**Occurrence of Polycyclic Aromatic**

**Hydrocarbons in the Waters from the Déganobo Lacustrine System**

**(Côte d’Ivoire): Contamination Levels, Sources, Ecological and Health Risks**

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ABSTRACT

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| This study aimed to assess the contamination levels of the waters in the Déganobo lake system by the sixteen polycyclic aromatic hydrocarbons (PAHs) designated as priority pollutants by the U.S. Environmental Protection Agency. Furthermore, it sought to determine their origins within these waters and evaluate the associated ecological and human health risks. To achieve this, 96 water samples were collected from the lake system over a one-year period, from August 2021 to July 2022, and subsequently analyzed in accordance with . The origin of PAHs in these waters was determined using molecular weight ratios of specific compounds. Contamination levels were assessed following the SEQ-Eau V2 guidelines. Ecological and human health risks were estimated using Directive 2013/39/EU, the hazard index, benzo[a]pyrene toxic equivalent quotients (TEQ-BaP), benzo[a]pyrene mutagenic equivalent quotients (MEQ-BaP), the incremental risk-related BaP dose (RI-BaP dose), and incremental lifetime cancerogenic risks. Phenanthrene was found to be the most abundant PAH in these waters, with an average concentration of (10.3±1.5) µg/L, while benzo(j)fluoranthene was the least abundant, at (0.02±0.01) µg/L. Low molecular weight PAHs (2 to 3 benzene rings) predominated, reaching (20.2±2.8) µg/L, whereas high molecular weight PAHs were less represented, at (2.1±0.1) µg/L. The origin of PAHs in these waters was primarily pyrolytic. The water quality was deemed poor, mainly due to the presence of benzo(b)fluoranthene ((1.7±0.3) µg/L) and indeno(1,2,3-c,d)pyrene ((0.5±0.1) µg/L). Overall, the associated health risks, particularly carcinogenic and mutagenic effects of these PAHs on the ecosystem's biota and human health, were found to be elevated. |

*Keywords: Côte d’Ivoire, City of San-pédro, Hydrocarbons, Lake pollution, Organic pollution, Surface*

*waters.*

1. INTRODUCTION

Naturally occurring in the environment (Fan et al., 2025), hydrocarbons have gained increasing prominence within numerous ecosystems since the advent of the Second Industrial Revolution, spanning from the late 19th to the early 20th century. This escalation is attributed to the rapid development of new technologies, the emergence of innovative industrial sectors, and the widespread adoption of transport modes predominantly powered by petroleum and its derivatives (Fan et al., 2025; Delcourt and Rocle, 2017). The substantial presence of these organic compounds in ecosystems constitutes a major source of ecological and health risks (Elobaid et al., 2025; Jin et al., 2025).

This concern is particularly salient for Polycyclic Aromatic Hydrocarbons (PAHs). These compounds, defined by the presence of at least two fused aromatic rings, are characterized by their low biodegradability, rendering them highly persistent in the environment. Due to this persistence, PAHs exhibit a strong capacity for bioaccumulation in living organisms and biomagnification throughout the food chain, thereby exposing all species, including humans, to serious toxicological risks. Their carcinogenic and mutagenic potential is now well-established (Moon et al., 2024; Qiao et M., 2025), designating them as priority pollutants requiring vigilant monitoring across various ecosystems. In this regard, the US-EPA (1982) identified a list of 16 PAHs deemed priority for environmental surveillance. These compounds include: acenaphthene (ACP), acenaphthylene (ACPT), anthracene (ANT), benzo(a)anthracene (BaA), benzo(a)pyrene (BaP), benzo(b)fluoranthene (BbF), benzo(k)fluoranthene (BkF), benzo(g,h,i)perylene (B(g,h,i)P), chrysene (CHRY), dibenzo[a,h]anthracene (D[a,h]A), fluorene (FLR), fluoranthene (FLRT), indeno[1,2,3-c,d]pyrene (I(1,2,3-c,d)P), naphthalene (NPT), phenanthrene (PNT), and pyrene (PYR).

Recent studies have highlighted a relatively high level of surface water pollution by these 16 PAHs, raising significant ecological and health implications. Probing examples include the work of Amine et al. (2024), Chen et al. (2025), and Wang et al. (2024). Broadly, these investigations reveal that the distribution and fate of PAHs in surface waters are contingent upon several determining factors. These encompass the geographical environment, the nature and intensity of industrial activities, the volume of road traffic, surface runoff, the physicochemical properties of water and sediments, biological activity, and hydroclimatic conditions (Cerciello et al., 2025; Wang et al., 2024). The distribution of PAHs in surface waters is also notably influenced by their number of benzene rings, as emphasized by Chen et al. (2025).

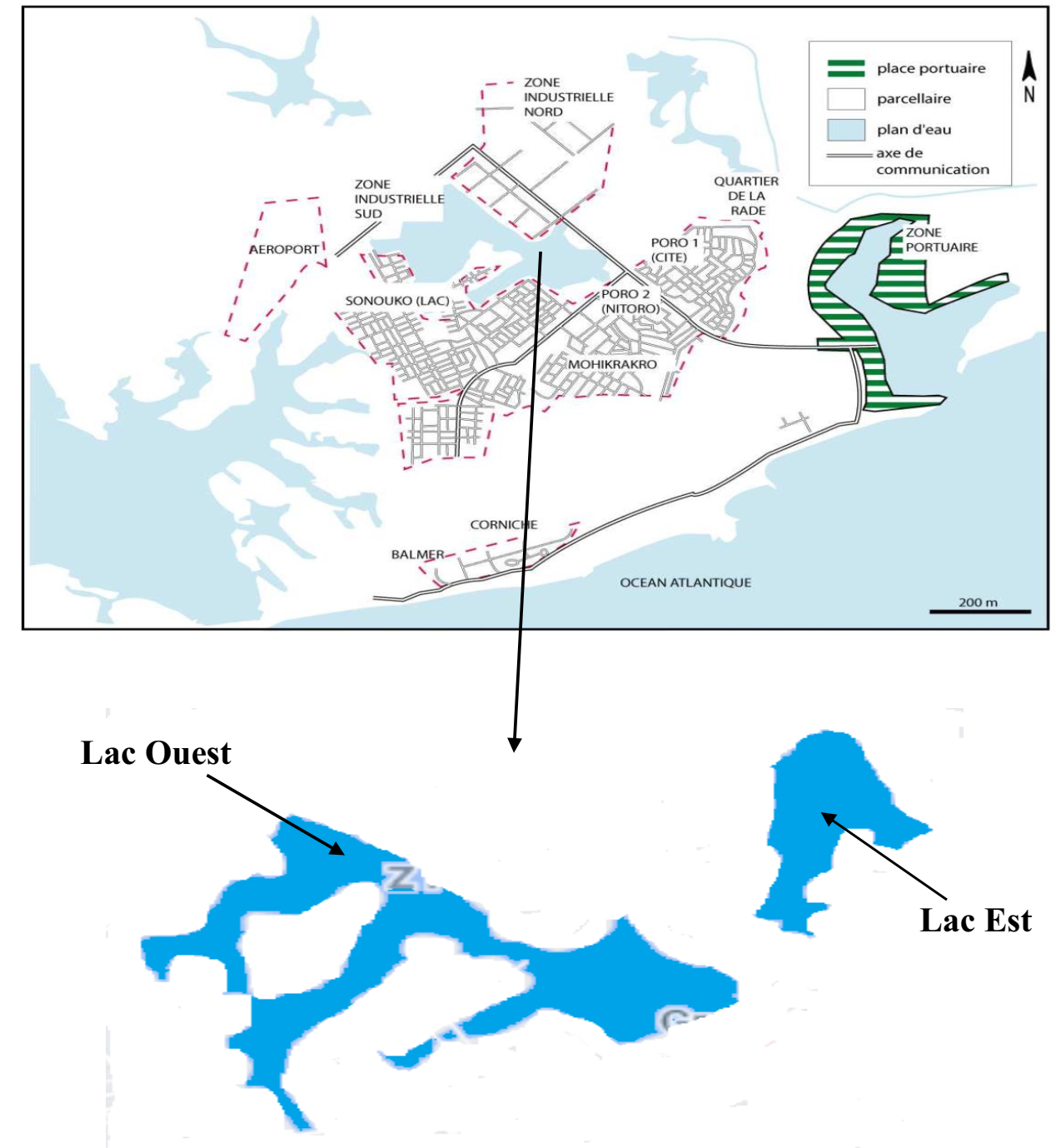
Hydrocarbon Pollution in Côte d'Ivoire: Focus on PAHs in the Déganobo Lake System

In Côte d'Ivoire, initial investigations into surface water pollution by hydrocarbons in general, and by PAHs in particular, along with their ecological and health implications, commenced in the 1970s with the work of Binet and Marchal (1970). This was followed by Marchand and Martin (1985), who focused on sediments in the Ébrié Lagoon. Currently, several recent scientific studies have addressed this form of pollution within Ivorian aquatic ecosystems. Among these, the works of Konan et al. (2020), Gnonsoro et al. (2023), and Irié et al. (2024), all conducted in the Ébrié Lagoon, have highlighted its alarming condition concerning priority PAHs. This type of research urgently needs to be extended to other unexplored Ivorian surface waters. The Déganobo lacustrine system, for instance, represents a vital aquatic ecosystem for the city of San-Pédro due to its self-purification capacity, socioeconomic significance, and rich biodiversity. However, this ecosystem is subject to severe anthropogenic pressures. One consequence of these pressures is its significant chemical pollution (Konan and Yao, 2023a), particularly from pesticides (Konan and Yao, 2023b), with serious implications for four fish species (Konan and Yao, 2025). Therefore, in-depth investigations in this direction are crucial. It is within this context that the present study was conducted. Its primary objective is to assess the level of PAH contamination in the waters of this lake system and to determine its ecological and health implications. Secondary objectives include determining the concentrations of the 16 PAHs deemed priority by the US-EPA (1982), thereby inferring water quality and evaluating carcinogenic and mutagenic risks for its biota and, by extension, for the local population dependent on it.

2. material and methods

**2.1. Study area description**

This lacustrine system is connected to the Digboué River to the west and the San-Pédro River to the east via extensive marshlands. The construction of the San-Pédro autonomous port in 1972, in immediate proximity to this lacustrine system, led to the establishment of two major industrial zones (North and South) along its banks (Fig. 1). Over the years, the expansion of these zones and the development of industrial activities, primarily agro-industry, have significantly polluted this lacustrine system. Furthermore, these developments have also contributed to the reduction of its surface area through landfilling. This pollution is notably attributed to intense atmospheric emissions and waters discharges from industrial units (Ogou and Bidi, 2019). The expansion of this port and increased traffic also contribute to this degradation (SGS Côte d’Ivoire, 2016; Kadio, 2023). The proliferation of automotive repair shops and intense road traffic adjacent to this aquatic ecosystem are additional contributing factors (Kouassi et al., 2017). Finally, hospital waste discharges and the incineration of waste at the San-Pédro municipal landfill further exacerbate the situation (Konan and Yao, 2023a,b; Koko et al., 2018).

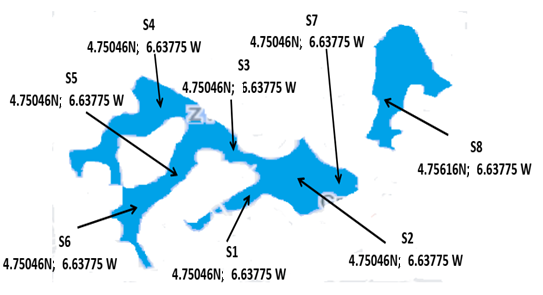


**Fig. 1. Geographical location of the study area (SGS Côte d'Ivoire, 2016).**

**2.2. Experimental techniques**

**2.2.1. Sampling collection**

This study was conducted in parallel with the work of Konan and Yao (2024a, b; 2025). It spanned a one-year period, from August 2021 to July 2022. During this timeframe, monthly sampling of water and superficial sediment samples was performed within the lacustrine system. Sampling was carried out at eight stations, selected to account for spatial heterogeneity and accessibility to the water body (Fig. 2).



**Fig. 2 : Geographical location of sampling stations used during the study period (Konan and Yao, 2024a,b; 2025).**

Given the shallow depth of the study area and to ensure optimal measurement precision, water samples were collected in duplicate from the water column at depths ranging from -0.5 to -1.5 m at each station. A 1-liter Niskin bottle was used for this purpose. After collection, the water was transferred, filling to the brim, into 1-liter borosilicate amber bottles equipped with a stopper featuring a Teflon-lined inner face. These containers were then hermetically sealed and stored at approximately 4°C. Each month, 16 water samples were collected, resulting in a total of 192 samples over the entire study period. In the laboratory, water samples were immediately stored in a refrigerator at 4°C. All these operations adhered to the recommendations of standard FD T90-52-523-4 (2023).

**2.2.2. Determination of 16 PAHs in Water Samples**

The sixteen PAHs classified as priority pollutants by the US-EPA (1982) were analyzed in the water samples. These included Acenaphthene ACP), Acenaphthylene (ACPT), Anthracene (ATC), Benzo[a]anthracene (BaA), Benzo[a]pyrene (BaP), Benzo[b]fluoranthene (BbF), Benzo[g,h,i]perylene (B(g,h,i)P), Benzo[k]fluoranthene (BkF), Benzo[j]fluoranthene (BjF), Chrysene (Chry), Fluoranthene (FLRT), Fluorene (FLR), Naphthalene (NPT), Phenanthrene (PNT), Pyrene (PYR), and Indeno[1,2,3-cd]pyrene (I(1,2,3-c,d)P). These analyses were performed following standard MA. 400 – PAH 1.1 (CEAEQ, 2024).

To prevent cross-contamination, all equipment used for the analysis was meticulously decontaminated using appropriate solvents, specifically acetone and dichloromethane. All glassware was cleaned by immersion in a specialized detergent bath for several hours. It was then rinsed sequentially with tap water, bi-distilled water, and finally Milli-Q ultrapure water, following a cascade rinsing protocol. After oven-drying at 60 °C, the glassware was rinsed with acetone using a wash bottle and then allowed to air dry. Clean equipment was finally stored hermetically sealed, either stoppered or wrapped in aluminum foil.

The following were used:

* Internal standards: Naphthalene-d₈ (CAS: 1146-65-2); Fluoranthene-d₁₀ (CAS: 93951-69-0); Benzo[a]pyrene-d₁₂ (CAS: 63466-71-7) at 10 ng/µL.
* Surrogate standard : Pyrene-d₁₀ (CAS: 1718-52-1) at 10 ng/µL.

*2.2.2.1. Extraction*

An 800 mL aliquot of homogenized water sample was introduced into a 1 L amber glass bottle. This was then fortified with 50 µL of a combined 10 ng/µL surrogate standard solution and 100 mL of dichloromethane. The bottle was temporarily sealed and manually agitated to homogenize the mixture, then briefly opened to release excess pressure. Following this, it was hermetically sealed and placed on a rotary shaker set at approximately 54 rpm for prolonged overnight agitation. After extraction, the mixture was transferred to a 1 L separatory funnel. The organic phase was collected and dried by passing it through a column packed with anhydrous sodium sulfate. The extract was then collected in a 250 mL evaporation flask. Concentration was performed under vacuum using a rotary evaporator, with the bath temperature controlled at ≤ 26 °C, until a final volume of approximately 2 mL was obtained.

*2.2.2.2. Final Preparation for GC-MS Analysis*

To the concentrated extract, 250 µL of isooctane was added, and the solution was then evaporated under a gentle stream of nitrogen to a volume of approximately 300 µL. Finally, 50 µL of a combined 10 ng/µL internal standard solution was added, and the final volume was adjusted to 500 µL with isooctane.

*2.2.2.3. GC-MS Analysis*

PAH analysis was performed using an Agilent 7890B gas chromatography system coupled to an Agilent 7000C triple quadrupole mass spectrometer (GC-MS/MS), equipped with an Agilent 7683A ALS automatic injector (Agilent Technologies, Santa Clara, CA). Chromatographic separation was achieved on an HP-5MS capillary column (30 m × 0.25 mm × 0.25 µm; Agilent J&W). High-purity helium (≥ 99.999%) was used as the carrier gas at a constant flow rate of 1.5 mL/min. Injections were performed in splitless mode, introducing 1 µL of sample at 240 °C. Mass spectrometric analysis was conducted in electron ionization (EI) mode at 70 eV. Temperatures were set at 280 °C for the transfer line, 230 °C for the ion source, and 150 °C for the quadrupole analyzer. A solvent delay of 4 minutes was applied. The GC oven temperature program was as follows:

* 50 °C for 2 min;
* Ramp at 5 °C/min to 155 °C, held for 2 min;
* Ramp at 3 °C/min to 280 °C, held for 2 min.

The total analysis time was 69 min.

Acquisition was performed in Multiple Reaction Monitoring (MRM) mode, with two specific transitions selected per analyte (precursor → product). The first transition was used for quantitative analysis, while the second provided structural confirmation. This acquisition mode offers enhanced selectivity, reducing potential interferences and significantly improving sensitivity compared to conventional SIM mode. Quantification was based on the peak area ratio (PAH / internal standard).

The calibration range was established from 0.01 to 5.00 ng/µL, distributed over seven concentration levels: 0.01 – 0.05 – 0.10 – 0.50 – 1.00 – 2.50 – 5.00 ng/µL. Each solution was prepared by successive dilution of a 10 ng/µL intermediate standard mixture in isooctane. A deuterated internal standard solution was added at a constant concentration of 0.5 ng/µL to each vial. Linearity of the curves was verified for each compound across the entire range with a correlation coefficient r² ≥ 0.995, in accordance with the requirements of method MA.400 – PAH 1.1. Limits of detection (LOD) and limits of quantification (LOQ) were calculated from the signal-to-noise ratio (3:1 for LOD; 10:1 for LOQ). The obtained LOQ value was 0.03 µg/L. Each calibration level was injected at least in duplicate, and the calculated coefficients of variation (%CV) were all less than 15%, ensuring good stability of the analytical system. PAH recovery rates were evaluated using blank samples fortified with surrogate standards. The obtained values ranged between 91% and 109%, adhering to the international acceptability range of 80% to 120% for trace compounds in environmental matrices.

**2.2.3. Determination of the origin**

Several ratios of PAH molecular weights are commonly employed to ascertain their origins in water and sediment samples. Notably, the ANT/(ANT + PHEN) and FLR/(FLR + PYR) ratios are frequently utilized.

For the ANT/(ANT + PHEN) ratio:

- a value of < 0.1 indicates a petrogenic origin for PAHs;

- a value of > 0.1 suggests a pyrogenic origin for PAHs;

- a value of < 0.4 points to a petrogenic source.

As for the FLR/(FLR + PYR) ratio:

- value between 0.4 and 0.5 indicates that the PAHs originate from petroleum-derived fuels;

- a value of > 0.5 suggests that the PAHs stem from the incomplete combustion of organic matter (e.g., coal, vegetation);

- a value of < 0.2 signifies a petrogenic origin (Howard et al., 2021).

**2.2.4. Determination of contamination levels of water by 16 PAHs**

The SEQ-Eau V2 guidelines (MEDD et Agence de l'eau, 2003) are utilized to define water quality based on the presence of polycyclic aromatic hydrocarbons (PAHs), including the 16 priority PAHs identified by the US-EPA (1982). These guidelines were employed for this purpose in the present study.

**2.2.5. Ecological and Health risk assessment**

*2.2.5.1. Ecological and human health risk assessment according to Environmental Quality Standards (EQS) defined by 2013/39/EU*

The EQS, as defined by Directive 2013/39/EU (OJEU, 2013), establish the risk levels for acute toxicity to both humans and aquatic biota. These levels are determined based on the Maximum Annual Concentrations (MACs) of specific PAHs in inland surface waters, notably ANT, BaP, BbF, BkF, FLR, NPT, and I(1,2,3-c,d)P. Risks become significant when concentrations exceed these thresholds.

Similarly, chronic risks, for both humans and aquatic biota, are assessed against the Annual Average Concentrations (AACs) of certain PAHs, specifically ANT, BaP, BbF, BkF, FLR, and NPT.

*2.2.5.2. Carcinogenic and mutagenic risks assessment for aquatic biota*

TEQ-BaP (Toxic Equivalency Quotient for Benzo[a]pyrene) and MEQ-BaP (Mutagenic Equivalency Quotient for Benzo[a]pyrene) indices are commonly used to estimate the potential carcinogenic and mutagenic risks to aquatic biota.

TEQ-BaP represents the carcinogenic toxicity equivalent potential of a given PAH relative to BaP for a specific biota. For a PAH i, it is calculated from its concentration (Ci) and its toxic equivalency factor (TEFi), according to the following formula:

Total TEQ-BaP for a mixture of PAHs is calculated as follows:

As for MEQ-BaP, it represents the mutagenic toxicity equivalent potential of a given PAH relative to BaP for a specific biota. For a PAH i, it is calculated from its concentration (Ci) and its mutagenic equivalency factor (MEFi), according to the following formula:

Total MEQ-BaP for a mixture of PAHs is calculated as follows:

These indices are typically calculated using PAHs of high molecular weight (HMW) and medium molecular weight (MMW), specifically BaA, BaP, BbF, BkF, and I[1,2,3-c,d]P. The TEF values for these specific PAHs are as follows (in µg/L): 0.1 for BaA, 1 for BaP, 0.1 for BbF, 0.1 for BkF, and 0.1 for I(1,2,3-c,d)P. The MEF values for these same PAHs are (in µg/L): 0.082 for BaA, 1 for BaP, 0.11 for BbF, 0.31 for BkF, and 0.25 for I(1,2,3-c,d)P (ATSDR, 2022)..

If the TEQ-BaP and MEQ-BaP values for a specific PAH are less than 1, the associated carcinogenic and mutagenic risks to aquatic biota are considered negligible. Conversely, if these values are greater than or equal to 1, the risks become significant. This same logic applies to the total TEQ-BaP and total MEQ-BaP values (ATSDR, 2022; Ezike et al., 2019; Hur et al., 2025).

*2.2.5.3. Human health risk assessment*

This section evaluates the short- and long-term human health risks associated with the presence of PAHs in the waters from this lacustrine system, considering frequently observed anthropogenic activities. Fishing is a daily activity for children and adolescents, particularly those aged 9 to 11, as well as for adults. Additionally, market gardening is practiced along the water body's banks. Adults also use this lacustrine system for bathing. Therefore, this study assessed the health risks related to PAH presence in this lacustrine system for children aged 9-11 and adults, accounting for their specific activities. Only dermal contact risks were considered, as inhalation and ingestion risks directly from the water body were deemed negligible.

Short-term risks evaluated for these individuals included non-carcinogenic risks and the carcinogenic and mutagenic risks posed by certain PAHs. Long-term health risk assessment primarily focused on lifetime carcinogenic risks associated with specific PAHs. The exposure variables detailed in Table 1 were utilized for these assessments.

**Table 1. Exposure variables used for the assessment of short- and long-term human health risks via dermal contact with water for the different categories of individuals considered in this study**.

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
| Exposure variables | Individuals | | | References |
| Child | Adult | |
| Market gardener | Bathing |
| Body weight (kg) (BW) | 35 | 70 | 70 |  |
| Dermal exposure area (cm2) (SA) | Hands  720 | Hands and feet  2,440 | Total body  21,000 | US-EPA, 2015 |
| Daily frequency/year (days/year) (EF) | 225/year | 365/year | 365/year |  |
| Annual exposition duration  (ED) (year) | 6 | 30 | 30 | Adeniji et al., 2019 |
| Daily exposition time (hour/day) (ET) | 6h/jour | 6h/jour | 0,5/jour |  |
| Event frequency (FE) | 1 event by year | | | US-EPA, 2005) |
| Dermal adsorption fraction (AC) (mg/cm2) | 0.13 | | | US-EPA, 2005) |
| Dermal adsorption rate (AR) | 0.02 mg/event | | | US-EPA, 2005 |
| Conversion factor | 0.001 | | | US-EPA, 2005 |
| Average exposure duration (AT) (AT = ED x 365) (days) | 2,190 | 10,950 | 10,950 | US-EPA, 2005 |
| Average lifetime exposure for the calculation of lifetime carcinogenic risk (AT) (days) | 365 x 58 years = 21,170 | | | US-EPA, 2005 |

*2.2.5.3.1 Short-term health risk assessment*

*2.2.5.3.1.1. Non-carcinogenic risk assessment*

The Hazard Index (HI) is used to assess non-carcinogenic risks to humans resulting from exposure to various chemical pollutants. It is derived from the Daily Exposure Dose (DED), which estimates or approximates the amount of pollutant received by an individual relative to their body weight.

For pollutant exposure via dermal contact, the expression for the DEDderm,i (in mg/kg/day) for a pollutant i is as follows:

where: Ci is the pollutant concentration (µg/l); Kp is the dermal permeability

coefficient (cm/h).

The non-carcinogenic risk value from dermal contact for humans, referring to a given pollutant (HQi), is expressed as:

The non-carcinogenic risk value for a mixture of pollutants (HI) is consequently derived using the following expression:

Among the 16 PAHs identified as priority by the US-EPA (1982), 13 exhibit the capacity to permeate the dermal barrier. However, only 5 have a dermal Reference Dose (RfDderm,i) value. These include ATC (0.04 mg/kg/day), FLR (0.04 mg/kg/day), FLRT (0.04 mg/kg/day), NPT (0.02 mg/kg/day), and (PNT) (0.04 mg/kg/day). Their respective dermal permeability coefficients (Kp) (cm/h), used to determine their daily dermal exposure (DEDderm,i), are: 3.46×10−4 for ATC; 6.26×10−3 for FLR; 0.36 for FLRT; 0.069 for NPT; and 0.27 for PNT (Hopf et al., 2019; US-EPA, 1992a,b).

When HI (HQ) < 1, no carcinogenic risk is identified in relation to the sum of pollutants (or a specific pollutant). Conversely, when HI (HQ) > 1, it is not possible to draw an absolute conclusion regarding the sum of pollutants (or a given pollutant), but the analyzed matrix appears to present a potential hazard to humans (Adenji et al., 2019; US-EPA, 1992a,b).

*2.2.5.3.1.2. Assessment of carcinogenic and mutagenic Risks at BaP-equivalent doses*

The carcinogenic and mutagenic risks of certain of the 16 PAHs to humans utilizing the waters of this lacustrine system for the aforementioned activities were assessed using the RI-BaP dose indices (TEQ and MEQ). This index is calculated from the BaP equivalent dose (BaP-TEQ ou BaP-MEQ) of the PAHs. Thus, for a given PAH i, the expression for the carcinogenic (mutagenic) BaP equivalent dose (mg/kg/day) is:

From this expression, the RI-BaP dose (carcinogenic or mutagenic) for this PAH is derived according to the following formula:

where: SFBaP equivalent dose is the slope factor of BaP equivalent dose, equal to

25 (mg/kg/day)−1 (Knafla et al., 2006); TEQ and MEQ values of PAH in water.

Finally, RI-BaP dose for a mixture of PAHs (Total RI-BaP dose) is expressed as:

If the carcinogenic (mutagenic) RI-BaPi dose values for a specific PAH are less than 1, the associated carcinogenic and mutagenic risks to human are considered negligible. Conversely, if these values are greater than or equal to 1, the risks become significant. This same logic applies to the total carcinogenic (mutagenic) RI-BaP dose values (ATSDR, 2022).

*2.2.5.3.2. Lifetime carcinogenic risk assessment*

Lifetime carcinogenic risks associated with dermal contact with certain PAHs (RIderm) in humans were determined based on the estimation of LADDderm and ILCRderm indices. ILCRderm represents the probability of an individual developing cancer over their lifetime due to dermal exposure to a potential carcinogen (US-EPA, 1989). For a given PAH i, LADDderm follows the same expression as DEDderm, with the exception that the AT value was replaced by 21,170 days, considering the life expectancy in Côte d'Ivoire, which was 58 years in 2023 (World Bank, 2023). The expressions for ILCRderm for a single PAH and RIderm for a mixture of PAHs are then established as follows:

Where CSF is the cancer slope factor of PAH i.

These risks were assessed considering 13 of the 16 PAHs, as Kp values for BjF, BkF, and D(a,h)A were unavailable. The same applies to the CSF for Chry. Thus, the Kp values (cm/h) for these PAHs are: 6.3×10−3 for ACPT; 3.46×10−4 for ATC; 0.81 for BaA; 1.2 for BaP; 1.2 for BbF; 1.2 for B(g,h,i)P; 0.81 for Chry; 6.26×10−3 for FLR; 0.36 for FLRT; 0.069 for NPT; 0.27 for PNT; 1.9 for I(1,2,3-c,d)P; 1.69×10−3 for PYR (Hopf et al., 2019; US-EPA, 1992a,b).

The CSF values were: 0.0073 for ACPT, FLR, FLRT, PNT, and PYR; 0.073 for ATC; 0.73 for BaA and I(1,2,3-c,d)P; and 7.3 for BaP (US-EPA, 1992b; US-EPA, 2009).

The regulatory guidance from the New York State Department of Health (NYS DOH, 2007) indicates that for ILCR (RIderm) values:

- ≤10−6, the lifetime carcinogenic risks of the PAH (or mixture of PAHs) for humans are

very low;

- 10−6 to 10−4, the lifetime carcinogenic risks of the PAH (or mixture of PAHs) for humans

are low;

- >10−4 to 10−3, the lifetime carcinogenic risks of the PAH (or mixture of PAHs) for

humans are moderate;

- >10−3 to 10−1, the lifetime carcinogenic risks of the PAH (or mixture of PAHs) for

humans are high;

- ≥10−1, the lifetime carcinogenic risks of the PAH (or mixture of PAHs) for humans are

very high.

**2.2.6. Statistical analysis of results**

Statistical analysis was performed using one-way ANOVA and Student's t-test, complemented by descriptive techniques including the mean (m), standard deviation (s), coefficient of variation (CV), as well as minimum (Min) and maximum (Max) values.

One-way ANOVA was used to assess for statistically significant differences between the annual means of the 16 PAHs across various matrices. When a significant difference was observed, three post-hoc tests were applied to identify the differentiating PAHs and determine homogeneous subgroups that did not exhibit significant differences. These tests included:

- Fisher's LSD test (Fisher, 1919) ;

- Tukey's HSD test (Tukey, 1949) ;

- Dunnett's test (Dunnett, 1955)

Student's t-test (Gosset, 1908) was applied to assess the statistical significance of the obtained results. This test allowed for the comparison of observed values against reference thresholds for ecological and health risk indices, as well as against the water quality Guidelines used.

Results from ANOVA one way and post-hoc tests, as well as Student's t-test, were considered statistically significant at *p* < 0.05. All these analyses were performed using Statistica software, version 7.

3. results and discussion

**3.1. Results**

**3.1.1. PAHs Abundance in the waters from this lacustrine system**

*3.1.1.1. Annual concentrations of the 16 PAHs*

ACP, Chry, and D[a,h]P were not detected in these waters during the study period. Among the thirteen PAHs that were detected, BjF exhibited the lowest mean concentration, while PNT showed the highest mean concentration throughout the study. At first glance, NPT appeared to be the least abundant, whereas PNT was the most prevalent among the six low molecular weight (LMW) PAHs. Regarding the medium molecular weight (MMW) PAHs, BjF seemed the least represented, while PYR was notably abundant in these waters during this period. Finally, among the high molecular weight (HMW) PAHs, BaA appeared to be the most abundant, whereas I-(1,2,3-c,d)P presented the lowest mean concentration within these waters throughout the study period. The variations in the annual mean concentrations of these PAHs were all low, with the exception of BkF (Table 2). This suggests their near-stability in these waters throughout the entire study period.

**Table 2. Annual mean concentrations (µg/l) of 16 PAHs in the Waters from the Déganobo lacustrine system during the Study Period (n=96).**

|  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- |
| Statistical parameters | LMW PAHs | | | | | |  |
| NPT | ACPT | FLR | PNT | ATC | FLRT | ACP |
| m±s | 0.4±0.1 | 0.8±0.1 | 2.3±0.3 | 10.3±1.5 | 1.8±0.2 | 4.7±0.6 | < LOD |
| VC (%) | 33.9 | 14.3 | 14.2 | 1.5 | 13.1 | 13.7 |  |
| Min-Max | 0.2-0.9 | 0.5-1.2 | 1.3-3.6 | 6.1-16.5 | 1.0-2.8 | 2.7-7.6 |  |
| Statistical parameters | MMW PAHs | | | | | |  |
| PYR | BbF | BjF | BkF | Chry |  |  |
| m±s | 3.4±0.5 | 1.7±0.3 | 0.02±0.01 | 0,4±0,1 | < LOD |  |  |
| VC (%) | 13.7 | 17.1 | 56.9 | 18,1 |  |  |  |
| Min-Max | 2.0-5.3 | 1.1-2.8 | 0.00-0.06 | 0,3-0,6 |  |  |  |
| Statistical parameters | HMW PAHs | | | | | |  |
| BaA | BaP | I(1,2,3-cd)P | D[a,h]P |  |  |  |
| m±s | 0.9±0.2 | 0.7±0.1 | 0.5±0.1 | < LOD |  |  |  |
| VC (%) | 17.4 | 16.5 | 16.2 |  |  |  |  |
| Min-Max | 0.6-1.6 | 0.4-1.2 | 0.3-0.9 |  |  |  |  |

One-way ANOVA revealed overall statistically significant differences among the annual mean concentrations of these PAHs in the water during the study period. Post-hoc tests conducted subsequently showed that only the annual mean concentration of BkF was not statistically different from that of NPT (*p* > .05). The same applied to the difference between the annual mean concentrations of NPT and I-(1,2,3-c,d)P (*p* > .05). Furthermore, the annual mean concentration of FLR showed no statistically significant difference compared to the annual mean concentrations of BaA (*p* > .05) and BaP (*p* >0.05). A similar observation was made for the differences between the annual mean concentration of I-(1,2,3-c,d)P and those of BkF (*p* > .05) and BaP (*p* > .05). Finally, the difference between the annual mean concentrations of BaP and BaA was not statistically significant (*p* > .05).

Thus, the ascending order of importance for the annual mean concentrations of these PAHs over the period is established as follows:

- NPT < ACPT < ATC < FLR < FLRT < PNT for LMW PAHs;

- BjF < BkF < BbF < Pyr for MMW PAHs;

- BaA ≃ BaP ≃I-(1,2,3-c,d)P for HMW PAHs;

- BjF < NPT = BkF ≃ BaA ≃ BaP ≃I-(1,2,3-c,d)P ≃ ACPT < BbF = ATC < FLR < PYR < FLRT < PNT for the 13 PAHs.

presents a significant concentration in these waters. This is also the case for , representing over 70%. . showed the lowest annual concentration. Variations in the annual concentrations of the sum of the different molecular weights of PAHs were low (Table 3), confirming the near-stability of these PAHs in the water throughout this period. One-way ANOVA and the post-hoc tests used in this study highlighted statistically significant differences among the annual mean concentrations of the different molecular weights (*p* < .05).

**Table 3. Annual mean concentrations (µg/l) of different molecular weight PAHs in these waters during the study period (n=96).**

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
| Statistical parameters |  |  |  |  |
| m±s | 20.2±2.8 | 5.5±0.8 | 2.1±0.3 | 27.8±3.9 |
| VC (%) | 13.8 | 14.5 | 14.7 | 13.9 |
| Min-Max | 12.0-32.5 | 3.3-8.7 | 1.3-3.7 | 16.5-44.9 |

**3.1.2. Distribution of PAHs in waters based on the number of benzene rings**

FLR, PNT, PYR, and BaA exhibited, on average, the highest proportions among PAHs with 2, 3, 4, and 5 benzene rings, respectively. Conversely, NPT, ATC, BjF, and I(1,2,3-c,d)P showed the lowest proportions among PAHs with 2, 3, 4, and 5 benzene rings, respectively, in these waters during the study period (Fig.3).

|  |  |
| --- | --- |
|  |  |
| 1. PAHs with 2 benzene rings | 1. PAHs with 3 benzene rings |
|  |  |
| 1. PAHs with 4 benzene rings | 1. PAHs with 5 benzene rings |

**Fig. 3. Mean proportions of PAHs Relative to by number of benzene rings in the waters from the Déganobo lacustrine system during the study period.**

PAHs with three aromatic rings were the most abundant, while those with five aromatic rings were the least abundant in these waters during this period (Fig. 4).

**Fig. 4. Mean proportions of PAHs by number of benzene rings in the Waters from the Déganobo lacustrine system during the study period.**

**3.1.3. Origins of PAHs in the waters from this lacustrine system**

The ATC/(ATC + PNT) ratio, with a mean value of 0.147±0.008, indicates that all 13 PAHs present in these waters during the study period are of pyrogenic origin, meaning they result from combustion (Figure 5a). This conclusion is corroborated by the FLR/(FLR + PYR) ratio, which has a mean value of 0.397±0.009. This latter ratio reveals that 46.9% of the studied PAHs originate from fuel combustion (Figure 5b). These findings were statistically confirmed by Student's t-test (*p* < .05).

a

b

**Fig. 5. Values of the ATC/(ATC +PNT) ratio (a) and of FLR/(FLR+PYR) ratio.**

**3.1.4. PAH contamination levels in waters**

Based on the annual mean concentrations of ACPT, ATC, BaP, BkF, FLR, and FLRT, these waters demonstrated moderate suitability for biological life and anthropogenic uses. Conversely, their quality was deemed poor due to high annual concentrations of BbF and I-(1,2,3-c,d)P. However, the water quality was good concerning the annual mean concentration of NPT, consistent with the guidelines of SEQ-Eau V2 (MEED and Agence de l'eau, 2003) (Table 4). These observations were statistically confirmed by Student's t-test (*p* < .05).

The contamination levels of these waters were moderate relative to ACPT, ATC, BaP, BkF, FLR, and FLRT, but very significant for BbF and I-(1,2,3-c,d)P. In conclusion, these waters are of poor quality for supporting biological life and for all anthropogenic uses.

**Table 4. Quality of the waters from the Déganobo lacustrine system according to SEQ-Eau V2 guidelines (MEDD and Agence de l'eau, 2003) during the study period.**

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| PAHs | Concentration thresholds (µg/l) | | | | Annual concentration in the waters from this lacustrine system (µg/L) | Water quality during this period |
| Very good | Good | Moderate | Poor |
| ACPT | < 0.04 | 0.04-0.4 | 0.4-99 | > 99 | 0.8±0.1 | Moderate |
| ATC | < 0.09 | 0.09-0.9 | 0.9-21 | > 21 | 1.8±0.2 | Moderate |
| BaA | < 0.0005 | 0.0005-0.005 | 0.005-1.2 | > 1.2 | 0.9±0.2 | Moderate |
| BaP | < 0.005 | 0.005-0.05 | 0.05-1.2 | > 1.2 | 0.7±0.1 | Moderate |
| BbF | < 0.0001 | 0.0001-0.001 | 0.001-0.3 | > 0.3 | 1.7±0.3 | Poor |
| BkF | < 0.0003 | 0.0003-0.003 | 0.003-0.8 | > 0.8 | 0.4±0.1 | Moderate |
| FLR | < 0.03 | 0.03-0.3 | 0.3-77 | > 77 | 2.3±0.3 | Moderate |
| FLRT | < 0.0024 | 0.0024-0.024 | 0.024-6 | > 6 | 4.7±0.6 | Moderate |
| I(1,2,3-cd)P | < 0.00016 | 0.00016-0.0016 | 0.0016-0.4 | > 0.4 | 0.5±0.1 | Poor |
| NPT | < 0.19 | 0.19-1.9 | 1.9-460 | > 460 | 0.4±0.1 | Good |
| PNT | < 0.011 | 0.011-0.11 | 0.11-27 | > 27 | 10.3±1.5 | Moderate |
| PYR | < 0.0024 | 0.0024-0.024 | 0.024-6 | > 6 | 3.4±0.5 | Moderate |

**3.1.5. Health risks associated with PAHs in these waters**

*3.1.5.1. Health risks associated with PAHs for biota and human*

The Environmental Quality Standards (EQS) defined by Directive 2013/39/EU (OJEU, 2013) indicate that the ecotoxicity and toxicity risks for the biota of this aquatic ecosystem, as well as for humans, were acute in the short term and chronic in the long term. This was due to the observed concentrations of ATC, BbF, BkF, and FLRT during this period. Risks were also acute in the short term when considering the maximum concentration of I-(1,2,3-c,d)P (Table 5). These observations were confirmed by Student's t-test (*p* < .05).

**Table 5. Toxicity Levels of the waters from the Déganobo lacustrine system related to some PAHs during the study period.**

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| PAHs | 1MACs  (µg/l) | 2AACs (µg/l) | Maximum concentration in waters (µg/l) | Mean concentration in the waters | Acute health risks | Chronic health risks |
| ATC | 0.1 | 0.1 | 2.8 | 1.8±0.2 | Severe | Severe |
| BaP | 0.0017 | 0.27 | 1.2 | 0.7±0.1 | Very low | Very low |
| BbF | 0.0017 | 0.017 | 2.8 | 1.7±0.3 | Severe | Severe |
| BkF | 0.0017 | 0.017 | 0.6 | 0.4±0.1 | Severe | Severe |
| FLRT | 0.063 | 0.12 | 7.6 | 4.7±0.6 | Severe | Severe |
| NPT | 2 | 130 | 0.9 | 0.4±0.1 | Very low | Very low |
| I(1,2,3-c,d)P | 0.0017 | - | 0.9 | 0.5±0.1 | Severe |  |

1Maximum Annual Concentrations (MACs); 2Annual Average Concentrations (AACs)

*3.1.5.2. Ecotoxicity risks of some PAHs to the aquatic biota of this lacustrine system*

Among the HMW PAHs evaluated in these waters, BaP exhibited the highest individual mean values for BaP-TEQ and BaP-MEQ indices, whereas BbF showed the lowest values. The mean BaP-TEQ and BaP-MEQ values for all five of these PAHs were greater than 1 (Table 6), an observation confirmed by Student's t-test (*p* < .05).

Referring to a value of 1 for BaP-TEQ and BaP-MEQ, the carcinogenic and mutagenic risks to the aquatic biota associated with the annual mean concentration of BaP in these waters were moderate. In contrast, these risks remained low when considering the concentrations of BaA, BbF, BkF, and I-(1,2,3-c,d)P. However, the carcinogenic and mutagenic risks for the aquatic biota associated with the entire group of these five PAHs were high for the aquatic fauna during the study period.

**Table 6. Mean BaP-TEQ and BaP-MEQ values for some HMW PAHs in aquatic fauna from the waters from the Déganobo lacustrine system during the study period.**

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
|  | Ci x TEFi (µg/l) | | | | | Total BaP-TEQ (µg/l) |
|  | BaA | BaP | BbF | BkF | I(1,2,3)P |
| m±s | 0.09±0.02 | 0.7±0.1 | 0.17±0.03 | 0.025±0.007 | 0.051±0.008 | 1.1±0.2 |
| Min-Max | 0.06-0.16 | 0.4-1.2 | 0.11-0.28 | 0.025-0.063 | 0.03-0.09 | 0.6-1.8 |
|  | Ci x MEFi (µg/l) | | | | | Total BaP-MEQ (µg/l) |
| m±s | 0.07±0.01 | 0.7±0.1 | 0.18±0.03 | 0.11±0.02 | 0.13±0.02 | 1.2±0.2 |
| Min-Max | 0.05-0.13 | 0.4-1.2 | 0.11-0.30 | 0.08-0.20 | 0.07-0.22 | 0.7-2.0 |

*3.1.5.3. Health Risks for Humans*

*3.1.5.3. Short-Term Risks*

*3.1.5.3.1. Short-Term Non-Carcinogenic Risks*

As indicated in Table 7, the mean DEDderm and HQderm values for children associated with ATC, FLR, FLRT, NPT, and PNT were lower than those for adults. Adults engaged in market gardening and fishing activities showed lower mean DJEderm and HQderm values related to these PAHs compared to individuals using these waters for bathing and swimming.

Short-term carcinogenic risks for children remained very low, as evidenced by their mean HIderm value being less than 1, confirmed by Student's t-test (*p* < 0.05). Conversely, for adults, the various mean HIderm values were greater than 1 (Table 7), also confirmed by Student's t-test (*p* < .05). While these values don't rule out short-term non-carcinogenic risks, they do indicate that using these waters for these purposes poses a hazard to adults.

**Table 7. Mean DEDderm, HQderm, and HIderm values for different individuals based on anthropogenic**

**activities practiced in the waters from the Déganobo lacustrine system during the study period.**

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| PAHs | Child  (Fisherman) | | Adult | | | |
| Fisherman and market gardener | | Bather | |
| DEDderm (mg/kg/day) | HQderm | DEDderm  (mg/kg/day) | HQderm | DEDderm  (mg/kg/day) | HQderm |
| ATC | 4.7x10-6 | 0.0001 | 1.3x10-6 | 0.0003 | 9.2x10-6 | 0.0002 |
| FLR | 0.0001 | 0.003 | 0.0003 | 0.007 | 0.0002 | 0.005 |
| FLRT | 0.01 | 0.3 | 0.04 | 0.9 | 0.03 | 0.6 |
| NPT | 0.0002 | 0.01 | 0.0006 | 0.03 | 0.0004 | 0.02 |
| PNT | 0.02 | 0.5 | 0.06 | 1.5 | 0.04 | 1.0 |
| m±s | 0.008±0.009 | 0.2±0.2 | 0.03±0.03 | 0.5±0.7 | 0.02±0.02 | 0.3±0.5 |
| Min-Max | 4.7x10 -6-0.02 | 0.0001-0.5 | 1.3x10-6-0.06 | 0.0003-1.5 | 9.2x10-6-0.04 | 0.0002-1.0 |
| **HIderm** | **0.8** | | **2.4** | | **1.6** | |

*3.1.5.3.2. Health risks associated with BaP equivalent dose*

The mean values for TEQ-BaPi and MEQ-BaPi for different individuals, considering water uses related to BaA, BaP, BbF, BkF, and I(1,2,3-c,d)P (except for the mean TEQ-BaP values of BkF and I-(1,2,3-c,d)P for children), as well as their resultant values (TEQ-BaP dose and MEQ-BaP dose), were found to be significantly greater than 1. The same applied to RI-BaP dose (TEQ) and RI-BaP dose (MEQ) (Table 8). These observations were corroborated by Student's t-test (*p* < .05).

These results clearly demonstrate that the carcinogenic and mutagenic risks associated with dermal contact with these waters, due to the presence of these five PAHs, were very significant for the individuals studied.

**Table 8. Mean values of TEQ-BaP dose, MEQ-BaP, RI-BaP (TEQ) dose, and RI-BaP (MEQ) for different individuals related to five PAHs in the Waters from the Déganobo lacustrine system during the study period.**

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| PAHs | TEQ-BaP dosei (x 10-7) (µg/l) | | | MEQ-BaP dosei (x 10-7) (µg/l) | | |
| Child  (Fisherman) | Adult | | Child  (Fisherman) | Adult | |
| Fisherman and market gardener | Bather | Fisherman and market gardener | Bather |
| BaA | 1.1±0.2 | 15±3 | 9±2 | 93±16 | 12.8±0.2 | 7±1 |
| BaP | 8±1 | 122±20 | 69±11 | 8±1 | 122±20 | 69±11 |
| BbF | 2.1±0.4 | 28±5 | 16±3 | 2.2±0.4 | 31±5 | 18±3 |
| BkF | 0.46±0.08 | 6±1 | 3.5±0.6 | 1.4±0.3 | 20±4 | 11±2 |
| I(1,23-c,d) | 0.6±0.1 | 8±1 | 4.9±0.8 | 1.6±0.3 | 22±4 | 12±2 |
| **TEQ-BaP dose (x 10-6) (µg/l)** | **1.3±0.2** | **18±2** | **1.0±0.2** |  |  |  |
| **RI-BaP dose (TEQ) (x 10-5) (µg/l)** | **3.3±0.5** | **45±7** | **25±4** |  |  |  |
| **MEQ-BaP dose (x 10-7) (µg/l)** |  |  |  | **15±2** | **208±31** | **117±18** |
| **RI-BaP dose (MEQ) (x 10-5) (µg/l)** |  |  |  | **3.8±0.5** | **51±8** | **29±4** |

*3.1.5.3. Lifetime carcinogenic risks*

For child, ACPT, ATC, FLR, NPT, PNT, and PYR presented very low lifetime carcinogenic risks (mean ILCRderm values < 10−6). In contrast, risks associated with BaA, FLRT, and I(1,2,3-c,d)P were moderate (mean ILCRderm values between 10−4 and 10−3), while those linked to BaP and BbF were high (mean ILCRderm values between 10−3 and 10−1). This finding is confirmed by Student's t-test (*p* <0.05). Regarding adults engaged in fishing and market gardening in and around these waters, the carcinogenic risks associated with ACPT, ATC, FLR, FLRT, PNT, and PYR were very low (mean ILCRderm values <10−6). Those related to BaA and PNT were moderate (mean ILCRderm values between 10−3 and 10−1), whereas risks due to BaP, BbF, and I-(1,2,3-c,d)P were high (mean ILCRderm values between 10−3 and 10−1). These results were also confirmed by Student's t-test (*p* < .05). Finally, for adults using these waters for bathing and swimming, the long-term carcinogenic risks associated with ACPT, ATC, FLR, NPT, and PYR were very low (mean ILCRderm values < 10−6). Those linked to FLRT and PNT were moderate (mean ILCRderm values between 10−3 and 10−1), while risks associated with BaA, BaP, BbF, and I-(1,2,3-c,d)P were high (mean ILCRderm values between 10−3 and 10−1). This finding is also confirmed by Student's t-test (*p* < .05).

The overall carcinogenic risks associated with all these PAHs combined were found to be extremely high (mean RIderm values >10−1) for all individuals, regardless of the anthropogenic use of these waters (Table 9). This observation was particularly pronounced in adults, as confirmed by Student's t-test (*p* < .05).

**Table 9: Mean values of LADDderm, ILCRderm, and RIderm for different individuals based on anthropogenic activities practiced in the waters from the Déganobo lacustrine System during the study period.**

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| PAHs | Child  (Fisherman) | | Adult | | | |
| Fisherman and market gardener | | Bather | |
| LADDderm  (x 10-4)  (mg/kg/day) | ILCRderm  (x 10-4) | LADDderm  (x 10-4)  (mg/kg/day) | ILCRderm  (x 10-4) | LADDderm  (x 10-4)  (mg/kg/day) | ILCRderm  (x 10-4) |
| ACPT | 0.04 | 2x10-4 | 5.4 | 0.004 | 0.4 | 0.003 |
| ATC | 0.005 | 3x10-4 | 0.07 | 0.005 | 0.05 | 0.004 |
| BaA | 5.8 | 4.3 | 80.1 | 58.4 | 57.4 | 41.9 |
| BaP | 6.8 | 49.3 | 92.8 | 677.3 | 66.5 | 485.7 |
| BbF | 15.7 | 11.4 | 215.4 | 157.2 | 154.5 | 112.8 |
| FLR | 0.1 | 8x10-4 | 1.5 | 0.01 | 1.1 | 0.008 |
| FLRT | 13.2 | 9.6 | 181.4 | 10-4 | 130.1 | 0.9 |
| NPT | 0.2 | 0.002 | 3.1 | 0.02 | 2.2 | 0.02 |
| PNT | 21.9 | 0.2 | 301.4 | 2.2 | 216.2 | 1.6 |
| I(1,2,3-cd)P | 7.6 | 5.5 | 104.2 | 76.1 | 74.8 | 54.6 |
| PYR | 0.05 | 3x10-4 | 0.6 | 0.005 | 0.5 | 0.003 |
| m±s | 6.5±7.6 | 7.3±14.5 | 89.6±103.8 | 88.3±20107 | 63,9±74,7 | 63.4±144.6 |
| Min-Max | 0.05-21.9 | 2x10-4-49.3 | 0.07-301.4 | 10-4-677.3 | 0,05-216,2 | 0.003-485.7 |
| **RI (x 10-4)** | **80.3** | | **971.2** | | **697.5** | |

**3.2. Discussion**

The concentrations of these 13 PAHs and their sum (∑13 PAHs) in the waters from this lacustrine system were higher than those reported by numerous recent studies. This was particularly true for the mean concentrations of ACPT, ATC, FLRT, and NPT in these waters during the study period, which exceeded those reported by Dovonou et al. (2019) in the waters from Sô River, where concentrations were 0, 0.01, 0, and 0 ng/L, respectively. Similarly, the concentrations of ATC (0.0-0.3 ng/l), BaA) (0.0-0.8 ng/l), BkF (0.0-2.0 ng/L), FLRT (0.3-117 ng/L), FLR (0.0-1.6 ng/L), NPT (3.7-59.2 ng/L), and PNT (0.0-20.6 ng/l) reported by Yang et al. (2024) for the period 2017-2022 in experimental lakes in Northwest Ontario (Canada) were also lower. The annual mean ∑13PAHs concentration obtained in the present study for these waters also surpassed the ∑16PAHs (the 16 PAHs deemed priority by the US-EPA (1982)) concentrations in the waters from urban lakes Longyang (26.95-36.5 ng/l) and Moshui (46.8-198.25 ng/l) in China (Zhang et al., 2025a). This is also the case when compared to ∑16PAHs concentrations in the transitional waters from three rivers (El Bared, Kebir, Abu Ali) in Northern Lebanon, which ranged between 115 and 600 ng/L (Amine et al., 2024), and also the mean concentration of 1.338 µg/L in the waters from Lake Baiyangdian (Northern China) (Zhang et al., 2025b).

These comparisons highlight high PAH pollution in the waters of this lacustrine system throughout the study period, particularly from the 13 detected PAHs. This situation was notably emphasized by the SEQ-Eau V2 guidelines (MEDD et Agence de l'eau, 2003) for BbF and I-(1,2,3-c,d)P. This contamination is primarily attributable to this lacustrine ecosystem proximity to two major industrial zones and the autonomous port of San-Pédro city. Furthermore, these waters receive PAH discharges from numerous non-modern automobile garages located along its banks, intense automobile traffic within the city, and domestic activities of riparian households. These industrial zones include many chemical and agro-industrial units. The industrial impact on surface water pollution by HAPs has been documented by various recent studies, including those by Amine et al. (2024) on the waters from Tripoli port (Northern Lebanon), where the mean ∑16 PAHs concentrations ranged from 800 to 8000 ng/L; Wang et al. (2024) in the waters from the Kuye River (China), with mean ∑16PAHs concentrations fluctuating between 299.34 and 10,390.9 ng/L; Olivares-Rubio and Ponce-Vélez (2025), who observed mean ∑16PAHs concentrations oscillating between 90,910 ± 13,000 and 131,500 ± 9,900 ng/L in the coastal waters from the Dos Bocas maritime terminal (Southern Gulf of Mexico); and Chen et al. (2025) in the waters from the Yangtze River (China), with a mean ∑16PAHs concentrations varying from 72 to 335 ng/L.

The high proportion of three-ring PAHs, followed by that of four-ring PAHs, observed in these waters confirms their pyrolytic origin throughout the study period, as demonstrated by the ATC/(ATC + PNT) and FLR/(FLR + PYR) ratios obtained in this research. This pyrolytic origin is primarily attributable to industrial emissions and vehicle exhaust, as previously mentioned. Indeed, Chen et al. (2025) noted that the strong presence of 2- to 3-ring PAHs in water is essentially linked to biomass combustion and also petroleum leakage. This is particularly relevant for agro-industrial factories, notably those in San-Pédro city, which utilize biomass and fuel oil. The low proportion of HMW PAHs in these waters, especially those originating from vehicle exhaust where they are abundant, is explained by their strong affinity for the particulate phase (Chen et al., 2025; Li et al., 2023). The effects of industrial emissions on air pollution in San-Pédro city have been documented by Ogou and Bidi (2019). The strong presence of LMW PAHs and MMW PAHs in waters associated with industrial emissions and vehicle exhaust has also been reported by Wang et al. (2024) in the Kuye River (China) and by Chen et al. (2025) in the Yangtze River (China).

The strong presence of three-ring PAHs in these waters is attributed to their relatively high solubility and significant atmospheric emissions within the study area. This is particularly true for FLRT (INERIS, 2015) and, more specifically, PNT (Olivares-Rubio and Ponce-Vélez, 2025), which exhibited the highest concentrations during this period. Although four-ring PAHs have lower solubility than three-ring PAHs, their solubility and substantial emission in the study area are nonetheless sufficient to facilitate their strong presence in these waters. This is notably the case for PYR, which recorded the highest annual mean concentration among four-ring PAHs in these studies. The low concentration of BjF, the lowest among all studied PAHs, is explained by its reduced emission in this study area. Furthermore, the lower mean proportion of five-ring PAHs, compared to three- and four-ring PAHs, is due to the progressive decrease in their solubility as the number of benzene rings increases (Camargo et al., 2024). Generally, the distribution and apportionment of PAHs in surface waters depend on several factors, including the geographical environment, the nature and intensity of industrial activities, the volume of road traffic, surface runoff, the physicochemical properties of the water and sediments, and hydroclimatic conditions (Cerciello et al., 2025; Chen et al., 2025). Thus, the low mean proportion of two-ring PAHs, compared to three- and four-ring PAHs, is explained by their high volatility (Cerciello et al., 2025). This observation also holds true for the non-statistically significant differences between the annual mean concentrations of NPT, BkF, BaA, BaP, I(1,2,3-c,d)P, and ACPT, as highlighted by one-way ANOVA. The same applies to the non-statistically significant difference between the annual mean concentrations of ATC and BbF, also revealed by one-way ANOVA.

The synthesis of results obtained from the Environmental Quality Standards (EQS) of Directive 2013/39/EU (OJEU, 2013), as well as the TEQ-BaP and MEQ-BaP indices, revealed acute and chronic risks for the aquatic fauna of this ecosystem. These risks were particularly high for ATC, BbF, BkF, and FLRT. Furthermore, the strong presence of I(1,2,3-c,d)P in these waters is likely to cause acute risks for this fauna. As for the carcinogenic and mutagenic risks associated with the combined group BaA, BaP, BbF, BkF, and I(1,2,3-c,d)P, they were found to be significant over the study period. Thus, the high concentrations of ATC, BbF, BkF, and FLRT in this lacustrine system can lead to immune system disruptions, physiological dysfunctions, and reproductive disorders in aquatic biota, particularly algae, invertebrates, and fish. These compounds can also bioaccumulate and biomagnify (Albarano et al., 2023; INERIS, 2020; Siquiera et al., 2023). The combined group of BaA, BaP, BbF, BkF, and I-(1,2,3-c,d)P can induce tumors in various aquatic species, both through dermal and internal exposure. This group of PAHs can cause genetic mutations in these species, including chromatid abnormalities, micronuclei, or other DNA damage. Such multi-genic damage increases the carcinogenic risks for this biota (Abouda et al., 2024; Kuhn et al., 2025).

A synthesis of results, also derived from the Environmental Quality Standards (EQS) of Directive 2013/39/EU (OJEU, 2013) and the RI-BaP dose (TEQ and MEQ), HIderm, and RIderm indices, revealed high short-term risks, with the exception of carcinogenic and mutagenic risks, linked to ATC, BbF, BkF, and FLRT. These waters proved to be hazardous, even though they did not present short-term carcinogenic risks for adult anthropogenic activities, particularly fishing and bathing, in relation to the combined PAHs ATC, FLR, FLRT, NPT, and PNT. However, this finding does not apply to children engaged in fishing in these waters. The short-term carcinogenic and mutagenic risks for the different categories of individuals linked to BaA, BaP, BbF, BkF, and I(1,2,3-c,d)P), as well as to the totality of these PAHs, were found to be very low. Conversely, long-term carcinogenic risks, considering eleven of the thirteen studied PAHs, were extremely high for individuals using these waters. This was particularly true for BaP and BbF in children engaged in fishing, and for BaA, BaP, BbF, and I(1,2,3-c,d)P in adults. In the short term, exposure to ATC, BbF, BkF, FLRT, FLR, NPT, and PNT can lead to skin irritation, itching, burns, skin thickening, pigmentation changes, lung lesions, and gastrointestinal disorders in humans (Albarano et al., 2023; INERIS, 2020; Siquiera et al., 2023). The carcinogenic risks for humans are manifold, particularly affecting the skin, bladder, and vital organs (ATSDR, 1995; Cerciello et al., 2025).

4. Conclusion

This study revealed significant PAH pollution in the waters of the Déganobo lacustrine system. This situation is primarily attributed to industrial and port activities in the vicinity of the study area. The PAHs in these waters were predominantly of pyrolytic origin and dominated by LMW PAHs. Both short- and long-term ecological and human health risks were found to be substantial for both the aquatic biota of this ecosystem and for humans.

Given the alarming PAH contamination and associated ecological and human health risks, future efforts must prioritize include: detailed source apportionment to enable targeted mitigation strategies against industrial, vehicular, domestic, and port discharges; robust long-term monitoring of PAHs in water, sediments, and biota to track trends and assess mitigation effectiveness; investigation of bioremediation and phytoremediation as cost-effective, nature-based solutions; ecological restoration initiatives to improve water quality and habitat integrity.

References.

Abouda, S., Galati, M., Conti, G.O., Cappello, T., Abelouah, M.R., Romdhani, I. et al. (2024). Metabolomic and biochemical disorders reveal the toxicity of environmental microplastics and benzo[a]pyrene in the marine polychaete Hediste diversicolor. Journal of Hazardous Materials, 477, 135404. [DOI: 10.1016/j.jhazmat.2024.135404]

Adeniji, A.O., Okoh, O.O., & Okoh, A.I. (2019). Distribution pattern and health risk assessment of polycyclic aromatic hydrocarbons in the water and sediment of Algoa Bay, South Africa. Environmental Geochemistry and Health, 41, 1303–1320. [DOI: 10.1007/s10653-018-0203-z]

Albarano, L., De Rosa, I., Santaniello, I., Montuori, M.M., Serafini, S.S., Toscanesi, M.M. et al. (2023). Synergistic, antagonistic, and additive effects of naphthalene, phenanthrene, fluoranthene and benzo(k)fluoranthene on Artemia franciscana nauplii and adult. Environmental Pollution, 335, 122286. [DOI: 10.1016/j.envpol.2023.122286]

Amine, H., Halwani, J., Gomez, E., & Merhabi, F. (2024). Research on polycyclic aromatic hydrocarbon pollution in sediments and waters of northern Lebanon: rivers, transition zones, and port sites. Region - Water Conservancy, 7(1), 195-212. [DOI: 10.1016/j.regwat.2024.01.002]

ATSDR (1995). Toxicological Profiles for Polycyclic Aromatic Hydrocarbons (PAHs). Agency for Toxic Substances and Disease Registry, Atlanta, GA: U.S. Department of Health and Human Services, Public Health Services. Retrieved from [http://www.atsdr.cdc.gov/toxpro2.html](https://www.google.com/search?q=http://www.atsdr.cdc.gov/toxpro2.html)

ATSDR (2022). Support document to the 2022 substance priority list, 9p.

Binet, D., & Marchal, E. (1970). Sur la présence des résidus d’hydrocarbures dans les eaux ivoiriennes. Rapport scientifique, Projet de développement de la pêche pélagique côtière. Ministère de la Production Animale et PNUD-FAO, 8 p.

Camargo, M.C.R., de Souza, E.T., Staudt, P.B., & Soares, R.P. (2024). Prediction of polycyclic aromatic hydrocarbons solubility in different solvents. Fluid Phase Equilibria, 579, 114013. [DOI: 10.1016/j.fluid.2023.114013]

CEAEQ, 2024. MA. 400 – HAP 1.1 2024-10-30 (révision 14) Détermination des hydrocarbures aromatiques polycycliques : dosage par chromatographie en phase gazeuse couplée à la spectrométrie de masse, Eds Ministère de l’Environnement, de la Lutte contre les Changements Climatiques, de la Faune et des Parcs (MELCCFP), 16 p.

Cerciello, F., Russo, C., Senneca, O., Migliaccio, R., Oliano, M.M., & Apicella, B. (2025). Volatile species and polycyclic aromatic hydrocarbons upon thermal (oxidative) decomposition of lignocellulosic biomass: The insightful case of reconstituted cast-leaf tobacco. Journal of Analytical and Applied Pyrolysis, 189, 107062. [DOI: 10.1016/j.jaap.2025.107062]

Chen, H., Shang, N., Zhao, C., Wang, C., Meng, L., Huang, T., et al. (2025). Factors influencing particle-water partition and ecological risks of polycyclic aromatic hydrocarbons in the Yangtze River mainstream. Environmental Pollution, 379, 126523. [DOI: 10.1016/j.envpol.2024.126523]

Delcourt, M., & Rocle, R. (2017). Des hydrocarbures issus de la biologie industrielle : la trajectoire et les perspectives de Global Bioenergies. Annales des Mines - Réalités industrielles, (1), 44–47. [DOI: 10.3917/rindu1.171.0044]

Dovonou, F. E., Ibikounle, M., Akouedegni, C. G., Aissi, V., Dossou, M. P., & Mama, D. (2019). Impacts des Hydrocarbures Aromatiques Polycycliques sur les poissons : Cas des Tilapias du lac Nokoué au Sud du Bénin (Afrique de l’Ouest). European Scientific Journal, 15(36), 458-474. [DOI: 10.19044/esj.2019.v15n36p458]

Dunnett, C. W. (1955). A multiple comparison procedure for comparing several treatments with a control. Journal of the American Statistical Association, 50, 1096–1121. [DOI: 10.1080/01621459.1955.10501252]

Elobaid, E. A., Yigiterhan, O., Al-Ansari, E. M. A. S., Chen, Z., Mohieldeen, Y. E., & Abdalla, R. (2025). Ecological risk assessment of heavy metals in the marine sediments associated with the petroleum hydrocarbon industry in the central Arabian Gulf. Journal of Hazardous Materials Advances, 100749. [DOI: 10.1016/j.jhazmat.2025.100749]

Ezike, C.O., Echor, F.O., Raymond, I., & Uwadiegwu, N.C. (2019). Benzo [a] Pyrene-carcinogenic and mutagenic equivalents of Clarias gariepinus (Burchell, 1822) to Nigerian-crude and diesel oils. International Journal of Fisheries and Aquatic Studies, 7(5), 62-67.

Fan, Q., Liu, D. D., Tian, H., Lu, X., Chen, Y., Feng, X., et al. (2025). Deciphering hydrocarbon evolution history of the Cambrian shales in tectonically complex basins. Marine and Petroleum Geology, 177, 107386. [DOI: 10.1016/j.marpetgeo.2025.107386]

FD T90-523-4 (2023). Qualité de l'eau - Guide d'échantillonnage pour le suivi de la qualité des eaux dans l'environnement - Partie 4 : échantillonnage d'eau de lac. Eds AFNOR, France, 50 p.

Fisher, R. A. (1919). The correlation between relatives on the supposition of Mendelian inheritance. Earth and Environmental Science Transactions of the Royal Society of Edinburgh, 52(2), 399–433. [DOI: 10.1017/s0000000000004558]

Gnonsoro, U. P., Soumahoro Sangaré, N., Kouassi, N. L. B., N’Goran, K. M., Yao, K. M., & Trokourey, A. (2023). Distribution of polycyclic aromatic hydrocarbons (PAHs) in M’Badon Bay (Ebrie Lagoon, Abidjan-Côte d’Ivoire). International Journal of Environmental Monitoring and Analysis, 11(4), 89–101. [DOI: 10.11648/j.ijema.20231104.14]

Gosset, W. S. (1908). The probable error of a mean. Biometrika, 6(1), 1–25. [DOI: 10.1093/biomet/6.1.1]

Hilly, M., Adams, M. L., & Nelson, S. C. (2002). A study of digit fusion in the mouse embryo. Clinical and Experimental Allergy, 32(4), 489-498. [DOI: 10.1046/j.1365-2222.2002.01358.x]

Hopf, N.B., Champmartin, C., Schenk, L., Berthet, A., Chedik, L., Du Plessis, J.L. et al. (2020). Reflections on the OECD guidelines for in vitro skin absorption studies. Regulatory Toxicology and Pharmacology, 117, 104752. [DOI: 10.1016/j.yrtph.2020.104752]

Howard, I. C., Okpara, K. E., & Techato, K. (2021). Assessment of polycyclic aromatic hydrocarbons in river bed sediment of artisanal crude oil refining area in the Niger Delta. Water, 13, 3295. [DOI: 10.3390/w13233295]

Hur, S., Kim, D.-W., Park, J.-G., Lee, H.R., Kim, Y.-J., Lee, B.-K. et al. (2025). Wintering loons in South Korea face an ongoing threat from polycyclic aromatic hydrocarbons: Shifting sources and potential DNA damage. Environmental Pollution, 367, 125659. [DOI: 10.1016/j.envpol.2024.125659]

INERIS (2015). Données technico-économiques sur les substances chimiques en France: Fluoranthène, DRC-14-136881-07005A, 28 p.

INERIS (2020). Benzo[b]fluoranthène, Verneuil-en-Halatte: Ineris - 200845 - v1.0, 12/10/2020, 38 p.

Irié, B. T. J.-G., Kando, A. M.-L., Touré, M., Koné, D., & Coulibaly, A. S. (2024). Concentration of polycyclic aromatic hydrocarbons and quality of sediments in Biétry bay (Ebrié lagoon, Côte d’Ivoire). GSC Advanced Research and Reviews, 20(3), 226–234. [DOI: 10.30574/gsrarr.2024.20.3.0248]

Jin, W., Song, K., Zhai, C., Wang, S., Zheng, M., & Feng, Z. (2025). Low ecological risks for heavy metals and petroleum hydrocarbons in coastal petrochemical discharge waters. Marine Pollution Bulletin, 218, 118163. [DOI: 10.1016/j.marpolbul.2025.118163]

Kadio, A. K. C. (2023). The role of the port of San Pedro in promoting port sustainability: an institutional perspective. Journal of Human Resource and Sustainability Studies, 11, 846–857. [DOI: 10.4236/jhrss.2023.116049]

Knafla, A., Phillipps, K.A., Brecher, R.W., Petrovic, S., & Richardson, M. (2006). Development of a dermal cancer slope factor for benzo[a]pyrene. Regulatory Toxicology and Pharmacology, 45(2), 159-168. [DOI: 10.1016/j.yrtph.2006.03.003]

Koko, A. T. N., Tuo, P., & Anoh, K. P. (2018). Gestion de l’environnement et santé de la population dans la ville de San Pédro. Regardsuds, 2018(1). Retrieved from [https://regardsuds.org/spip.php?article121](https://www.google.com/search?q=https://regardsuds.org/spip.php%3Farticle121)

Konan, K. F. A., & Yao, M. K. (2023a). Occurrence, ecological and health risks of organic pollution in tropical lake waters. International Journal of Environment and Climate Change, 13(11), 2509–2521. [DOI: 10.9734/ijecc/2023/v13i113063]

Konan, K. F. A., & Yao, M. K. (2023b). Seasonal contamination of Déganobo Lake waters by herbicide residues. GSC Advanced Research and Reviews, 17(2), 2509–2521. [DOI: 10.30574/gsrarr.2023.17.2.0232]

Konan, K. F. A., & Yao, M. K. (2025). Seasonal concentrations of insecticide residues in Déganobo Lake System. Journal of Applied Sciences and Environmental Management, 29(1), 147–156. [DOI: 10.4314/jasem.v29i1.15]

Konan, K. F. A., Yao, M. K., Dongui, B. K., & Trokourey, A. (2020). Sources and toxic potential of PAHs in Vridi canal sediments. Research Journal of Chemical Sciences, 10(1), 23–33.

Kouassi, D. S., Bakary, N. M., & Aloko, N. G. (2017). Optimisation du trafic des ports ivoiriens. Revue de Géographie Tropicale et d’Environnement, 2, 153–164.

Kuhn, A.V., Alliprandini da Costa, J.H., Souza, U.P., de Lima, G.V., Pereira, P.H.C., & Dolatto, R.G. (2025). Ecotoxicological assessment of polycyclic aromatic hydrocarbons and fish biomarkers in a marine protected area. Regional Studies in Marine Science, 85, 104155. [DOI: 10.1016/j.rsma.2025.104155]

Li, C., Li, Z., & Wang, H. (2023). Characterization and risk assessment of polycyclic aromatic hydrocarbons (PAHs) pollution in particulate matter in rural residential environments in China-A review. Sustainable Cities and Society, 96, 104690. [DOI: 10.1016/j.scs.2023.104690]

Marchand, M., & Martin, J.-L. (1985). Détermination de la pollution chimique dans la lagune d’Abidjan. Océanographie Tropicale, 20(1), 26–39.

MEDD et Agence de l'eau (2003). Système d'Évaluation de la Qualité des Eaux (SEQ-Eaux), grilles d'évaluation, France.

Moon, H. G., Bae, S., Chae, Y., Kim, Y.-J., Kim, H.-M., Song, M., et al. (2024). Ecological risk of PAHs in urban soils with high PM levels. Ecotoxicology and Environmental Safety, 272, 116014. [DOI: 10.1016/j.ecoenv.2024.116014]

Mortazavi, M. S., Sharifian, S., Nozar, S. L. M., & Koohkan, H. (2025). Ecological risks posed by heavy metals and petroleum hydrocarbons in Persian Gulf sediments. Journal of Trace Elements in Medicine and Biology, 87, 127574. [DOI: 10.1016/j.jtemb.2024.127574]

NYS DOH (New York States Department of Health) (2007). Hopewell precision area contamination: Appendix C-NYS DOH. In Procedure for Evaluating Potential Health Risks for Contaminants of Concern. States Department of Health: New York, NY, USA. Retrieved from <http://www.health.ny.gov/environmental/investigations/hopewell/appendc.htm>

Ogou, A. W. A., & Bidi, T. J. (2019). Port, aménagement et développement durable à San-Pédro. European Scientific Journal Edition, 15, 110–131. [DOI: 10.19044/esj.2019.v15n36p110]

OJEU (Official Journal of the European Union) (2013). Directive 2013/39/EU of 12 August 2013. L 226, 14–17.

Olivares-Rubio, H.F., & Ponce-Vélez, G. (2025). Polycyclic aromatic hydrocarbons during the 21st Century in Southern Gulf of Mexico, a prominent petroleum area: A review. Marine Pollution Bulletin, 210, 117343. [DOI: 10.1016/j.marpolbul.2024.117343]

Qiao, M., Fu, L., Li, Z., Liu, D., Bai, Y., & Zhao, X. (2020). Distribution and ecological risk of substituted and parent PAHs in rivers of northern China. Environmental Pollution, 257, 113600. [DOI: 10.1016/j.envpol.2019.113600]

SGS Côte d’Ivoire (2016). Étude d’impact environnemental et social: terminal industriel polyvalent à San-Pédro. PRO-ENV 054B-2016, 32 p.

Siqueira, A.S., Almeida, L.F., & Fragoso, W.D. (2023). Determination of anthracene, phenanthrene, and fluorene in tap water and sediment samples by fluorescence spectroscopy on nylon membranes and second-order calibration. Talanta, 253, 124002. [DOI: 10.1016/j.talanta.2022.124002]

Tukey, J. (1949). Comparing Individual Means in the Analysis of Variance. Biometrics, 5(2), 99–114. [DOI: 10.2307/3001913]

US-EPA (1982). Appendix A: Priority pollutants. Federal Registration, 47, 52309.

US-EPA (1989). Risk Assessment Guidance for Superfund Volume I, Human Health Evaluation Manual (Part A): InterimFinal. EPA/540/1 -89/002; PB90—155581, pp. 289.

US-EPA (1992a). Dermal Exposure Assessment: Principles and Applications (Interim Report). EPA/600/8-91/011B. Washington, DC.

US-EPA (1992b). Guidelines for Exposure Assessment. EPA/600/Z-92/001. Washington, DC.

US EPA (2005). Guidelines for carcinogen risk assessment. 70 FR 17765-17817, United States Environmental Protection Agency, Washington, DC., United States.

US-EPA (2009). Risk Assessment Guidance for Superfund. Vol. 1: Human Health Evaluation Manual (F). EPA/540/R/070/002. Washington, DC.

Wang, Z., Liu, Y., Zhang, A., Yang, L., Wei, C., Chen, Y., Liu, Z., & Li, Z. (2024). Occurrence characteristics, environmental trend, and source analysis of polycyclic aromatic hydrocarbons in the water environment of industrial zones. Environmental Research, 245, 118053. [DOI: 10.1016/j.envres.2024.118053]

World Bank (2023). <https://donnees.banquemondiale.org/pays/cote-divoire>, consulted le 19th May 2023.

Yang, Z., Shah, K., Hollebone, B., Laforest, S., MacGillivray, M., Dey, D. et al. (2024). Occurrence, characterization, and ecological risk analysis of petroleum hydrocarbons in water and sediments following large-scale field simulated oil spills at the experimental lakes area, Northwestern Ontario, Canada. Marine Pollution Bulletin, 209, 117235. [DOI: 10.1016/j.marpolbul.2024.117235]

Zhang, S., Xing, X., Yu, H., Du, M., Zhang, Y., Li, P., Li, X., Zou, Y., Shi, M., Liu, W., & Qi, S. (2025a). Fate of polycyclic aromatic hydrocarbon (PAHs) in urban lakes under hydrological connectivity: A multi-media mass balance approach. Environmental Pollution, 366, 125556. [DOI: 10.1016/j.envpol.2024.125556]

Zhang, Q., Hu, R., Xie, J., Hu, X., Guo, Y., & Fang, Y. (2025b). Effects of microplastics on polycyclic aromatic hydrocarbons migration in Baiyangdian Lake, northern China: Concentrations, sorption–desorption behavior, and multi-phase exchange. Environmental Pollution, 366, 125408. [DOI: 10.1016/j.envpol.2024.125408]