



Occurrence, concentration and distribution of 50 organic contaminants in water and bottom sediment from urban streams affected by stormwater discharges

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ARTICLE INFO

Key words:

Urban runoff
Micropollutants
Urban river
Wet weather
Emerging contaminants

ABSTRACT

Stormwater runoff transports organic contaminants from urban areas to receiving water bodies, yet its contribution to these pollutants in the aquatic environment is still poorly understood. Additionally, contaminants behave differently in receiving waters, with some binding to particles and accumulating in sediments while others stay dissolved in the water. This study was carried out in three Swedish urban streams receiving stormwater discharges through separate sewer systems, under dry and wet weather conditions. Stream water and bottom sediment samples were collected along an urbanization gradient, from rural upstream to urban downstream sections, and analyzed for 50 stormwater-related organic contaminants to assess the impact of stormwater on contaminant levels. Polycyclic aromatic hydrocarbons (PAHs) and phthalates were more prevalent in sediment samples, with concentrations increasing along the urbanization gradient, indicating contributions from urban areas and stormwater runoff. In contrast, organotin compounds and phenols showed no clear pattern indicating transport through stormwater runoff in the water phase. Per and polyfluoroalkyl substances (PFAS) behaved differently from other contaminant groups by exhibiting a clear contribution from stormwater runoff in both phases. Though carried out in streams passing through relatively small urban settings, the findings clearly demonstrate that stormwater discharges can impact receiving waters. Of the 50 analyzed contaminants, three exceeded toxicity-based limits in dry weather (DW), seven in wet weather (WW), and twenty in bottom sediments. In the water phase, under DW and WW conditions, the three contaminants with the highest exceedance of toxicity-based limits were Perfluorooctanesulfonic acid (PFOS), Tributyltin (TBT), and 4-nonylphenol (4-NP). In the sediment phase, 4-tert-octylphenol (4-t-OP), Tributyltin (TBT), and di-2-ethylhexyl phthalate (DEHP) were the three compounds with the highest exceedance of toxicity-based limits. Compared to relatively hydrophilic contaminants (e.g., PFAS), hydrophobic organic contaminants, particularly those accumulating in sediments (e.g., phenols, phthalates), posed a greater risk to the aquatic environment with exceedance levels reaching up to 10^5 times the thresholds. These findings raise concerns about the long-term impact on aquatic environments and highlight the need for mitigation strategies, including regulatory or operational restrictions on the contaminant sources and implementation of stormwater treatment facilities.

1. Introduction

In an urban environment, contaminants originate from many sources including atmospheric deposition, vehicular activity, building and surface materials. Once they enter surface waters (through stormwater discharges or waste water effluents), they may pose risk to aquatic ecosystems, as many are persistent, highly mobile, potentially bioaccumulative and/or toxic (Schwarzenbach et al., 2006). Additionally,

these substances can undergo various transformation processes, resulting in transformation products, some harmless, some potentially more dangerous than the original compounds (Zahn et al., 2024). Due to these potential risks, some (e.g. polycyclic aromatic hydrocarbons (PAHs), alkylphenols) have been designated priority substances under the European Water Framework Directive, with established environmental quality standards for surface water bodies (Directive 2013/39/EU, 2013).

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<https://doi.org/10.1016/j.watres.2025.123847>

Received 19 March 2025; Received in revised form 7 May 2025; Accepted 15 May 2025

Available online 16 May 2025

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Stormwater has emerged as a significant pathway, often conveying a mix of compounds from urban areas (Masoner et al., 2019; Müller et al., 2020; Mutzner et al., 2022; Pamuru et al., 2022). Recent studies have identified numerous organic compounds, including contaminants of emerging concern (CECs) (phthalates, alkylphenols, and per- and poly-fluorinated substances (PFAS)) in urban stormwater (Björklund et al., 2009; Gasperi et al., 2022, 2014; Mutzner et al., 2022; Spahr et al., 2019; Wicke et al., 2021a; Zgheib et al., 2012). Potential sources of contaminants in stormwater runoff vary, depending on the type of chemical compound and urban activity. PAHs are primarily derived from incomplete combustion processes, roofing materials, transportation activities, and road and pavement wear (Müller et al., 2020). Phthalates originate from a wide range of sources, including atmospheric deposition, roofing and façade materials, soft plastics, sealants, tire and road wear (such as paint, asphalt, and concrete), vehicle oil and chemical spills, car components, and automotive care products (Björklund, 2010; Celeiro et al., 2018; Müller et al., 2019). Phenols are typically associated with construction and building materials, commercial car wash detergents, and traffic-related sources (Flanagan et al., 2021; Gasperi et al., 2022; Markiewicz et al., 2017). Organotin is commonly found in paints, sealants, PVC materials, biocides, and surface disinfectants (Blunden and Evans, 1990; Hoch, 2001). Finally, PFAS are introduced into stormwater from building materials, traffic-related sources, consumer products, aqueous film forming foams and lubricants (Glüge et al., 2020; Liu et al., 2024; Zhu and Kannan, 2020). Since stormwater runoff is still often discharged into receiving water bodies with minimal or no treatment, understanding its contribution to water quality degradation is essential, particularly regarding CECs.

Despite research into organic contaminants in stormwater itself, its actual contribution to these pollutants in the aquatic environments remains poorly understood. While several studies have examined the presence of organic contaminants in the urban streams' water phase following runoff events, most are limited in scope, typically focusing on a single water body (Glaser et al., 2023; Meyer et al., 2011; Müller et al., 2021; Peter et al., 2020; Wicke et al., 2021a), one (Müller et al., 2021) or two (Glaser et al., 2023; Meyer et al., 2011) runoff events, and/or combined sewer overflows (i.e. a mixture of stormwater and wastewater) (Glaser et al., 2023; Müller et al., 2021; Paijens et al., 2022). Investigating bottom sediments in urban streams provides valuable insights for predicting the long-term environmental impacts and ecological risks of organic contaminants. However, few studies have focused on bottom sediment in urban rivers (Bai and Son, 2021; Cladière et al., 2013; Teil et al., 2014). Fully understanding the behavior and distribution of stormwater discharges on pollutants in urban aquatic systems requires a study of both the water and sediment phases in parallel. Kali et al. (2025) investigated both the water and sediment phase providing a detailed analysis of PFAS and their precursors. Given the complexity of the stormwater matrices, variable nature of water and sediment quality in urban streams, comprehensive research is needed involving a wide range of contaminant groups with different characteristics.

This study aimed to: i) assess the occurrence and concentration of 50 organic compounds in both the water phase and the bottom sediment across three urban streams with distinct characteristics, ii) analyze how stormwater runoff may influence contaminant levels in these receiving waters by comparing concentrations of these compounds during both dry weather (DW) and wet weather (WW) events, and iii) examine changes in compound concentrations along the streams, from rural up-stream areas to more urbanized downstream sections.

2. Material & methods

2.1. Study areas

This study was carried out in three streams in Sweden flowing through urban areas—Uppsala, Norrköping, and Söderköping. These streams were selected based on their distinct characteristics, including

differences in population density and land use by the cities they flow through, as well as variations in flow conditions and size (i.e. width) (Table 1). Streams' abbreviations are based on the cities they pass through: Fyrisån in Uppsala (Upp), Ljurabäcken in Norrköping (Nrk), and Storån in Söderköping (Sdk). Each city operates a separate sewer system, resulting in stormwater runoff being discharged into the streams through multiple outlets along the watercourse, typically without prior treatment (Kali et al., 2025).

Table 1
Characteristics of study areas.

Water body	Fyrisån	Ljurabäcken	Storån
Municipality	Uppsala	Norrköping	Söderköping
Abbreviation for stream	Upp	Nrk	Sdk
Population (person) ¹	242 000 +/- 1000	145 000 +/- 1000	15 000 +/- 1000
Annual average rainfall (mm) ²	561	568	546
Long-term average daily flow (m ³ /s) ³	7.9	0.6	1.9
Sampling stretch (km)	8.5	7.5	5.5
Average width of the streams along the sampling stretch (m)	ca. 25m	ca. 5m	ca. 18m
Distance between sampling points	UppA - UppB1 = ca. 6.5 km UppB1 - UppB2 = ca. 200 m UppB2 - UppC = ca. 1.8 km	NrkA - NrkB = ca. 5 km NrkB - NrkC = ca. 2.5 km	SdkA - SdkB = ca. 3 km SdkB - SdkC = ca. 2.5 km
Total municipal catchment area draining to the sampling stretch (km ²) ⁴	22.1	11.9	3.8
Total municipal catchment area draining to the sampling sites (km ²) ⁴	UppA = 0 UppB1 = 20.8 UppB2 = 0.8 UppC = 1.3	NrkA = 0 NrkB = 6.2 NrkC = 5.7	SdkA = 0 SdkB = 3.8 SdkC = 0
Distances between sampling points and the closest stormwater outlet ⁴	Upp A = NA* UppB1 = 120 m (upstream) UppB2 = 50 m (upstream) 45 m (downstream) UppC = 240 m (upstream)	NrkA = No outlets upstream and downstream in 1 km range NrkB = 110 m (upstream) 55 m (downstream) NrkC = 255 m (upstream)	SdkA = No outlets upstream and downstream in 1 km range Sdk B = 106 m (upstream) SdkC = No outlets upstream and downstream in 1 km range
Percentage of separate sewers (%) ¹	The sewer system is mostly separated, however data regarding percentages are not available	95	100

Key:

¹ Information gathered from the municipalities.

² Information gathered from the weather measurement stations operated by the Swedish Meteorological and Hydrological Institute (SMHI, 2024).

³ Information gathered from the flow measurement stations operated by the Swedish Meteorological and Hydrological Institute (SMHI, 2024).

⁴ Information gathered from GIS data provided by the municipalities.

* NA: No stormwater outlets are operated by the municipality at these sites. There are military base outlets located upstream of the sampling locations. The exact locations of these outlets are not publicly available.

2.2. Sampling sites

Water and sediment samples were collected at different sites along the river course (Fig. 1). Sites NrKA, SdkA, and UppA upstream of the urbanized areas acted as reference points to represent stream conditions without urban stormwater impact. Sites directly downstream from the main urban areas, NrKB, SdkB, and UppB1, were chosen to assess the potential impact of separate urban storm sewer discharges. The further downstream sites NrKC, SdkC, and UppC were selected to monitor potential dilution effects. The sampling stretches were 5–9 km, depending on the stream (Table 1) (Kali et al., 2025). Details of the study areas are presented in Table 1. Additional information on catchment land use types is available in Supporting Information-I (SI-I Table S1).

Potential point sources were present in the cities (shown in Fig. 1). In Uppsala, a wastewater treatment plant (WWTP) serving 200,000 person equivalents (Uppsala vatten, 2024) discharged into the stream. Thus, an additional sampling site, UppB2, was added approximately 100 m downstream from the WWTP discharge point and 200 m downstream from UppB1. Although upstream of the urbanized area, UppA was potentially affected by a military air force base. In Norrköping, an airport discharged its stormwater into the studied stream between NrKB and NrKC. An industrial area and an old landfill site were located upstream of the sampling site NrKC. The NrKC site was selected as a sampling point located downstream of urban areas and known point sources. Since this stream joins another water body further downstream and is relatively short, there was limited opportunity to establish a site where the effects of dilution could be observed. No known point sources were present in Söderköping (Kali et al., 2025).

2.3. Sampling procedure

Water samples were collected under DW and WW conditions. DW conditions were classified as days with under 1 mm of precipitation, to capture compound concentrations in the streams unaffected by

stormwater runoff. In contrast, WW conditions involved periods of >3 mm of precipitation or snowmelt, with sampling during these events representing concentrations in the streams when impacted by stormwater runoff. WW events captured stormwater runoff, as evidenced by higher total suspended solid and total zinc concentrations observed downstream in urban areas compared to both DW events and the unaffected upstream locations as described by Kali et al. (2025) (SI-I Figure S1).

Samples were collected using autosamplers (Teledyne ISCO 6712) approximately 0.5 m beneath the water surface, following the SS-EN ISO 5667-6:2014 standard. Further, samples were collected from as close as possible to the mid-channel at each site, where mixing tends to be more thorough. Time-proportional samples were collected, with subsamples of 300 mL taken every hour over 24 h for DW events. For WW events, sampling intervals ranged from 15 min to 4 h depending on the event duration, resulting in 20–56 subsamples, with individual subsample volumes ranging from 180 to 360 mL. WW sampling was extended by 1–2 h post-WW events to capture runoff from contributing catchments. The precipitation depth for WW ranged between 3.2 – 39.6 mm and antecedent dry periods ranged from 0 to 31 days (SI-I Table S2). Sub-samples were retrieved within 24 h of sampling, mixed in the field to form a 7.2 L composite sample and transported to the laboratory in cooled bags.

Sampling occurred from June 2022 to October 2023, covering six WW ($n = 6$) and four DW events ($n = 4$) in Uppsala, seven WW ($n = 7$, including one snowmelt event) and three DW events ($n = 3$) in Norrköping, and eight WW ($n = 8$, including one snowmelt event) and three DW events ($n = 3$) in Söderköping.

As sediment quality is usually more stable than water quality, a single sediment sampling event was completed in September 2023, near the aforementioned water sampling sites (maximum distance 550 m; SI-I Table S3). Sediment samples were collected using a powder-coated brass Ekman sediment sampler (Hydrobios Denmark). Several grab samples were taken from each sampling site, composited (9 L bucket) and

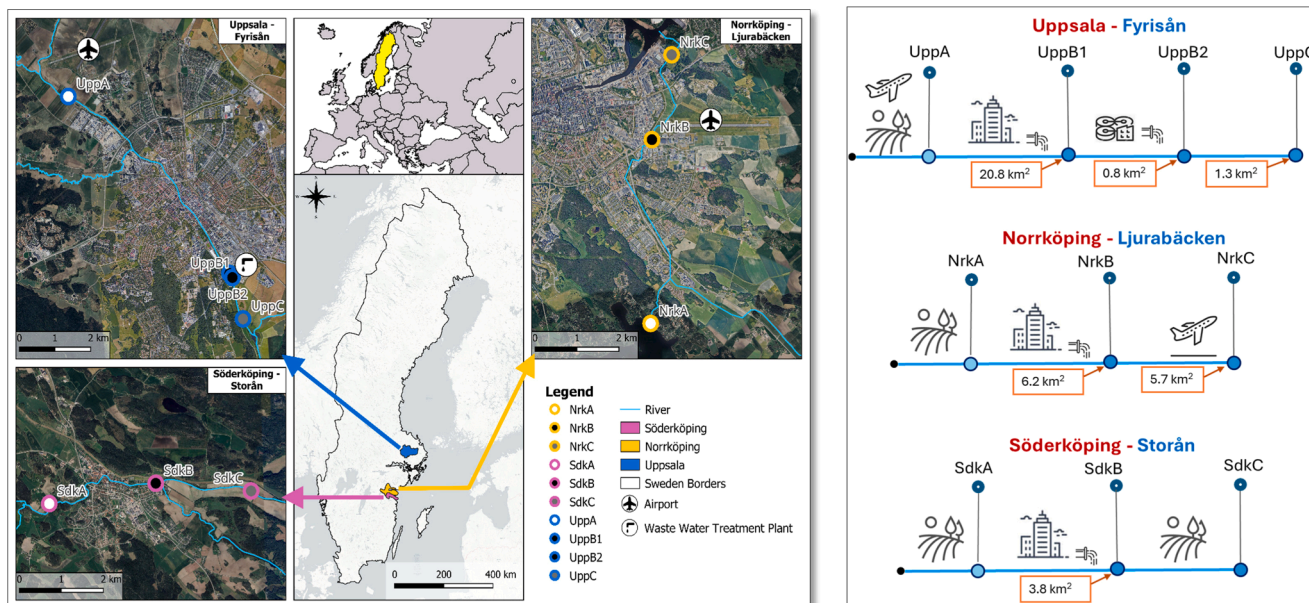


Fig. 1. A map showing the locations of the study areas in Sweden: Uppsala (marked in blue), Norrköping (yellow), and Söderköping (pink). Middle insets provide information on location of Sweden in Europe and location of study areas within Sweden. Outer insets provide details on the sampling sites along the Fyrisån (upper left), Storån (lower left), and Ljurabäcken (lower right), showing the locations of airports and the wastewater treatment plant. Sampling sites are denoted by rings, color-coded to correspond with their respective cities. Reference sites are represented by rings filled with white. Sites positioned in or downstream of urbanized areas are marked with black, whereas those further downstream are depicted with grey. An illustration showing urbanization gradient along each stream represented by the total municipal catchment area draining to each sampling site. Upstream locations typically have minimal or no urban inputs, while downstream sites are increasingly influenced by stormwater runoff from urban areas. (Revised figure from the supporting information of Kali et al. (2025)). (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.).

homogenized in the field. Detailed descriptions of sediment sampling and homogenization procedures are described in Kali et al. (2025).

2.4. Sample analysis

All water and sediment samples were analyzed for various organic compounds, including 13 phthalates, 10 organotin compounds (OTCs), 16 PAHs, and 9 phenols (i.e., nonylphenols, octylphenols and -ethoxylates). Sediment samples were analyzed for eight phenols, excluding Bisphenol A. Contaminants were selected based on literature identifying emerging substances commonly found in both particulate and dissolved phases of stormwater runoff and in sediments from treatment facilities (Flanagan et al., 2021; Gasperi et al., 2022; Masoner et al., 2019; Müller et al., 2020; Mutzner et al., 2022; Wei et al., 2023; Zgheib et al., 2012). In this study, concentrations of PFOS, PFOA, and sum of 35 PFAS from the study by Kali et al. (2025) are presented for comparison with other stormwater-related organic compounds to provide a holistic overview of urban runoff impacts in receiving waters. Details of all analyses (analytical methods and reporting limits (RLs) are presented in Table 2. Physicochemical properties of analyzed organic contaminants are presented in a separate file (SI-II).

Stream water from the respective site was used to rinse all equipment in contact with the samples before each sampling event. Equipment blanks for the ISCO autosamplers were carried out for the two most quantified compound families in water samples, phenolic substances and organotin compounds, at two temperatures, 4 °C and 25 °C, to represent the warm and cold seasons during which sampling occurred. In the equipment blanks MBT was quantified at 2.22 ng/L at 4 °C. Despite that, MBT was not detected in any of the water samples collected during the colder season. Nevertheless, results regarding MBT should be interpreted with caution due to this variability.

All analyses carried out by laboratories accredited by the Swedish Board for Accreditation and Conformity Assessment (SWEDAC), which adheres to international QA standards and implements quality assurance and control measures. The water and sediment samples were analyzed by ALS Scandinavia AB, except for PFAS in sediment, which were analyzed by Eurofins Environment Testing Sweden AB.

2.5. Risk estimation for the streams

The concentrations of contaminants measured in the water phase of streams under DW and WW conditions were compared against the respective Annual Average Environmental Quality Standards (AA-EQS) set by the European Union Water Framework Directive (Directive 2013/39/EU, 2013), when such standards were available.

If no AA-EQS was available for a given substance, the lowest predicted no-effect concentrations (PNEC) from the NORMAN Database System (2025) were used for comparison. For concentrations measured in the bottom sediment, where no official quality standards exist, only the lowest available PNEC values were applied. If neither AA-EQS nor PNEC values were available for a compound, it was excluded from the analysis (e.g., most organotin compounds). Details regarding AA-EQS and PNEC values are presented in SI-I (Table S4).

To assess potential risks, contaminant concentrations measured under DW, WW, and in bottom sediment were divided by their corresponding toxicity-based limits (either AA-EQS or the lowest PNEC values), yielding a ratio as outlined in Eq. (1). Substances with a ratio greater than 1 were classified as critical contaminants, suggesting a potential risk to receiving water bodies. Substances posing the highest risk were identified based on: (1) exceedance rate, which reflects the extent to which a substance's concentration surpasses its toxicity-based limit, and (2) exceedance frequency, which represents how often a substance exceeded the toxicity-based limits, expressed as a percentage.

$$\text{Ratio} = \frac{\text{Contaminant Concentration}}{\text{Toxicity based limit(AA EQS or PNEC)}} \quad (1)$$

Table 2

The analytical methods and reporting limits (RL) for all substances analyzed in water and sediment.

Media	Compound Family	Analysis method	Standard(s)	Compounds (abbreviations, reporting limits – RL- in µg/L for water and in µg /kg dry weight (dw) for sediment samples)
Water	Phenols	GC-MS ^a or GC-MS/MS	Based on ISO 18857-2	4-tert-octylphenol (4-t-OP, 0.01-0.02), 4-tert-octylphenol monoethoxylate (4-t-OP1EO, 0.01 – 0.1), 4-tert-octylphenol diethoxylate (4-t-OP2EO, 0.01-0.627), 4-tert-octylphenol triethoxylate (4-t-OP3EO, 0.01 – 0.02) 4-nonylphenols (tech. mixture) (4-NP, 0.1-0.57), 4-nonylphenol monoethoxylate (4-NP1EO, 0.1 – 0.26), 4-nonylphenol diethoxylate (4-NP2EO, 0.1 – 0.94), 4-nonylphenol triethoxylate (4-NP3EO, 0.1 – 1.09), Bisphenol A (BPA, 0.05 – 0.3)
	OTCs	GC-ICP-MS ^b	According to SE-SOP-0037 (ISO 17353:2005).	Monobutyltin (MBT, 0.001- 0.002), dibutyltin (DBT, 0.001- 0.002), tributyltin (TBT, 0.001), tetrabutyltin (TTBT, 0.001- 0.002), monoocetyltn (MOT, 0.001- 0.002), dioctyltin (DOT, 0.001- 0.002), tricyclohexyltin (TcyT, 0.001- 0.002), monophenyltin (MPhT, 0.001- 0.002), diphenyltin (DPhT, 0.001- 0.002), triphenyltin (TPhT, 0.001- 0.002)
	Phthalates	GC-MS	According to DIN ISO 18856:2005	Dimethyl phthalate (DMP, 1), diethyl phthalate (DEP, 1), di-n-propyl phthalate (DPP, 1), di-n-butyl phthalate (DBP, 1), diisobutyl phthalate (DIBP, 1), di-n-pentyl phthalate (DNPP, 1), di-n-octyl phthalate (DNOP, 1), di-2-ethylhexyl phthalate (DEHP, 1), butylbenzylphthalate (BBP, 1), dicyclohexyl phthalate (DCP, 1), diisodecyl phthalate (DIDP, 10), diisononyl phthalate (DINP, 10), di-n-hexylphthalate (DNHP, 1)
	PAHs	GC-MS or GC-MS/MS	According to US EPA 8270D, US EPA 8082A, CSN EN ISO 6468 and US EPA 8000D	Naphthalene (NAP, 0.03), acenaphthylene (ACY, 0.01), acenaphthene (ACE, 0.01), fluorene (FL, 0.01), phenanthrene (PHE, 0.01), anthracene (ANT, 0.01),

(continued on next page)

Table 2 (continued)

Media	Compound Family	Analysis method	Standard(s)	Compounds (abbreviations, reporting limits – RL- in µg/L for water and in µg /kg dry weight (dw) for sediment samples)
				fluoranthene (FLA, 0.01), pyrene (PYR, 0.01), benzo[a]anthracene (BaA, 0.01), chrysene (CHR, 0.01), benzo[b]fluoranthene (BbF, 0.01-0.06), benzo[k]fluoranthene (BkF, 0.01-0.07), benzo[a]pyrene (BaP, 0.01), dibenzo[a,h]anthracene (DahA, 0.01), benzo[g,h,i]perylene (BPer, 0.01), indeno[1,2,3-cd]pyrene (IcdP, 0.01)
	PFAS	LC-MS/MS ^c	Based on US EPA 537 and CSN P CEN/TS 15968 or US EPA 533	Perfluorooctanoic acid (PFOA 0.0003), Perfluorooctanesulfonic acid (PFOS, 0.0003)
Sediment	Phenols	GC-MS or GC-MS/MS	Based on EU Standard BT WI CSS99040	4-tert-octylphenol (4-t-OP, 10), 4-tert-octylphenol monoethoxylate (4-t-OP1EO, 10-13), 4-tert-octylphenol diethoxylate (4-t-OP2EO, 10), 4-tert-octylphenol triethoxylate (4-t-OP3EO, 10-19), 4-nonylphenols (tech. mixture) (4-NP, 100-120), 4-nonylphenol monoethoxylate (4-NP1EO, 100), 4-nonylphenol diethoxylate (4-NP2EO, 100), 4-nonylphenol triethoxylate (4-NP3EO, 10-24)
	OTCs	GC-ICP-MS	According to SE-SOP-0036 (SS-EN ISO 23161:2018)	Monobutyltin (MBT, 1), dibutyltin (DBT, 1), tributyltin (TBT, 1), tetrabutyltin (TTBT, 1), monoethyltin (MOT, 1), dioctyltin (DOT, 1), tricyclohexyltin (TcyT, 1), monophenyltin (MPHT, 1), diphenyltin (DPHT, 1), triphenyltin (TPHT, 1)
	Phthalates	GC-MS	According to DIN 19742:2014-08	Dimethyl phthalate (DMP, 50), diethyl phthalate (DEP, 50), di-n-propyl phthalate (DPP, 50), di-n-butyl phthalate (DBP, 50), diisobutyl phthalate (DiBP, 50), di-n-pentyl phthalate (DNPP, 50), di-n-octyl phthalate (DNOP, 50-300), di-2-ethylhexyl phthalate (DEHP, 50), butylbenzylphthalate (BBP, 50), dicyclohexyl phthalate (DCP, 50), diisodecyl phthalate (DIDP, 2500), diisononyl phthalate (DINP, 2500), di-n-hexylphthalate (DNHP, 50)

Table 2 (continued)

Media	Compound Family	Analysis method	Standard(s)	Compounds (abbreviations, reporting limits – RL- in µg/L for water and in µg /kg dry weight (dw) for sediment samples)
	PAHs	GC-MS	Based on SS-EN ISO 18287:2008	Naphthalene (NAP, 100), acenaphthylene (ACY, 100), acenaphthene (ACE, 100), fluorene (FL, 100), phenanthrene (PHE, 100), anthracene (ANT, 100), fluoranthene (FLA, 100), pyrene (PYR, 100), benzo[a]anthracene (BaA, 50), chrysene (CHR, 50), benzo[b]fluoranthene (BbF, 50), benzo[k]fluoranthene (BkF, 50), benzo[a]pyrene (BaP, 50), dibenzo[a,h]anthracene (DahA, 50), benzo[g,h,i]perylene (BPer, 100), indeno[1,2,3-cd]pyrene (IcdP, 50)
	PFAS	LC-MS/MS	Based on DIN 38414-14 and Powley et al. (2005)	Perfluorooctanoic acid (PFOA 0.03 - 0.078), Perfluorooctanesulfonic acid (PFOS, 0.03)

^a Gas chromatography-mass spectrometry.

^b Gas chromatography – inductively coupled plasma analysis – mass spectrometry.

^c Liquid chromatography-mass spectrometry.

2.6. Data treatment

Descriptive statistics, including minimum, maximum, and median concentrations, were calculated for compounds based on each sampling site. Matrix effects in some samples led to higher RLs for certain compounds, resulting in a left-censored dataset (Helsel, 2012). Therefore, appropriate statistical methods for left-censored data were applied, using the Nondetects and Data Analysis for Environmental Data (NADA and NADA2) package in R (Julian and Helsel, 2024). Median concentrations were calculated using the robust regression on order statistics method (Helsel, 2012). All statistical analyses, including descriptive statistics, conducted in R and were limited to compounds detected in at least 20 % of the samples ($DF \geq 20$).

3. Results and discussion

3.1. Occurrence and concentration of compounds

Under DW conditions, eight out of the 50 targeted compounds were detected (i.e. above detection limit) in water samples at least once (Fig. 2). These included three OTCs (DBT, MBT, TBT), two phenols (BPA, 4-NP), one phthalate (DEHP), and two PFAS (PFOA, PFOS). During WW events, the number of detected compounds increased to 15, comprising six OTCs (DBT, DOT, MBT, MOT, TBT, TTBT), four PAHs (BbF, BkF), three phenols (BPA, 4-NP, 4, 4-NP2EO), and two PFAS (PFOA, PFOS) (Fig. 2). These findings suggest that stormwater discharges introduce a greater variety of contaminants, including PAHs and OTCs, into the streams during WW.

In bottom sediments, 29 out of the 50 analyzed compounds were detected in at least one sample (Fig. 2), with OTCs, PAHs, and phthalates being particularly prevalent, indicating that sediments act as a sink for these contaminants. OTCs were present in both water and sediment, but

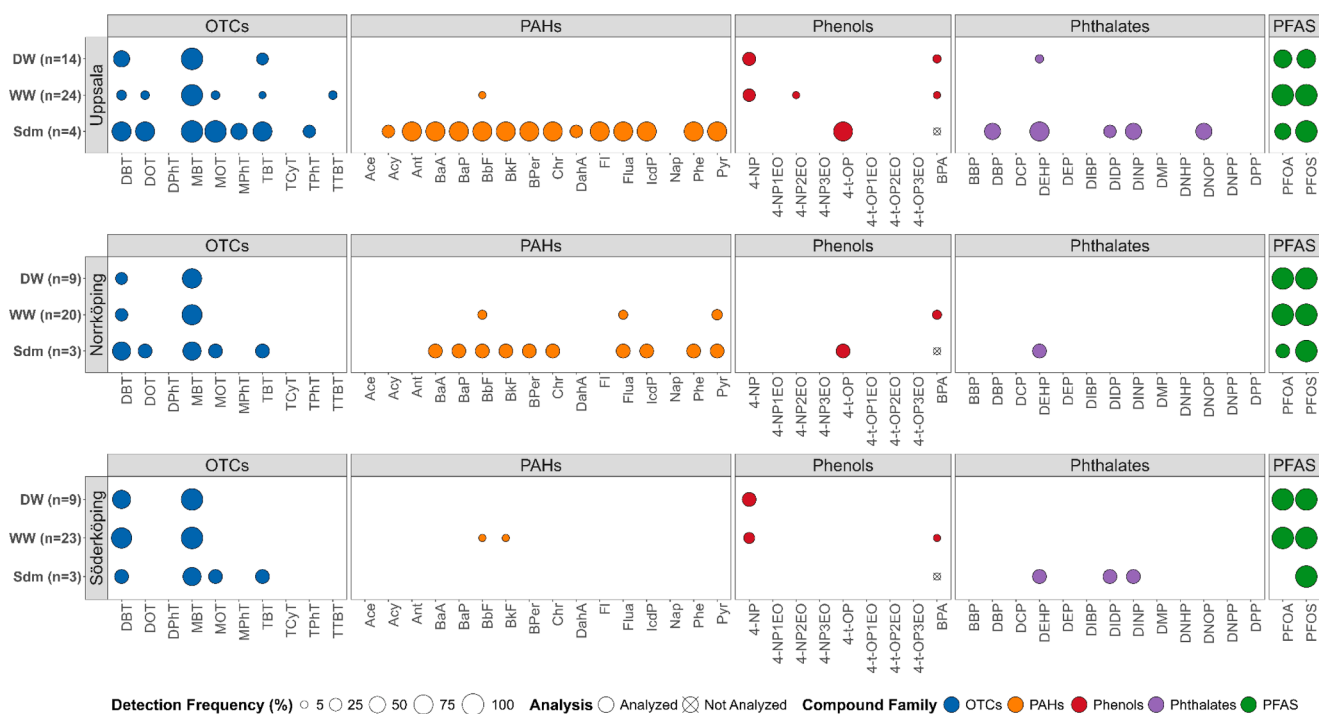


Fig. 2. Detection frequencies (%) of compounds from various families (OTCs, PAHs, Phenols, Phthalates, and PFAS) across three sampling sites (from top: Uppsala, Norrköping, Söderköping) under different conditions: dry weather (DW), wet weather (WW), and sediment (Sdm). The size of each circle represents the detection frequency (%) of the compound, with larger circles indicating higher frequencies. Compounds are grouped by family and analyzed across water (DW and WW) and sediment (Sdm) samples, while compounds not analyzed are indicated with a cross (BPA in sediment). (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.).

were predominantly found in sediments, as with PAHs and phthalates. PFAS, however, were consistently detected across all sample types, while phenols showed no clear distribution pattern.

Fig. 3 shows the median concentrations measured in sampling sites under DW and WW conditions in the water phase and measured concentrations in the sediment phase. The partly higher concentrations observed under DW compared to WW conditions suggested that stormwater discharges diluted the median concentrations of OTCs and phenols (Fig. 3). In contrast, PFAS (e.g. PFOA, PFOS) showed a clear increase during WW events, being particularly evident when considering the increase in median concentration of sum of all quantified PFAS (total of 34 PFAS) under WW across all sampling sites (Fig. 3 and SI-III).

Analysis of bottom sediments revealed that all analyzed compound families (OTCs, PAHs, phenols, phthalates, and PFAS) followed the urbanization gradient, with higher concentrations – or only reported concentrations – downstream of urban areas or near known point sources (e.g. airports, industrial areas). Although stream water samples during WW discharges did not show elevated concentrations of stormwater-driven and relatively hydrophobic compounds (PAHs and phthalates), the pattern observed in bottom sediments indicates that urbanization – and consequently, stormwater runoff – contributes to their accumulation in stream sediments. These findings emphasize that relying solely on water samples does not provide a comprehensive assessment of stormwater impact. Accumulation of pollutants (e.g. PAHs and phthalates) in bottom sediments of the streams reveals a clear stormwater impact.

3.1.1. OTCs

Of the 10 OTCs analyzed, MBT was detected in over 85 % of the water samples across all streams under both wet and dry weather conditions. Overall, MBT was the most abundant OTC in the water samples, followed by DBT which was also quantified in all streams but with varying frequencies (Fig. 2). Concentrations varied across the streams and sampling sites within each stream. Median MBT concentrations

were 3.64–259 ng/L in DW and 3.01–62.1 ng/L in WW. For DBT, median concentrations were 1.31–12.5 ng/L in DW and 0.05–4.82 ng/L in WW (Fig. 3). DBT showed a dilution pattern during runoff events across all streams, while MBT did not follow a pattern, with concentrations varying between sampling sites along the streams.

The sources of OTCs compounds in urban environments can include PVC pipes, paints, automotive parts, pesticides, wood preservatives, and food packaging (Fromme et al., 2005; Hoch, 2001). In addition, the occurrence of MBT and DBT may also be linked to the microbial or photochemical degradation of TBT (Hoch, 2001). TBT, a compound formerly used as a biocide in antifouling paints on ships and boats, was listed as a priority hazardous substance and banned in the EU in 2003 (Regulation (EC) No 782/2003). TBT was quantified in Uppsala, although at low frequencies (21 % in DW and 4 % in WW) and predominantly in DW samples, showing a clear dilution trend with WW discharges (concentrations below reporting limits in almost all WW samples). The streams in Uppsala and Söderköping are used for recreational sailing, a potential source of organotin compounds. In Norrköping, the stream is not suitable for sailing, thus, other MBT and DBT sources must dominate. For example, the highest median concentration measured in this study for MBT (259 ng/L) during DW was measured at NrkC, pointing to the presence of a continuous source (unidentified) impacting the stream under DW conditions.

The occurrence and concentrations of MBT and DBT under DW and WW conditions highlighted the variability in OTC concentrations along the streams, with stormwater and potential local sources impacting each site differently. For example, in Uppsala, at UppB2, median MBT and DBT concentrations during DW were approximately 1.6 times lower than at UppB1, despite the short distance between these sites, implying that the WWTP discharge might dilute OTC concentrations in the stream. In Söderköping, unlike other streams, results indicated that stormwater contributed to the MBT levels in the stream. Notably, MBT concentrations were consistently higher along all sampling points in Söderköping compared to other streams. This was unexpected, given

Organotin concentrations in river sediments vary across different rivers due to differences in urbanization, pollutant sources, and hydrological processes. In Norway, riverine particles originating from urban runoff, erosion, and wastewater discharges showed higher concentrations, as reported by [Cornelissen et al. \(2008\)](#): 60–120 µg/kg for MBT, 95–210 µg/kg for DBT, and 60–230 µg/kg for TBT. Contrastingly, [David et al. \(2012\)](#) found lower OTC concentrations (sum of MBT, DBT, and TBT: 0.4–24 µg/kg dw) in the bottom sediment of the intermittent Vène River, which was impacted by urban runoff and domestic WWTP. Organotin concentrations in the bottom sediment decreased during high-flow periods, suggesting that stormwater transports contaminated sediments and pollutants downstream ([David et al., 2012](#)). However, present study found that organotin levels remained high near urban and industrial areas which indicates either reduced flushing or more dominant local contamination sources (e.g. stormwater outlets). Similarly, [Cornelissen et al. \(2008\)](#) demonstrated that riverine particles transport organotin compounds from urban areas into Norwegian harbors, emphasizing the contribution of urban runoff in conveying OTCs.

3.1.2. PAHs

PAHs were prevalent in bottom sediments, but their presence varied by stream, reflecting differences in urbanization and industrial activity. Of the 16 PAHs analyzed, 14 were detected in at least one sediment sample from Uppsala, 10 in Norrköping, and none in Söderköping. In Uppsala, 12 of the 14 quantified PAHs were found at three of the four sampling sites. However, no PAHs were detected at the most upstream site (UppA), i.e. the contamination originates from downstream urban or industrial sources rather than from areas further upstream. The highest total PAH-16 concentration (9.5 mg/kg dw) was measured at UppB1, directly downstream of an urban area. Similarly, in Norrköping, PAHs were only quantified at the sampling site (NrC) located downstream of urban areas, a local airport, an industrial zone, and an old landfill. No PAHs were quantified in Söderköping, likely due to its smaller urban footprint and lack of significant industrial activity. Additionally, the stream in Söderköping is proportionally larger relative to its contributing urban area compared to those in Norrköping and Uppsala ([Table 1](#)), potentially leading to greater dilution.

The observed total PAH-16 concentrations (<RL-9.5 mg/kg dw) and trends (urbanization gradient) in this study were consistent with previous research on bottom sediment ([Valentyne et al., 2018](#)) carried out on three streams the Root River, the Plover River, and the Eau Claire River, in Wisconsin, USA, which flow from relatively undeveloped upstream areas through developed urban regions. Reported concentrations were <RL to 12.4 mg/kg dw. Both studies observed an increase in PAH-16 concentrations along the urbanization gradient, with higher levels in bottom sediments downstream of urbanized areas.

Diagnostic ratios ([Baldwin et al., 2017](#); [Yunker et al., 2002](#)), such as LMW to HMW PAHs, were calculated for all bottom sediment samples (with no non-detects) to identify potential sources of PAHs in the bottom sediment (SI-I Table S5). The calculated ratios (LMW/HMW <0.30) suggested that most PAHs in bottom sediment were likely derived from combustion processes, particularly vehicle emissions ([Soclo et al., 2000](#)), indicating that stormwater runoff is a vector transporting PAHs from urban areas to the streams. PAH concentrations are typically high in runoff from catchments with roads and vehicle-related activities ([Gasperi et al., 2022](#); [Wicke et al., 2021a](#)). In this study, the highest or only quantifiable PAH concentrations in bottom sediments were measured downstream of urban areas, where receiving waters collect runoff from roads and vehicle-related sources.

In stream water, none of the 16 PAHs analyzed exceeded the RL in any stream under DW conditions. Also, under WW conditions, a few PAHs (e.g. BbF, BkF, Flua, Pyr -[Fig. 2](#)) were rarely quantified (DF<25). Given that PAHs are common stormwater contaminants ([Mutzner et al., 2022](#)), with vehicle-related (i.e. exhaust emissions, tire wear, and road abrasion) and atmospheric deposition being their primary sources ([Müller et al., 2020](#)), most probably they should be present in the

stormwater discharged in the three investigated catchments. Thus, the non-detection (i.e. concentrations <RL) is likely due to dilution effects in the streams. Due to their hydrophobicity, PAHs are typically found in the solid phase or absorbed by sediments ([Mackay et al., 2006](#)). The relatively low TSS concentrations in the streams during WW (median concentration of TSS was 3.2–41.5 mg/L), compared to typical TSS concentrations observed in stormwater (mean TSS = 104mg/L ([Pamuru et al., 2022](#))) may partly explain the low PAH detection frequency observed. However, studies by [Hwang and Foster \(2006\)](#) and [Kim and Young \(2009\)](#) found similar TSS levels in urban streams during runoff events, yet PAHs were still detected, suggesting that TSS alone does not fully explain the differences. Factors, such as the size of the contributing catchment, proximity to stormwater outlets, and the hydrology and morphology of the streams (e.g. flow rate, width, and depth) may also influence PAH occurrence. Nevertheless, [Hwang and Foster \(2006\)](#) and [Kim and Young \(2009\)](#) suggested that stormwater is a significant source of elevated PAH concentrations during runoff events in receiving waters. The findings of this study showed a similar pattern, but in the sediment phase rather than water phase.

The median concentrations were calculated for Pyr, BbF, and sum of PAHs only for NrC site due to DF (i.e. DF>20 %). The median concentrations were 0.009 µg/L for Pyr, 0.007 µg/L for BbF, and 0.023 µg/L for total PAH-16. The total median PAH concentrations under WW in the observed in this study were up to 543 times lower than those reported in the tidal Anacostia River, Washington, DC, USA, during runoff events (\sum PAHs: 1.51–12.5 µg/L), where river water samples were collected from six branches along the lower river during WW events, including sites near industrial and urban contamination sources such as the Washington Navy Yard and a fossil fuel-based power plant ([Hwang and Foster, 2006](#)). Similarly, PAH concentrations in this study were 165 times lower than those measured during storm events in Arcade Creek, an urban tributary to the Sacramento River in California, USA (\sum PAHs: 0.13–3.79 µg/L), which receives runoff from residential and commercial areas ([Kim and Young, 2009](#)). Total PAH-16 concentrations in this study were up to 140 times lower than those in the Panke River, Berlin, a small urban river that receives stormwater discharges from separate sewer systems during wet weather events (\sum PAHs: 0.094–4.8 µg/L) ([Wicke et al., 2021a, 2021b](#)). Low concentrations measured in the streams we studied may be due to the differences in urbanization levels, as the other regions studied (Seattle, Washington, DC, and Berlin) are metropolitan cities, whereas the streams we studied are in less densely urbanized areas.

3.1.3. Phenolic substances

In bottom sediment, 4-t-OP was the only phenols quantified with 75 % detection frequency in Uppsala and 33 % detection frequency in Norrköping. Notably, although 4-NP is more hydrophobic ([Mackay et al., 2006](#)) and accounts for a larger proportion of industrial applications than OP ([Bergé et al., 2012](#); [Mackay et al., 2006](#)), it was not detected in the bottom sediment of any of the studied streams. This lack of quantification might be related to 4-NP's higher reporting limit compared to 4-t-OP for sediment ([Table 2](#)). The concentrations of 4-t-OP were 0.11–1.57 mg/kg dw ([Fig. 3](#)) and followed the same pattern as PAHs, with the highest concentrations measured directly downstream of the urban area (UppB1) and potential sources (i.e. industrial area) (NrC). Notably, upstream sampling sites and the Söderköping stream, all of which are less impacted by urban and industrial impacts, showed no detectable levels of 4-t-OP. This suggests that stormwater may contribute to 4-t-OP concentrations in streams, as sediments and particles from stormwater also indicated an association between high OP concentrations and traffic-related activities ([Flanagan et al., 2018](#); [Gasperi et al., 2022](#); [Markiewicz et al., 2017](#); [Müller et al., 2020](#)).

Of phenolic substances, 4-NP (DF: 29 % in Uppsala, 33 % in Söderköping) and BPA (DF: 7 %, only in Uppsala) were the only compounds quantified in the stream water under DW. Under WW conditions, 4-NP occurrence decreased, and BPA was rarely quantified (DF < 20 %)

in any stream. 4-t-OP, with a reporting limit 10 times lower than 4-NP, was not detected in any stream water samples. Median 4-NP concentrations were 0.10–0.15 µg/L in DW, and 0.06–0.11 µg/L in WW, suggesting a dilution pattern in the streams under WW events. BPA concentrations were <RL–0.09 µg/L in DW and <RL–0.33 µg/L in WW. Out of 99 samples analyzed, one alkylphenol ethoxylate (4-NP2EO) was detected above the RL in just one instance, indicating they were almost never present at reportable levels, possibly related to their transformation to alkylphenols after release (Thiele et al., 1997).

The 4-NP concentrations in the studied streams (<RL–0.268 µg/L in dry weather, <RL–0.333 µg/L in wet weather) were higher than those measured in the Seine River, France (0.028–0.157 µg/L, median 0.061 µg/L), where 4-NP levels remained stable along the river from rural upstream to downstream of Paris (Cladière et al., 2013). This is particularly notable given that the Seine flows through a heavily urbanized area (Paris), whereas the streams analyzed in this study pass through less urbanized regions with lower population densities. Despite this difference in urbanization levels, the 4-NP concentrations measured in the water phase in both studies suggest that sources other than direct urban discharges – such as groundwater and historical contamination – may contribute to 4-NP levels in urban streams. Also WWTP can be a significant source of 4-NP to receiving waters (Zhao et al., 2021). However, concentrations measured in Uppsala (UppB1 vs. UppB2) showed no contribution from WWTP to the stream.

3.1.4. Phthalates

Of 13 phthalates, five were quantified in at least one of the sediment samples from Uppsala, one in Norrköping, and three in Söderköping. These quantified phthalates included DIDP, DINP, DEHP, and DNOP (Fig. 2), all of which are high molecular weight compounds with more hydrophobic characteristics, except for DBP, which has a relatively lower molecular weight (Mackay et al., 2006). DEHP was the only phthalate quantified in at least one sediment sample from all streams. As with other compound families (PAHs, OTCs) in this study, DEHP concentrations measured in the bottom sediment followed an urbanization gradient, with the highest or sole concentrations consistently found downstream of urban and/or industrial areas (e.g. UppB1, SkdB, NrC). Other phthalates, including DBP, DNOP, DIDP, and DINP, also exhibited this pattern.

DEHP concentrations in the bottom sediments of the studied streams were 0.1–3.6 mg/kg dw, while DIDP and DINP levels were <RL–4.90 mg/kg dw and <RL–4.40 mg/kg dw, respectively. A study in the Seine River (Paris) and its tributary, the Orge River, reported DEHP concentrations of 0.082–16.5 mg/kg dw in the Seine and 0.062–0.983 mg/kg dw in the Orge, with levels increasing 202-fold downstream of Paris, highlighting the influence of urban runoff, WWTP discharge, and industrial activity (Teil et al., 2014). Similarly, an analysis of suspended particulate matter from major German rivers (e.g. Rhine, Elbe, and Danube) found DEHP (0.225–2.08 mg/kg dw), DIDP (0.074–2.070 mg/kg dw), and DINP (0.101–4.150 mg/kg dw), identifying urban runoff and WWTP as primary contamination pathways, particularly in industrialized and densely populated regions (Nagorka and Koschorreck, 2020).

The findings of this study align with these previous studies, showing a clear upstream-to-downstream increase in phthalate concentrations in the bottom sediment, with the highest levels – or the only detectable concentrations – found downstream of urban areas (UppB1, SdkB) and known pollution sources such as WWTPs (UppB2) (Fig. 3). Further, the phthalates detected in this study have also been found in previous research in stormwater sediments and particulate matter (Björklund et al., 2009; Flanagan et al., 2021; Wei et al., 2023; Zgheib et al., 2012), further supporting the connection between stormwater runoff and the introduction of phthalates into urban streams. While DEHP concentrations in this study were generally within the range observed in German rivers, they were lower than those reported in the Seine River, where contamination was amplified by urbanization. This reinforces the notion

that urban runoff and point sources (e.g. WWTPs) are drivers of phthalate pollution, with the extent of contamination dependent on local land use, population density, and industrial activity.

In water, only DEHP was quantified, and this occurred on a single occasion (at DW in Uppsala). No phthalates exceeded the RL in any WW samples across the streams. While phthalates from various sources (e.g. plastics, tires, roofs, façades, and traffic-related activities) are present in stormwater (Björklund et al., 2007; Gasperi et al., 2022; Müller et al., 2022; Wicke et al., 2021a), their concentrations in streams may fall below RLs due to dilution and settling processes. Since phthalates tend to adsorb onto particulates, lower TSS concentrations might have limited their quantification in the water column (Mackay et al., 2006).

3.1.5. PFAS

PFAS concentrations, unlike other organic substances, consistently increased during WW discharges, highlighting their strong association with urbanization and stormwater runoff (Kali et al., 2025). Notably, PFAS concentrations – including PFOA, PFOS, and sum of quantified PFAS – not only rose with WW discharges but also exhibited a clear pattern with urbanization gradients in both water and sediment. In contrast, other organic substances typically showed such trends only in sediment, not in stream water. This distinct behavior highlights the unique transport potential of PFAS through stormwater runoff, driven by their dual hydrophilic and hydrophobic properties and widespread use in urban environments, which was particularly evident when compared to other organic contaminants analyzed in this study.

3.2. Risk assessments for streams

Comparing measured concentrations to Predicted No-Effect Concentration (PNEC), the results for bottom sediments were particularly concerning. In the sediments, 20 out of 50 compounds included in this analysis exceeded their corresponding PNEC values at least once. The concentrations observed in the sediments exhibited both greater exceedance rates and higher exceedance frequencies compared to concentrations of the same compounds measured in water (both under DW and WW). The critical substances identified in the sediment included 4-t-OP, TBT, DEHP, PAHs (Ant, BaP, BkF), and PFOS, with maximum exceedance rates of 140–10⁵ (Fig. 4 and SI-I Table S6). Since DW conditions reflect the existing state of streams, AA-EQS values, which are more relevant for chronic exposure, were used to assess measured concentrations. To compare potential risks under WW conditions, the same AA-EQS values were applied. While stormwater discharges are episodic and thus Maximum Allowable Concentrations (MAC) values might be used for such short-term exposures, the goal here was to highlight differences between DW and WW conditions. Therefore, AA-EQS values were also used for WW. Moreover, because AA-EQS values are more conservative, substances not exceeding them are unlikely to pose risks under MAC thresholds either (SI-I Table S4).

Concentrations in the water phase, compared to their respective AA-EQS or PNEC, showed lower exceedance rates and frequencies under both dry and wet conditions than in bottom sediments (Fig. 4). The most critical compounds under DW conditions were PFOS, NP, and TBT, both in terms of exceedance rates and frequency (Fig. 4), with maximum exceedance rates 1.3–55 times the AA-EQS (SI-I Table S7).

As WW discharges introduced a broader range of compounds to the streams (Fig. 2), the number of substances exceeding toxicity-based limits (i.e. AA-EQS or PNEC) in water increased with stormwater discharges. For instance, the concentrations of PAHs (BbF, Pyr, Fla) and BPA exceeded their respective AA-EQS and PNEC during at least one WW event in the streams. A total of seven out of 50 contaminants (PFOS, NP, BbF, Pyr, Chr, TBT, and BPA) exceeded toxicity-based limits. PFOS was the most concerning compound, with maximum exceedance rates reaching 60 times the AA-EQS (Fig. 4 and SI-I Table S8).

There are several factors explaining the notable difference in exceedance rates between bottom sediments and the water phase. For

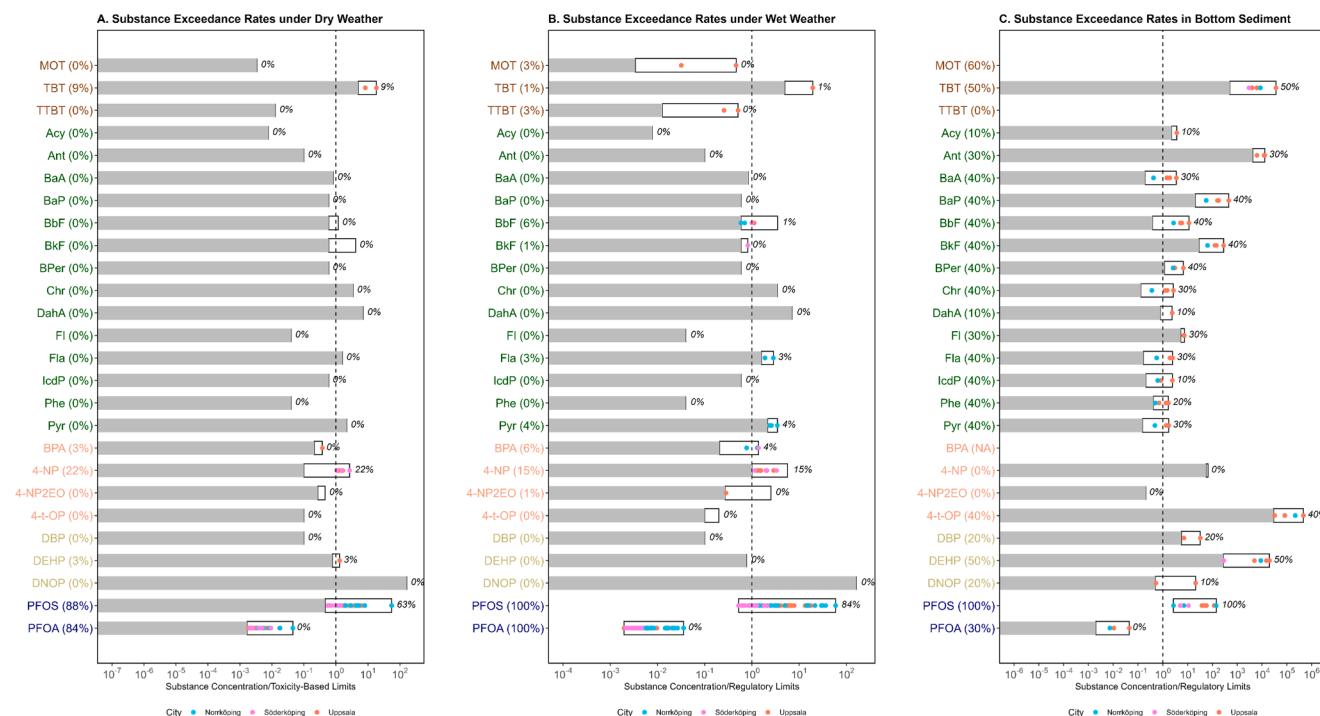


Fig. 4. Substance exceedance rates under dry (A) and wet weather (B) conditions, as well as in bottom sediment (C), for various substances across three urban streams: Norrköping (blue), Söderköping (pink), and Uppsala (orange). The x-axis represents the ratio of substance concentrations to their respective AA-EQS or PNEC on a logarithmic scale. The y-axis lists individual substances, color-coded by substance group (OTCs: brown, PAHs: green, Phenols: salmon, Phthalates: yellow, PFAS: blue). Detection frequencies (percentage of samples in which a substance was quantified) are shown next to each substance name. White bars indicate the range (minimum to maximum) of quantified concentrations (i.e. concentrations above reporting limits), while extended grey bars represent values below reporting limits. Percentages next to the bars indicate the overall exceedance rates. The dashed vertical line represents the critical threshold of 1. Ratios exceeding this threshold indicate that the substance concentration surpasses the AA-EQS or PNEC, suggesting a potential risk to receiving water bodies due to likely toxic effects on organisms. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

example, the processes influencing contaminant concentrations in each medium differ. Sampling sites downstream of urban and industrial areas are subject to continuous inputs of contaminants, particularly during stormwater events, where multiple sources – both point and non-point – contribute to pollution. Sediments act as a long-term sink for hydrophobic and particulate-bound substances. Contaminants such as PAHs, DEHP, and TBT, which have low solubility in water and strong affinities to organic matter, tend to bind to sediments rather than remain in the water phase. This behavior allows sediments to integrate pollution over extended periods, reflecting cumulative contaminant inputs, whereas concentrations in the water phase vary more in the short term due to the episodic nature of WW discharges. The fact that most of the quantified contaminants in sediments were detected at sites downstream of urban areas reinforces the role of stormwater discharges in degrading water quality and suggests that urban runoff is a contributor to the accumulation of contaminants in receiving water bodies.

While this risk estimation provides valuable insights, it excludes compounds lacking established EQS and PNEC values – such as most organotin compounds – preventing an assessment of their potential risks to aquatic ecosystems. Additionally, some substances were not quantified during measurements. However, if a substance's reporting limit exceeded its toxicity-based threshold, if detected, it would immediately pose a risk with a ratio above 1 (Fig. 4 - e.g. DNOP, Pyr, Chr). This does not confirm the absence of risk but rather underscores limitations in quantification. Consequently, these substances show an exceedance rate of 0%, despite their potential risks.

Further, risk estimation primarily focuses on ecological thresholds (i.e. PNEC and AA-EQS). However, the exceedance of these benchmarks, particularly for substances such as PFOS, PAHs, and phthalates, also raises potential concerns for human health as these compounds are known for their persistence, bioaccumulative properties, and toxic

effects, including endocrine disruption and carcinogenicity (Podder et al., 2021; Schwarzenbach et al., 2006; Spahr et al., 2019). Although this study did not directly evaluate human health risks, the presence of these contaminants in urban streams that may connect to recreational areas or downstream water supplies highlights the need for further research into possible exposure pathways.

Moreover, the co-occurrence of multiple contaminant groups in both sediment and water, even at low concentrations, presents complex environmental challenges. The effects of exposure to contaminant mixtures remain poorly understood and, while individual compounds may fall below toxic thresholds, their combined and long-term effects could result in cumulative or synergistic impacts on aquatic ecosystems. These "cocktail effects" highlight the need to move beyond single-compound assessments and adopt a more comprehensive approach to evaluating the environmental impact of stormwater-related contaminants.

4. Conclusions

This study comprised a comprehensive analysis of potentially stormwater-related contaminants in water (under DW and WW) and bottom sediments in urban streams. The findings highlight the role of stormwater runoff in transporting contaminants from urban areas to receiving water bodies. This was particularly evident in the analysis of bottom sediments, where sampling sites downstream of urban and industrial areas were found to contain a greater variety of contaminants as well as higher concentrations of analyzed compounds. Such contamination was concerning, as sediments can act both as a sink and source, potentially leading to long-term exposure risks.

Despite their known presence in stormwater, PAHs and phthalates were rarely detected in the water phase in the receiving waters, likely due to their strong affinity for sediments and dilution effects. However,

their accumulation in bottom sediments downstream of urban areas indicated that stormwater runoff is a transport pathway for these contaminants. In contrast, organotin compounds in the water phase likely originate from sources other than stormwater runoff, as their concentrations in the streams were diluted during WW, suggesting lower levels in stormwater compared to baseflow conditions in the streams. Phenols did not exhibit a clear pattern indicative of runoff transport. Given that contaminants distribute differently between water and sediments depending on their physicochemical properties, investigating both matrices is important when assessing stormwater impacts on receiving water. Hydrophobic compounds tend to accumulate in sediments, while hydrophilic substances are more likely to remain in the water phase. Thus, focusing on only one matrix can lead to an incomplete assessment of stormwater-related contaminants and associated risks in the streams as contaminants may be below the detection limit in one of the phases while present in the other. Therefore, future research should preferably aim to analyze both phases in parallel to provide a more comprehensive understanding of stormwater impacts on receiving waters.

The risk assessment revealed that hydrophobic organic contaminants, particularly those accumulating in sediments (e.g., phenols, phthalates, and PAHs), posed a greater risk than relatively hydrophilic ones (e.g., PFAS) based on exceedance rates. This is especially concerning due to the potential long-term impact of sediment-accumulated contaminants on aquatic ecosystems. On the other hand, although PFAS – particularly PFOS – had lower exceedance rates in sediment compared to contaminants like phenols and phthalates, unlike others, it was uniquely identified as a critical contaminant in both the water and sediment phases. Its widespread presence across all environmental compartments is further intensified by stormwater discharges increasing associated risks in both water and sediment, raising additional concerns for aquatic ecosystems, and highlighting the need for mitigation strategies. Source control is an effective strategy for reducing the impact of stormwater runoff on receiving waters, as it targets contaminants at their origin, minimizing the need for downstream treatment systems. Measures may include regulatory or operational restrictions on anthropogenic sources, such as traffic and industrial activities, as well as substituting sources (e.g. of PFAS) with alternatives. Further, as not all sources can be eliminated, stormwater treatment technologies are also needed for pollution reduction. However, their efficiency varies considerably depending on pollutant characteristics such as hydrophobicity (Beryani et al., 2023; Bodus et al., 2024; David et al., 2015). Source elimination should especially target persistent, hard-to-remove substances such as PFAS. However, as this is complicated due to political processes and societal dependence on certain products, future research should focus not only on identifying urban contaminant sources but also improving treatment efficiency of stormwater treatment facilities for a wider range of compounds.

Variability in the occurrence and concentrations of contaminants across the studied streams highlights the role of stormwater runoff in transporting pollutants. Their associated risks are affected by local factors such as the presence of known point sources and the degree of urbanization. Even though this study was carried out in relatively small cities, the findings clearly demonstrate that such stormwater discharges can also impact receiving waters. The scale of the urbanization in this study may be seen as a limitation in terms of generalizing the results to larger metropolitan areas; however, comparisons with other studies of urban rivers have shown that concentrations in smaller streams, such as those studied, can be similar to those observed in larger rivers (e.g., Rhine, Elbe, or Seine) and larger cities (e.g., Berlin, Paris) for certain compounds (e.g., phthalates). This finding suggests that streams affected by relatively small urban areas are not necessarily less impacted and may be equally vulnerable to contamination from stormwater discharges, depending on the characteristics of the surrounding catchments and streams. As a result, management strategies need to be tailored to the specific conditions of each catchment, taking into account factors such as urbanization intensity and the presence of identified sources (i.e.

industrial zones and airports) as well as the sensitivity of the receiving waters.

CRediT authorship contribution statement

S.E. Kali: Writing – original draft, Visualization, Validation, Software, Project administration, Methodology, Investigation, Conceptualization. **H. Österlund:** Writing – review & editing, Validation, Supervision, Project administration, Methodology, Funding acquisition, Conceptualization. **M. Viklander:** Writing – review & editing, Supervision, Project administration, Methodology, Funding acquisition, Conceptualization. **G. Blecken:** Writing – review & editing, Supervision, Project administration, Methodology, Funding acquisition, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

We gratefully thank Peter Rosander from Luleå University of Technology and Karin Gundberg from Tyréns for help with fieldwork. We would also like to acknowledge the staff at Tyréns and WRS for helping with the sampling, Uppsala, Norrköping/Nodra and Söderköping Municipalities as well as Åängens B & B, Ensjöholms By, and Tom Lindström for authorizing access to sampling sites and providing necessary information. The work was carried out as part of the research cluster Stormwater & Sewers. Financial support for this work was provided by the Swedish Environmental Protection Agency (Naturvårdsverket), Grant numbers 06920-21 and 03808-23, the Swedish Research Council Formas, Grant number 2023-02529, and the DRIZZLE Centre for Stormwater Management, funded by the Swedish Governmental Agency for Innovation Systems (Vinnova), Grant number 2022-03092.

Declaration of generative AI and AI-assisted technologies in the writing process

During the preparation of this work, the authors used Grammarly and Microsoft Copilot to check grammar and spelling. After using this tool/service, the authors reviewed and edited the content as needed and take full responsibility for the content of the publication.

Supplementary materials

Supplementary material associated with this article can be found, in the online version, at [doi:10.1016/j.watres.2025.123847](https://doi.org/10.1016/j.watres.2025.123847).

SI-I: Sampling site details, sampling occasions details, details of chemical analysis, details of diagnostic ratio calculations for PAHs, and a comparison of concentrations with quality standards (PDF)

SI-II: Physicochemical properties of organic contaminants (XLSX)

SI-III: Detection frequencies, min, max, and median concentrations of analyzed compounds across studied streams and sampling sites under dry and wet weather conditions as well in bottom sediment (XLSX)

Data availability

The dataset supporting this study is referenced at: <https://doi.org/10.5878/hpqb-k059>.

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