RSC Advances

PAPER

View Article Online
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Cite this: *RSC Adv.*, 2025, **15**, 38646

# Assessment of bisphenol A analogues in water and sediment of an urban river system in Dhaka, Bangladesh: a regionally focused environmental investigation

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Bisphenol A analogues (BPs), now commonly replacing BPA in industrial applications, are emerging contaminants of concern due to their pervasive environmental presence and possible endocrine-disrupting effects. Further research is needed to elucidate their ecological implications. Although their production and application have expanded rapidly, comprehensive data regarding their occurrence, spatial distribution, correlations, and environmental distribution remain limited. BPs are almost unreported in the environmental matrices in Bangladesh. Herein, a total of 120 samples (80 from water and 40 from sediment) were analyzed for four BPs (bisphenol B, bisphenol E, bisphenol F, bisphenol AF) in two seasons (rainy and winter) from the Turag River in Dhaka City, Bangladesh. GC-MS was used to determine the BPs in water and sediment samples. In the case of water samples in the rainy season, the results indicated that BPF was the predominant BP, which ranged from 1.3950 to 7.2352 μg $L^{-1}$, followed by BPAF (0.3222 to 6.9578 μg $L^{-1}$), BPB (0.3046 to 2.5262 μg $L^{-1}$) and BPE (0.3230 to 1.8309 μg $L^{-1}$). Water samples in the winter season also indicated BPF as the predominant BP, which ranged from 1.1186 to 7.3094 μg $L^{-1}$, followed by BPB (0.0387 to 9.2240 μg $L^{-1}$), BPAF (0.3497 to 5.9782 μg $L^{-1}$) and BPE (0.5280 to 3.5314 μg $L^{-1}$). However, in sediment samples, BPF was found in the range from 27.2740 to 234.4540 μg $g^{-1}$ dw, followed by BPAF (56.3560 to 129.1900 μg $g^{-1}$ dw), BPB (24.3860 to 141.4120 μg $g^{-1}$ dw) and BPE (3.7340 to 33.8920 μg $g^{-1}$ dw). A significant seasonal influence was observed in the occurrence of BPs in river water. Significant positive and moderate positive correlations were found in the BPs of water samples, and no significant correlations were observed in the BPs of sediment samples.

Received 7th July 2025
Accepted 17th September 2025

DOI: 10.1039/d5ra04861c

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## 1. Introduction

Bisphenol analogues (BPs) are a bulky cluster of synthetic organic chemicals that share a common structure with two hydroxyphenyl functionalities.[1] These compounds are created through the reaction of phenol with acetone under the conditions of high temperature and low pH, with the participation of catalysts.[2] BPs are widely used in the production of polycarbonates and epoxy resins as well as in several industrial and commercial appliances, such as fire-resistant polymers, plastic linings for beverage cans, dentistry sealants, thermo-sensitive coatings for paper materials, medical equipment, toys, electronics, and water pipes.[3–5] Wastewater (WW), originating from both residential and industrial sources, serves as a primary pathway through which bisphenol analogues (BPs) are introduced into the environment.

The global consumption of BPA was over 7.7 million metric tons in 2015.[6] Because of its extensive use, huge amounts of BPA have been inevitably released into aquatic environments.[7] BPA has been categorized as a "pseudo-persistent" chemical, leading to its widespread presence and potential agglomeration in diverse environments, including indoor dust, water, sediment, *etc.*[8–11] It has been well acknowledged that BPA is associated with several adverse health effects in wildlife and humans, such as cancer, infertility, cardiovascular diseases, diabetes and neurodegenerative diseases.[5] As a result, the production and usage of BPA have been prohibited or strictly regulated in the European Union, North America, Japan and China.[12–14] To reduce potential risks to the environmental quality and health outcomes, it is essential to develop safer substitutes. Several BPA alternatives that are structurally similar to BPA have been synthesized and introduced into the market.[5,13] Representative BPA substitutes examined in this study include BPB, BPE, BPF, and BPAF. Bisphenol AF and bisphenol S dominate among the

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commonly utilized bisphenol A replacements. For instance, the estimated annual BPAF production was in the range of 10 000–500,000 pounds in the U.S. between 1986 and 2002.[15] BP substitutes were also frequently spotted in environmental and biota samples.[16,17] In which the detected BPF caused genotoxicity.[18] To date, several studies have revealed that BPA alternatives (*e.g.*, BPAF, BPB, BPF, and BPE) present similar or even greater toxicological effects, such as nerve toxicity, developmental toxicity, and estrogenic and/or antiandrogenic activities.[19–21]

In thermal paper,[19] bisphenols (BPAF, BPB, BPF, BPE, BPA and others) have been identified and categorized (U.S. Environmental Protection Agency 2015); different scientific reports have indicated that they all have similar endocrine-disrupting potential,[22,23] and the ecotoxicological risk of mixtures of bisphenols (BPs).[24] In India, Ponnaiah *et al.*[25] synthesized a $Ru^0$/PANI/g-$C_3N_4$ nanocomposite using ultrasonic energy, which demonstrated exceptional electrochemical performance for detecting bisphenol compounds, particularly BPA in human and animal urine. The study highlights BPA's strong estrogenic activity and its potential to cause reproductive and developmental disorders even at trace concentrations.

Although scientists in the mid-1930s noted the first signs of the embryotoxic and teratogenic effects of BPs, it is only in the last few years that this compound has received widespread attention.[26,27] Due to its adverse impact on human health, more countries are imposing bans or restrictions on the usage of BPA.[28] Pollutants that exist in water have an impact on the adjacent environment and human health; therefore, it is indispensable to test the quality of water and assess the existence of pollutants.[29]

The central focus of the investigation is to determine the occurrence of BPs in water along with sediments of the Turag River by applying a combined gas chromatography and mass spectrometry procedure. There have been no previous reports based on the urban river systems in Bangladesh; this study has revealed the presence of BPs in the water and sediments of the Turag River, which is strongly impacted by human activities. This work also aims to assess the concentration, percentage contribution, correlation, cluster analysis, and spatial distribution of BPs in Turag, one of the leading urban rivers of Bangladesh.

## 2 Materials and methods

### 2.1 Study area

In total, twenty[20] water samples were collected from the Turag River, a vital waterway in Bangladesh, flowing through the heart of Dhaka;[10] samples were collected in the rainy season (June–July) and another 10 in the winter (December–January) season (Fig. 1). Each sample was tested for four BPs, namely BPB, BPF, BPAF, and BPE (BPA and BPS were not included due to the

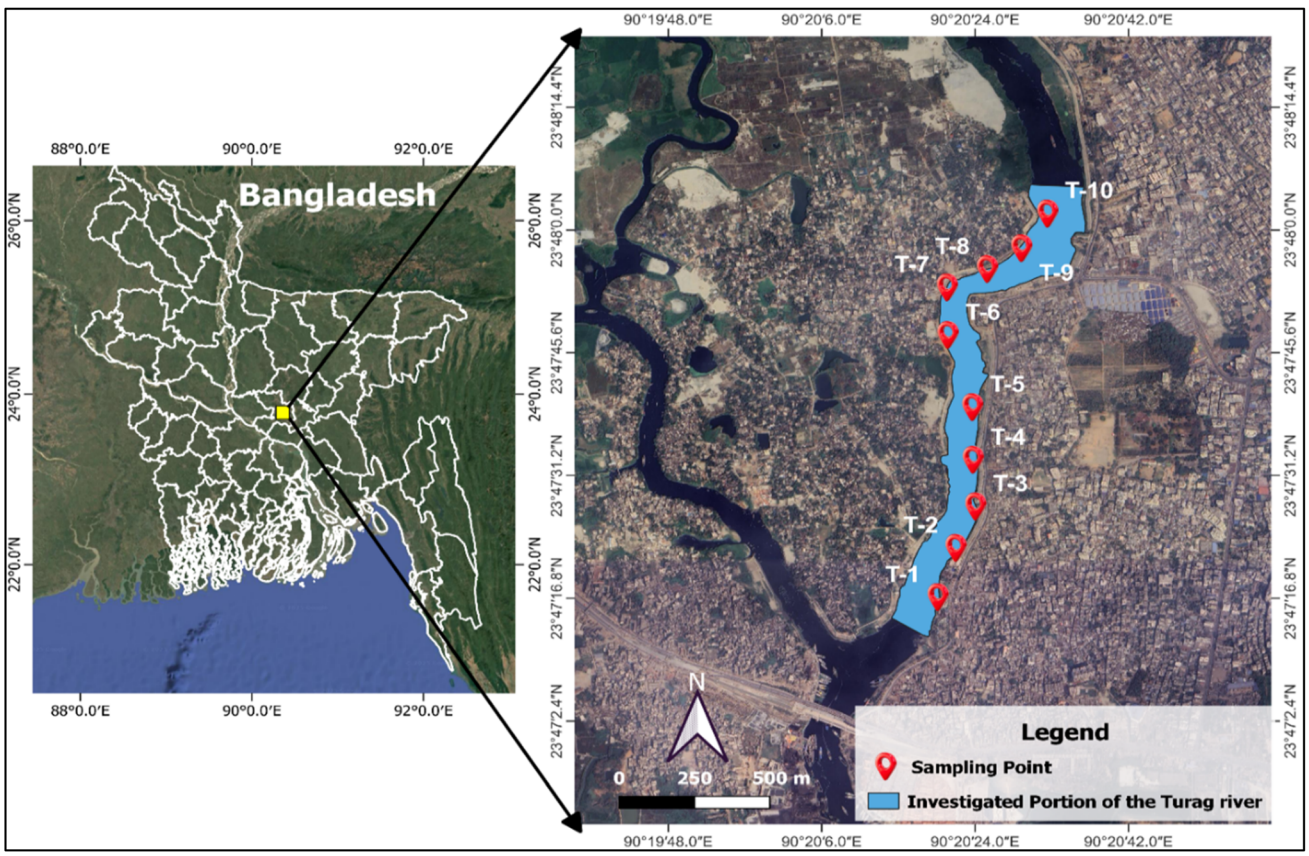


Fig. 1 Study area map of the Turag River, Dhaka District, Bangladesh, showing the sampling locations for bisphenol analogue (BP) analysis.

unavailability of certified standards at the time of analysis) over two seasons, and a total of eighty (80) data points were found. Water samples were collected at a depth of 0.5 m from the water surface using a water sampler and then stored in amber storage bottles [500 mL] after collection.

Ten sediment samples were collected from the Turag riverbed in the winter season (each sample was tested for four BPs (BPB, BPF, BPAF, BPE), and forty data points were obtained) using a stainless-steel grab sampler at each designated site. The wet sediment samples were dried, homogenized, sieved (80-mesh sieve), and stored at −16 °C in a PE package that was wrapped in aluminum foil until extraction.

### 2.2. Chemicals used in this research

Four BPs, including BPB, BPE, BPF, and BPAF, were selected as target compounds. Sodium chloride (NaCl) and a 1 : 1 solution of hydrochloric acid (HCl) were used for sample treatment. Sodium sulphate (anhydrous) was used to absorb water due to its highly hygroscopic nature; 99% BSTFA [N,O-bis(trimethylsilyl) trifluoroacetamide] and 1% TMCS [trimethylchlorosilane, trimethylsilylchloride] were used as derivatization agents. All the compounds were sourced from Sigma-Aldrich, Germany, except for DCM, which was supplied by Riedel-de Haen Honeywell, Germany.

### 2.3. Water sample extraction

Five hundred milliliters (500 mL) of each water sample were taken, and then NaCl (150 g) and 2–3 drops of HCl (1 : 1) were added. The prepared solution was poured into a separatory funnel, where 20 mL of dichloromethane was added. It was then subjected to mechanical shaking at 1500 rpm for one minute to facilitate efficient recovery. The underlying organic phase was subsequently isolated and transferred into a round-bottom flask preloaded with anhydrous sodium sulfate to remove residual moisture. The organic phase was then directly transferred to a 2 mL GC vial using a 0.22 μm filter, and 50 μL of derivatization agent was added. The mixture was then subjected to vortex agitation for one minute at ambient temperature and set in an ultrasonic sonicator at 45 °C for 30 minutes. The sample was then spiked with an internal standard solution before analysis and prepared to be injected into the GC-MS. A calibration curve was constructed using solutions of bisphenol analogues of known concentrations treated in the same way.[30]

### 2.4. Sediment sample extraction

For sediment samples, 10 g of the measured portion of the homogenized sediment sample was accurately weighed and placed into a 200 mL conical flask. The sediment sample was extracted with 20 mL DCM and placed in an orbital shaker at 30 °C with a shaking speed of 250 rpm for 60 minutes to ensure uniform mixing and extraction efficiency (repeating the shaking process once again). The supernatant was then transferred into a round-bottom flask filled with anhydrous sodium sulfate (to dehydrate) using Whatman filter paper (100 mm). The rest of the procedures were the same as for the water samples.

### 2.5. GC-MS profiling

A 2 μL portion of the extract was loaded into the system for analysis. Separation was performed on an Rxi-5Sil MS capillary column (30 m × 0.25 mm i.d., 0.25 μm film thickness) and high-purity helium (99.999% purity) was employed as the carrier gas, maintained at a constant flow rate of 1 mL $min^{-1}$ throughout the chromatographic run. The injector was kept at 300 °C, while the oven temperature program began at 40 °C (held for 1 minute), followed by a ramp of 10 °C $min^{-1}$ to 300 °C, where it was held for an additional 5 minutes, with a total run time of 32 minutes. The ion source was maintained at 200 °C, and a temperature of 250 °C was applied to the interface during analysis to ensure optimal ionization and transfer efficiency during GC-MS analysis. The solvent delay time was 6 minutes, and the column pressure was maintained at 49.7 kPa throughout the analysis.

### 2.6. Quality control and assurance

Quality control and assurance were prioritized throughout the study, including field sampling, sample preparation, and analytical instrumentation. To minimize analytical interference, a secondary rinse was performed using deionized water and methyl alcohol (methanol). For each extraction batch, the procedural blank and recovery percentage were assessed. To quantify BP concentrations, a six-point internal calibration curve (5, 10, 20, 50, 100, 200 μg $mL^{-1}$) was used, yielding correlation coefficients ($R^2$) greater than 0.998, indicating excellent linearity across the concentration range. The LOD, LOQ, and MDL values for each BP congener were estimated mathematically from the slope of the linear calibration curve ($slope_m$) and the standard deviation of the response using the following eqn (1)–(3):

$$\mathrm{LOD} = 3.3 \times \frac{\sigma}{S} \quad (1)$$

$$\mathrm{LOQ} = 10 \times \frac{\sigma}{S} \quad (2)$$

$$\mathrm{MDL} = \text{Student's } t \text{ value} \times \text{standard deviation} \quad (3)$$

where $\sigma$ is the standard deviation of the response, and $S$ is the slope of the calibration curve.

Technical replicates were performed during instrumental analysis, with each QC sample analyzed in triplicate to confirm reproducibility and analytical precision. Environmental samples were not processed in technical replicates due to resource constraints.

Tables 1 and 2 show that the detection limit and quantification limit for the desired bisphenols were in the ranges of 0.8968–1.0594 μg $L^{-1}$ and 2.7175–3.2102 μg $L^{-1}$ in water and 0.9103–1.0721 μg $g^{-1}$ dw and 2.8310–3.2289 μg $g^{-1}$ dry weight in collected sediment. All concentrations of the selected bisphenol compounds in the procedural blanks exhibited analyte levels below their respective detection limits. The spike recovery percentage was determined, which ranged from 67.52% to 73.51% and 68.57% to 74.42% for water and sediment,

 

Table 1 Recovery percentage, LOD, LOQ, and MDL of each BP in water

| Name of the analyte | LOD (μg $L^{-1}$) | LOQ (μg $L^{-1}$) | MDL (μg $L^{-1}$) | Recovery (%) |
|---|---|---|---|---|
| Bisphenol B | 1.0594 | 3.2102 | 0.4766 | 73.51 |
| Bisphenol AF | 0.9089 | 2.7544 | 0.6659 | 67.52 |
| Bisphenol E | 0.8968 | 2.7175 | 0.7132 | 69.31 |
| Bisphenol F | 0.9759 | 2.9571 | 0.4739 | 70.96 |

Table 2 Recovery percentage, LOD, LOQ, and MDL of each BP in sediment

| Name of the analyte | LOD (μg $g^{-1}$ dw) | LOQ (μg $g^{-1}$ dw) | MDL (μg $g^{-1}$ dw) | Recovery (%) |
|---|---|---|---|---|
| Bisphenol B | 1.0721 | 3.2289 | 0.4792 | 74.42 |
| Bisphenol AF | 0.9103 | 2.8310 | 0.6781 | 68.57 |
| Bisphenol E | 0.9782 | 2.9361 | 0.7159 | 69.84 |
| Bisphenol F | 0.9975 | 2.9853 | 0.4838 | 71.19 |

respectively. The recovery percentage was calculated according to eqn (4):

$$\mathrm{RR}\% = \frac{C_{\mathrm{found}} - C_{\mathrm{real}}}{C_{\mathrm{added}}} \times 100 \quad (4)$$

Here, $C_{\mathrm{found}}$ = the concentration of the analyte measured in the spiked sample after analysis. $C_{\mathrm{real}}$ = the concentration of the analyte originally present in the unspiked sample. $C_{\mathrm{added}}$ = the known concentration of analyte intentionally added to the sample. RR% = the percentage of the added analyte that was successfully recovered.

In the spiking procedure, representative blank or low-background samples were selected and fortified with a known quantity of bisphenol standard. These spiked samples were then processed using the same extraction and analytical method (*e.g.*, GC-MS) applied to environmental samples. Following the analysis, the measured concentration was used to calculate the recovery percentage, accounting for native levels and spiked amounts.

To verify the absence of background interference, quality control blanks were run after every ten analytical samples to ensure the extraction accuracy. Tables 1 and 2 show the recovery percentage, LOD, LOQ, and MDL for each BP in the collected water and sediment matrices, respectively.

### 2.7. Statistical analysis

The presence of bisphenol pollutants in sediment matrices was expressed based on the dry mass of the sample. To assess data distribution and variance, the Shapiro–Wilk procedure was employed to examine normality and homogeneity. An independent *t*-test was conducted to determine the significant differences for the bisphenols in the obtained samples. To determine the relationships between the BPs, Spearman's correlation coefficient was employed, while clustering was employed to identify similarities between BPs and sampling locations. All statistical procedures were carried out *via* Origin Pro (Version 2022 V.9.9.0225), with a significance threshold set at $p < 0.05$. Spatial distribution mapping was completed employing QGIS (Version 3.22.1), an open-source geospatial tool.

## 3. Outcomes and analysis

### 3.1. Concentration profiles of BPs in the water matrix

Four bisphenol analogues—BPB, BPE, BPAF, and BPF—were detected in the water matrices with high frequency: 73.51%,

Table 3 Concentration of Bis-B, Bis-E, Bis-AF, and Bis-F in water and sediment

| Compounds | | | Bis-B | Bis-E | Bis-AF | Bis-F |
|---|---|---|---|---|---|---|
| Water (μg $L^{-1}$) | Rainy season | Mean | 1.23 | 0.82 | 1.93 | 3.66 |
| | | Median | 1.22 | 0.49 | 0.42 | 3.39 |
| | | Max | 2.53 | 1.83 | 6.96 | 7.24 |
| | | Min | 0.30 | 0.32 | 0.32 | 1.40 |
| | | Range | 0.30–2.53 | 0.32–1.83 | 0.32–6.96 | 1.40–7.24 |
| | | S.D. | 0.81 | 0.61 | 2.30 | 1.75 |
| | Winter season | Mean | 4.14 | 1.51 | 3.39 | 4.56 |
| | | Median | 4.17 | 1.42 | 3.50 | 4.67 |
| | | Max | 9.22 | 3.53 | 5.98 | 7.31 |
| | | Min | 0.04 | 0.53 | 0.35 | 1.12 |
| | | Range | 0.04–9.22 | 0.53–3.53 | 0.35–5.98 | 1.12–7.31 |
| | | S.D. | 3.51 | 0.91 | 1.96 | 1.85 |
| Sediment (μg $g^{-1}$) | | Mean | 67.77 | 18.04 | 95.85 | 123.51 |
| | | Median | 55.86 | 17.13 | 95.60 | 123.81 |
| | | Max | 141.41 | 33.89 | 129.19 | 234.45 |
| | | Min | 24.39 | 3.73 | 56.36 | 27.27 |
| | | Range | 24.39–141.41 | 3.73–33.89 | 56.36–129.19 | 27.27–234.45 |
| | | S.D. | 37.50 | 7.41 | 23.61 | 64.44 |

67.52%, 69.31%, and 70.96%, respectively. The total measured bisphenol concentrations (ΣBPs) ranged from 0.0387 to 9.2240 μg $L^{-1}$ (Table 3), indicating substantial contamination across sites.

Seasonal analysis revealed distinct distribution patterns. In the rainy season, median concentrations followed the order: BPF (3.39 μg $L^{-1}$) > BPB (1.22 μg $L^{-1}$) > BPE (0.49 μg $L^{-1}$) > BPAF (0.42 μg $L^{-1}$), while in winter, the order shifted slightly to BPF (4.67 μg $L^{-1}$) > BPB (4.17 μg $L^{-1}$) > BPAF (3.50 μg $L^{-1}$) > BPE (1.42 μg $L^{-1}$). Mean concentrations echoed this trend, with BPF consistently exhibiting the highest levels in both seasons—3.66 μg $L^{-1}$ in the rainy season and 4.56 μg $L^{-1}$ in winter—suggesting its widespread presence.

BPF concentrations ranged from 1.1186 to 7.3094 μg $L^{-1}$ overall, with seasonal ranges of 1.3950–7.2352 μg $L^{-1}$ (rainy) and 1.1186–7.3094 μg $L^{-1}$ (winter). BPAF, the second most abundant in the rainy season and third in winter, showed ranges of 0.3222–6.9578 μg $L^{-1}$ and 0.3497–5.9782 μg $L^{-1}$, respectively. BPB ranked second in winter and third in the rainy season, with a notably wide winter range (0.0387–9.2240 μg $L^{-1}$) compared to the rainy season (0.3046–2.5262 μg $L^{-1}$). BPE remained the least prevalent, ranging from 0.3230 to 3.5314 μg $L^{-1}$, with a slightly higher winter concentration. BPAF and BPF were the second and third dominant BPs, respectively, and these two chemicals are the major BPA substitutes because they are more resistant to biological degradation in aquatic environments.[1,31–33]

Seasonal hydrological changes significantly influence the concentrations of bisphenol compounds (BPs) in aquatic environments. During periods of heavy rainfall, increased river flow and water volume enhance the dilution capacity of the system, reducing the concentrations of pollutants such as BPs in surface waters. This natural purification process is most noticeable in monsoon-affected regions. During the dry winter period, reduced hydrological activity limits the dispersal of contaminants, allowing them to accumulate and concentrate in aquatic systems. These seasonal dynamics underscore the importance of considering hydrological variability when assessing the environmental behavior and risks of bisphenol compounds.[34]

Although BPF was introduced as a substitute for bisphenol A (BPA), its structural similarity and comparable endocrine-disrupting potential raise concerns about its environmental safety. Zaborowska *et al.*[35] emphasized that BPF, like BPA and BPS, can negatively affect soil microbiome and enzyme activity, potentially disrupting ecological balance and plant development. The seasonal persistence of BPF, with slightly elevated levels in winter, may reflect reduced dilution, increased industrial discharge, or accumulation due to lower microbial degradation rates in colder conditions.

These findings underscore the need for regulatory scrutiny of bisphenol analogues and comprehensive environmental monitoring. Substituting BPA with structurally similar compounds like BPF may not mitigate ecological risks unless accompanied by robust toxicity assessments and pollution control strategies.

### 3.2. Concentration profiles of BPs in sediment

The observed levels, including the range, mean, median, minimum, and maximum data of specific bisphenol compounds in sediment matrices from the Turag River, are presented in Table 3. Like BPs in water samples, four BPs (BPB, BPE, BPAF, and BPF) were also detected in sediment samples with detection rates of 74.42%, 68.57%, 69.84%, and 71.19%, respectively. Bisphenol concentrations in sediment matrices from the Turag River revealed significantly higher levels than those observed in water samples, indicating strong sorption potential and long-term accumulation in benthic environments. Among the four detected bisphenols, BPF was again predominant, with concentrations reaching 234.454 μg $g^{-1}$ dw and a median of 123.81 μg $g^{-1}$ dw. Elevated BPF levels at

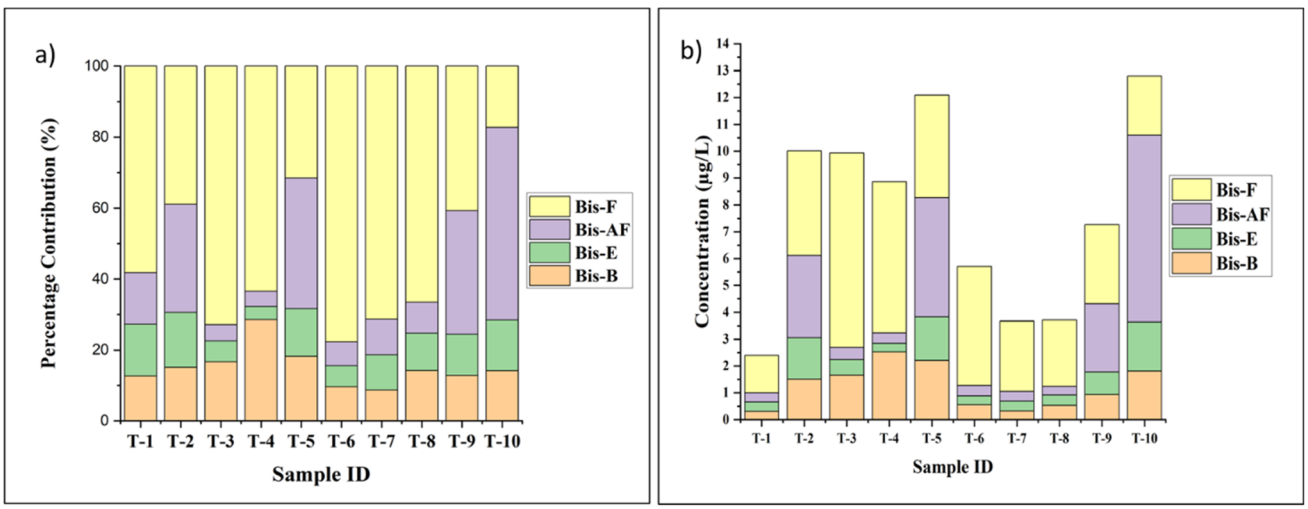


Fig. 2 Percentage compositions (a) and concentrations (b) of bisphenol analogues (BPs) in the water samples from ten sites (T-1 to T-10) along the Turag River during the rainy season. The figure highlights spatial variability and elevated BP levels near industrial zones, influenced by seasonal runoff and discharge.

sampling points TS-2, TS-8, and TS-10 located near industrial zones suggest direct input from chemical, textile, packaging, and electronics facilities. BPAF and BPB also showed substantial concentrations, reinforcing the influence of industrial discharge on sediment contamination.

The high concentrations of bisphenol analogues, particularly BPF, BPAF, and BPB, in Turag River sediments underscore their strong affinity for sediment particles and long-term persistence in benthic environments. This accumulation is driven by the physicochemical properties of bisphenols, along with sediment factors such as organic matter content and grain size. Recent photocatalytic studies have shown promising degradation pathways for bisphenol compounds under controlled conditions.[36–38] However, such mechanisms are often absent in natural sediment environments.

These findings are consistent with broader assessments of the Turag River, which have reported significant pollutant loads, including heavy metals and persistent organics in sediments, posing ecological risks to aquatic life.[39]

### 3.3. Percentage distribution of BPs in water

The percentage shares of river water at ten[10] sampling points are shown in Fig. 2. The figure highlights spatial variability and elevated BP levels near industrial zones, influenced by seasonal runoff and discharge. In the rainy season at sampling point 1 (T-1), BPF showed the maximum proportion (58%), and BPB showed the lowest representation of 13%. BPAF and BPE each comprised 15%. Fig. 3 reflects limited hydrological dilution and site-specific accumulation patterns of BPs, with distinct variations among individual compounds. In the winter season at sampling point 1 (TW-1), BPF declined to 41% and BPB rose to 40%; BPAF was 15% and BPE 5%. At sampling point 3 (T-3), BPF dominated the profile at 73%, followed by BPB (17%), and BPE and BPAF showed the least contribution (5%). In winter, at sampling point 3 (TW-3), BPAF and BPF each accounted for 37%, BPB showed 17% and BPE showed 9%. At point T-4 in the rainy season, BPF prevailed at 63%, whereas in winter at point TW-4, BPB led at 45%. At T-5 (rainy), BPAF represented the highest percentage at 37%, whereas at TW-5 (winter), BPB topped the profile at 39%. Overall, BPF emerged as the dominant compound across both seasons. In the rainy season, the percentage share was BPF > BPAF > BPB > BPE. In the winter season, the percentage share was BPF > BPB > BPAF > BPE.

BPF has become a widely adopted substitute for bisphenol A (BPA) in the production of epoxy resins, coatings, and thermal paper. The Turag River flows through highly industrialized zones, including Tongi and Gazipur, where numerous chemical, plastic, and textile industries operate. These industries release untreated or inadequately treated wastewater directly into the river. Moreover, many industrial facilities along the Turag lack adequate wastewater treatment systems. As a consequence, bisphenol F (BPF) and other bisphenol analogs are frequently discharged into aquatic environments due to their widespread use as BPA substitutes. These compounds frequently bypass standard wastewater treatment methods, leading to their prolonged presence in aquatic systems and contributing to ecological risks for aquatic biota.[40]

These findings align with regional studies in South Asia. Yamazaki *et al.*[41] reported that BPF concentrations in Indian rivers, particularly near Chennai, were often significantly higher, reflecting its growing use as a BPA substitute in industrial applications. The study also detected BPAF and BPB at lower levels, supporting the seasonal variability observed in this dataset.

In Bangladesh, BPS, BPA, BPF, BPB, and BPAF were examined in thermal paper cash receipts, in which BPA and BPS were found at concentrations ranging from 0.83% to 1.71% and 0.61% to 0.96%, respectively. No other analogues were detected from any of the samples analyzed there.[42]

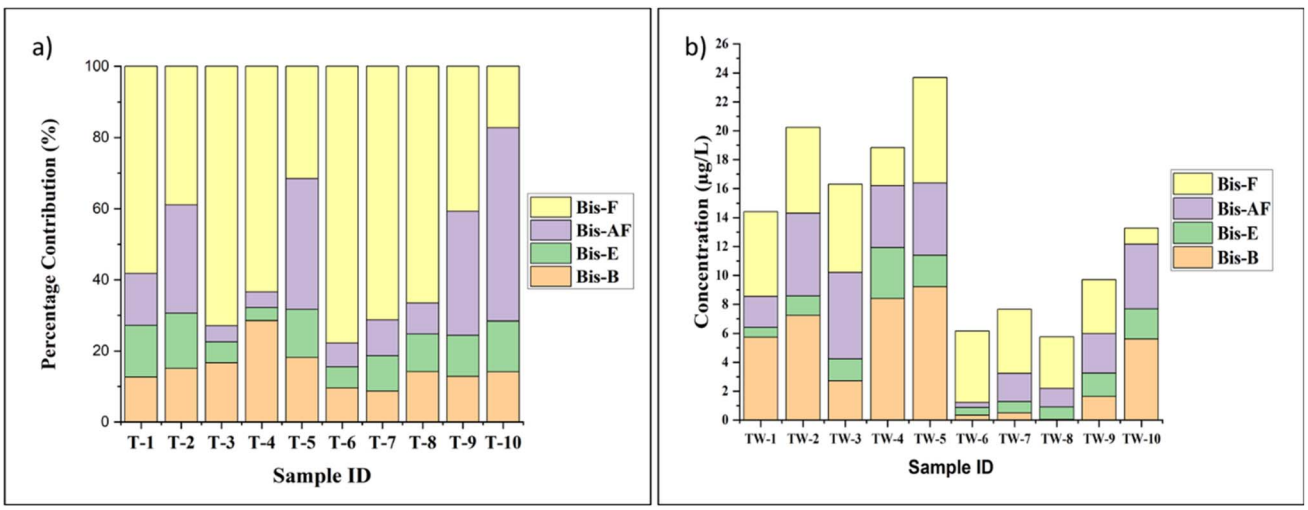


Fig. 3 Percentage compositions (a) and concentrations (b) of bisphenol analogues (BPs) in the water samples from ten sites (TW-1 to TW-10) along the Turag River during the winter season. The figure reflects limited hydrological dilution and site-specific accumulation patterns of BPs, with distinct variations among individual compounds.

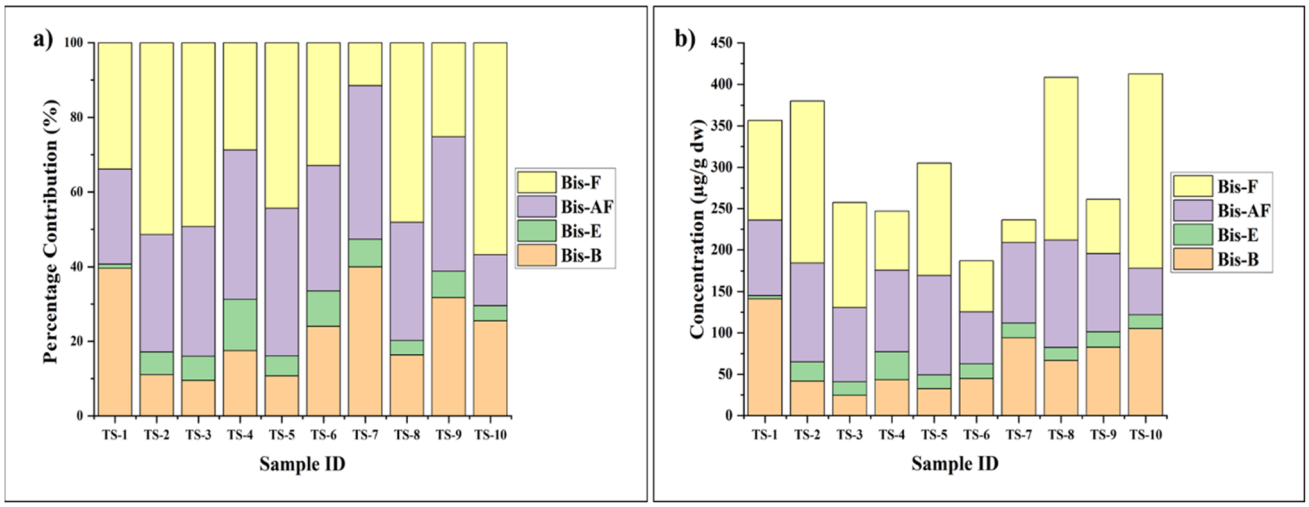


Fig. 4 Percentage compositions (a) and concentrations (b) of bisphenol analogues (BPs) in the sediment samples from ten sites (TS-1 to TS-10) along the Turag River. The figure illustrates compound-specific variability and localized accumulation patterns, reflecting limited sediment dispersion during the sampling period.

### 3.4. Percentage distribution of BPs in sediment

Fig. 4 depicts compound-specific variability and spatially localized accumulation of bisphenol analogues, suggesting restricted sediment dispersion during the sampling period. In the case of sediment samples, the percentage contribution showed maximum contributions of BPF (34%, 51%, 49%, 29%, 44%, 33% 12%, 48%, 25%, and 57%), followed by BPAF (25%, 31%, 35%, 40%, 40%, 34%, 41%, 32%, 36%, and 14%), then BPB and BPE. At point TS-1, the distribution was BPB (40%) > BPF (34%) > BPAF (25%) > BPE (1); here, BPB and BPF dominated the profile. At TS-2, BPF and BPAF showed the highest percentages (51% and 31%, respectively). BPF (49%) and BPAF (35%) were dominant at point TS-3, BPAF (40%) and BPF (29%) at point TS-4, BPF (44%) and BPAF (40%) at point TS-5, BPAF (34%) and BPF (33%) at point TS-6, BPAF (41%) and BPB (40%) at point TS-7, BPF (48%) and BPAF (32%) at point TS-8, BPAF (36%) and BPB (32%) at point TS-9, and BPF (57%) and BPB (25%) at point TS-10. BPE showed the lowest percentage presence at all sampling points. The extensive presence of BPs in this area may be attributed to the use and disposal of BP-containing products. BPs are reported to be widely used in the manufacture of epoxy and polycarbonate resins, plastics for food contact materials, and thermal receipt papers.[13,32]

This overspreading aligns with findings from Yamazaki *et al.*[41], who reported elevated BPF concentrations in surface waters and sediments in India, especially near Chennai, due to its use in epoxy resins and coatings. Similarly, Luo *et al.*[43] documented BPF and BPA as the main contributors to sediment contamination in irrigation canals in southern China, with ecological risks linked to aquaculture and domestic discharge.

### 3.5. Correlation coefficient

The normality of the datasets was tested using the Shapiro–Wilk test (Table 4). Since some variables showed significant deviations from normality ($p < 0.05$), non-parametric methods were applied. Therefore, Spearman's rank correlation (Fig. 5 exhibits statistically pronounced association among bisphenol analogues, indicating possible common sources or analogous environmental fates and transport mechanisms) was chosen to assess the associations between the levels of bisphenol compounds in collected water (rainy and winter) and sediment matrices.

Table 4 The Shapiro–Wilk test for the normality of bisphenol analogues in water and sediment samples from the Turag River

| Season | Analyte/matrix | W statistic | *p*-Value | Normality decision ($\alpha = 0.05$) |
|---|---|---|---|---|
| Rainy | Bis-B (water) | 0.9107 | 0.2857 | Cannot reject normality |
| | Bis-E (water) | 0.7767 | 0.0075 | Reject normality |
| | Bis-AF (water) | 0.7544 | 0.0040 | Reject normality |
| | Bis-F (water) | 0.9390 | 0.5414 | Cannot reject normality |
| Winter | Bis-B (water) | 0.8972 | 0.2039 | Cannot reject normality |
| | Bis-E (water) | 0.8952 | 0.1940 | Cannot reject normality |
| | Bis-AF (water) | 0.9406 | 0.5595 | Cannot reject normality |
| | Bis-F (water) | 0.9722 | 0.9102 | Cannot reject normality |
| | Bis-B (sediment) | 0.9206 | 0.3622 | Cannot reject normality |
| | Bis-E (sediment) | 0.8376 | 0.0412 | Reject normality |
| | Bis-AF (sediment) | 0.9310 | 0.4569 | Cannot reject normality |
| | Bis-F (sediment) | 0.9410 | 0.5646 | Cannot reject normality |

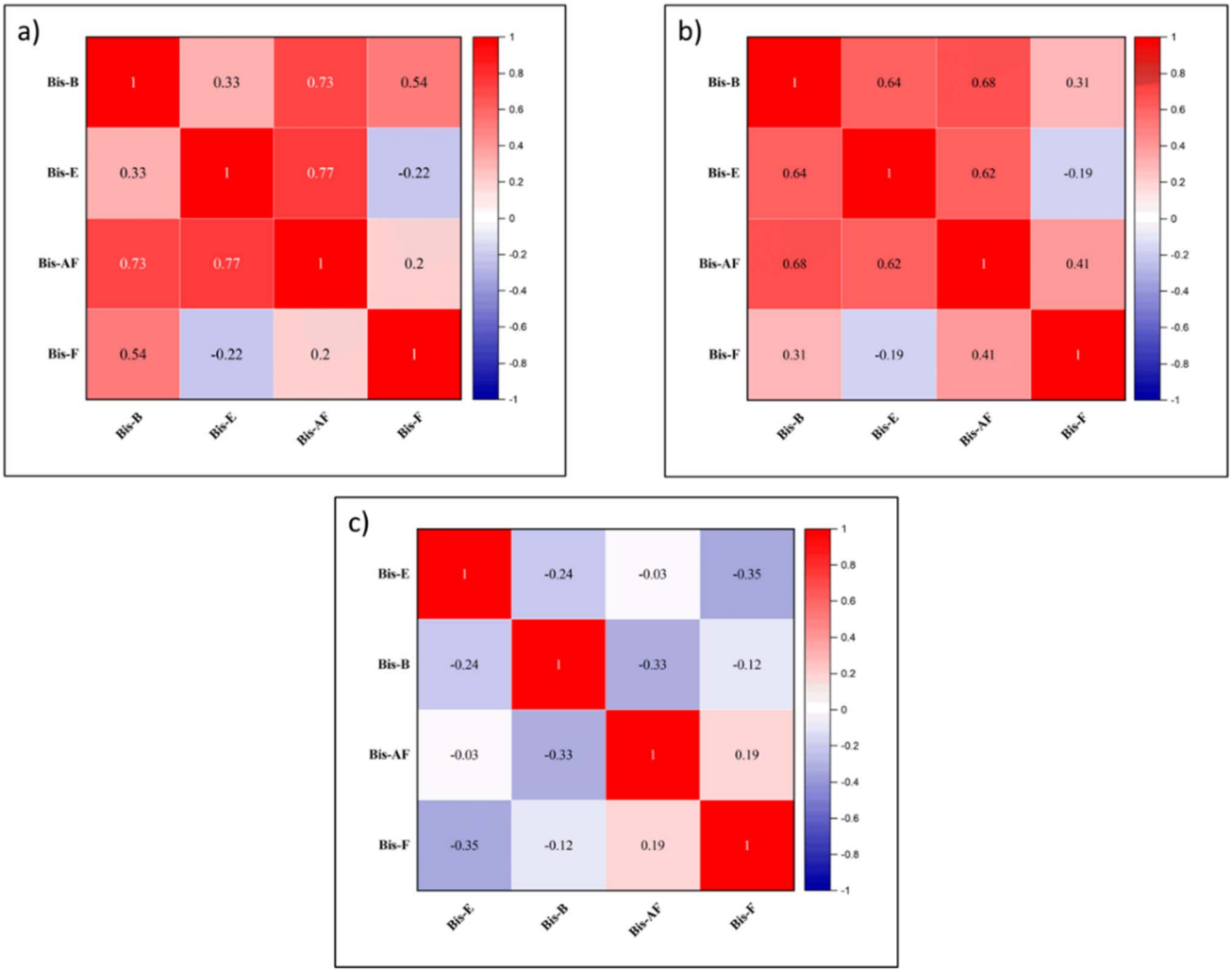


Fig. 5 Correlation matrix of the bisphenol analogues (BPs) across seasonal and sample conditions in the Turag River (a) in the rainy season, (b) in the winter season, and (c) in sediment samples. The figure highlights statistically significant associations, indicating potential shared sources or similar environmental behaviors.

Significant positive correlations were detected between BPB and BPAF ($r = 0.73$, $p < 0.05$), BPE and BPAF ($r = 0.77$, $p < 0.05$), Bis AF and BPB ($r = 0.73$, $p < 0.05$) and BPAF and BPE ($r = 0.77$, $p < 0.05$) in the rainy season. In the winter season, moderate positive correlations were detected between BPB and BPE ($r = 0.64$, $p < 0.05$), BPB and BPAF ($r = 0.68$, $p < 0.05$), BPE and BPB ($r = 0.64$, $p < 0.05$), BPE and BPAF ($r = 0.62$, $p < 0.05$), BPAF and BPB ($r = 0.68$, $p < 0.05$), and BPAF and BPE ($r = 0.62$, $p < 0.05$). Correlation coefficients for the winter season were nearly the same, and correlations were the same for the rainy season. Similar correlations indicated that these BPs may share similar sources or similar environmental behaviors.[44] In sediments, no significant correlations between BPs were observed.

### 3.6. Cluster analysis

A hierarchical cluster analysis was conducted to identify the distinct pollution characteristics of BPs (BPB, BPF, BPAF, BPE) across different seasons (rainy and winter) and environmental compartments (water and sediments). The analysis was performed using the group average linkage method with Euclidean distance as the similarity metric, and the clustering patterns were visualized through a dendrogram (Fig. 6) that displays compound interrelationships and spatial clustering across sampling sites, indicating a distinct environmental dispersion profile and potential anthropogenic or natural input routes. Clusters were interpreted based on the relative distances at which branches merged, enabling identification of groups with comparable contamination profiles. The analysis revealed a significant difference in bisphenol concentrations between sediments and the water column, indicating different environmental behaviors for these compounds. The clusters demonstrated that seasonal variations have a considerable impact on bisphenol levels in the water, generally separating samples collected during the rainy season from those collected in winter. However, BPB concentrations in water remained notably consistent across both seasons. Additionally, a unique and significant clustering was observed between BPF in rainy season

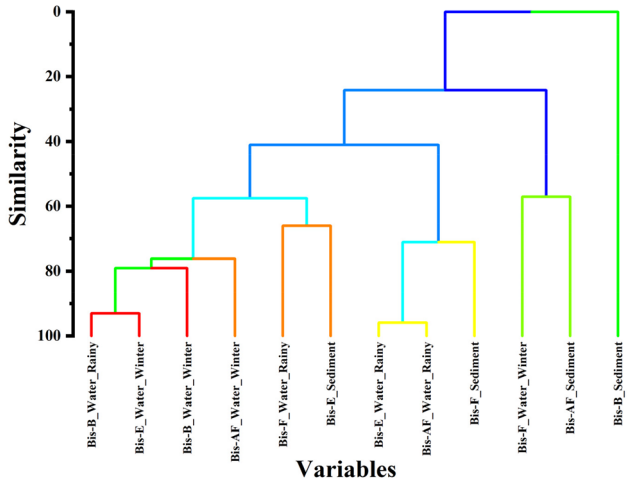


Fig. 6 Hierarchical clustering dendrogram of bisphenol analogues (BPs) in the water and sediment samples from the Turag River (a) in the rainy season, (b) in the winter season, and (c) in sediment samples. The figure highlights compound linkages and spatial grouping patterns, reflecting environmental dispersion and input pathways.

water and BPE in sediment samples. This finding suggests a specific inter-compartmental relationship or shared fate mechanism for these compounds, which warrants further investigation.

The widespread application of bisphenol analogues like BPF, BPAF, and BPB in packaging materials, resins, and consumer goods emphasizes their environmental persistence and notable estrogenic activity.[45] The study highlighted that those structural similarities among these compounds contribute to their shared environmental fate and clustering behavior in aquatic systems.

### 3.7. Spatial distributions of BPs

The spatial dispersion of each bisphenol type (BPB, BPF, BPAF, BPE) was effectively visualized using the spatial analyst tool in QGIS, revealing continuous spatial distribution patterns within the Turag River water and sediments. Fig. 7 indicates spatial and temporal trends in concentration that suggest distinct source contributions and dynamic seasonal transport processes. In the water column along the study area, BPB, BPE, and BPF concentrations consistently showed hotspots around sampling points 4, 5, and 10 during the rainy season. A similar pattern, but with significantly higher overall concentrations, was observed for BPB, BPE, BPAF, and BPF in the winter season. The presence of a drainage outlet between sampling points 4 and 5 may serve as a major point source contributing to bisphenol contamination in the river water. The elevated concentrations observed at points 4 and 5, regardless of the season, directly confirm the impact of this outlet. As a result, multiple bisphenols could be released into the environment through this point *via* rainfall runoff.[46] and industrial discharge.[47]

The consistently high concentrations in the upstream at sampling point 10 further suggest the presence of additional upstream sources, or significant diffuse pollution entering the river before this point. The general increase in bisphenol

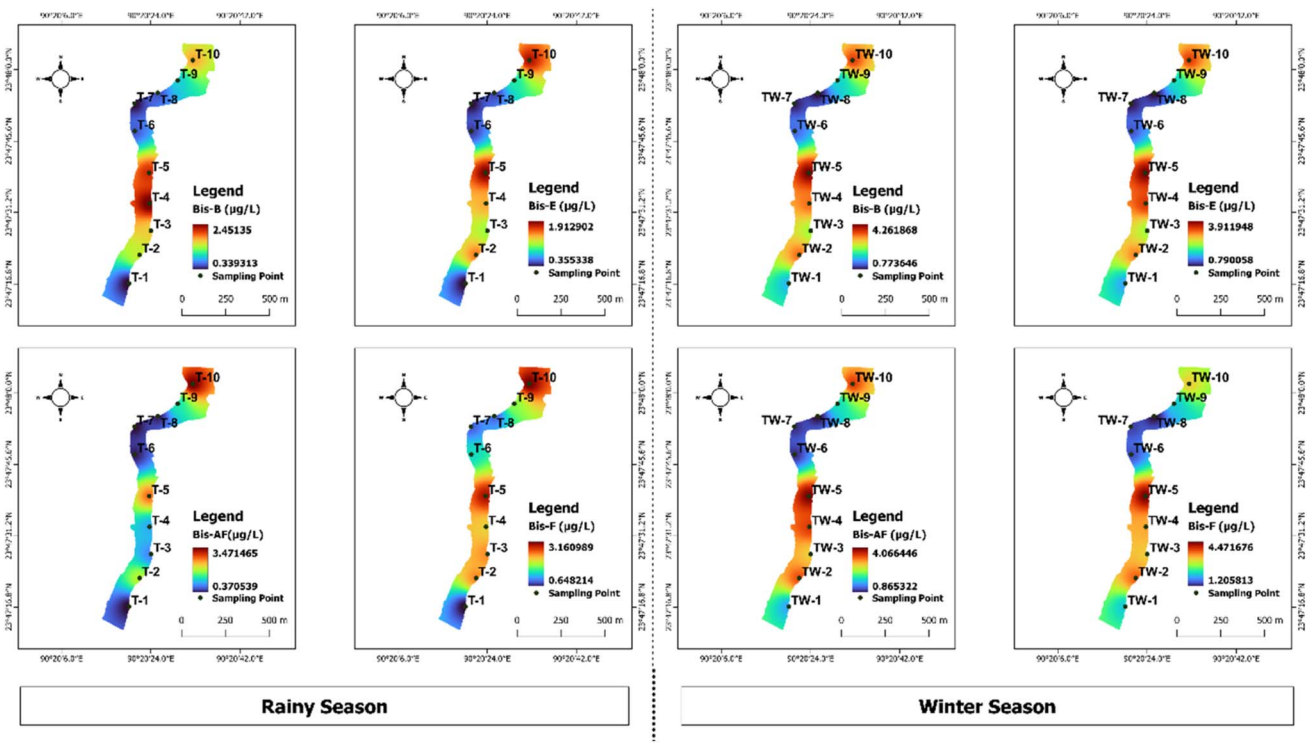


Fig. 7 Spatial arrangement map of different BP concentrations in water in a stretch of the Turag River during the winter and rainy seasons. Concentration patterns suggest differential input sources and seasonal transport dynamics.

concentrations in water during the winter season compared to the rainy season could be attributed to reduced river flow and lower dilution capacity in the dry winter months, concentrating pollutants from continuous discharges. Conversely, the rainy season might lead to some dilution due to increased water volume. Fig. 8 illustrates localized concentration gradients and site-specific accumulation patterns of bisphenol analogues. In the sediment samples, the concentrations of BPB, BPE, and BPF exhibited different spatial hotspots, showing higher values predominantly around sampling points 1 and 10. This distinct pattern, compared to water, highlights the role of sediments as sinks for bisphenol accumulation. High concentrations upstream around sampling point 10 in the sediment reinforce the notion of an upstream source where bisphenols settle out of the water column. The accumulation around sampling point 1, the downstream end of the study area, could be due to sedimentation processes where bisphenols, adsorbed to suspended particles, settle out as the water velocity decreases, or potentially another unidentified local input in the lower reaches of the river.

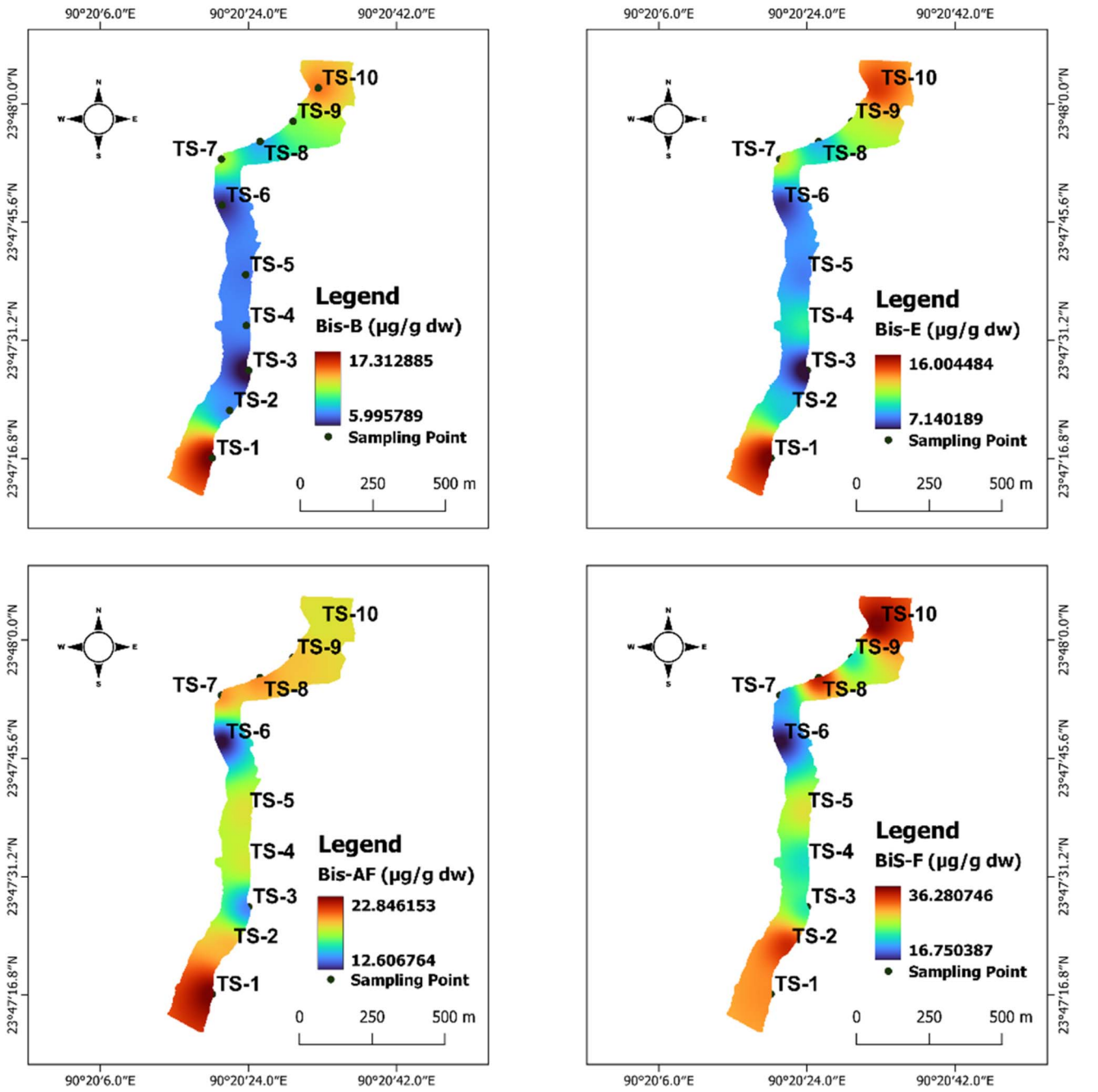


Fig. 8 Spatial distribution map of different BP concentrations in the sediment in a stretch of the Turag River. The figure highlights localized concentration gradients and differential accumulation of BPs across sampling sites.

## 4. Conclusion

Herein, we have assessed the occurrence, dispersion, correlation, and spatial organization of BPs from the Turag River in Dhaka City, Bangladesh. Four BPs (BPB, BPE, BPAF, BPF) were found in the water and sediment fractions. BPB was predominant, and BPE was the least prevailing BP within the collected water and sediment matrices. In water samples, significant positive and moderate positive correlations were observed for BPs, but no significant correlation was detected in sediment samples. BP concentrations of 3.734–234.454 μg $g^{-1}$ dw were found in the sediment, which are remarkably higher than the concentrations of 0.03872 to 9.22404 μg $L^{-1}$ found in water.

## 5. Recommendations

Based on the recent findings, actionable recommendations for environmental monitoring and regulatory policy in Bangladesh include establishing routine surveillance of bisphenol compounds (BPs) in aquatic environments using standardized protocols and sensitive analytical methods. The World Bank Group[48] emphasizes the need to strengthen institutional capacity and improve pollutant tracking systems to support environmental governance. Regulatory frameworks should be updated to include BPs in industrial effluent standards, particularly for sectors involved in plastics and resin production. Additionally, the Bangladesh Biosafety Policy[49] outlines strategies for enhancing chemical safety, which can be extended to endocrine-disrupting compounds like BPs. Developing publicly accessible pollutant databases and expanding policy instruments, such as economic incentives and eco-labeling, are essential for effective implementation. Stakeholder engagement and public awareness campaigns should also be prioritized to foster compliance and promote sustainable practices.

## Conflicts of interest

Authors confirm that there are no financial interests or personal affiliations that could be perceived as influencing the outcomes of this research.

## Data availability

Data will be available upon request from the corresponding author.

Supplementary information is available. See DOI: https://doi.org/10.1039/d5ra04861c.

## Acknowledgements

I offer my sincere gratitude to the Institute of National Analytical Research and Service (INARS), BCSIR, for their full technical assistance and laboratory support, which greatly contributed to demonstrating the significance of this study. Mr Nahid Hasan is especially acknowledged for his invaluable support for statistical analysis.

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