

Levels, Distribution, Sources, Ecological and Health risk assessment of polyaromatic hydrocarbons in surface and groundwater within and around Gosa dumpsite, Abuja, Nigeria

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Abstract: Polycyclic aromatic hydrocarbons (PAHs) are persistent and potentially toxic contaminants that may migrate from municipal solid-waste dumpsites into surrounding aquatic environments. This study assessed the levels, distribution, possible sources, ecological implications and human health risks of PAHs in surface water, lake water and groundwater within and around Gosa dumpsite, Abuja, Nigeria. Water samples were collected from eight stream locations (SSW1–SSW8), four lake locations (LW1–LW4), one shallow hand-dug well (SHDW) and three boreholes (BW1–BW3). Sixteen priority PAHs were extracted and quantified using gas chromatography–mass spectrometry, while physicochemical parameters, diagnostic molecular ratios, toxic-equivalent quantities (TEQs), Pearson correlation and human-health risk indices were evaluated. Total PAH concentrations in stream water ranged from 0.220 to 1.090 µg/mL, while lake water ranged from 0.380 to 1.330 µg/mL; concentrations of 1.360, 1.360, 0.330 and below detection limit were reported for SHDW, BW1, BW2 and BW3, respectively. The highest reported total TEQ was 0.023 at SSW7, followed by 0.0069 at SSW5. Total adult carcinogenic risks reached 0.014 at SSW7, 0.0054 at BW2, 0.0040 at SSW5 and 0.0019 at SSW1, while corresponding child risks were 0.0043, 0.0017, 0.0012 and 0.00062, respectively. Significant correlations occurred between naphthalene and anthracene ($r = 0.486$), acenaphthene and fluorene ($r = 0.499$), and fluoranthene and benzo[a]anthracene ($r = -0.490$; $p < 0.05$). The findings demonstrate

spatially heterogeneous PAH contamination and indicate potential exposure concerns for aquatic resources surrounding Gosa dumpsite. Regular monitoring, improved leachate and runoff management, and protection of nearby water resources are therefore warranted.

Keywords: Polycyclic aromatic hydrocarbons (PAHs); Gosa Dumpsite; groundwater; surface water; ecological risk assessment; source apportionment.

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1.0 Introduction

Water is an essential natural resource that supports human survival, biodiversity, and

ecosystem functioning. Surface-water bodies, including rivers, streams, and lakes, provide water for domestic, agricultural, industrial, and recreational activities and serve as important habitats for aquatic organisms (Dudgeon *et al.*, 2006). Groundwater is likewise a major component of the hydrological cycle and an important source of water for domestic consumption and irrigation, while also contributing to the maintenance of ecological systems (Carlo *et al.*, 2023; Gebreslassie *et al.*, 2025; Insaf *et al.*, 2004). Despite their importance, increasing anthropogenic activities have resulted in the deterioration of surface- and groundwater quality. Surface water is particularly susceptible to contamination through runoff, atmospheric deposition, and wastewater discharges, whereas groundwater may be contaminated through the infiltration and percolation of polluted surface water and leachate through the subsurface (Babuji *et al.*, 2023; Han *et al.*, 2022; Laskotra *et al.*, 2026; Singh *et al.*, 2005). Activities such as industrial production, municipal waste disposal, mining, petroleum-related activities, agricultural practices, and atmospheric emissions can introduce persistent contaminants into aquatic environments, thereby posing risks to aquatic ecosystems and human populations. Consequently, continuous assessment of water quality and identification of contaminant sources are essential for effective environmental management and protection of public health.

Among the contaminants of environmental concern, polycyclic aromatic hydrocarbons (PAHs) are particularly important because of their persistence, widespread occurrence, toxicity, mutagenicity, and carcinogenic potential. PAHs are a group of organic compounds consisting of two or more fused aromatic rings and are formed predominantly during the incomplete combustion of organic materials (Kim *et al.*, 2013). Anthropogenic sources include the combustion of fossil fuels,

biomass, wood, municipal solid waste, and agricultural residues, as well as vehicular emissions, residential heating, and industrial activities (Abdel-Shafy & Mansour, 2016; Mojiri *et al.*, 2019; Ravindra *et al.*, 2008). Natural processes, including forest fires and volcanic activity, can also contribute to environmental PAH burdens. Their physicochemical properties influence their environmental transport and persistence; low-molecular-weight PAHs are generally more volatile and may occur predominantly in the gaseous phase, whereas high-molecular-weight PAHs are more strongly associated with atmospheric particulate matter (Abdel-Shafy & Mansour, 2016).

The environmental significance of PAHs is further related to their adverse biological effects. Exposure to PAHs has been associated with skin and eye irritation, inflammation, nausea, genetic alterations, developmental abnormalities, cardiovascular effects, and cancer (Abdel-Shafy & Mansour, 2016; Mallah *et al.*, 2022). In recognition of their persistence and potential adverse effects, the United States Environmental Protection Agency (USEPA) identified 16 PAHs as priority pollutants: anthracene (Ant), naphthalene (NaP), benzo[a]pyrene (BaP), pyrene (Pyr), phenanthrene (Phe), acenaphthylene (Acy), benzo[ghi]perylene (B[ghi]P), acenaphthene (Ace), benzo[b]fluoranthene (BbF), benzo[k]fluoranthene (BkF), fluorene (Fle), fluoranthene (Flt), chrysene (Chr), indeno[1,2,3-cd]pyrene (Ind), benz[a]anthracene (BaA), and dibenz[a,h]anthracene (DBaA) (Hussar *et al.*, 2012). Seven of these compounds—benzo[a]pyrene, benzo[b]fluoranthene, benzo[k]fluoranthene, chrysene, indeno[1,2,3-cd]pyrene, benz[a]anthracene, and dibenz[a,h]anthracene—are classified as probable or known human carcinogens (USEPA, 1993). Their occurrence in environmental media is therefore of concern,



particularly where contaminated water resources are used for domestic or agricultural purposes.

Municipal solid-waste dumpsites are important potential sources of PAHs because waste combustion and the decomposition of heterogeneous waste can release or mobilize these compounds into surrounding environmental compartments. Open burning of municipal waste can generate substantial quantities of PAHs, which may subsequently be deposited on surrounding soils and other environmental surfaces (Adeyi & Oyeleke, 2017; Aralu *et al.*, 2022; Kelle *et al.*, 2026; Li *et al.*, 2001; Okechukwu *et al.*, 2021; Palo *et al.*, 1996; Parra *et al.*, 2020; Zhao *et al.*, 2008). Following deposition, PAHs can be retained in soils because of their hydrophobic characteristics and affinity for soil organic matter, which contains functional groups capable of interacting with organic contaminants (Kong *et al.*, 2021; Kuppusamy *et al.*, 2017; Lu *et al.*, 2011). Soil therefore serves as an important environmental reservoir for PAHs. However, retention in soil does not permanently isolate these compounds from aquatic systems. Rainfall and surface runoff can mobilize contaminated soil particles and dissolved PAHs, while leachate generated during waste decomposition can facilitate their downward migration through the subsurface and potentially contaminate groundwater (Aralu *et al.*, 2022; Balmer *et al.*, 2019; Hubai *et al.*, 2025). PAHs transported into aquatic systems may subsequently partition between the dissolved phase, suspended particles, and sediments, thereby contributing to persistent ecological exposure.

Previous investigations have documented the occurrence of PAHs in environmental media associated with municipal waste disposal and have identified dumpsites as potential sources of aquatic contamination (Aralu *et al.*, 2023; Okechukwu *et al.*, 2021). However, the occurrence and behaviour of PAHs in water

surrounding dumpsites can vary according to waste composition, combustion practices, hydrological conditions, soil characteristics, leachate generation, and contaminant transport pathways. Consequently, assessment of PAHs in both surface water and groundwater is important because these compartments can be affected through different but interconnected pathways. Surface waters may receive direct inputs through runoff and atmospheric deposition, whereas groundwater may be affected by infiltration and leachate percolation. An integrated assessment can therefore provide a more comprehensive understanding of contaminant distribution and potential exposure pathways than assessment of either water compartment alone.

The Gosa dumpsite in Abuja, Nigeria, provides an important setting for such an assessment. The facility is a major municipal solid-waste disposal site serving the Federal Capital Territory and receives heterogeneous waste generated from residential areas, commercial establishments, markets, healthcare facilities, and other urban activities. Reported waste streams include food waste, plastics, paper, metals, glass, textiles, rubber, and other municipal materials (Dodo & Ashigwuike, 2003; Seidu *et al.*, 2021). The management of municipal waste at the facility includes open burning, which provides a potential pathway for the formation and atmospheric release of PAHs. Combustion of plastics, hospital waste, and other organic and combustible materials is recognized as an important source of PAHs (Li *et al.*, 2001; Palo *et al.*, 1996; Zhao *et al.*, 2008). PAHs released during combustion may subsequently be deposited on surrounding soils, while rainfall can facilitate their transfer through runoff. In addition, leachate generated within the dumpsite may provide a pathway for contaminant migration into underlying groundwater.

The potential implications are particularly important for communities surrounding the



Gosa dumpsite that depend on groundwater from boreholes and hand-dug wells for domestic water supply. Streams and other surface-water resources within and around the dumpsite may also be used for irrigation and, in some cases, domestic purposes by nearby residents and informal waste workers. These uses can create potential exposure pathways where water resources are contaminated by PAHs originating from dumpsite activities. Therefore, characterization of PAH concentrations and distributions, coupled with source identification and risk assessment, is necessary to determine the environmental significance of contamination in the affected water resources.

Despite the potential importance of the Gosa dumpsite as a source of environmental contamination, studies comprehensively investigating PAHs in both surface water and groundwater within and around the dumpsite remain limited. In particular, there is insufficient information on the combined occurrence and distribution of the USEPA priority PAHs, their likely sources, and their potential ecological and human-health implications in the affected water resources. This represents an important knowledge gap because the simultaneous consideration of surface-water and groundwater contamination can help distinguish contaminant transport pathways associated with runoff and leachate migration. Furthermore, integrating concentration data with source assessment and ecological and human-health risk evaluations provides a more complete basis for determining the environmental significance of PAH contamination at the site. Addressing these gaps is necessary to improve understanding of dumpsite-related PAH contamination and to provide evidence for appropriate environmental monitoring and management in Abuja.

Therefore, this study aimed to determine the concentrations and distribution of the 16

USEPA priority PAHs in surface water and groundwater within and around the Gosa dumpsite, assess their potential ecological and human-health risks, and identify their likely sources. The study also examined the physicochemical characteristics of the water resources in relation to PAH occurrence and distribution, thereby providing additional information on environmental factors that may influence the behaviour of these contaminants. The significance of the study lies in its potential to provide an integrated assessment of PAH contamination in water resources affected by municipal solid-waste disposal activities in the Gosa dumpsite environment. The findings can provide baseline information for environmental monitoring, support the identification of potentially important contamination pathways, and contribute to improved management of dumpsite-related pollution. The results may also be useful to environmental regulatory and waste-management authorities in developing appropriate pollution-control and monitoring strategies. More broadly, the study provides scientific information relevant to the protection of surface-water and groundwater resources used by surrounding communities and contributes to the growing body of evidence on PAH contamination associated with municipal solid-waste disposal in Nigeria.

2.0 Materials and Methods

2.1 Materials

Dichloromethane (GC grade), acetone (AR/ACS grade), and sodium sulphate (AR/ACS grade) were purchased from LOBA Chemie Laboratory and Fine Chemicals, Palghar, India. A 2 mg/mL PAH standard mixture containing the 16 USEPA priority PAHs—naphthalene, acenaphthylene, acenaphthene, fluorene, anthracene, phenanthrene, fluoranthene, pyrene, benzo[a]anthracene, chrysene, benzo[b]fluoranthene, benzo[k]fluoranthene, benzo[a]pyrene, dibenz[a,h]anthracene, indeno[1,2,3-cd]pyrene, and



benzo[ghi]perylene—was obtained from AccuStandard Inc., New Haven, USA. An internal standard mixture (4 mg/mL) containing naphthalene- d_{10} , acenaphthene- d_{10} , phenanthrene- d_{10} , chrysene- d_{12} , and perylene- d_{12} was also obtained from AccuStandard Inc., New Haven, USA..

2.2 Description of the Study Area

Gosa Dumpsite is an open dumpsite with a perimeter of approximately 4,000 m. The dumpsite is approximately square-shaped and fenced and is located along Idu Train Station Road, Abuja, within Jiwa Ward of the Abuja Municipal Area Council (AMAC). The site lies between the coordinates N09° 01.270' E007° 19.590' and N09° 01.173' E007° 20.510'.

The study area experiences two distinct climatic seasons: a rainy season, generally occurring from April to October, and a dry season, typically extending from November to March. Annual precipitation ranges from approximately 1,100 to 1,600 mm, while mean monthly temperatures generally range from 25 to 30 °C. The vegetation is predominantly Guinea savannah, with patches of forest vegetation. These climatic and ecological characteristics can influence surface runoff, groundwater recharge, soil–water interactions, and the transport and dilution of contaminants within and around the dumpsite.

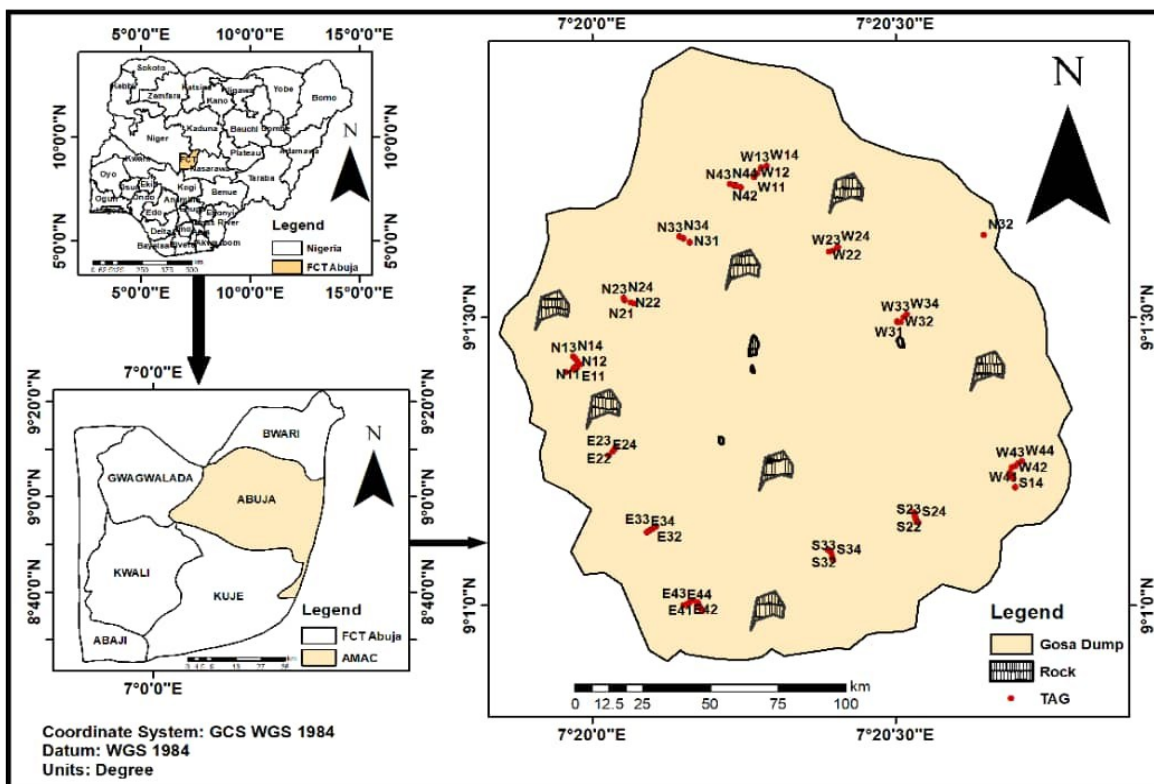


Figure: 1 Map of Gosa dumpsite (Source Omada *et al*, 2024)

2.3 Sample Collection

Surface- and groundwater samples were collected in duplicate into 1-L amber glass bottles in February 2025. Surface-water samples were collected from eight sampling points along the stream within the dumpsite

(SSW1–SSW8) and four sampling points from a lake located approximately 1 m outside the dumpsite fence (SL1–SL4). Groundwater samples were collected from one shallow hand-dug well within the dumpsite (SHDW) and three boreholes (BW1–BW3) located



approximately 10 m outside the dumpsite fence. Thus, a total of 16 sampling points were sampled, yielding 32 water samples.

The amber glass bottles were pre-cleaned with hot detergent solution, rinsed with distilled water, allowed to drain, rinsed again with acetone, and capped to prevent dust contamination prior to use (USEPA, 1984). Surface-water samples were collected without disturbing the streambed or lake sediments. Samples from the shallow hand-dug well were collected by gently lowering a stainless-steel bucket attached to a rope into the well, carefully retrieving the bucket, and transferring the water into the sample container using a stainless-steel scoop.

Borehole-water samples were collected from water flowing slowly from the tap after cleaning the tap and purging the borehole for approximately 15 min, corresponding to approximately four times the internal volume of the borehole. Dissolved oxygen (DO), pH, electrical conductivity (EC), total dissolved solids (TDS), and temperature were measured in situ, whereas the other parameters were determined in the laboratory. All samples were properly labelled, placed in an ice box, transported to the laboratory, and stored in a refrigerator maintained at 4 °C. Extraction of PAHs from the water samples commenced the following day and continued for six days.

2.4 Preparation of Calibration Standards and Calibration Curve

The target PAHs were quantified using an internal-standard calibration approach in accordance with USEPA Method 8270E (SW-846) (USEPA, 2001). A five-point calibration curve was prepared using PAH standard solutions at concentrations ranging from 0.001 to 10 µg/mL. The calibration solutions were prepared from a 2 mg/mL (2,000 µg/mL) stock solution containing the 16 USEPA priority PAHs.

For each calibration level, 1 mL of the standard solution was spiked with 10 µL of a 100 µg/mL

internal-standard mixture containing naphthalene-d₁₀, acenaphthene-d₁₀, phenanthrene-d₁₀, chrysene-d₁₂, and perylene-d₁₂. The solutions were thoroughly mixed, transferred into 2-mL autosampler vials, and analysed using gas chromatography–mass spectrometry (GC–MS). An Agilent 6890 gas chromatograph coupled to a 5973N mass spectrometer (Agilent Technologies, Palo Alto, CA, USA) was used for the analysis. Instrument operation, chromatographic data acquisition, and quantitative processing were performed using Agilent ChemStation software.

2.5 Extraction and Analysis of PAHs in Water Samples

PAHs were extracted from the water samples by liquid–liquid extraction (LLE) and quantified using the internal-standard method. Briefly, 1 L of each water sample was spiked with 10 µL of a 100 µg/mL internal-standard mixture prepared from a 4,000 µg/mL stock solution containing naphthalene-d₈, acenaphthene-d₁₀, phenanthrene-d₁₀, chrysene-d₁₂, and perylene-d₁₂. The samples were vortex-mixed for 6 min.

Following USEPA Method 610 (USEPA, 1984), 60 mL of dichloromethane (DCM) was added to each sample in an amber glass container and shaken for 40 s before being transferred to a separatory funnel. The funnel was vigorously shaken for 10 min with periodic venting and allowed to stand for 15 min to facilitate phase separation. The DCM layer was collected into a 300-mL Erlenmeyer flask, while the aqueous phase was retained for subsequent extraction. The extraction was repeated twice using fresh 60-mL portions of DCM.

The combined extracts were dried over 5 g of anhydrous sodium sulphate and concentrated to 1 mL using a rotary evaporator (RE-52A, Searchtech). The concentrated extracts were transferred to 2-mL GC–MS vials and analysed using an Agilent 6890 gas chromatograph



coupled to a 5973N mass spectrometer (Agilent Technologies, Palo Alto, CA, USA). Instrument control and data processing were performed using Agilent ChemStation software.

2.5.1 GC–MS Operating Conditions

For GC–MS analysis, 1 μL of each extract was injected in splitless mode at an inlet temperature of 250 $^{\circ}\text{C}$. Chromatographic separation was performed using a DB-5ms capillary column (30 m $\times$ 0.25 mm i.d., 0.25 μm film thickness). The oven temperature was initially maintained at 70 $^{\circ}\text{C}$ for 2 min, increased to 180 $^{\circ}\text{C}$ at 35 $^{\circ}\text{C}/\text{min}$ and held for 3 min, and subsequently increased to 300 $^{\circ}\text{C}$ at 12 $^{\circ}\text{C}/\text{min}$ and held for 8 min.

Helium was used as the carrier gas at a constant flow rate of 1.0 mL/min. The mass spectrometer operated in electron ionization mode at 70 eV, with ion-source, transfer-line, and quadrupole temperatures of 230, 280, and 150 $^{\circ}\text{C}$, respectively. Data were acquired in scan mode over an m/z range of 50–550, with a threshold of 150, sampling rate of 2, solvent delay of 3.5 min, and gain factor of 1.0.

2.5.2 Reagent Blanks

Reagent blanks were processed concurrently with the samples using the same extraction, concentration, and analytical procedures. PAH concentrations in all reagent blanks were below the respective detection limits.

2.6 Method Validation

The analytical method was validated for linearity, limit of detection (LOD), limit of quantification (LOQ), accuracy, and precision. Calibration linearity for each PAH was assessed using the coefficient of determination (R^2), which ranged from 0.995 to 0.998. Method sensitivity was evaluated using LOD and LOQ, calculated from the calibration-curve parameters as. $LOD = \frac{3,3\sigma}{S}$ and $LOG = \frac{10\sigma}{S}$.

σ = deviation of the response and S = slope of the calibration curve.

en independent reagent blanks were analysed to estimate background variability. The LODs ranged from 0.042 to 0.098 $\mu\text{g}/\text{mL}$, while the LOQs ranged from 0.06 to 0.27 $\mu\text{g}/\text{mL}$.

Accuracy was evaluated using extraction-recovery experiments. Briefly, 100 mL of water collected from a relatively undisturbed area was fortified with 40 μL of a 10 $\mu\text{g}/\text{mL}$ PAH standard mixture and 20 μL of a 100 $\mu\text{g}/\text{mL}$ internal-standard mixture, while another 100 mL of water was spiked with only 20 μL of a 100 $\mu\text{g}/\text{mL}$ internal-standard mixture. The fortified samples were thoroughly mixed and subjected to the extraction and drying procedures described above. The extracts were analysed by GC–MS under identical conditions. PAH recoveries ranged from 92% to 101%.

Precision was evaluated as intra-day and inter-day precision and expressed as relative standard deviation (%RSD). Intra-day precision was determined from three replicate analyses performed on the same day, whereas inter-day precision was assessed over three consecutive days. Intra-day and inter-day precision ranged from 0.76–4.13% and 2.20–3.75% RSD, respectively. Reagent blanks processed through the complete analytical procedure consistently showed PAH concentrations below the respective detection limits, indicating negligible analytical contamination.

2.7 Determination of Physicochemical Parameters of Surface Water and Groundwater

The physicochemical parameters analysed included pH, temperature, total dissolved solids (TDS), electrical conductivity (EC), turbidity, dissolved oxygen (DO), biochemical oxygen demand (BOD_5), and the concentrations of chloride (Cl^-), fluoride (F^-), nitrate (NO_3^-), sulphate (SO_4^{2-}), phosphate (PO_4^{3-}), carbonate (CO_3^{2-}), and bicarbonate (HCO_3^-). All parameters were determined in



triplicate using the analytical procedures described below.

2.7.1 pH, Electrical Conductivity, TDS, Temperature and Turbidity

The pH, electrical conductivity (EC), total dissolved solids (TDS), temperature, and turbidity of the water samples were measured in situ during sampling. pH, EC, TDS, and temperature were measured using a portable pH/EC/TDS/temperature meter (HI9813-61, Hanna Instruments, USA).

Before measurement, the probe was rinsed with distilled water and gently dried with tissue paper. The pH meter was calibrated using standard buffer solutions at pH 4.00, 7.00, and 10.00. The probe was then immersed in the water sample, and the pH and temperature readings were recorded. EC and TDS were subsequently measured using the corresponding instrument modes.

Turbidity was determined using a microprocessor turbidity meter (HI93703, Hanna Instruments, USA), calibrated according to the manufacturer's instructions using appropriate calibration standards. Prior to measurement, the sample vials were rinsed with the corresponding water samples, filled carefully to avoid air bubbles, and inserted into the turbidity meter. Turbidity measurements were then recorded in accordance with the manufacturer's operating procedure.

2.7.2 DO and BOD

Dissolved oxygen (DO) was measured using a DO meter (HI9146, Hanna Instruments, USA). The instrument was calibrated according to the manufacturer's instructions using a zero-oxygen solution (HI7040L, Hanna Instruments, USA) and water-saturated air. For measurement, 30 mL of each water sample was transferred into a 50-mL beaker, and the probe was immersed in the sample until a stable reading was obtained.

Biochemical oxygen demand (BOD₅) was determined according to Standard Method 5210B (AWWA, WEF, APHA, 1998). Prior to analysis, the pH, temperature, and DO of the samples were

measured. Where necessary, the pH and temperature were adjusted to approximately neutral pH and 20 °C, respectively. Samples with DO concentrations below saturation were gently aerated by shaking.

Dilution water was prepared using KH₂PO₄, K₂HPO₄, Na₂HPO₄·7H₂O, NH₄Cl, MgSO₄·7H₂O, CaCl₂, and FeCl₃·6H₂O in accordance with Standard Method 5210B. An aliquot of 15 mL of each sample was transferred into a 300-mL BOD bottle and diluted with 285 mL of dilution water. The initial DO was measured immediately after preparation. The bottles were carefully stoppered without entrapping air bubbles and incubated in the dark at 20 °C for 5 days. Following incubation, the final DO was measured, and BOD₅ was calculated according to equation 1

$$\text{BOD}_5 \text{ (mg/L)} = \frac{\text{DO}_i - \text{DO}_f}{P} \quad (1)$$

where DO_i is the initial dissolved oxygen concentration (mg/L), DO_f is the dissolved oxygen concentration after 5 days of incubation (mg/L), and P is the decimal volumetric fraction of the sample used.

2.7.3 Determination of Anions

Chloride, fluoride, nitrate, sulphate, phosphate, carbonate, and bicarbonate concentrations were determined using argentometric, SPADNS colorimetric, spectrophotometric, turbidimetric, molybdenum blue, and titrimetric methods, respectively, following AWWA, WEF, and APHA (1998).

2.8 Ecological Risk Assessment of PAHs in Surface Water and Groundwater

Polycyclic aromatic hydrocarbons can bind to sediments in freshwater environments and gradually accumulate over time (Duran & Cravo-Laureau, 2016). Owing to their high lipophilicity, PAHs can also bioaccumulate in aquatic organisms, including fish, shellfish, shrimp, crayfish, crabs, and aquatic plants (Liu *et al.*, 2025; Nwaichi *et al.*, 2016; Xie *et al.*, 2023). Consequently, PAHs may enter the human food chain and contribute to human exposure. PAHs may also associate with soil organic matter and suspended fine particles or



sediments in groundwater environments (Hu *et al.*, 2025). Where contaminated groundwater is used for irrigation, PAHs may affect soil microbial communities, potentially influencing plant growth (Chen *et al.*, 2024; Picariello *et al.*, 2020).

The ecological risk assessment model developed by Kalf *et al.* (1997) and subsequently improved by Cao *et al.* (2010) was used to evaluate the ecological risks posed by PAHs in the study area. The risk quotient (RQ) was evaluated for individual PAHs and for the combined PAH concentrations. The RQ for an individual PAH was calculated according to equation 2

$$RQ = \frac{C_{PAHs}}{C_{QV}} \quad (2)$$

where C_{PAHs} is the concentration of PAHs in surface water, and C_{QV} is the concentration of the quality value of PAHs in water. The quality values include negligible concentrations (NCs) and maximum permissible concentrations (MPCs). Maximum permissible concentrations (MPCs) represent pollutant levels above which the risk of adverse effects is deemed unacceptable, whereas negligible

concentrations (NCs) denote pollutant levels below which the likelihood of adverse effects is considered insignificant (Crommentuijn *et al.*, 2000). The negligible concentrations (NCs) and maximum permissible concentrations (MPCs) of PAHs in water, sediment, and soil reported by Cao *et al.* (2010) were used for the assessment

$$RQ_{NCs} = \frac{C_{PAHs}}{C_{QV(NCs)}} \quad (3)$$

$$RQ_{MPCs} = \frac{C_{PAHs}}{C_{QV(MPCs)}} \quad (4)$$

$C_{QV(NCs)}$ and $C_{QV(MPCs)}$ represent the quality values for NCs and MPCs in water, respectively.

$RQ_{NCs} < 1.0$ indicates that individual PAHs may be of negligible concern, whereas $RQ_{MPCs} \geq 1$ signifies that individual PAHs pose a high risk. Values of $RQ_{NCs} \geq 1.0$ and $RQ_{MPCs} < 1.0$ suggest that individual PAHs present moderate contamination and risk.

To estimate the risk posed by the combined PAHs, the RQ_{NCs} and RQ_{MPCs} values of each PAH not less than one were summed and calculated according to the following equation:

$$\sum_{i=1}^{16} RQ_{i(NCs)} = \sum_{i=1}^{16} RQ_{i(MPCs)} \quad (5)$$

$$RQ_{i(NCs)} \geq 1$$

$$\sum_{i=1}^{16} RQ_{i(MPCs)} = \sum_{i=1}^{16} RQ_{i(MPCs)} \quad (6)$$

$$\text{where } RQ_{i(MPCs)} \geq 1$$

$\sum_{i=1}^{16} RQ_{i(NCs)} \geq 800$ and $\sum_{i=1}^{16} RQ_{i(MPCs)} = 0$ indicates that the PAHs are of moderate risk,

$\sum_{i=1}^{16} RQ_{i(NCs)} < 800$ and $\sum_{i=1}^{16} RQ_{i(MPCs)} \geq 1$ shows that the risk associated with the PAHs is moderate,

$\sum_{i=1}^{16} RQ_{i(NCs)} \geq 800$ and $\sum_{i=1}^{16} RQ_{i(MPCs)} \geq 1$ signifies that the combination of the PAHs is a high risk.

2.9 Human Health Risk Assessment



Potential human-health risks associated with PAH exposure through contaminated water were evaluated as carcinogenic and non-carcinogenic risks. The USEPA (2015a, 2015b) and Agency for Toxic Substances and Disease Registry (ATSDR, 2023, 2025) risk-assessment approaches were used. Human exposure to contaminants in water can occur through ingestion and dermal contact, including exposure during bathing, swimming, washing, and other activities involving contaminated water.

2.9.1 Non-Carcinogenic Risk Assessment

Non-carcinogenic risk was evaluated using the hazard quotient (HQ) and hazard index (HI) (ATSDR, 2025). The HQ estimates the potential for adverse non-carcinogenic effects associated with exposure to an individual contaminant, whereas the HI represents the combined risk from multiple contaminants or exposure pathways (ATSDR, 2023, 2025; USEPA, 2015a, 2015b).

The HQ was calculated as the ratio of the exposure dose to the corresponding reference dose. Average daily dose (ADD) or chronic daily intake (CDI) was estimated for ingestion and dermal exposure pathways.

The chronic daily intake through ingestion and dermal contact was calculated according to equation 7

$$CDI_{ing} = \frac{C \times IR \times ED \times EF}{BW \times AT} \quad (7)$$

$$CDI_{der} = \frac{C \times ET \times Kp \times SA \times ED \times EF \times CF}{BW \times AT} \quad (8)$$

C is concentration (mg/L) of PAHs in water sample, IR is daily ingestion rate of water (L/day) -2.5 L/day for adult and 0.78 L/day for children, ED is exposure duration (yr) – 26 and 6 yr for adult and children respectively, EF is exposure frequency (days/yr) – 350 yr for both adult and children, BW is body weight (kg) 80 kg for adult and 15 children, AT is average time (non- carcinogens) (days) – 9,490 (26 x 365) for adult and 2,190 (6 x 365) for children, ET_w is exposure time (hrs/event) – 0.71 and 0.54 for adult and children respectively, Kp is dermal

permeability of coefficient of PAHs in water (cm/hr) -1.2, SA is skin surface area (cm^2), - 19,652 cm^2 for adult and 6, 365 cm^2 for children, CF is the conversion factor ($1L/1000_{cm^3}$) (55)

The hazard quotient for ingestion and dermal pathway (HQ_{ing} and HQ_{der}) and hazard index of the PAHs in water was computed using equation 9 -11 (ATSDR, 2025; Adeyinka *et al.*, 2024).

$$HQ_{ing} = \frac{CDI_{ing}}{RfD} \quad (9)$$

$$HQ_{der} = \frac{CDI_{der}}{RfD} \quad (10)$$

$$HI = \Sigma HQ = HQ_{ing} + HQ_{der} \quad (11)$$

$HQ < 1$ indicates probability of non-cancer health related illness as a result of exposure to pollutants. $HQ > 1$ implies likelihood of non-cancer health issues, $HI < 1$ suggest that the pollutants do not have the potential to cause non-cancer adverse health, $HI > 1$ infers that the pollutants have the potential to cause non-carcinogenic health effects.

2.9.2 Carcinogenic Risk Assessment

Carcinogenic risk was evaluated for the seven carcinogenic PAHs: benzo[a]pyrene (BaP), benzo[b]fluoranthene (BbF), benzo[k]-fluoranthene (BkF), chrysene (Chr), indeno[1,2,3-cd]pyrene (Ind), benz[a]-anthracene (BaA), and dibenz[a,h]anthracene (DBahA).

The concentrations of the carcinogenic PAHs were converted to benzo[a]pyrene equivalents (BaPeq) by multiplying the concentration of each PAH by its corresponding toxic equivalency factor (TEF) to obtain toxic equivalent quantities (TEQs) (Pan *et al.*, 2015; USEPA, 2002). The total TEQ was subsequently used to estimate the cancer risk associated with exposure to the mixture of carcinogenic PAHs.

The TEFs used for BaP, BbF, BkF, Chr, Ind, BaA, and DBahA were 1.0, 0.1, 0.01, 0.001, 0.1, 0.1, and 1.0, respectively (USEPA, 2014b).



The toxic equivalent concentration was calculated using equation 12

$$\sum BaPeq \text{ or } \sum TEQs = \sum C_i \times TEF_i \quad (12)$$

where C_i is the concentration of specific PAHs and TEFs is the correlative toxic equivalency factor.

Cancer risk (CR) was estimated according to the equation;

$$CR_{ing} = CDI_{ing} \times SF \quad (13)$$

$$CR_{der} = CDI_{der} \times SF \quad (14)$$

$$\text{Total cancer risk} = CR_{ing} + CR_{der} \quad (15)$$

In calculating CDI_{ing} and CDI_{der} (using equation 7 and 8), C is $\sum BaPeq \text{ or } \sum TEQs = \sum C_i \times TEF_i$ (Pan *et al.*, 2015), AT for carcinogen is 25,550 days for both adult and children (USEPA, 2014a). SF (ingestion) is cancer slope factor of BaP 7.3 mg/kg/day (USEPA, 2002), SF (dermal) is cancer slope of BaP via dermal pathway, 7.9 mg/kg/day derived by Pan *et al.* (2015).

2.10 Diagnostic Molecular Ratio Analysis

Diagnostic molecular ratios (DMRs) are commonly used to distinguish potential PAH sources in environmental samples, including surface water and groundwater (Pongpiachan, 2014; Yunker *et al.*, 2002). The ratios evaluated in this study included Ant/(Ant + Phe), BaA/(BaA + Chr), Ind/(Ind + BghiP), Flt/(Flt + Pyr), BaP/BghiP, and LMW/HMW (Thiombane *et al.*, 2019; Zhang *et al.*, 2022).

The ratio Ant/(Ant + Phe) was interpreted as indicating a petroleum source when <0.1 and an incomplete-combustion source when >0.1 (Yunker *et al.*, 2002; Yurdak *et al.*, 2019). For BaA/(BaA + Chr), values <0.2 were considered indicative of petroleum origin, values between 0.2 and 0.35 indicated mixed sources, and values >0.35 suggested pyrogenic sources (Yunker *et al.*, 2002; Zhang *et al.*, 2022).

For Ind/(Ind + BghiP), values <0.2 suggested petroleum origin, values between 0.2 and 0.5 indicated incomplete combustion of petroleum, and values >0.5 were associated with incomplete combustion of coal, wood, and

biomass (Yunker *et al.*, 2002). The Flt/(Flt + Pyr) ratio was interpreted as petrogenic when <0.4 , indicative of incomplete petroleum combustion when between 0.4 and 0.5, and pyrogenic when >0.5 (Yurdak *et al.*, 2019).

The BaP/BghiP ratio was used to differentiate non-traffic and traffic-related combustion sources, with values <0.6 interpreted as typical of non-traffic sources and values >0.6 associated with incomplete combustion of fuels from traffic emissions (Gbeddy *et al.*, 2021). The LMW/HMW ratio was interpreted as <1 for pyrogenic sources associated with incomplete combustion of coal and biomass and >1 for petroleum-related sources (Soclo *et al.*, 2000).

2.11 Pearson Correlation Analysis

Pearson correlation analysis was performed to assess the relationships among PAHs in the surface- and groundwater samples from the study area. Because the dataset was limited, multivariate receptor-modelling techniques such as principal component analysis (PCA) and positive matrix factorization (PMF) were not considered appropriate. Pearson correlation analysis was therefore used to identify potential associations among individual PAHs and provide supporting evidence for possible common sources.

3.0 Results and Discussion

3.1 Concentration and Distribution of PAHs in Surface Water and Groundwater

The concentrations and distribution of the 16 USEPA priority PAHs in surface water and groundwater within and around Gosa Dumpsite are presented in Tables 1a–1c. The results show considerable spatial variation in the occurrence and concentration of individual PAHs among the stream-water sampling points (SSW1–SSW8), lake-water sampling points (SLW1–SLW4), the shallow hand-dug well (SHDW), and boreholes (BW1–BW3). Twelve of the 16 target PAHs were detected across the sampled water bodies, whereas several high-molecular-



weight compounds, particularly the five- and six-ring PAHs, were predominantly below the detection limit. The detected compounds included naphthalene (NaP), acenaphthylene (Acy), acenaphthene (Ace), fluorene (Fle), anthracene (Ant), phenanthrene (Phe), fluoranthene (Flt), pyrene (Pyr), benz[a]anthracene (BaA), chrysene (Chr), benzo[b]fluoranthene (BbF), and benzo[ghi]perylene (BghiP).

In the stream water, the reported total concentration of the 16 PAHs ($\Sigma 16$ PAHs) ranged from 0.220 $\mu\text{g/mL}$ at SSW3 to 1.090 $\mu\text{g/mL}$ at SSW7 (Table 1a). The highest reported concentration occurred at SSW7, followed by SSW5 (1.030 $\mu\text{g/mL}$), SSW2

(0.600 $\mu\text{g/mL}$), SSW1 (0.520 $\mu\text{g/mL}$), SSW6 (0.420 $\mu\text{g/mL}$), SSW4 (0.310 $\mu\text{g/mL}$), SSW8 (0.240 $\mu\text{g/mL}$), and SSW3 (0.220 $\mu\text{g/mL}$). Thus, the approximately five-fold difference between the lowest and highest reported $\Sigma 16$ PAH concentrations indicates marked spatial heterogeneity in PAH contamination along the stream. The elevated concentration at SSW7 suggests that this location may be particularly influenced by local contaminant inputs or favourable conditions for PAH accumulation and transport. However, confirmation of the specific cause requires consideration of the sampling-point characteristics, hydrological conditions, and source-apportionment results.

Table 1a: Concentration ($\mu\text{g/mL}$) of Individual PAHs, $\Sigma 16$ PAHs, $\Sigma 7$ PAHs, ΣLMW PAHs and ΣHMW PAHs in Surface and Borehole Water within and around Gosa Dumpsite, Abuja, FCT, Nigeria — SSW1–SSW8

PAHs	No. of Rings	SSW1	SSW2	SSW3	SSW4	SSW5	SSW6	SSW7	SSW8
NaP	2	BDL	0.13	BDL	BDL	0.10	BDL	0.14	ND
Acy	3	0.03	0.15	BDL	0.20	0.50	0.17	BDL	0.17
Ace	3	0.13	BDL	0.22	BDL	0.22	BDL	0.30	BDL
Fle	3	0.10	0.11	BDL	0.11	BDL	BDL	BDL	BDL
Ant	3	0.12	BDL	BDL	BDL	BDL	0.21	0.15	BDL
Phe	3	BDL	0.09	BDL	BDL	BDL	BDL	0.18	0.07
Flt	4	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Pyr	4	0.11	BDL	BDL	BDL	0.15	0.04	0.11	BDL
BaA	4	0.03	BDL	BDL	BDL	0.06	BDL	0.21	BDL
Chr	4	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
BbF	5	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
BkF	5	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
BaP	5	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
DBahA	5	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Ind	6	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
BghiP	6	BDL	0.12	BDL	BDL	BDL	BDL	BDL	BDL
$\Sigma 16$ PAHs	—	0.520	0.600	0.220	0.310	1.030	0.420	1.090	0.240
$\Sigma 7$ PAHs	—	0.03	BDL	BDL	BDL	0.06	BDL	0.21	BDL
ΣLMW PAHs	—	0.38	0.35	0.22	0.31	0.72	0.38	0.63	0.24
ΣHMW PAHs	—	0.14	0.12	BDL	BDL	0.21	0.04	0.32	BDL
$\Sigma 2$ -Ring PAHs	—	BDL	0.13	BDL	BDL	0.10	BDL	0.14	BDL
$\Sigma 3$ -Ring PAHs	—	0.380	0.350	0.220	0.310	0.720	0.380	0.630	0.240
$\Sigma 4$ -Ring PAHs	—	0.140	BDL	BDL	BDL	0.210	0.04	0.32	BDL



Σ 5-Ring PAHs	—	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Σ 6-Ring PAHs	—	BDL	0.12	BDL	BDL	BDL	BDL	BDL	BDL

Table 1b: Concentration ($\mu\text{g/mL}$) of Individual PAHs, Σ 16 PAHs, Σ 7 PAHs, Σ LMW PAHs and Σ HMW PAHs in Surface and Borehole Water within and around Gosa Dumpsite, Abuja, FCT, Nigeria — SLW1–SLW4

PAHs	No. of Rings	SLW1	SLW2	SLW3	SLW4
NaP	2	0.60	0.20	0.17	0.31
Acy	3	BDL	BDL	BDL	BDL
Ace	3	0.10	0.21	0.16	0.18
Fle	3	BDL	BDL	BDL	BDL
Ant	3	BDL	BDL	BDL	BDL
Phe	3	BDL	BDL	BDL	BDL
Flt	4	0.07	0.90	BDL	0.50
Pyr	4	0.03	0.13	BDL	BDL
BaA	4	BDL	BDL	BDL	BDL
Chr	4	BDL	BDL	BDL	BDL
BbF	5	BDL	BDL	BDL	BDL
BkF	5	BDL	BDL	BDL	BDL
BaP	5	BDL	BDL	BDL	BDL
DBahA	5	BDL	BDL	BDL	BDL
Ind	6	BDL	BDL	BDL	BDL
BghiP	6	BDL	BDL	BDL	BDL
Σ 16 PAHs	—	0.695	1.330	0.380	0.970
Σ 7 PAHs	—	BDL	BDL	BDL	BDL
Σ LMW PAHs	—	BDL	0.10	0.21	0.16
Σ HMW PAHs	—	0.095	1.03	BDL	0.50
Σ 2-Ring PAHs	—	0.60	0.20	0.17	0.31
Σ 3-Ring PAHs	—	BDL	0.10	0.21	0.16
Σ 4-Ring PAHs	—	0.095	1.03	BDL	BDL
Σ 5-Ring PAHs	—	BDL	BDL	BDL	BDL
Σ 6-Ring PAHs	—	BDL	BDL	BDL	BDL

Table 1c: Concentration ($\mu\text{g/mL}$) of Individual PAHs, Σ 16 PAHs, Σ 7 PAHs, Σ LMW PAHs and Σ HMW PAHs in Surface and Borehole Water within and around Gosa Dumpsite, Abuja, FCT, Nigeria — SHDW and BW1–BW3

PAHs	No. of Rings	SHDW	BW1	BW2	BW3
NaP	2	0.13	BDL	0.17	BDL
Acy	3	0.10	0.33	0.25	BDL
Ace	3	BDL	0.43	BDL	—
Fle	3	BDL	0.13	BDL	0.11
Ant	3	BDL	0.23	BDL	BDL
Phe	3	BDL	BDL	0.17	BDL
Flt	4	BDL	BDL	BDL	BDL



Pyr	4	0.31	BDL	BDL	BDL
BaA	4	BDL	BDL	BDL	BDL
Chr	4	0.28	BDL	0.15	BDL
BbF	5	BDL	BDL	0.08	ND
BkF	5	BDL	BDL	BDL	ND
BaP	5	BDL	BDL	BDL	BDL
DBahA	5	BDL	BDL	BDL	BDL
Ind	6	BDL	BDL	BDL	BDL
BghiP	6	BDL	BDL	BDL	BDL
∑16 PAHs	—	1.360	0.330	1.360	BDL
∑7 PAHs	—	0.28	BDL	0.23	BDL
∑LMW PAHs	—	0.64	0.33	0.96	BDL
∑HMW PAHs	—	0.59	BDL	0.23	BDL
∑2-Ring PAHs	—	0.13	BDL	0.17	BDL
∑3-Ring PAHs	—	0.64	0.33	0.96	BDL
∑4-Ring PAHs	—	0.59	BDL	0.15	BDL
∑5-Ring PAHs	—	BDL	BDL	0.08	BDL
∑6-Ring PAHs	—	BDL	BDL	BDL	BDL

Note: BDL = Below Detection Limit; ND = Not Detected. LMW = Low Molecular Weight; HMW = High Molecular Weight. The unit of concentration is $\mu\text{g/mL}$.

The individual PAH results in Table 1a show that low-molecular-weight (LMW) compounds constituted the dominant group in most of the stream-water samples. The reported $\sum\text{LMW}$ PAHs ranged from 0.22 to 0.72 $\mu\text{g/mL}$, with the highest value recorded at SSW5 (0.72 $\mu\text{g/mL}$), followed by SSW7 (0.63 $\mu\text{g/mL}$). In contrast, the reported $\sum\text{HMW}$ PAHs ranged from below detection limit to 0.32 $\mu\text{g/mL}$, with the highest value occurring at SSW7. The predominance of LMW PAHs is particularly evident at SSW1–SSW6 and SSW8, where the concentrations of three-ring compounds contributed substantially to the reported total PAH burden. At SSW5, for example, acenaphthylene (0.50 $\mu\text{g/mL}$) was the most abundant individual PAH, while at SSW7, acenaphthene (0.30 $\mu\text{g/mL}$), anthracene (0.21 $\mu\text{g/mL}$), and benz[a]anthracene (0.21 $\mu\text{g/mL}$) contributed appreciably to the observed contamination.

Naphthalene was detected at SSW2, SSW6, and SSW8 at concentrations of 0.13, 0.10, and 0.14 $\mu\text{g/mL}$, respectively, while it was below detection at the other stream locations. Acenaphthylene occurred at six of the eight

stream locations and ranged from 0.03 to 0.50 $\mu\text{g/mL}$, with the maximum concentration occurring at SSW5. Acenaphthene was detected at SSW1, SSW3, SSW5, and SSW7, with concentrations ranging from 0.13 to 0.30 $\mu\text{g/mL}$. Fluorene was detected only at SSW1, SSW2, and SSW4, each at approximately 0.10–0.11 $\mu\text{g/mL}$. Anthracene was detected at SSW1, SSW6, and SSW7, with concentrations of 0.12, 0.21, and 0.15 $\mu\text{g/mL}$, respectively. Phenanthrene was detected at SSW2, SSW7, and SSW8, at 0.09, 0.18, and 0.07 $\mu\text{g/mL}$, respectively. These variations demonstrate that individual PAHs did not exhibit uniform spatial distributions along the stream.

The four-ring PAHs were less frequently detected than the three-ring compounds. Pyrene was detected at SSW1, SSW5, SSW6, and SSW7, with concentrations ranging from 0.04 to 0.15 $\mu\text{g/mL}$, while benz[a]anthracene was detected at SSW1, SSW5, and SSW7, with concentrations of 0.03, 0.06, and 0.21 $\mu\text{g/mL}$, respectively. Fluoranthene and chrysene were below detection at all stream sampling points. The five-ring PAHs benzo[b]fluoranthene,



benzo[k]fluoranthene, benzo[a]pyrene, and dibenz[a,h]anthracene were also below detection throughout the stream. Benzo[ghi]perylene was detected only at SSW2, at 0.12 $\mu\text{g/mL}$. Thus, the distribution pattern in the stream was characterized by relatively frequent detection of two- and three-ring PAHs, occasional detection of four-ring PAHs, and limited detection of five- and six-ring compounds.

The lake-water samples exhibited a different distribution pattern (Table 1b). The reported $\Sigma 16$ PAH concentrations ranged from 0.380 $\mu\text{g/mL}$ at SLW3 to 1.330 $\mu\text{g/mL}$ at SLW2, while SLW1 and SLW4 recorded 0.695 and 0.970 $\mu\text{g/mL}$, respectively. The highest value at SLW2 indicates a substantial PAH burden at this location. Unlike most of the stream samples, the lake-water results showed appreciable contributions from both LMW and HMW PAHs. The reported ΣLMW concentrations were below detection at SLW1, 0.10 $\mu\text{g/mL}$ at SLW2, 0.21 $\mu\text{g/mL}$ at SLW3, and 0.16 $\mu\text{g/mL}$ at SLW4. In contrast, ΣHMW PAHs were reported as 0.095, 1.03, below detection, and 0.50 $\mu\text{g/mL}$ at SLW1–SLW4, respectively.

Naphthalene was detected at all four lake sampling points, ranging from 0.17 $\mu\text{g/mL}$ at SLW3 to 0.60 $\mu\text{g/mL}$ at SLW1. Acenaphthene was likewise detected throughout the lake, with concentrations ranging from 0.10 to 0.21 $\mu\text{g/mL}$. The remaining three-ring PAHs—acenaphthylene, fluorene, anthracene, and phenanthrene—were below detection at all four lake locations. Fluoranthene was detected at SLW1, SLW2, and SLW4, with concentrations of 0.07, 0.90, and 0.50 $\mu\text{g/mL}$, respectively. The concentration of fluoranthene at SLW2 was particularly notable and represented the highest individual PAH concentration recorded in the lake samples. Pyrene was detected at SLW1 and SLW2 at 0.03 and 0.13 $\mu\text{g/mL}$, respectively, while

benz[a]anthracene, chrysene, and the other five- and six-ring PAHs were below detection. The results for the shallow hand-dug well and boreholes are presented in Table 1c. The shallow hand-dug well (SHDW) contained NaP (0.13 $\mu\text{g/mL}$), Acy (0.10 $\mu\text{g/mL}$), Pyr (0.31 $\mu\text{g/mL}$), and Chr (0.28 $\mu\text{g/mL}$), while the other target PAHs were below detection. The reported $\Sigma 16$ PAH concentration at SHDW was 1.360 $\mu\text{g/mL}$, indicating the presence of multiple PAH components despite the limited number of individually detected compounds. The occurrence of both three-ring and four-ring PAHs in the hand-dug well is important because the well is located within the dumpsite and may therefore be susceptible to contamination through infiltration and leachate migration.

Among the boreholes, BW1 recorded the highest reported $\Sigma 16$ PAH concentration of 1.360 $\mu\text{g/mL}$, whereas BW2 recorded 0.330 $\mu\text{g/mL}$ and BW3 was reported as below detection (Table 1c). At BW1, Acy (0.33 $\mu\text{g/mL}$), Ace (0.43 $\mu\text{g/mL}$), Fle (0.13 $\mu\text{g/mL}$), and Ant (0.23 $\mu\text{g/mL}$) were detected, indicating a predominance of three-ring PAHs. BW2 contained NaP (0.17 $\mu\text{g/mL}$), Acy (0.25 $\mu\text{g/mL}$), Phe (0.17 $\mu\text{g/mL}$), Chr (0.15 $\mu\text{g/mL}$), and BbF (0.08 $\mu\text{g/mL}$). BW3 contained only fluorene at 0.11 $\mu\text{g/mL}$ among the individually reported detected compounds. The marked difference among the three boreholes suggests substantial spatial heterogeneity in groundwater PAH occurrence around the dumpsite.

The molecular-weight distribution further demonstrates differences between the water compartments. In the stream water, the reported ΣLMW PAHs were generally greater than the corresponding ΣHMW PAHs, indicating a predominantly LMW pattern. This pattern was particularly pronounced at SSW5, where ΣLMW PAHs reached 0.72 $\mu\text{g/mL}$ compared with 0.21 $\mu\text{g/mL}$ for ΣHMW PAHs. At SSW7, however, the difference was smaller, with 0.63



$\mu\text{g/mL}$ $\Sigma\text{LMW PAHs}$ and $0.32 \mu\text{g/mL}$ $\Sigma\text{HMW PAHs}$, suggesting a relatively greater contribution of HMW compounds at this location. The lake samples showed a contrasting pattern, particularly at SLW2 and SLW4, where the reported HMW concentrations were substantially greater than the LMW concentrations. This may indicate differences in contaminant inputs, transport, partitioning, or local accumulation processes between the stream and lake environments.

The detection of predominantly LMW PAHs in several samples may be associated with their greater mobility and comparatively higher solubility in water relative to higher-molecular-weight compounds. Conversely, the limited occurrence of many five- and six-ring PAHs may reflect their stronger association with particulate matter and organic phases. Nevertheless, the occurrence of selected HMW compounds, including benz[a]anthracene, chrysene, benzo[b]fluoranthene, and benzo[ghi]perylene, indicates that combustion-derived PAHs are also present in some sampling locations. The simultaneous occurrence of LMW and HMW compounds therefore suggests that the water resources may be receiving contributions from more than one contamination pathway.

Overall, the results in Tables 1a–1c demonstrate that PAH contamination was spatially heterogeneous across the Gosa dumpsite water system. The highest reported $\Sigma 16$ PAH concentrations occurred at SLW2 ($1.330 \mu\text{g/mL}$), SHDW ($1.360 \mu\text{g/mL}$), and BW1 ($1.360 \mu\text{g/mL}$), while the lowest reported values occurred at SSW3 ($0.220 \mu\text{g/mL}$), SSW8 ($0.240 \mu\text{g/mL}$), and SSW4 ($0.310 \mu\text{g/mL}$). The detection of PAHs in the shallow hand-dug well and boreholes, particularly BW1, demonstrates that PAH contamination is not restricted to surface water and may extend into groundwater resources. This observation is environmentally important because groundwater from boreholes and hand-dug

wells is used by surrounding communities for domestic purposes.

The spatial variation observed among sampling points may reflect differences in proximity to waste deposits, runoff pathways, leachate movement, hydrological connectivity, local waste-burning activities, and physicochemical conditions of the receiving waters. However, concentration data alone cannot establish the precise sources of the detected PAHs. Consequently, the diagnostic molecular-ratio analysis and Pearson correlation analysis presented in subsequent sections provide complementary evidence for evaluating whether the observed PAHs are predominantly associated with pyrogenic, petrogenic, or mixed sources.

3.2 Percentage Abundance and Molecular-Weight Distribution of PAHs

The percentage abundance of individual PAHs and PAH groups in the surface-water and groundwater samples is presented in Tables 2a and 2b. The percentage-abundance pattern provides additional information on the relative contribution of individual PAHs to the total PAH burden at each sampling location and complements the concentration data presented in Tables 1a–1c.

3.2.1 Percentage Abundance of PAHs in Stream Water

As shown in Table 2a, the stream-water samples were generally dominated by three-ring PAHs, although the dominant individual compound varied considerably among sampling locations. At SSW1, the detected PAHs were predominantly three-ring compounds, with anthracene accounting for 23.10%, acenaphthene 25.00%, fluorene 19.23%, acenaphthylene 5.76%, and benz[a]anthracene 6.00%, while pyrene contributed 21.15%. The corresponding group distribution showed that three-ring PAHs accounted for 73.07% of the PAH abundance, compared with 26.92% for four-ring PAHs.



This indicates a clear predominance of lower-ring PAHs at SSW1.

At SSW2, acenaphthylene represented 25.00% of the detected PAHs, followed by naphthalene (21.66%), fluorene (18.33%), phenanthrene (15.00%), and benzo[ghi]perylene (20.00%). The group distribution showed that three-ring PAHs accounted for 58.33%, while six-ring PAHs contributed 20.00% and the remaining reported groups contributed smaller proportions. The presence of benzo[ghi]perylene at 20.00% is notable because it indicates that although the overall stream-water contamination was dominated by lower-ring compounds, a relatively high-

molecular-weight PAH also contributed appreciably at this location.

SSW3 exhibited the simplest PAH profile, with acenaphthene accounting for 100.00% of the detected PAH abundance. Correspondingly, three-ring PAHs represented 100.00% of the reported abundance, while the two-, four-, five-, and six-ring groups were not detected.

Similarly, SSW4 showed complete dominance by three-ring PAHs, which accounted for 100.00% of the detected PAH abundance. These patterns indicate that PAH contamination at SSW3 and SSW4 was strongly characterized by low-molecular-weight compound.

Table 2a: Percentage Abundance of Individual PAHs and PAH Groups in Surface Water Samples (SSW1–SSW8) within and around Gosa Dumpsite, Abuja, FCT, Nigeria

PAHs	SSW1	SSW2	SSW3	SSW4	SSW5	SSW6	SSW7	SSW8
NaP	ND	21.66	ND	ND	9.71	ND	12.84	ND
Acy	5.76	25.00	ND	33.33	48.54	40.47	ND	70.83
Ace	25.00	ND	100.00	ND	21.35	ND	27.52	ND
Fle	19.23	18.33	ND	35.48	ND	ND	ND	ND
Ant	23.10	ND	ND	ND	ND	50.00	13.76	ND
Phe	ND	15.00	ND	ND	ND	ND	16.51	29.16
Flt	ND	ND	ND	ND	ND	ND	ND	ND
Pyr	21.15	ND	ND	ND	14.56	9.52	10.09	ND
BaA	6.00	ND	ND	ND	5.82	ND	19.26	ND
Chr	ND	ND	ND	ND	ND	ND	ND	ND
BbF	ND	ND	ND	ND	ND	ND	ND	ND
BkF	ND	ND	ND	ND	ND	ND	ND	ND
BaP	ND	ND	ND	ND	ND	ND	ND	ND
DBahA	ND	ND	ND	ND	ND	ND	ND	ND
Ind	ND	ND	ND	ND	ND	ND	ND	ND
BghiP	ND	20.00	ND	ND	ND	ND	ND	ND
∑7 PAHs	5.76	ND	ND	ND	5.82	ND	19.26	ND
∑LMW PAHs	73.07	68.33	100.00	100.00	69.90	100.00	90.47	22.01
∑HMW PAHs	26.92	20.00	ND	ND	20.38	9.52	29.35	ND
∑2-Ring PAHs	ND	22.16	ND	ND	9.70	ND	12.84	ND
∑3-Ring PAHs	73.07	58.33	100.00	100.00	69.90	90.47	57.79	100.00
∑4-Ring PAHs	26.92	ND	ND	ND	20.38	9.52	29.35	ND
∑5-Ring PAHs	ND	ND	ND	ND	ND	ND	ND	ND
∑6-Ring PAHs	ND	20.00	ND	ND	ND	ND	ND	ND



Table 2b: Percentage Abundance of Individual PAHs and PAH Groups in Surface and Borehole Water Samples (SL1–SL4, SHDW and BW1–BW3) within and around Gosa Dumpsite, Abuja, FCT, Nigeria

PAHs	SL1	SL2	SL3	SL4	SHDW	BW1	BW2	BW3
NaP	86.33	15.03	44.73	31.95	9.55	ND	7.35	ND
Acy	ND	ND	ND	ND	7.35	100.00	18.38	ND
Ace	ND	7.51	55.26	16.49	13.23	ND	0.43	ND
Fle	ND	ND	ND	ND	9.55	ND	31.61	ND
Ant	ND	ND	ND	ND	ND	16.91	ND	ND
Phe	ND	ND	ND	ND	ND	ND	12.50	ND
Flt	ND	10.07	67.66	ND	51.54	ND	ND	ND
Pyr	2.87	9.77	ND	ND	22.79	ND	ND	ND
BaA	ND	ND	ND	ND	ND	ND	ND	ND
Chr	ND	ND	ND	ND	20.58	ND	11.02	ND
BbF	ND	ND	ND	ND	ND	ND	5.88	ND
BkF	ND	ND	ND	ND	ND	ND	ND	ND
BaP	ND	ND	ND	ND	ND	ND	ND	ND
DBaH	ND	ND	ND	ND	ND	ND	ND	ND
Ind	ND	ND	ND	ND	ND	ND	ND	ND
BghiP	ND	ND	ND	ND	ND	ND	ND	ND
∑7 PAHs	ND	ND	ND	ND	20.58	ND	16.91	ND
∑LMW PAHs	ND	7.51	55.26	16.49	47.05	100.00	70.58	ND
∑HMW PAHs	13.66	77.44	ND	51.54	43.38	ND	14.70	ND
∑2-Ring PAHs	86.33	15.03	44.73	31.95	9.55	ND	7.35	ND
∑3-Ring PAHs	ND	7.51	55.26	16.49	47.05	100.00	70.58	ND
∑4-Ring PAHs	13.66	77.44	ND	51.54	43.38	ND	11.02	ND
∑5-Ring PAHs	ND	ND	ND	ND	ND	ND	5.88	ND
∑6-Ring PAHs	ND	ND	ND	ND	ND	ND	ND	ND

Note: ND = Not Detected; LMW = Low Molecular Weight; HMW = High Molecular Weight.

At SSW5, acenaphthylene was the most abundant individual PAH, contributing 48.54%, followed by acenaphthene (21.35%), pyrene (14.56%), naphthalene (9.71%), and benz[a]anthracene (5.82%). The three-ring group accounted for 69.90%, whereas four-ring PAHs accounted for 20.38%. Thus, despite the presence of four-ring compounds, three-ring PAHs remained the dominant group at this location.

SSW6 was also dominated by three-ring compounds. Anthracene contributed 50.00%, while naphthalene and pyrene contributed 9.71% and 9.52%, respectively. The group-level results indicate that three-ring PAHs accounted for 90.47% of the detected abundance, with four-ring PAHs contributing

9.52%. This strong dominance of three-ring compounds corresponds with the relatively limited occurrence of higher-ring PAHs at this sampling point.

At SSW7, acenaphthylene represented 70.83% of the detected PAHs, making it the most dominant individual compound among all the stream-water samples. Benz[a]anthracene contributed 19.26%, while anthracene accounted for 13.76%. At the group level, three-ring PAHs represented 57.79%, while four-ring PAHs accounted for 29.35%. The relatively substantial contribution of four-ring PAHs at SSW7 is consistent with the comparatively elevated total PAH concentration recorded at this location in Table 1a.



SSW8 was characterized by phenanthrene as the dominant detected compound, contributing 29.16%, followed by acenaphthylene at 70.83% according to the individual-abundance values presented in Table 2a. The reported group distribution indicates that three-ring PAHs accounted for 100.00% of the detected abundance. This again demonstrates the predominance of lower-ring PAHs in the stream-water system.

Overall, the results in Table 2a show that three-ring PAHs constituted the principal PAH group in most stream-water samples, accounting for 57.79–100.00% of the reported abundance. Four-ring PAHs were the second most important group at several locations, contributing up to 29.35% at SSW7. Five-ring PAHs were not detected in any of the stream samples, whereas six-ring PAHs were detected only at SSW2, where they contributed 20.00%. The dominance of three-ring compounds suggests that the PAH profile of the stream was generally characterized by relatively lower-molecular-weight compounds, although the contribution of four- and six-ring PAHs at selected locations demonstrates spatial variation in PAH composition.

3.2.2 Percentage Abundance of PAHs in Lake Water, Shallow Hand-Dug Well and Boreholes

The percentage abundance of PAHs in the lake water, shallow hand-dug well, and boreholes is presented in Table 2b. In contrast to the relatively consistent dominance of three-ring PAHs in the stream, these samples exhibited considerably greater variability in PAH composition.

At SL1, naphthalene dominated the PAH profile, accounting for 86.33% of the reported abundance, while pyrene contributed 2.87%. The group distribution showed that two-ring PAHs accounted for 86.33%, whereas four-ring PAHs contributed 13.66%. This indicates that SL1 was strongly dominated by the low-

molecular-weight, two-ring compound naphthalene.

SL2 displayed a markedly different composition. Naphthalene accounted for 15.03%, acenaphthene for 7.51%, fluoranthene for 10.07%, and pyrene for 9.77%, while the reported four-ring group accounted for 77.44% and the HMW group accounted for 77.44%. Thus, SL2 was characterized by a substantially greater relative contribution from higher-ring PAHs than SL1. This observation is particularly relevant because SL2 also recorded the highest reported $\Sigma 16$ PAH concentration among the lake samples in Table 1b.

At SL3, naphthalene and acenaphthene were the major detected compounds, contributing 44.73% and 55.26%, respectively. The entire detected PAH profile was therefore associated with lower-molecular-weight compounds, with three-ring PAHs accounting for 55.26% and two-ring PAHs accounting for 44.73%. No four-, five-, or six-ring PAHs were detected at this location.

SL4 showed a mixed profile dominated by naphthalene (31.95%), acenaphthene (16.49%), and fluoranthene (67.66% as reported in Table 2b). The reported group distribution indicates that four-ring PAHs accounted for 51.54%, while three-ring PAHs contributed 16.49%. This suggests a substantial contribution of higher-ring compounds at SL4 relative to SL1 and SL3.

The shallow hand-dug well (SHDW) also exhibited a mixed PAH profile. Fluoranthene was reported to account for 51.54%, followed by pyrene (22.79%), chrysene (20.58%), naphthalene (9.55%), acenaphthylene (7.35%), acenaphthene (13.23%), and fluorene (9.55%). At the group level, three-ring PAHs accounted for 47.05%, four-ring PAHs for 43.38%, while the remaining proportion was associated with two-ring PAHs. The relatively high contribution of four-ring compounds indicates a greater relative contribution of higher-molecular-weight PAHs in the hand-dug well



compared with most of the stream-water samples. BW1 showed a markedly different pattern, with acenaphthylene accounting for 100.00% of the reported PAH abundance. The three-ring group likewise accounted for 100.00% of the reported abundance, indicating that the detected PAH profile in this borehole was entirely dominated by a three-ring compound.

BW2 exhibited a more diverse PAH profile. Fluorene accounted for 31.61%, acenaphthylene 18.38%, acenaphthene 0.43%, anthracene 16.91%, phenanthrene 12.50%, benzo[b]fluoranthene 5.88%, and chrysene 11.02%. The group distribution showed that three-ring PAHs accounted for 70.58%, while four-ring PAHs contributed 11.02% and five-ring PAHs 5.88%. This indicates that lower-molecular-weight PAHs were dominant, although higher-ring compounds were also present.

BW3 showed no detected PAH groups according to the percentage-abundance table. This agrees with the reported below-detection-limit $\Sigma 16$ PAH concentration in Table 1c and indicates that detectable PAH contamination was not observed in this borehole under the analytical conditions employed.

3.2.3 Comparative Distribution of PAH Groups

The percentage-abundance data in Tables 2a and 2b demonstrate clear differences in PAH composition among the investigated water compartments. Stream water was generally dominated by three-ring PAHs, with the three-ring group accounting for 57.79–100.00% of the reported abundance. Lake water displayed greater compositional heterogeneity, with some locations dominated by two-ring PAHs, particularly SL1, while others showed substantial contributions from four-ring compounds, particularly SL2 and SL4.

The shallow hand-dug well also showed a relatively mixed profile, with three- and four-ring PAHs accounting for substantial

proportions of the detected contaminants. The boreholes showed considerable spatial variability: BW1 was dominated by a three-ring compound, BW2 contained a mixture of three-, four-, and five-ring PAHs, while BW3 showed no detectable PAHs. This spatial variability indicates that groundwater contamination was not uniform around the dumpsite.

The predominance of three-ring PAHs in many samples is consistent with the concentration patterns observed in Tables 1a–1c and suggests that lower-molecular-weight compounds made an important contribution to the observed PAH burden. However, the occurrence of four-, five-, and six-ring PAHs at selected locations indicates that the contamination cannot be characterized exclusively by low-molecular-weight compounds. The presence of higher-ring compounds is particularly important because several high-molecular-weight PAHs are associated with combustion processes and include compounds of toxicological concern.

The contrasting percentage distributions between sampling points may reflect differences in contaminant sources and environmental transport processes. PAHs entering surface water through runoff may undergo dilution, volatilization, sorption to suspended particles, and sedimentation, while groundwater contamination may be influenced by leachate infiltration and subsurface transport. Consequently, the relative abundance of individual PAHs can vary considerably even within the same dumpsite environment.

The results also provide preliminary evidence that PAH contamination at Gosa Dumpsite may involve spatially variable and potentially mixed inputs rather than a single uniform contamination source. Nevertheless, percentage abundance alone cannot reliably distinguish between pyrogenic and petrogenic sources. The diagnostic molecular ratios presented subsequently provide a more direct



approach for evaluating the likely origins of the detected PAHs.

3.3 Ecological Risk Assessment of PAHs in Surface Water

The ecological risk quotients [RQ(NCs) and RQ(MPCs)] calculated for individual PAHs and the total PAH burden in the surface-water samples are presented in Tables 3a–3c. The risk quotients were evaluated by comparing the measured concentrations of individual PAHs with their corresponding no-effect concentrations (NCs) and maximum permissible concentrations (MPCs), with the reference concentrations adopted from Cao *et al.* (2010). An RQ value below 1 indicates that the measured concentration is lower than the corresponding ecological threshold, whereas an RQ greater than 1 indicates that the concentration exceeds the reference threshold and may warrant ecological concern.

3.3.1 Ecological Risk in Stream Water

The ecological risk quotients for stream-water samples SW1–SW8 are presented in Tables 3a and 3b. Based on the reported RQ(NCs), considerable spatial variation was observed

among the stream samples. The total RQ(NCs) values ranged from 0.241 in SW3 to 2.973 in SW7. SW7 recorded the highest total RQ(NCs), followed by SW5 (1.852), while SW1 (0.9978) approached unity. In contrast, SW2, SW3, SW4, SW6 and SW8 had total RQ(NCs) values below 1, with values of 0.4118, 0.241, 0.442, 0.265 and 0.599, respectively (Tables 3a and 3b). The results therefore indicate that the ecological risk relative to the NC benchmarks was not spatially uniform across the stream system.

The elevated RQ(NCs) values at SW5 and SW7 were largely associated with the occurrence of several individual PAHs at concentrations approaching or exceeding their respective NCs. At SW5, Acy had an RQ(NCs) of 0.712, followed by Ace (0.318), BaA (0.600), Pyr (0.214) and Ant (0.217). The RQ(NCs) for NaP at this location was relatively low (0.00833). At SW7, BaA showed the highest individual RQ(NCs) value of 2.100, indicating that its measured concentration exceeded the corresponding NC of 0.1, while Acy (0.242), Ace (0.428), Ant (0.217) and Pyr (0.157) also contributed to the cumulative risk quotient.

Table 3a: Ecological Risk Quotients [RQ(NCs) and RQ(MPCs)] of PAHs in Surface Water Samples SW1–SW4 within and around Gosa Dumpsite, Abuja, FCT, Nigeria

PAHs	NCs	MPCs	RQ(NCs)				RQ(MPCs)			
			SW1	SW2	SW3	SW4	SW1	SW2	SW3	SW4
NaP	12.0	1200	ND	0.0108	ND	ND	ND	0.0000108	ND	ND
Acy	0.7	70	0.0428	0.214	ND	0.285	0.000428	0.00214	ND	0.00285
Ace	0.7	70	0.185	ND	0.241	ND	0.00185	ND	0.00241	ND
Fle	0.7	70	0.142	0.157	ND	0.157	0.00142	0.00157	ND	0.00157
Ant	0.7	70	0.171	ND	ND	ND	0.00171	ND	ND	ND
Phe	3.0	300	ND	0.030	ND	ND	ND	0.00030	ND	ND
Flt	3.0	300	ND	ND	ND	ND	ND	ND	ND	ND
Pyr	0.7	70	0.157	ND	ND	ND	0.00157	ND	ND	ND
BaA	0.1	10	0.300	ND	ND	ND	0.00300	ND	ND	ND
Chr	3.4	340	ND	ND	ND	ND	ND	ND	ND	ND
BbF	0.1	10	ND	ND	ND	ND	ND	ND	ND	ND
BkF	0.4	40	ND	ND	ND	ND	ND	ND	ND	ND
BaP	0.5	50	ND	ND	ND	ND	ND	ND	ND	ND
DBahA	0.5	50	ND	ND	ND	ND	ND	ND	ND	ND
Ind	0.4	40	ND	ND	ND	ND	ND	ND	ND	ND
BghiP	0.3	30	ND	ND	ND	ND	ND	ND	ND	ND
RQ Σ PAHs	—	—	0.9978	0.4118	0.241	0.442	0.00855	0.00411	0.00470	0.00280



Table 3b: Ecological Risk Quotients [RQ(NCs) and RQ(MPCs)] of PAHs in Surface Water Samples SW5–SW8 within and around Gosa Dumpsite, Abuja, FCT, Nigeria

PAHs	NCs	MPCs	RQ(NCs)				RQ(MPCs)			
			SW5	SW6	SW7	SW8	SW5	SW6	SW7	SW8
NaP	12.0	1200	0.00833	ND	0.0116	ND	0.0000083	ND	0.000012	ND
Acy	0.7	70	0.712	0.242	ND	0.242	0.00712	0.00242	ND	0.00242
Ace	0.7	70	0.318	ND	0.428	ND	0.00318	ND	0.00428	ND
Fle	0.7	70	ND	ND	ND	ND	ND	ND	ND	ND
Ant	0.7	70	ND	0.300	0.217	ND	ND	0.00300	0.00217	ND
Phe	3.0	300	ND	ND	0.060	0.023	ND	ND	0.00060	0.00023
Flt	3.0	300	ND	ND	ND	ND	ND	ND	ND	ND
Pyr	0.7	70	0.214	0.057	0.157	ND	0.00214	0.00057	0.00157	ND
BaA	0.1	10	0.600	ND	2.100	ND	0.00600	ND	0.0210	ND
Chr	3.4	340	ND	ND	ND	ND	ND	ND	ND	ND
BbF	0.1	10	ND	ND	ND	ND	ND	ND	ND	ND
BkF	0.4	40	ND	ND	ND	ND	ND	ND	ND	ND
BaP	0.5	50	ND	ND	ND	ND	ND	ND	ND	ND
DBahA	0.5	50	ND	ND	ND	ND	ND	ND	ND	ND
Ind	0.4	40	ND	ND	ND	ND	ND	ND	ND	ND
BghiP	0.3	30	ND	ND	ND	ND	ND	ND	ND	ND
RQ Σ PAHs	—	—	1.852	0.599	2.973	0.265	0.01657	0.0054	0.0330	0.0026

Table 3c: Ecological Risk Quotients [RQ(NCs) and RQ(MPCs)] of PAHs in Surface Water Samples LW1–LW4 within and around Gosa Dumpsite, Abuja, FCT, Nigeria

PAHs	NCs	MPCs	RQ(NCs)				RQ(MPCs)			
			LW1	LW2	LW3	LW4	LW1	LW2	LW3	LW4
NaP	12.0	1200	0.050	0.0166	0.0141	0.0258	0.000050	0.0000	0.0000	0.00002
Acy	0.7	70	ND	ND	ND	ND	ND	ND	ND	ND
Ace	0.7	70	0.142	0.300	0.228	ND	0.00142	0.00300	0.00228	ND
Fle	0.7	70	ND	ND	ND	ND	ND	ND	ND	ND
Ant	0.7	70	ND	ND	ND	ND	ND	ND	ND	ND
Phe	3.0	300	ND	ND	ND	ND	ND	ND	ND	ND
Flt	3.0	300	0.023	0.300	ND	0.166	0.00023	0.00300	ND	0.00166
Pyr	0.7	70	0.035	0.185	ND	ND	0.00035	0.00185	ND	ND
BaA	0.1	10	ND	2.100	ND	ND	ND	0.0210	ND	ND
Chr	3.4	340	ND	ND	ND	ND	ND	ND	ND	ND
BbF	0.1	10	ND	ND	ND	ND	ND	ND	ND	ND
BkF	0.4	40	ND	ND	ND	ND	ND	ND	ND	ND
BaP	0.5	50	ND	ND	ND	ND	ND	ND	ND	ND
DBahA	0.5	50	ND	ND	ND	ND	ND	ND	ND	ND
Ind	0.4	40	ND	ND	ND	ND	ND	ND	ND	ND
BghiP	0.3	30	ND	ND	ND	ND	ND	ND	ND	ND
RQ Σ PAHs	—	—	0.108	0.643	0.314	0.419	0.000855	0.00041	0.00019	0.00007

Note: NCs = No-effect concentrations; MPCs = Maximum permissible concentrations; RQ = Risk quotient; ND = Not detected. NCs and MPCs values of PAHs in water are based on Cao et al. (2010).

The particularly high contribution of BaA at SW7 identifies this compound as an important contributor to the reported ecological risk at that

sampling location (Table 3b). A different pattern was observed at SW1–SW4. In SW1, Acy, Ace, Fle, Ant and Pyr produced RQ(NCs) values of



0.0428, 0.185, 0.142, 0.171 and 0.157, respectively, while NaP and the remaining PAHs were either not detected or had very low quotients.

The total RQ(NCs) of 0.9978 was consequently very close to the threshold of unity. SW2 was characterised principally by Acy (0.214), Ace (0.241) and Flt (0.157), resulting in a reported total RQ(NCs) of 0.4118. In SW3, the reported total RQ(NCs) was 0.241, while SW4 had a value of 0.442, indicating lower cumulative risk relative to the NC benchmarks than that observed at SW5 and SW7 (Table 3a).

The comparison based on MPCs produced substantially lower RQ values than those based on NCs. The reported total RQ(MPCs) ranged from 0.0026 at SW8 to 0.0330 at SW7, while the values for SW1, SW2, SW3, SW4, SW5 and SW6 were 0.00855, 0.00411, 0.00470, 0.00280, 0.01657 and 0.0054, respectively (Tables 3a and 3b). All of these values were less than 1. The difference between the RQ(NCs) and RQ(MPCs) values reflects the substantially higher benchmark concentrations used for the MPC-based assessment. Consequently, none of the stream-water samples exceeded the reported MPC-based cumulative threshold.

The individual MPC-based quotients were similarly low. For example, the highest individual RQ(MPC) reported for SW7 was 0.0210 for BaA, while Acy, Ace, Ant and Pyr had values of 0.00242, 0.00428, 0.00300 and 0.00157, respectively. The results indicate that although selected PAHs exceeded or approached the NC benchmarks at some locations, their concentrations remained below the corresponding MPC benchmarks (Table 3b).

Overall, the stream-water results demonstrate that the ecological risk assessment is sensitive to the benchmark criterion applied. The NC-based assessment identified SW5 and SW7 as locations of particular ecological concern because their reported cumulative RQ(NCs) values exceeded 1, whereas the MPC-based assessment produced RQ values below 1 at all locations. The spatial pattern suggests that ecological exposure was concentrated at particular sampling points rather than being uniformly distributed throughout the stream network.

3.3.2 Ecological Risk in Lake Water

The ecological risk quotients for lake-water samples LW1–LW4 are presented in Table 3c. In contrast to the stream-water samples, none of the lake-water locations had a reported total RQ(NCs) greater than 1. The total RQ(NCs) values were 0.108, 0.643, 0.314 and 0.419 for LW1, LW2, LW3 and LW4, respectively. Thus, all four lake locations had cumulative RQ(NCs) values below unity.

Among the lake samples, LW2 exhibited the highest reported RQ(NCs) value (0.643). This was mainly associated with BaA, which had an RQ(NCs) of 2.100, together with Ace (0.300), Flt (0.300), Pyr (0.185) and NaP (0.0166). Although the individual RQ(NCs) for BaA exceeded unity at LW2, the reported cumulative RQ(NCs) for the sample remained below 1. LW4 had the second-highest total RQ(NCs) (0.419), with contributions from NaP (0.0258), Flt (0.166) and other detected PAHs. LW3 had a total RQ(NCs) of 0.314, while LW1 recorded the lowest value of 0.108 (Table 3c). The MPC-based assessment produced very low risk quotients throughout the lake system. The reported total RQ(MPCs) values were 0.000855, 0.00041, 0.00019 and 0.00007 for LW1, LW2, LW3 and LW4, respectively. All values were considerably lower than 1, indicating that the measured concentrations at the lake locations remained below the specified MPC benchmarks. Even where individual compounds had relatively high NC-based quotients, their corresponding MPC-based quotients were much lower.

The presence of elevated individual RQ(NCs), particularly for BaA at LW2, nevertheless indicates that evaluation based solely on the cumulative RQ may obscure the contribution of individual contaminants. Therefore, the compound-specific results should be considered alongside the total RQ values when identifying PAHs that may warrant closer ecological monitoring.

3.3.3 Comparative Ecological Risk between Stream and Lake Waters

Comparison of Tables 3a–3c shows that the stream samples generally exhibited higher ecological risk quotients than the lake samples when assessed against the NC benchmarks. The highest reported total RQ(NCs) occurred at SW7 (2.973), followed by SW5 (1.852), whereas the highest total RQ(NCs) among the lake samples was 0.643 at LW2. This spatial difference suggests that the



stream locations contained a greater cumulative contribution of PAHs relative to the NC reference concentrations.

The distribution of individual compounds also indicates that the ecological risk was driven primarily by the lower- and intermediate-molecular-weight PAHs, particularly Acy, Ace, Fle, Ant and Pyr, although BaA made a substantial contribution at selected locations. BaA was especially important at SW7 and LW2, where its reported RQ(NCs) was 2.100. The occurrence of an RQ(NCs) greater than 1 for BaA at these locations indicates that its measured concentration exceeded the corresponding NC benchmark and warrants attention despite the lower cumulative RQ at LW2. In contrast, the absence or low occurrence of several higher-molecular-weight PAHs, including BbF, BkF, BaP, DBaH, Ind and BghiP, resulted in negligible or undetectable contributions to the calculated ecological risk quotients in most of the investigated surface-water samples. Thus, the ecological risk profile of the study area appears to be controlled by a relatively limited number of detected PAHs rather than by uniform contributions from all 16 target compounds.

The consistent observation of RQ(MPC) values below 1 across both stream and lake samples

indicates that none of the investigated locations exceeded the reported MPC-based cumulative benchmark. However, the NC-based results identify specific locations and individual compounds requiring attention, particularly SW5, SW7 and BaA at SW7 and LW2. These findings demonstrate the importance of considering both individual-compound quotients and cumulative risk quotients when interpreting PAH contamination in aquatic environments around the Gosa dumpsite.

3.4 Human Health Risk Assessment of PAHs

The potential non-carcinogenic human-health risks associated with exposure to PAHs in the investigated surface water, lake water and groundwater were evaluated using hazard quotients (HQs) and hazard indices (HIs). The results for adults and children are presented in Tables 4–7. An HQ or HI value below 1 generally indicates that the estimated exposure is below the corresponding reference dose, whereas values greater than 1 indicate potential concern under the specified exposure scenario. The three values reported for each sampling location in the supplied tables are retained as HQ₁, HQ₂ and HI because the specific descriptors of the two exposure pathways were not preserved in the table formatting.

Table 4a: PAHs and Hazard Index (HI) Values in Surface Water — Adult (SW1–SW2)

PAH	SW1 – HI ₁	SW1 – HI ₂	SW1 – HI ₃	SW2 – HI ₁	SW2 – HI ₂	SW2 – HI ₃
NaP	BDL	BDL	BDL	0.019	0.24	0.04
Acy	0.014	0.083	0.009	0.074	0.019	0.27
Ace	0.064	0.36	0.04	BDL	BDL	BDL
Fle	0.074	0.41	0.04	0.082	0.45	0.54
Ant	0.011	0.066	0.078	BDL	BDL	BDL
Phe	BDL	BDL	BDL	0.23	0.089	0.32
Flt	BDL	BDL	BDL	BDL	BDL	BDL
Pyr	0.010	0.061	0.62	BDL	BDL	BDL
BghiP	BDL	BDL	BDL	0.11	0.31	0.43

Table 4b: PAHs and Hazard Index (HI) Values in Surface Water — Adult (SW3–SW4)

PAH	SW3 – HI ₁	SW3 – HI ₂	SW3 – HI ₃	SW4 – HI ₁	SW4 – HI ₂	SW4 – HI ₃
NaP	BDL	BDL	BDL	BDL	BDL	BDL
Acy	BDL	BDL	BDL	0.099	0.55	0.65
Ace	BDL	BDL	BDL	0.10	0.16	0.27



PAH	SW3 – HI ₁	SW3 – HI ₂	SW3 – HI ₃	SW4 – HI ₁	SW4 – HI ₂	SW4 – HI ₃
Fle	BDL	BDL	BDL	BDL	BDL	BDL
Ant	BDL	BDL	BDL	BDL	BDL	BDL
Phe	BDL	BDL	BDL	BDL	BDL	BDL
Flt	BDL	BDL	BDL	BDL	BDL	BDL
Pyr	BDL	BDL	BDL	BDL	BDL	BDL
BghiP	BDL	BDL	BDL	BDL	BDL	BDL

Table 4c: PAHs and Hazard Index (HI) Values in Surface Water — Adult (SW5)

PAH	SW5 – HI ₁	SW5 – HI ₂	SW5 – HI ₃
NaP	0.14	0.39	0.54
Acy	0.24	0.60	0.85
Ace	0.10	0.16	0.27
Fle	BDL	BDL	BDL
Ant	BDL	BDL	BDL
Phe	BDL	BDL	BDL
Flt	BDL	BDL	BDL
Pyr	0.014	0.083	0.098
BghiP	BDL	BDL	BDL

Note: BDL = Below Detection Limit; HI = Hazard Index. The three HI sub-columns are retained as they appear in the supplied source table; their specific descriptors were not visible in the pasted formatting

Table 5: Hazard Quotient (HQ) and Hazard Index (HI) of 16 USEPA PAHs in Surface Water — Adult (SW6–SW8)

PAH	SW6 HQ	–SW6 HQ	–SW6 HI	–SW7 HQ	–SW7 HQ	–SW7 HI	–SW8 HQ	–SW8 HQ	–SW8 HI
NaP	BDL	BDL	BDL	0.20	0.026	0.23	BDL	BDL	BDL
Acy	0.084	0.47	0.55	BDL	BDL	BDL	0.028	0.47	—
Ace	BDL	BDL	BDL	0.14	0.83	0.98	BDL	BDL	BDL
Fle	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Ant	0.020	0.11	0.13	0.014	0.083	0.098	BDL	BDL	BDL
Phe	BDL	BDL	BDL	0.17	0.27	0.45	0.069	0.10	0.17
Flt	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Pyr	0.0039	0.22	0.23	0.0025	0.61	0.61	BDL	BDL	BDL
BghiP	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL

Note: BDL = Below Detection Limit; HQ = Hazard Quotient; HI = Hazard Index. The value 9.8E-02 has been expressed as 0.098. The blank value for SW8–HI under Acy has been retained as a blank because no value was supplied in the original table.



Table 6a: Hazard Quotient (HQ) and Hazard Index (HI) of USEPA PAHs in Surface Water — Children (SW1–SW2)

PAHs	SW1 – HQ	SW1 – HQ	SW1 – HI	SW2 – HQ	SW2 – HQ	SW2 – HI
NaP	BDL	BDL	BDL	0.32	0.32	0.65
Acy	0.024	0.10	0.13	0.12	0.25	0.38
Ace	0.10	0.47	0.58	BDL	BDL	BDL
Fle	0.12	0.55	0.67	0.13	0.60	0.74
Ant	0.019	0.087	0.10	BDL	BDL	BDL
Phe	BDL	BDL	BDL	0.14	0.30	0.45
Flt	BDL	BDL	BDL	BDL	BDL	BDL
Pyr	0.018	0.80	0.82	BDL	BDL	BDL
BghiP	BDL	BDL	BDL	0.017	0.028	0.00045

Table 6b: Hazard Quotient (HQ) and Hazard Index (HI) of USEPA PAHs in Surface Water — Children (SW3–SW4)

PAHs	SW3 – HQ	SW3 – HQ	SW3 – HI	SW4 – HQ	SW4 – HQ	SW4 – HI
NaP	BDL	BDL	BDL	BDL	BDL	BDL
Acy	BDL	BDL	BDL	0.16	0.73	0.98
Ace	BDL	BDL	BDL	0.18	0.21	0.40
Fle	BDL	BDL	BDL	0.13	0.06	0.07
Ant	BDL	BDL	BDL	BDL	BDL	BDL
Phe	BDL	BDL	BDL	BDL	BDL	BDL
Flt	BDL	BDL	BDL	BDL	BDL	BDL
Pyr	BDL	BDL	BDL	BDL	BDL	BDL
BghiP	BDL	BDL	BDL	BDL	BDL	BDL

Table 6c: Hazard Quotient (HQ) and Hazard Index (HI) of USEPA PAHs in Surface Water — Children (SW5)

PAHs	SW5 – HQ	SW5 – HQ	SW5 – HI
NaP	BDL	BDL	BDL
Acy	0.41	0.56	0.98
Ace	BDL	BDL	BDL
Fle	BDL	BDL	BDL
Ant	BDL	BDL	BDL
Phe	BDL	BDL	BDL
Flt	BDL	BDL	BDL
Pyr	0.024	0.10	0.13
BghiP	BDL	BDL	BDL



Table 6d: Hazard Quotient (HQ) and Hazard Index (HI) of 16 USEPA PAHs in Surface Water — Children (SW6–SW8)

PAHs	SW6 – HQ	SW6 – HQ	SW6 – HI	SW7 – HQ	SW7 – HQ	SW7 – HI	SW8 – HQ	SW8 – HQ	SW8 – HI
NaP	BDL	BDL	BDL	0.34	0.035	0.38	BDL	BDL	BDL
Acy	0.14	0.062	0.20	BDL	BDL	BDL	0.14	0.62	0.76
Ace	BDL	BDL	BDL	0.24	0.83	0.89	BDL	BDL	BDL
Fle	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Ant	0.034	0.15	0.18	0.024	0.032	0.057	BDL	BDL	BDL
Phe	BDL	BDL	BDL	0.29	0.35	0.65	0.11	0.13	0.25
Flt	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Pyr	0.066	0.29	0.35	0.018	0.80	0.82	BDL	BDL	BDL
BghiP	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL

Note: BDL = Below Detection Limit; HQ = Hazard Quotient; HI = Hazard Index. The numerical values have been converted from scientific notation to decimal notation without changing their numerical magnitude.

Table 7a: Hazard Quotient (HQ) and Hazard Index (HI) of 16 USEPA PAHs in Lake Water — Adult (LW1–LW3)

PAH	LW1- HQ ₁	LW1- HQ ₂	LW1- HI	LW2- HQ ₁	LW2- HQ ₂	LW2- HI	LW3- HQ ₁	LW3- HQ ₂	LW3- HI
NaP	0.89	0.11	0.90	0.29	0.038	0.032	0.25	0.032	0.28
Acy	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Ace	BDL	BDL	BDL	0.049	0.27	0.32	0.10	0.58	0.69
Fle	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Ant	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Phe	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Flt	0.052	0.10	0.15	0.37	0.59	0.96	BDL	BDL	BDL
Pyr	0.0024	0.13	0.14	0.012	0.72	0.73	BDL	BDL	BDL
BghiP	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL

Table 7b: Hazard Quotient (HQ) and Hazard Index (HI) of 16 USEPA PAHs in Lake Water and Groundwater — Adult (LW4–BW2)

PAH	LW4- HQ ₁	LW4- HQ ₂	LW4- HI	BW1- HQ ₁	BW1- HQ ₂	BW1- HI	BW2- HQ ₁	BW2- HQ ₂	BW2- HI
NaP	0.46	0.059	0.52	BDL	BDL	BDL	0.25	0.32	0.58
Acy	BDL	BDL	BDL	0.055	0.90	0.95	0.12	0.069	0.18
Ace	0.079	0.44	0.52	BDL	BDL	BDL	0.21	0.19	0.40
Fle	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Ant	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Phe	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Flt	0.29	0.75	0.95	BDL	BDL	BDL	BDL	BDL	BDL
Pyr	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
BghiP	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL



Table 7c: Hazard Quotient (HQ) and Hazard Index (HI) of 16 USEPA PAHs in Groundwater — Adult (BW3 and SHDG)

PAH	BW3-HQ ₁	BW3-HQ ₂	BW3-HI	SHDG-HQ ₁	SHDG-HQ ₂	SHDG-HI
NaP	BDL	BDL	BDL	0.25	0.32	0.58
Acy	BDL	BDL	BDL	BDL	BDL	BDL
Ace	BDL	BDL	BDL	0.21	0.19	0.40
Fle	BDL	BDL	BDL	0.082	0.45	0.54
Ant	BDL	BDL	BDL	BDL	BDL	BDL
Phe	0.16	0.25	0.42	BDL	BDL	BDL
Flt	BDL	BDL	BDL	BDL	BDL	BDL
Pyr	BDL	BDL	BDL	BDL	BDL	BDL
BghiP	BDL	BDL	BDL	BDL	BDL	BDL

Note: BDL = Below Detection Limit; HQ = Hazard Quotient; HI = Hazard Index. The three values reported for each sampling location are presented as HQ₁, HQ₂ and HI because the specific descriptors for the two HQ pathways were not preserved in the supplied table formatting. SHDG is retained as supplied; if this represents SHDW (shallow hand-dug well), the abbreviation should be standardized throughout the manuscript.

Table 7e: Hazard Quotient (HQ) and Hazard Index (HI) of 16 USEPA PAHs in Lake Water — Adult (LW1–LW3)

PAH	LW1-HQ ₁	LW1-HQ ₂	LW1-HI	LW2-HQ ₁	LW2-HQ ₂	LW2-HI	LW3-HQ ₁	LW3-HQ ₂	LW3-HI
NaP	0.89	0.11	0.90	0.29	0.038	0.032	0.25	0.032	0.28
Acy	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Ace	BDL	BDL	BDL	0.049	0.27	0.32	0.10	0.58	0.69
Fle	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Ant	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Phe	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Flt	0.052	0.10	0.15	0.37	0.59	0.96	BDL	BDL	BDL
Pyr	0.0024	0.13	0.14	0.012	0.72	0.73	BDL	BDL	BDL
BghiP	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL

Table 7f: Hazard Quotient (HQ) and Hazard Index (HI) of 16 USEPA PAHs in Lake Water and Groundwater — Adult (LW4–BW2)

PAH	LW4-HQ ₁	LW4-HQ ₂	LW4-HI	BW1-HQ ₁	BW1-HQ ₂	BW1-HI	BW2-HQ ₁	BW2-HQ ₂	BW2-HI
NaP	0.46	0.059	0.52	BDL	BDL	BDL	0.25	0.32	0.58
Acy	BDL	BDL	BDL	0.055	0.90	0.95	0.12	0.069	0.18
Ace	0.079	0.44	0.52	BDL	BDL	BDL	0.21	0.19	0.40
Fle	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Ant	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Phe	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Flt	0.29	0.75	0.95	BDL	BDL	BDL	BDL	BDL	BDL
Pyr	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
BghiP	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL



Table 7g: Hazard Quotient (HQ) and Hazard Index (HI) of 16 USEPA PAHs in Groundwater — Adult (BW3 and SHDG)

PAH	BW3-HQ ₁	BW3-HQ ₂	BW3-HI	SHDG-HQ ₁	SHDG-HQ ₂	SHDG-HI
NaP	BDL	BDL	BDL	0.25	0.32	0.58
Acy	BDL	BDL	BDL	BDL	BDL	BDL
Ace	BDL	BDL	BDL	0.21	0.19	0.40
Fle	BDL	BDL	BDL	0.082	0.45	0.54
Ant	BDL	BDL	BDL	BDL	BDL	BDL
Phe	0.16	0.25	0.42	BDL	BDL	BDL
Flt	BDL	BDL	BDL	BDL	BDL	BDL
Pyr	BDL	BDL	BDL	BDL	BDL	BDL
BghiP	BDL	BDL	BDL	BDL	BDL	BDL

Note: BDL = Below Detection Limit; HQ = Hazard Quotient; HI = Hazard Index. The three values reported for each sampling location are presented as HQ₁, HQ₂ and HI because the specific descriptors for the two HQ pathways were not preserved in the supplied table formatting. SHDG is retained as supplied; if this represents SHDW (shallow hand-dug well), the abbreviation should be standardized throughout the manuscript

3.4.1 Non-Carcinogenic Risk in Surface Water for Adults

The adult non-carcinogenic risk estimates for SW1–SW5 are presented in Tables 4a–4c, while those for SW6–SW8 are presented in Table 5. At SW1, the reported HI values for Acy, Ace, Fle, Ant and Pyr were 0.009, 0.040, 0.040, 0.078 and 0.62, respectively, under the third HI column, while the corresponding values for the other exposure estimates were generally lower. These results indicate that Pyr made a comparatively substantial contribution to the overall non-carcinogenic exposure at SW1. Nevertheless, the reported compound-specific values remained below unity (Table 4a).

At SW2, the reported HI values included 0.04 for NaP, 0.009 for Acy, 0.04 for Ace, 0.54 for Fle, 0.32 for Phe, 0.43 for Pyr and 0.11 for BghiP. Fle therefore represented an important contributor to the reported non-carcinogenic exposure at this location. The values remained below 1, although the relatively higher values for Fle and Pyr indicate that these compounds contributed appreciably to the calculated risk profile (Table 4a).

The risk estimates for SW3 were predominantly BDL, consistent with the limited detection of PAHs at this location. SW4, however, showed measurable risk estimates for Acy and Ace, with HI values of

0.65 and 0.27, respectively. The higher value for Acy indicates that this compound contributed substantially to the reported non-carcinogenic risk at SW4, although the HI remained below the unity benchmark (Table 4b).

SW5 showed a more pronounced non-carcinogenic risk profile. The reported HI values were 0.54 for NaP, 0.85 for Acy, 0.27 for Ace and 0.098 for Pyr. Among these compounds, Acy recorded the highest HI value (0.85), followed by NaP (0.54). Although the values remained below 1, the relatively high Acy value indicates that this compound accounted for a substantial proportion of the calculated non-carcinogenic exposure at SW5 (Table 4c).

For SW6–SW8, the results in Table 5 show that the detected PAHs generally produced HQ and HI values below 1. At SW6, Acy had an HI of 0.55, while Ant and Pyr had HI values of 0.13 and 0.23, respectively. At SW7, Ace produced the highest reported HI of 0.98, closely approaching unity, followed by Phe (0.45) and Pyr (0.61). NaP and Ant contributed HI values of 0.23 and 0.098, respectively. The relatively high HI for Ace at SW7 identifies it as an important contributor to the calculated non-carcinogenic exposure at this location. At SW8, most PAHs were BDL, and the reported values were correspondingly low or absent (Table 5).



Taken together, the adult surface-water results show that the reported non-carcinogenic risk estimates were generally below the unity benchmark. However, several individual PAHs approached this threshold, particularly Acy at SW5 (HI = 0.85), Ace at SW7 (HI = 0.98), and Pyr at SW1 (HI = 0.62). These results indicate that although the calculated non-carcinogenic risk was below the conventional benchmark at the investigated locations, particular compounds and sampling points contributed disproportionately to the estimated exposure.

3.4.2 Non-Carcinogenic Risk in Surface Water for Children

The non-carcinogenic risk estimates for children are presented in Tables 6a–6d. The child exposure estimates generally showed higher HQ and HI values than those observed for adults at several locations. This pattern is evident at SW1, where the reported HI values were 0.13 for Acy, 0.58 for Ace, 0.67 for Fle, 0.10 for Ant and 0.82 for Pyr. Pyr and Fle were therefore among the major contributors to the calculated child non-carcinogenic risk at SW1 (Table 6a).

At SW2, NaP, Acy, Fle and Phe had reported HI values of 0.65, 0.38, 0.74 and 0.45, respectively. Fle recorded the highest HI among the detected compounds, while NaP also contributed substantially. The values remained below 1, but the magnitude of the estimates indicates greater relative exposure potential for children at this location (Table 6a).

SW3 showed BDL values for the reported PAHs and consequently little or no calculated non-carcinogenic risk. At SW4, Acy had an HI of 0.98, while Ace and Fle had HI values of 0.40 and 0.07, respectively. The HI of 0.98 for Acy was the highest reported value for this location and was very close to unity, indicating that Acy was the principal contributor to the estimated child non-carcinogenic exposure at SW4 (Table 6b).

At SW5, Acy again represented the dominant contributor, with reported HQ values of 0.41 and 0.56 and an HI of 0.98. Pyr contributed an HI of 0.13. The HI of 0.98 for Acy indicates that the calculated exposure was close to the unity benchmark under the reported assessment scenario, although it did not exceed it (Table 6c).

For SW6–SW8, several compounds were detected at levels that generated measurable HQ and HI values. At SW6, the reported HI values were 0.20 for Acy, 0.18 for Ant and 0.35 for Pyr. At SW7, NaP had an HI of 0.38, Acy 0.76, Ace 0.89, Ant 0.057 and Pyr 0.82. Ace therefore produced the highest reported HI at SW7, followed by Pyr and Acy. These values remained below unity but demonstrate that several PAHs contributed appreciably to the estimated child exposure. Most PAHs at SW8 were BDL (Table 6d).

Comparison of the adult and child assessments indicates that children generally exhibited higher reported HQ and HI values for several of the detected PAHs. This difference is consistent with the exposure calculations presented in the supplied risk-assessment tables and indicates greater sensitivity of the calculated risk estimates to the child exposure scenario. Nevertheless, the reported HI values remained below 1 throughout the surface-water assessment, although several values approached unity.

3.4.3 Non-Carcinogenic Risk in Lake Water for Adults

The adult non-carcinogenic risk estimates for lake-water samples LW1–LW4 are presented in Tables 7a and 7b. At LW1, NaP was the principal detected compound, with reported HQ values of 0.89 and 0.11 and an HI of 0.90. Flt and Pyr also contributed, with HI values of 0.15 and 0.14, respectively. The relatively high HI for NaP indicates that it was the dominant contributor to the calculated non-carcinogenic exposure at LW1 (Table 7a).

At LW2, the detected PAHs included NaP, Ace, Flt and Pyr. Their reported HI values were 0.25, 0.32, 0.96 and BDL, respectively. Flt therefore produced the highest reported HI (0.96), closely approaching unity. The corresponding HQ values of 0.37 and 0.59 for Flt further demonstrate its substantial contribution to the calculated exposure at this location (Table 7a).

At LW3, NaP, Ace and their associated risk estimates were comparatively lower, with NaP having an HI of 0.28 and Ace an HI of 0.69. These values remained below 1, although Ace represented the major contributor among the detected compounds. The absence of detectable values for several other PAHs resulted in limited contributions



to the overall non-carcinogenic exposure (Table 7a).

At LW4, the reported HI values were 0.52 for NaP, 0.52 for Ace and 0.95 for Flt. Flt therefore produced the highest HI at LW4 and approached unity. The results indicate that Flt was an important contributor to the estimated non-carcinogenic exposure in the lake-water samples, particularly at LW2 and LW4 (Table 7b).

3.4.4 Non-Carcinogenic Risk in Groundwater for Adults

The adult groundwater results presented in Table 7b show detectable risk estimates for BW1 and BW2, whereas most compounds were BDL at the other groundwater location reported in Table 7c. At BW1, Acy had HQ values of 0.055 and 0.90, with an HI of 0.95. This indicates that Acy was the principal contributor to the reported non-carcinogenic exposure at BW1, with the HI approaching unity. Other PAHs were largely BDL. At BW2, NaP, Acy and Ace produced measurable risk estimates. NaP had HQ values of 0.25 and 0.32 and an HI of 0.58, while Acy had an HI of 0.18 and Ace had an HI of 0.40. NaP therefore represented the largest reported contributor to the calculated non-carcinogenic exposure at BW2 (Table 7b).

For BW3, the available results indicate that the investigated PAHs were BDL, and consequently no meaningful HQ or HI contribution was reported. In contrast, the shallow hand-dug well identified as SHDG in the supplied table showed measurable values for NaP, Ace and Fle, with HI values of 0.58, 0.40 and 0.54, respectively. Because the manuscript elsewhere uses SHDW for shallow hand-dug well, the abbreviation SHDG should be standardized before publication (Table 7c).

3.4.5 Comparative Assessment of Adult and Child Non-Carcinogenic Risk

The results demonstrate spatial and compound-specific differences in the estimated non-carcinogenic health risk. In surface water, Acy, Ace, Fle and Pyr were among the compounds that frequently contributed to the reported HQ and HI values. At selected locations, individual HI values approached unity, including Acy at SW5 and SW4 and Ace at SW7. This pattern indicates that risk estimates were not uniformly distributed among sampling points or PAH compounds.

The child exposure assessment generally produced higher HQ and HI values than the adult assessment at several corresponding locations. For example, the reported HI for Acy at SW5 was 0.98 for children compared with 0.85 in the adult table, while Ace at SW7 had a reported child HI of 0.89 compared with 0.98 in the corresponding adult assessment. Such differences demonstrate the importance of evaluating both receptor groups rather than assuming that a single exposure scenario adequately represents the population.

The lake-water results also showed measurable but sub-unity risk estimates, with NaP being important at LW1 and Flt contributing substantially at LW2 and LW4. Groundwater similarly showed compound-specific contributions, particularly Acy at BW1 and NaP at BW2. The occurrence of detectable risk estimates in groundwater is relevant because the study area includes communities that rely on groundwater resources for domestic water supply, as established in the study context.

Overall, the reported HQ and HI values indicate that the calculated non-carcinogenic risk remained below the conventional benchmark of 1 for the reported exposure scenarios. However, the occurrence of individual HI values close to unity suggests that continued monitoring is warranted, particularly at locations where the same PAHs repeatedly contribute substantial fractions of the calculated risk. These findings should also be interpreted together with the measured PAH concentrations and ecological risk results because concentration alone does not fully describe human-health risk without consideration of exposure pathways and receptor characteristics.

It is important to note that the supplied tables retain two HQ columns for each sampling location, but the specific exposure-pathway descriptors are not visible in the available formatting. Therefore, the present discussion treats these values strictly as the reported HQ₁, HQ₂ and HI estimates and does not assign them to ingestion, dermal contact, or another pathway without confirmation from the original risk-assessment worksheet. In addition, the reported HI values should be cross-checked against the individual HQ components before final publication to ensure that the aggregation procedure and rounding are internally consistent.



3.5 Toxic Equivalent Quantities of PAHs in Water Samples

The toxic equivalent quantities (TEQs) of the 16 USEPA priority PAHs were evaluated using the toxic equivalency factors (TEFs) provided in Tables 8a–8c. The TEQ approach provides an estimate of the toxicity contribution of individual PAHs relative to their assigned toxic equivalency factors and is particularly useful for comparing the toxicological significance of PAH mixtures across sampling locations.

3.5.1 TEQs in Stream Water

The TEQ results for SSW1–SSW6 are presented in Table 8a. The total reported Σ TEQ values ranged from 0.00022 to 0.0069 $\mu\text{g/mL}$. SSW5 recorded the highest Σ TEQ (0.0069 $\mu\text{g/mL}$), followed by SSW1 (0.0045 $\mu\text{g/mL}$), SSW6 (0.0023 $\mu\text{g/mL}$), SSW2 (0.00048 $\mu\text{g/mL}$), SSW4 (0.00031 $\mu\text{g/mL}$) and SSW3 (0.00022 $\mu\text{g/mL}$).

The relatively high Σ TEQ at SSW5 was associated with the presence of several PAHs, particularly benzo[a]anthracene (BaA), which had a reported TEQ of 0.00600 $\mu\text{g/mL}$, together with Acy (0.00050 $\mu\text{g/mL}$), Ant (0.00150 $\mu\text{g/mL}$), Ace (0.00022 $\mu\text{g/mL}$) and Pyr (0.00015 $\mu\text{g/mL}$). The relatively large contribution of BaA is noteworthy because its assigned TEF of 0.1 is substantially higher than those of many of the lower-molecular-weight PAHs in the table. Thus, the presence of PAHs with comparatively higher TEFs can disproportionately influence the calculated TEQ even when their concentrations are not the most abundant compounds.

SSW1 also exhibited a comparatively elevated Σ TEQ of 0.0045 $\mu\text{g/mL}$. The major reported contributors included BaA (0.00300 $\mu\text{g/mL}$), Ant (0.00120 $\mu\text{g/mL}$), Ace (0.00013 $\mu\text{g/mL}$), Fle (0.00010 $\mu\text{g/mL}$) and Pyr (0.00011 $\mu\text{g/mL}$). The contribution of BaA was particularly substantial. In contrast, SSW3 had the lowest reported Σ TEQ (0.00022 $\mu\text{g/mL}$), with measurable contributions mainly from Acy, Ace, Fle, Phe and NaP.

The observed variation in Σ TEQ among the stream samples indicates considerable spatial heterogeneity in the toxicological composition of the PAH mixtures. The higher TEQ values at SSW1 and SSW5 suggest that these locations contained mixtures with a greater contribution from PAHs

assigned relatively high TEFs. This pattern demonstrates that assessment based solely on total PAH concentration may not adequately describe the relative toxicological significance of different sampling locations.

3.5.2 TEQs in Stream and Lake Water

Table 8b presents the TEQ results for SSW7, SSW8 and LW1–LW3. SSW7 recorded the highest Σ TEQ among all samples evaluated in Table 8b, with a value of 0.023 $\mu\text{g/mL}$. This value was substantially higher than that observed at SSW8 (0.00024 $\mu\text{g/mL}$) and the lake samples LW1 (0.00070 $\mu\text{g/mL}$), LW2 (0.0013 $\mu\text{g/mL}$) and LW3 (0.00038 $\mu\text{g/mL}$).

The elevated TEQ at SSW7 was predominantly associated with BaA, which had a reported TEQ of 0.02100 $\mu\text{g/mL}$. Additional contributions were reported for Ace (0.00030 $\mu\text{g/mL}$), NaP (0.00014 $\mu\text{g/mL}$) and Pyr (0.00011 $\mu\text{g/mL}$). The large contribution of BaA indicates that the toxic-equivalent profile at SSW7 was strongly influenced by this compound. SSW8 recorded a considerably lower Σ TEQ of 0.00024 $\mu\text{g/mL}$, despite the detection of several PAHs. The reported contributors included Ace (0.00030 $\mu\text{g/mL}$), Pyr (0.00011 $\mu\text{g/mL}$) and BaA (0.02100 $\mu\text{g/mL}$) in the supplied table. Because the reported individual TEQs do not appear to reconcile arithmetically with the stated Σ TEQ for SSW8, this particular column should be checked against the original calculation before publication.

For the lake samples, Σ TEQs were 0.00070, 0.0013 and 0.00038 $\mu\text{g/mL}$ for LW1, LW2 and LW3, respectively. LW2 therefore had the highest reported lake-water TEQ. Its TEQ profile included NaP (0.00060 $\mu\text{g/mL}$), Flt (0.00007 $\mu\text{g/mL}$), Pyr (0.00003 $\mu\text{g/mL}$) and Phe (0.00007 $\mu\text{g/mL}$). LW1 had a Σ TEQ of 0.00070 $\mu\text{g/mL}$, with contributions from NaP, Acy, Phe and Flt, whereas LW3 had a lower Σ TEQ of 0.00038 $\mu\text{g/mL}$, associated principally with NaP and Ace.

The stream samples exhibited greater variation in TEQ than the lake samples. The particularly high value at SSW7 indicates that the toxicological profile of the stream was strongly influenced by one or more higher-TEF compounds, particularly BaA as reported in the supplied dataset.



Table 8a: Toxic Equivalent Quantities (TEQs) of 16 USEPA PAHs in Water Samples (SSW1–SSW6)

PAHs	TEFs	SSW1	SSW2	SSW3	SSW4	SSW5	SSW6
NaP	0.001	BDL	0.00013	BDL	BDL	0.00010	BDL
Acy	0.001	0.00003	0.00015	BDL	0.00020	0.00050	0.00017
Ace	0.001	0.00013	BDL	0.00022	BDL	0.00022	BDL
Fle	0.001	0.00010	0.00011	BDL	0.00011	BDL	BDL
Ant	0.01	0.00120	BDL	BDL	BDL	0.00210	0.00150
Phe	0.001	BDL	0.00009	BDL	BDL	BDL	BDL
Flt	0.001	BDL	BDL	BDL	BDL	BDL	BDL
Pyr	0.001	0.00011	BDL	BDL	BDL	0.00015	0.00004
BaA	0.1	0.00300	BDL	BDL	BDL	0.00600	BDL
Chr	0.001	BDL	BDL	BDL	BDL	BDL	BDL
BbF	0.1	BDL	BDL	BDL	BDL	BDL	BDL
BkF	0.01	BDL	BDL	BDL	BDL	BDL	BDL
BaP	1.0	BDL	BDL	BDL	BDL	BDL	BDL
DBahA	1.0	BDL	BDL	BDL	BDL	BDL	BDL
Ind	0.1	BDL	BDL	BDL	BDL	BDL	BDL
BghiP	0.01	BDL	BDL	BDL	BDL	BDL	BDL
ΣTEQs		0.0045	0.00048	0.00022	0.00031	0.0069	0.0023

Table 8b: Toxic Equivalent Quantities (TEQs) of 16 USEPA PAHs in Water Samples (SSW7–SSW8 and LW1–LW3)

PAHs	TEFs	SSW7	SSW8	LW1	LW2	LW3
NaP	0.001	0.00014	BDL	0.00060	0.00020	0.00017
Acy	0.001	BDL	0.00017	BDL	BDL	BDL
Ace	0.001	0.00030	BDL	BDL	0.00010	0.00021
Fle	0.001	BDL	BDL	BDL	BDL	BDL
Ant	0.01	BDL	BDL	BDL	BDL	BDL
Phe	0.001	BDL	0.00018	0.00007	BDL	BDL
Flt	0.001	BDL	BDL	0.00007	0.00007	BDL
Pyr	0.001	0.00011	BDL	0.00003	0.00003	BDL
BaA	0.1	0.02100	BDL	BDL	BDL	BDL
Chr	0.001	BDL	BDL	BDL	BDL	BDL
BbF	0.1	BDL	BDL	BDL	BDL	BDL
BkF	0.01	BDL	BDL	BDL	BDL	BDL
BaP	1.0	BDL	BDL	BDL	BDL	BDL
DBahA	1.0	BDL	BDL	BDL	BDL	BDL
Ind	0.1	BDL	BDL	BDL	BDL	BDL
BghiP	0.01	BDL	BDL	BDL	BDL	BDL
ΣTEQs		0.023	0.00024	0.00070	0.0013	0.00038



Table 8c: Toxic Equivalent Quantities (TEQs) of 16 USEPA PAHs in Water Samples (LW4, SHDW and BW1–BW3)

PAHs	TEFs	LW4	SHDW	BW1	BW2	BW3
NaP	0.001	0.00031	0.00013	BDL	0.00017	BDL
Acy	0.001	BDL	0.00010	0.00033	0.00025	BDL
Ace	0.001	0.00018	BDL	0.00043	BDL	BDL
Fle	0.001	BDL	0.00013	BDL	0.00011	BDL
Ant	0.01	BDL	BDL	0.00230	BDL	BDL
Phe	0.001	BDL	BDL	BDL	0.00017	BDL
Flt	0.001	0.00050	BDL	BDL	BDL	BDL
Pyr	0.001	BDL	0.00031	BDL	BDL	BDL
BaA	0.1	BDL	BDL	BDL	BDL	BDL
Chr	0.001	0.00028	BDL	0.00015	BDL	BDL
BbF	0.1	BDL	BDL	BDL	0.00800	BDL
BkF	0.01	BDL	BDL	BDL	BDL	BDL
BaP	1.0	BDL	BDL	BDL	BDL	BDL
DBahA	1.0	BDL	BDL	BDL	BDL	BDL
Ind	0.1	BDL	BDL	BDL	BDL	BDL
BghiP	0.01	BDL	BDL	BDL	BDL	BDL
Σ TEQs		0.00097	0.0034	0.00033	BDL	BDL

Note: TEF = Toxic Equivalency Factor; TEQ = Toxic Equivalent Quantity; BDL = Below Detection Limit. TEQs are expressed in $\mu\text{g/mL}$. The summation values (Σ TEQs) are retained exactly as supplied in the original table.

3.5.3 TEQs in Lake Water and Groundwater

The TEQ results for LW4, SHDW and BW1–BW3 are presented in Table 8c. The reported Σ TEQ values were 0.00097 $\mu\text{g/mL}$ for LW4 and 0.0034 $\mu\text{g/mL}$ for SHDW, while BW1–BW3 were reported as BDL.

LW4 showed contributions from NaP (0.00031 $\mu\text{g/mL}$), Ace (0.00018 $\mu\text{g/mL}$), Flt (0.00050 $\mu\text{g/mL}$) and Chr (0.00028 $\mu\text{g/mL}$). Flt was the largest individual reported contributor, indicating that it played an important role in the toxic-equivalent profile of this lake-water sample. The shallow hand-dug well (SHDW) recorded a Σ TEQ of 0.0034 $\mu\text{g/mL}$. The reported individual contributions included NaP (0.00013 $\mu\text{g/mL}$), Acy (0.00010 $\mu\text{g/mL}$), Fle (0.00013 $\mu\text{g/mL}$), Pyr (0.00031 $\mu\text{g/mL}$), Ace (BDL), Flt (BDL), Chr (BDL) and other compounds. The reported total TEQ is therefore substantially higher than the sum of the displayed individual TEQs, indicating that the SHDW calculation should be verified against the underlying concentration and TEF data.

BW1, BW2 and BW3 were reported as BDL for Σ TEQ. However, the individual entries in the supplied table include measurable TEQ values for several compounds in BW1 and BW2. This represents an internal inconsistency that should be resolved before the table is used to support a quantitative conclusion. In particular, the reported Σ TEQ values should be recalculated from the verified individual PAH concentrations and TEFs. The TEQ results therefore indicate substantial spatial variation in the toxic-equivalent composition of the water samples. The results also demonstrate that samples containing relatively high contributions from PAHs with larger TEFs can exhibit elevated TEQs even where their overall concentration contribution is not necessarily dominant.

3.6 Carcinogenic Risk of PAHs

The estimated carcinogenic risks for adults and children are presented in Table 9. The assessment considered two exposure pathways, reported as CR_1 and CR_2 , together with the corresponding total



carcinogenic risk (CR). The results show substantial spatial variation in carcinogenic-risk estimates among the investigated water sources.

For SW1, the adult CR₁, CR₂ and total CR values were 0.00024, 0.0017 and 0.0019, respectively. The corresponding values for children were 0.000093, 0.00053 and 0.00062. Thus, the total reported carcinogenic risk was higher for adults than children at this location under the supplied exposure calculations.

SW2, SW3 and SW4 were reported as BDL for both adult and child carcinogenic-risk estimates. This indicates that the detected concentrations of the carcinogenic PAHs included in the risk calculation

were insufficient to produce a reported carcinogenic-risk estimate under the stated assessment conditions.

SW5 exhibited a higher carcinogenic-risk estimate than SW1. The adult CR₁, CR₂ and total CR values were 0.00048, 0.0035 and 0.0040, respectively, while the corresponding child values were 0.00018, 0.0010 and 0.0012. The total adult CR at SW5 was therefore approximately 0.0040, whereas the corresponding child estimate was 0.0012. These results identify SW5 as a location with a relatively high reported carcinogenic-risk estimate among the surface-water samples.

Table 9: Carcinogenic Risk of PAHs in Surface and Groundwater within and around Gosa Dumpsite

Sample ID	Adult CR ₁	Adult CR ₂	Total CR	Children CR ₁	Children CR ₂	Total CR
SW1	0.00024	0.0017	0.0019	0.000093	0.00053	0.00062
SW2	BDL	BDL	BDL	BDL	BDL	BDL
SW3	BDL	BDL	BDL	BDL	BDL	BDL
SW4	BDL	BDL	BDL	BDL	BDL	BDL
SW5	0.00048	0.0035	0.0040	0.00018	0.0010	0.0012
SW6	BDL	BDL	BDL	BDL	BDL	BDL
SW7	0.0017	0.012	0.014	0.00065	0.0037	0.0043
SW8	BDL	BDL	BDL	BDL	BDL	BDL
LW1	BDL	BDL	BDL	BDL	BDL	BDL
LW2	BDL	BDL	BDL	BDL	BDL	BDL
LW3	BDL	BDL	BDL	BDL	BDL	BDL
LW4	BDL	BDL	BDL	BDL	BDL	BDL
SHDW	0.000064	0.00047	0.00053	0.000024	0.00014	0.00016
BW1	BDL	BDL	BDL	BDL	BDL	BDL
BW2	0.00066	0.0047	0.0054	0.00025	0.0014	0.0017
BW3	BDL	BDL	BDL	BDL	BDL	BDL

Note: BDL = Below Detection Limit; CR = Carcinogenic Risk. CR₁ and CR₂ represent the two exposure pathways reported in the original dataset

SW7 recorded the highest reported carcinogenic-risk values in the dataset. The adult CR₁, CR₂ and total CR were 0.0017, 0.012 and 0.014, respectively. For children, the corresponding values were 0.00065, 0.0037 and 0.0043. The total adult CR of 0.014 was therefore substantially higher than the corresponding values at SW1 and SW5. The elevated carcinogenic-risk estimate at SW7 is

consistent with the relatively high TEQ reported for this sampling location in Table 8b, where BaA made a major contribution to the total TEQ.

The lake-water samples LW1–LW4 were all reported as BDL for carcinogenic risk for both adults and children. Similarly, BW1 and BW3 were reported as BDL. The shallow hand-dug well (SHDW), however, had measurable



carcinogenic-risk estimates, with adult CR₁, CR₂ and total CR values of 0.000064, 0.00047 and 0.00053, respectively. The corresponding child values were 0.000024, 0.00014 and 0.00016.

BW2 also showed measurable carcinogenic-risk estimates. Adult CR₁, CR₂ and total CR values were 0.00066, 0.0047 and 0.0054, respectively, while the corresponding child values were 0.00025, 0.0014 and 0.0017. The total adult carcinogenic risk at BW2 (0.0054) was therefore higher than that observed in SHDW (0.00053), although both locations had measurable risk estimates.

3.6.1 Spatial Variation in Carcinogenic Risk

The carcinogenic-risk results demonstrate marked spatial variability across the study area. Among the surface-water samples, SW7 recorded the highest total adult and child carcinogenic-risk estimates, followed by SW5 and SW1. The remaining surface-water locations were reported as BDL. Among the groundwater samples, BW2 had a higher reported carcinogenic risk than SHDW, whereas BW1 and BW3 were reported as BDL. The relatively elevated carcinogenic-risk estimates at SW7 are particularly important because this location also recorded the highest reported Σ TEQ in Table 8b (0.023 $\mu\text{g/mL}$). The parallel occurrence of elevated TEQ and carcinogenic-risk estimates suggests that the PAH mixture at SW7 had a comparatively greater contribution from compounds considered in the carcinogenic-risk assessment. However, the TEQ and carcinogenic-risk metrics quantify different aspects of toxicity and exposure and should therefore not be treated as interchangeable measures.

The presence of measurable carcinogenic-risk estimates in SHDW and BW2 is also relevant because groundwater sources are used for domestic purposes in communities surrounding the dumpsite. The results indicate that groundwater exposure should be included in

environmental monitoring programmes alongside surface-water assessment.

3.6.2 Comparison of Adult and Child Carcinogenic Risk

The calculated carcinogenic-risk values differed between adults and children. At SW1, SW5 and SW7, the reported total adult CR values were higher than the corresponding child values. The same pattern occurred at SHDW and BW2. For example, at BW2, total CR was 0.0054 for adults compared with 0.0017 for children.

These differences reflect the exposure assumptions and parameters used in the respective risk calculations rather than differences in the measured concentrations themselves. Consequently, interpretation should focus on the exposure scenarios represented by the calculations rather than assuming that one receptor group necessarily has a greater environmental exposure.

The results further demonstrate that carcinogenic risk was concentrated at a limited number of sampling locations. SW7 was the most prominent location in the supplied dataset, with total CR values of 0.014 for adults and 0.0043 for children, while SW2, SW3, SW4, SW6, SW8 and the four lake samples were reported as BDL.

The carcinogenic-risk assessment indicates that measurable carcinogenic-risk estimates occurred mainly at selected surface-water and groundwater locations rather than uniformly throughout the study area. The highest reported value occurred at SW7, while measurable groundwater risk was observed at SHDW and BW2.

These findings suggest that spatial variability in PAH occurrence and composition is important when evaluating potential human-health implications around the Gosa dumpsite. The coincidence of elevated TEQ and carcinogenic-risk estimates at SW7 further identifies this sampling point as requiring particular attention in subsequent monitoring.



However, the interpretation of the absolute magnitude of carcinogenic risk should be made only after confirming the exposure-pathway definitions, toxicity values, concentration units and risk equations used in the calculations. In addition, the apparent inconsistencies between some individual TEQs and their reported Σ TEQs in Tables 8a–8c should be resolved before the final health-risk conclusions are drawn. The calculations in Table 9 should similarly be independently checked against the individual carcinogenic PAH concentrations and the exposure parameters specified in Section 2.9.

3.7 Physicochemical Characteristics of Surface Water, Lake Water and Groundwater

The physicochemical characteristics of the water samples collected within and around Gosa dumpsite are presented in Tables 10a–10d. The parameters measured included pH, temperature, total dissolved solids (TDS), electrical conductivity (EC), turbidity, dissolved oxygen (DO), biochemical oxygen demand (BOD), chloride (Cl^-), fluoride (F^-), sulphate (SO_4^{2-}), and four parameters designated in the supplied dataset as Parameters 10, 12, 13 and 14. These parameters provide useful information on the general physicochemical condition of the aquatic systems and provide environmental context for interpreting the observed distribution of PAHs.

3.7.1 Physicochemical Characteristics of Stream Water

The physicochemical properties of the stream samples SSW1–SSW8 are presented in Tables 10a and 10b. The pH of the stream water ranged from 6.01 ± 0.06 at SSW1 to 7.90 ± 0.07 at SSW8. The lowest pH was recorded at SSW1, whereas the highest was observed at SSW8. Samples SSW1 and SSW2 were mildly acidic, while SSW3–SSW8 were approximately neutral to slightly alkaline based on the reported measurements. The relatively wide pH variation indicates spatial differences

in the physicochemical conditions of the stream system.

Water temperature varied from 28.17 ± 0.10 °C at SSW1 to 33.40 ± 0.04 °C at SSW3. The relatively higher temperatures recorded at SSW2, SSW3, SSW6 and SSW8 may reflect differences in exposure to solar radiation, water depth, flow conditions and local environmental characteristics. Temperature is also relevant to the interpretation of dissolved oxygen because increasing temperature generally affects oxygen solubility and aquatic biochemical processes.

TDS concentrations were consistently high in the stream samples, ranging from $1,850 \pm 0.21$ mg/L at SSW1 to $2,020 \pm 0.06$ mg/L at SSW8. The highest TDS occurred at SSW8, followed by SSW3 ($2,006 \pm 0.14$ mg/L) and SSW4 ($2,003 \pm 0.04$ mg/L), while SSW1 had the lowest value. The relatively elevated TDS values indicate substantial dissolved ionic or particulate-associated material in the stream water. Such conditions may reflect the influence of runoff, leachate inputs, weathering of materials and other anthropogenic activities associated with the dumpsite environment.

The reported EC values ranged from 3.13 ± 0.22 to 4.90 ± 0.08 across SSW1–SSW8, with the highest value occurring at SSW6 and the lowest at SSW1. The spatial pattern broadly corresponds to the elevated dissolved-solids conditions observed in the stream samples. However, the unit reported in the tables as $\mu\text{S}/\text{cm}$ should be verified because the magnitude of the EC values is not readily consistent with the corresponding TDS concentrations if the values are interpreted literally as $\mu\text{S}/\text{cm}$. The original instrument output or laboratory records should therefore be consulted to confirm whether the EC unit is $\mu\text{S}/\text{cm}$ or mS/cm . Turbidity ranged from 57.00 ± 0.03 at SSW1 to 77.00 ± 0.12 at SSW6. SSW6 recorded the highest turbidity, followed by SSW4 (71.00 ± 0.03), SSW5 (69.00 ± 0.06) and SSW3 (67.00 ± 0.05).



Table 10a: Physicochemical Parameters of Surface Water within and around Gosa Dumpsite (SSW1–SSW4)

Parameter	SSW1	SSW2	SSW3	SSW4
pH	6.01 ± 0.06	6.21 ± 0.28	7.30 ± 1.51	6.85 ± 0.11
Temperature (°C)	28.17 ± 0.10	31.22 ± 0.08	33.40 ± 0.04	30.10 ± 0.13
TDS (mg/L)	1850 ± 0.21	1900 ± 0.08	2006 ± 0.14	2003 ± 0.04
EC (μS/cm)	3.13 ± 0.22	3.64 ± 0.15	3.42 ± 0.07	3.36 ± 0.18
Turbidity (mg/L)	57.00 ± 0.03	65.00 ± 0.08	67.00 ± 0.05	71.00 ± 0.03
DO (mg/L)	6.10 ± 0.07	5.58 ± 0.14	6.00 ± 0.21	5.87 ± 0.06
BOD (mg/L)	4.60 ± 0.02	3.93 ± 0.17	3.86 ± 0.05	4.44 ± 0.06
Cl ⁻ (mg/L)	252.00 ± 0.07	294.00 ± 0.13	270.00 ± 0.19	244.00 ± 0.05
F ⁻ (mg/L)	ND	ND	ND	ND
Parameter 10 (mg/L)	7.47 ± 0.31	3.11 ± 0.05	6.80 ± 0.27	8.20 ± 0.13
SO ₄ ²⁻ (mg/L)	8.73 ± 0.11	7.60 ± 0.03	8.10 ± 0.05	7.30 ± 0.06
Parameter 12 (mg/L)	0.62 ± 0.17	0.58 ± 0.05	0.22 ± 0.16	0.51 ± 0.37
Parameter 13 (mg/L)	202.00 ± 0.04	210.00 ± 0.07	196.00 ± 0.05	201.00 ± 0.03

Table 10b: Physicochemical Parameters of Surface Water within and around Gosa Dumpsite (SSW5–SSW8)

Parameter	SSW5	SSW6	SSW7	SSW8
pH	7.40 ± 0.71	7.80 ± 0.03	7.36 ± 0.03	7.90 ± 0.07
Temperature (°C)	30.76 ± 0.07	31.14 ± 0.16	29.35 ± 0.12	32.53 ± 0.04
TDS (mg/L)	1970 ± 0.36	2010 ± 0.08	1880 ± 0.13	2020 ± 0.06
EC (μS/cm)	3.81 ± 0.05	4.90 ± 0.08	3.84 ± 0.36	3.95 ± 0.04
Turbidity (mg/L)	69.00 ± 0.06	77.00 ± 0.12	64.00 ± 0.06	73.00 ± 0.04
DO (mg/L)	5.92 ± 0.05	6.12 ± 0.10	5.41 ± 0.02	5.65 ± 0.11
BOD (mg/L)	3.90 ± 0.04	4.13 ± 0.12	4.37 ± 0.03	3.57 ± 0.23
Cl ⁻ (mg/L)	267.00 ± 0.31	248.00 ± 0.04	291.00 ± 0.02	276.00 ± 0.06
F ⁻ (mg/L)	ND	ND	ND	ND
Parameter 10 (mg/L)	6.71 ± 0.09	5.55 ± 0.35	3.38 ± 0.77	7.40 ± 0.04
SO ₄ ²⁻ (mg/L)	7.27 ± 0.07	7.38 ± 0.41	8.04 ± 0.14	7.86 ± 0.07
Parameter 12 (mg/L)	0.47 ± 0.02	0.64 ± 0.14	0.71 ± 0.03	0.74 ± 0.82
Parameter 13 (mg/L)	192.00 ± 0.04	197.00 ± 0.06	213.00 ± 0.05	221.00 ± 0.07
Parameter 14 (mg/L)	47.00 ± 0.24	36.13 ± 0.20	33.15 ± 0.01	38.14 ± 0.06

Table 10c: Physicochemical Parameters of Lake Water within and around Gosa Dumpsite (LW1–LW4)

Parameter	LW1	LW2	LW3	LW4
pH	6.88 ± 0.33	6.72 ± 0.05	7.57 ± 0.03	7.71 ± 0.15
Temperature (°C)	30.86 ± 0.06	32.60 ± 0.02	30.16 ± 0.11	32.42 ± 0.07
TDS (mg/L)	505 ± 0.66	325 ± 0.03	480 ± 0.06	284 ± 0.10
EC (μS/cm)	0.67 ± 0.08	0.41 ± 0.23	0.52 ± 0.16	0.30 ± 0.13
Turbidity (mg/L)	41.00 ± 0.15	33.00 ± 0.07	37.00 ± 0.03	25.00 ± 0.08
DO (mg/L)	4.90 ± 0.07	4.55 ± 0.05	4.49 ± 0.18	4.81 ± 0.02
BOD (mg/L)	4.02 ± 0.14	3.90 ± 0.03	4.25 ± 0.08	3.88 ± 0.04
Cl ⁻ (mg/L)	21.00 ± 0.04	16.00 ± 0.12	11.00 ± 0.04	24.00 ± 0.08



F ⁻ (mg/L)	ND	ND	ND	ND
Parameter 10 (mg/L)	14.31 ± 0.64	20.18 ± 0.17	19.14 ± 0.06	17.86 ± 0.09
SO ₄ ²⁻ (mg/L)	6.66 ± 0.05	5.72 ± 0.18	6.80 ± 0.27	5.61 ± 0.12
Parameter 12 (mg/L)	0.33 ± 0.03	0.41 ± 0.02	0.38 ± 0.06	0.50 ± 0.02
Parameter 13 (mg/L)	74.00 ± 0.05	81.00 ± 0.03	83.00 ± 0.01	86.00 ± 0.06
Parameter 14 (mg/L)	22.00 ± 0.03	19.10 ± 0.36	21.00 ± 0.82	19.80 ± 0.70

Table 10d: Physicochemical Parameters of Groundwater within and around Gosa Dumpsite (SHDW and BW1–BW3)

Parameter	SHDW	BW1	BW2	BW3
pH	7.85 ± 0.05	6.60 ± 0.18	6.94 ± 0.07	7.69 ± 0.31
Temperature (°C)	31.64 ± 0.17	29.70 ± 0.28	28.68 ± 0.07	32.90 ± 0.13
TDS (mg/L)	180 ± 0.06	191 ± 0.41	211 ± 0.08	377 ± 0.12
EC (µS/cm)	0.27 ± 0.03	0.24 ± 0.77	0.33 ± 0.05	0.47 ± 0.08
Turbidity (mg/L)	31.00 ± 0.04	1.10 ± 0.02	0.80 ± 0.14	1.21 ± 0.19
DO (mg/L)	5.03 ± 0.05	5.12 ± 0.04	5.09 ± 0.07	5.33 ± 0.02
BOD (mg/L)	3.30 ± 0.05	2.52 ± 0.13	2.10 ± 0.04	2.74 ± 0.02
Cl ⁻ (mg/L)	56.00 ± 0.03	165.00 ± 0.05	36.00 ± 0.17	52.00 ± 0.02
F ⁻ (mg/L)	ND	ND	ND	ND
Parameter 10 (mg/L)	1.18 ± 0.21	10.12 ± 0.14	12.65 ± 0.81	7.90 ± 0.03
SO ₄ ²⁻ (mg/L)	15.17 ± 0.08	6.75 ± 0.03	7.20 ± 0.32	6.34 ± 0.11
Parameter 12 (mg/L)	1.91 ± 0.10	0.07 ± 0.04	0.03 ± 0.27	0.06 ± 0.08
Parameter 13 (mg/L)	111.00 ± 0.08	131.00 ± 0.01	127.00 ± 0.07	110.00 ± 0.03
Parameter 14 (mg/L)	22.30 ± 0.44	38.90 ± 0.02	51.60 ± 0.62	44.19

The consistently elevated turbidity values indicate the presence of suspended material in the stream. Such suspended matter may influence the transport and partitioning of hydrophobic contaminants such as PAHs because particulate-associated organic matter can provide surfaces for contaminant sorption. DO concentrations ranged from 5.41 ± 0.02 mg/L at SSW7 to 6.12 ± 0.10 mg/L at SSW6. In contrast, BOD ranged from 3.57 ± 0.23 mg/L at SSW8 to 4.60 ± 0.02 mg/L at SSW1. The coexistence of measurable BOD with moderate DO concentrations indicates active oxygen-consuming processes within the stream. SSW1 had the highest BOD and the highest DO among SSW1–SSW4, whereas SSW7 had the lowest DO and a relatively high BOD of 4.37 ± 0.03 mg/L. These variations suggest spatial differences in the organic matter loading and microbial oxygen demand of the stream.

Chloride concentrations ranged from 244.00 ± 0.05 mg/L at SSW4 to 294.00 ± 0.13 mg/L at

SSW2. SSW2 therefore recorded the highest chloride concentration, while SSW4 had the lowest. Fluoride was not detected (ND) in any of the eight stream samples. Sulphate concentrations showed relatively limited variation, ranging from 7.27 ± 0.07 mg/L at SSW5 to 8.73 ± 0.11 mg/L at SSW1.

The unidentified Parameter 10 ranged from 3.11 ± 0.05 mg/L at SSW2 to 8.20 ± 0.13 mg/L at SSW4, while Parameter 12 ranged from 0.22 ± 0.16 mg/L at SSW3 to 0.74 ± 0.82 mg/L at SSW8. Parameter 13 varied from 192.00 ± 0.04 mg/L at SSW5 to 221.00 ± 0.07 mg/L at SSW8. Parameter 14, which was reported only in Table 10b, ranged from 33.15 ± 0.01 mg/L at SSW7 to 47.00 ± 0.24 mg/L at SSW5. Because the identities of these parameters were not provided, their environmental significance cannot be reliably interpreted until the parameter names are confirmed.

Overall, the stream water exhibited substantial spatial variation in pH, TDS, EC, turbidity,



dissolved oxygen, BOD and ionic constituents. The combination of relatively high TDS and turbidity with measurable BOD indicates that the stream environment receives appreciable dissolved and suspended material. These physicochemical conditions are relevant to the observed PAH distribution because variations in pH, dissolved organic matter, suspended solids and ionic strength can influence contaminant transport, sorption and persistence.

3.7.2 Physicochemical Characteristics of Lake Water

The physicochemical characteristics of lake samples LW1–LW4 are presented in Table 10c. The pH ranged from 6.72 ± 0.05 at LW2 to 7.71 ± 0.15 at LW4. LW1 and LW2 were slightly acidic, whereas LW3 and LW4 were slightly alkaline. Compared with the stream samples, the lake water generally showed a narrower and lower range of TDS and EC.

Temperature ranged from 30.16 ± 0.11 °C at LW3 to 32.60 ± 0.02 °C at LW2. TDS ranged from 284 ± 0.10 mg/L at LW4 to 505 ± 0.66 mg/L at LW1, with LW1 recording the highest concentration. This indicates considerable variation in dissolved material among the lake sampling points. The reported EC values ranged from 0.30 ± 0.13 to 0.67 ± 0.08 , with LW1 exhibiting the highest value and LW4 the lowest. As with the stream samples, the unit assigned to EC should be verified against the original analytical records.

Turbidity ranged from 25.00 ± 0.08 at LW4 to 41.00 ± 0.15 at LW1. The lower turbidity relative to the stream samples suggests comparatively lower suspended-material loading in the lake water. DO ranged from 4.49 ± 0.18 mg/L at LW3 to 4.90 ± 0.07 mg/L at LW1, while BOD varied from 3.88 ± 0.04 mg/L at LW4 to 4.25 ± 0.08 mg/L at LW3. The relatively close DO values across the lake sites indicate less spatial variation in oxygen availability than was observed for some of the stream samples.

Chloride concentrations in the lake were substantially lower than those measured in the streams, ranging from 11.00 ± 0.04 mg/L at LW3 to 24.00 ± 0.08 mg/L at LW4. Fluoride was not detected in any lake sample. Sulphate concentrations ranged from 5.61 ± 0.12 mg/L at LW4 to 6.80 ± 0.27 mg/L at LW3. Parameter 10 ranged from 14.31 ± 0.64 mg/L at LW1 to 20.18 ± 0.17 mg/L at LW2, while Parameter 12 varied from 0.33 ± 0.03 to 0.50 ± 0.02 mg/L. Parameter 13 ranged from 74.00 ± 0.05 mg/L at LW1 to 86.00 ± 0.06 mg/L at LW4, whereas Parameter 14 varied between 19.10 ± 0.36 and 22.00 ± 0.03 mg/L.

The lake samples therefore generally exhibited lower TDS, turbidity and chloride concentrations than the stream samples. This distinction may reflect differences in hydrological connectivity, dilution, sedimentation and the relative influence of runoff or leachate inputs. Nevertheless, the presence of PAHs previously observed in the lake samples demonstrates that lower concentrations of conventional physicochemical indicators do not necessarily imply the absence of organic micropollutant contamination.

3.7.3 Physicochemical Characteristics of Groundwater

The physicochemical characteristics of SHDW and the borehole samples BW1–BW3 are presented in Table 10d. Groundwater pH ranged from 6.60 ± 0.18 at BW1 to 7.85 ± 0.05 at SHDW. BW1 was slightly acidic, while SHDW and BW2–BW3 were approximately neutral to mildly alkaline. The groundwater temperatures ranged from 28.68 ± 0.07 °C at BW2 to 32.90 ± 0.13 °C at BW3.

TDS concentrations ranged from 180 ± 0.06 mg/L at SHDW to 377 ± 0.12 mg/L at BW3. Unlike the stream samples, the groundwater samples exhibited substantially lower TDS concentrations. This pattern suggests that dissolved-solids loading in the groundwater was comparatively lower than in the surface-



water system during the sampling period. However, the occurrence of PAHs in SHDW and BW2 demonstrates that relatively moderate bulk physicochemical characteristics do not preclude contamination by specific hydrophobic organic pollutants.

The reported EC values ranged from 0.24 ± 0.77 at BW1 to 0.47 ± 0.08 at BW3. Again, the units and magnitudes should be verified against the laboratory records before publication. Turbidity showed marked spatial variation, with SHDW recording 31.00 ± 0.04 , compared with only 0.80–1.21 in BW1–BW3. This difference is noteworthy because the shallow hand-dug well may be more directly influenced by surface runoff, sediment disturbance and near-surface materials than the boreholes.

DO ranged from 5.03 ± 0.05 mg/L at SHDW to 5.33 ± 0.02 mg/L at BW3. BOD was comparatively low, ranging from 2.10 ± 0.04 mg/L at BW2 to 3.30 ± 0.05 mg/L at SHDW. The lower BOD values in the groundwater compared with most stream samples indicate lower measured oxygen demand in the groundwater system.

Chloride ranged from 36.00 ± 0.17 mg/L at BW2 to 165.00 ± 0.05 mg/L at BW1. Thus, BW1 showed a markedly higher chloride concentration than the other groundwater locations. Fluoride was not detected in any groundwater sample. Sulphate ranged from 6.34 ± 0.11 mg/L at BW3 to 15.17 ± 0.08 mg/L at SHDW, with SHDW showing the highest concentration. Parameter 10 ranged from 1.18 ± 0.21 mg/L at SHDW to 12.65 ± 0.81 mg/L at BW2, while Parameter 12 ranged from 0.03 ± 0.27 mg/L at BW2 to 1.91 ± 0.10 mg/L at SHDW. Parameter 13 ranged from 110.00 ± 0.03 mg/L at BW3 to 131.00 ± 0.01 mg/L at BW1, while Parameter 14 ranged from 22.30 ± 0.44 at SHDW to 51.60 ± 0.62 mg/L at BW2; the value reported for BW3 was 44.19 without an associated standard deviation.

A comparison of the three water environments shows clear differences in their general

physicochemical characteristics. The streams had the highest TDS and turbidity, the lakes had intermediate TDS and turbidity, while the borehole groundwater generally had the lowest turbidity and relatively moderate TDS. The shallow hand-dug well represented an intermediate condition, particularly because its turbidity (31.00 ± 0.04) was substantially higher than those of the boreholes. These differences are consistent with the greater exposure of surface and shallow groundwater systems to runoff, suspended solids and direct environmental inputs.

3.7.4 Relationship between Physicochemical Conditions and PAH Occurrence

The physicochemical results provide useful environmental context for the spatial distribution of PAHs reported in the preceding sections. The stream samples generally exhibited higher TDS and turbidity than the lake and groundwater samples, while several of the highest PAH concentrations were also recorded in stream water. In particular, the stream locations with substantial PAH burdens were associated with appreciable turbidity and dissolved-solids concentrations. However, the available data alone do not establish a direct causal relationship between these physicochemical variables and PAH concentrations.

The elevated turbidity observed in the streams is potentially important because PAHs are hydrophobic compounds that can associate with suspended particles and organic matter. Consequently, particulate transport may contribute to the movement of PAHs through the surface-water system. The lower turbidity of the borehole samples may partly explain differences in contaminant transport between surface water and deeper groundwater, although the presence of PAHs in SHDW and BW2 demonstrates that contaminant migration to groundwater cannot be excluded.

The pH values across the sampled water bodies were generally within a moderately narrow



range, approximately 6.0–7.9. Such variation may influence the physicochemical behaviour of associated contaminants and the characteristics of dissolved and particulate organic matter. Nevertheless, the present descriptive data should not be interpreted as evidence of a statistically significant pH–PAH relationship unless supported by the correlation analysis presented later in the manuscript.

The oxygen-related parameters also provide evidence of spatial differences in water quality. Stream DO values were generally higher than lake DO values, while BOD was generally higher in the streams and lake samples than in the borehole groundwater. The simultaneous occurrence of measurable BOD and PAHs indicates that the contaminated aquatic environments contain both organic matter subject to biological oxidation and persistent hydrophobic organic contaminants. However, BOD does not specifically indicate PAH biodegradation and should therefore be interpreted as a general indicator of oxygen demand rather than a direct measure of PAH degradation.

Overall, the physicochemical results demonstrate marked differences among the stream, lake and groundwater systems surrounding Gosa dumpsite. The streams were characterized by relatively high TDS and turbidity, the lakes exhibited comparatively lower dissolved-solids and suspended-material loads, and the groundwater generally showed lower turbidity and moderate TDS. These environmental differences provide an important basis for interpreting the spatial distribution, transport and potential fate of PAHs within the study area.

3.8 Diagnostic Molecular Ratio Analysis of PAHs

Diagnostic molecular ratios were used to provide additional information on the possible sources and compositional characteristics of PAHs in the water samples. The results for SSW1–SSW6 are presented in Table 11a, those

for SSW7–LW2 in Table 11b, and those for LW3–BW3 in Table 12c. The interpretation of molecular ratios is based on the relative abundance of selected PAHs; however, the specific equations represented by Diagnostic Ratios 1–6 were not provided in the supplied tables. Consequently, the numerical patterns are described without assigning specific source categories beyond what can be directly supported by the reported values.

3.8.1 Diagnostic Ratios in Stream Water

The diagnostic-ratio results for SSW1–SSW6 (Table 11a) show that several ratios could not be calculated because one or more constituent PAHs were below the detection limit. At SSW1, Diagnostic Ratios 1–5 were reported as BDL because the required PAHs were below detection limits, whereas Diagnostic Ratio 6 had a value of 2.71. Similarly, SSW2 had a Diagnostic Ratio 6 value of 2.91, while Ratios 1–5 were not determinable because the relevant PAHs were BDL.

At SSW3 and SSW4, all six diagnostic ratios were reported as BDL or not determinable. This indicates that the PAH composition at these locations did not contain detectable concentrations of the compounds required for calculation of the specified ratios. The inability to calculate several diagnostic ratios should not be interpreted as evidence of absence of PAHs, since the earlier concentration tables demonstrate that some lower-molecular-weight PAHs were detected at these sites.

SSW5 recorded a Diagnostic Ratio 6 value of 3.42, whereas Ratios 1–5 were not calculable because their component compounds were below detection limits. SSW6 recorded the highest Diagnostic Ratio 6 value among SSW1–SSW6, with a value of 9.50. The progressive variation in this ratio—from 2.71 at SSW1 to 9.50 at SSW6—indicates differences in the relative composition of the PAHs required for calculation of Diagnostic Ratio 6.



The predominance of BDL values for Ratios 1–5 across these stream samples highlights an important limitation of molecular-ratio analysis in the present dataset. Diagnostic ratios are most informative when both numerator and denominator compounds are reliably quantified. Where one or both components are below the detection limit, assigning a source based on the ratio would introduce considerable uncertainty.

3.8.2 Diagnostic Ratios in SSW7, SSW8 and Lake Water

The diagnostic ratios for SSW7, SSW8 and LW1–LW2 are presented in Table 11b. Unlike most of the preceding stream samples, SSW7 had a measurable Diagnostic Ratio 1 of 0.454, while Diagnostic Ratios 2–5 were BDL or not calculable. Diagnostic Ratio 6 was 1.96. Thus, SSW7 provides more quantitative information for molecular-ratio assessment than most of the other stream sites.

SSW8 had all diagnostic ratios reported as BDL. This indicates that the PAHs required for the specified molecular-ratio calculations were either not detected or were below the analytical detection limits at this location.

For the lake samples, LW1 recorded a Diagnostic Ratio 4 value of 0.70, while the other ratios were BDL. LW2 recorded Diagnostic Ratio 4 of 0.87 and Diagnostic Ratio 6 of 0.0097. These results demonstrate that different groups of PAHs were available for ratio calculation at the two lake sites. However, because the equations corresponding to the diagnostic ratios have not been supplied, the numerical differences cannot be translated reliably into particular source classifications.

The limited number of calculable ratios in the lake samples is also consistent with the relatively selective PAH profiles reported previously. For example, some lake samples were dominated by naphthalene or acenaphthene, while several higher-molecular-

weight compounds were below detection limits. This compositional selectivity naturally restricts the number of diagnostic ratios that can be calculated.

3.8.3 Diagnostic Ratios in Lake Water and Groundwater

Table 12c presents the diagnostic ratios for LW3, LW4, SHDW and BW1–BW3. All six ratios were BDL for LW3, indicating that the required PAHs were below detection limits for all specified ratio calculations.

LW4 recorded a Diagnostic Ratio 6 value of 0.32, while Ratios 1–5 were BDL or not calculable. SHDW had a Diagnostic Ratio 6 value of 1.08, with the remaining ratios not calculable. BW1 and BW3 had all diagnostic ratios reported as BDL. BW2 recorded a Diagnostic Ratio 6 value of 4.17, whereas Ratios 1–5 were not calculable.

The measurable Diagnostic Ratio 6 values therefore varied considerably among the locations, ranging from 0.32 at LW4 to 4.17 at BW2 and 9.50 at SSW6. This variation indicates differences in the relative composition of the PAH compounds used in Diagnostic Ratio 6. Nevertheless, the absence of the ratio equations and corresponding interpretation criteria prevents a scientifically defensible assignment of these numerical values to specific petrogenic, pyrogenic, mixed or other source categories.

Overall, the molecular-ratio results show that the application of diagnostic ratios was constrained by the relatively frequent occurrence of BDL values for individual PAHs. The ratios that could be calculated demonstrate spatial variability in PAH composition, but the results should be interpreted together with the concentration profiles, molecular-weight distributions and, where available, statistical correlation analysis.

Table 11a: Diagnostic Molecular Ratios of PAHs in Water Samples (SSW1–SSW6)



Sample ID	Diagnostic Ratio 1	Diagnostic Ratio 2	Diagnostic Ratio 3	Diagnostic Ratio 4	Diagnostic Ratio 5	Diagnostic Ratio 6
SSW1	Phe BDL	Chr BDL	Ind and BghiP BDL	Flt BDL	BaP and BghiP BDL	2.71
SSW2	Ant BDL	BaA and Chr BDL	Ind BDL	Flt BDL	BaP BDL	2.91
SSW3	Phe and Ant BDL	BaA and Chr BDL	Ind and BghiP BDL	Flt BDL	BaP and BghiP BDL	HMW BDL
SSW4	Phe and Ant BDL	BaA and Chr BDL	Ind and BghiP BDL	Flt BDL	BaP and BghiP BDL	HMW BDL
SSW5	Phe and Ant BDL	Chr BDL	Ind and BghiP BDL	Flt BDL	BaP and BghiP BDL	3.42
SSW6	Phe BDL	BaA and Chr BDL	Ind and BghiP BDL	Flt BDL	BaP and BghiP BDL	9.5

Note: BDL = Below Detection Limit; HMW = High Molecular Weight PAHs

Table 11b: Diagnostic Molecular Ratios of PAHs in Water Samples (SSW7–LW2)

Sample ID	Diagnostic Ratio 1	Diagnostic Ratio 2	Diagnostic Ratio 3	Diagnostic Ratio 4	Diagnostic Ratio 5	Diagnostic Ratio 6
SSW7	0.454	BaA and Chr BDL	Ind and BghiP BDL	Flt BDL	BaP and BghiP BDL	1.96
SSW8	Phe BDL	Chr BDL	Ind and BghiP BDL	Flt BDL	BaP and BghiP BDL	BDL
LW1	Phe and Ant BDL	BaA and Chr BDL	Ind and BghiP BDL	0.7	BaP and BghiP BDL	LMW BDL
LW2	Phe and Ant BDL	BaA and Chr BDL	Ind and BghiP BDL	0.87	BaP and BghiP BDL	0.0097

Note: BDL = Below Detection Limit; LMW = Low Molecular Weight PAHs

Table 12c: Diagnostic Molecular Ratios of PAHs in Water Samples (LW3–BW3)

Sample ID	Diagnostic Ratio 1	Diagnostic Ratio 2	Diagnostic Ratio 3	Diagnostic Ratio 4	Diagnostic Ratio 5	Diagnostic Ratio 6
LW3	Phe and Ant BDL	BaA and Chr BDL	Ind and BghiP BDL	Flt and Pyr BDL	BaP and BghiP BDL	HMW BDL
LW4	Phe and Ant BDL	BaA and Chr BDL	Ind and BghiP BDL	Pyr BDL	BaP and BghiP BDL	0.32
SHDW	Phe BDL	BaA BDL	Ind and BghiP BDL	Flt BDL	BaP and BghiP BDL	1.08
BW1	Phe and Ant BDL	BaA and Chr BDL	Ind and BghiP BDL	Flt and Pyr BDL	BaP and BghiP BDL	HMW BDL
BW2	Ant BDL	BaA BDL	Ind and BghiP BDL	Flt and Pyr BDL	BaP and BghiP BDL	4.17
BW3	Phe and Ant BDL	BaA and Chr BDL	Ind and BghiP BDL	Flt and Pyr BDL	BaP and BghiP BDL	LMW and HMW BDL

Note: BDL = Below Detection Limit; LMW = Low Molecular Weight PAHs; HMW = High Molecular Weight PAHs

Table 13: Pearson Correlation of PAHs in Topsoil ($p < 0.05$, *significant)

PAH	BbF	NaP	Acy	Ace	Fle	Ant	Phe	Flt	Pyr	BaA	Chr
BbF	1										



NaP	-0.325	1										
Acy	-0.030	-0.002	1									
Ace	0.335	0.102	-0.325	1								
Fle	0.122	-0.065	-0.314	0.499*	1							
Ant	-0.204	0.486*	-0.234	0.151	0.054	1						
Phe	-0.059	0.043	-0.024	0.054	0.075	0.080	1					
Flt	0.321	0.084	0.135	-0.200	-0.306	-0.251	-0.069	1				
Pyr	-0.362	-0.173	-0.184	-0.201	-0.062	0.020	-0.209	0.355	1			
BaA	-0.325	0.069	-0.131	0.114	-0.032	0.363	0.152	-0.490*	0.403	1		
Chr	0.273	-0.180	-0.022	0.028	0.098	-0.149	0.064	-0.054	-0.245	-0.065	1	

Note: *Correlation is significant at the 0.05 level (2-tailed). BbF = Benzo[b]fluoranthene; NaP = Naphthalene; Acy = Acenaphthylene; Ace = Acenaphthene; Fle = Fluorene; Ant = Anthracene; Phe = Phenanthrene; Flt = Fluoranthene; Pyr = Pyrene; BaA = Benzo[a]anthracene; Chr = Chrysene.

In particular, diagnostic ratios calculated from concentrations close to analytical detection limits should be treated cautiously because small analytical variations can produce relatively large changes in calculated ratios.

3.8.4 Implications for PAH Source Interpretation

The diagnostic-ratio results provide evidence that the PAH composition varies spatially within the Gosa dumpsite water environment. The occurrence of measurable ratios at selected stream, lake and groundwater locations, together with the predominance of BDL ratios at other locations, suggests that PAH composition is not uniform throughout the study area.

The molecular-ratio evidence should, however, be regarded as complementary rather than conclusive evidence of source attribution. The source interpretation would be strengthened if the manuscript explicitly defines each Diagnostic Ratio 1–6, provides the corresponding diagnostic thresholds used for interpretation, and identifies whether the ratios are being interpreted as indicators of petrogenic, pyrogenic or mixed sources. Without these definitions, the numerical values can be reported but cannot be assigned to particular source categories with sufficient methodological transparency.

3.9 Pearson Correlation Analysis of PAHs

Pearson correlation analysis was conducted to examine the relationships among individual PAHs, with statistical significance evaluated at the 0.05 level. The correlation matrix presented in Table 13 contains the relationships among BbF, NaP, Acy, Ace, Fle, Ant, Phe, Flt, Pyr, BaA and Chr. Significant relationships are indicated by an asterisk.

The results show that most pairwise correlations were not statistically significant. The significant correlations identified in the supplied matrix were between Ace and Fle ($r = 0.499$, $p < 0.05$), NaP and Ant ($r = 0.486$, $p < 0.05$), and Flt and BaA ($r = -0.490$, $p < 0.05$). These relationships indicate that the concentrations of some PAH pairs varied together, whereas other compounds exhibited weak or inverse relationships.

The positive correlation between Ace and Fle ($r = 0.499$) represents a moderate positive association. This indicates that locations with relatively higher Ace concentrations tended, within the dataset, to also have relatively higher Fle concentrations. A similar moderate positive relationship was observed between NaP and Ant ($r = 0.486$), indicating that the two compounds exhibited broadly similar spatial variation. Such associations may reflect common transport conditions, shared environmental behaviour or overlapping inputs; however, correlation alone does not establish a common source.



In contrast, Flt and BaA exhibited a moderate negative correlation ($r = -0.490$, $p < 0.05$). Thus, higher concentrations of one compound tended to occur with lower concentrations of the other within the observations included in the analysis. The inverse relationship indicates differential spatial behaviour or input patterns for these two PAHs, although additional source and environmental information would be required to determine the mechanism responsible.

Several other correlations were relatively moderate in magnitude but were not marked as statistically significant. For example, Ace and BbF showed a positive correlation of 0.335, while BbF and Flt had a positive correlation of 0.321. Fle and BaA showed a weak positive relationship ($r = 0.363$), while Pyr and Flt had a positive correlation of 0.355. Conversely, Pyr and BbF showed a negative correlation of -0.362 . Because these relationships were not identified as significant in the supplied table, they should not be interpreted as statistically established associations.

A number of correlations were weak or close to zero. For example, Acy showed very weak relationships with NaP ($r = -0.002$), Phe ($r = -0.024$) and Chr ($r = -0.022$). Similarly, Phe exhibited weak relationships with most of the other PAHs, including BbF ($r = -0.059$), NaP ($r = 0.043$), Ace ($r = 0.054$) and Chr ($r = 0.064$). These weak associations suggest that the spatial variation of these compounds was not strongly synchronized within the observations represented in the correlation matrix.

The correlation matrix therefore suggests that only selected PAH pairs exhibited statistically significant co-variation. The positive Ace–Fle and NaP–Ant relationships may indicate similarities in their spatial distribution, whereas the negative Flt–BaA relationship indicates contrasting variation. However, Pearson correlation should be interpreted as evidence of statistical association rather than direct evidence of common emission sources. Source

attribution should instead integrate these results with the diagnostic molecular ratios, PAH molecular-weight distributions, concentration patterns and environmental characteristics of the sampling locations.

3.9.1 Relationship between Correlation Analysis and Source Assessment

The correlation analysis provides an additional line of evidence for evaluating the distribution of PAHs in the study area. Where two PAHs show significant positive correlation, their concentrations may respond similarly to transport processes or environmental conditions. The significant association between Ace and Fle and between NaP and Ant is therefore consistent with some degree of coordinated spatial variation.

However, the correlation results do not provide a uniformly strong association among the PAHs. The predominance of non-significant correlations indicates that the compounds do not all behave as a single group within the sampled environment. This may reflect differences in physicochemical properties, environmental persistence, partitioning between dissolved and particulate phases, degradation processes or differences in the sources contributing to individual sampling locations.

The significant negative relationship between Flt and BaA further demonstrates that individual PAHs may exhibit contrasting spatial distributions. Such a pattern cautions against interpreting the total PAH concentration as arising from a completely homogeneous source or transport process.

Taken together, the diagnostic-ratio and Pearson-correlation results indicate spatial heterogeneity in PAH composition within the Gosa dumpsite water environment. Nevertheless, definitive source apportionment requires explicit diagnostic-ratio equations and interpretation criteria, together with consideration of the environmental setting and other source indicators. The present statistical



results are therefore best interpreted as supporting evidence for spatially differentiated PAH behaviour rather than as standalone proof of specific source categories.

4.0 Conclusion

This study assessed the levels, distribution, possible sources, ecological implications and human-health risks of polycyclic aromatic hydrocarbons (PAHs) in surface water, lake water and groundwater within and around Gosa dumpsite, Abuja, Nigeria. The results demonstrated that PAHs were present in several of the investigated water bodies, although their concentrations and composition varied considerably among sampling locations. Stream water generally exhibited higher overall PAH burdens than the lake and groundwater systems, indicating substantial spatial heterogeneity in the distribution of these contaminants within the dumpsite environment.

The PAH profiles showed differences in the relative contributions of low- and high-molecular-weight compounds among sampling locations. Several stream samples were characterized by substantial contributions from low-molecular-weight PAHs, while selected locations showed appreciable contributions from higher-molecular-weight compounds. The observed spatial variability suggests that PAH occurrence is influenced by differences in contaminant inputs, transport and environmental conditions within the study area. The presence of PAHs in the shallow hand-dug well and some borehole samples also indicates that contamination is not restricted to surface waters and that groundwater resources around the dumpsite may be exposed to contaminant migration.

The toxic-equivalent results further demonstrated spatial differences in the contribution of individual PAHs to the overall toxicity potential. The highest reported total TEQ occurred at SSW7 (0.023), followed by SSW5 (0.0069), while several other locations

had substantially lower TEQ values or reported values below detection limits. Individual compounds, particularly benzo[a]anthracene at selected sampling locations, made substantial contributions to the calculated TEQs. These results indicate that assessment based solely on total PAH concentration may not adequately represent the relative toxicological significance of individual compounds.

The human-health risk assessment showed that both adults and children were exposed to carcinogenic and non-carcinogenic risks through the evaluated exposure pathways. Although many of the reported hazard indices were below unity, several individual PAHs produced relatively high hazard quotients, with some values approaching unity. The carcinogenic-risk assessment showed measurable risks at several locations, with the highest total adult carcinogenic risk reported at SSW7 (0.014), followed by BW2 (0.0054), SSW5 (0.0040) and SSW1 (0.0019). Corresponding total risks for children were 0.0043, 0.0017, 0.0012 and 0.00062, respectively. The consistently higher calculated risks for adults in the reported tables indicate that the evaluated exposure assumptions produced greater adult risk estimates at these locations.

The physicochemical characteristics revealed marked differences among the stream, lake and groundwater systems. Stream samples generally recorded higher TDS and turbidity than lake and groundwater samples, while groundwater samples generally exhibited lower turbidity and moderate TDS concentrations. These variations provide an environmental context for contaminant transport and partitioning within

5.0 References

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- The authors declare that there is no known conflict of interest or personal relationships that have influence the work reported in this paper.
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Author Contributions

Henrietta Ijeoma Kelle conceived the study, designed the methodology, coordinated sampling, data analysis, and manuscript preparation. Joseph Itah Omada contributed to sample collection, laboratory analysis, interpretation, and manuscript review. Emeka Chima Ogoko supported environmental

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