mg/L).

The concentrations of Na in the water samples varied widely between 26.3 and 200.6 mg/L, with a mean concentration value of 49.6 mg/L. The Na contents in the groundwater samples were below the NSDWQ desirable permissible limit, except for Well AB4 (200.6 mg/L) that showed a concentration level slightly higher than 200 mg/L NSDWQ permissible limit. Potassium concentrations in the groundwater samples ranged between 72.3 and 108.5 mg/L, with mean concentration value of 86.7 mg/L. The Ca levels in the well water samples varied widely between 6.3 and 50.4 mg/L (a mean concentration value of 23.9 mg/L). The water samples of the area are characterised by low Mg (28–136.1 mg/L, with a mean concentration value of 68.5 mg/L) contents. The Fe and Mn concentrations in the groundwater samples ranged from 3.60 to13.40 μg/L (mean concentration of values of 7.8 μg/L) and 0.96 to 14.93 μg/L (mean concentration value of 5.6 μg/L), respectively. Heavy metals, such as Cu (0.2–4.5 μg/L), Zn (2.0–2.5 μg/L), and Ni (0.23–4.4 μg/L) were below the WHO and NSDWQ permissible limits for drinking water; while Cd and Pb were below detection limits in all the groundwater samples.

It can be observed that cations, including Fe, Mn, Co, Zn, and Ni, indicate reducing trends with increasing distances from the MSW dumpsite; while anions, such as NO_3^-, SO_4^{2-}, PO_4^{3-}, and Cl^-, do not show any discernible increasing or decreasing trends with increasing distances from the dumpsite (Fig. 4). The major ionic constituents of the groundwater samples respectively expressed as the percentages of the total cations and anions in milliequivalents per litre are represented on Piper trilinear plot (Fig. 5). As can be observed in Fig. 5, most of the groundwater samples are within the field of no-dominant cations except a few samples indicated in the Na + K and Mg fields. Similarly, the triangle of the anion shows the groundwater samples generally plotting in the no-dominant anions region, except a sample proximal to the MSW disposal facility that plots in the Cl^- field.

Table 3 Physical and chemical analyses of Aba-Eku MSW leachate

Parameters	Minimum values	Maximum values	Mean	WHO wastewater discharge limits
Colour	Black	Black		
pH	8.80	9.70	9.25	6–9
EC (μS/m)	0.57	12.55	11.46	
Turbidity (mg/L)	396.01	441.01	418.51	
TSS (mg/L)	421.00	480.00	450.50	60
TDS (mg/L)	396.01	1102.01	1050.25	1500
Total solids (mg/L)	1418.50	1583.00	1500.75	
Total alkalinity (mg/L)	400.00	420.00	410.00	
Total hardness (mg/L)	840.00	920.00	880.00	
Dissolved oxygen (mg/L)	0.00	0.00	0.00	> 1.0
BOD (mg/L)	50.41	66.70	59.06	60
COD (mg/L)	166.10	186.10	176.10	150
BOD_5/COD	0.30	0.35	0.33	
Nitrate (mg/L)	84.60	98.16	91.38	45
Phosphate (mg/L)	46.21	66.21	56.21	15
Sulphate (mg/L)	820.61	894.61	857.61	300
Ammonia (mg/L)	98.01	134.01	116.01	
Total chloride (mg/L)	1620.00	1920.00	1770.00	
Ca (mg/L)	134.41	139.59	121.60	200
Mg (mg/L)	119.09	218.75	133.65	60
K (mg/L)	791.36	1852.89	1169.71	
Na (mg/L)	713.76	795.23	605.39	200
P (mg/L)	0.46	12.41	7.03	
S (mg/L)	35.37	77.96	64.00	
Mn (mg/L)	1.65	6.40	3.39	0.2
Fe (mg/L)	2.08	23.03	12.36	5
Al (mg/L)	0.05	14.67	6.84	2
Co (μg/L)	5.8	41.09	23.09	50
Ni (μg/L)	9.75	103.77	55.15	200
Cu (μg/L)	4.31	1077.56	477.84	200
Zn (μg/L)	4.59	1971.25	988.11	5000
As (μg/L)	0.60	15.40	7.36	50
Se (μg/L)	0.90	< 7.81		50
Cd (μg/L)	0.04	19.39	7.82	10
Hg (μg/L)	0.03	1.98	0.86	2
Pb (μg/L)	0.53	261.23	120.61	200
Mo (μg/L)	2.73	53.90	30.56	10
Cr (μg/L)	3.55	312.79	138.93	20
V (μg/L)	2.08	59.81	29.12	
U (μg/L)	1.50	2.29	1.81	

TSS, total suspended solids; *TDS*, total dissolved solids; *TS*, total solids; *DO*, dissolved oxygen; *BOD*, biochemical oxygen demand; *COD*, chemical oxygen demand

Table 4 Description of wells around Aba-Eku MSW dumpsite

Location code	Well depth (m)	Concrete rings	Uses	Observation
AB1	~ 5	Present	Domestic purpose	Covered with iron plate
AB2	ND	Present	Domestic purpose	Covered with iron plate
AB3	~ 5	Present	Domestic purpose	Covered with steel
AB4	~ 2	Absent	Abandoned	No cover and very shallow
AB5	ND	Present	Domestic purpose	Covered with aluminium plate
AB6	ND	Present	Domestic purpose	Covered with iron steel
AB7	~ 2	Present	Domestic purpose	Covered with iron plate and very shallow
AB8	ND	Absent	Domestic purpose	Covered with a plank
AB9	~ 6	Present	Domestic purpose	Covered with iron plate
AB10	ND	Absent	Domestic purpose	Well dug along the root of a palm tree
AB11	ND	Present	Domestic purpose	Covered with iron plate
AB12	ND	Present	Domestic purpose	Covered with iron plate
AB13	~ 7	Present	Domestic purpose	Close to an unlined septic well
AB14	~ 7	Present	Domestic purpose	Covered with iron plate and beside a rock outcrop

ND, not determined

Correlation among ionic constituents of the groundwater samples

The correlation matrix for the physicochemical parameters of the groundwater samples around the dumpsite is displayed in Table 6. Moderate to high positive correlations of Mg with Ca (0.51), K (0.55), Na^+ (0.59), Fe (0.73), and between K and Fe (0.68) were observed in this study. Strong correlations of TDS with macrocomponents including Na (0.63), Mg (0.96), K (0.50), Fe (0.79), Cl^- (0.63), SO_4^{2-} (0.40), TA (0.55), and heavy metals such as Zn^{2+} (0.70), Ni^{2+} (0.74), and Cu^{2+} (0.32) were observed. Electrical conductivity shows significant positive correlation with Mg (0.93), K (0.71), Fe^{2+} (0.90), Zn^{2+} (0.81), Cl^- (0.88), TDS (0.93), TA (0.66), and moderate positive correlation with Ni^{2+} (0.51) and Mn^{2+} (0.47) (Table 6) but indicate very weak positive to negative correlation with SO_4^{2-} (0.12) and NO_3^- (0.06), NH_4^+ (− 0.69).

Principal component analysis

The result of the principal component analysis (PCA) (Fig. 6) indicates two major components, which account for the potential sources of ionic enrichment in the groundwater samples. The first component (PC1) comprises the macro components, including Na, Mg, K, and SO_4^{2-}; while the second component (PC2) consists of heavy metals, such as Ni, Cu, Pb, Co, Zn, Mn, and Fe. Calcium is observed to show a negative association with PC1 and positive affinity with PC2.

Trace elements in the stream receiving leachate from Aba-Eku MSW dumpsite

The results of the analysis of the trace elements in the stream receiving leachate directly from Aba-Eku dumpsite are presented in Fig. 7. There were high concentrations of trace metals in the stream at 100-m distance away from the dumpsite, while the trace element concentrations of the stream at 200 m away from the dumpsite were significantly reduced. Zn (15.3 μg/L) was the most abundant trace element in the stream followed by Cu (9.0 μg/L) and the least being Cd (0.06 μg/L). The trend of dominance among the heavy metals in the stream was Zn > Cu > Ni > Cr > Pb > Cd.

Leachate pollution index

The leachate pollution index (LPI) value for the MSW leachate is shown in Table 7. LPI is a useful tool in determining the environmental pollution risk associated with a municipal solid waste leachate. The organic and inorganic ions present in the leachate samples were used to calculate the leachate pollution potentials. Higher LPI values were observed during the wet season (14.46), as compared with the dry season (12.70) in this study.

Table 5 Physicochemical characteristics of groundwater around Aba-Eku MSW dumpsite

Parameters	Location code																		
	AB1	AB2	AB3	AB4	AB5	AB6	AB7	AB8	AB9	AB10	AB11	AB12	AB13	AB14	Min	Max	Mean	WHO*	NSDWQ**
T (°C)	27.2	26.6	26.9	25.5	26.6	27.2	28.4	28.2	27.3	28.0	27.9	28.8	28.3	28.2	25.5	28.8	27.5		
EC (μS/m)	2.6	1.6	1.4	1.9	0.5	1.0	0.6	1.0	0.7	1.0	1.1	0.9	0.8	1.1	0.5	2.6	1.2		1
pH	6.7	6.7	6.7	6.6	6.4	6.6	6.2	6.2	6.0	6.3	6.7	6.1	6.1	6.4	6.0	6.7	6.4		6.5–8.5
TDS (g/L)	0.6	0.4	0.3	0.6	0.2	0.2	0.3	0.2	0.1	0.2	0.3	0.2	0.2	0.3	0.1	0.6	0.3		
TH (mg/L)	649.5	397.1	359.6	637.4	155.8	295.2	130.6	263.9	146.9	255.5	321.5	363.9	326.5	462.2	130.6	649.5	340.4		
TA (mg/L)	80.0	92.5	75.0	52.5	55.0	52.5	50.0	27.5	35.0	37.5	50.0	27.5	22.5	50.0	22.5	92.5	50.5		
NO_3^- (mg/L)	1.0	1.4	0.8	1.0	2.3	2.2	0.7	1.0	0.4	1.0	0.7	0.5	1.5	0.3	0.3	2.3	1.1	50	50
NH_4^+ (mg/L)	0.05	0.06	0.07	0.08	0.07	0.08	0.08	0.09	0.07	0.07	0.09	0.07	0.08	0.09	0.1	0.1	0.1		
SO_4^{2-} (mg/L)	82.0	64.2	94.7	141.7	87.7	58.3	90.0	108.5	96.8	113.3	121.4	91.9	92.4	88.2	58.3	141.7	95.1	500	100
PO_4^- (mg/L)	0.35	0.46	4.87	0.50	0.45	0.44	0.42	0.36	0.52	0.46	0.38	0.47	0.49	0.33	0.33	4.87	0.75		
Cl^- (mg/L)	426.0	71.0	62.1	142.0	142.0	71.0	71.0	150.9	115.4	97.6	115.4	133.1	115.4	133.1	62.1	426.0	131.9	250	250
Na (mg/L)	46.0	43.7	39.7	200.6	36.3	40.7	30.3	35.1	36.4	39.8	42.0	34.0	42.9	26.3	26.3	200.6	49.6		200
Mg (mg/L)	136.1	82.9	71.3	128.7	32.4	60.9	28.0	56.8	31.2	56.4	61.7	68.2	62.0	82.0	28.0	136.1	68.5		0.2
Ca (mg/L)	36.6	22.9	26.9	43.9	9.2	18.2	6.3	12.4	7.6	9.7	27.4	33.7	28.9	50.4	6.3	50.4	23.9		
K (mg/L)	108.5	98.5	86.5	80.5	86.2	80.2	86.2	95.2	80.5	72.5	78.5	82.5	86.2	92.3	72.5	108.5	86.7		
P (mg/L)	1.33	3.14	5.13	4.90	1.16	1.93	3.10	1.24	4.54	3.43	5.12	2.09	3.01	1.19	1.2	5.1	3.0		
Fe (μg/L)	13.4	11.1	9.6	9.6	8.9	7.4	6.6	5.9	6.6	11.1	5.9	5.1	3.6	4.4	3.6	13.4	7.8		300
Mn (μg/L)	14.9	3.1	ND	1.0	ND	5.7	ND	ND	ND	ND	12.0	1.3	ND	0.96	1.0	14.9	5.6		200
Cu (μg/L)	3.4	3.0	3.2	3.4	3.6	4.5	4.3	3.8	4.0	3.8	3.6	0.2	0.4	0.47	0.2	4.5	3.0	200	1000
Co (μg/L)	2.1	1.8	1.4	1.1	0.4	0.4	0.1	BDL	BDL	11.1	10.3	9.0	7.8	6.6	BDL	11.1	4.3		
Zn (μg/L)	2.5	2.5	2.4	2.3	2.3	2.2	2.1	2.0	2.1	2.4	2.0	2.0	2.0	2.0	2.0	2.5	2.2	300	3000
Cd (μg/L)	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL		3	3
Pb (μg/L)	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL		10	10
Ni (μg/L)	1.11	1.01	ND	4.14	ND	0.49	ND	ND	ND	ND	0.44	0.34	ND	0.23	0.23	4.1	1.1	7	20

ND, not determined; *T*, temperature; *EC*, electrical conductivity; *TDS*, total dissolved solids; *TH*, total hardness; *TA*, total alkalinity

**WHO*, World Health Organization guidelines for drinking water

***NSDWQ*, Nigeria Standard for Drinking Water Quality; BDL, below detection limits

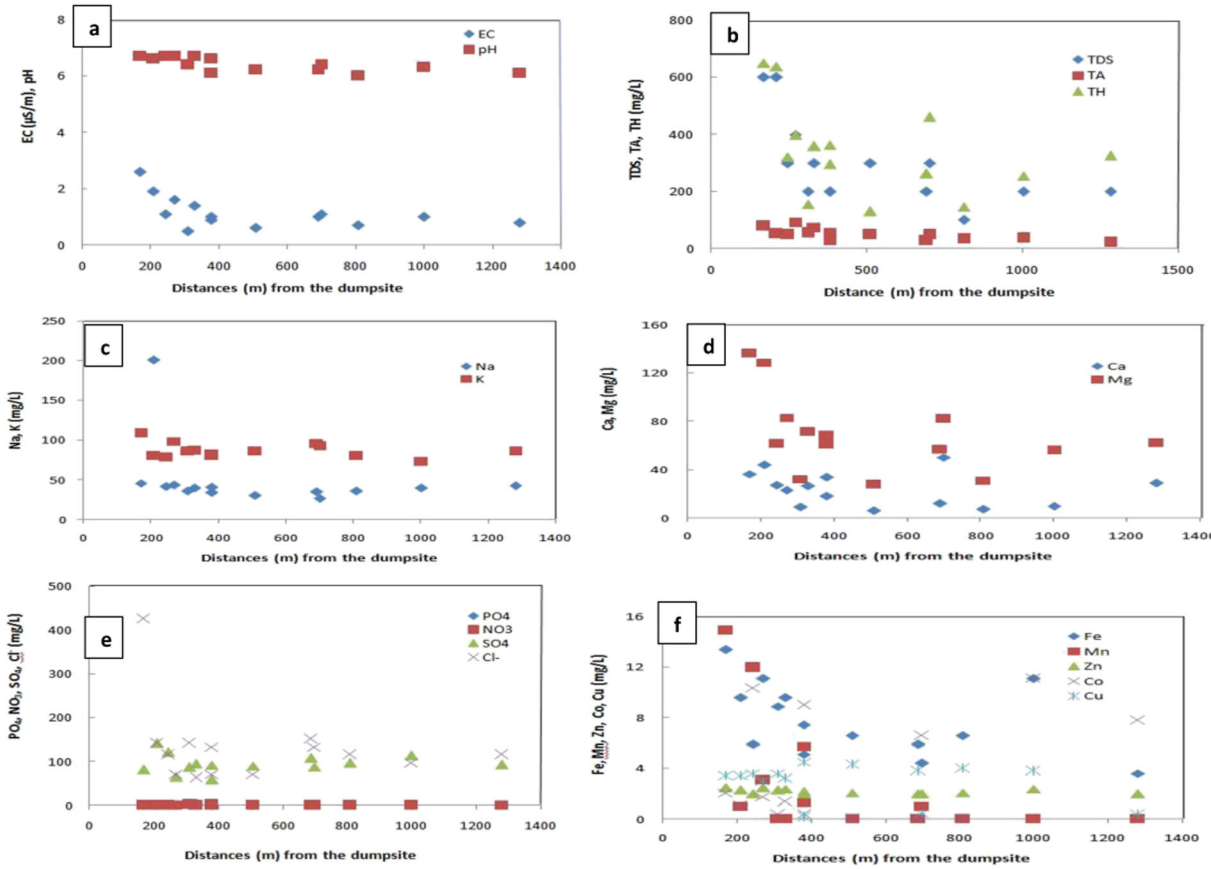


Fig. 4 Variation of some parameters with distances away from Aba-Eku dumpsite **a** EC and pH; **b** TDS, TA, and TH; **c** Na and K; **d** Ca and Mg; **e** PO_4, NO_3^-, SO_4^{2-}, and Cl^-; **f** Fe, Mn, Zn, Co, and Cu

Geophysical survey

The vertical electrical sounding (VES) imaging carried out at the dumpsite is presented in Fig. 8. Figure 8 consists of Traverse 1-3, having 9 VES in the south-north direction, downslope of the dumpsite. Traverse 1 comprises VES 1-3; underlain by topsoil, lateritic clay layer, and fresh bedrock. The delineated overburden lithologic layers are characterised by very low resistivity values (5–24 Ωm). The impact of the low anomalous resistivity is up to a depth greater than 4 m along this traverse. Traverse 2, consisting of VES 4-6, reveals subsurface layers comprising the topsoil, the lateritic clay layer, and the weathered bedrock. Layers above the fresh bedrock indicated very low resistivity values (3–33 Ωm) from the surface to depth of about 5 m. Traverse 3 shows VES 7-9, with similar lithologic characteristics along Traverse 1. The delineated subsurface sequence (topsoil and lateritic clay layer) are characterised by very low electrical resistivity (2–20 Ωm).

The ranges of resistivity values for the topsoil, the lateritic clay layer, and the weathered bedrock along the traverses were 5–33 Ωm, 2–24 Ωm, and 15–146 Ωm, respectively. This shows an anomalous deviation from that of the control flow direction values of 37 Ωm, 86 Ωm, and 416 Ωm respectively (Fig. 9). Figure 10 shows the leachate flow model, as deciphered from topographic data and geoelectrical imaging diagram.

The extent of infiltration of leachate from the MSW disposal facility into the soil subsurface was determined by 2D-resistivity imaging (Fig. 11a). The result shows low resistivity values (5–20 Ωm) zone at the first layer of the profile to 5 m depth, which positively correlates with the high EC values of the leachate samples and the VES carried out. Moderately low resistivity value (47 Ωm) was also observed in the second layer and the resistivity increased at depth. High resistivity values (164 and 167 Ωm) were observed at a depth beyond 7 m towards the NW and SE directions. When compared with the control traverse (Fig. 11b), higher resistivity

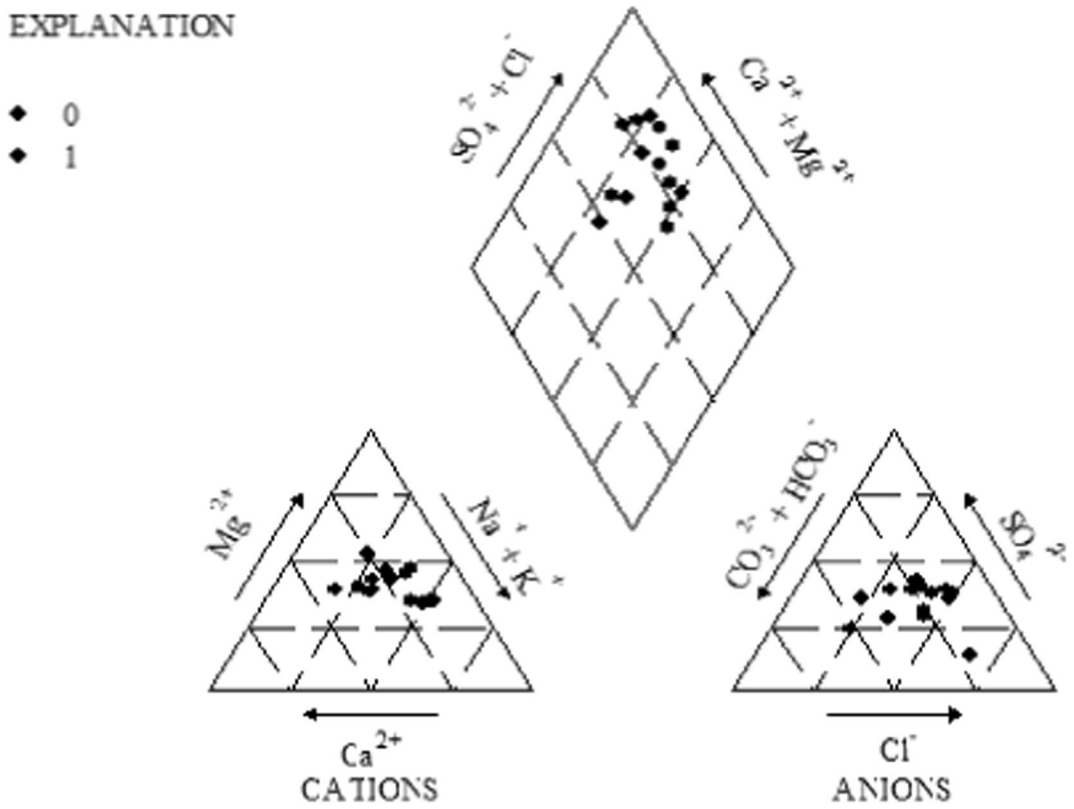


Fig. 5 Piper diagram showing the hydrochemical composition of water samples around Aba-Eku dumpsite

values (70 Ωm) were observed at 5 m, underlain by layer > 200 Ωm resistivity value.

Discussion

The presence of coliforms in the leachate samples confirms the disposal of human and animal wastes into the waste stream. The heterotrophic count (179×10^6 CFU/mL) indicated that there was an abundant diversity of bacteria in the leachate, and this can be attributed to the presence of organic nutrients which support the growth of these microorganisms. *Pseudomonas* spp. are ubiquitous in nature because of their tremendous ability to degrade many organic compounds (Atlas and Bartha 1993). The result of this study is similar to the findings of Xie et al. (2012), who have earlier reported the abundance of *Pseudomonas* spp. in an aged landfill leachate treatment plant in China; thus indicating that these bacteria play an important role in the decomposition of organic matter. *Bacillus* is known to be responsible for the decomposition of cellulose materials in the waste (Song et al. 2015; Jabari et al. 2016). Huang et al. (2005) confirm the abundance of *Bacillus* in municipal waste leachate. *Enterobacter aerogenes* and *Escherichia coli* are members of the coliform group, which are normal flora of humans and warm-blooded animals in the tropics. Their presence in the leachate samples could be attributed to the discharge of faecal materials into the waste streams. The bacterial taxonomic distribution in the MSW leachate obtained from this study is consistent with the reports of landfill leachate studies in some countries, including China, Japan, Colombia, India, and Hong Kong, in which the phylum Proteobacteria was the most dominant irrespective of the geographical location or method of determination (Huang et al. 2004; Sawamura et al. 2010; Gomez et al. 2011; Xie et al. 2012; Krishnamurthi and Chakrabarti 2013; Ye and Zhang 2013; Song et al. 2015). Proteobacteria have been reported to play an important role in wastewater decomposition (Jabari et al. 2016). Ye and Zhang (2013) reported 20.25% of Firmicutes in the influent samples of municipal wastewater treatment plant by 16S rDNA pyrosequencing method.

The presence of coliforms in the leachate samples, especially *Escherichia coli* and *Enterobacter aerogenes*,

Table 6 Correlation matrix among the parameters of groundwater samples around Aba-Eku dump site

	T °C	EC	pH	TDS	TA	SO_4^{2-}	NO_3^-	NH_4^+	Cl^-	Na	Mg	Ca	K	P	Fe	Mn	Cu	Co	Zn	Ni
T °C	1.00																			
EC	− 0.58	1.00																		
pH	− 0.64	0.54	1.00																	
TDS	− 0.72	0.93	0.52	1.00																
TA	− 0.52	0.66	0.75	0.55	1.00															
SO_4^{2-}	− 0.26	0.12	0.00	0.40	− 0.38	1.00														
NO_3^-	− 0.47	0.06	0.48	− 0.06	0.37	− 0.51	1.00													
NH_4^+	0.19	− 0.69	− 0.16	− 0.46	− 0.62	0.42	− 0.22	1.00												
Cl^-	− 0.02	0.80	0.19	0.63	0.30	0.03	− 0.21	− 0.61	1.00											
Na	− 0.79	0.38	0.19	0.63	− 0.04	0.71	0.05	0.10	− 0.01	1.00										
Mg	− 0.60	0.93	0.31	0.96	0.41	0.33	− 0.11	− 0.51	0.73	0.59	1.00									
Ca	0.02	0.23	− 0.32	0.39	− 0.25	0.51	− 0.75	0.21	0.33	0.34	0.51	1.00								
K	− 0.07	0.71	0.28	0.51	0.75	− 0.44	− 0.06	− 0.77	0.69	− 0.29	0.55	0.16	1.00							
P	− 0.46	− 0.03	0.37	0.22	− 0.05	0.73	− 0.02	0.35	− 0.35	0.60	0.00	− 0.13	− 0.54	1.00						
Fe	− 0.66	0.90	0.64	0.79	0.82	− 0.15	0.40	− 0.81	0.59	0.28	0.73	− 0.17	0.68	0.00	1.00					
Mn	0.06	0.47	0.58	0.27	0.35	− 0.05	0.12	− 0.33	0.65	− 0.28	0.20	− 0.29	0.36	0.02	0.44	1.00				
Cu	− 0.64	0.36	0.85	0.32	0.46	− 0.02	0.77	− 0.11	0.06	0.29	0.17	− 0.57	− 0.05	0.40	0.55	0.52	1.00			
Co	0.78	− 0.53	− 0.48	− 0.49	− 0.58	0.32	− 0.74	0.43	− 0.14	− 0.41	− 0.50	0.18	− 0.32	0.15	− 0.70	0.06	− 0.57	1.00		
Zn	− 0.69	0.81	0.61	0.70	0.88	− 0.28	0.48	− 0.80	0.43	0.24	0.64	− 0.22	0.68	− 0.06	0.97	0.27	0.51	− 0.78	1.00	
Ni	− 0.85	0.51	0.26	0.74	0.11	0.64	0.08	− 0.05	0.08	0.99	0.69	0.33	− 0.13	0.54	0.42	− 0.23	0.31	− 0.50	0.39	1.00

confirms the disposal of faecal materials into the waste. These organisms have been implicated in some infectious diseases, such as diarrhoea and gastroenteritis. There is a serious public health threat when the leachate is released into the environment without any form of treatment, as is the case in MSW Aba-Eku dumpsite. The organisms found in the leachate are relatively stable in water and could infiltrate into the groundwater through seepage from soil pore space, rock fractures, and percolation through the vadose zone (Darnault et al. 2004; Gerba et al. 2011; Pandey et al. 2014). Many communities around the dumpsite rely on water from the untreated hand-dug wells as their main source of water supply. The outbreak of some diseases has been attributed to microbial contamination of groundwater from municipal wastes. Oguntoke et al. (2009) reported a high incidence of cholera and diarrhoea in some parts of Ibadan; out of which, 23.6% of the well water sampled were positive for *Vibrio cholerae*.

The colour of the leachate was black, and this could be ascribed to the formation of ferric hydroxide colloids from the oxidation of ferric ions (Christensen et al. 2001; Mor et al. 2006). Abdelwaheb et al. (2012) reported a pH value of 8.2 for landfill leachate in Jabel Chakir, Tunisia, which is similar to the findings of this study. The alkaline nature of the leachate indicates the presence of carbonate, bicarbonate, and hydroxide compound of calcium, sodium, and potassium (Jorstad et al. 2004; Niloufer et al. 2013). This result is also similar to pH 8.3 reported by Song et al. (2015) for a municipal waste leachate in China. Increased pH from 6 to 8 has been reported to indicate the production of volatile fatty acids and carbon dioxide in landfill dumpsite (Christensen et al. 2001; Kehew 2001; Kjeldsen et al. 2002). Electrical conductivity is used to measure the concentration of ions in water samples. The extremely high values of EC (0.57–12.55 μS/m) can be attributed to high concentrations of cation and anions in the leachate samples and these invariably reflected the high salinity of the leachate.

The high total suspended solids (421–480 mg/L) of the leachate samples will make it possible for microorganisms to adsorb to solid particle surfaces, thereby enhancing biodegradation of organic pollutants (Mohod and Dhote 2013). Total dissolved solids (997.50–1102.01 mg/L) found in the leachate samples possibly indicate a high amount of soluble ions (Oketola and Akpotu 2015). The total hardness of the leachate samples ranges between 840 and 920 mg/L. The high total hardness can be attributed to the high concentration of Ca and Mg ions in the leachate samples (Christenson et al. 1999; Niloufer et al. 2013). Non-detectable DO values showed that the leachate was highly polluted with organic matter, invariably pointing to high organic activities and consequently high oxygen consumption by organisms. The concentrations of BOD ranging between 50.41 and 66.70 mg/L (mean value of 59.06 mg/L) and COD ranging between 166.1 and 186.10 mg/L (mean value of 176.10 mg/L) found in Aba-Eku MSW leachate samples were lower when compared with studies of Oketola and Akpotu (2015), who reported BOD and COD of 507 mg/L and 1780 mg/L, respectively. The variations in values may possibly be attributed to the fact that different decomposition stages are taking place simultaneously at various locations within an open dumps waste management system. Also, the BOD and COD values decrease as the dumpsite ages (Kjeldsen et al. 2002).

The mean BOD_5/COD value (0.33) indicated that substantial components of the organic matter in the leachate were not readily biodegradable, as they possibly contain humic and fulvic acids (Kjeldsen et al. 2002; Banar et al. 2006).

The presence of phosphate and nitrate in the leachate could be as a result of the discharge of agro-allied and laundry wastes into the waste disposal site (Kale et al. 2010; Niloufer et al. 2013). Industrial and hospital wastes containing sulphur and the possible oxidation of sulphide ores might be responsible for the high sulphate values observed in the leachate (Raman and Narayanan 2008). High concentrations of ammonia and chloride ions in the leachate samples of this study could be attributed to the discharge of domestic waste into the dumpsite, and this could be hazardous when released to water bodies. The high concentrations of K and Na in the leachate samples indicate that they were not readily affected by microbial activities within the waste site. The leaching of fertiliser and potassium-bearing minerals (K-feldspar and mica) from the waste soil into the leachate are possible sources of high concentration levels of potassium in the leachate samples (Naveen et al. 2014).

Concentrations of Ca (90.80–139.59 mg/L) and Mg (63.11–218.75 mg/L) in the waste leachate indicate the deposit of animal bones and horns into the waste (Oketola and Akpotu 2015). The dumping of textile and cosmetic waste materials into the waste site are other major contributions of Mg into the leachate

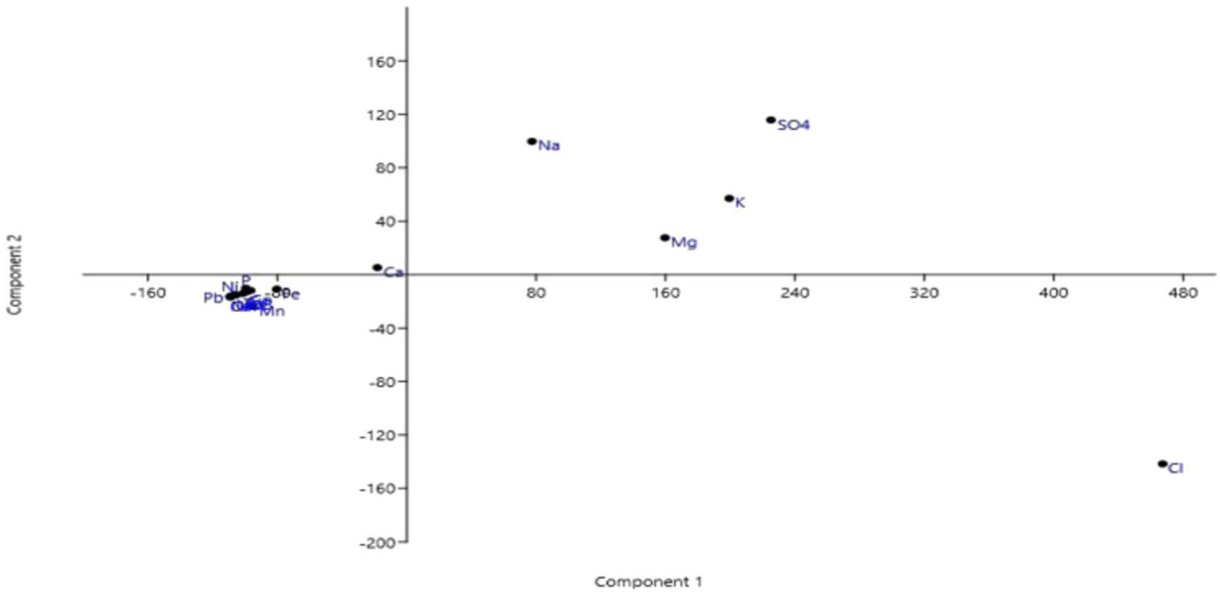
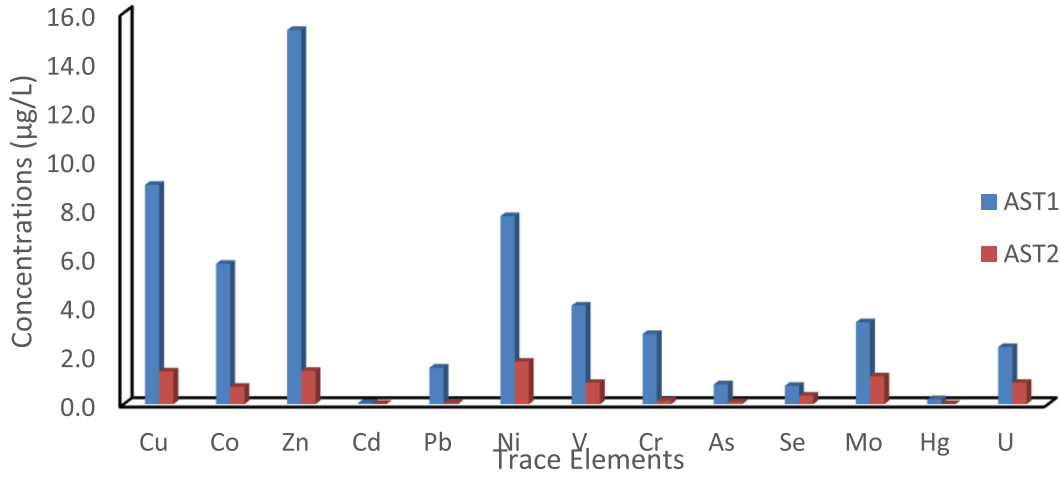


Fig. 6 Principal component analysis of the chemical parameters in the groundwater

(McBean et al. 1995). The presence of Al in the leachate samples showed that wastes containing aluminium were disposed into the waste site. The high concentration of heavy metals, such as Zn, Hg, Ni, Cd, and Pb, in the leachate samples indicates the disposal of household hazardous chemicals, lead-based materials (Zn rods and lead-acid accumulator batteries), discarded automobile parts (metal scraps and metal foils), fluorescence bulbs, and other electronic wastes, as well as degreasing agents into the dumpsite. These could be very hazardous when released into the receiving stream and the surrounding groundwater of the area (Mor et al. 2006; Kale et al. 2010).

The modest temperature values recorded for the water samples invariably reflected low levels of organic substrate and apparently low organic activities in the water; while the slightly acidic pH values (mean pH value of 6.4) of the groundwater samples indicated the



AST1 ~100 m away from dumpsite; AST2 ~200 m away from dumpsite

Fig. 7 Trace elements in stream receiving leachate from Aba-Eku dumpsite. AST1 ~ 100 m away from dumpsite; AST2 ~ 200 m away from dumpsite

Table 7 The LPI values of Aba-Eku MSW leachate samples

S/N	Pollutants			Individual pollution rating (Pi)		Pollutant weight (Wi)	Overall rating pollution (WiPi)	
		Dry season	Wet season	Dry season	Wet season		Dry season	Wet season
1	pH	8.80	9.70	10	35	0.055	0.55	1.925
2	TDS	396.01	1102.01	6	6	0.05	0.3	0.3
3	BOD	66.70	50.41	7	6	0.061	0.427	0.366
4	COD	186.10	166.10	8	8	0.062	0.496	0.496
5	NH_4^+	98.01	134.01	10	11	0.051	0.51	0.561
6	Fe	2.08	23.03	5	5	0.045	0.225	0.225
7	Cu	0.00	1.08	5	7	0.05	0.25	0.35
8	Ni	0.01	0.10	5	5	0.052	0.26	0.26
9	Zn	0.01	1.97	5	5	0.056	0.28	0.28
10	Pb	0.00	0.26	5	5	0.063	0.315	0.315
11	Cr	0.00	0.31	5	5	0.064	0.32	0.32
12	Hg	0.00	0.00	5	5	0.062	0.31	0.31
13	As	0.00	0.02	5	5	0.061	0.305	0.305
14	Chloride	1920.00	1620.00	17	17	0.048	0.816	0.816
15	CFU/mL	$150x10^6$	$179x10^6$	100	100	0.052	5.2	5.2
	Total					0.832	10.564	12.029
	LPI						12.697	14.458

TDS, total dissolved solid; *BOD*, biochemical oxygen demand; *COD*, chemical oxygen demand; *CFU/mL*, colony forming unit per millilitre. Readings in mg/L except pH

contributions from the soil and water-rock interactions. Ehinola (2002) reported a pH range of 6.0 to 7.4 in groundwater samples of some hand-dug wells in some parts of southwest Nigeria, which is similar to the findings of this study. Salami et al. (2014) also reported an acidic pH range of 4.99 and 5.67 in well waters around Ofe-Afa MSW dumpsite in Lagos, southwest Nigeria. The EC and TDS values of the water samples can, therefore, be attributed to the ionic loadings from aquifer solids of the area with minor contributions from the MSW dumpsite. Electrical conductivity, TDS, and pH values in the well water samples show systematic reduction with increasing distance away from the dumpsite (Fig. 4). While the elevated values of EC and TDS close to the MSW dumpsite can be attributed to possible leachate contamination, their decreasing trends farther away from the dumpsite indicated local groundwater dilution of the MSW effluents (Christensen et al. 2001). The high nitrate contents may be due to local input from septic tank leakage rather than contamination from the MSW dumpsite. However, the nitrate levels in all other wells were below the permissible limits of the 50 mg/L of the World Health Organization (WHO 2017) and the Nigerian Standard for Drinking Water Quality (NSDWQ 2008). The low nitrate contents may be attributed to low agricultural activities in the area. Elevated levels of sulphate in Well AB4 (141.7 mg/L), Well AB8 (108.5 mg/L), Well AB10 (113.3 mg/L), and Well AB11 (121.4 mg/L) wells were obviously not from the dumpsite but could be due to absence of well covers and possible inputs from local discharge of domestic wastewaters containing detergents, sulphate-bearing chemicals, and leakage from septic tanks. Consumption of sulphate-bearing water has been indicated to have a laxative effect on human health (WHO 2017).

Possible sources of phosphate in the well water may include effluents from cleaning and laundry activities and leakages from septic tanks. Groundwater sample from shallow Well AB3 shows the highest (4.89 mg/L) concentration and may require specialised treatment to make the water suitable for drinking purpose. Although, no health-related phosphate threshold value has been established for drinking waters; however, elevated phosphate levels in groundwater have been noted to over-

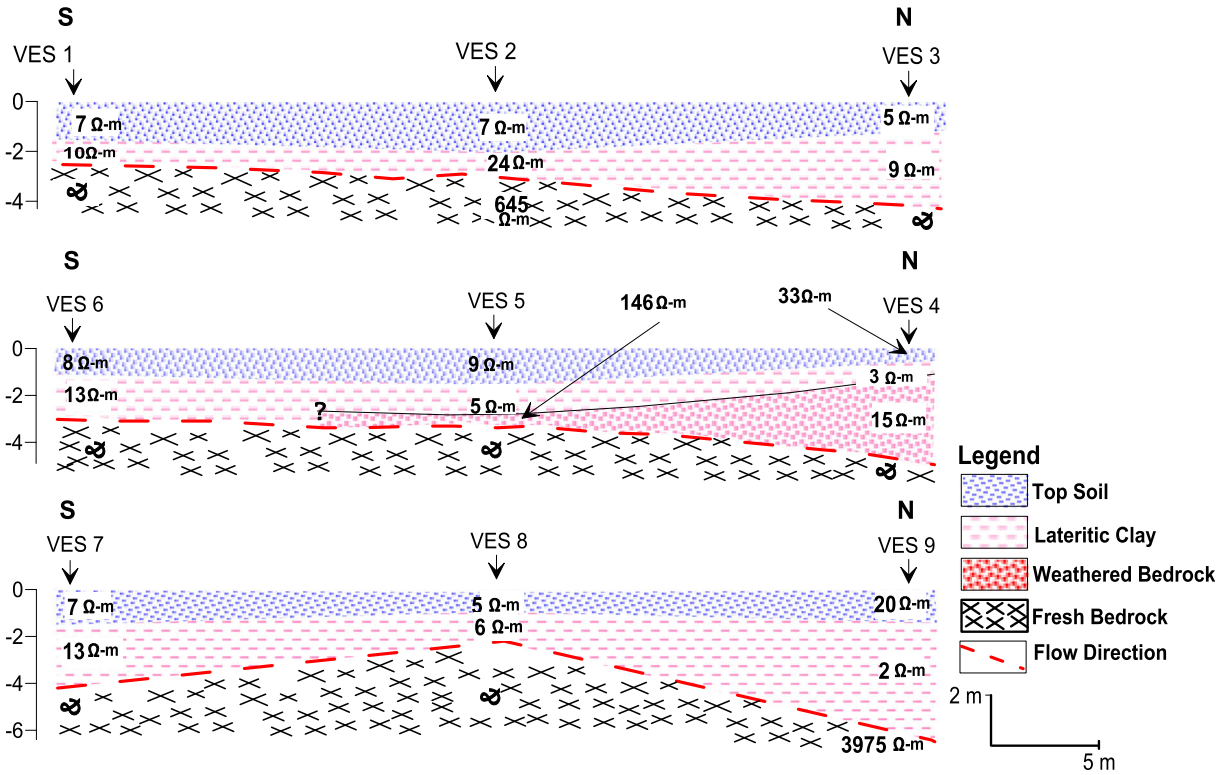


Fig. 8 Geoelectric models of the bedrock topography across three traverses indicating leachate probable flow direction

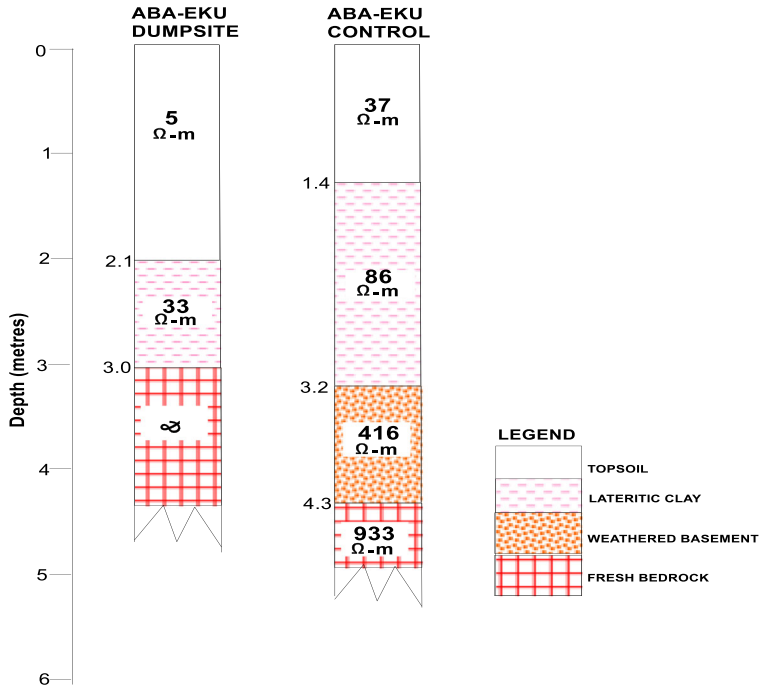


Fig. 9 Vertical electrical sounding of Aba-Eku MSW dumpsite and the control site

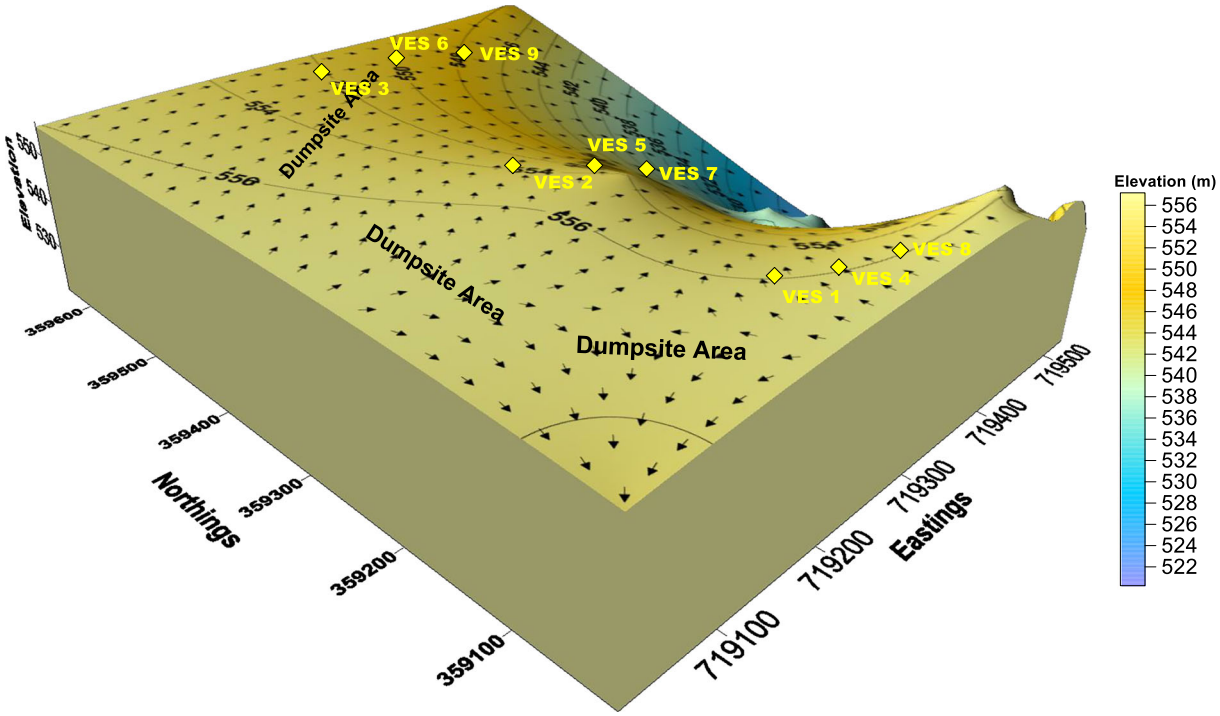


Fig. 10 Model of surface leachate flow pattern within the dumpsite

fertilise aquatic plants and catalyse eutrophication (Fadiran et al. 2008). This could be particularly deleterious in drinking waters subjected to long storage period. The high chloride ion concentration in Well AB1 (426 mg/L) sample indicated profound leachate pollution in the immediate vicinity of the waste disposal site. However, the chloride ion concentration levels in all other well water samples in the study area were below the WHO and NSDWQ permissible limits for drinking water (Table 5). These can be ascribed to progressive aquifer dilution with distances farther away from the MSW dumpsite. Chloride ion is a non-reactive anion and shows conservative chemical behaviour in aquifers and water bodies (Christensen et al. 2001; Olayinka and Olayiwola 2001; Pedretti et al. 2017; Blackmore et al. 2018). High concentration levels of Cl^- (and Na^+) in drinking waters have been linked to some heart diseases and kidney-related problems (Raman and Narayanan 2008).

Bicarbonate (HCO_3^-) and carbonate (CO_3^-) ions mainly account for the total alkalinity (TA) of the water samples. Since the study area is underlain by biotite-granites, biotite-hornblende gneisses and quartzite and not by carbonate-rick rocks, TA in the groundwater samples is possibly a contribution from the biodegradation of organic matter in the refuse dump into inorganic carbon, which ultimately resulted in bicarbonate ions (Fernandez et al. 2014; Olayinka and Olayiwola 2001). TA values signify decrease with distances from the dumpsite (Fig. 4), which attest to some contributions from MSW dumpsite. Well water sample AB1 shows noticeable high TH value (649.51 mg/L), which is traceable to the anthropogenic inputs from the nearby waste dumpsite. The reducing trends of the TH with distances away from the dumpsite also confirm the leachate inputs into the groundwater.

It is remarkable to note that the Na contents in the groundwater samples were below the NSDWQ desirable permissible limit, except for AB4 (200.6 mg/L) that showed concentration level slightly higher than 200 mg/L NSDWQ permissible limit. This could be due to leakage from septic tanks in the immediate vicinity, as high concentration levels of SO_4^{2-} (141.7 mg/L), Cl^- (142 mg/L), Mg (128.7 mg/L), and Ca (43.9 mg/L) were also recorded for the same well. The lack of concrete rings and well cover, as well as a long period of abandonment (Table 4), has invariably resulted in the well-being prone to diverse contaminations. Lithogenic inputs from K-feldspar rich biotite granites and biotite-hornblende gneisses, underlying the study area, are the

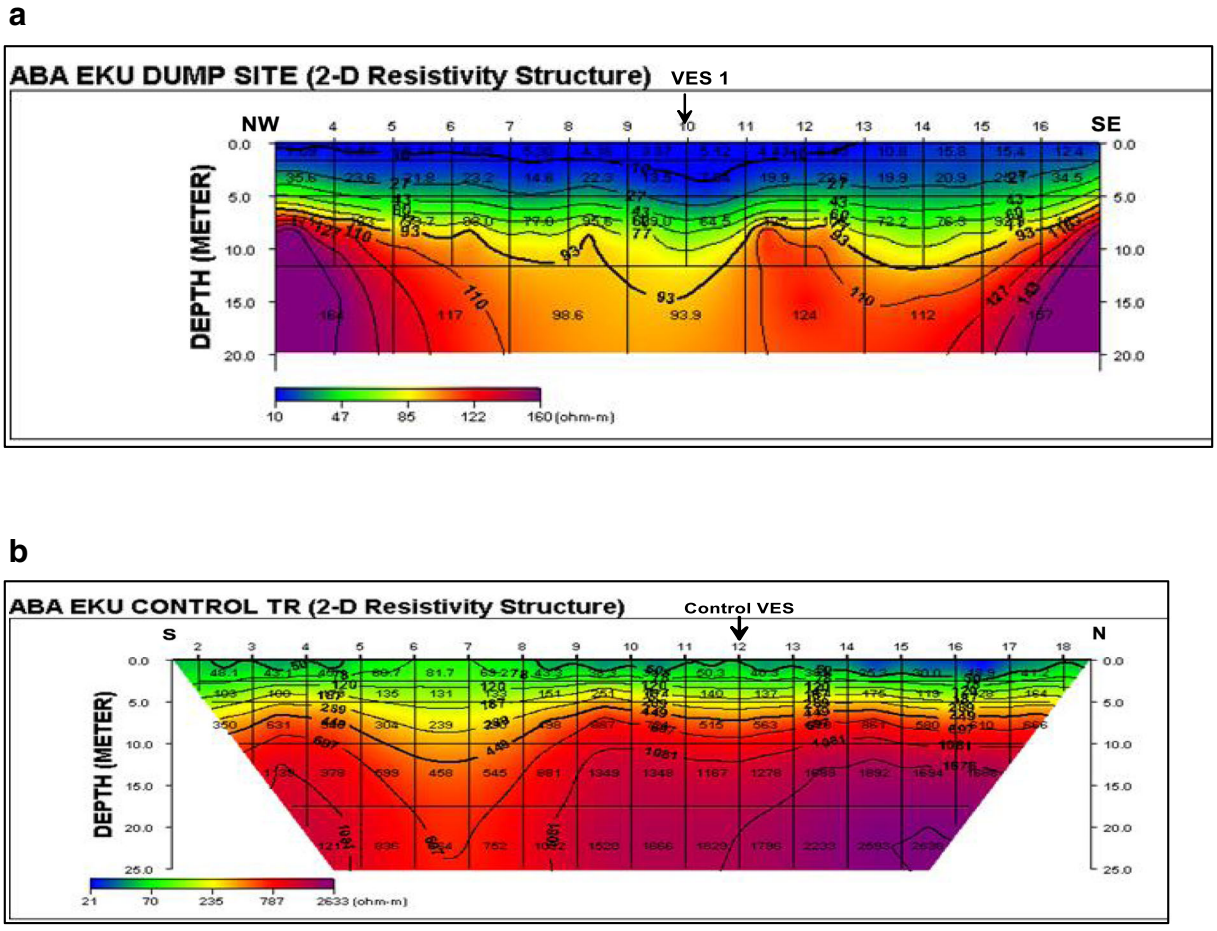


Fig. 11 **a** Resistivity image of Aba-Eku MSW site. **b** Resistivity image of the control site (100 m uphill away from Aba-Eku MSW site)

main possible sources of potassium in the water samples. However, elevated concentration of potassium in Well AB1 (close the MSW dumpsite) is an indication of leachate infiltration into groundwater of the area. The low Ca contents in the groundwater samples are consistent with the absence of carbonate-rich rocks and the very low Ca-plagioclase bearing rocks in the area as well as calcium-deficient leachate (Table 3) discharge from the Aba-Eku solid waste facility. The obvious decreasing trend of the Mg concentrations from the immediate vicinity of the dumpsite (Fig. 4) points to the MSW leachate as the source of Mg in the groundwater and not from water-rock interaction of the low Mg-bearing lithological units of the area. However, the obvious reducing Fe and Mn trends with distances away from the Aba-Eku waste disposal facility show minor inputs of these metals from MSW leachate, while reduction-oxidation processes and groundwater dilution are also possible reasons for their low concentrations observed in the water samples.

Although the MSW dumpsite receives heavy metal wastes from discarded metallic items and degreasing agents; the reduced heavy metal contents in the groundwater samples could, however, be attributed to attenuation processes by sorption, precipitation, and aquifer dilution of the MSW leachate (Christensen et al. 2001; Kehew 2001; Kjeldsen et al. 2002).

The TDS values and the concentrations of some inorganic macro-components, including Fe, Mn, Cl^-, SO_4^{2-}, NH_4^+, have been used as indicators of groundwater contaminated with leachates (Han et al. 2014; Fernandez et al. 2014). It can be observed that cations, including Fe, Mn, Co, Zn, and Ni, indicate reducing trends with increasing distances from the MSW dumpsite; while anions, such as NO_3^-, SO_4^{2-}, PO_4^{3-}, and Cl^-, do not show any discernible increasing or decreasing trends with increasing distances from the

dumpsite (Fig. 4). The above observations are consistent with higher cation exchange sites available in the clay-dominated vadose zone surrounding the MSW dumpsite, which facilitate attenuation through sorption, adsorption, absorption, and ion exchange (Christensen et al. 2001), as opposed to a few anion exchange sites in aquifer solids which invariably limit attenuation of anion constituents to groundwater mixing, redox processes, and biodegradation.

The major ionic constituents of the groundwater samples respectively expressed as the percentages of the total cations and anions in milliequivalents per litre are represented on Piper trilinear plot (Fig. 5) (Piper 1944). The Piper plot is generally employed to show the sources of dissolved constituents in water, so as to characterise the water into appropriate hydrochemical facies and precisely elucidate the hydrochemical evolution of the groundwater. As can be observed in Fig. 5, most of the groundwater samples are within the field of no-dominant cations except few samples indicated in the Na + K and Mg fields. Similarly, the anion triangle shows the groundwater samples generally plotting in the no-dominant anions region, except a sample proximal to the MSW disposal facility that plots in the Cl^- field. The groundwater samples of the study area can, therefore, be classified as no-dominant Na·Ca·Mg–HCO_3·Cl water type, which invariably reflects the neutralising effects of the ionic loadings from the MSW dumpsite leachate over the ionic contributions from the local aquifer solids via water-rock interaction.

The moderate to high positive correlations of Mg with Ca (0.51), K (0.55), Na^+ (0.59), Fe (0.73), and between K and Fe (0.68) suggest a lithogenic source from chemical dissolution of biotite, hornblende, and feldspar in the rock types of the study area. The strong correlations of TDS with macro components including Na (0.63), Mg (0.96), K (0.50), Fe (0.79), Cl^- (0.63), SO_4^{2-} (0.40), TA (0.55), and heavy metals such as Zn^{2+} (0.70), Ni^{2+} (0.74), and Cu^{2+} (0.32) are indicative of the major ionic loading from geogenic source and MSW leachate into the groundwater samples. Electrical conductivity shows a significant positive correlation with Mg (0.93), K (0.71), Fe^{2+} (0.90), Zn^{2+} (0.81), Cl^- (0.88), TDS (0.93), TA (0.66), and moderate positive correlation with Ni^{2+} (0.51) and Mn^{2+} (0.47) (Table 6) but indicates very weak positive to negative correlation with SO_4^{2-} (0.12) and NO_3^- (0.06), NH_4^+ (− 0.69). These statistically significant EC correlations with the aforelisted cations and anions further substantiate the greater inputs from water-rock interactions (geogenic source) via weathering and leaching and minor contributions from the MSW leachate, while the very weak correlation of EC with SO_4^{2-}, NO_3^-, and NH_4^+ is indicative of the intenseness of attenuations witnessed by these ions in the course of the migration of MSW leachates into the surrounding groundwater.

The result of the PCA (Fig. 6) indicates two major components, which account for the potential sources of ionic enrichment in the groundwater samples. The first component (PC1) comprises the macro components, including Na, Mg, K, and SO_4^{2-} that point to the lithogenic source, while the second component (PC2) is indicative of the minor contributions of the heavy metals (Ni, Cu, Pb, Co, Zn, and Mn) and Fe from the MSW leachate into the ambient groundwater of the area. Calcium is observed to show a negative association with PC1 and positive affinity with PC2, which reflect the weak dual contributions of the cation from both geogenic and anthropogenic sources.

The high concentrations of trace metals in the stream at 100 m distance away from the dumpsite can be attributed to the impact of the leachate on it; while the trace element concentrations of the stream at 200 m away from the dumpsite was significantly reduced, which signifies the effect of dilution along the water course. Zn (15.3 μg/L) was the most abundant trace element in the stream followed by Cu (9.0 μg/L) and the least being Cd (0.06 μg/L). Kanownik and Policht-Latawiec (2016) reported a mean value of 15 μg/L for Zn in a stream below a municipal landfill in Malopolskie Province of Krakow, which is consistent with the results of this study. The trend of dominance among the heavy metals in the stream was Zn > Cu > Ni > Cr > Pb > Cd. The concentrations of all the heavy metals were below the WHO permissible limits for drinking water (WHO 2017).

Higher LPI values observed during the wet season (14.46), as compared with the dry season (12.70) in this study, could be attributed to the influx of excess rainwater into the waste, thus dislodging more soluble materials into the leachate. This is in agreement with the report of Munir et al. (2014). A value of LPI greater than 7.4 indicates polluting leachate (Kumar and Alappat 2005; Somashekar and Sonza 2013). Aba-Eku MSW dumpsite has been in operation for 25 years, and this could account for the high LPI observed in this dumpsite because LPI values are greatly influenced by the age of the waste site (Umar et al. 2010; Munir et al. 2014).

Young MSW landfills (< 5 years) usually have low LPI values, while older landfills (> 10 years) have high LPI values (Munir et al. 2014). Higher values of LPI in Aba-Eku MSW leachate, greater than the set-mark for non-polluting MSW leachate, also suggest that the waste leachate is not yet stabilised, as indicated by the low BOD_5/COD value (0.33) and high pH values (8.80–9.70) (Umar et al. 2010; Somashekar and Sonza 2013). The LPI of this study is comparable with the report of Munir et al. (2014), who reported LPI of 13.7 and 16.7 for two aged MSW dumpsite in Pakistan. Also, Kumar and Alappat (2005) reported 15.97 for a landfill in China, which is similar to this study.

Low resistivity values at the subsurface represent the presence of leachate; this is because leachates are rich in soluble ions which allow the flow of electric current to pass through them, thus giving low resistivity value (Kaya et al. 2007). The extremely low resistivity values (2–33 Ωm) recorded from the lithologic sequence beneath VES 1-9, as compared with the Aba-Eku control section (Fig. 9) showing high geoelectrical resistivity values (37–416 Ωm), are indicative of the possible impact of leachate migration within the overburden materials in the vicinity of the dumpsite. Quantitative assessment of the layer parameters further justify the assertion, as revealed from the discriminant values observed when compared with those of the control section.

The resistivity imaging profile carried out in this study revealed the evidence of leachate percolation from the waste into the soil subsurface. Dipole-dipole profiling gives better information of the subsurface, as it relates to leachate infiltration from the dumpsite. The result shows low resistivity values (5–20 Ωm) zone at the first layer of the profile to 5 m depth, which positively correlates with the high EC values of the leachate samples and the VES carried out. The low resistivity values observed in this zone indicate the presence of leachate infiltration from the dumpsite. Moderately low resistivity value (47 Ωm) was also observed in the second layer and the resistivity values further increased at depth. High resistivity values (164 and 167 Ωm) observed at a depth beyond 7 m towards the NW and SE directions could be attributed to the presence of hard crystalline bedrock in this region, which hinders the percolation of the leachates. When compared with the control traverse (100 m uphill away from the dumpsite), higher resistivity values (70 Ωm) were observed at 5 m, underlain by layer > 200 Ωm resistivity value, thus indicating that the control site is not polluted with the leachate (Fig. 11b). The relatively low resistivity value of 30 Ωm observed at a segment towards the northern section of the traverse was, however, due to an unlined hand-dug septic well located at this point, which leaked directly into the soil subsurface.

Conclusions

The results of this study have shown the evidence of the heterogeneous nature of Aba-Eku MSW open dumpsite in Nigeria. There is a diversity of bacteria in the leachate samples and the predominant phylum being proteobacteria (83%), followed by firmicutes (17%), which is similar to many findings in the literature. Increased concentrations of cations, such as copper, aluminium, iron, manganese, molybdenum, and chromium in the leachate, confirm the discharge of hazardous chemicals into the waste. The hydrochemical status of the groundwater around the municipal waste site is largely due to the soil-water and rock-water interactions, while a minor contribution from the MSW leachate, of which the groundwater can be classified as no-dominant Na·Ca·Mg-HCO_3-Cl water type. Higher values of leachate pollution index in Aba-Eku leachate, greater than the set-mark for non-polluting MSW leachate, also suggest that the waste leachate is not yet stabilised, which is indicated by the low BOD_5/COD value and high pH values. Three geoelectrical layers were inferred from the geophysical data, which include the topsoil, the lateritic clay layer, and the weathered basement rock. Low resistivity values observed within the topsoil and the lateritic clay layers signify leachate infiltration and leachate plume generation. Although the pollutants from the leachate have not strongly infiltrated the groundwater, the geophysical data revealed a continuous migration of the leachate plume into the soil strata. Therefore, it is strongly recommended that Aba-Eku MSW dumpsite should be closed for reclamation process, in order to mitigate further percolation of the leachate into the soil subsurface, which could result in groundwater pollution in the nearest future. Groundwater abstraction around the MSW facility should also be discontinued, so as to forestall further widening of the subsurface leachate plume.

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