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Human exposure to per- and poly-fluoroalkyl substances (PFAS) in Asia and contributing factors, with a focus on East Asia

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The production and use of PFAS† in some countries, coupled with uncertainties about their applications across Asia, underscore the urgent need to assess human exposure—particularly beyond China, Japan, and South Korea, which account for 80–90% of existing environmental PFAS studies. Exposure levels vary significantly across the region, with industrial activities, including textile and automotive manufacturing, contributing to severe contamination, especially in freshwater sources. Studies have detected PFAS in surface and groundwater across 20 Asian countries (~3000 samples), sometimes at concerning concentrations. Contamination extends to drinking water and food products, further increasing human exposure risks. There is now substantial evidence, particularly from China, South Korea, and Japan, indicating a widespread presence of long-chain PFAS in human serum and breast milk. Additionally, replacement compounds and their degradation products, such as 6:2 chlorinated polyfluorinated ether sulfonate (6:2 Cl-PFESA) and the dimer and trimer acids of hexafluoropropylene oxide (HFPO-DA, and HFPO-TA), are increasingly detected in human samples in China, where they are produced. Although the situation in the rest of Asia is currently unclear due to limited data, given the widespread PFAS contamination in water and food sources in the studied areas of Asia, human exposure is highly likely. Beyond direct contamination, additional risk factors in certain Asian regions are likely to exacerbate exposure, including industrially impacted freshwater resources, self-supplied and untreated drinking waters, and high reliance on fish and seafood (including wastewater-fed fisheries) in some countries. Conversely, dietary patterns, such as vegetarianism in some regions (*e.g.* India), may influence PFAS exposure differently. Despite these concerns, PFAS regulations in Asia typically fall behind those in Western countries, resulting in significant gaps in risk assessment and regulatory oversight. There is also less pressure to systematically characterize exposure levels and associated health risks. This article examines the pathways of PFAS exposure in Asia, focussing on East Asia due to the availability of data. It examines the main factors contributing to exposure, including PFAS production and associated industries, as well as the consumption of contaminated food and water. The article also identifies future research needs aimed at enhancing the understanding and mitigation of PFAS risks in Asia.

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Environmental significance

Per- and polyfluoroalkyl substances (PFAS) pose a growing environmental and public health concern across Asia, driven by industrial production, widespread use, and limited regulatory oversight. While most existing research has focused on China, Japan, and South Korea, emerging evidence suggests significant PFAS contamination across freshwater systems, drinking water, and food in at least 20 Asian countries. Human exposure—particularly to long-chain PFAS and newer replacement compounds—is increasingly documented in blood and breast milk samples in Asia, especially in the data-rich eastern region. Additional risk factors such as self-supply of untreated drinking water, industrial pollution, and high reliance on local aquatic food sources further elevate exposure risks. Currently, PFAS regulation and exposure monitoring in much of Asia remain limited, highlighting an urgent need for greater surveillance, risk assessment, and targeted mitigation strategies across the region.

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† The full forms of acronyms used in this article are given in Table S1 (SI).

1. Introduction

The investigation and management of environmental contamination by PFAS have increased substantially since the beginning of this century, particularly in the past decade.[1–3] These activities have predominantly occurred in Western nations, including North America, Europe and Australasia. However, significant PFAS production and use has also occurred in Asia, necessitating an understanding of environmental and human exposures as well as the associated risks in this region. Although PFAS production in China, Japan and South Korea has received considerable attention over the last couple of decades, much less information is available related to PFAS production, use, and human exposure in Asia outside China.

In the 20th century, PFAS manufacturing in Western nations was dominated by electrochemical fluorination (ECF), *e.g.* perfluorooctanoic acid (PFOA) and perfluorooctanesulfonyl fluoride (POSF)-based chemistries by 3M,[1] and fluorotelomer production by several companies such as DuPont. Under pressure from the US EPA, 3M ceased production of PFOA and POSF-based materials in 2000–2002, as well as related C_6 and C_{10} chemistries, and later a US EPA-sponsored program phased out the production of other long-chain perfluoroalkyl substances in the US from 2009 to 2015. During this period, major producers in Western countries started replacing long-chain PFAS with shorter-chain homologues or other fluorinated and non-fluorinated substances.

Consistent with Bowles *et al.* 2024,[4] for this paper, we define:

- Long-chain perfluoroalkyl acids (PFAAs) by the length of the fluorinated chain (n) with $n \geq 6$ for perfluoroalkyl sulfonates (PFSAs) and $n \geq 7$ for perfluoroalkyl carboxylic acids (PFCAs).
- Ultra-short-chain PFAAs as having $n = 1$ to 3.
- Short-chain PFAAs as those with chain lengths between the above two definitions.

For this paper, discussion of long-chain and short-chain PFAS implicitly includes the corresponding polyfluoroalkyl precursors to the long-chain and short-chain PFAAs, although the strict definition applies only to the perfluoroalkyl moiety.

Trends in chemical regulation and market pressures, in Asia and globally, have shifted both production chemistries and their geographic distribution. The industry phase-outs and tighter PFAS regulation in Western Countries shifted the production of long-chain PFAS to Asia, mainly China, India, and Russia.[2] Moreover, the PFAS produced in Asia are often at least slightly different from those dominating production and use in Western nations. As the toxicology of some of these different PFAS chemistries has generally been less well studied, this limits our understanding of any associated risks.

Several studies have provided substantial data on PFAS emissions from China, Japan, and South Korea.[5–10] While the PFAS literature from other Asian regions is smaller, it is rapidly expanding. As of 2023, PFAS research has been conducted in 24 out of 49 Asian countries, with >80% of these studies originating from East Asia (predominantly from mainland China, Taiwan, Japan, and South Korea).[8] Studies in other countries primarily examine the presence of PFAS in the environment, with less emphasis on the exposure in humans and other organisms. Some studies have focussed on the impact of PFAS manufacturing on environmental and human exposure.

Available geographical information for PFAS contamination in Asia is largely limited to areas of PFAS production and intense industrial activity. In contrast, there are significant remote and rural regions in Asia where PFAS use is likely to be limited, but little information exists to support this assumption. PFAS have even been found in snow and stream waters on Mt. Everest, likely due to their extensive use in nearby countries.[11] Three out of the 14 PFAS tested for (PFOA, perfluorooctanesulfonic acid – PFOS and perfluorohexanoic acid – PFHxA) were detected, with PFOS identified at the highest concentrations of 26.14 ng L^{-1} and 10.34 ng L^{-1} in snow and meltwater, respectively. There is significant diversity in economic development within and among Asian countries. Cultural practices affecting dependence on specific food types or sources of drinking water also influence human exposure to PFAS. Consequently, it is anticipated that human exposure to PFAS will vary widely within Asia and may not exactly replicate exposure patterns in Western countries.

Noting that PFAS production in China and Japan has been significant, and that data from China, Japan, and South Korea currently influence the scientific and public health discussion on PFAS in Asia, we aimed to acknowledge the differences within and among Asian countries (*e.g.*, economic development, food and water sources, cultural practices, dietary preferences) to draw conclusions, where possible, for other countries in East, Southeast, South, and West Asia.

Our aim for this study was first to examine if sufficient data exist demonstrating the problem of human exposure in Asia. We subsequently examined factors contributing to that exposure, notably, PFAS production and related industries, environmental concentrations, and concentrations in drinking water and food products. Finally, we attempted to briefly address societal aspects in parts of Asia that may result in different patterns of PFAS exposure compared to Western nations. In doing so, we identified critical gaps that must be addressed to better understand the PFAS risk in this region.

In this article, we examine four themes:

(1) Evidence of human exposure to PFAS in Asia in comparison with other countries/regions.

(2) Production and use of PFAS in Asia, especially in manufacturing sectors.

(3) The status (which PFAS, where and how much) of key pathways of human exposure (water, food, *etc.*) and Asia-specific influences on exposures, such as from dietary and drinking water sources.

(4) Regulatory responses, knowledge gaps and research needs in Asia.

It is worth emphasizing that we have opted to discuss human exposure, rather than estimating health risks. This approach was taken due to (i) the limited dataset for PFAS, (ii) the restricted availability of health-based guidance values for individual PFAS, and (iii) significant variations in health-based guidance values across different countries and regions. Consequently, comparing health risk estimates between geographical

areas may be misleading. The conclusions identify research gaps that require further understanding and provide recommendations for additional investigation, regulation, and management.

2. Methodology

A comprehensive search was conducted across Web of Science, PubMed, Scopus, and Google Scholar using defined terms related to PFAS in water, drinking water, human serum, breast milk, fish/seafood *etc.* The search incorporated general and specific PFAS-related keywords, including "per- and polyfluoroalkyl substances" OR PFAS* OR "perfluorinated compounds", combined with relevant environmental or biological matrices using Boolean operators. Specifically, searches included (PFAS* OR "perfluorinated compounds" OR "per- and polyfluoroalkyl substances") AND (water OR groundwater OR surface water OR wastewater) for general PFAS in water; (drinking water OR tap water OR potable water) for PFAS in drinking water; (serum OR blood OR plasma) for PFAS in human serum or blood; (breast milk OR human milk) for PFAS in breast milk; and (fish OR seafood OR marine organisms OR shellfish) for PFAS in aquatic food sources.

As the keyword "Asia" is not commonly used by researchers; the search was performed for each country individually. Countries with published data on specific matrices were first identified using Web of Science and then refined using other databases such as Scopus. To ensure accuracy, review articles were excluded, focusing solely on original research studies to prevent duplication and to count only primary data. Additionally, search results were cross-checked for duplicate studies across databases, and irrelevant records (*e.g.*, studies on non-PFAS fluorinated compounds or unrelated environmental contaminants) were removed based on title and abstract screening. After refining the dataset, only studies with clear country attribution (*e.g.*, research conducted in Japan) were retained for further analysis.

The SI shows the distribution of studies in various Asian countries for categories like PFAS in water and PFAS in serum, blood, or plasma, *etc.* Web of Science returned a greater number of studies for any given country, perhaps due to its coverage of conference papers and grey literature. PubMed and Scopus yielded similar numbers, focusing on peer-reviewed journals. Scopus was used to gather data on study distribution of PFAS in various matrices, presented in the SI. China, Japan, and South Korea dominated 80–90% of the studies, depending on the subtopic. Fig. 1 displays the country-wise distribution of the total number of studies in Asia (355) on PFAS in human serum/plasma or blood. For "PFAS* AND water", nearly 1100 studies were identified, whereas for other matrices such as "PFAS* AND drinking water or tap water", the number was much smaller (185) and even lower in some other cases, such as PFAS in fruits and vegetables (25). However, in all cases China contributed the largest proportion of the total, as shown by the data in the SI.

Based on criteria such as data availability, study duration and volume of data, 24 research studies reporting PFAS in humans across Asia were selected for detailed analysis,

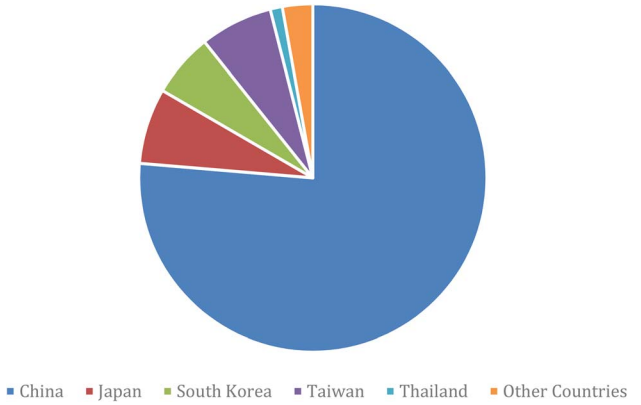

Fig. 1 Country-wise distribution of number of published studies on PFAS in human serum/blood/plasma in Asia (355 in total), based on the Scopus database.

including temporal trends in serum PFAS levels. The majority of the studies (18) reported serum concentrations and three each reported PFAS concentrations in whole blood or plasma samples. The data were harmonized to serum equivalents using published matrix-conversion factors (Kuo *et al.* 2023).[12] To convert whole-blood concentrations to serum concentrations, a whole-blood-to-serum ratio of 1 : 2 was used and to convert plasma concentrations to serum concentrations, a conversion ratio of 1 : 1 was applied.

3. Existing evidence of human exposure to PFAS in Asia

Although much of the scientific literature has focused on evidence of PFAS exposure in Asia associated with PFAS production facilities (which is relevant to only a few countries in Asia such as China, Japan and India), PFAS measurements in serum and breast milk demonstrate that PFAS exposure in Asian countries is widespread and varies greatly in different regions.

3.1. PFAAs in human samples in Asia

During the last two decades, 21 out of 49 (43%) Asian countries have published PFAS studies on human samples, with China leading the rest of Asian nations, accounting for about 75% of the total (Fig. 1). Three countries, *i.e.* China, Japan and South Korea together represent about 90% of total number of studies. In Fig. 2, serum concentrations of PFAAs in some Asian countries sampled between 2013 and 2021 are compared with data from other regions during the same period. For China, data were selected from cities without major industrial sources to avoid complication with exposures from PFAS production areas. The serum concentrations of C_8–C_{11} PFCAs are higher in China and South Korea than in Europe, the U.S. and some other Asian countries, which is likely due to the use of long-chain PFAAs in China. For short-chain PFAAs (PFHxA, perfluoroheptanoic acid – PFHpA and perfluorobutanesulfonic acid – PFBS), concentrations in the Asian population are generally comparable to

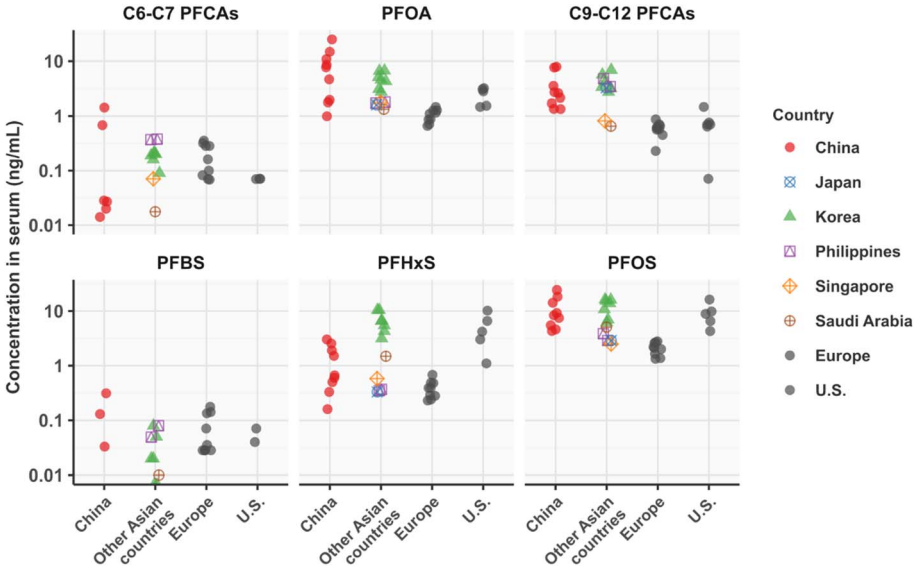

Fig. 2 Concentrations of PFAAs in serum in Asia and other regions sampled between 2013 and 2021 extracted from published studies. Markers represent median or mean concentrations depending on the data provided. Human samples collected include whole blood and serum. These were harmonised to serum equivalents using published approaches. The U.S. data are sourced from various health agencies.[13–17] See Table S2 (SI) for source data and more details.

those in other regions. PFHxS and PFOS serum concentrations in China and South Korea are higher than in Europe, some other Asian countries and the general U.S. population (4.3 ng mL^{-1} in 2017–2018) as reported by CDC (2024)[13] but similar to the levels, in exposed communities (medians 6.6–16.1 ng mL^{-1}, respectively), reported by various U.S. agencies.[14–17] A small number of studies on PFAS in human samples is available for other Asian countries as well (*e.g.* Thailand, Malaysia, Vietnam, Singapore, and UAE). A serum study conducted in Jeddah analysed 17 PFAS in 208 individuals (aged 40–89 years) and 7 PFAS were detected in >80% of samples.[18] Four PFAS, namely PFOS (median concentration: 5.08 ng mL^{-1}), perfluorohexanesulfonic acid – PFHxS (1.49 ng mL^{-1}), PFOA (1.33 ng mL^{-1}) and perfluorononanoic acid – PFNA (0.55 ng mL^{-1}) accounted for 94% of the total PFAS serum levels. In a nationwide survey in India, 25 PFAS were measured in 39 human hair samples collected from 14 cities in India.[19] PFHxS, PFOS and PFOA were dominant among the nine compounds quantified.

Individuals working in fluorochemical production plants are at greater risk of exposure to PFAS. For example, substantial occupational exposure was noted in workers at a fluorochemical plant located in Hubei province, China. Very high median serum concentrations of PFHxS (764 ng mL^{-1}), PFOA (427 ng mL^{-1}) and PFOS (1725 ng mL^{-1}) were found in these workers ($n = 302$).[20] Even higher concentrations (with median values of 2250, 603 and 5544 ng mL^{-1}, for the three compounds, respectively) were observed in the employees ($n = 101$) working in the sulfonation department, in this study. Furthermore, exceptionally high serum concentrations of C_4 to C_{12} PFAS were also found in fishery employees from Tangxun Lake, located near a production base of the fluorochemical industry in Wuhan, China.[21] The median serum concentration of PFOS in these employees ($n = 39$) was as high as 10 400 ng mL^{-1} (highest reported globally), as compared to 18.7 ng mL^{-1} in a reference group from the same city ($n = 9$). However, the authors observed that this was not a typical case of occupational exposure.

Breast milk can be a primary source of infant exposure to PFAS and is also a marker of maternal exposure.[22] Yao *et al.*[23] reported the widespread occurrence of PFAS in human milk in China, based on a nationwide survey (21 cities). PFOA was found to be the dominant PFAS, accounting for 63% of the total load, followed by PFOS (13%). The contamination levels varied widely among cities, reflecting the local sources such as manufacturing and other industrial activities (*e.g.*, electroplating). Contamination of breast milk with PFAS has been reported in several other Asian countries including Japan, South Korea, Vietnam, Malaysia, Jordan and Philippines.[24–26] Fig. 3 shows the concentrations of PFAAs in breast milk sampled between 2015 and 2020 in different countries/regions (for additional details see SI, Table S3). Generally, PFAA concentrations in breast milk in Asia are broadly comparable to those in other regions/countries, except for the higher concentrations of C_9–C_{12} PFCAs and PFOS observed in China. PFOA levels in breast milk in China were found to be 4–24 times higher than those reported in the U.S. and several European countries.[23] However, the levels of short-chain PFCAs (perfluorobutanoic acid – PFBA and PFHpA) were lower than those in European countries (Fig. 3). A number of PFAS were detected in the breast milk samples from several Pacific Island nations during 2008–2019.[27] The samples from Kiribati showed particularly high concentrations of PFHxS (111.0 pg g^{-1}) and PFOS (211.9 pg g^{-1}) in 2018. While the causes of this are still not clear, this points to the potential for significant exposure even at locations remote from PFAS manufacture and heavy industry.

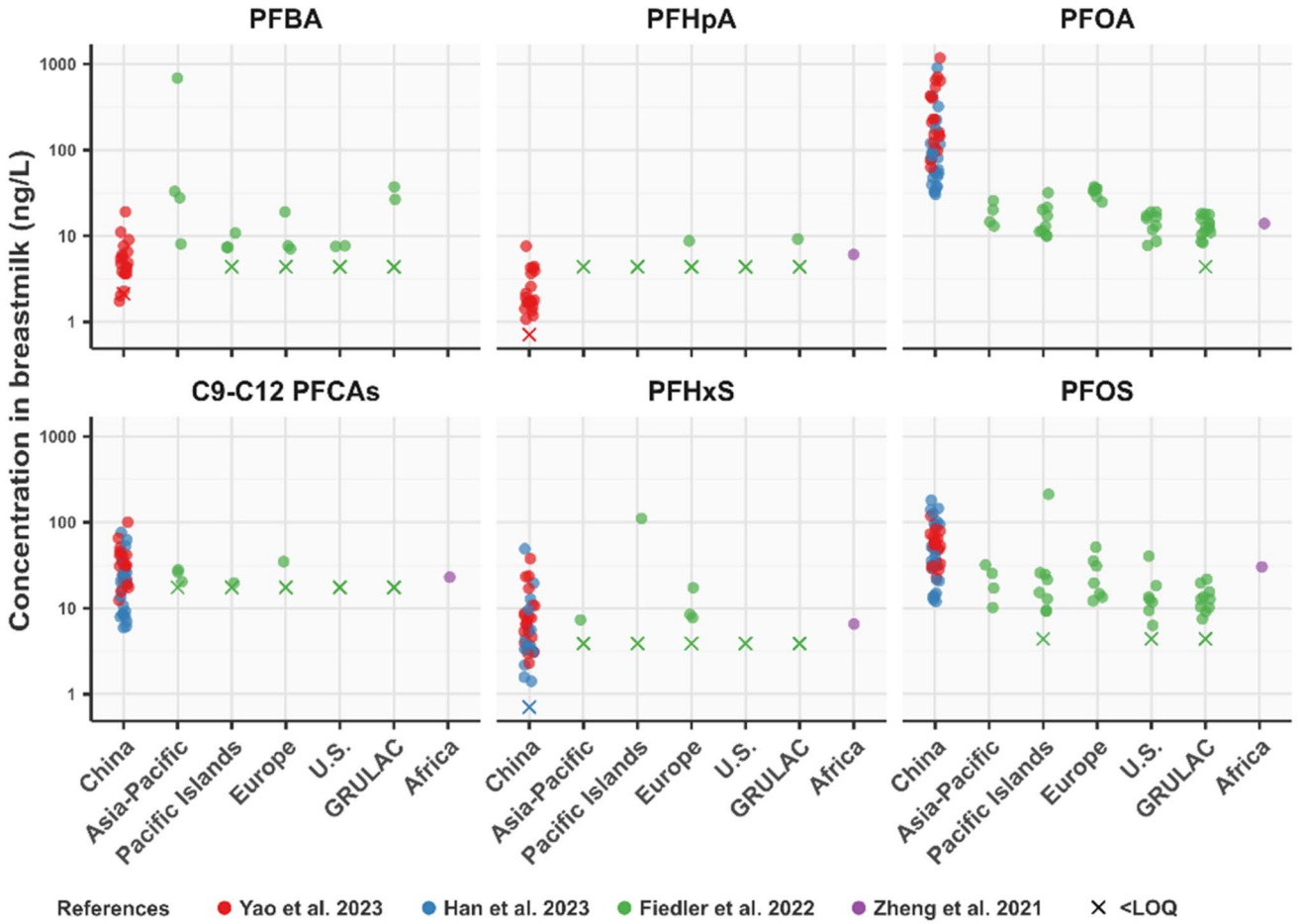

Fig. 3 Concentrations of PFAAs in breast milk sampled between 2015 and 2020. Markers represent median or mean concentrations depending on the data provided in the literature. GRULAC represents the Group of Latin America and the Caribbean. See Table S3 (SI) for source data and more details.

3.1.1. Temporal trends of PFAS in humans in Asia and other continents. Serum concentrations for two important legacy PFAS (PFOS and PFOA) over two decades (2000–2021) in several Asian countries, the US and Europe are shown in Fig. 4. Long-term data are predominantly available for a few countries (mainland China, Taiwan, Japan, and Korea), but they are missing for most Asian countries. Clear temporal trends are absent for many Asian countries (Fig. 4). While no clear temporal trend was observed for both PFOA and PFOS in the population of South Korea between 2006 and 2015,[28] a statistically significant ($p < 0.05$) decline was observed in Japan for PFOS in blood but not for PFOA (Liu *et al.* 2020; Fig. 4). The longer-term data in the populations of the U.S., Australia[29,30] and some European countries[31] indicates a decline in levels of PFAS.

In China, both PFOA and PFOS are produced and used,[32–34] and their concentrations in human samples can be greatly influenced by local industrial sources, such as electroplating and fluoropolymer production. However, following their listing under the Stockholm Convention, these compounds have largely been phased out in China in recent years. For example, PFOA was phased out in 2023 and PFOS was included in China's Strictly Restricted Toxic Chemical List in 2018.[35] In areas where PFAS in the Chinese population have been reported for at least two time periods during the past two decades, the PFOA serum concentration revealed an increasing trend with time ($p < 0.05$), while the PFOS concentration generally decreased from 2004 to 2017 (Fig. 5). Similar time-trends for PFOA and PFOS have also been observed in breast milk samples from China (Fig. 5)

3.2. Emerging PFAS in human samples in Asia

Alternative compounds, such as F-53B and HFPO-TA, have been produced and used locally in China for some time.[36] F-53B, a PFOS alternative, is extensively used as a mist suppressor in electroplating (chrome) industry. Its main component, 6:2 Cl-PFESA, is now widely detected in the Chinese population. According to Chen *et al.*,[37] it is now the third most prevalent PFAS in Chinese population, after PFOS and PFOA. In a national human milk survey between 2017 and 2020, Han *et al.*[38] found 6:2 Cl-PFESA in 97 of 100 pooled human milk samples collected from 24 provinces of China. Yao *et al.*[23] also detected 6:2 Cl-PFESA in 99.7% of 1151 human milk samples collected from 21 Chinese cities between 2020 and 2021. Higher levels of PFOS

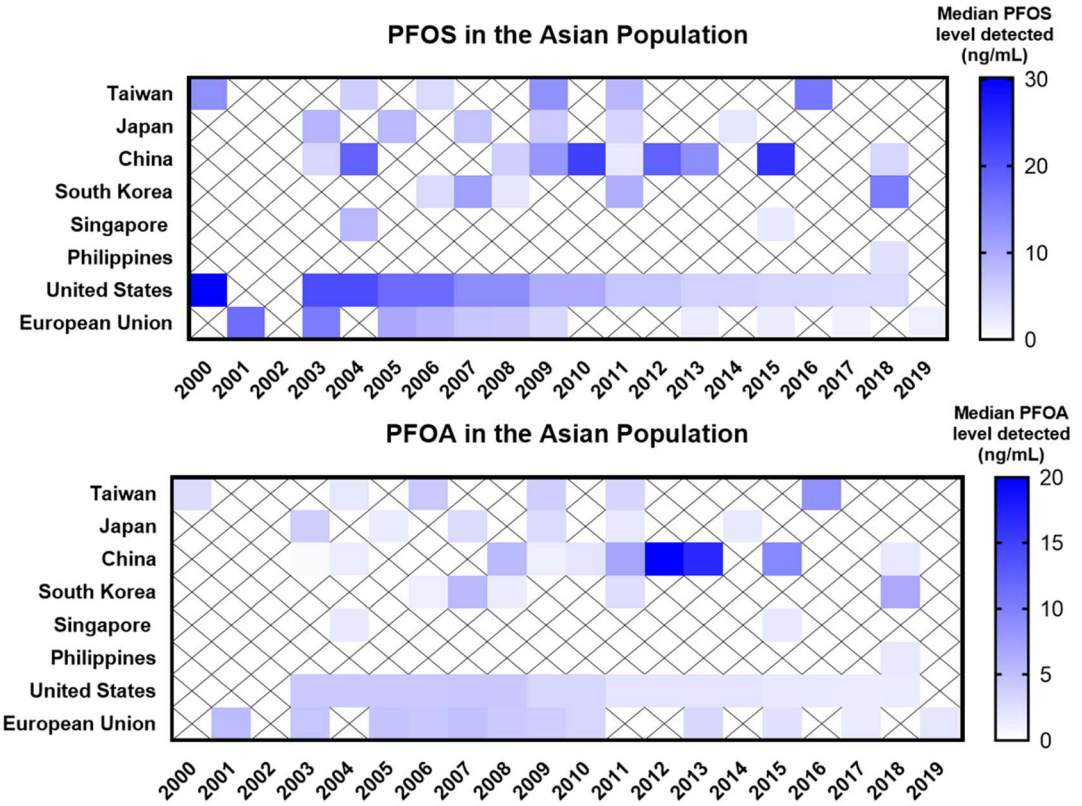

Fig. 4 PFOS and PFOA in the Asian population contrasted with the US and Europe. Median PFOS and PFOA levels were determined in Asian countries by comparing studies on PFAS conducted between 2000 and 2021. Human samples collected included whole blood, serum, and cord blood, which were harmonized for serum equivalents.

and F-53B were found in the cities of eastern China, possibly due to the presence of the local electroplating industry. Several studies have detected 6:2 Cl-PFESA in serum samples from the Chinese population[38–40] Liu *et al.*[41] reported a median serum concentration of 5.48 ng mL^{-1} (2.3–15.8 ng mL^{-1}, 1st–3rd quartile) for 480 pregnant females in Tianjin, China between 2010 and 2012, while Duan *et al.*[42] reported that the median concentration in the general population in the same city was 8.64 ng mL^{-1} (3.39–22.3 ng mL^{-1}, 1st–3rd quartile, $n = 294$) in 2017. Generally, the PFOS level in serum is higher than 6:2 Cl-PFESA but in some cases the differences are small. For example, in the above-mentioned study by Duan *et al.*,[42] the median PFOS serum concentration was in the similar range *i.e.* 14–24 ng mL^{-1} (8.69–22.92 ng mL^{-1}, 1st–3rd quartile). Liu *et al.*[43] summarized the 6:2 Cl-PFESA/PFOS ratios in human blood in China from published studies, which ranged from 0.01 to 1.22. The ratios for breast milk were between 0.07 and 0.87 calculated based on the median concentrations by Han *et al.*[38] and Yao *et al.*[23] Although 6:2 Cl-PFESA is mainly used in China, it has also been detected (albeit at a trace level) in surface waters of several other countries, including some in the West.[44] This suggests its potential long-range dispersal *via* products exported from China. However, no report is available on its occurrence in human samples from any other Asian country, so far.

Perfluoroether carboxylic acids (PFECAs) are used as alternatives to legacy PFAAs in fluoropolymer resin manufacturing. These have also been detected in human samples. For example, Yao *et al.*[45] analysed 13 novel PFAS (PFECAs and perfluoroether sulfonic acids-PFESAs) in 977 serum samples collected from residents living near fluorochemical plants in Shandong, China. Perfluoromethoxyacetic acid (PFMOAA), perfluoro-4-oxahexanoic acid (PFO4DA), and perfluoro-5-oxaoctanoic acid (PFO5DoDA) were frequently detected (>95%), at respective median concentrations of 12.91, 0.142, and 0.987 ng mL^{-1}. The study raised concerns about the potential health impacts associated with these alternative PFAS.

3.3. Overall human exposure to PFAS in Asia

Clearly, the data and the discussion presented above provides substantial evidence of human exposure to PFAS occurring in Asia. Although most (>80%) of the available data originate from China, Japan, and South Korea, there is growing evidence of human exposure across other parts of Asia as well. PFAS exposure in Asia is rising and requires urgent mitigation. A thorough understanding of potential sources of PFAS exposure is essential. Although production is a relevant source of exposure for certain countries such as China and Japan, in most other instances, water and food are expected to be the primary sources of exposure. The following sections provide a detailed discussion of these sources, particularly within the context of Asia.

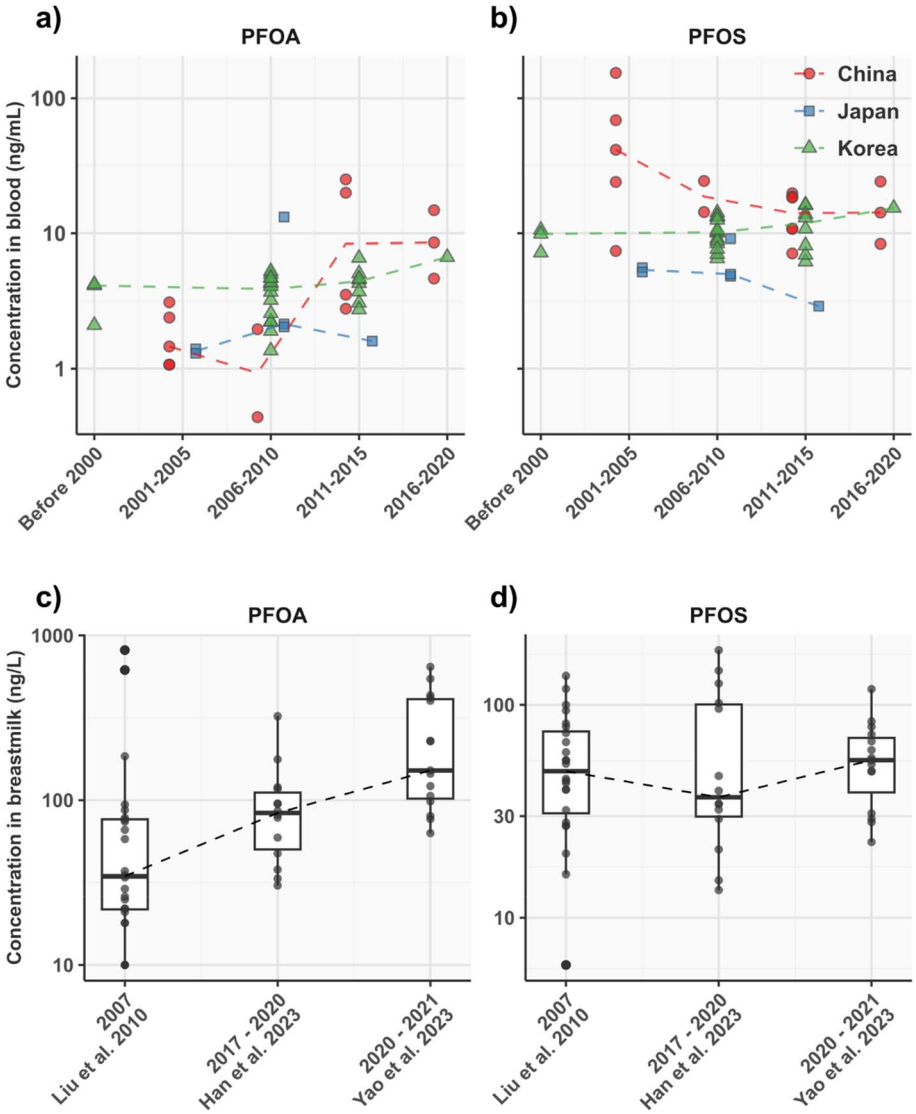

Fig. 5 Time-trends in PFOA and PFOS concentrations in blood (a and b) and breast milk (c and d) in the Asian population. Data are extracted from published studies and mainly originate from China, Japan and Korea (covering >80% of published data). Each marker represents the median, arithmetic mean or geometric mean concentration of each sample group, depending on the descriptive statistics provided in the corresponding study. For the blood concentrations (a and b), the dashed lines connect the geometric mean concentrations in each period. For the breast milk concentration plot, the horizontal line inside each boxplot indicates the median of the dataset. The grey dashed lines that connect the medians in plots c and d are for visual guidance. See Tables S4 and S5 (SI) for source data and more details.

4. Production and use of PFAS in Asia

Global trends in PFAS production have influenced PFAS production and usage in Asia, with an increase (at least temporarily) in long-chain PFAS and perfluoroether chemistries in Asia in the 21st century. Increases in industrial productivity have correspondingly resulted in greater volumes of PFAS production in Asia and in substantial releases to the environment in some regions. Many industries in Asia use PFAS, with fluoropolymer production, electroplating, automotive production, electronics, and textile treatment, being major examples. As in Western nations, the use of aqueous film-forming foams (AFFFs) has impacted many commercial sectors and locations. Production trends in Asia of both legacy PFAS and their replacement products are discussed below.

4.1. Fluoropolymer production

Fluoropolymer production is a major source of PFCAs and many other PFAS.[46] The fluoropolymer industry in China has expanded rapidly since the 2000s. For example, according to the China Association of Fluorine and Silicone Industry (CAFSI), the production capacity of polytetrafluoroethylene (PTFE) in China surged from 0.66×10^4 t per year in 1999 (ref. 2) to 18.8×10^4 t per year in 2021.[47] The production capacity of polyvinylidene fluoride (PVDF) reached 10.3×10^4 t per year in 2022.[48]

In Japan, major fluorochemical industries have been a significant contributor to PFAS production and contamination.[49,50] These companies have sales offices throughout Asia. For example, AGC sells a range of fluoropolymer-based products in several Asian countries.[51] PFAS production in Japan has

declined in recent years. According to the OECD, Japan has banned the production and import of PFOS (2010) and PFOA (2021), and PFHxS (2024).[52] It is also addressing PFOS containing firefighting foams, promoting alternatives, and tracking stockpiles, which have decreased by 2024. A survey conducted in November 2024 showed that the total volume of AFFF decreased by 45%, and that of PFOS decreased by 36% compared to the previous report in 2020. There is limited publicly available data on the production, distribution, and use of PFAS in India. After 2010, the production of PTFE increased in India (*ca.* 2.3×10^3 t in 2011 and 7.5×10^3 t in 2012).[53] In 2021, the production capacity of PTFE in India and Russia was estimated to be 1.5×10^4 t per year and 1.4×10^4 t per year, respectively. This may have resulted in increased use of commercial PFOA in fluoropolymer production.[2]

Historically, PFOA was used as a processing aid in the production of PTFE.[3] While the major fluoropolymer manufacturers in developed countries were replacing PFOA with PFOA-free processing aids during the early 2000s, PFOA was still used in PTFE production in China. The global emission inventories for C_4–C_{14} PFCA homologues by Wang *et al.* (2014)[2] highlighted that PFOA production in China increased from negligible before the 1990s to an annual production of about 50–80 tonnes in 2009. Consistent with this, Li *et al.*[5] estimated that a total of about 480 tonnes of commercial PFOA was produced during 2004–2012, primarily for use by the fluoropolymer industry in China. Following the listing of PFOA in Annex A of the Stockholm Convention in 2019, China initiated measures to restrict the use of PFOA. In recent years, the fluoropolymer industry in China has replaced PFOA with C_6 alternatives[47] such as PFHxA, HFPO-DA and HFPO-TA.[54] A major fluoropolymer manufacturer in China stopped producing PFOA in 2020 and modified the facility to produce PFHxA in 2024. Considering the above, the emission of shorter chain PFCAs in China may increase in future.

4.2. PFAS used in electroplating

Electroplating is a major industry in Asia, especially due to the rapid growth of automobile production in several Asian countries.[55] While China remains the largest manufacturer of automobiles in the world, India surpassed Japan in automobile sales and moved to third place globally in 2022.[56] In China, the electroplating industry was already well developed by 2005, with >15 000 factories and 5000 production lines.[57] PFOS has long been used in electroplating in China in mist suppressants to protect factory workers from exposure to Cr(VI) during the electrolytic process.[32] In 2016, PFOS use in the electroplating industry in China was approximately 17 t per year.

China started producing PFOS in the late 1980s. After the listing of PFOS under the Stockholm Convention in 2009, China remained the only known manufacturer of POSF-based chemistries. Production peaked at around 250 t per year in 2006, carried out by some 15 different enterprises mostly located in the Hubei and Fujian provinces.[32] The production volume in China sharply declined to about 100–150 t per year in 2012.[6] In 2015, China produced around 100 tons of POSF annually and supplied it to other downstream producers to produce PFOS-related substances.[58]

Supported by the World Bank, China initiated a series of actions under the framework of the "China – Reduction and Phaseout of PFOS in Priority Sectors" project.[59] According to the Implementation Status and Results Report of this project, by the end of 2020, there was zero use of PFOS in metal-plating (previously ~17 t per year) as well as in sulfluramid formulations for pest control (previously ~1 t per year) and in chemically driven enhanced oil recovery in oil production (previously ~26 t per year).

An important fluorinated mist suppressant that is uniquely used in China is 6:2 Cl-PFESA (F-53B). F-53B has been produced in China since 1975 for local use even before the introduction of PFOS.[39,40] According to a survey of about 4000 chromium plating plants in China, 58% used F-53B suppressants in 2013.[60] While examining the fate of F-53B, Liu *et al.*[61] observed that the electroplating industry was the major source of F-53B in the environment. Based on data from Ti *et al.*,[60] up to 86% of F-53B discharged from manufacturing plants in China ended up in inland waters, and up to 30% in oceans. Oceanic transport of PFAS from China is also evident, with F-53B detected in seawater from the Pacific and Indian Oceans[62] as well as in mammals from remote regions.[63] Typically, the concentrations of F-53B in Chinese freshwater are now higher than those of PFOS, and in the blood serum of people from China, F-53B is found at concentrations that are second to PFOS and PFOA.[64] With the elimination of PFOS use in metal plating, the use of F-53B is expected to increase.

4.3. PFAS use in firefighting

Asian countries are primarily focused on compliance with international agreements, such as the Stockholm Convention, without clear domestic policies targeting C_6-based AFFFs. However, some Asian countries, namely Japan, China, South Korea and Singapore, are taking steps to phase out the production, import, and use of AFFF and any firefighting foams that contain PFAS listed under the Stockholm Convention.[65,66] In fact, AFFF products are still readily available for purchase in most Asian countries. In China, the use of PFOS in foam formulations for firefighting was approximately 61 t per year in 2016,[58] and foam-based extinguishing agents with PFOS still accounted for about 35% of the local market in 2022.[67] Notifications regarding AFFF use for firefighting from Cambodia, China and Vietnam were received in the recent past by the Stockholm Convention.[68] In the West, market supply of AFFF products containing C_6 fluorotelomers appears to be decreasing due to increased regulatory pressures. No clear domestic policies targeting C_6-based foams are currently available in Asia.

4.4. PFAS use in the textile industry

In the textile industry, PFAS are used to impart water-, stain- and oil-resistant properties to garments.[69,70] Asia dominates textile manufacture globally, with 55% of global exports coming from East Asia.[71] Unlike automobile manufacturing which is dominated by a few Asian countries (China, Japan, Korea, and India

accounting for about 50%), textile manufacturing is widespread and also significant in countries like Bangladesh, Cambodia, Vietnam and Indonesia. PFAS, including PFOA, PFDA (perfluorodecanoic acid) and fluorotelomer alcohols (FTOHs), were detected in various environmental matrices (wastewater, air and dust particles) at a typical textile manufacturing plant located in the Yangtze River delta in China.[72] PFAS in textile effluent suggested that long-chain PFAS were preferably used in China at that time. Due to some restrictions imposed by China in 2017,[35] textile treatment agents containing PFAS may have been shifting from C_8- to C_6-based products. In a study on the long-term trend (2013–2021) of FTOHs in five municipal wastewater treatment plants, impacted by the textile industry in Wuxi city of China, Ma *et al.*[73] noted a significant correlation between FTOH concentrations in water and the output of textile products from the city. They noted a shift in the FTOH homologue profile in influent samples from 8:2 FTOH to 6:2 FTOH in 2020–2021.

In Bangladesh, another country with a large textile sector, PFAS were detected in air and surface water samples collected from nine locations. The surface water samples from locations in the vicinity of the textile industry (*e.g.* Cannel Savar Lake) had higher concentrations of Σ_{16}PFAS than others.[74] While 6:2 FTOH dominated the 16 PFAS detected in both air and water samples, a range of long-chain and short-chain PFAS, including PFOA and PFOS were detected. Given its volatility and reactivity, the detection of 6:2 FTOH in water may be indicative of a large or ongoing release. Bangladesh ratified the PFOS and PFOA listings under the Stockholm Convention only in 2023.

4.5. PFAS used in lithium-ion batteries

Ionic liquids are attractive for applications in metal plating, batteries and capacitors, lubricants and many (50 known) other applications. Guelfo *et al.*[75] recently demonstrated that bis-perfluoroalkyl sulfonimides (bis-FASIs) are associated with proximity to production facilities for lithium (Li)-ion batteries in the USA and Europe. Asia dominates global Li-ion battery production, accounting for approximately 81.1% of the world's output in 2022, with China, Japan, and South Korea leading the industry and China by a large margin which is growing.[76] China is projected to maintain this dominance, with an estimated 67% of global production capacity by 2030. This indicates the need for a better understanding of PFAS occurrence in Asia associated with Li-ion battery production and possibly areas of recycling or disposal. PFAS-free alternatives for Li-ion batteries would need to be developed.[77]

Importantly, the use of fluorine in Li-ion batteries reveals why one must be cautious when using extractable organofluorine (EOF) to characterise the potential presence of PFAS. A study by Jiao *et al.*[78] analysed tap water from Shanghai, revealing ultra-short PFAS as dominant contributors to ΣPFAS. However, over 40% of the EOF was accounted for by the inorganic anions BF_4^- and PF_6^-. BF_4^- and PF_6^- are also used as electrolytes in Li-ion batteries to improve the stability and performance of the electrolyte. This caution should also be applied in studies globally.

5. Pathways of human exposure

Given the heavy use and local production of PFAS in some Asian countries, elevated exposure to these contaminants likely occurs in populations working in or living near contamination hotspots *e.g.*, fluorochemical industrial parks.[79] For example, Heydebreck *et al.*[72] noted that workers' exposure to FTOHs at a textile manufacturing plant was up to five orders of magnitude higher than background exposure to the general population. In addition to exposure levels, the types of PFAS involved will also depend on legacy and current industrial practices in Asia, including PFAS uniquely used in Asia. In the following sections, reported levels of PFAS in Asian water, food and dust are summarised as important pathways of human exposure.

5.1. PFAS in Asian waters

5.1.1. PFAS in freshwater resources of Asia. Understanding the level of PFAS contamination in freshwaters is important as these can be a source of drinking water and a pathway for food contamination (including seafood, eggs, dairy, plant crops and meat products). The largest data set available in Asia on PFAS is for the freshwater resources. According to the Scopus database, 20 out of 48 countries have so far published studies on PFAS occurrence in freshwater. Most studies (about 90%) over the past decade have originated from China, Japan, and South Korea, with only a limited number from other countries in East Asia, Southeast Asia, and the Middle East. However, as reviewed by Baluyot *et al.*[80] and Tang *et al.*,[81] these include surface water and drinking water in the Philippines and Thailand, river water and ground water in Vietnam, surface water in Malaysia and reservoir water in Singapore, river water and groundwater in India.

Grunfeld *et al.* (2024) compiled data from 45 000 water samples worldwide to map PFAS occurrences and concentrations.[82] We extracted the data originating from Asia ($n = 3008$ samples) from this set and categorized them based on PFAS concentrations (Fig. 6). Notably, 74% of the samples ($n = 2209$) had PFAS concentrations below 10 ng L^{-1}. The highest PFAS concentrations were found in samples from contaminated sites, such as those impacted by AFFF used in firefighting.

The temporal trends in the concentrations of legacy PFAS (PFOA, PFOS, and PFHxS) in freshwater resources of various Asian countries are presented in Fig. 7. Despite data variability, a statistically significant increase over time was observed for all three compounds in China. In contrast, PFOA in Japan showed a declining trend during this period.

5.1.2. Impact of PFAS production on surface waters. Surface waters are often the major sources of drinking water in Asia. For example, in a nationwide centralized drinking water survey in China,[83] 71% of 1093 drinking water samples were sourced from lakes, reservoirs and rivers, and the 28.6% from groundwaters. Obviously, the production and various applications of PFAS, as outlined in Section 3 above, contribute to the contamination of freshwater resources. In China, manufacturing and other industrial sources have been found to be the key contributors. For example, Liu *et al.*[84] estimated the

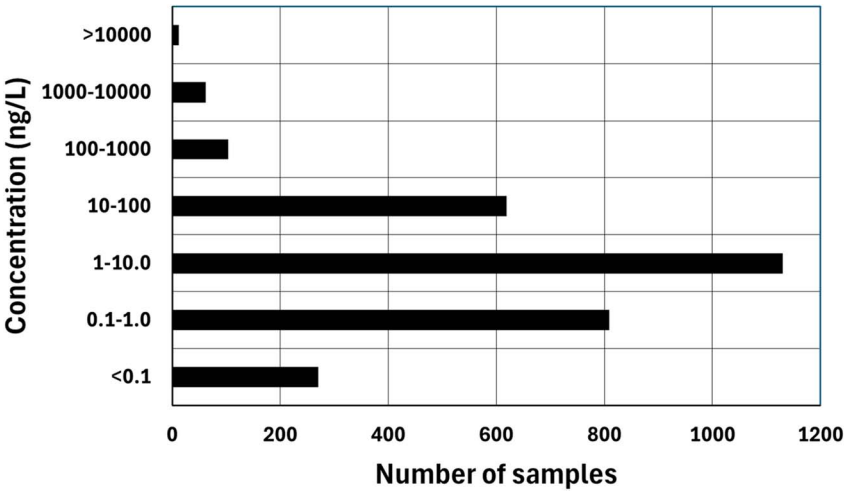

Fig. 6 Distribution of PFAS concentrations (sum of 20 PFAS subject to EU guidance) in Asian water samples (surface water, groundwater and drinking water), based on data compiled by Grunfeld *et al.* 2024.[82]

emissions of PFOS and PFOA from different sources in Central and Eastern China. Their findings indicated that industrial sources were the primary contributors, accounting for 86–87% of the annual emissions of PFOS (70 tonnes per year) and PFOA (40 tonnes per year) during that period. Consistent with this, the highest level of ∑PFAAs (1.86 mg L^{-1}) and PFOA (1.71 mg L^{-1}) were detected in surface water of the Xiaoqing River,[85] receiving wastewater from the fluorochemical industry. The fluorochemical industrial park in this region has been discharging PFAS into the river through the Dongzhulong tributary, which merges with the Xiaoqing River approximately 15 kilometres downstream from the park. Pan *et al.*[86] reported that the ∑PFAS concentration (C_4–C_{14} PFCAs, C_6 and C_8 PFSAs, Cl-PFESAs, HFPO-DA and HFPO-TA) peaked at 282 μg L^{-1} in this tributary in 2015. Downstream of the industrial park, ∑PFAS concentrations were 800-fold higher than upstream. PFOA dominated the ∑PFAS concentrations (197 μg L^{-1}) followed by HFTO-TA, an alternative compound. Similarly, Feng *et al.*[87] found a high ∑PFAS concentration (780 μg L^{-1}) at approximately the same location in 2019. HFPO-TA and HFPO-DA together accounted for ~20% of the ∑PFAS concentrations after the confluence point and were dominated by HFPO-TA (>90%) in that case.

The emission pattern seems to be consistent with the PFAS use profiles of different regions in China. The spatial and temporal variations in annual riverine mass discharge of PFOA in China, during 2003–2020, are shown in Fig. 8. Pan *et al.*[44] estimated the total riverine PFAS load and found that nearly 80% of the total load in 2016 originated from three rivers, namely Yangtze (~67%), Xiaoqing (~23%) and the Pearl River (~9%) in China. The fluoropolymer industry-impacted Xiaoqing River system contributed 73% of PFOA and 22% of HFPO-TA to the total PFAS load.[44] In comparison, the contribution of the Pearl River to the total PFAS emission in this study was in the following order: PFOA (23%) > PFOS (18%) > HFPO-DA (17%) > HFPO-TA (13%). While the fluoropolymer industry is the primary source of PFAS pollution in the Xiaoqing River, the Pearl River (South China) receives most of the load from PFAS-containing products. Wang *et al.*[88] compared the riverine discharges of PFAS among 13 rivers (including Xiaoqing and Yellow rivers running in parallel) in the Shandong province, a key location for the fluoropolymer industries (7 known facilities) in China. They found that of the 13 rivers, Xiaoqing River (26.5 t) and Yellow River (0.8 t) contributed ~95 and ~3% of the total PFAS mass discharge to the surrounding Bohai and Yellow seas.

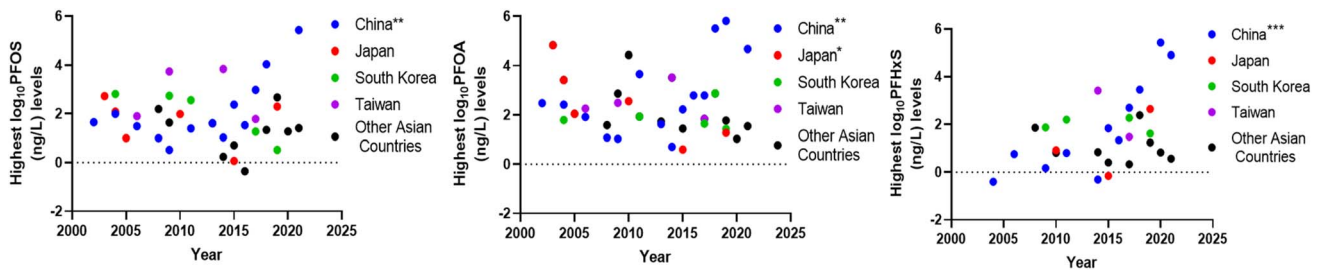

Fig. 7 Legacy PFAS in freshwater resources of various Asian countries. Freshwater resources include surface water, tap water, groundwater and drinking water. Asterisk (*) indicates a significant change in PFOS, PFOA, and PFHxS levels with time. Spearman correlation and regression analysis was undertaken using Graphpad Prism 8.0.

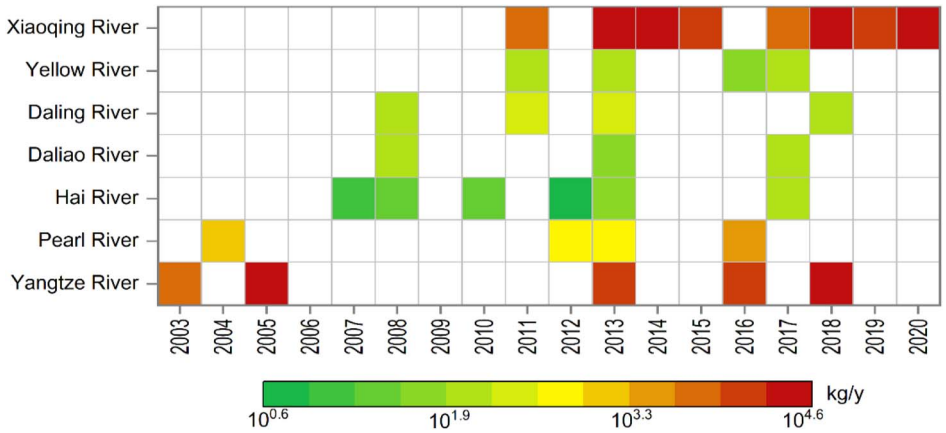

Fig. 8 PFOA annual riverine mass discharge in China (kg per year). The selected rivers are located in the main PFAS emission regions of China. Spaces without colours indicate data that are not available. For data sources and details see Table S6 in SI.

In addition to China, contamination of rivers with legacy PFAS has been reported in other Asian countries, *e.g.* Malaysia (Langat River, 43.5 μg L^{-1}), Japan (Samondogawa River, 16.2 μg L^{-1}), Taiwan (Nanmen River, 6.6 μg L^{-1}), South Korea (Daegu River, 1.1 μg L^{-1}), and India (Ganges river, 9.4 ng L^{-1}).[89–92] In India, Koulini *et al.*[93] reported the detection of both short and long-chain PFAS in surface water samples collected from a range of water bodies from different cities in India, at concentrations ranging from <0.04 ng L^{-1} to 93 ng L^{-1}. The highest levels of PFOS (36 ng L^{-1}), PFOA (180 ng L^{-1}) and PFBA (200 ng L^{-1}) were found in the Meenachi River in India.[94] In many instances, only a limited selection of PFAS (*e.g.* only PFOS and PFOA) were analysed, which may result in an underestimation of the total load in river systems. Direct comparisons among studies are also challenging due to varying analysis methods and study periods. Nevertheless, they show that PFAS contamination is present in freshwaters across several Asian countries.

5.1.3. PFAS alternatives in surface waters. Due to stricter regulations, short chain homologues such as PFBS and functionalized perfluoropolyethers (*e.g.*, PFECAs and PFESAs) have increasingly been adopted as replacements for long-chain PFAS. Certain PFAS produced are unique to China or are produced and utilized more extensively there than in Western nations, influencing human exposure to specific PFAS types. The replacement products for PFOS and PFOA in Asia differ from those in Europe and North America, with distinct usage patterns, such as F-53B suppressants employed by the electroplating industry in China. In China, the trimer acid of HFPO (HFPO-TA, trade name T-5) is also used as a substitute for PFNA in the production of PVDF.[95] GenX is utilized in fluoropolymer production as an alternative to PFOA.

As mentioned earlier, the main component of the PFOS alternative compounds F-53B (6:2 Cl-PFESA) is now ubiquitous in Chinese population. There is a strong association between the presence of this compound in river water and human exposure. In a survey of five major rivers in China,[44] Cl-PFESAs in these rivers were found at concentrations comparable to those of PFOS. In this instance, 85% of the total riverine ∑PFESA discharge (4.3 tons per year) was attributed to 6:2 Cl-PFESA, which was comparable to the discharge of PFOS (4.6 tons per year). HFPO-TA and HFPO-DA have also been detected in 83–100% of water samples from the five rivers.[44] The oligomers of HFPO, including HFPO-DA and HFPO-TA were detected downstream from a major fluorochemical industrial park in China.[86,95] Feng *et al.*[87] estimated that from this industrial park in 2021, annual emissions of HFPO oligomers to water and air were up to 5040 kg and 1026 kg, respectively. Comparatively, PFOA emissions that year were 7560 kg to water and 1890 kg to air. They also confirmed the presence of HFPO-TA in various environmental compartments in this area (soil, dust, tree leaf, tree bark, and groundwater). HFPO-TA was also detected in the serum of residents living in the vicinity of the plant.[45] However, this chemical is less common in European and North American environments.[44,96] The trace concentrations of F-53B found in rivers of Western countries, could well be from products originating in China, as China is the only known source of F-53B.[44] Despite these few studies on occurrences of alternatives in the environment, there is limited understanding of their fate, transport characteristics and toxicity currently in the literature.

5.1.4. PFAS in Asian groundwaters. A substantial body of work on PFAS in groundwaters has started to emerge in Asia.[97] Groundwater contamination with PFAS has been noted in several Asian countries, including China, Japan, South Korea, Vietnam, Thailand and India.[93,98–102] In a South Korean study,[99] the groundwaters in the vicinity of industrial areas were found to be contaminated, especially with the shorter chain compounds (<C_8). The ∑PFAS concentrations ranged from non-detectable levels to 36.9 ng L^{-1} in this study. In Thailand, among seven target compounds, three PFAS (PFOA, PFOS, and PFHxS) were found at concentrations ranging from 1.68–42.0 ng L^{-1} in groundwaters at the sites impacted by industrial and municipal wastes.[100] Similarly, Duong *et al.*[102] analysed 16 PFCAs and PFSAs in 22 groundwater samples collected from four major cities in Vietnam. In this study, PFOS, PFOA and PFNA were the most common among the 11 PFAS detected, at maximum concentrations of 8.2, 4.5 and 0.45 ng L^{-1}, respectively. In a study of groundwater samples collected from the

Ganges River basin in India, Sharma *et al.*[101] reported frequent occurrence of 14 out of 21 PFAS analysed. PFBA and PFBS were detected at higher concentrations compared to PFOA and PFOS, with levels reaching up to 9.2 ng L^{-1} and 4.9 ng L^{-1}, respectively. PFHxA and PFHpA were present in all groundwater samples, with concentrations up to 4.9 ng L^{-1}, while other PFAS were below 2.2 ng L^{-1}. A separate study conducted in Chennai, India, identified both short and long-chain PFAS in groundwater samples, with significantly higher concentrations of PFBS (136.27 ng L^{-1}) and PFOA (77.61 ng L^{-1}).[93] In India, groundwater is commonly utilised as a source of drinking water.

Most studies on PFAS contamination in groundwater come from China. Calore *et al.*[96] reviewed 24 studies, about half of which reported PFAS in Chinese groundwaters. These studies often found the highest PFAS concentrations near manufacturing facilities. PFAS detected in groundwaters also included PFAS alternatives, such as Cl-PFESAs, HFPOs, PFECHS (perfluoroethylcyclohexanesulfonate), and ADONA (4,8-dioxa-3H-perfluorononanoic acid).[8,34,103,104] High levels of C_4 to C_8 PFAAs (reaching mg L^{-1}) were found by Jia *et al.*[34] inside two POSF manufacturing sites. PFAS have also been reported in groundwater in regions far from major production areas, where primarily short-chain PFAS are present.[105] PFAS such as F-53B, Cl-PFESAs, 6:2 FTS, ADONA, and HFPOs have also been reported at non-industrialized locations in Jiangxi province and the Loess Plateau of China.[104,106,107] Overall, groundwater contamination by PFAS is a significant concern in China, particularly due to the country's reliance on groundwater for drinking purposes.[105]

5.1.5. PFAS in municipal drinking water supplies in Asia. Considerable data are available on PFAS levels in drinking waters and their sources in Asia (Fig. 9). For instance, 12 PFAS were detected in bottled drinking water tested in the Philippines and Thailand, with total levels ranging from 7.2 to 59 ng L^{-1}. This compares to levels of 16 to 67 ng L^{-1} found in source waters such as rivers, lakes, and dams, where 15 compounds were identified.[108] Long-chain PFAAs were present in all water samples tested in this case. Additionally, PFOS alternatives or precursors, including 6:2 FTS, FBSA (perfluorobutanesulfonamide), and *N*-MeFOSAA (*N*-methylperfluorooctanesulfonamidoacetic acid), were detected in some samples. Given the absence of PFAS production in these two countries, consumer products are considered the potential sources of contamination.

The local populations in certain regions of China, particularly near contamination hotspots like fluorochemical industrial parks, are exposed to higher concentrations of PFAS compared to other Asian countries.[41] In a comprehensive survey of PFAS in drinking water in China, Liu *et al.*[41] examined data on PFAS concentrations in drinking water supplies for 452 million residents across 66 cities, finding $\sum_{18}$PFAS levels ranging from 0.1 to 502.9 ng L^{-1}. Elevated concentrations were observed in East, South, and Southwest China, regions known for high industrial activity and population density. Previously, Fang *et al.*[109] reported notably high levels of PFOA in drinking water near a fluorochemical plant, with PFOA, PFOS, and PFBA being the most prevalent compounds. Emerging PFAS such as F-53B and HFPO-TA have also been detected in Chinese drinking water.[110] Since 2010, short-chain PFAS concentrations have shown an increase there.[41]

5.2. PFAS in Asian food

PFAS are known to bioaccumulate in both plants and animals, presenting pathways for human exposure *via* food. Dietary intake (particularly fish, seafood, meat, eggs and dairy) is often a major exposure pathway for several long-chain PFAS in humans.[111,112] A considerable body of data on levels of PFAS in food products is available from some Asian countries.[110–112] However, once again the majority of this literature is from a small number of countries in East Asia, whereas very little data are available from South Asia and the Middle East. Given PFAS production and high local use of PFAS in some Asian countries, elevated exposure is likely for populations living near contamination hotspots (*e.g.* fluorochemical industrial parks), as indicated by the serum biomonitoring discussed above.

Dietary preferences and food habits are diverse in Asia and may vary from region to region within a country. For example, diets may be rich in seafood in some areas, while a vegetarian diet may be common in others. This influences not only total exposure but also the types of PFAS to which individuals are exposed. Short-chain PFAS are known to preferentially accumulate in plants, whereas long-chain PFAS exhibit greater accumulation in animals. The occurrence of PFAS in various food commodities in Asia is discussed below.

5.2.1. PFAS in fish and seafood. Fish and seafood are a significant part of the Asian diet. Nearly two-thirds of global fish/seafood consumption occurs in Asia,[113] with much of it sourced locally. Given the concentrations observed in environmental waters across Asia, human exposure is expected in various Asian regions, especially in areas involved in PFAS manufacturing and other industrial activities.

Our review revealed that most data on fish and seafood contamination (~90%) once again originate from China, Japan, and Korea. Studies have detected various PFAS in fish and seafood from East Asia, sometimes at higher levels than those in other regions. For example, a study on edible Pacific cod (*Gadus macrocephalus*) muscle samples sourced from various coastal waters of the North Pacific Ocean (USA, Canada, Russia, Japan, Korea) found mean concentrations of C_8–C_{14} PFCAs ranging from 0.22 to 1.71 ng per g (wet weight), varying by location.[114] In this study, the concentrations of the long chain PFCAs (C_{11}–C_{13}) in samples from Japanese and Korean waters were found to be two to four times higher than those from the USA, Canada, and Russia. A study, on six seafood fish species, sourced from the Eastern Red Sea (Saudi Arabia), detected 16 PFAS (PFOS primarily) in liver and muscle tissues, with mean concentrations ranging from 3.9–59 ng per g (dry weight).[115] In some cases, the combined levels of four PFAS (PFOS, PFOA, PFNA and PFHxS) exceeded the European Food Safety Authority's 2020 tolerable weekly intake (4.4×10^{-3} ng g^{-1} of body weight). PFAS have been detected in various types of seafood, depending on the location and species. The concentrations of $\sum_{15}$PFAS in seven edible species of shrimps from Japan and

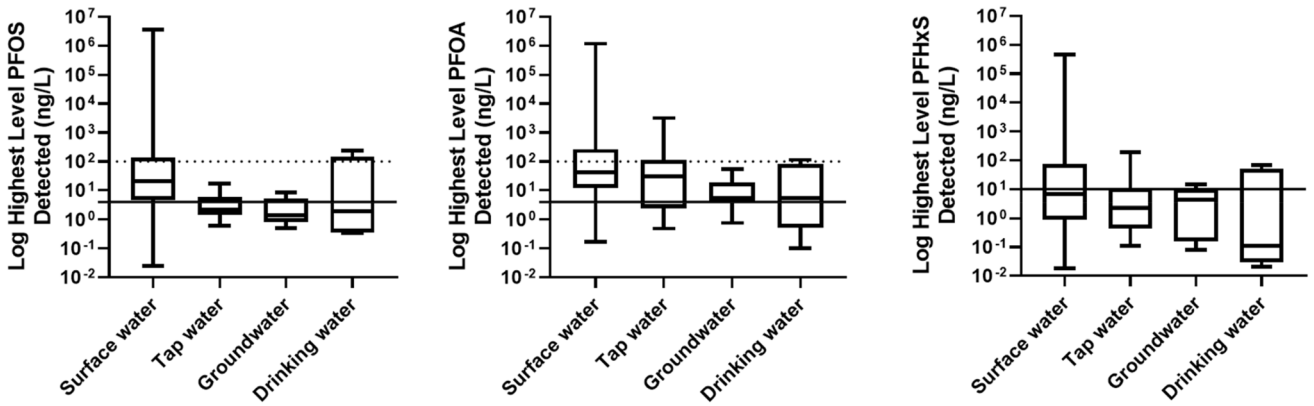

Fig. 9 The highest levels of PFOS, PFOA, and PFHxS detected in drinking waters and their sources in Asia. The data were procured from various studies published during 2000–2024. The dotted lines indicate the health advisory levels for PFOS and PFOA in drinking water (100 ng L^{-1}) set by the World Health Organization. The solid lines indicate the maximum concentration levels set by USEPA.

neighbouring countries ranged from 1.5 to 10 ng per g (wet weight), varying by location and species.[11] PFOS and PFUnDA (perfluoroundecanoic acid) were identified as the main contributors to the total PFAS load in this case.

In China, Liu *et al.*[116] reported that fish livers from Tangxun Lake (Hubei province, China) had higher levels of total PFSAs (50–950 ng g^{-1}) compared to those from the Yangtze River (7–25 ng g^{-1}), with PFOS accounting for 93–97% of the total PFAS measured. These high concentrations in fish appear to be reflected in data on PFAS levels in human serum. For example, Zhou *et al.*[21] reported exceptionally high serum concentrations of C_4 to C_{12} PFAS in fishery employees from Tangxun Lake (median PFOS 10,400 ng mL^{-1}; $n = 39$). Based on a comparison of different exposure pathways, the authors concluded that the contaminated fish from Tangxun Lake was the primary source of PFAS exposure for these employees. The estimated daily intake of PFAS through fish consumption was found to be several orders of magnitude higher than that for the population exposed to background levels.[117] Similarly, Shi *et al.*[118] reported higher PFOS serum concentrations in high fish consumers than in electroplating workers.

5.2.1.1 Emerging PFAS in fish and seafood. Among emerging PFAS unique to China, F-53B and its homologues (8:2 and 10:2 Cl-PFESA) were detected in fish samples collected from Tangxun Lake and from the Yangtze River receiving effluents from a manufacturing facility.[116] In this study, four classes of PFAS were found in fish for the first time, including two classes of PFSAs (diether- and cyclic ether-PFSAs), that are perhaps manufactured intentionally as PFOS alternatives in China. In another study,[86] HFPO-TA was found to bioaccumulate in common carp (*Cyprinus carpio*) in an area located near a fluoropolymer production facility in Huantai County, China. However, the concentrations of HFPO-TA in human serum were low (median 2.93 ng mL^{-1}), which may be due to either lower concentrations in the environment, or due to the differences in the bioaccumulation potential of various PFAS. Elevated concentrations of emerging PFAS have also been reported in seafood from coastal waters of China. Pan *et al.*[119] found that marine organisms collected from the Beibu Gulf in the South China Sea were contaminated with a range of PFAAs, including PFOS and F-53B, both of which were found to be highly accumulative with bioaccumulation factors >5000. Data on PFAS exposure in Asian countries outside China, Japan, and Korea are limited. However, nations without a PFAS industry likely face exposure through contaminated fish and seafood from industrial activities like electroplating and textile manufacturing.

5.2.2. PFAS in eggs and vegetables. Among other sources, eggs are an important part of dietary intake in Asia. Eggs are known to accumulate PFAS, depending on the feed and feeding environment of poultry. For example, Gao *et al.*[120,121] found that free-range chicken near an electrochemical fluorination plant in China had higher $\sum$PFAS concentrations than farm chicken, consistent with higher PFAS concentrations found in dust, water and feed in the former case. The dominance of PFBA and PFOS in the tissues and organs of chicken were found to be consistent with the production facility, indicating the manufacturing as a direct source of exposure. $\sum$PFAS concentrations in egg yolks were greater than those in the tissues and organs in this case. In other studies, PFAS in homegrown eggs from PFAS manufacturing areas have been found to be elevated, (up to 368 ng per g PFOA) and more contaminated than commercially produced eggs.[120,121]

Like eggs, vegetables produced in the industrial regions can be contaminated with PFAS, especially the short-chain PFAAs, as shown by a Japanese study on field grown vegetables.[122] An association between vegetable consumption and serum concentrations of PFDA was noted in a recent study on Korean adolescents.[123] Zhang *et al.*[124] noted significant geographical variations in the dietary exposure to PFAS between the regions of China that were manufacturing PFAS and those that were not. They found the estimated dietary intakes (EDIs) to be much higher (by two orders of magnitude) in the PFAS manufacturing area (Hubei province) than those in the urbanized coastal areas (Zhejiang province). In this study, the consumption of vegetables was found to be an important exposure pathway to PFAS. Liu *et al.*[85] found very high ΣPFAS levels in agricultural soils (8.28–84.3 ng g^{-1}). About 70% of this contamination originated from fluorochemical manufacturing and metal processing

industries, with PFOS being the dominant compound (41.1%). The sum of PFAS concentrations in vegetables from these areas ranged from 163 to 7176 ng g^{-1}, which is 1–5 orders of magnitude higher than levels found in global market-purchased vegetables. Short-chain PFAS (C_4–C_7, particularly PFBA) were present in higher concentrations in the vegetables, despite PFOS being the dominant contaminant in the soils. This study also detected novel compounds such as Cl-PFESAs, HPFO-DA, HPFO-TA, and ADONA in various vegetables, with HFPO-TA displaying the highest levels in both soils (maximum 5.30 ng g^{-1}) and vegetables (maximum 32.8 ng g^{-1}).

5.3. PFAS in indoor dust and particulate matter

Inhalation of indoor dust and total suspended particles can be a significant pathway of exposure to PFAS, both in occupational workers and the general population.[125] A range of PFAS has been detected in indoor dust samples and particulate matter in air from several Asian countries (Table 1), including China, Japan, Korea, India, Thailand and Vietnam.[126–129] The largest dataset is from China. The reports from China, Japan and Thailand (Table 1) showed higher median concentrations of total PFAS and total PFCA (118–230 ng g^{-1}) in indoor dust, as compared to those in Korea and Vietnam (24–42 ng g^{-1}). Tan *et al.*[128] carried out a major study on PFAS in house dust collected from six different regions covering four continents (Asia, Australia, North and South America). PFAS were detected in all indoor dust samples ($n = 153$), with total median concentrations ranging from 17.3 to 197 ng g^{-1} (Table 1). Indoor dust samples from offices in China were found to contain similar concentrations to those in homes.[128] Among the legacy PFAS, PFOS and PFOA were most common (with 100% detection frequency in Tianjin) and dominated the total load (Table 1). In China, PFOA and PFOS are the main PFAS in particulate matter, while short-chain PFAS dominated in India, Japan, and South Korea.[126] A geographical comparison of PFAS levels in dust shows similar median or maximum concentrations between Asian countries and Australia or the US,[128] except for lower levels in Thailand and Vietnam (Table 1).

Fewer studies have included precursors and emerging PFAS in their analysis.[128,130] However, a range of PFAS, such as 6:2 diPAP (polyfluoroalkyl phosphate diester), 6:2 Cl-PFESA, and HFPO-DA, have been frequently detected in indoor dust and particulate matter in China[126,131] and other Asian countries.[128] At three Asian sites (Tianjin, Hong Kong and Hanoi) the detection frequencies for 6:2 diPAP (used in paints, cleaning products, and food packaging) and HFPO-DA ranged from 86–100%, whereas those for 6:2 Cl-PFESA ranged from 38–100%.[128] Geographically, the Asian samples had higher concentrations of 6:2 Cl-PFESA and HFPO-DA than those from the US and Australia, but the converse was true for 6:2 and 8:2 diPAPs.[128] This is not surprising, as 6:2 Cl-PFESA and HFPO-DA are used in metal plating and fluoropolymer manufacturing in China and not used in Australia. HFPO-DA is used in the US to some extent. Lifestyle differences may explain geographical variations. The use of carpet strongly predicts PFCA and PFSA levels, as well as total PFAS in house dust ($p \leq 0.05$).[128] Carpets are uncommon in Asian homes, especially in tropical areas with high humidity, where they are less appealing.

6. Asia-specific influences on exposures through drinking water and dietary sources

Human exposure in Asia is quite variable depending on PFAS manufacturing, industrial use, food habits, drinking water sources, lifestyle, cultural practices, *etc.*[126–129] Indeed, there are major differences in reliance on certain foods. For example,

Table 1 Legacy PFAS in indoor dust samples and air particulate matter in Asian countries[a]

			Min–max (median)			
Country	City/year	n	PFOS	PFOA	∑PFAS	Data source
China	Tianjin/2024	36	4.08–297 (20.3)	5.14–410 (32.5)	33.9–1511 (118)	Tan *et al.* 2024 (ref. 128)
China	Hong Kong/2024	25	n.d.–108 (14.1)	n.d.–55.7 (11.2)	21.5–478 (67.1)	Tan *et al.* 2024 (ref. 128)
China*	All provinces (Urban)/2016	58	n.d.–14.0 (1.1–2.7)	<0.04–160 (0.48–1.2)	7.9–177 (5.9–20.0)	Yao *et al.* 2016 (ref. 131)
China	Guangzhou (Offices)/2024	41	8.7–24.4 (11.2)	0.02–8.42 (3.48)	52.6–219 (107)	Cheng *et al.* 2024 (ref. 132)
Japan	Kyoto/2015	5	2.95–21.1 (3.91)	n.d.–111 (25.5)	118–355 (230)[b] 4.0–124 (7.1)[c]	Eriksson, Kärrman 2015 (ref. 130)
Korea	Nine cities/2016	15	0.7–52.1 (11.4)	0.6–10.7 (4.5)	29.9–97.6 (42.0)	Tian *et al.* 2016 (ref. 133)
Thailand	Bangkok; Nakhon/2011	20	3.0–130 (16.0)	1.0–290 (18.0)	54.0–1314 (161)	Goosey & Harrad 2011 (ref. 134)
Vietnam	Hanoi/2024	21	n.d.–23.0 (4.91)	n.d.–12.8 (2.64)	4.0–147 (24.1)	Tan *et al.* 2024 (ref. 128)
Australia	Adelaide/2024	21	4.43–168 (19.9)	6.12–94.0 (17.7)	66.7–1999 (197)	Tan *et al.* 2024 (ref. 128)
Canada	Ottawa/2015	10	1.69–699 (7.29)	n.d.–310 (21.0)	9.3–1197 (112)[b] 2.2–1182 (21)[c]	Eriksson, Kärrman 2015 (ref. 130)
Sweden	Three cities/2015	10	n.d.–9.67 (1.96)	n.d.–49.5 (14.4)	n.d.–196 (53)[b] n.d.–15.8 (5.8)[c]	Eriksson, Kärrman 2015 (ref. 130)
Spain	Catalonia/2015	10	2.12–7.16 (5.29)	n.d.–39.7 (8.81)	1.4–315 (26)[b] 2.12–12.1 (6.7)[c]	Eriksson, Kärrman 2015 (ref. 130)
USA	Carbondale/2024	17	0.77–72.7 (6.28)	1.10–64.8 (4.94)	33.1–1624 (91.0)	Tan *et al.* 2024 (ref. 128)

[a] n.a. – not reported; n.d. – not detected. [b] ∑PFCA. [c] ∑PFSA.

seafood is abundant in southeast Asian diets[135] whereas a large fraction of the Indian population (47%) following vegetarian or vegan diets.[136] Furthermore, differences in drinking water sources (*e.g.* untreated groundwater and surface water in parts of Asia) are also likely to result in higher or lower exposures in Asia as compared to high-income Western countries.

6.1. Self-supply of drinking water

As in other parts of the world, drinking water in Asia can be sourced from either centralised municipal supplies or self-sourcing. Due to an inadequate water supply by the utilities, self-supply plays a far greater role in some Asian countries than in high-income countries. Foster *et al.*[137] analysed 77 datasets from 26 countries to estimate the prevalence of self-supply of drinking water in low- and middle-income countries of South Asia, Southeast Asia and the Pacific. They found that ~32% of the population (~780 million people) relied on self-supply for drinking water in 2018. Of these, 85% were in South Asia (India, Bangladesh, Pakistan) and Indonesia.

In terms of sewer connectivity, the contrast between the lower-income (low and lower-middle) and high-income countries is stark. In high-income countries (*e.g.* Australia, USA, UK, Japan, and South Korea) 90–100% of the population is connected to sewers whereas a much smaller fraction of the population in lower-income countries is connected to sewerage systems. In India, more than half the population in its megacities (Kolkata, Chennai, Mumbai and Delhi) live in informal settlements.[93,138] In some Asian countries, partially treated or untreated wastewater is discharged into river systems, which in some cases serve as a source of drinking water downstream. For example, the River Yamuna in India receives partially treated or untreated wastewater and industrial effluents from the city of New Delhi.[139] In 900 Indian cities, treatment capacity was exceeded by 60–70% in 2013.[140] Furthermore, PFAS are not effectively removed during the wastewater treatment process anyway. For example, Wang *et al.*[141] collected wastewater from treatment plants in 17 Chinese cities and found 13 different classes of PFAS (including 49 emerging PFAS) which were only partially removed during the treatment.

Groundwater constitutes the most common source of self-supply in Asia.[137] More than 50% of the rural population in Cambodia, Vietnam, Myanmar, Timor-Leste, Lao PDR, and Indonesia utilise groundwater as a drinking resource.[142] Groundwater supplies a significant portion of drinking water in China too, although the reliance on groundwater is decreasing with time.[83] According to Zhang *et al.*,[83] the lakes and reservoirs in China account for about 40% of the centralized drinking water sources, whereas rivers and groundwaters account for around 30% each. However, this can vary regionally. For instance, North China, which contains 45.2% of the country's total population, possesses only 19.1% of the nation's water resources. Consequently, the population in this region relies extensively on groundwater.[143] Similarly, many water users in India either drill private borewells or purchase tanker water to supplement the water provided by local utilities. However, such sources are generally not subject to water quality control.

Groundwater resources may be particularly vulnerable to contamination by PFAS, due to their persistence and high mobility, especially of the shorter chain compounds ($<C_8$). This can be a problem in groundwater located in the vicinity of industrial areas. As discussed above, groundwater contamination with PFAS is widespread in Asian countries (Japan, South Korea, Vietnam, Thailand and India).[93,98–102] The maximum $\sum$PFAS concentrations in these reports have ranged from 36.9 ng L^{-1} in South Korea,[99] 42.0 ng L^{-1} in Thailand and 136.3 ng L^{-1} of PFBA alone in India.[93]

Similar reliance on groundwater may also be the case in some Western countries, notably the USA and Sweden, where considerable PFAS contamination of aquifers used for drinking water has occurred.[144,145] However, a difference may be the self-supply and higher frequency of drawing groundwater from small local wells in Asia, rather than centralised sources with treatment. Also, the dependence on groundwater in some parts of Asia is, in fact, an attempt to avoid surface water, which can suffer from significant contamination with pathogens.

6.2. Greater reliance on seafood as a protein source

Seafood consumption tends to vary across different parts of the world due to differences in availability and cultural practices.[146] Seafood, as a protein source, plays a vital role in Asian diets. High intake of fish and seafood (*e.g.*, shrimp) has been identified elsewhere,[147] and discussed earlier,[116] as an important contributor to PFAS exposure. Asia is an important producer of seafood. Bangladesh, China, India, Indonesia, the Philippines, Myanmar, Thailand, and Vietnam are among the ten largest aquaculture-producing developing countries.[148] Most (89%) of the farmed fish produced in these countries (except in Thailand and Vietnam) is consumed locally. Vietnam is among the top ten seafood exporters worldwide (US$ 8.97 billion in 2023). Per capita consumption of seafood in Asian countries is much higher than the global average, mainly due to their coastal geographical location and the rich supply.[149] While comparing the seafood consumption and the related nutritional–toxicological conflict in different regions, Sieon *et al.*[150] reported that most seafood is consumed (69.0 g per person per day) in the 'diet-cluster' that included Japan, Korea, Madagascar, and the Philippines, followed by the Nordic-Baltic countries (49.2 g per person per day) and South-East Asia (45.0 g per person per day). While pelagic marine fish were most commonly consumed in the first two clusters, freshwater fish were highly consumed in Southeast Asia. United Arab Emirates (UAE) residents also have relatively high seafood consumption rates (~71.0 g per person per year).[151] GDP may be linked to fish and seafood consumption. According to FAO,[152] annual fish consumption in China increased from about 5 kg per capita in 1980s to approximately 35 kg in 2010s, consistent with its increasing GDP. Fish protein intake in Asia has been steadily increasing over recent years, commensurate with increasing GDP and people's real income.

Carp, tilapia, and catfish are staple freshwater fish across many Asian populations. Large yellow croaker (*Pseudosciaena crocea*) is widely consumed in China. The orange-spotted grouper (*Epinephelus coioides*) and the kingfish (*Scombermorus*

commerson) are two highly consumed marine fish species in the UAE. Interestingly, the processing and eating methods in Asia differ from those in Western countries, and this may impact exposure to contaminants such as PFAS. For example, eating raw fish (sashimi or sushi) is a common traditional dietary practice in Japan and Korea but is less common in China and Vietnam. Fermented shrimp products (kimchi and saeujeot) are popular in Korea and other Asian countries. More data are needed on the potential PFAS contamination of popular freshwater fishes in Asia. Using whole seafood, such as boiling whole animals in soups and stews, which is common in some Asian cuisines, may also be a point of difference. Cooking practices in Western countries generally are dominated by muscle fillets for finfish, and claws or deheaded animals for crustaceans. Levanduski *et al.* found that in US finfish, PFAA concentrations were almost double (conversion factor of 1.86) in whole fish compared to fillets,[153] which suggests that PFAS exposures may be greater in populations that consume whole animals.

6.3. Sewage-fed fisheries and vegetable production

A striking feature unique to Asia is the long-standing practice of sewage-fed aquaculture. The East Kolkata Wetland (EKW) is the 'world's largest integrated wastewater fisheries' serving as a natural sewage treatment for the city of Kolkata.[154] The sewage-fed fisheries in the EKW are an important resource not only for West Bengal but also for neighbouring states in India. The EKW model of wastewater pisciculture is also used in several other countries (Bangladesh, Pakistan, Myanmar, and Thailand). Concerns about the presence of contaminants in fish tissues have been raised previously.[127] However, currently no study is available in the literature on PFAS or other organic contaminants in fish tissues from wastewater-based pisciculture in India or elsewhere.

Wastewater is also often used for irrigation in peri urban areas in Asia. This water is commonly untreated or only partially treated and thus it not only serves as a valuable source of irrigation water but also of nutrients. In India, wastewater originating from 36% of cities is used to irrigate agricultural land.[155] Organic contaminants such as antibiotics have been found to accumulate in roots and shoots of various crops irrigated with wastewater.[156] PFAS uptake in vegetables grown in industrially impacted biosolids and wastewater has also been reported.[157,158] However, the contribution of PFAS *via* vegetables and other plant foods grown in wastewater irrigated or biosolids-amended lands in Asia is not yet known.

6.4. Large vegetarian population

Unlike most high-income countries, a significant fraction of the population (approximately 10–40% in countries like India, Nepal, and Sri Lanka) follows a vegetarian diet, due to their religious beliefs (*e.g.* Hindus and Buddhist). India has the largest number of vegetarians and vegans, as up to 45% of India's population prefer meat-free diet, as compared to only 4% in the USA.[136] The total number of vegetarians in the world's two most populous countries (India and China) exceeds the entire US population. In a study conducted in China,[124] the consumption of vegetables was found to be an important PFAS exposure pathway. The consequence of this may be comparatively greater exposure to short chain PFFAs *versus* long chain PFAAs, due to the greater relative uptake of the former by plants.

It is noteworthy that unlike in Western Countries, fish and vegetables consumed in Asia are often procured from local sources rather than from a large market system sourcing produce from diverse sources, although supermarket share is growing in many Asian countries.[159] Large supply chains with centralised processing facilities (abattoirs, fish processing, *etc.*) and distribution networks including supermarkets chains can create a market dilution effect whereby individuals are far less likely to receive multiple PFAS-contaminated products than if they receive food regularly from a single area (that may have been impacted). In many parts of Asia, such market dilution is likely to be lower than that in Western nations, and this may increase exposure to PFAS in some areas.

7. Regulatory response to manage exposure pathways in Asia

The current state of regulatory response in Asian countries has been somewhat guided by the Stockholm Convention and is summarised in Table S7 (SI). The Stockholm Convention is a major instrument for controlling production and use of persistent organic pollutants. The Convention came into effect in 2004, and PFOS was listed under Annex B for restriction in 2009. PFOA and PFHxS were listed under Annex A for elimination in 2019. The response of Asian countries to the Stockholm Convention is highly variable. For example, by 2015 major producers in Japan had participated in the phase-out initiative targeting long-chain PFAS and their precursors. However, a detailed phase-out time plan was not available by that time in China, India, and Russia.[160] While Jordan banned PFOS *via* three relevant laws during 2005–2012,[161] Singapore restricted PFOS, PFOA, and PFHxS as a ratifier of the Stockholm Convention, and Vietnam implemented some policy measures on PFOS, but not on other PFAS. Malaysia is not yet a party to the Stockholm Convention, while some member states, such as India and Bangladesh, have not ratified the PFAS listings and have essentially provided no response to restrict their use and emissions.

7.1. Regulatory impact on production of PFAS in China

China, being a signatory to the Stockholm Convention, responded to it by significantly reducing the production of PFOS after its listing in Annex B by the Convention in 2009. For example, in one of the largest production plants located in the Hubei province of China, PFOS production declined post 2009 and was replaced by PFHxS production. According to Fu *et al.*,[20] PFOS production decreased from 30 tonnes in 2009 down to 10 tonnes in 2011. In contrast, the production of PFHxS increased from 10 tonnes to 30 tonnes during the same period. Nevertheless, due to market forces the production of PFOS in the same plant increased to 65 tonnes in 2012 and the production of PFHxS ceased. China ratified the PFOS listing in 2013 and subsequently, PFOS was included in China's Strictly Restricted

Toxic Chemical List in 2018.[35] The Chinese government banned the import and export of PFOS substances in 2023.[162]

In the case of PFOA, restrictions in China were implemented later. In 2011, PFOA-related technology and products came under special scrutiny in China,[160] and some restrictions were placed on the installation of new PFOA and POSF production facilities, and the elimination of paints containing POSF-based substances was flagged. In 2013, China recognised PFOA-based fluoropolymers as high-risk products and placed them in the comprehensive Catalogue for Environmental Protection. PFOA and related compounds were included into the list of key controlled new pollutants in 2023.[162] Furthermore, the environmental impact assessment and pollutant discharge permits for new, renovated, and expanded factories involving PFOS and PFOA are no longer approved.

7.2. Regulatory response in Asia to manage environmental exposure

Beyond controlling production, environmental regulations are needed to ensure effective management of legacy contamination, and to control exposures from ongoing releases during production, use and following disposal. For PFAS, such regulation in Asia is lagging behind that of the West, as reported by IPEN in a paper on PFAS pollution in 12 Asian and Middle Eastern countries.[161] No guidelines for natural and drinking water have been established in most countries at this stage.[41,161] The Standards for Drinking Water Quality in China (GB5749-2022 standard) established for PFAS are limited to PFOS and PFOA, with limits of 40 and 80 ng L^{-1}, respectively. Japan has set provisional drinking water quality guidelines for PFOS and PFOA at 50 ng L^{-1} each. South Korea has established maximum contaminant levels for PFOS and PFOA in drinking water at 70 ng L^{-1}. Taiwan has set a limit of 50 ng L^{-1} for PFOA plus PFOS and 70 ng L^{-1} for PFOS plus PFHxS.[163] Singapore does not yet have specific regulatory limits for PFAS in drinking water, but the country follows international guidelines set by the World Health Organization (WHO). There are currently no specific regulations for PFAS in drinking water in India. This is likely to be common across several other countries of Asia, such as in the Middle East.[161] Drinking water guideline values (DWGVs) in the tens of ng L^{-1} in Asia are similar to Australia's DWGVs for PFOS + PFHxS (70 ng L^{-1}) and higher than the US Maximum Contaminant Level for PFOS and PFOA (4 ng L^{-1}).[164] Some Western nations focus on limiting DWGVs to PFOS and PFOA, while the US has DWGVs for 6 PFAS. European nations may include up to 21 PFAS, and Canada includes a DWGV for the sum of measurable PFAS. Another key difference between Asia and Western countries is the lack of guideline values for human health or ecological protection in environmental media like groundwater and soil.

The PFAS found to occur predominantly in Asia, cannot be assumed to be of less concern for human exposure.[165] Mahoney *et al.*[64] ranked modelled toxicities (highest to lowest) as PFOS > F-53B > PFOA > HFPO-DA > PFECHS. They expressed particular concern, noting that the ranking of F-53B is higher than that of PFOA. They also indicated that, in certain exposure scenarios, HFPO-DA may be more toxic than PFOA. The database on these novel replacement compounds is currently small and the ranking may change as it grows.[165]

Noting that the persistence, bioaccumulation potential, and toxicity to fish of F-53B are in the same order as PFOS, Ti *et al.*[60] concluded that F-53B is a poor choice as a PFOS substitute. They predicted that the emission of F-53B is likely to increase as regulatory measures to phase out PFOS take effect in Asia, and therefore, the risk associated with F-53B on the environment and human health is likely to be greater in the future. Since F-53B is known to be bioaccumulative and has a long half-life in humans,[64,118] it is critical that data on its production volume and environmental emission are gathered for the risk analysis.

8. Key observations, knowledge gaps and recommendations

From the information summarised above, several observations can be made.

(1) Data regarding PFAS in the Asian environment and human exposure are limited outside China, Japan, and Korea, and extremely scarce beyond East Asia. Furthermore, these data are often restricted by the range of analytes included in studies. Nonetheless, substantial evidence indicates notable human exposure to PFAS in certain regions of Asia.

(2) PFAS concentrations in the environment vary dramatically between and within different Asian countries, likely due to differences in production and industrial use of PFAS as well as the degree of regional development, but limited data from areas outside PFAS production sites limit the ability to draw firm conclusions.

(3) Many PFAS used in Western countries are also observed in Asia.

(4) Some PFAS are predominantly found in Asia (*e.g.* F-53B and HFPO-TA), and cannot be assumed to be less hazardous than more widely studied PFAS.

(5) The higher prevalence in Asia of practices such as self-sourced drinking water, lower levels of wastewater treatment prior to discharge along major rivers used as drinking water sources, and use of wastewater for aquaculture and peri-urban agriculture, may increase human exposure *via* impacted food and waters.

(6) Greater reliance on certain foods known to accumulate PFAS, such as seafood and home-produced poultry eggs, may result in higher exposure in some areas especially in areas near PFAS production sites or intense industrial manufacturing activity.

(7) A large population of Asia relies on a vegetarian diet (*e.g.* in India). While this may limit exposure to long-chain PFAS present in meat from dietary sources, the potential for exposure to short-chain PFAS, which are preferentially taken up by plants, cannot be discounted. This may become more important with time, due to increased production and use of short-chain PFAS.

(8) Regulatory controls on PFAS production and use lag behind those in Western countries. There is little, if any, region-

specific guidance for the management of PFAS-contaminated land and water resources.

Beyond these observations, some assumptions can be made about possible future trends. For example, geographical shifts in PFAS production to other emerging economies in Asia, *e.g.*, India and Vietnam, may occur if regulation increases in China. The existing example of the relocation of the textile industry from China to other Asian countries, such as Vietnam, Bangladesh, Indonesia, Myanmar, and Cambodia, suggests that emissions from the textile processing industry will shift to these countries as well.[166] Currently there are 11 textile exporting countries in Asia.[166]

Despite major information gaps, considerable evidence indicates significant human exposure to PFAS in parts of Asia, necessitating further study, emission controls, and guidance on managing contaminated land. This issue should also be considered within global supply management trends. Asian products often have a price advantage, driving their importation, including those containing PFAS. However, data on PFAS in imported products are limited. Given the broader impact on goods trade, monitoring human exposure to PFAS in Asia is necessary, including potential alternative compounds.

Chemical management, pollution controls, and regulatory responses to environmental contamination vary between countries depending on the priorities and resources available. It is not appropriate to dictate what other countries or regions should adopt. Nevertheless, the following recommendations may be prudent to consider:

(1) Further information on PFAS across Asia is needed, to add to the existing knowledge base from China and other East Asian countries.

(2) Greater understanding of PFAS use and emissions from industries such as automotive, textile and Li-ion battery manufacturing is needed to help inform current and future trends in human exposure.

(3) A global perspective on chemical supply chains is needed to both minimise adverse outcomes, such as increased human exposure to PFAS in Asia, and to control PFAS movement from Asia to other countries.

(4) Multinational companies should practice global stewardship by using the same manufacturing processes and controls as they do in Western countries.

(5) Raising awareness of PFAS-related issues among policy makers and the public should be prioritized. As highlighted in the Zurich Statement (2018) this is a critical requirement to galvanise any response in Asian and other developing economies.[167]

(6) Supporting the availability of PFAS analytical capability in less developed parts of Asia may be needed to improve understanding of human and environmental impacts.

(7) Conducting stocktakes, and developing toxicological profiles for PFAS, that are used primarily in Asia, is needed to support risk assessment and management.

(8) Efforts are needed to evaluate how current wastewater reuse practices, like those used in fisheries and vegetable production, affect PFAS exposure. These efforts should consider ways to improve these practices, taking into account all relevant contaminants, not just PFAS.

This study demonstrates that human exposure to PFAS is significant and likely to change as industries shift geographically due to market pressures and changing regulation. PFAS production and export to other parts of the world, in and from Asia, will continue to be important for the foreseeable future and will continue to have a significant global impact. There is a need to ensure that economic benefits from PFAS production are not prioritised without considering the potential health impacts associated with these activities. Inadequate management of PFAS in Asia will lead to long-term global consequences, given that these chemicals are highly persistent and can travel long distances and impact remote areas where they were neither produced nor used. New research is needed to develop a sound knowledge base, especially in areas outside the major PFAS manufacturing centres in China, as well as in export-driven industries such as textile and Li-ion battery production. While resource constraints in parts of Asia tend to limit broader chemical regulation, the development of effective region-specific regulations for PFAS and broader public awareness are needed to mitigate human impacts.

Author contributions

RSK, KCB, MK, DN, and CPH conceptualised the review. RSK led and coordinated the writing. General contributions: PFAS data on serum and breastmilk – BS, MCV, and JCB. PFAS in water: JCB, MCV, and GGY. Uniqueness of Asia: RSK and KCB. Regulatory aspects: LPP and MK. Review literature: MK and DN. Global context and improvements in the manuscript: CPH and ITC. All authors made essential contributions to writing, reviewing, discussions and the data presented in this review.

Conflicts of interest

There are no conflicts of interest to declare.

Data availability

The data supporting this article have been included as part of the SI.

The full form of acronyms used in this article; distribution of number of studies on PFAS in various matrices in different Asian countries; concentrations of PFAAs in serum in Asia and other regions; data (and their sources) associated with Fig. 3, 5 and 8; PFAS regulatory status in Asian countries. See DOI: **https://doi.org/10.1039/d5em00396b**.

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