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Assessment of poly- and perfluoroalkyl substances (PFAS) in commercially available drinking straws using targeted and suspect screening approaches

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ABSTRACT

Many food contact materials (FCMs) and reusable plastics in the food industry contain poly- and perfluoroalkyl substances (PFAS), a group of synthetic pollutants that are known to be potentially harmful for wildlife, humans, and the environment. PFAS may migrate from FCMs to food consumed by humans. As a replacement for plastics, often paper and other plant-based materials are used in commercial settings. This also applies to drinking straws, where plant-based and other presumably eco-friendly straws are increasingly used to reduce plastic pollution. In order to make these materials water-repellent, PFAS are added during manufacturing but can also already be present early in the supply chain due to the use of contaminated raw materials. In the present study, we examined the PFAS concentrations in 39 different brands of straws, made from five materials (i.e. paper, bamboo, glass, stainless steel, and plastic) commercially available on the Belgian market. We combined both targeted and suspect-screening approaches to evaluate a wide range of PFAS. PFAS were found to be present in almost all types of straws, except for those made of stainless steel. PFAS were more frequently detected in plant-based materials, such as paper and bamboo. We did not observe many differences between the types of materials, or the continents of origin. The presence of PFAS in plant-based straws shows that they are not necessarily biodegradable and that the use of such straws potentially contributes to human and environmental exposure of PFAS.

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Introduction

Per- and polyfluoroalkyl substances (PFAS) are a large family of more than 12000 manmade resistant perfluorinated chemicals (US EPA 2023). Their unique water-repellent and fire-resistant properties, as well as their thermal and chemical stability, ensure that these organic compounds are produced and used on a large industrial scale for various applications. However, most PFAS barely break down and are both accumulative and potentially toxic to humans, animals, and the environment (EEA 2022).

Intake through food and drinking water are the main routes of general human exposure to PFAS. In addition, many food packaging

materials (Food Contact Materials, FCMs) and reusable plastic bags used in the food industry can contain PFAS (Sznajder-Katarzyńska et al. 2019; EEA 2022). Begley et al. (2005) has shown that during storage of food in FCMs, migration of PFAS to the food occurs, thus increasing human dietary exposure. The extent of PFAS migration from FCMs depends on the amount, type, and chain length of PFAS used, the type of food, the duration of contact and the temperature (Lerch et al. 2023). Prolonged food contact at high temperatures usually increases migration, especially with fatty foods (Trier et al. 2011; Schaidler et al. 2017; Lerch et al. 2023). Furthermore, shorter PFAS migrate more easily

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from FCMs to food since their lower weight and higher volatility increases their mobility (Lerch et al. 2023).

A recent study in the U.S. by Timshina et al. (2021) revealed that PFAS can also be found in straws made from plant-based materials. By drinking through such straws, humans may ingest a so far undetermined amount of PFAS. As a study on PFAS in plant-based straws has never been explored in Europe before, we investigated how PFAS concentrations compare in commercially available straws available on the Belgian market and extended the research of Timshina et al. (2021) to straws made from four materials (i.e. bamboo, glass, stainless steel and plastic) other than paper. We subjected 39 different brands of straws to an extensive targeted analysis for 29 individual PFAS compounds. From the obtained data, we compared the \sum PFAS concentrations based on the continents where the straws were produced, and among types of straws. In addition to the targeted analytical approach used by Timshina et al. (2021), we performed suspect screening analysis on a subset of straws to investigate the potential presence of other non-targeted or new generation PFAS. High-resolution mass spectrometry (HRMS), such as time-of-flight (TOF) MS, combined with a suspect screening data mining approach enables an evaluation of a wide range of substances without available reference standards. Previous studies have reported substantial amounts of unreported PFAS in environmental matrices (Koch et al. 2021; Gan et al. 2022) and human serum (Miaz et al. 2020). Therefore, a comprehensive investigation of diverse PFAS classes is necessary in order to identify potential sources of these unreported PFAS. Until now, suspect screening has been applied to PFAS research for various matrices such as river water (Joerss et al. 2022; Ng et al. 2022), wastewater (Wang et al. 2020; Jeong et al. 2022; Koronaiou et al. 2022; Ng et al. 2022), tap water (Koronaiou et al. 2022), and serum (Miaz et al. 2020; Bao et al. 2022) resulting in successful identification of a number of emerging PFAS groups. This indicates the power of suspect screening as a complementary tool to investigate additional or new PFAS that would have been missed by a targeted analytical approach.

Materials and methods

Collection of straws

For this study, we collected the widest possible range of straws available on the Belgian market, manufactured from different materials, purchased from several stores and with an as diverse source as possible (Table 1). The names of (online) stores and restaurants have been removed from this table and an umbrella category name such as 'supermarket' was used. Based on the available market supply, it was decided to subject 20 paper straws, five glass straws, five bamboo straws, five stainless steel straws, and four plastic straws to extensive analysis for 29 different PFAS.

PFAS extraction and analysis

Samples were processed by homogenizing each type of straw in order to fit into a 50 mL polypropylene (PP) tube. Paper and plastic straws were cut, using stainless steel scissors, into pieces of approximately 0.5 by 1 cm and bamboo straws were cut into larger pieces due to the sturdiness of the material. The stainless-steel straws were folded to break them into 2–3 pieces and the glass straws were crushed with a hammer into 5 mm shards. Three replicates of each sample were examined.

Table 1. Inventory of straws purchased and examined, grouped by type of straw: paper (PAP), glass (G), bamboo (B), stainless steel (S), and plastic (PLA).

Type	Store	CO	Type	Store	CO
PAP1	Supermarket	Europe	G1	Variety store	Asia
PAP2	Variety store	Asia	G2	Variety store	Unknown
PAP3	Variety store	Asia	G3	Variety store	Unknown
PAP4	Drugstore	Asia	G4	Variety store	Asia
PAP5	Supermarket	Europe	G5	Supermarket	Europe
PAP6	Variety store	Asia	B1	Variety store	Asia
PAP7	Variety store	Unknown	B2	Variety store	Asia
PAP8	Fast-food chain	Unknown	B3	Supermarket	Europe
PAP9	Variety store	Asia	B4	E-commerce	Asia
PAP10	Fast-food chain	Unknown	B5	Furniture store	Asia
PAP11	Variety store	Europe	S1	Variety store	Asia
PAP12	Variety store	Asia	S2	Variety store	Asia
PAP13	Toy store	Asia	S3	Variety store	Asia
PAP14	Toy store	Asia	S4	Supermarket	Asia
PAP15	Toy store	Asia	S5	Variety store	Asia
PAP16	Supermarket	Asia	PLA1	Toy store	Unknown
PAP17	E-commerce	Europe	PLA2	Unknown	Unknown
PAP18	Supermarket	Europe	PLA3	Unknown	Europe
PAP19	Supermarket	Unknown	PLA4	Toy store	Unknown
PAP20	Supermarket	Unknown			

For each straw the type of store it was purchased from and its continent of origin (CO) are shown. Unknown is used when the origin of the straw was not provided.

The extraction procedure followed a protocol based on Powley et al. (2005), with some adjustments. Each sample was spiked with 10 ng of a mass-labelled internal standard (ISTD; MPFAC-MXA, Wellington Laboratories, Guelph, Canada), containing seven mass-labelled PFCAs and two mass-labelled PFSAs (Supplementary Table A1). Subsequently, 10 mL of methanol (MeOH; VWR International BVBA, Belgium) was added to the samples and they were vortex-mixed. Hereafter, the samples were sonicated (3×10 min, with vortex-mixing in between periods, Branson 2510) and left overnight on a shaking plate (135 rpm, room temperature, GFL 3020). The following day, the samples were centrifuged ($1037 \times g$, 4°C , 10 min, Eppendorf centrifuge 5804 R, rotor A-4-44) and the supernatant was transferred to a 15 mL PP tube. The supernatants were then dried, using a rotational vacuum concentrator (RVC 2-25), to approximately 0.5 mL. The dried samples were then rinsed twice with 250 μL methanol (MeOH, VWR International BVBA, Belgium) and transferred to Eppendorf tubes containing 0.1 mL of activated carbon powder (graphitized carbon powder; Supelclean ENVI-Carb, Sigma-Aldrich, Belgium) and 50 μL acetic acid (Acetic acid glacial, Acros Organics BVBA, Belgium), to reduce impurities. After vortex mixing the samples for 1 min and centrifuging them for 10 min ($9279.4 \times g$, 4°C , 10 min, Eppendorf centrifuge 5804 R, rotor A-4-44), the supernatant was transferred into a series of new Eppendorf tubes and dried again using the previously mentioned rotational-vacuum-concentrator. The dried samples were then reconstituted with 300 μL of a 2% ammonium hydroxide solution (Fisher Scientific B.V., Belgium) diluted in acetonitrile (ACN, Acros Organics BVBA, Belgium). Samples were finally filtered through an Acrodisc syringe filter (Waters, N.V., Belgium), with 0.2 μm Supor polyethersulfone membrane, into injectorvials (Waters N.V., Belgium) to be further analyzed.

The samples were analysed on 29 individual PFAS compounds using Ultra Performance Liquid Chromatography Tandem Mass Spectrometry (UPLC-MS/MS, ACQUITY, TQD, Waters, Milford, MA, VS) operating in negative electrospray ionization. An ACQUITY BEH C18 column (2.1×50 mm; $1.7 \mu\text{m}$, Waters, USA) was used to

separate the analytes. An ACQUITY UPLC BEH C18 VanGuard Precolumn (2.1×5 mm; $1.7 \mu\text{m}$, Waters, USA) was placed on this analytical column to protect it from particulates and chemical contamination. An ACQUITY BEH C18 delay column (2.1×30 mm; $1.7 \mu\text{m}$, Waters, USA), was inserted between the injector and solvent mixer to retain any PFAS contamination originating from the system. The mobile phase solvent gradient started at 65% of 0.1% formic acid in water, linearly changed to 100% of 0.1% formic acid in ACN in 3.4 min, and returned to 65% of 0.1% formic acid in water at 4.7 min. We used an injection volume of 6 μL (partial loop) with a flow rate of 450 $\mu\text{L}/\text{min}$. Multiple reaction monitoring (MRM) of two diagnostic transitions per analyte were used to quantify the target PFAS. Further details on instrumental settings (cone voltages, collision energy) and diagnostic transitions are provided in Supplementary Table A1 and were validated by Groffen et al. (2021).

Furthermore, a 1290 ultra-high performance liquid chromatography (UHPLC; Agilent Technologies, Santa Clara, CA, USA) coupled with 6530 Q-TOF (Agilent Technologies) was employed for suspect screening of PFAS in selected straw samples. These straws were selected based on the PFAS concentrations and the diversity in PFAS profiles detected after the targeted analysis and included one glass straw (G2), one plastic straw (PLA2), three bamboo straws (B2, B3 and B4), and eight paper straws (PAP3, PAP5, PAP6, PAP7, PAP8, PAP10, PAP12 and PAP15). The extraction of samples for suspect screening was done in the same way as for the target analysis. Chromatographic separation was performed using Zorbax Eclipse Plus RRHD C18 column (2.1×100 mm, $1.8 \mu\text{m}$; Agilent Technologies) connected with a guard column (Eclipse Plus C18, 2.1×5 mm, $1.8 \mu\text{m}$; Agilent Technologies). The mobile phases consisted of (A) 2 mM ammonium acetate in water and (B) MeOH with a flow rate of 0.25 mL/min. The LC gradient program started with 10% B, changed to 90% B at 25 min, increased to 100% B at 30 min, and then back to the initial condition from 33 min (total run time: 40 min). The column temperature was set to 40°C . The Q-TOF-MS was operated with electrospray ionization

(ESI) negative polarity and the data-dependent acquisition (DDA; Auto-MSMS, 5 abundant precursors selection/cycle) mode was applied at a 2 GHz extended dynamic range mode (1700 m/z) with collision energy settings of 10 and 30 eV. The source parameters were set as following: gas temperature 200 °C, gas flow 8 mL/min, nebulizer pressure 40 psi, sheath gas temperature 250 °C, sheath gas flow 12 mL/min, and nozzle voltage 500 V. Mass range for MS and MS/MS ranged from 100 to 1400 m/z and 50 to 1300 m/z, respectively, with an acquisition rate of 4 spectra/s. Isolation width was set to narrow (1.3 m/z). A real-time calibration was performed during the analytical run by monitoring reference ions which were 119.0363 (purine) and 980.0163 (hexakis [1H, 1H, 3H-tetrafluoropropoxy] phosphazine).

For suspect screening, two suspect lists were applied which were PFAS Master List provided by US EPA Comptox Dashboard and Liu et al. (2019). Initially, the features in straw samples were aligned and extracted by batch recursive feature extraction algorithm using Profinder 10.0 (Agilent Technologies). The processed data were exported and introduced to Mass Profiler Professional 15.0 (Agilent Technologies) and categorized into blank ($n=2$) and samples ($n=13$) to perform statistical analysis such as fold-change analysis. For the straw samples, a fold-change of 3 was used to keep the features with abundances at least 3 times higher than those in the blanks.

The filtered features were matched with suspect lists with ID Browser 10.0 (Agilent Technologies) with a match tolerance of ± 10 ppm + 2 mDa and an adduct of $[M-H]^-$ was considered. The matched features were exported and compounds with matching score over 50 were manually investigated with Agilent MassHunter Qualitative Analysis 7.0. using the find-by-formula option. Samples were selectively re-injected to achieve clearer MS/MS spectra of annotated compounds. The confidence level (CL) of a compound was given based on (Schymanski et al. 2014), which were confirmed MS, MS/MS, and RT by standard (CL1); MS/MS library matching (CL2); tentative structure confirmation with a substructure or class (CL3); unequivocal molecular formula matching (CL4). As a quality control, a procedural blank (10 mL

MeOH) was introduced in the extractions and measurements for each batch of 15–20 samples. In the blanks, no PFAS contamination could be derived from the UPLC-MS/MS analysis, given that all concentrations were below the Limits of Quantification (LOQs). In addition, an instrumental solvent blank (100% ACN) was regularly injected during the UPLC-MS/MS analysis to prevent carry-over between samples and procedural blanks. The LOQs can be found in [Supplementary Table A2](#) and were determined in the matrix as the PFAS concentrations in the straws corresponding to a cut-off signal-to-noise ratio of 10.

Statistical analysis

R Studio (Version 4.0.3) was used to statistically compare PFAS concentrations among material and continent of origin (CO). Statistical analysis was always run on the sum of all PFAS components measured in a single straw, with three replicates for each straw. PFAS concentrations < LOQ were assigned a replacement concentration following a maximum likelihood estimation method (Villanueva 2005; De Solla et al. 2012). To analyze potential differences in PFAS concentrations between materials and origin, an analysis of variance (one-way ANOVA) was used twice. Only paper straws were used when determining whether a significant difference could exist between concentration and CO since the dataset of straws made from alternative materials as well as the variance between CO would be too small. We chose to work with larger regions such as Europe and Asia instead of separate countries to avoid there being countries of origin with $N=1$. The normality of the dataset was verified using a Shapiro-Wilk's test and diagnostic plots of the models. For non-normally distributed data, a non-parametric Kruskal–Wallis test was used. The level of significance was set at $p \leq 0.05$. For a p -value < 0.05, a Pairwise Wilcoxon test was used to evaluate the significant differences. Data were visualized using Graphpad Prism (version 8).

Results and discussion

Our hypothesis was that straws made of paper were more likely to be contaminated with PFAS than other types of straws, as manufacturers aim

to make the paper water-repellent. Following this hypothesis, we expected that there would be virtually no PFAS present in straws made of stainless steel or glass. For plastic straws, it was difficult to predict the presence of PFAS due to the variety of plastics and additives used. For both straws of plant origin (paper and bamboo), we hypothesized that PFAS contamination could already have occurred during plant growth on contaminated soils. Moreover, we would assume that there would be differences in PFAS concentrations based on the country and/or continent of origin due to different regulations regarding the use of PFAS in FCMs.

Of the 29 target PFAS analyzed, 16 components were observed above the LOQ, namely PFHxA, PFOA, PFNA, PFUnDA, PFDoDA, PFTTrDA, PFTeDA, PFBS, PFPeS, PFHpS, PFOS, PFDS, 4:2 FTS, 6:2 FTS, 8:2 FTS and NaDONA. PFNA was only detected in one of the three replicates of PAP6. Since the average concentrations of the triplicates was < LOQ, we omitted PFNA from the results and further analyses. The components with concentrations below the LOQ, i.e. PFBA, PFPeA, PFHpA, PFDA, PFHxS, PFEESA, FBSA, 11Cl-PF3OUdS, 9Cl-PF3ONS, HFPO-DA, PF5OHxA, PF4OPeA, and 3,6-OPFHpA, were omitted from the results and statistical analysis.

Target analysis

PFAS were detected in almost all paper-based straws, with highly variable concentrations between brands, ranging from < LOQ to 7.15 ng/g (Figure 1). PFOA was the most frequently detected component. Specific concentrations can be found in [Supplementary Table A3](#). In the other types of straws, more often all PFAS were below the LOQ. In bamboo straws, PFAS were detected in the range < LOQ to 3.47 ng/g in four out of five brands. In glass straws two brands showed concentrations above the LOQ, ranging from < LOQ to 6.65 ng/g, while the concentrations for the other brands were found to be below the LOQ. In the stainless steel straws, no PFAS concentrations above the LOQ were observed in any of the brands. Finally, three out of four plastic straw brands contained quantifiable PFAS concentrations, ranging from < LOQ

to 0.924 ng/g. There was a significant variation in PFAS profiles between straws from the same materials. This variation, in combination with the relatively small dataset, made it impossible to compare statistically the materials and continent of origin (CO) for each PFAS individually. Therefore, we used the Σ PFAS concentrations instead.

Focusing on the PFAS profiles of each straw (Figure 2), it can be seen that 8:2 FTS was exclusively detected in the paper straws. 8:2 FTS is a precursor of 8:2 FTOH which eventually degrades to PFOA and other carboxylic acids (Dasu et al. 2012). Apart from 8:2 FTS, PFHxA, PFBS, PFPeS, PFOS, PFDS, and NaDONA were only detected in paper straws, albeit with low detection frequencies (i.e. often in only one brand of paper straws). Furthermore, PFHpS and 4:2 FTS were only detected in bamboo straws, but again with detection in only one brand. In general, PFCAs contributed more than 40% to the PFAS profile in each of the materials, with PFOA being dominant in paper and plastic straws and PFTeDA being dominant in glass and bamboo straws. Finally, we did not observe any pattern related to the continent of origin. This is in agreement with the study by Timshina et al. (2021) who reported no associations between the country of manufacture and the type of PFAS present in commercially available straws from the US.

The median Σ PFAS concentration was found to be < 2 ng/g (or 0.402 ng/straw) for straws from all tested materials (Figure 3) but Σ PFAS concentrations did not differ significantly among the different materials ($p = 0.245$). Despite the variation among brands and the absence of significant differences in Σ PFAS concentrations among types of straws, it is noteworthy that almost all plant-based straws contained PFAS. This was expected as PFAS are known to be used to confer stain and water repellency to FCMs (Borg and Ivarsson 2017; Trier et al. 2017). Stainless steel is often made of chromium(III)oxide or other metal oxides and thus has no net charge on the surface of the straws. Therefore, adsorption of pollutants is expected to be limited or even absent in stainless steel. Some glass straws are made of borosilicate glass so the presence of PFAS in these straws might be due to adsorption to silica minerals (Du et al. 2014). However, since we have no

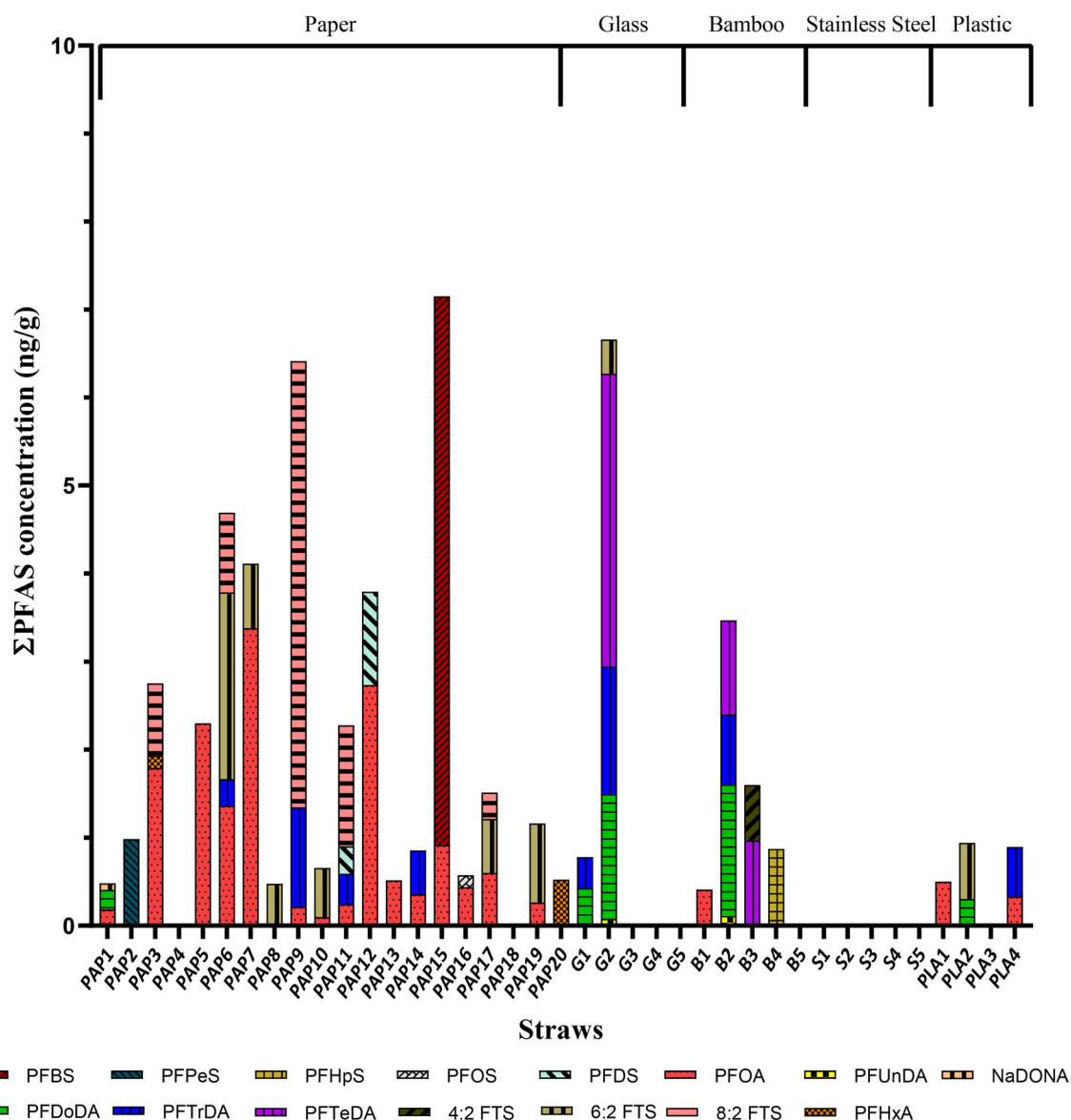


Figure 1. Mean ($N=3$ per straw) Σ PFAS concentrations in ng/g, divided into the different PFAS components measured. Details on PFAS concentrations are reported in [Supplementary Table A3](#).

information regarding the type of glass for each of the brands, further examination of how glass can be contaminated with PFAS is required.

Although median Σ PFAS concentrations for Asian straws seemed higher than those from Europe and those with unknown origin, this difference was not significant ($p=0.781$; [Figure 4](#)). The absence of differences among continent of origin is in agreement with the study by [Timshina et al. \(2021\)](#) in straws commercially available in the US.

Commercially available plant-based straws on the Belgian market contained higher PFAS concentrations (median Σ PFAS of 1.01 ng/straw; the

present study) than those on the US market (median Σ PFAS of 0.554 ng/straw; [Timshina et al. 2021](#)), which might be attributed to differences in origin and background contamination due to non-intended addition of PFAS (see below). To the best of our knowledge, no other studies have identified PFAS in straws.

Suspect screening

In total, 454 features were annotated with a matching score above 50 and those were manually investigated for peak shapes, MS isotopic patterns, and MS/MS fragmentation patterns. The

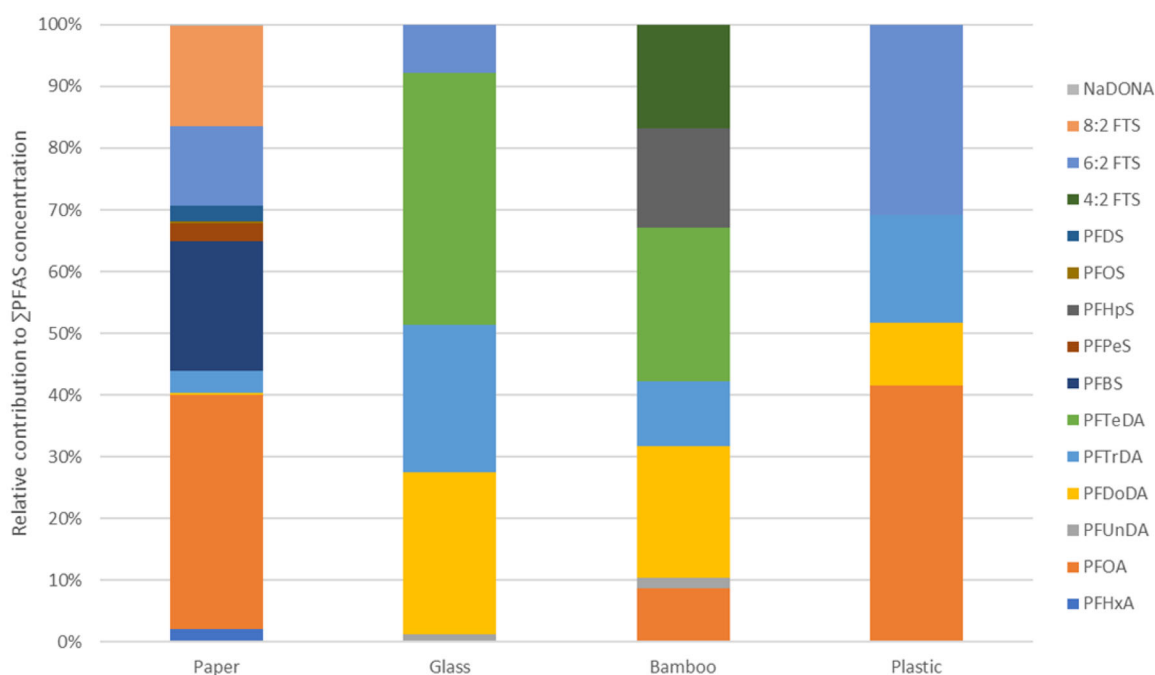


Figure 2. Relative contribution of PFAS to the \sum PFAS concentration in each of the types of straws. Values $<$ LOQ were considered 0 in the calculation of the sum concentrations. paper ($N=20$), bamboo ($N=5$) glass ($N=5$), plastic ($N=4$). Stainless steel was omitted from the Figure due to no detection of any PFAS in straws of this material.

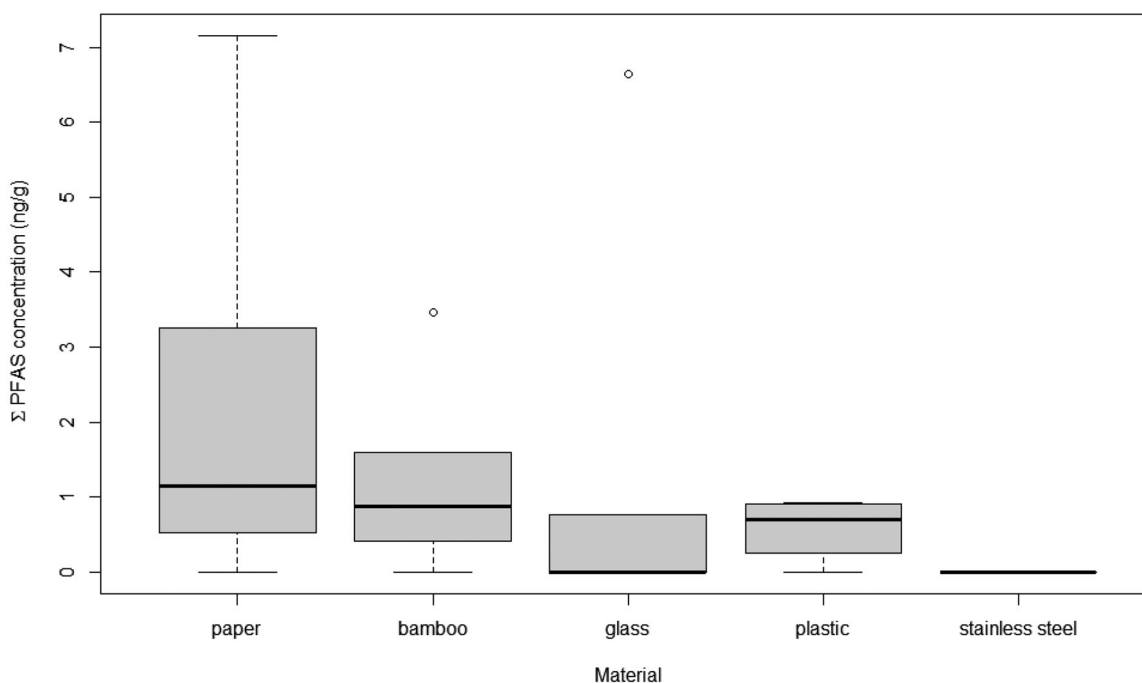


Figure 3. Mean \sum PFAS concentrations (ng/g) in the different materials studied: paper ($N=20$), bamboo ($N=5$) glass ($N=5$), plastic ($N=4$) and stainless steel ($N=5$). Values $<$ LOQ were considered 0 in the calculation of the sum concentrations in the replicates.

confidence level (CL) of each compound was given based on Schymanski et al. (2014), which were confirmed MS, MS/MS and RT by standard (CL1); MS/MS library matching (CL2); tentative structure confirmation with a substructure or

class (CL3); and unequivocal molecular formula matching (CL4). As a result, two compounds were newly identified as CL1 which were not included in the target analysis (Supplementary Table A4). The identified compounds in straw

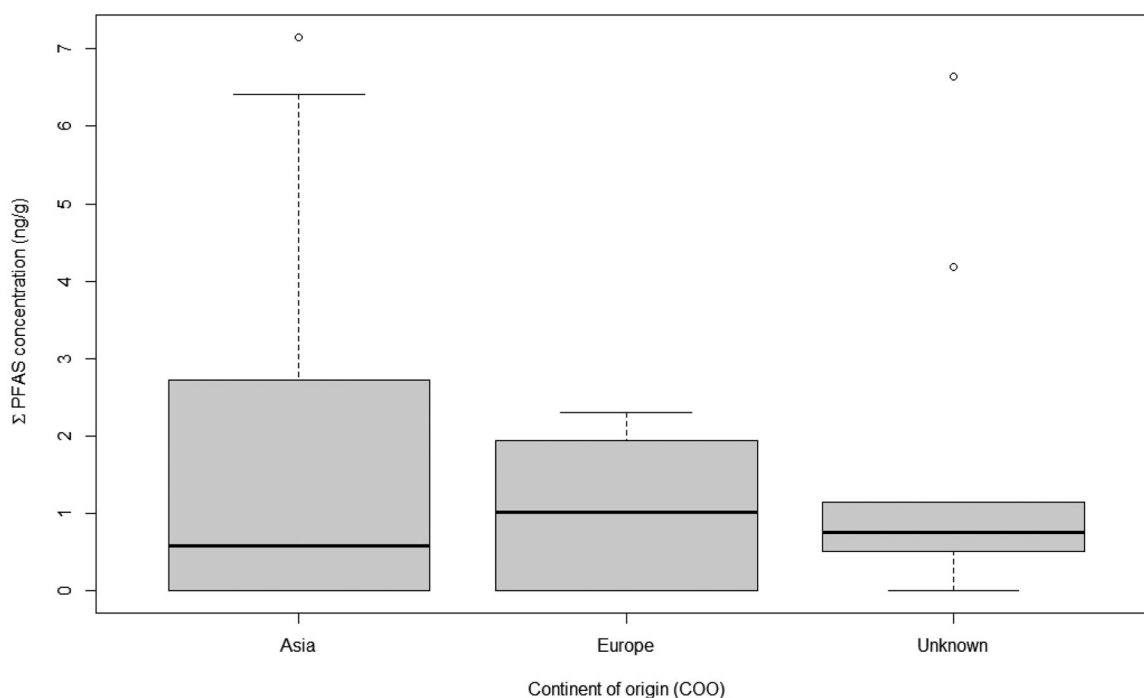


Figure 4. Mean Σ PFAS concentrations (ng/g) for paper straws produced in different continents of origin (COO). Asia ($N=30$), Europe ($N=15$) and Unknown ($N=15$).

samples were trifluoroacetic acid (TFA; average m/z : 112.9864; average mass error: 7.1 ppm) and trifluoromethanesulfonic acid (TFMS; 148.953; 2.7 ppm) which are classified as ultra-short chain PFAS (PFAS with carbon chain ≤ 2 for PFCAs and ≤ 3 for PFSAs). The MS/MS spectrum used for compound confirmation are presented in [Supplementary Figure A1](#).

In straw samples, TFA was detected in five out of eight paper straws and in one bamboo straw. The highest abundance of TFA was observed in sample PAP 10, but the continent of origin is unknown. TFMS was detected in six paper straws and one bamboo straw, and the highest abundance was found in one paper straw (PAP 7), but the sampling information is absent.

Since the industrial use of TFA is limited, it is likely that the TFA present in the straws is coming from the breakdown of halogenated hydrocarbons (Franklin 1993; Boutonnet et al. 1999; Solomon et al. 2016; Sun et al. 2020), but leaching from landfills has also been reported as a source of TFA (Björnsdotter et al. 2019). The potential sources of other ultra-short chain PFAS, including TFMS, are still unclear but have been linked to sites where firefighting foams have been used. Furthermore, ultra-short chained PFAS

have also been reported in wastewater, surface water, and groundwater that were connected to landfills, military training sites, and waste management facilities (Zhang et al. 2013; Barzen-Hanson and Field 2015; Björnsdotter et al. 2019). Nonetheless, both chemicals are highly water soluble, meaning that there is a chance of them migrating from the straw into the drink.

Ultra-short chain PFAS meet the criteria of persistent, mobile, and toxic (PMT) or very persistent and very mobile (vPMT) substances proposed by the German Environment Agency (UBA) (Neuwald et al. 2022). Therefore, there should be continued and ever-increasing attention on the occurrence, environmental distribution, and toxicity of ultra-short chain PFAS. Our study confirmed that straws might be an exposure route of TFA and TFMS which might result in increased exposure to PFAS for humans and subsequent potential toxic effects (Carlson et al. 2022).

Our study provides first-time evidence that suspect screening can be used to identify various PFAS without available reference standards in food contact materials such as drinking straws. As mentioned earlier, suspect screening has been used before to investigate the presence of PFAS in various matrices. Although suspect screening

is less useful for matrices with low concentrations of markers of interest (Pourchet et al. 2020), our study adds to the scientific proof that this technique is a suitable tool to determine additional or new PFAS that have not been included in targeted analytical approaches.

Non-intentional addition of PFAS

To date, it remains unclear which portion of the total PFAS concentration in plant-based FCMs has intentionally been added to the product (for the purpose of e.g. water repellency) and which portion can be considered as background concentration due to recycling procedures. The occurrence of non-intentionally added PFAS could be due to the usage of recycled contaminated paper fibers in the production of new FCMs or to contamination of source materials or the processing water (D'Eon et al. 2009; Hoffman et al. 2011). This contamination can also be traced back to biosolids, which are being used as a fertilizer in agriculture (Fernandes et al. 2019). Plants grown on contaminated soils can take up PFAS and eventually this pollution can end up in FCMs when these plants are used in the production process (Costello and Lee 2020). The complexity and variation in potential background contamination makes it difficult to pinpoint an exact source for PFAS in drinking straws. Plant-based straws marketed as eco-friendly alternatives are as such not necessarily more sustainable for the planet than plastics, because they may contribute to the current prominent PFAS pollution issues (Langberg et al. 2020; Helmer et al. 2022). Plant-based FCMs, particularly those made of paper, are supposed to be recyclable products, but will most likely end up in landfills or be incinerated, releasing PFAS further into the environment (Monge Brenes et al. 2019; Masoner et al. 2020; Stoiber et al. 2020).

Conclusions

PFAS were found to be present in almost all types of straws, but primarily in those made from plant-based materials. These 'eco-friendly' plant-based straws are not necessarily a more sustainable alternative to plastic straws, because they can

be considered as an additional source of PFAS exposure in humans and the environment (e.g. after degradation in landfills or through incomplete incineration). The most sustainable alternative seems to be stainless-steel straws, which can be reused, do not contain PFAS and can be fully recycled. More research is needed on PFAS in FCM, the factors affecting the migration of PFAS into food and drinks, and the potential human risks posed by the use of these FCMs. Our results contribute to a further understanding of the potential human exposure pathway through FCMs, and the results are important in the context of public health.

Author contributions

Pauline Boisacq: Investigation, Formal analysis, Writing, Visualization; Maarten De Keuster: Investigation, Formal analysis, Writing, Visualization; Els Prinsen: Conceptualization, Writing, Funding Acquisition; Yunsun Jeong: Formal analysis, Writing; Lieven Bervoets: Writing, Funding Acquisition; Marcel Eens: Writing, Funding Acquisition; Adrian Covaci: Writing; Tim Willems: Conceptualization, Methodology, Writing; Thimo Groffen: Conceptualization, Methodology, Validation, Writing, Funding acquisition, Supervision.

Disclosure statement

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Data availability statement

The datasets generated and/or analyzed during the current study are not publicly available. The test data is restricted to the relevant personnel of the project and is not allowed

to be disclosed to the public but are available from the corresponding author on reasonable request.

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