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Recycling of paper, cardboard and its PFAS in Norway

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ABSTRACT

Per- and polyfluoroalkyl substances (PFAS) have been used in paper products since the 1960s. PFAS emissions during the life cycle of these products have been substantial sources of these substances to the environment. Here, a total of 37 PFAS were analysed in 13 paper products sampled in 2021, for which some were made of virgin paper and others recycled paper, including food contact materials (FCM), packaging, and a notebook. In addition, different fractions of the recycling process of corrugated paper and board were sampled at a Norwegian cardboard recycling plant and analysed for the same PFAS, to get an overview of PFAS in the recycling stream of these materials in Norway. PFAS were found in recycled paper and board, indicating PFAS can be a non-intentionally added substance (NIAS) in recycled paper products. Sum of targeted PFAS detected in paper products (including products made of virgin and recycled materials) ranged between 0.4 and 971 μg kg^{-1}, dominated by SAmPAP diester and 6:2 FTS (0–62% and 0–98%, respectively). The sum of targeted PFAS in returned corrugated paper and board in Norway is at least 32 kg per year (6 mg per capita per year). Recycling is important to ensure sustainability. As part of the broad PFAS restriction in Europe, PFAS are to be restricted in paper and board materials. Thus, lower levels of PFAS entering the recycling stream are expected in the future. Monitoring is necessary to assess reductions of PFAS in the paper waste stream due to PFAS regulations in Europe.

Introduction

Concerns regarding per- and polyfluoroalkyl substances (PFAS), including their persistence and mobility (Cousins et al., 2019; Hale et al., 2020), potential for bioaccumulation (Ng and Hungerbühler, 2014), and adverse health effects (Schrenk et al., 2020), have captured the attention of the scientific community since the early 2000s. Among the numerous applications of PFAS (Glüge et al., 2020), their use in paper products is one of the most well-investigated (Trier et al., 2017, 2011). The scientific community has particularly focused on the use of PFAS in food contact materials (FCM), particularly in terms of migration from FCM into food, and subsequently into human serum (Begley et al., 2008; Glenn et al., 2021; Susmann et al., 2019; Trier et al., 2017).

Production and disposal of paper products containing PFAS have been demonstrated to pollute the local environment. In 2018, based on documents from the United States Food and Drug Administration (US FDA), it was estimated that PFAS emissions through wastewater from so-called "typical" paper mills were in the range of 40 to 100 kg PFAS per day (Neltner, 2018). In lake Tyrifjorden, Norway, a paper FCM manufacturing facility was found to have polluted the entire lake (Langberg et al., 2021, 2020), and investigations indicated that tens of tons of PFAS have been emitted at the site between the 1960s and 2013 (Langberg et al., 2021). In the U.S. state of Michigan, paper production and disposal of papermaking waste is suspected to be the main reason for PFAS pollution in the City of Parchment's municipal water system (Michigan PFAS Action Response Team, 2020). Additionally, paper sludge from FCM may end up on agricultural land. Compost containing FCM has been reported to have elevated concentrations of PFAS compared to compost without FCM (Choi et al., 2019). In Rastatt, Baden-Württemberg, Germany, the application of compost mixed with paper sludge in agriculture has been identified to be the source of PFAS pollution in soil and drinking water (Bugsel and Zwiener, 2020).

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Recycling paper is considered more sustainable than disposal or incineration (Villanueva and Wenzel, 2007). However, elevated fluorine concentrations and concentrations of PFAS have been reported for recycled paper (Curtzwiler et al., 2021; Glenn et al., 2021; Robel et al., 2017). Differentiating between intentionally added PFAS and PFAS as a non-intentionally added substance (NIAS) in recycled products is a complex task due to the residues contained in the material (Curtzwiler et al., 2021) as well as unknown uses of PFAS in the recycling machinery. The concentrations and types of PFAS incorporated to products vary depending on manufacturer and usage period (Glenn et al., 2021; Trier et al., 2011), resulting in variations between countries and over time. As part of the broad PFAS restriction proposed in Europe, the conclusion is drawn that "Given sufficiently strong evidence pointing to the existence of technically and economically feasible alternatives at [the Entry into Force of the restriction] no derogation is proposed" (ECHA, 2023). This implies a potential restriction of PFAS in European paper and board materials 18 months after the proposal is adopted, reducing PFAS entering the paper recycling stream. However, PFAS already present in paper products may still be transported within the recycling stream as NIAS and thus, potentially exposing humans through recycled paper products or be emitted into the environment during the recycling process.

The aim of the present study was to investigate PFAS concentrations in paper returned to a paper recycling facility in Norway, along with assessing environmental emissions associated with this industry. The facility mostly processes returned corrugated paper (i.e. cardboard). A subsequent aim was to perform mass estimates of PFAS in this fraction of the recycling stream, extrapolated for the whole country. Targeted analyses were conducted on 37 PFAS. The results are discussed in the context of potential solutions to minimise the presence of PFAS in recycled wastepaper.

Methods

Sampling

Paper products on the Norwegian market

To screen for the concentrations of PFAS in paper products available on the Norwegian market, 13 paper products, sampled in 2021, were chosen for analyses (Table S1). The paper products were selected based on searching Norwegian shops in early 2021 for products with available information about whether they were manufactured with virgin or recycled paper. Additionally, as PFAS have been reported to be used in FCMs, different FCMs were included in the study. Hence, the sampled products represented both recycled paper items (6 different products, 4 of them were FCMs) and products made of virgin materials (6 FCMs and 1 facemask container). Furthermore, two samples of recycled paper from the recycling plant (see section "Fractions on the paper recycling plant") were included.

Fractions on the paper recycling plant

To investigate the present-day paper recycling stream in Norway, different fractions from the recycling process were collected at a Norwegian paper recycling plant (Fig. 1 and Table S2). The facility recycles corrugated paper and board into new paper based products. Samples were collected in April 2021 by subsampling large piles of paper, reject and pulp by hand (using nitrile gloves) into polyurethane bags (VWR Norway). Wastewater was sampled in HDPE bottles by combining grab samples collected every day for one week. The process of paper recycling at this facility involved the following steps. (Fig. 1): Removal of plastic and other debris through visual inspection. Thereafter, the paper was pulped and submitted to a screening step to remove plastic, metals, glue granulates and other impurities using sieving and trommel screening. The sampled fractions included: (1) reject from the initial sorting; (2) pulp of the paper sorted for recycling; (3 and 4) wastewater filtered to separate the dissolved, and particulate phases; (5 and 6) reject from the paper machines PM5 and PM6); and (7 and 8) recycled paper from PM5 and PM6. The two paper machines produced paper of different quality, where PM5 made board and PM6 mad thin paper (Table S2). Pulp and reject fractions were sampled by combining several grab samples (>5). Wastewater was filtered in the laboratory using glass microfiber filters GF/F diameter 47 mm (Whatman™, Middlesex, U.K.) with 0.45 µm of pore size and the obtained particulate matter was analysed separately. Reject PM5 and PM6 were frozen at −20ºC and freeze-dried (−21ºC, 6 mbar). Samples were then homogenised and kept at 4ºC until analysis.

Chemical analysis

Samples were extracted following two different protocols. Homogenised, freeze-dried paper samples (0.1 g) and wastewater (50 mL) were spiked with isotope labelled internal standards. Briefly, solid samples (paper and reject fractions from the recycling plant), were

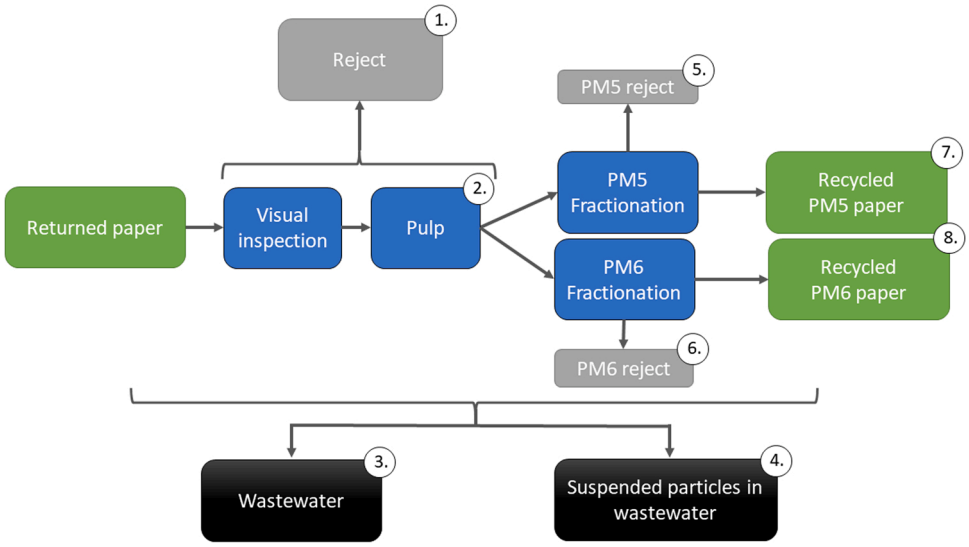

Fig. 1. Overview of recycling process at the paper recycling plant. The numbers indicate the different sampled fractions: 1. Reject from initial sorting, 2. Pulp, 3. Wastewater (dissolved PFAS), 4. Filtered particles in the wastewater, 5. And 6. Reject after removal of impurities during PM5 and PM6 fractionation, 7. Recycled PM5 paper (board) and 8. Recycled PM6 paper (thin paper).

extracted with ethyl acetate using ultrasound assisted extraction (45 min, 35°C) as described previously (Sørmo et al., 2023). Wastewater was extracted following the protocol proposed by (Arvaniti et al., 2014), applying solid phase extraction using RP-Strata X cartridges.

Extracts were analysed for 37 PFAS using liquid chromatography coupled to tandem mass spectrometry (UPLC-QqQ-MS). Information on extraction and instrumental analysis is included in the SI. Tables S3 and S4 list the PFAS included.

Quality control and quality assurance

To avoid cross contamination, clothes and equipment suspected to contain PFAS (such as water and stain repellent clothing) were avoided during sampling and sample treatment. Nitrile gloves were used during sampling and handling of the samples. Samples were kept in suitable containers, see section "Fractions on the paper recycling plant". During sample treatment and analyses, procedural and solvent blanks, as well as a standard solution, were used to control for cross contamination and carryover, and to evaluate signal variations and drifting time (details in the SI).

Statistics and mass estimates

For calculations of sum PFAS, concentrations below the LOQ were treated as LOQ/2 while non-detects were excluded from sum concentrations. Averages are presented as arithmetic means. Statistical analyses (Mann-Whitney U test) were carried out using R (wilcox.test) (R Core Team, 2023)

Pulp and reject fractions were chosen for estimating the amounts of PFAS in the recycling stream as they, due to the mixing and pulping at the factory, represent homogenised masses of the returned paper during the time of sampling. Therefore, the pulp and reject fractions were considered to be more representative for the average PFAS concentrations compared to samples of paper products. To estimate the mass of the targeted PFAS in the returned corrugated paper and board received by the paper recycling facility per year, PFAS concentrations in the pulp ($C_{PFAS\ pulp}$, μg kg^{-1}), the reject fraction (mostly plastic) ($C_{PFAS\ re}$, μg kg^{-1}), filtered wastewater ($C_{PFAS\ ww}$, μg L^{-1}), and particles from the wastewater ($C_{PFAS\ pw}$, μg kg^{-1}),were multiplied with the yearly masses of the respective sample type and summed, as shown in Eq. 1:

$$\dot{m}_{PFAS\ fac} = \frac{C_{PFAS\ pulp} \times \dot{m}_{pulp} + C_{PFAS\ re} \times \dot{m}_{re} + C_{PFAS\ ww} \times Q_{ww} + C_{PFAS\ pw} \times \dot{m}_{pw}}{10^9} \quad (1)$$

$\dot{m}_{PFAS\ fac}$ is the mass of PFAS in the returned paper received by the paper recycling facility per year (kg/year), $\dot{m}_{Pulp}$ is the mass of the pulp produced per year (kg/year), $\dot{m}_{re}$ is the mass of the reject produced per year (kg/year), Q_{ww} is the volume of wastewater produced per year (L/year), $\dot{m}_{pw}$ is the mass of the particles in the wastewater produced per year (kg/year).

In 2021, the paper-recycling facility received 119,733 tons of returned corrugated paper and board, of which approximately 90,000 tons derived from the Norwegian market. A total of 236,081 tons of corrugated paper and board were returned in Norway in 2021 (Norsk Resy, 2022). Thus, approximately 40% of all corrugated paper and board returned in Norway was recycled at this paper-recycling facility. According to the facility, the dry matter content of returned paper is 87%. An overview of the mass fractions used for mass estimates is presented in Table S5. These numbers were used to extrapolate the results from the recycling facility to the total corrugated paper and board in the recycling stream in Norway. It was assumed that the samples from the paper recycling facility were representative for all returned corrugated paper and board in Norway, and the total mass of targeted PFAS in returned corrugated paper and board in Norway in 2021 was estimated using Eq. 2:

$$\dot{m}_{PFAS\ Nor} = \frac{\dot{m}_{PaperNor}}{\dot{m}_{Paper\ fac}} \times \dot{m}_{PFAS\ fac} \quad (2)$$

$\dot{m}_{PFAS\ Nor}$ is the mass of PFAS in all the returned paper in Norway per year (kg/year), $\dot{m}_{Paper\ Nor}$ is the total mass of returned paper in Norway per year (kg/year), and $\dot{m}_{PFAS\ fac}$ is the mass of PFAS in the returned paper received by the paper recycling facility per year (kg/year).

Results and discussion

PFAS in paper products in the Norwegian market

14 of the 37 targeted PFAS were detected in paper products: PFBA, PFHxA, 7H-PFHpA, PFDoDA, PFHpS, PFOS, SAmPAP monoester, SAmPAP diester, 4:2 FTS, 6:2 FTS, 8:2 FTS, 10:2 FTS, GenX and ADONA (SI Table S6). Substantial PFBA concentrations were detected in five of the 15 paper products: V4 (6863 μg kg^{-1}), V5 (347 μg kg^{-1}), V6 (1857 μg kg^{-1}), Re5 (2492 μg kg^{-1}), and Re6 (2945 μg kg^{-1}). However, confirmation of PFBA was based in only one mass fragmentation, and as previously discussed by (Abraham et al., 2021) some uncertainty was associated with this quantification. PFBA was therefore not included in mass estimates in the main text. In the SI, PFBA results are included, though they should be considered with caution. Concentrations of other targeted PFAS are shown in Fig. 2. Sum targeted PFAS concentrations without PFBA (∑PFAS) varied between 0.5 and 971 μg kg^{-1}. Excluding PFBA, 6:2 FTS was the PFAS most commonly exhibiting the highest concentrations. Three paper products showed substantial 6:2 FTS concentrations, which consisted of virgin materials (sample V5 [559 μg kg^{-1}, 88% of ∑PFAS] and V6 [952 μg kg^{-1}, 98% of ∑PFAS]), and recycled paper (Re5 [579 μg kg^{-1}, 97% of ∑PFAS]). SAmPAP diester also presented high concentrations in two virgin paper products (sample V4 [26 μg kg^{-1}, 26% of ∑PFAS] and sample V5 [70 μg kg^{-1}, 11% of ∑PFAS]) and in the two samples produced at the recycling facility (Re7 [38 μg kg^{-1}, 62% of ∑PFAS] and Re8 [27 μg kg^{-1}, 48% of ∑PFAS]). Concentrations above 10 μg kg^{-1} were detected for GenX (sample V2 [12 μg kg^{-1}]), SAmPAP monoester (one of the samples produced at the recycling facility, Re8 [21 μg kg^{-1}]), 4:2 FTS (sample V6 [11 μg kg^{-1}], and PFHxA (sample V4 [67 μg kg^{-1}]).

No difference in PFAS concentrations between virgin and recycled paper materials was apparent ($p = 0.49$ for sum PFAS, Mann-Whitney U test). However, all samples containing virgin materials for which PFAS were detected in significant concentrations were FCMs (sample V4: carton for pumpkin seeds, sample V5: flour carton, sample V6: paper bag for fruit). This suggest intentionally added PFAS, as these substances can be used to add water and fat repellent properties, and FCMs have been reported to contain significant concentrations of PFAS (Begley et al., 2008; Glenn et al., 2021; Susmann et al., 2019; Trier et al., 2017). In contrast, the presence of PFAS in products containing recycled paper, such as sample Re5 (cardboard box for transportation or storage), is presumed to be derived from the PFAS content in the original paper used to manufacture these products, making it a NIAS. The PFAS content in the samples of paper produced at the recycling facility, Re7 and Re8, supports this hypothesis. Thus, SAmPAPs and other PFAS in this material are non-intentionally added substances (NIAS), with uncertain

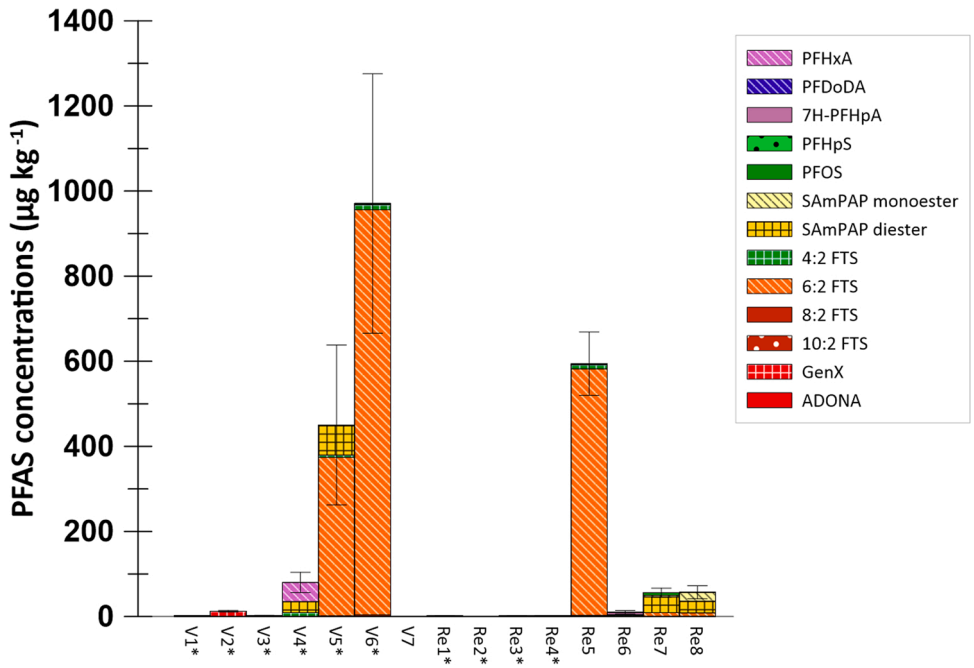

Fig. 2. PFAS concentrations in paper products: seven products on the Norwegian market consisting of virgin materials (V1: paper straw, V2: burger wrapping, V3: strawberry box, V4: carton for pumpkin seeds, V5: flour carton, V6: paper bag, V7: box for face masks), six products in the Norwegian market produced from recycled paper (Re1: burger box, Re2: strawberry box, Re3: carton for chocolate eggs, Re4: egg carton, Re5: cardboard packaging, Re6: notebook), and the two samples of recycled paper produced in the recycling facility (Re7: PM5 paper [board], Re8: PM6 paper [thin paper]). Results are the average of triplicate analyses (n = 3). Asterix (*) denotes FCMs. Error bars are showing the standard error of the mean (SEM) for the sum PFAS concentrations.

exposure routes.

Herein, 6:2 FTS dominates the PFAS profiles, which differs with previously reported profiles. Perfluoroalkyl carboxylic acids (PFCA), 8:2 FTS (1091 µg kg^{-1}), and 10:2 FTS (399 µg kg^{-1}) were dominating in a paper plate produced at a factory in Norway in 2007, while lower concentrations of 6:2 FTS (71 µg kg^{-1}) were reported (Langberg et al., 2021). Targeted PFAS vary between studies (Granby and Håland, 2018; Langberg et al., 2021), complicating comparisons. PFCA and fluorotelomer alcohols (FTOH) were reported to dominate in FCMs in the Norwegian market in 2017 (Granby and Håland, 2018). However, 6:2 FTS and SAmPAP diester (which are the dominating PFAS in the present study) were not targeted in that study (Granby and Håland, 2018). A study conducted in six European countries (The Czech Republic, Denmark, France, Germany, the Netherlands, and the United Kingdom) in 2020 showed that PFAS are still used in disposable food packaging and tableware in Europe, and that a large fraction of the total organic fluorine in the products could not be assigned to targeted PFAS (Strakov et al., 2021). That study also found fluorotelomers in their samples, though were dominated by 6:2 FTOH, not included in this study. PFAS used for paper products have changed over time (Glenn et al., 2021; Langberg et al., 2021) following a general trend of replacing the classical long-chained PFAS in many products (Brendel et al., 2018). However, it has been reported that the production of classical long-chained PFAS, such as PFOS and PFOA, has undergone a global shift; whereby manufacturers primarily located in Asia have increased the production of long-chain PFAS in contrast to other regions (Land et al., 2018). As a result, paper products may also differ in PFAS composition and content depending on where they were produced. In addition to differences in the targeted PFAS between studies, time and country of production may also explain some of the discrepancy in PFAS composition reported between studies.

PFAS in the fractions on the paper recycling plant

15 of the 37 targeted PFAS were detected in reject, pulp, suspended particles in wastewater, reject from the paper machines, or wastewater: PFBA, PFDoDA, PFTrDA, 7H-PFHpA, PFOS, MeFOSA, SAmPAP monoester, SAmPAP diester, 4:2 FTS, 6:2 FTS, 8:2 FTS, 10:2 FTS, ADONA, 6:2 Cl-PFESA, and PFECHS. However, PFDoDA and 4:2 FTS, were only detected at concentrations below the LOQ. In wastewater, only 8:2 FTS (0.9 ng L^{-1}) was detected. PFAS concentrations at the paper recycling facility are compiled in Fig. 3. High concentrations of SAmPAP diester were detected in reject PM5 (1631 µg kg^{-1}) and in suspended particles in wastewater (1111 µg kg^{-1}), while lower concentrations were detected in reject (19 µg kg^{-1}), pulp (36 µg kg^{-1}), and reject PM6 (20 µg kg^{-1}). MeFOSA was also detected in reject (23 µg kg^{-1}) and suspended particles in wastewater (156 µg kg^{-1}); SAmPAP monoester was detected in the reject (44 µg kg^{-1}), pulp (111 µg kg^{-1}), and reject PM5 (46 µg kg^{-1}); 6:2 FTS in reject (11 µg kg^{-1}), pulp (10 µg kg^{-1}), suspended particles in wastewater (120 µg kg^{-1}), reject PM5 (10 µg kg^{-1}), and reject PM6 (20 µg kg^{-1}); 8:2 FTS in suspended particles in wastewater (30 µg kg^{-1}); PFECHS, 7H-PFHpA and PFTrDA were detected in suspended particles in wastewater (49, 17 and 25 µg kg^{-1}, respectively); and PFOS in reject PM5 (11 µg kg^{-1}).

SAmPAPs dominated the fractions collected at the recycling facility, however, concentrations varied between the sampled fractions. SAmPAP mono-, di-, and tri-ester have been reported as ingredients in fluorochemical products, such as 3M's Scotchban (Martin et al., 2010; Trier et al., 2017). SAmPAP diester was the dominating PFAS in sediments in lake Tyrifjorden, Norway, which is polluted due to emissions from paper industry (Langberg et al., 2021) Therefore, the historic use of SAmPAPs for paper has likely been high, which may be the explanation for concentrations in recycled paper reported here. The differences in PFAS concentrations between the different fractions likely reflect differences in affinity of the specific PFAS. For example, the high concentration of SAmPAP diester in particles in wastewater compared to the wastewater is likely a result of the very low solubility of SAmPAP diester in water (Benskin et al., 2012). The relatively large differences between reject PM5 and reject PM6 was unexpected but may be due to differences in these fractions as PM5 produces a product of different quality

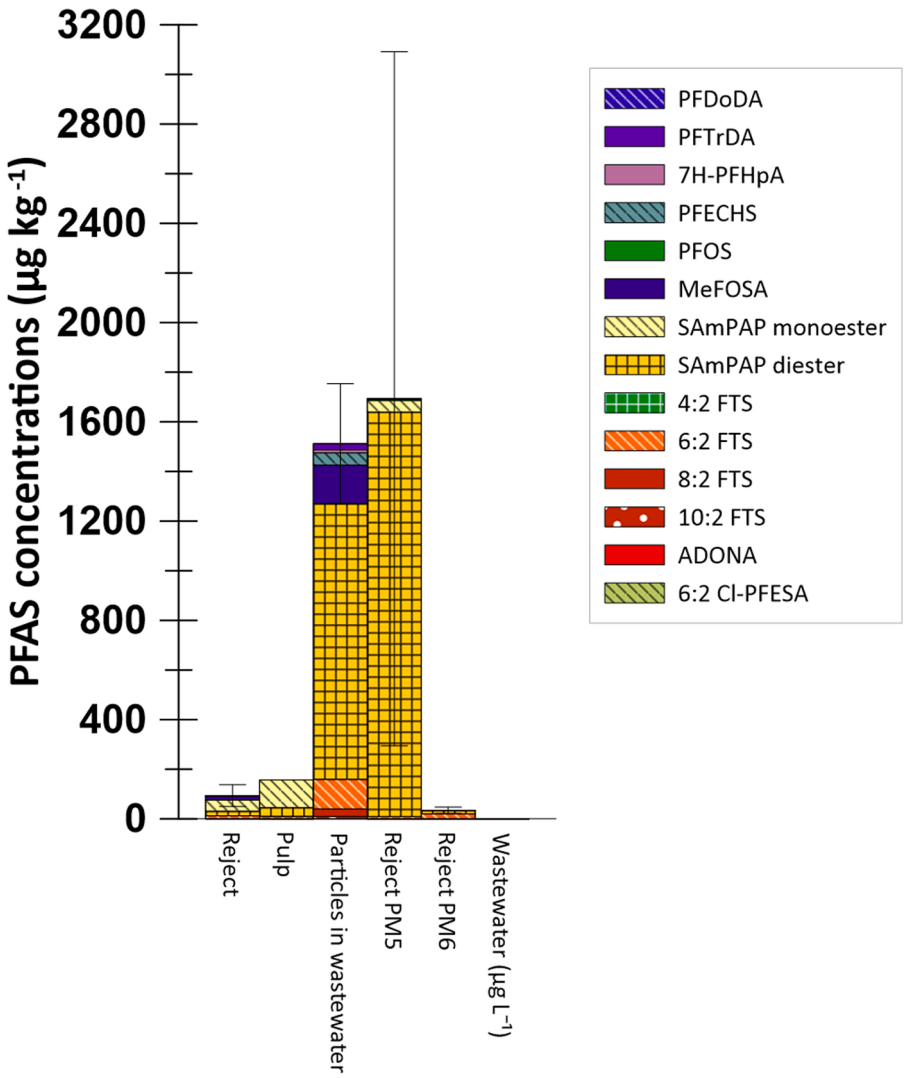

Fig. 3. PFAS detected in the fractions collected at the recycling facility [average of triplicate analyses (n = 3), except for the pulp (n = 1)]. Error bars are showing the standard error of the mean (SEM) for the sum PFAS concentrations.

(cardboard) compared to PM6 (thin paper). It should be noted that there were large variations for the detected PM5 concentrations.

PFAS in the recycling stream and mass estimates

SAmPAP diester and 6:2 FTS were the dominating PFAS in the paper products in the present study. Following the development of new chemical mixtures and increasing focus on the negative effects of PFOS and related chemicals (including SAmPAPs), a shift towards other PFAS has been reported (Langberg et al., 2021). In lake Tyrifjorden, Norway, this shift occurred around 1990 in the PFAS mixture emitted from the paper factory – before the phase out of PFOS and related compounds in the early 2000s. Emitted PFAS changed from being mainly SAmPAPs to a mixture dominated by long-chain FTS (8:2, 10:2, 12:2, and 14:2 FTS) (Langberg et al., 2021). A shift towards shorter PFAS has also been reported (Glenn et al., 2021). The predominance of 6:2 FTS reported here may be attributed to a shift towards shorter FTSs.

SamPAPs and FTSs have been identified as precursors to the persistent perfluoroalkyl acids (PFAA) (Wang et al., 2011; Zhang et al., 2018). Apart from the potential direct toxicological risk these precursors may pose, there exists and indirect risk due to transformation into PFAA, which are well-documented to be harmful for human health and the environment. The concentrations of precursor PFAS reported here indicate that PFAS in paper products could potentially be a significant source of PFAA to the environment – especially if paper products containing PFAS are not handled properly, as previously reported (Bugsel and Zwiener, 2020; Michigan PFAS Action Response Team, 2020). Alternatively, there is a possibility that precursor PFAS undergo transformation into PFAAs during the recycling process, although no conclusive evidence supporting this was observed in the present study.

Based on Eq. 1, the corrugated paper received by the recycling facility contained a total of 16.1 kg PFAS per year (for the sum of the PFAS targeted by the chemical analyses in the present study; see the SI Table S5). Amounts of PFAS in emitted water (particles + dissolved) were less than 0.2 kg per year. By extrapolating the results to the total amount of returned corrugated paper in Norway, a total of 31.7 kg PFAS per year was estimated. The population in Norway was reported to be 5,391,369 at the start of 2021 (Statistics Norway, 2023). Thus, the estimated mass of PFAS in returned corrugated paper in Norway euqalled approximately 6 mg per capita per year. Although substantial numbers, these amounts are small compared to the amounts previously reported for production of FCMs, i.e., tons of emitted PFAS per factory per year, see e.g., (Neltner, 2018) or (Langberg et al., 2021). Therefore, part of these lower emissions is due to FCMs not being the main source of paper at the facility, as well as a shift in how much of the target PFAS are used in FCMs. It has been reported that PFAS emissions from wastewater

treatment plants in Sweden euqalled 16 mg per person per year (Gobelius et al., 2023) – indicating that more PFAS is emitted via wastewater per person per year than the whole yearly mass of PFAS in the corrugated paper recycling stream per person. Only a small fraction of the sum PFAS in corrugated paper received by the recycling facility was emitted via water. The relatively low mass estimates reported here compared to FCM production and wastewater emissions are likely reflecting low levels of PFAS in the specific fraction of returned paper investigated here (corrugated paper and board) compared to e.g., a fraction dominated by FCMs. This is also evident from Fig. 2, where Re7 and Re8 had much lower PFAS concentrations than the most contaminated FCMs (V5 and V6). Reduced use of PFAS due to previous and ongoing phase out is also likely contributing. However, this estimate is only accounting for target PFAS considered. Other PFAS than those considered here, such as 6:2 diPAP which was recently found ubiquitous in toilet paper, could contribute to emissions (Thompson et al., 2023).

Perspectives and future research needs

In the present study, PFAS were reported for both virgin materials and recycled paper products sampled in Norway. Although the recycling facility receives mostly returned corrugated paper, PFAS detected in the different fractions at the facility reflects the presence of PFAS in the recycling stream as a NIAS. Due to their extensive use, PFAS contamination in recycling streams is expected. It has been argued that permitting trace levels of PFAS may be necessary to facilitate the use of recycled materials (Ng et al., 2021). At present, only a few PFAS are restricted in Norway and the EU. The use of PFOS has been restricted in Norway since 2008 (Norwegian Government, 2006), followed by the restriction of PFOA in 2014 (Norwegian Government, 2013). The use of most other PFAS has mostly been unregulated. However, current and future phase-out of PFAS as part of the proposed PFAS restriction in Europe (ECHA, 2023) will likely lead to reductions in recycling streams over time. Monitoring of target PFAS, non-target and total organofluorine (Schaider et al., 2017) is necessary to confirm the reduction of PFAS entering recycled paper. Those methods are particularly of relevance as there may be a shift towards new PFAS mixtures, which pose a challenge for analytical monitoring. Therefore, it is crucial to implement and standardise approaches for monitoring PFAS in paper products during both entry into and recycling within the system. This essential activity is necessary to monitor the success of transitioning towards a safe and sustainable circular economy.

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CRediT authorship contribution statement

Håkon Austad Langberg: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Project administration, Visualization, Writing – original draft, Writing – review & editing. **Hans Peter H. Arp:** Conceptualization, Funding acquisition, Methodology, Supervision, Writing – review & editing. **Gabriela Castro:** Data curation, Formal analysis, Investigation, Writing – original draft, Writing – review & editing. **Alexandros G. Asimakopoulos:** Project administration, Resources, Supervision, Writing – review & editing. **Heidi Knutsen:** Conceptualization, Funding acquisition, Project administration, Writing – review & editing.

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.

Data availability

Data is included as supplementary material.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.hazl.2023.100096.

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