

Simulation of Contaminants Transport from a Dumpsite into Shallow Aquifer of the Upper Deltaic Plain Deposits, Western Niger Delta

K. E. Aweto^{*}, G. Ovwamuedo

Department of Geology, Delta State University, Abraka, Delta State, Nigeria

Received 20 August 2022, accepted in final revised form 16 February 2023

Abstract

Dumpsites are a copious point source of groundwater pollution in the Niger Delta. The objective of this study seeks to assess contaminants transported from a dumpsite around Ekpan into groundwater. Dumpsite leachate and groundwater samples were collected for analysis, and the Modular 3D Solute Transport Model (MT3D) was used to evaluate the impact of leachate on the quality and flow direction of groundwater. The chemical analyses of leachate samples showed significant potential to contaminate groundwater. The pH of greater than 7 and BOD₅/COD ratio of less than 0.1 indicated that the dumpsite is old with stabilized leachate. Groundwater models showed that the dumpsite has the propensity for Na and TDS increasing to 1500 and 3700 mg/L respectively after 10 years. The contaminants infiltrate into successive layers beneath the dumpsite and spread laterally in the direction of groundwater flow (southwestern direction). This study revealed that groundwater near the dumpsite did show some significant contamination. A pattern of decreasing concentration of contaminants with distance from the dumpsite was observed due to natural attenuation.

Keywords: Dumpsite; Leachate; Plume; Groundwater; Infiltration; Attenuation.

© 2023 JSR Publications. ISSN: 2070-0237 (Print); 2070-0245 (Online). All rights reserved.
doi: <http://dx.doi.org/10.3329/jsr.v15i2.61307> J. Sci. Res. **15** (2), 301-312 (2023)

1. Introduction

Groundwater supply systems remain a significant source of drinking water in the Niger Delta; availability of good quality water supply, however, remains a challenge. This is because groundwater quality tends to degrade as the population pressure in any geographic region increases from industrialization, urbanization, and indiscriminate disposal [1]. Open dumping has remained a widely accepted waste disposal method due to its economic advantage. However, leachate production is one of the significant weaknesses of open dumping. The leachate formed is discharged directly into the aquifer, thus contaminating the groundwater [2]. The contaminant plume may persist in groundwater undetected for an extended period, especially in the Niger Delta, because of the absence of or limited monitoring wells [3].

The absence of centralized, properly managed/lined waste disposal sites has made every household have disposal sites that are usually indiscriminately and randomly

^{*} Corresponding author: kizaweto@yahoo.com

located; presently, there are several dumpsites in the Niger Delta without suitable adequate liners. This exposes the aquifers in the Niger Delta, thus to contamination by the clusters of high densities dumpsites. Permeable layers above aquifers further compound the problem coupled with high water table, hydraulic conductivity and porosity of these aquifers. It is not unconventional to find dumpsites within a built-up environment inhabited by several people. These dumps have, over the years, become point sources for pollutants injection into the aquifer.

According to Aweto [4], groundwater use as a source of water supplies has been on the rise in recent years; these water sources present in shallow aquifers were generally poor in quality. The change of groundwater chemistry in these aquifers appears to be driven by leachate contaminant plume, an essential source of major cations and anions. Thus potential contamination of groundwater by leachate has been suggested by Akpoborie and Aweto [3]. A study by Aweto [5] using a single-point resistance log and electrical resistivity imaging at a dumpsite in Niger Delta reported significant groundwater contamination up to depths of 14 m. This study aims to assess the dumpsite's effect on the study area's shallow groundwater resource and ascertain the extent of influence on the qualities of groundwater.

2. Location and Geology

Ekpan-Warri metropolis is located in the Niger Delta. It extends between latitudes $5^{\circ} 31' 48''\text{N}$, $5^{\circ} 34' 48''\text{N}$ and longitudes $5^{\circ} 43' 30''\text{E}$, $5^{\circ} 48' 00''\text{E}$. A thick unconsolidated sequence of Miocene to Recent deposits/formations underlies the area. Lithological data from resistivity sounding and wells show that the area consists of predominantly clean sands characterized by large lateral homogeneity [6]. The exposed unit belongs to the Quaternary, while the subsurface litho-stratigraphic unit range from Oligocene. The aquifer is mostly unconfined, mainly of alluvial origin, and characterized by high transverse resistance, thus indicating good aquifer potential [3,6,7].

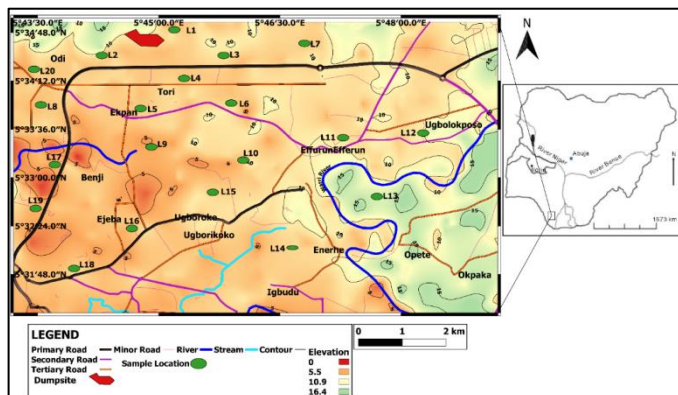


Fig. 1. Map of the study area showing dumpsite and sampling location.

3. Methodology

3.1. Geochemical analysis

Five (5) leachate samples from the dumpsite and groundwater samples from twenty (20) locations within the vicinity of the dumpsite and other control sites were collected and taken to the laboratory for analysis. Physico-chemical parameters such as pH, Electrical Conductivity (EC), and Total Dissolved Solids (TDS) were analyzed using a pH meter, and conductivity/TDS meter. Biochemical Oxygen Demand (BOD₅), Chemical Oxygen Demand (COD), calcium, magnesium, sodium, potassium, nitrates, sulfates, chlorides, chromium, lead, copper, zinc, and iron were analyzed using applicable flame photometric and atomic absorption spectrometric methods in accordance to standard procedure described by American Public Health Association, APHA [8]. Samples for heavy metals were preserved separately by adding 1.0 ml concentrated nitric acid to prevent precipitation of the heavy metals.

3.2. Groundwater modeling

The MODFLOW package was first used to simulate the 3D flow direction of the study area. Warri river in the Southeastern part of the model was used as a river boundary condition; the hydraulic heads simulated from the 1D flow direction of the study area were used as active cells in the model. The top elevation was determined from the Digital Elevation Model (DEM) of the study area, and the lowest elevation was assigned using the drillers' log of the study area. Five (5) layers: topsoil, silty clay, fine sand, fine-medium sand, and medium sand obtained from the driller's log were used while the hydraulic conductivity of these layers was assigned according to Guideal [9].

The recharge rate of the average annual rainfall of the Warri metropolis used by Oyerinde [10] was adopted for this study. A conductance of 700 m²/day was used for the Warri River, and a pumping rate of 313.92 m³/day was used. The Modular 3D Solute Transport Model (MT3D) was used to simulate leachate transport from the dumpsite.

4. Results and Discussion

4.1. Leachate characterization

The results of physical, chemical parameters and heavy metals for leachate collected from the dumpsite are presented in Table 1. The pH value ranged between 7.1 and 7.5, indicating that the leachate was significantly alkaline. Ohwohere-Asuma and Aweto [11] had earlier studied the leachate in this dumpsite and found that the pH ranged from 5.2 to 6.8. Low pH indicates the acidic phase of decomposition, Fatta *et al.* [12] observed that low pH values characterize the initial stage of leachate formation, then later with higher pH values as the dumpsite gets older. Electrical conductivity (EC) and Total dissolved solids (TDS) values range are 786.1 – 4152.4 $\mu\text{s}/\text{cm}$. and 527.8 – 2984 mg/L respectively.

The high concentration of EC and TDS can be attributed to high levels of various cations and anions. The concentration of Biochemical oxygen demand (BOD₅) and Chemical oxygen demand (COD) in the leachate ranged between 38 – 203 mg/L and 301 – 1885 mg/L, respectively. The result of this study was at variance with those observed in the work of Ohwohere-Asuma and Aweto [11] which revealed that BODs and COD in leachate ranged between 65 – 1040 mg/L and 200 – 10, 200 mg/L respectively. The lower values in this study may be due to the dumpsite's aging and leachate maturing. The BOD₅/COD ratio varied from 0.096 – 0.107 is characteristic of an old dumpsite. Aziz [13], Oketola, and Akpotu [14] observed that the BOD₅/COD ratio decreases as the age of the dumpsite increase. For young dumpsites, the ratio is about 0.5, while in old and matured dumpsites, the ratio is generally less than 0.1.

The concentration of major cations was: 16.8 – 36 mg/L (Ca), 9.5 – 30.2 mg/L (Mg), 335.6 – 1966.2 mg/L (Na), 80.6 – 317.4 mg/L (K) respectively while anions concentration was 666 – 1017 mg/L (NO₃), 195.1 – 810.4 mg/L (SO₄), 310 – 1021.3 mg/L (Cl). The concentration of heavy metals in leachate showed the Cr content ranged from 0.25 – 0.61 mg/L, while Pb and Cu had values of 0.01 – 0.32 mg/L and 1.95 – 5.02 mg/L, respectively; Zn had concentration values of 0.15 – 0.63 mg/L, the concentration of Fe ranged from 4.9 – 12.7 mg/L. The values obtained for heavy metals were inconsistent with those of Ohwohere-Asuma and Aweto [11]. The relatively lower values of heavy metals in this study could be due to the age of the dumpsite. Aziz [13] has shown that leachate from a dumpsite older than 10 years has lower heavy metal contents.

Table 1. Summary statistics of physicochemical parameters and heavy metals in leachate and groundwater.

	Leachate			Groundwater		
	Minimum	Maximum	Mean + SD	Minimum	Maximum	Mean + SD
P ^H	7.1	7.3	7.7 ± 0.72	5.8	8.3	6.7 ± 1.37
EC	786.1	4152.4	2728.8 ± 1301.6	22.1	1059.0	422.44 ± 226.77
TDS	527.8	2984.0	2415.2 ± 846.57	17.4	642.5	230.7 ± 239.6
BOD ₅	38.0	203.0	106.5 ± 77.48	7.1	19.5	12.33 ± 6.50
COD	301.0	1885.0	1101.75 ± 654.6	BDL	40.1	20.37 ± 15.04
Ca	16.8	36.0	26.70 ± 8.22	12.0	30.4	19.48 ± 6.92
Mg	9.5	30.2	21.14 ± 8.63	0.18	21.9	10.02 ± 9.27
Na	335.6	1966.2	863.85 ± 757.7	10.4	180.3	75.94 ± 67.09
K	80.6	317.4	227.25 ± 125.03	0.08	26.5	13.12 ± 10.13
NO ₃	666.0	1017.0	720.25 ± 288.94	BDL	61.6	25.08 ± 22.74
SO ₄	195.1	810.4	498.13 ± 279.33	1.40	28.7	11.18 ± 9.80
Cl	310.0	1021.3	657.33 ± 303.16	6.2	228.5	109.02 ± 86.50
Cr	0.25	0.61	0.41 ± 0.15	BDL	0.075	0.022 ± 0.033
Pb	0.01	0.32	0.19 ± 0.14	0.001	0.035	0.014 ± 0.016
Cu	1.95	5.02	2.84 ± 1.46	0.01	0.15	0.048 ± 0.07
Zn	0.15	0.643	0.36 ± 0.25	0.001	0.80	0.014 ± 0.33
Fe	4.90	12.7	7.8 ± 3.4	BDL	1.7	0.23 ± 0.58

BDL – Below detection limits

4.2. Groundwater characterization

The results of physiochemical and heavy metals analyses of groundwater samples are given in Table 2. The result shows that the groundwater quality is significantly different from leachate.

The pH values of 5.8 – 8.3 were within the permissible limit acceptable for drinking water except from the location (pH 5.6). EC values ranged between 22.1 – 1059 $\mu\text{S}/\text{cm}$, and TDS ranged between 17.4 – 642.5 mg/L. These values were below the permissible limits for drinking water except at location 2 (L2), which was within proximity of the dumpsite. Musa and Akuh [15] observed high TDS in groundwater from wells and boreholes closely located to dumpsites. The concentrations of BOD₅ and COD were 7.1 – 19.5 mg/L and 0.00 – 40.1 mg/L, respectively. The low values of COD indicate that there is adequate oxygen available in the groundwater system, depletion of oxygen is usually due to microbial activities.

The concentrations of Ca, Mg, Na, and K were; 1.2 – 30.4 mg/L, 0.18 – 21.9 mg/L, 10.4 – 180.3 mg/L, and 0.08 – 26.5 mg/L. The concentration of NO₃, SO₄, and Cl was 0.00 – 11.6 mg/L, 1.4 – 28.7 mg/L, and 6.2 – 228.5 mg/L. The concentration of heavy metals in the ground shows significant variation from those of leachate. Heavy metal concentrations were: Cr (0.00 – 0.075 mg/L), Pb (0.001 – 0.035 mg/L), Cu (0.01 – 0.15 mg/L), Zn (0.001 – 0.81 mg/L) and Fe (0.00 – 1.7 mg/L). Most of the parameters in groundwater are within the maximum acceptable limits specified by World Health Organization (WHO) [16]. TDS (642.5 mg/L), EC (1059 $\mu\text{S}/\text{cm}$), NO₃ (61.6 mg/L), Cl (228.5 mg/L), Pb (0.035 mg/L) and Fe (1.7 mg/L) were generally higher in localities around the dumpsite; this may be due to the impact of the dumpsite. High Fe content was observed at other locations (L7, L 14, L 16, and L18) far from the dumpsite. Akpoborie and Aweto [17] have established the occurrence of Fe in shallow groundwater in the Quaternary Sombreiro – Warri Deltaic Plain. Another source of Fe contamination is geologic. According to Etu-Efeotor [18], the source of Fe in groundwater can be related to the leaching of Fe²⁺ into the groundwater from iron-rich minerals such as hematite, limonite, and goethite that are abundant within sediments of the Sombreiro – Warri Deltaic Plain that underlie the study area

4.3. Groundwater flow direction

The hydraulic head of the study area ranged from 2.5 – 8.5 m (Figs. 2 and 3). From the Figs, the groundwater flow is towards the southwestern part of the study area. A similar flow direction was observed in the Niger Delta at Agbarho by Efobo *et al.* [19]. However, in areas around the Warri River, the groundwater flows toward the river; which indicates interaction between the river and the groundwater in the study area. When the top layer is recharged, water infiltrates the top layer into the bottom layers as shown in Fig. 3.

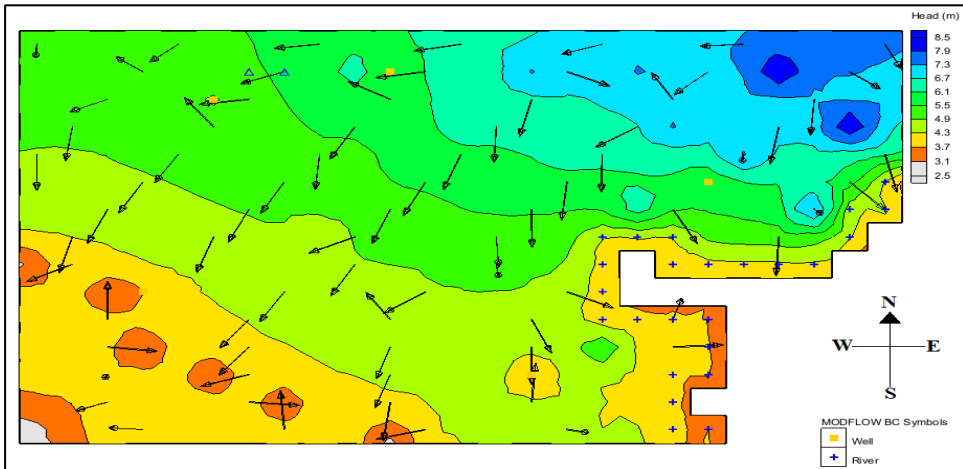


Fig. 2. Groundwater flow direction (plane view).

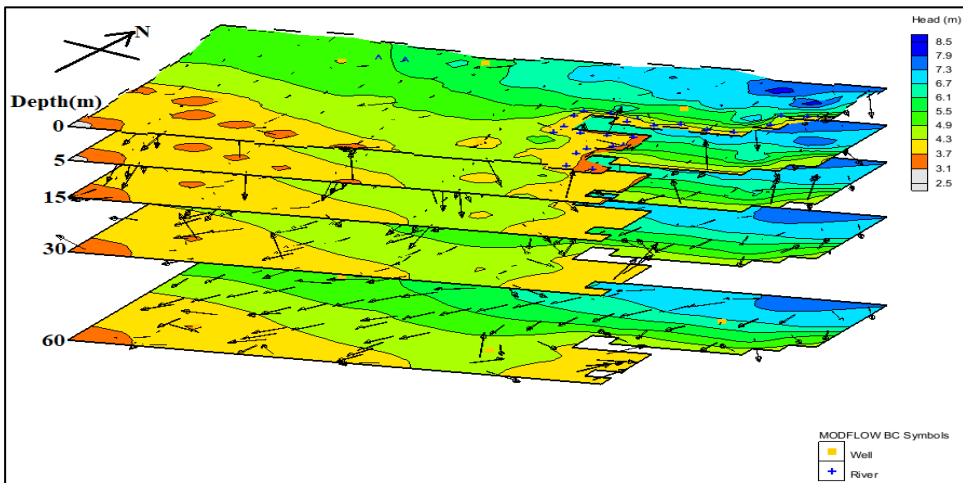


Fig. 3. Groundwater flow direction (Oblique view).

4.4. *Contaminant model*

MT3D package was used to model the dumpsite in the study area using an average concentration of Na (863.85 mg/L) and TDS (2415.2 mg/L) in leachate. The spread and concentration of Na and TDS are shown in Figs 4 and 5. The leachate is introduced at the top layer and infiltrates through the subsequent layers downward; the leachate is attenuated as it infiltrates through the layers, causing a decrease in concentration as the leachate moves downward. For Na, the initial concentration decreased to 70 mg/L, and for TDS, it decreased to between 100 – 700 mg/L at depths of 30 m. Some factors tend to affect the spread of the leachate when introduced at the top layer, such as; recharge rate, pumping rate, dispersion, and advection. The higher the rainfall in the area, the more

water will recharge the aquifer; hence, more water will aid infiltration from top to bottom. Advection describes the spread of the contaminants in the same direction and velocity as the groundwater flow in an area. The spread of the leachate from the model is mainly in the same direction as the groundwater flow; therefore, more mass of the plume is encountered in that direction. Dispersion describes the transport of the contaminants from areas of high concentration to low concentration at rates greater or less than the average linear velocity of groundwater flow; therefore, the solutes disperse along the flow direction, and their concentrations diminish. The groundwater extraction rate in the study area also affects the spread of the contaminants; with much abstraction, less concentration will be found in the aquifer because much of the contaminants have been extracted through pumping. According to National Research Council (NRC) [20], attenuation of the solute is affected by the type of geologic material in an area and the process is much more effective parallel to the plume than transverse. The area under study is underlain mainly by silty-clay, fine to medium sand; these geologic materials are the best in terms of attenuation of contaminants. Thus, at greater distances from the plume, the effect of the contaminants diminishes greatly.

Natural attenuation is the process that naturally transforms contaminants into less harmful forms [21]. Local hydrogeological conditions, time frame for infiltration of leachate, and size and heterogeneity of dumpsite have been suggested by Christensen *et al.* [22] and Bjerg *et al.* [23] as critical for natural attenuation.

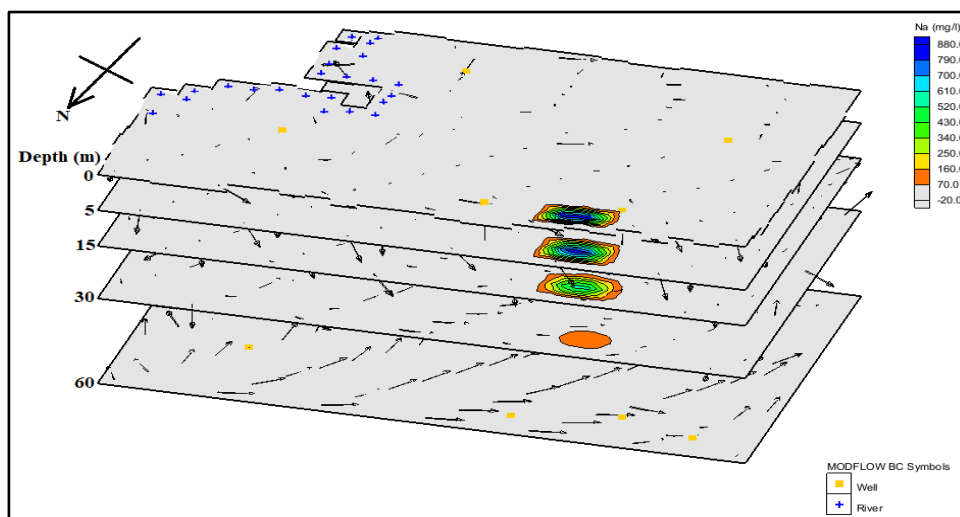


Fig. 4. Spread of Na in contaminant plume with depth.

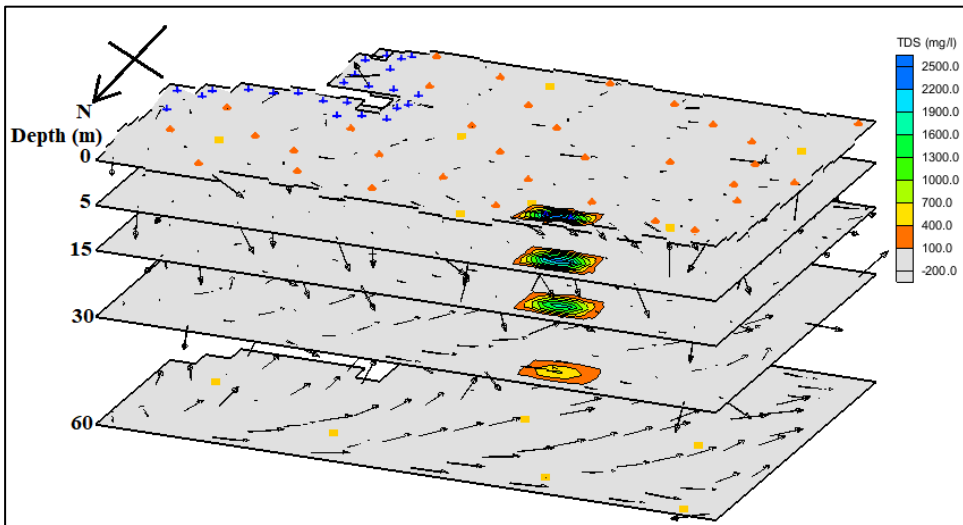


Fig. 5. Spread of TDS in contaminant plume with depth.

4.5. Ionic concentration within dumpsite with time

The graphs in Fig. 6 show the concentration of Na and TDS in the top layer of the dumpsite over 10 years. For Na, the concentration increases from zero (0) to 480 mg/L within a year, and further increases to about 850 mg/L within $2\frac{3}{26}$ years, and after 10 years, it will be 1500 mg/L. For TDS, the concentration increases from zero (0) to 1400 mg/L within a year, further increases to 2400 mg/L within $2\frac{3}{26}$ years, and after 10 years, the concentration will be 3700 mg/L. It then implies that the concentration of leachates increases as more waste is disposed of at the dumpsite and with time.

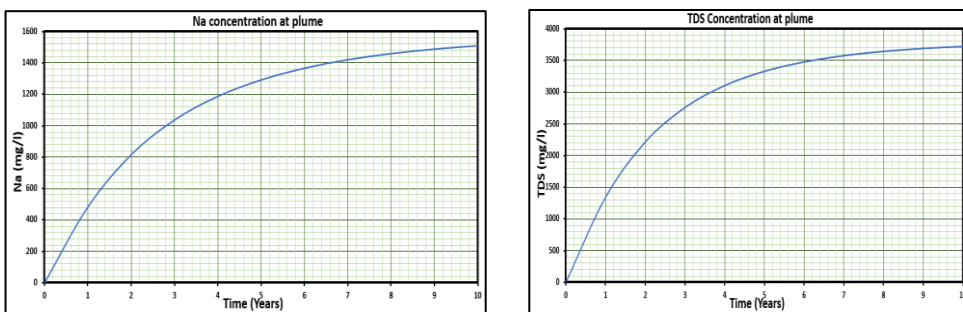


Fig. 6. Ionic concentration within the top layer of dumpsite with time.

4.5.1. Spread of solute within the aquifer

The spread of the contaminant plume within the aquifer is mainly affected by the groundwater flow in the study area. The graphs in Fig. 7 show the spread of Na in

different directions away from the plume; the concentration of Na in the plume within the aquifer directly beneath the dumpsite increases from 0 – 143 mg/L. This then spreads towards the southern and western directions with an increase in ionic concentration; at 125 m south and west of the dumpsite, the concentration of Na increases from 0 – 3.15 mg/L and 0 – 13.8 mg/L, respectively. At 180 m southwest of the dumpsite, the concentration of Na increases from 0 – 6.5 mg/L. At 125 m east and north of the dumpsite, there is no spread of leachate; hence, the concentration of Na decreases. The southwestern part of the study, which is the direction of the groundwater flow, is at more risk.

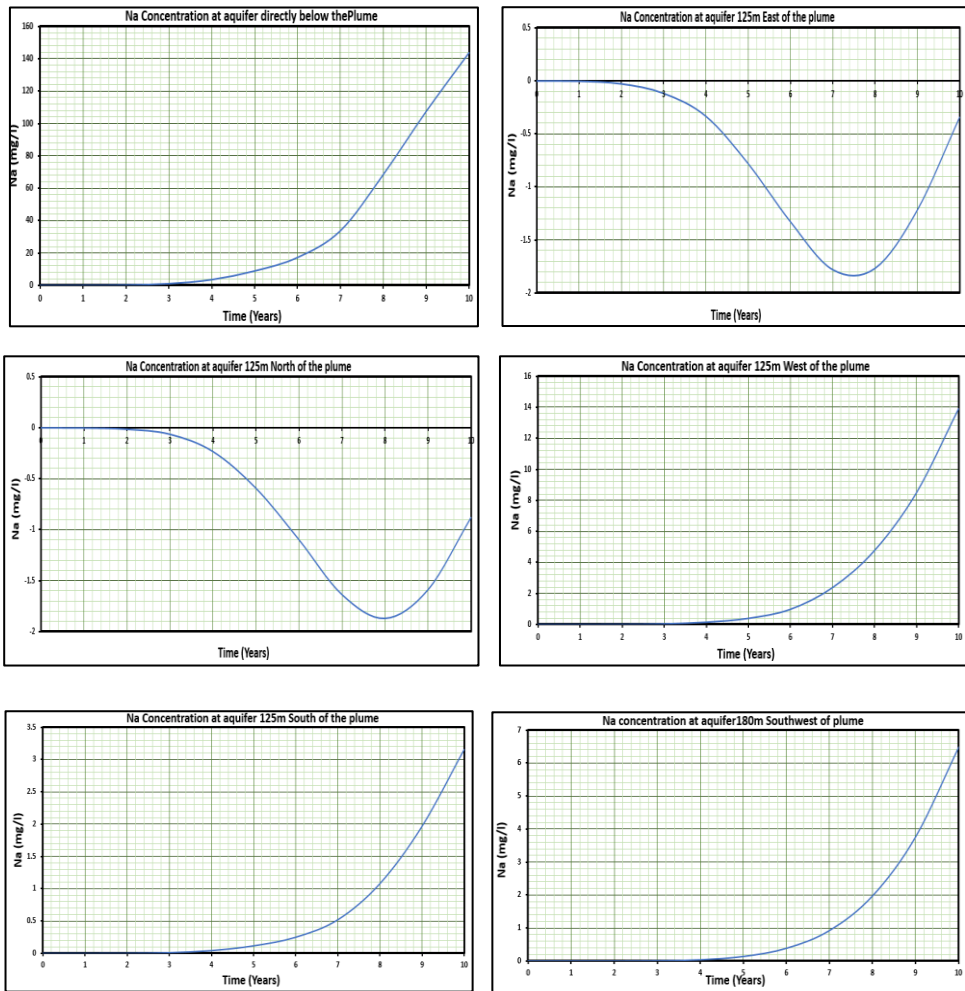


Fig. 7. Spread of Na within the aquifer.

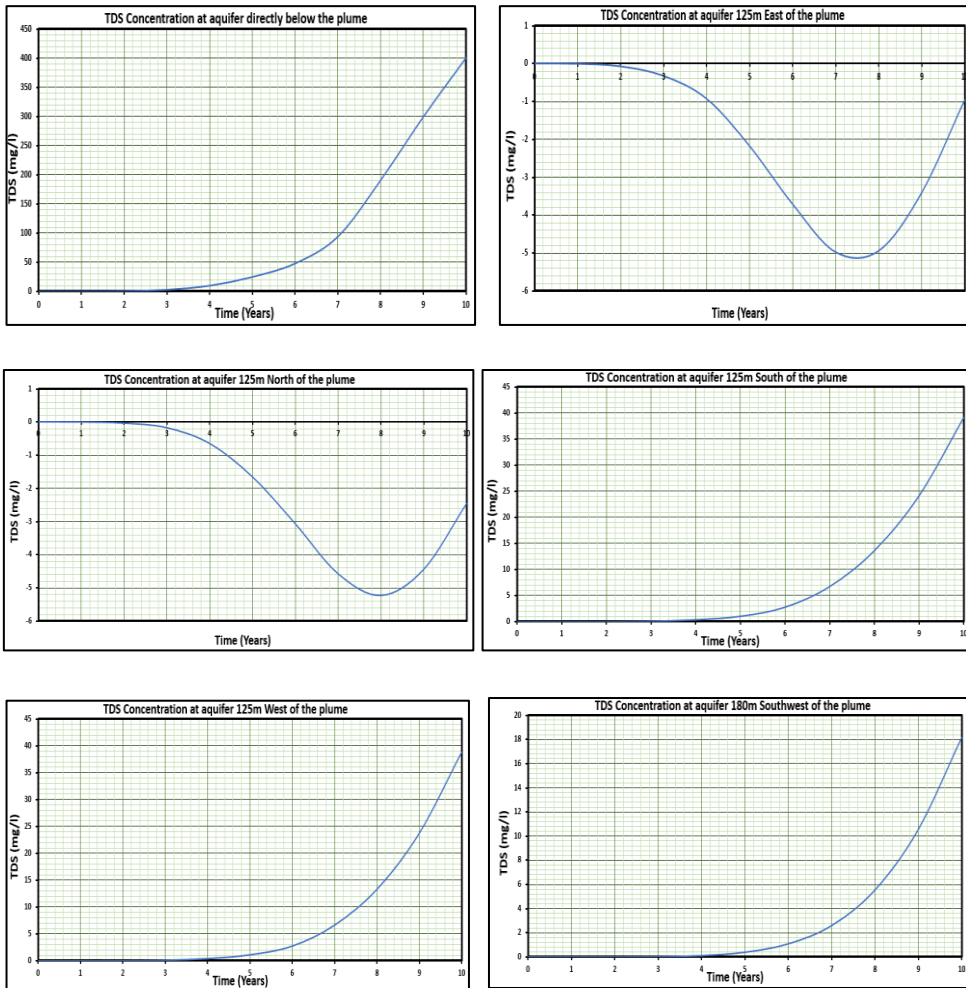


Fig. 8. Spread of TDS within the aquifer.

The graphs in Fig. 8 show TDS in different directions and distances away from the plume. From the graphs, the concentration in the aquifer directly below the dumpsite increases from 0 – 400 mg/L. At 125 m south of the dumpsite, it increases from 0 – 38 mg/L, while at 125 m west of the dumpsite, it increases from 0 – 37.5 mg/L. At 125 m north and east of the dumpsite, TDS decreases. At the 180 m southwestern part of the dumpsite, TDS increased from 0 – 18 mg/L. The spread pattern of Na and TDS confirms that the southwestern direction is the groundwater flow direction. Therefore, 200 m north and east of the plume and 400 m south and west of the dumpsite, the aquifer is safe from the contaminant effect of the dumpsite.

5. Conclusion

The models have been able to show the spread of contaminant plumes at a dumpsite in the Ekpan-Warri area using chemical parameters obtained from the leachate at the dumpsite. The result indicates local enrichment of significant cations, anions, and heavy metals around the dumpsite. Groundwater in the vicinity of the dumpsite has been affected negatively by leachate produced in the dumpsite; groundwater samples had elevated concentrations of TDS, EC, NO₃, Cl, Pb, and Fe; in contrast to groundwater sampled far from the dumpsite. The plume spreads towards the southwestern part, which is also the groundwater flow direction in the study area. Therefore, the spread of leachates from the dumpsite is mainly influenced by groundwater flow in the study area. The dominant permeable sand and heavy precipitation within the study area aided the introduction of pollutants into the groundwater system. If not suitably managed, leachate from the dumpsite can adversely affect the nearby groundwater. Hence, for sustainable environmental management and groundwater protection, there is an urgent need for the following remedial initiatives; the dumpsite should be located far from built-up areas to reduce the pollution of water resources or modification of the dumpsite to well-engineered dumpsite with clay or plastic liners to preclude infiltration of leachate into the groundwater zone.

References

1. A. O. Akankpo and M. U. Igboekwe, *Arch. Appl. Sci. Res.* **4**, 3 (2012).
2. S. Kanmani and R. Gandhimathi, *Appl. Water Sci.* **3**, 1 (2013).
<https://doi.org/10.1007/s13201-012-0072-z>
3. K. E. Aweto and I. A. Akpoborie, *British J. Appl. Sci. Technol.* **6**, 5 (2015).
<https://doi.org/10.9734/BJAST/2015/14541>
4. K. E. Aweto, *J. Sci. Res.* **12**, 3 (2020). <http://dx.doi.org/10.3329/jsr.v12i3.44136>
5. K. E. Aweto - *Proc. 4th Delta State University Faculty of Science Int. Conf.* (Abraka, 2017) pp. 110-115.
6. K. E. Aweto, *Nigerian J. Pure Appl. Sci.* **32**, 2 (2019).
7. O. Ohwoghre-Asuma, K. E. Aweto, and F. C. Ugbe, *Hydrol.* **6**, 31 (2019).
<https://doi.org/10.3390/hydrology6020031>
8. American Public Health Association (APHA), *Standard Methods for the Examination of Water and Waste Water*, 24th Ed. (Washington, D. C., 2011).
9. R. Guideal, A. E. Bala, and A. E. Ikpokonte, *J. Appl. Sci.* **11** (2011).
<https://doi.org/10.3923/jas.2011.542.548>
10. O. O. Oyerinde, *Euro. J. Environment Earth Sci.* **2**, 2 (2021).
<https://doi.org/10.24018/ejgeo.2021.2.2.122>
11. O. Ohwoghre-Asuma and K. E. Aweto, *J. Environ. Earth Sci.* **3**, 9 (2013).
12. D. Fatta, A. Papadopoulos, and M. Loizidou, *Environ. Geochem. Health* **21**, 2 (1999).
<https://doi.org/10.1023/A:1006613530137>
13. S. Q. Aziz - *Proc. 16th Int. Conf. on Petroleum, Mineral Resources and Development* (Cairo, 2013).
14. A. A. Oketola and S. O. Akpotu, *Chem. Ecol.* **31**, 2 (2014).
<https://doi.org/10.1080/02757540.2014.907280>
15. O. K. Musa, B. O. Adama, and J. B. Ahmed II, *Water Res.* **31** (2021).

16. World Health Organization (WHO), Guideline for Drinking Water Quality, 4th Edition (Geneva, 2017) pp. 631.
17. A. I. Akpoborie and K. E. Aweto, *J. Environ. Hydrol.* **20**, 6 (2012).
18. J. O. Etu-Efeotor, *J. Min. Geo.* **18**, 1 (1981).
19. O. Efobo, F. C. Ugbe, and I. A. Akpoborie, *J. Sci. Res.* **12**, 14 (2020).
<https://doi.org/10.3329/jsr.v12i4.45187>
20. K. E. Aweto and L. I. Mamah, *Int. J. Environ. Prot.* **4**, 3 (2014).
21. National Research Council (NRC), *Natural Attenuation for Groundwater Remediation* (National Academic Press, 2000) pp. 274.
22. T. H. Christensen, P. Kjeldsen, H. -J. Albrechtsen, G. Heron, P. H. Nielsen, P. L. Bjerg, and P. E. C. Holm, *Rev. Environ. Sci. Technol.* **24**, 2 (1994).
<https://doi.org/10.1080/10643389409388463>
23. P. L. Bjerg, N. Tuxen, L. A. Reitzel, H. -J. Albrechtsen, and P. Kjeldsen, *Ground Water* **49** (2011).