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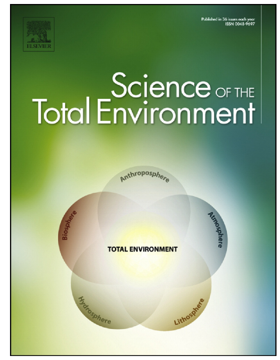

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Environmental Levels and Human Body Burdens of Per- and Poly-fluoroalkyl Substances in Africa: A Critical Review

Patrick Ssebugere [a,*], Mika Sillanpää [b,c,d,*], Henry Matovu [a,e], Zhanyun Wang [f], Karl-Werner Schramm [g], Solomon Omwoma [h], William Wanasolo [i], Emily Chelangat Ngeno [a], Silver Odongo [a]

[a] Department of Chemistry, Makerere University, P.O. Box 7062, Kampala, Uganda

[b] Institute of Research and Development, Duy Tan University, Da Nang 550000, Vietnam

[c] Faculty of Environment and Chemical Engineering, Duy Tan University, Da Nang 550000, Vietnam

[d] School of Civil Engineering and Surveying, Faculty of Health, Engineering and Sciences, University of Southern Queensland, West Street, Toowoomba, 4350 QLD, Australia

[e] Department of Chemistry, Gulu University, P. O. Box 166, Gulu, Uganda

[f] Institute of Environmental Engineering, ETH Zürich, 8093 Zürich, Switzerland

[g] Helmholtz Zentrum Müenchen, German National Research Centre for Environmental Health (GmbH), Molecular EXposomics (MEX), Ingolstaedter Landstrasse 1, Neuherberg, Munich, Germany

[h] Department of Physical Sciences, Jaramogi Oginga Odinga University of Science and Technology, P. O. Box 210-40601, Bondo, Kenya

[i] Department of Chemistry, Kyambogo University, P.O. Box 1, Kyambogo, Uganda

* Corresponding author, *E-mail address:* ssebugere@cns.mak.ac.ug (P. Ssebugere).

Abstract

Per- and polyfluoroalkyl substances (PFASs) are known organic pollutants with adverse health effects on humans and the ecosystem. This paper synthesises literature about the status of the pollutants and their precursors, identifies knowledge gaps and discusses future perspectives on the study of PFASs in Africa. Limited data on PFASs prevalence in Africa is available because there is limited capacity to monitor PFASs in African laboratories. The levels of PFASs in Africa are higher in samples from urban and industrialized areas compared to rural areas. Perfluorooctane sulfonic acid (PFOS) and perfluorooctanoic acid (PFOA) are the dominant PFASs in human samples from Africa. Levels of PFOS and PFOA in these samples are lower than or comparable to those from industrialised countries. PFOA and PFOS levels in drinking water in Africa are, in some cases, higher than the EPA drinking water guidelines suggesting potential risk to humans. The levels of PFASs in birds' eggs from South Africa are higher, while those in other environmental media from Africa are lower or comparable to those from industrialised countries. Diet influences the pollutant levels in fish, while size and sex affect their accumulation in crocodiles. No bioaccumulation of PFASs in aquatic systems in Africa could be confirmed due to small sample sizes. Reported sources of PFASs in Africa include municipal landfills, inefficient wastewater treatment plants, consumer products containing PFASs, industrial wastewater and urban runoff. Relevant stakeholders need to take serious action to identify and deal with the salient sources of PFASs on the African continent.

Keywords

Per- and polyfluoroalkyl substances; PFOA; PFOS; Human body burdens; Africa

List of abbreviations

Per- and polyfluoroalkyl substances (PFASs); perfluoroalkyl acids (PFAAs); perfluoroalkyl carboxylic acids (PFCAs); perfluoroalkane sulfonic acids (PFSAs); perfluorobutanoic acid (PFBA); perfluoropentanoic acid (PFPeA); perfluorohexanoic acid (PFHxA); perfluoroheptanoic acid (PFHpA); perfluorooctanoic acid (PFOA); perfluorononanoic acid (PFNA); perfluoroundecanoic acid (PFUnA); perfluorodecanoic acid (PFDA); perfluorododecanoic acid (PFDoA); perfluoroundecanoic acid (PFUnDA); Perfluorotridecanoic acid (PFTriDA); perfluorotetradecanoic acid (PFTeDA); Perfluorohexadecanoic acid (PFHxDA); perfluorooctanedecanoic acid (PFOcDA); perfluorobutane sulfonic acid (PFBS); perfluorohexane sulfonic acid (PFHxS); perfluoroheptane sulfonic acid (PFHpS); perfluorooctane sulfonic acid (PFOS); perfluorinated sulfonamides (FASAs); perfluorooctane sulfonyl fluoride (POSF); fluorotelomer alcohols (FTOHs); perfluorodecane sulfonic acid (PFDS); polyfluoroalkyl phosphoric acid esters (PAPs); fluorotelomer carboxylates (FTCAs); fluorotelomer unsaturated carboxylates (FTUCAs); fluorotelomer sulfonates (FTSAs); fluorotelemer sulfonate (FTSA); Perfluorooctane sulfonamide (FOSA); perfluorooctane sulfonamidoethanol (FOSE); methylperfluorooctane sulfonamidoethanol (MeFOSE); ethylperfluorooctane sulfonamidoethanol (EtFOSE); methylperfluorooctane sulfonamide (MeFOSA); ethylperfluorooctane sulfonamide (EtFOSA); wastewater treatment plants (WWTPs).

1. Introduction

Per- and polyfluoroalkyl substances (PFASs) are organic compounds containing at least one fully fluorinated carbon moiety (Buck et al., 2011). These include long-chain perfluoroalkane sulfonic acids (PFSAs, $C_nF_{2n+1}SO_3H$, n ≥6) such as perfluorooctane sulfonic acid (PFOS, $C_8F_{17}SO_3H$), and perfluoroalkyl carboxylic acids (PFCAs, $C_nF_{2n+1}COOH$, n ≥7) such as perfluorooctanoic acid

(PFOA, $C_7F_{15}CO_2H$) and their precursors. These compounds exhibit high chemical and thermal stability because of the high-energy carbon-fluorine bond. Many of them cannot be readily degraded by microbial metabolism (Kissa, 2001; Kotthoff et al., 2015), do not readily degrade either in presence of strong acids or oxidizing agents, and are stable in air at high temperatures.

Many PFASs also exhibit specific physicochemical characteristics such as oil- and water-repellence, fire-resistance, weather resistance and surfactant properties, which make them very useful for a wide range of consumer and industrial applications (Franke et al., 2019; Herzke et al., 2012; Kotthoff et al., 2015; Rankin et al., 2016). Some of the application areas include insecticides in agrochemicals, food packaging and stain-resistant coatings, fire-fighting foams, oil well and mining surfactants, furniture and carpets, leather and textile industry, and consumer products for cleaning and polishing (Herzke et al., 2012; Laitinen et al., 2014; Lindstrom et al., 2011).

There are serious health concerns over the widespread exposure of these pollutants in environmental media due to adverse effects reported in animals and humans. Animal exposure to PFASs has been shown to cause hepatic effects, impaired response to antigens and decreased locomotor activity (ATSDR, 2018). In addition, exposure of mice to long-chain PFASs causes disruption of endocrine system (White et al., 2011) and lipid metabolism (Thorsten Stahl et al., 2011). Furthermore, exposure to PFOS and PFOA can lead to cardiovascular disease due to elevated serum cholesterol, decreased sperm count, thyroid dysfunction, low birth weight and size in mice (Gutshall et al., 1989; Olsen et al., 2000). In humans, occupational exposure to PFAS has been associated with liver damage, immunotoxicity, decreased fertility, thyroid diseases, hypertension and/or preeclampsia (ATSDR, 2018); while prenatal exposure to PFASs has been linked to adiposity (Starling et al., 2019), lower birth weight, lower growth outcomes in early infancy decreased antibody response to childhood vaccines (Granum et al., 2013), a higher prevalence of cardiovascular diseases, thyroid diseases, hypertension, increased proportion of days with fever in early stages of life, and respiratory problems such as colds and asthma (Dalsager et al., 2016).

Due to the bioaccumulative, persistent and toxic nature of long-chain PFASs, there have been intentional global efforts to reduce human and environmental exposure to the pollutants. For instance, 3M, the company in the United States that was known to be the major manufacturer of PFASs worldwide since the 1940s, phased-out its products based on C_6, C_8 and C_{10} chemistry in 2001 (Renner, 2001). In May 2009, the Stockholm Convention on persistent organic pollutants (POPs) listed PFOS and perfluoro-octane sulfonyl fluoride (POSF) for global restriction in their production and use (Wang et al., 2017a). In 2012, PFCAs with C_{11}-C_{14} were included on the list of compounds of very high concern under the European chemicals regulation (ECHA, 2013) while in 2013, PFOA and PFOA-related compounds such as ammonium perfluorooctanoate (APFO) were also added to the list (ECHA, 2013). In addition, PFHxS and its salt have been listed as potential persistent organic pollutants by the Stockholm Convention (https://chm.pops.int). Furthermore, the long-chain PFCAs, PFSAs and their precursors have been replaced with their shorter-chain homologues or other PFASs with similar structures such as those with fluorinated segments joined by ether linkages in many parts of the world (Fromme et al., 2017; Wang et al., 2013). However, although the alternative shorter-chain PFASs are less bio-accumulative, they can still persist in the environment the same way the long-chain compounds do (Buck et al., 2011; Fernandez et al., 2016).

Many countries in Africa endorsed the Stockholm convention which entered into force in 2004, pledged their commitment to protect the environment and living organisms from pollutants of global concern and have developed National Implementation Plans (NIPs). However, there are country variations in the efforts by African countries in minimising exposure to PFASs. For instance, in many African countries, NIPs were developed before 2009 and have never been updated. These NIPs covered the initial 'dirty dozen' chemicals but not PFASs. Moreover, in some countries (like Zimbabwe) ratification and enforcement of the convention wasn't possible until 2012, while in other countries like Botswana, the convention has not been ratified to-date (https://chm.pops.int). In addition, there are large data gaps regarding baseline levels of PFASs in the different human and environmental samples in many countries in Africa which limits the setting and implementation of countermeasures towards PFAS exposure, as well as the evaluation of the effectiveness of the global efforts.

While several reviews have been published synthesizing data about the occurrence of organic pollutants in Africa such as pharmaceuticals (Madikizela et al., 2017), endocrine disrupting organic chemicals (including organochlorines and pharmaceuticals) (Gwenzi & Chaukura, 2018), PCDD/Fs and dioxin-like PCBs (Ssebugere et al., 2019); and brominated flame retardants (Brits et al., 2016), none of these reviews has systematically synthesized literature on the current status of PFASs in Africa. Therefore, this review is aimed at raising awareness among stakeholders of the existence of the wide range of PFASs in human and environmental samples from Africa, and to evaluate the effectiveness of global restrictions on PFASs in the African context. In addition, we identify research gaps, and offer suggestions for the next steps in research and regulation.

2. Materials and methods

Literature reviewed in this paper was obtained by searching institutional online databases in Africa and scholarly databases (PubMed, Web of Science®, ScienceDirect and Google

Scholar®) using the search terms 'PFASs' and 'Africa or a specific country name in Africa' AND each of the words 'food, drinking water, surface water, lake, river, sediments, plants, soil, fish, invertebrates, atmosphere, ambient air, food packaging material, dust, wastewater treatment plants, eggs, dumpsites, human blood, or breast milk'. A complimentary search using the terms 'perfluorinated organic compounds', 'perfluoroalkyl acids', 'PFOS', 'PFOA' or 'persistent organic compounds' instead of PFASs was also conducted. In cases where the term 'persistent organic pollutants' was used, the authors first read through the abstracts of the papers for hints on PFASs as part of the analytes before selecting the papers for full-text review. Only peer-reviewed journal articles and theses for which full-text was available in English and published in the period 2005-2020 (after adoption of the Stockholm Convention) were selected. Data from approximately 38 empirical studies on samples from Africa has been reviewed in this paper. Literature on the status of PFASs in other continents has also been included for comparison purposes. For consistency, all the PFASs levels have been presented in nanogram (ng) units, ng g^{-1} or ng mL^{-1}.

3. PFASs exposure pathways

The major sources of PFASs in the environment are point sources such as manufacturing and processing sites of PFASs and their related products, firefighting foam training sites, sewage treatment plants, landfills, and diffuse sources such as use and disposal of consumer products containing PFASs (Boucher et al., 2019; Dauchy et al., 2017; Wang et al., 2017b; Wang et al., 2014a, 2014b).

Some PFASs may also be introduced into the environment through transformation of their precursors. For example, polyfluoroalkyl phosphoric acid esters (PAPs), fluorotelomer carboxylates (FTCAs), fluorotelomer unsaturated carboxylates (FTUCAs) and fluorotelomer sulfonates (FTSAs) may be degraded to form perfluoroalkyl acids (PFAAs) (Kannan, 2011; Yang et al., 2014) or they may undergo transformation via metabolism to form PFSAs and PFCAs particularly in the aquatic environment (Bossi et al., 2008; Kunacheva et al., 2011). Furthermore, some PFAS precursors (e.g. fluorotelomer alcohols, FTOHs; perfluorinated sulfonamides, FASAs) are subject to several transformation pathways in the atmosphere and/or under aerobic/anaerobic conditions in other environmental compartments which could form PFCAs (Gawor et al., 2014; Shoeib et al., 2016). Overall, the exact sources of exposure for a given location may vary depending on the contaminant type, consumption pattern and country.

In Africa, the sources of the pollutants have not been clearly identified which makes it hard for mitigation measures to be carried out. However, releases of effluents from wastewater treatment plants, hospitals, solid waste dumpsites and urban centers have been cited as sources responsible for the occurrence of chemical pollutants in water bodies via runoff and/or atmospheric deposition (Arinaitwe et al., 2016; Groffen et al., 2018; Ibor et al., 2020). Discharges from industries and hospitals have also been reported as contributors to the amounts of PFASs released into the environment, such as in the Nakivubo Channel and Lake Victoria in Uganda (Dalahmeh et al., 2018). Furthermore, there are several fluorochemical industries in South Africa such as Pelchem, a manufacturer of fluoropolymers and perfluorocarbon liquids from fluorspar (https://www.pelchem.com/). However, since these industries have not reported manufacture of PFASs, it is not clear if such industries could be point sources of PFASs. Following the aforementioned sources, the warm tropical temperatures and high relative humidity in Africa would favor the volatilization of organic pollutants, such as (semi-) volatile PFASs, and their deposition in remote places (Rankin et al., 2016; Ssebugere et al., 2019).

When released into the environment, PFASs can enter bodies of humans through different routes including indoor dust ingestion (Egeghy & Lorber, 2011) and dietary intake through consumption of contaminated drinking water and food (Denys et al., 2014; Domingo & Nadal, 2017; Domingo & Nadal, 2019; Eriksson et al., 2013; Essumang et al., 2017). Studies have shown that, since the compounds can bioaccumulate through aquatic and terrestrial food chains, humans get exposed to PFAAs through food especially that of animal origin. For instance, Vestergren et al. (2013) showed that cows receiving naturally contaminated feed and drinking water bioaccumulate long chain PFAAs in their tissues (liver, beef and blood), which consequently leads to human exposure. Other intake pathways include inhalation of contaminated air/dust particles (Laitinen et al., 2014), to a lesser extent dermal contact via skin through use of personal care products (Fujii et al., 2013; Schultes et al., 2018); and through breastfeeding for infants (Papadopoulou et al., 2016).

Studies to assess PFAS exposure, especially through diet, have been carried out in other parts of the world (Domingo & Nadal, 2017), but such data are relatively scarce for Africa. However, some studies have suggested human dietary exposure to PFAS in Africa. For instance, a recent study by Vaccher et al. (2020) reported detectable levels of PFASs in beef samples and fish samples marketed in West Africa, while in Uganda, Dalahmeh et al. (2018) found detectable levels of PFASs in yams (*Dioscorea* spp.), sugarcane (*Saccharum officinarum)* and maize (*Zea mays)* samples. Since fish, beef, maize, sugarcanes and yams are important dietary components in many African countries, the detection of PFASs in these samples shows that consumption of these foods continues to be a human exposure pathway to PFASs. Literature on PFAS exposure through indoor and outdoor dust is also still limited for Africa. Further evaluation of potential PFAS exposure of the general public in African countries through these sources needs to be carried out. This will allow the relevant institutions to lay measures against continued exposure to PFASs in Africa.

4. Monitoring of PFASs in human and environmental matrices in Africa

This section presents a critical analysis of PFASs concentrations in the different human and environmental matrices in Africa such as: sewerage sludge (Table 1), waste water, drinking water, surface water and pore water (Table 2), sediments and suspended solids (Table 3), fish (Table 4), soils and crops, foods and food contact materials, indoor dust and ambient air, wildlife, as well as human blood from different locations in Africa.

4.1 Wastewater

In African environments, wastewater from industrial, urban centres and hospitals, and landfill leachates has been reported to be a major source of organic pollutants such as pharmaceuticals (Agunbiade & Moodley, 2016; Madikizela & Chimuka, 2017), polycyclic aromatic hydrocarbons (Edokpayi et al., 2016) and other halogenated contaminants (Daso et al., 2012; Nomngongo et al., 2012). A similar case can be made regarding PFASs. For instance, in Kenya, Chirikona et al. (2015) reported concentration ranges of 1.3-28 ng L^{-1} for PFOA and 0.9-9.8 ng L^{-1} for PFOS in wastewater from WWTPs, while Orata et al. (2009) reported maximum concentrations of PFOA and PFOS as 96.4 and 11.7 ng L^{-1} in municipal waste treatment ponds in the Lake Victoria catchment area on the Kenya side.

In South Africa, Adeleye (2016) analysed PFASs in WWTPs of Beaufort West, Scottsdene and Zandvliet in and around Cape Town. The maximum concentrations in effluents were 48.53 ng L^{-1} of PFHpA from Zandvliet; 13.10 ng L^{-1} of PFOA from Zandvliet; 18.8 ng L^{-1} of PFNA from Beaufort West; 6.21 ng L^{-1} of PFDA from Beaufort West and 4.22 ng L^{-1} of PFUnDA from

Scottsdene, and 10.24 ng L^{-1} of PFOS from Bellville. Recently, Kibambe et al. (2020) determined levels of sixteen PFASs in three WWTPs in Gauteng Province, South Africa. In wastewater from the three WWTPs, ∑PFASs in the influents and effluents were 129-626 and 35.8-224 ng L^{-1}, respectively. PFOS dominated the profile with concentrations up to 508 ng L^{-1}. In another study, Dalahmeh et al. (2018) investigated 26 PFASs in influents and effluents from a WWTP that handles wastewater from Kampala, the most urbanized city in Uganda. Total (∑) PFASs in effluents from the treatment plant (5.6-9.1 ng L^{-1}) were higher than those in the corresponding influents (3.4-5.1 ng L^{-1}), which can be attributed to poor removal of PFASs by the treatment plant. However, desorption of the pollutants from biosolids within the plant (Orata et al., 2009) and the formation of the contaminants through transformation of precursor compounds during the wastewater treatment processes (Arvaniti et al., 2012) could be other factors responsible for the higher levels in the effluents than in the influents.

The concentrations of PFASs in WWTPs in Africa appear to be variable within countries. The levels of PFASs in wastewater in South Africa are comparable to those in Kenya but are higher than those in Uganda. These variations could be due to a difference in the levels of urbanization, nature of consumer products used, and efficiency of wastewater treatment processes in the different countries. In many countries in Africa, treatment of wastewater is not widely carried out due to energy and financial resource constraints. In many cases, wastewater is released into waste stabilization ponds (Edokpayi et al., 2017) or directly into natural wetlands for natural treatment (Dalahmeh et al., 2018; Wang et al., 2014). These wetlands may reduce PFASs concentrations in water but the removal efficiency is never perfect (Dalahmeh et al., 2018; Kibambe et al., 2020), which leads to continued exposure of aquatic life to the pollutants.

In major cities in Africa, wastewater from households and industries is treated using conventional primary treatment methods, as well as secondary treatment methods in WWTPs using activated sludge and biofilms (Kibambe et al., 2020; Wang et al., 2014). In countries like South Africa ultrafiltration is applied and reverse osmosis is being implemented (Adeleye, 2016). In some cases, the WWTPs are relatively effective in removing PFASs from wastewater for example, Kibambe et al. (2020) reported removal efficiencies of PFOS as 94% at Zeekoegat WWTP in South Africa. However, in other areas in Africa, removal efficiencies still need to be improved upon as some WWTPs reported removal efficiencies as low as 25% for PFDA at Phola WWTP (Kibambe et al., 2020), while in Uganda, the Bugolobi WWTP showed higher levels of PFASs in effluents than in influents (Dalahmeh et al., 2018). Therefore, PFAS contaminated effluents still find their way into the environment. Specifically, issues related to poor maintenance and operation of the WWTPs in Africa need to be addressed. Furthermore, disposal of PFASs contaminated membranes and membrane reject water continues to be a challenge which needs urgent attention in many countries, especially the developing ones (Eschauzier et al., 2012).

It must be noted that the detection of PFAS concentrations higher in WWTP effluents than in the influents is not uncommon, even in developed countries outside Africa (Arvaniti et al., 2012; Chen et al., 2018; Coggan et al., 2019; Ratola et al., 2012). This is due to the unique physicochemical properties of PFASs, such as high thermal and chemical stabilities. For instance, the C-F bond in PFASs is strong and does not easily break during conventional water treatment methods such as ozonation (Eschauzier et al., 2012). In addition, PFAAs especially <C8 tend to be desorbed from the adsorbents as a result of displacement by more highly sorptive matrix components (Eschauzier et al., 2012). To address these challenges, a variety of novel

methods of PFAS removal from wastewater such as nanofiltration are being used in developed countries (Mudumbi et al., 2017) but, due to resource constraints, the methods are not popular in many countries in Africa. In addition, currently used methods of water treatment such as the use of granular activated carbon (GAC) may be effective for removal of longer chain PFAAs but not the shorter chain congeners (Eschauzier et al., 2012; Guelfo et al., 2018). Therefore, researchers in Africa need to explore the feasibility of using relatively cheaper alternative water remediation techniques such as agro-based adsorbents (Omo-Okoro et al., 2018a, 2018b). In addition, policy makers should consider investing more resources in the WWTPs infrastructure.

4.2. Sewerage sludge

Within WWTPs, adsorption onto biosolids is an important mechanism for removal of PFASs from wastewater. However, disposal of these biosolids is a serious problem because more advanced methods such as sludge incineration are not affordable in many African countries. As such, in majority cases, biosolids which exit the WWTPs as sewerage sludge end up being sent to poorly constructed landfills or applied as fertilizers in soils (Yan et al., 2012). Consequently, these solids become a significant exposure pathway to pollutants in the environment (Ssebugere et al., 2019). Available literature on sludge samples in Africa shows that the levels of PFASs are low compared to those in industrialized countries in Asia, Europe and USA (Table 1).

In Nigeria, Sindiku et al. (2013) determined PFCAs and PFSAs in sludge from domestic, hospital and industrial WWTPs. The authors reported PFOA and PFOS in ranges of 0.01-0.416 and <0.01 to 0.540 ng g^{-1} dry weight (dw), respectively. The long chain PFCAs (>8 fluorinated carbons) were detected at higher levels compared to the short chain ones (<8 fluorinated carbon chain). The dominance of long chain over short chain PFCAs was also observed by Chirikona et al. (2015) for sludge samples from five similar WWTPs in Kenya. Chirikona et al. (2015) reported concentration ranges of PFOA in sludge as 0.032-0.345 ng g^{-1} and <0.015-0.673 ng g^{-1} for PFOS. In both studies, the observed levels were generally attributed to hospital discharges such as medical devices with PFAS layers including tetrafluoroethylene copolymer (ETPE) and radio-opaque ETFE production, in vitro diagnostic medical devices and CCD colour filters.

The levels by Sindiku et al. (2013) and Chirikona et al. (2013) were several magnitudes lower than those reported in highly industrialized continents such as in North America (Higgins et al., 2005; Loganathan et al., 2007; Yoo et al., 2009), Asia (Guo et al., 2008; Yu et al., 2009) and Europe (Gómez-Canela et al., 2012; Sun et al., 2011) (Table 1). However, there's still need to carry out background studies in all African countries and to set strict guidelines on the permissible levels of PFASs in sludge that can be applied to soils. In addition, environmental management authorities in African countries should ensure that, where possible, construction of municipal landfills receiving these biosolids is done according to standard guidelines as leachates from these may contaminate ground water (Hepburn et al., 2019), and lead to adverse health effects.

4.3. Drinking water

The detection of PFASs in drinking water sources such as tap water and bottled water may shed light on the possible route of continued exposure of humans to PFASs. So far, only two studies have investigated the presence of PFASs in drinking water in Africa. A study by Essumang et al. (2017) reported ∑PFAAs (i.e. PFCAs and PFSAs) as 197-200 ng L^{-1} in tap water in Ghana. The authors reported PFOA and PFOS as the most predominant compounds in the samples (both compounds contributed about 99% to the ∑PFAAs). The levels of PFOS and PFOA reported in this study were, in majority cases, higher than the Environmental Protection Agency (EPA) safe drinking water concentrations of 70 ng L^{-1} for the sum of the two compounds (Dauchy, 2019), suggesting that consumption of tap water in Ghana may result in long term effects related to these pollutants. Ghana is a developing country with no reported production of PFASs. The exceptionally high levels of PFASs in tap water reported by Essumang et al. were as a result of inefficiencies of water treatment plants in reducing PFAS levels in water.

In another study, Kaboré et al. (2018) reported low levels of PFASs in tap and bottled water from Burkina Faso and Ivory coast. The levels of PFOA and PFOS were <0.06-1.89 and 0.12-3.85 ng L^{-1} in samples from Burkina Faso, while those in samples from Ivory Coast were <0.06-0.44 and <0.03-1.32 ng L^{-1}, respectively. Generally, PFOS dominated the distribution of PFASs in the samples. However, some samples from tap water near municipal landfills showed exceptionally high levels of 5:3 FTCA (up to 32 ng L^{-1}), suggesting that these landfill sites contaminate ground water. The levels in Burkina Faso and Ivory Coast were comparable to those found in drinking water in Canada, USA, China, Chile in the same study (Kaboré et al., 2018), and Germany (Shafique et al., 2017).

At a continental level, the two studies highlight a broader problem of pollution of water sources including ground water from which the tap water is drawn by industries and WWTPs in the vicinity of the sources. The studies also emphasize the need to carry out more monitoring studies of this nature across Africa and, to set and enforce strict guidelines regarding drinking water quality. In addition, more monitoring of legacy and emerging pollutants in hotspot areas such as landfills, industrial areas and WWTPs should be done as these tend to contaminate ground water and expose humans to pollutants such as PFASs.

4.4. Environmental waters, sediments and suspended solids

In the aquatic environment, water is one of the major transport media for substances with hydrophilic functionalities such as PFASs, while sediments normally act as reservoirs for these compounds since they also have hydrophobic functionalities (Chen et al., 2017). Once PFASs are in the aquatic environment, they undergo partitioning processes, whereby long-chain PFCAs and PFSAs tend to bind to solid particles whereas large amounts of short-chain PFSAs and PFCAs are distributed in the water phase (Ahrens & Bundschuh, 2014). Some studies have been carried out to establish the levels of PFASs in surface water and pore water (Table 2), and sediment samples and suspended solids from African water bodies (Table 3).

4.4.1. Surface and pore waters

In Uganda, Dalahmeh et al. (2018) determined PFASs (PFCAs: PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFUnDA, PFDoDA, PFTriDA, PFTeDA, PFHxDA, PFOcDA; and PFSAs: PFBS, PFHxS, PFOS, PFDS) levels in surface water from Lake Victoria and Nakivubo Channel. The channel flows directly through the industrial and urban areas of Kampala (the capital city of Uganda) before channeling its water into Lake Victoria. PFAS concentrations were 5-fold higher in water from the Nakivubo Channel (8.5-12 ng L^{-1}) than that from Lake Victoria (1.0-2.5 ng L^{-1}). In Kenya, Orata et al. (2009) reported concentrations of PFOA and PFOS as 0.4-9.64 and <0.4-13.2 ng L^{-1} in river water, and 0.4-11.7 and <0.4-2.53 ng L^{-1} in Lake Victoria water, respectively. These pollutants were also reported at levels of 0.8-2.8 and 3.9-10.1 ng L^{-1}, respectively, in surface water; while the levels in porewater were 1.71-16.2 ng L^{-1} and 10.9-20.4 ng L^{-1}, respectively, from seven rivers in Nigeria (Ololade, 2014; Ololade et al., 2018).

In a related study, Ahrens et al. (2016a) reported concentrations of the PFASs: PFSAs (PFBS, PFHxS, PFOS, PFDS) and PFCAs (PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFUnDA, PFDoDA, PFTriDA, PFTeDA, PFHxDA, PFOcDA) in water from Lake Tana, Ethiopia as 0.073-5.6 ng L^{-1} (mean; 2.9 ng L^{-1}). In Ghana, Essumang et al. (2017) reported mean levels of ∑PFASs (PFHxA, PFHpA, PFOA, PFDA, PFOS, PFPeA) in surface water from Kakum and Pra Rivers as 281 and 398 ng L^{-1}, respectively in Ghana. PFOA was the dominant compound.

Shafique et al. (2017) reported levels of PFCAs in River Siosani, Kenya. The concentrations of the pollutants in sparsely populated areas were: PFDoDA (mean; 23.3 ng L^{-1}), PFDA (10.8 ng L^{-1}), PFUnDA (10.5 ng L^{-1}), PFNA (8.6 ng L^{-1}), PFOA (1.6 ng L^{-1}), PFPeA (1.3 ng L^{-1}), PFHxA (1.3 ng L^{-1}), PFBA (0.6 ng L^{-1}) and PFHpA (0.4 ng L^{-1}). In densely populated areas, the levels were higher: PFDoDA (mean; 31.6 ng L^{-1}), PFNA (25.9 ng L^{-1}), PFDA (21.6 ng L^{-1}), PFUnDA (15.9 ng L^{-1}), PFOA (8.8 ng L^{-1}), PFOA (1.6 ng L^{-1}), PFHxA (2.5 ng L^{-1}), PFPeA (1.3 ng L^{-1}), PFHpA (1.0 ng L^{-1}) and PFBA (0.7 ng L^{-1}).

In South Africa, PFOA was detected at maximum concentrations of 310 and 1089 ng L^{-1} for PFOA in aMatikulu and uMvoti estuaries, respectively; while the maximum levels of PFOS were 54.2 and <14.6 ng L^{-1}, respectively (Fauconier et al., 2019). In this study, the maximum levels of PFBA, PFNA, PFDA and PFDoDA in the two rivers were 771, 126, 95, 67.6 ng L^{-1}, respectively; while PFTrDA, PFTeDA, PFHxA, PFHpA and PFDS could not be quantified (LOQs; 2.11-1177 ng L^{-1}). Groffen et al. (2018) reported maximum levels of PFASs in water of up to 38.5 ng L^{-1} in Vaal River; while Verhaert et al. (2017) reported that all PFASs were less than the limit of quantification (LOQ) in all water samples in Olifants river. In another study, PFOA and PFOS were detected at maximum levels of 314 and 182 ng L^{-1} for Diep River; 390

and 47 ng L^{-1} for Salt River; and 146 and 23 ng L^{-1} for Eerste River, respectively (Mudumbi et al., 2013); and at levels ranging from 12.8 to 62.6ng L^{-1} and <0.06 to 12.4 ng L^{-1}, respectively in Plankenburg River (Fagbayigbo et al., 2018).

4.4.2 Sediments and suspended solids

Ahrens et al. (2016a) reported concentrations of the PFASs: PFSAs (PFBS, PFHxS, PFOS, PFDS) and PFCAs (PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFUnDA, PFDoDA, PFTriDA, PFTeDA, PFHxDA, PFOcDA) as 0.22-0.55 ng g^{-1} dw (mean; 0.30 ng g^{-1} dw) in sediments from Lake Tana, Ethiopia. In another study, Orata et al. (2011) reported levels of PFOA and PFOS as <1-24.1 and <1-4.0 ng g^{-1} dw, respectively, in the sediments from the Winam Gulf of Lake Victoria, Kenya. The pollutants PFOA and PFOS were also detected in sediments of rivers on the Kenya side of the Lake Victoria basin as 1.4-99.1 and <1-57.5 ng g^{-1} dw, respectively (Orata et al., 2011). The levels were higher in the rivers than in the lake sediments possibly because the rivers are closer to the point sources of PFASs such as the wastewater settlement ponds, industries and urban centres. The pollutants levels also get diluted as they flow into the lake.

In Western Cape (South Africa), PFOS and PFOA levels were reported as 10.7-772 and 2.53-121.1 ng g^{-1} dw in Diep river; 38.6-187 and n.d. to 19.9 ng g^{-1}dw for Salt River; and 15.2-193 and 0.72-75.1 ng g^{-1} dw, respectively (Mudumbi et al., 2014). In the same rivers, the same pollutants were reported at mean concentrations of 28 and 26 ng g^{-1} dw for Diep River, 16 ng g^{-1} dw and non-detectable (n.d) for Eerste River, and 14 and 5 ng g^{-1} dw for Salt River, respectively, in suspended solids (Mudumbi et al., 2013). Fagbayigbo et al. (2018) determined levels of PFOA and PFOS in sediments from Plankenburg River in South Africa. The levels were in the range

0.14 to 0.33 ng g^{-1} dw (PFOA) and <0.02 to 0.7 ng g^{-1} dw (PFOS). Another study by Groffen et al. (2018) determined levels of PFASs in sediments of Vaal River in South Africa. The authors reported PFASs in the sediments were <LOQ, with the exception of PFOS at Thabela Thabeng (2.36 ng g^{-1} dw). The PFOS levels at Thabela Thabeng were attributed to effluents containing sewage and wastewater from Suikerbosrant and/or Klip rivers which flow into the Vaal River.

In general, reviewed literature shows regional variations in the concentrations of PFASs in fresh-water bodies in Africa, possibly due to differences in pollutant sources. The highest levels of PFOA in surface water in Africa have been reported from uMvoti Estuary, South Africa (Fauconier et al., 2019) while the highest levels in sediments have been reported in the rivers in the highly industrialised region of Western Cape, South Africa. Overall, discharges from WWTPs (Orata et al., 2011; Orata et al., 2009), urban and industrial discharges (Ahrens et al., 2016b; Fagbayigbo et al., 2018; Fauconier et al., 2019; Groffen et al., 2018; Ololade, 2014; Ololade et al., 2018; Shafique et al., 2017), agricultural run-off (Ololade, 2014), run-off from municipal landfills (Mudumbi et al., 2014), as well as poor disposal of used PFAA-containing products (such as carpets, electronics and fire-fighting foams) (Essumang et al., 2017; Fabayigo et al., 2018) have been identified as the major sources of PFASs in aquatic systems.

The levels of PFASs in sediments reported in Africa are in majority cases higher than those reported in sediments elsewhere such as those from Dalian Bay, China (range; 1.49-2.66 ng g^{-1} dw) (Ding et al., 2018) Bohai, Yellow and East China seas (<LOD-2.78 ng g^{-1} dw) (Gao et al., 2014) and marine sediments in Hong Kong (0.1-1.59 ng g^{-1} dw) (Loi et al., 2013). For water samples, the levels of PFOS and PFOA in water samples from Africa were, in majority cases, in the same range of data as those reported by Ahrens et al. (2010) in Northern Europe, Atlantic and Southern Ocean (maximum values of 0.232 and 0.223 ng L^{-1}, respectively). However, in some

cases, the reported concentrations of PFOS and PFOA in Africa were even higher than those reported in some areas in developed countries such as in the Great Lakes (range of 21-70 ngL^{-1} for PFOS and 27-50 ng L^{-1} for PFOA) (Boulanger et al., 2004) and, Michigan and New York waters (1.8-17 ng L^{-1}, PFOS and 4.4-22 ng L^{-1}, PFOA) (Sinclair et al., 2004).

Having higher concentrations of PFASs in sediment and water samples from Africa compared to those from developed countries is surprising since no production of PFASs in Africa has been reported. Thus, importation of products that contain PFASs coupled with improper waste management could be some of the factors responsible for the observed levels. Therefore, environment management authorities in the different countries in Africa must set strict guidelines for waste management to protect and limit exposure of water bodies to PFASs. In addition, continuous monitoring of emerging and legacy persistent organic pollutants should be carried out in areas with high intensity of anthropogenic activities.

4.5. PFASs levels in fish and invertebrates

4.5.1. Fish species

In Africa, fish is a big part of diet as a source of proteins, and a source of livelihood for those engaged in the fishing activities (Kolding et al., 2016). However, water bodies are normally prone to contamination due to the large population densities following industrial and commercial developments around them (Orata et al., 2011). As such, many pollutants enter the water bodies and may end up bioaccumulating in fish, posing adverse health risks to humans who consume the fish. Consumption of PFAS contaminated fish has been reported elsewhere for example, in France and Italy (Denys et al., 2014; Squadrone et al., 2014; Squadrone et al., 2015). However, the extent of human exposure depends on several factors, among which is the method of fish preparation.

In Africa, the fish for consumption is mainly prepared by cooking, deep-frying in oil, sun-drying and smoking. Studies have shown that these conventional food preparation methods are not effective in reducing the levels of PFASs in edible fish (Bhavsar et al., 2014; Taylor et al., 2019) and may even increase the levels of PFASs (Vassiliadou et al., 2015). Del Gobbo et al. (2008) reported that baking is the most effective way of reducing PFASs load in edible fish. Unfortunately, baking in electric ovens is not possible for many people in Africa due to lack of electricity. Therefore, it is likely that, using the preparation methods currently available, the continued consumption of PFAS-contaminated fish in Africa could have deleterious effects on humans. These effects must be assessed. In addition, governments and policy makers in different countries in Africa should try to ensure access to affordable and sustainable energy resources to the general population which could later influence the preparation methods used.

Currently, data on the occurrence of PFASs in fish from water bodies in Africa is limited (Table 4). In these samples, the PFAS levels were generally higher in liver samples than in muscles (Bangma et al., 2017; Fauconier et al., 2019; Verhaert et al., 2017). It should be noted that metabolism of pollutants including PFASs occurs in the liver and the compounds are *proteinophilic*, which explains why levels of PFASs in protein-rich tissues such as livers, kidney and plasma are higher than those in muscles (Groffen et al., 2018; Houde et al., 2011; Verhaert et al., 2017).

In South Africa, Verhaert et al. (2017) reported low levels of PFASs in muscle and liver tissues of several fish species from the Olifants River Basin. PFOS, PFOA and PFNA were the only detected PFASs in muscle tissues at maximum levels of 2.7, 0.42 and 0.14 ng g^{-1} wet weight (ww), respectively. The mean concentrations of the pollutants in liver tissues were 11, 0.48 and 0.22 ng g^{-1} ww, respectively. The levels reported by Verhaert et al. (2017) were higher than those by Mwakalapa et al. (2018) who reported non-detectable levels of PFASs in muscles of farmed and wild milkfish from Mtwara in Tanzania and, islands of Pemba and Unguja in Zanzibar. However, this comparison should be interpreted with care since the detection limits of Mwakalapa et al. (2018) were high (0.08-0.42 ng g^{-1} ww) so the difference in levels of PFASs in the two studies could have been insignificant due to the high method detection limits in the later study.

The levels reported by Verhaert et al. (2017) and Mwakalapa et al. (2018) were lower than those reported in Kenya, Ethiopia and other parts of South Africa. For instance, in Kenya, a study by Orata et al. (2008) determined levels of PFOS in Nile tilapia (*Oreochromis niloticus*) and Nile perch (*Lates niloticus*) from the Kenya side of Lake Victoria. The maximum concentrations in *L. niloticus* were 10.5 and 35.7 ng g^{-1} ww in muscle and liver tissues, respectively, while those in *O. niloticus* were 12.4 and 23.7 ng g^{-1} ww for muscle and liver tissues, respectively. The authors attributed the occurrence of PFOS in fish to inefficient sewer treatment plants and industrial activities in the catchment of Lake Victoria.

In Ethiopia, Ahrens et al. (2016) assessed the levels of PFASs in fish species from Lake Tana. The mean value of PFASs in muscle tissues was 1.2 ng g^{-1} ww with a maximum concentration of 5.8 ng g^{-1} ww. PFAS concentrations in piscivorous fish species (*Labeobarbus megastoma* and *Labeobarbus gorguari*) were reported to be significantly higher than those in non-piscivorous species (*Labeobarbus intermedius*, *O. niloticus* and *Clarias gariepinus*). The high concentrations in the piscivorous fish could be attributed to their foraging character and given the fact that they are bottom feeders and could be exposed to the pollutants by the contaminated sediments.

Bangma et al. (2017) examined concentrations of 15 PFAAs in tissues of male *Oreochromis mossambicus* collected from Loskop Dam, Mpumalanga in South Africa. PFAAs and PFOS were detected at a median concentration of 41.6 ng g^{-1} ww while PFOA was the least detected (0.0825 ng g^{-1} ww). ∑PFAA levels in the tissues were in the order plasma (median; 22.2 ng g^{-1} ww), liver (11.6 ng g^{-1} ww) and kidney (9.04 ng g^{-1} ww). In another study, Groffen et al. (2018) determined levels of PFASs in muscles and liver of two fish species (*Labeo capensis* and *Labeobarbus aeneus*) from Vaal river in South Africa. PFASs were up to 289 ng g^{-1} ww in liver and 34 ng g^{-1} ww in muscle. The authors noted that PFAS-contaminated sediments were the probable source of fish contamination. PFOS was the most dominant PFASs in this study and its concentrations were higher than those reported by (Verhaert et al., 2017) but were comparable to those reported by Fauconier et al. (2019) from uMvoti and aMatikula Estuaries in South Africa. Fauconier et al. (2019) reported levels of 15 PFASs in tissues of fish (*Ambassis natalensis*, *O. mossambicus*, and *Rhabdosargus holubi*) caught from the two estuaries in South Africa. PFOS was the dominant compound in fish samples at levels of 0.09-2.25 and 1.5-27.9 ng g^{-1} ww in muscle and liver tissues, respectively. PFOA was detected in all samples at levels of 0.08-0.67 ng g^{-1} ww in muscle tissue and 0.17–1.48 ng g^{-1} ww in liver tissues.

In another study, Ojemaye & Petrik (2019) determined levels of PFASs in wild pelagic fish species *Thyrsites atun* (snoek), *Sarda orientalis* (bonito), *Pachymetopon blochii* (panga) and *Pterogymnus laniarius* (hottentot) from Kalk Bay harbour, Cape Town. The authors reported that long chain PFAAs such as PFDA (20.13–179 ng g^{-1} d.w), PFNA (21.22–114 ng g^{-1} d.w), and PFHpA (40.06–138.3 ng g^{-1} d.w) were the most predominant among the PFASs. PFOA had the lowest concentrations across all the fish species, which could be due to the fact that the bioaccumulation potential of PFAAs increases with chain length (Lescord et al., 2015). Furthermore, no specific trend of PFASs concentrations was observed in the different parts of the fish samples, implying that bioaccumulation of the pollutants occurs in all the parts of fish. Because these results were reported in ng g^{-1} d.w units, they could not be compared with other results in Africa in which the levels are reported in ng g^{-1} ww units.

Overall, the studies on fish show a wide variation in levels and profiles of PFASs from different regions of Africa, as well as those in water bodies within the same country as reflected in fish samples from South Africa. Samples from areas with greater influences from industries (such as the Vaal River and uMvoti Estuary in South Africa) and wastewater treatment plants (Lake Victoria, Kenya) show higher PFAS concentrations than areas with lower intensities of anthropogenic activities. When compared, the levels of PFASs in Africa are, in majority cases, lower than those reported in other countries in the world (Table 4). These variations continue to emphasize the differences in existing sources of environmental exposure of the compounds. However, the available literature about Africa is only from studies which have been carried out in Kenya, Tanzania, Ethiopia and South Africa only. It is not clear whether fish from other countries in Africa contain higher or lower levels. This needs further investigation.

In addition, diet and physiological status appear to influence the levels of PFASs in fish species (Fauconier et al., 2019). With respect to diet, piscivorous fish (such as *L. niloticus* and *A. natalensis*) have higher levels of PFASs in their tissues than herbivorous (e.g. *O. niloticus*) and omnivorous (e.g. *O. mossambicus*) fish species in the same water bodies (Ahrens et al., 2016b; Fauconier et al., 2019). The piscivorous species are higher in the trophic food chain and are therefore likely to accumulate higher concentrations of PFASs. In terms of physiological conditions, fish with health conditions such as pansteatitis have been reported to have lower PFASs burdens than healthy fish, possibly due to lower dietary intake, as a result of reduced appetite or foraging success (Bangma et al., 2017), as well a reduction in the abundance of proteins (e.g. albumin) to which the PFAAs could bind (Bowden et al., 2016). It should be noted that the relationship between health status and PFASs burdens in fish and wild life is still understudied. However, a study by Van de Vijver et al. (2003) reported that harbor seals which were affected by bronchopneumonia had lower levels of PFOS compared to health ones. We argue that researchers must therefore critically assess the health status of their samples to aid interpretation and discussion of their PFASs data, and to generate broader implications.

4.5.2. Invertebrates

In South Africa, Groffen et al. (2018) determined levels of PFASs in different invertebrate taxa (Ephemeroptera, Decapoda, Hirudinea, zooplankton, *Caridina nilotica*, Gyrinidae) from the Vaal River. PFASs varied from n.d. to 34.5 ng g^{-1} ww in these species. Authors attributed the higher levels to influences from the tributaries of the river which are polluted by industries and urban centres. The study also noted that PFOS levels in water and invertebrates were negatively correlated suggesting that PFAS uptake by invertebrates occurs through multiple sources including water, sediment and food.

Compared to the levels reported by Groffen et al. (2018), a recent study by Fauconier et al. (2019) reported lower levels of PFASs in several invertebrate species including shrimps (*Caridina nilotica*), crabs, worms and gastropods in two aquatic (uMvoti and aMatikula) estuaries in South Africa. In *Caridina nilotica*, levels of PFOA, PFOS, PFDA, PFDoDA were 0.64-0.74, 0.5-0.58, 1.31-1.73 and <0.06-0.24 ng g^{-1} ww, respectively in uMvoti Estuary, while those in aMatikula Estuary were 0.34-6.5, 0.49-1.76, <0.73-1.73 and 0.16-1.3 ng g^{-1} ww, respectively. In gastropods, the levels of same pollutants were 2.26-3.57, <0.09-0.54, 0.85-1.54 and 0.41-1.10 ng g^{-1} ww in uMvoti Estuary, respectively, while those in aMatikulu Estuary the levels were 2.26-5.37, <0.09-0.54, <0.71-1.54 and 0.41-1.09 ng g^{-1} respectively. Levels of PFBA, PFPeA, PFHpA, PFNA, PFDS, PFTrDA were all below detectable limits. The authors noted that shrimps had higher levels of PFASs than snails, worms and crabs. The levels of PFOA in worms were comparable to those in higher organisms, suggesting no biomagnification of the pollutant.

4.5.3. Is there evidence for biomagnification of PFASs in African aquatic systems?

Factors that influence the bioaccumulation potential of PFAS include the chain length of the PFAS and the functional group attached to the carbon chain. For instance, the sulphonate in PFSAs (such as PFOS) has a higher bioconcentration factor, half-life and uptake than the carboxylate group in PFAAs (Ahrens et al., 2016b; Verhaert et al., 2017). This explains why PFOS is a major PFAS in fish tissues. In addition, elsewhere, long-chain PFCAs and PFOS have been shown to bio-magnify in freshwater food webs such as in France (Munoz et al., 2017; Simmonet-Laprade et al., 2019), Hong Kong (Loi et al., 2011) and China (Fang et al., 2014).

To date, no clear conclusions can be reached about bioaccumulation and trophic magnification of PFASs in the aquatic systems in Africa due to inconsistent results reported by several studies. Verhaert et al. (2017) reported no significant relationship between ∑PFAS and trophic level of fish species sampled from the Olifants River Basin, South Africa; Fauconier et al. (2019) did not observe tropic magnification in uMvoti and aMatikulu estuaries, South Africa; while Groffen et al. (2018) observed trophic transfer and biomagnification for PFBA, PFDA and PFTrA in the aquatic environment of Vaal River, South Africa. However, all these results need further confirmation since, the sample sizes were very small to allow generalizations. Besides, all the fish species sampled in Groffen et al. (2018) were omnivorous which could lead to higher levels of PFASs in the samples due to their multiple diets.

More carefully designed studies to investigate the levels of PFASs in matched components of the aquatic system (sediments, water, invertebrates, fish), as well as those in fish-eating birds and crocodiles need to be carried out to allow critical assessment of bioaccumulation in the food web, as well as the health risk posed by PFAS to birds and humans who feed on the different species. Bigger sample sizes must be targeted in such studies to allow generalizations be made from these findings.

4.6. Soils and crops

Literature on the levels of PFASs in soils and crops in Africa is very limited. However, the data available shows that the levels are low and vary widely among locations with industrialized areas showing higher levels than the less industrialized ones. Generally, the use of sewerage sludge as soil conditioners and/or fertilizers in agricultural fields, as well as direct release of wastewater from WWTPs into the environment, and urban and/or industrial areas, or its use for irrigation are the major sources of PFAS accumulation in plants (Ghisi et al., 2019). Several studies in Africa have highlighted the need for environmental protection agencies in African countries to closely monitor waste management practices as these are highly influencing the levels of PFASs in the environment.

In West Africa, a recent study by Ibor et al. (2020) revealed that PFASs are present in soils around solid waste dumpsites in Calabar, Nigeria, at mean levels of 0.05, 1.4, 0.6, 2.3, 1.0, 5.0, 0.6 and 2.2 ng g^{-1} dw for PFBS, PFOS, PFHpA, PFOA, PFNA, PFDA, PFUnDA and PFDoDA, respectively. The study suggests that the poorly managed solid waste dumpsites are responsible for exposing the environment to PFASs. In another study by Rankin et al. (2016), PFASs were ubiquitously detected in each of the five soil samples collected from five different locations in Africa. The samples were collected from Cameroon (n =2), Mapunguwe National Park in South Africa (n =1), Nigeria (n =1) and Mabira Forest Reserve in Uganda (n=1). PFCAs and PFSAs were detectable in ranges of 0.12-1.49 and <LOQ-0.144 pg g^{-1} dw. These levels were found to be lower compared to the levels detected in soils sampled from North America and Asia in the same study. For the samples from Africa, the sample size (n=1 for each location) was not sufficient to allow for statistical comparisons. However, the fact that PFASs were detected in samples from remote locations in conservation areas such as Mapunguwe National Park and Mabira Forest Reserve highlights the importance of long-range transport of PFASs as another source of the pollutants on the African continent.

Mudumbi et al. (2014) reported levels of PFOA in riparian reeds of three rivers (Diep, Salt and Eerste Rivers) which go through highly industrialised areas in Western Cape (South Africa) as 11.7-38 ng g^{-1} dw. In a recent study, Mudumbi et al. (2019) showed that PFOA, PFOS and PFBS can bioaccumulate in tissues of *Tagetes erecta L*. which is a known medicinal plant in South Africa. The authors reported that the mean levels of the pollutants in the leaves of *Tagetes erecta L*. plants which were irrigated by PFAS-contaminated water from Salt River were 94.8, 5.03 and 1.44 ng g^{-1} dw, respectively. The study suggested that medicinal plants may be a minor exposure pathway of PFASs into humans. In another study, Dalahmeh et al. (2018) evaluated the concentrations of 26 PFASs (PFSAs and PFCAs) in soils and crops such as yams, maize and sugarcane grown in Nakivubo wetland and along the Lake Victoria shoreline, Uganda. PFASs occurred at 1.7-7.9 ng g^{-1} dw in soil, 0.16 ng g^{-1} dw in maize cobs and 0.38 ng g^{-1} dw in sugarcane stems. While the dominant compound was PFOS in soil, PFHpA and PFOA dominated the PFAS profile in the different plant tissues, reflecting PFAS-specific partitioning behaviour in the different matrices.

The studies by Mudumbi et al. (2014) and Dalahmeh et al. (2018) emphasized that uptake of PFASs from contaminated soils may lead to the accumulation of the pollutants in wetlands, as well as in plants. This uptake of PFASs from soils into plant tissues is clearly documented in literature (Lechner & Knapp, 2011; Stahl et al., 2009; 2013). The process is known to be dependent on several factors such as the PFAS levels in soil (Stahl et al., 2009), soil organic matter content, and the chain length of the PFAS. For example, in their study, (Stahl et al., 2009) showed that PFOA had higher accumulation in plants than PFOS, while the latter had higher concentrations in soil. Other studies (Müller et al., 2016; Wen et al., 2013) have showed that PFASs with longer carbon chain lengths (>C7) tend to be restricted to roots while those with shorter chain lengths are more easily absorbed and translocated. More investigation needs to be carried out in the case of Africa to ascertain whether uptake of PFASs by plants from contaminated fields may influence the levels of the pollutants in foods consumed by humans.

4.7. Marketed foods and food packaging materials

Reports on the occurrence of PFASs in food samples in Africa is very limited due to lack of total diet studies. One study (Vaccher et al., 2020) determined levels of PFASs in fatty foods (eggs, fish, meat, milk/dairy products, nuts/seeds, oil/fat) collected from Benin, Cameroon, Mali and Nigeria in 2017. Generally, PFASs were undetectable in eggs, milk/dairy products, oil/fat, nuts/seeds. In beef samples from Nigeria, only PFOS and PFNA could be quantified at levels of 0.1 ng g^{-1} ww and 0.03 ng g^{-1} ww, respectively, while no other PFASs could be quantified in beef samples from other areas. PFOS occurred in smoked fish at levels of 0.02-10.4 ng g^{-1} ww, with samples from Mali containing exceptionally higher levels compared to the samples from the other countries. The authors noted that the high levels of PFOS in samples from Mali could be a result of application of pesticides such as EtFOSA which can transform into PFOS and PFOA. The results of the study showed that, although the levels of PFASs in food in Africa are still low, upper-bound levels in fish were comparable or lower to those in Europe (Vaccher et al., 2020). In addition, the levels of PFASs were variable among countries due to differences in pollution sources and may be dependent on regional agricultural practices.

Although the levels detected by Vaccher et al. (2020) were low, differences in dietary habits and social-economic status/living conditions between people in urban and rural areas could be responsible for the different human exposure levels to PFASs. For instance, many people especially those in urban centres in Africa use food packaging materials on a regular basis and could potentially be exposed to PFASs and/or their precursors through these materials, as observed elsewhere (Muncke et al., 2017; Rosenmai et al., 2016; Susmann et al., 2019; Trier et al., 2011). In addition, PFAS precursors such as FTOHs and PAPs enter foods when these packaging materials are heated at oven temperatures (Fengler et al., 2011; Susmann et al., 2019).

To date, only one study in Africa (Shoeib et al., 2016) has evaluated human exposure to PFASs through food packaging materials. Low levels of PFASs and their neutral precursors (PAPs) were detected in food packaging materials in Egypt. In the study, 6:2 monoPAPs and 8:2 monoPAPs were detected in only three samples of packaging materials a French fries cardboard box and two sandwich wrapping papers, respectively. PFAAs (PFCAs and PFSAs) were detected at median levels of 2.4, 0.5, 0.22, 0.32, 0.13 and 0.29 ng g^{-1} for PFOA, PFHxA, PFHpA, PFNA, PFDA and PFOS, respectively. The authors also observed an increase in levels of PFCAs for processed popcorn and pasta containers compared to the unprocessed ones. However, the levels of PFASs reported in this study were lower than those detected in food packaging materials in developed countries such as in Catalonia, Spain (Moreta & Tena, 2013).

It should be noted that the African continent is endowed with many varieties of foods of plant and animal origin. However, given the poorly regulated nature of the food and food packaging products on the African market, these items could be exposing many people to PFASs. In addition, a number of raw food products (fruit, vegetables, nuts, etc.) that are sold in rural African markets come directly from the fields. Therefore, any PFAS exposure through long-range transport or contaminated irrigation water would not be controlled through regulations that would be in place for produce to be sold in markets in urbanised regions or Europe / North America. Therefore, more market-based surveys, as well as assessment of levels of PFASs in food contact materials across Africa are needed to evaluate the levels of human exposure.

4.8. Indoor dust and ambient air samples

4.8.1. Indoor dust

Inhalation of contaminated dust is a major exposure route of humans to organic pollutants such as PFASs (Beesoon et al., 2012). For instance, although carpets have historically been treated with PFAS and/or their precursors as oil and water-repellents, these carpets continue to be used in homes, offices and cars. Measurements of PFASs in dust samples from such microenvironments have been carried out elsewhere in the world (Beesoon et al., 2012; de la Torre et al., 2019; Eriksson & Kärrman, 2015; Tian et al., 2016). To date, only one study has investigated the occurrence of the pollutants in indoor dust. In Egypt, Shoeib et al. (2016) determined levels of ionic and neutral PFAS precursors in dust from microenvironments (homes, cars and workplaces) in Cairo, Egypt. In home dust, ∑PFAAs (PFHxA, PFOA, PFNA, PFDA, PFBS, PFHxS, PFOS, FOSA) ranged from 0.23 to 14.1 ng g^{-1}, while levels of ∑FTOHs (6:2 FTOH, 8:2 FTOH and 10:2 FTOH) were 1.09-25.7 ng g^{-1} and ∑FOSA/FOSE (MeFOSA, EtFOSA, MeFOSE, EtFOSE) were <0.02-24.7 ng g^{-1}. The authors also reported levels of PFAS precursors (FOSA/FOSE and FTOHs) as 4.94-30.9 and 3.04-21.4 ng g^{-1} in workplace dust and car dust, respectively.

The study showed that, similar to what has been reported worldwide (Jian et al., 2017), FTOHs and FOSA/FOSE levels are higher in indoor dust than PFASs. FOSA/FOSE and FTOHs are important components in manufacturing paints, carpets and cleaning agents, as well as for impregnation onto furniture (Jian et al., 2017). The levels of PFASs and their precursors in dust samples in Africa were less than those in developed countries such as in Boston, USA (Fraser et al., 2013), Oslo, Norway (Haug et al., 2011), Vancouver, Canada (Shoeib et al., 2011) and Brisbane, Newcastle and Sydney, Australia (Goosey & Harrad, 2011). However, levels of FTOHs in this study were unexpectedly higher than those reported in Catalonia, Spain (Jogsten et al., 2012).

The occurrence of PFASs and their precursors in dust samples in Egypt suggests that PFAS-treated carpets are imported and used in Africa. It is also important to note that infants, especially those at the crawling stage, tend to experience higher uptake doses of pollutants through dust inhalation than adults and teenagers because of their constant hand-to-mouth activities (Anderko & Pennea, 2020). Moreover, their metabolic systems are not fully developed which might lead to more adverse effects (Zeng et al., 2019). Literature reports significant contributions by indoor dust contamination to human exposure to organic pollutants such as organochlorine pesticides, polychlorinated biphenyls (PCBs) and flame retardants (Ali et al., 2013; Bräuner et al., 2011). In many cases, such contamination has been linked to adverse health effects, for example, residential exposure to PCBs from carpet dust conferred a two-fold risk of childhood leukemia in California (Ward et al., 2009). Therefore, dust contamination is potential source of human exposure to PFASs in Africa. The issue warrants more investigation to assess the human health risks associated with this PFAS exposure in the different parts of Africa.

4.8.2. Ambient air

Literature on the occurrence of PFASs in ambient air in Africa is limited. Using high volume active air samplers and XAD resin-based air samples, Gawor et al. (2014) measured levels of neutral PFASs in the atmosphere of Botswana. Levels of 6:2 FTOH, 8:2 FTOH, 10:2 FTOH, MeFOSA, EtFOSA and MeFOSE were <0.54, 0.7-7.1, 0.59-7.4, <0.05-0.15, <0.06-0.3, <0.42-0.77 and <0.53 ng m^{-3} from sites near Okavango Delta, Botswana. The reported levels were lower than the levels measured elsewhere in the world such as San Antonio de Belen, Costa Rica and Ontario, Canada (Gawor et al., 2014), Büsum, Germany (Zhen Wang et al., 2014), Antarctic Peninsula (Wang et al., 2015) and Asian environments (Li et al., 2011). In a similar study, Jahnke et al. (2007) determined neutral, volatile PFASs in high-volume air samples collected on board the German research vessel Polar Stern during cruise ANTXXIII-1 between Bremerhaven, Germany (53° N) and Cape town, South Africa (33° S). The authors observed a decreasing concentration gradient from the European continent towards the less industrialised regions in Africa. For instance, south of the equator, trace amounts of FTOHs and FOSAs (maximum of 14 pg m^{-3} for 8:2 FTOH) were detected.

The results of these two studies showed that airborne PFASs are mainly restricted to the northern hemisphere (maximum concentration of 190 pg m^{-3} for 8:2 FTOH; Jahnke et al. (2007)) compared to the southern hemisphere where one third of Africa is located. In addition, levels of these pollutants were lower in remote areas than in urban and industrialized areas suggesting localized sources in urban/industrialized areas. Such sources might include emissions from industries, WWTPs and waste management areas such as municipal landfills. Furthermore, the detection of PFASs in remote areas is evidence for possible contribution of atmospheric deposition of semi-volatile PFASs to the total load of PFASs in the environment.

4.9. PFASs in wildlife

In South Africa, Bouwman et al. (2015) reported the median levels of ∑PFASs (PFOS,PFOA, PFNA, PFDA and PFUdA) as 12 ng g^{-1} ww and 12 ng g^{-1} ww for the eggs of African penguin from Robben Island and Bird Island, respectively. In another study in South Africa, Fredriksson (2018) determined PFASs in 10 eggs of African Darter (*Anhinga rufa*) along the Vaal River/Orange River as 2100 ng g^{-1} ww. Reported concentrations were higher than those reported in Sweden (670 ng g^{-1} ww), Iceland (72 ng g^{-1} ww) and Faroe Islands (67 ng g^{-1} ww) from the same study. The most prevalent group was PFOS (constituted 96% to the PFASs) at concentrations of up to 2,000 ng g^{-1} ww. The PFOS levels in eggs of *A. rufa* from South Africa were higher than those reported in eggs from Germany (540 ng g^{-1} ww) (Rüdel et al., 2011), Canada (83 ng g^{-1} ww) (Miller et al., 2015), China (20.4 ng g^{-1} ww) (Yuan Wang et al., 2008), Sweden (590 ng g^{-1} ww), Faroe Islands (39 ng g^{-1} ww) and Iceland (25 ng g^{-1} ww) (Fredriksson, 2018). It is not entirely clear why the eggs of birds in South Africa showed higher levels of PFASs than those from other countries. However, one possible reason for this observation is the poor disposal of PFAS containing consumer products in Africa, which can expose the birds to particle-bound organic pollutants due to their free-range feeding nature (Ssebugere et al., 2019).

Bouwman et al. (2014) determined eight PFASs (PFHxS, PFOS, PFOA, PFNA,PFDA, PFUdA, PFDoA, PFTriA) in eggs of Nile Crocodiles (*Crocodylus niloticus*) from a site where high crocodile mortalities (Olifants River Gorge) occurred and a reference site, both located in Kruger National Park, South Africa. The PFASs varied from 1 to 25 ng g^{-1} ww. Levels of PFASs in eggs from Gorge were significantly higher than those from reference site. In another study, Christie et al. (2016) investigated the occurrence of PFAAs in plasma of Nile crocodiles from the reference site. The median concentration of PFOS (13.5 ng g^{-1} ww) was high compared to other PFASs compounds. The authors observed differences in PFAA accumulation between crocodile sizes and/or sexes when crocodile length and PFAA load were correlated. Further to this, statistical analysis showed significant differences between the different sites ($p < 0.05$). The highest median concentrations of PFOS were observed at Flag Boshielo Dam (50.3 ng g^{-1} ww) while the levels at other sites were <14.0 ng g^{-1} ww. The two studies by Bouwman et al. (2014) and Christie et al. (2016) show that PFOS was the dominant compound in the eggs and plasma samples. However, the levels of PFAAs were higher in crocodile plasma than in their eggs.

Lesch et al. (2017) analysed eight PFASs (PFOS, PFOA, PFNA, PFDA, PFUnA, PFDoA, PFHxA and PFHxS) in adult male dragonflies from six sites in South Africa. The levels in dragonflies collected from industrial areas in the south (ΣPFASs = 9.3 ng g^{-1} ww) were a magnitude higher than those that were close to farming areas in the north (0.32 ng g^{-1} ww). The highest PFOA concentrations were from Bloemhof Dam (ΣPFASs = 21 ng g^{-1} ww).

In this review, all the studies found on PFSAs in wildlife have been done in South Africa. These studies suggest the ubiquity of PFASs in the environment. Research into the occurrence of the pollutants in other parts of Africa, as well as follow-up studies in South Africa are warranted to establish clear spatial and temporal trends. In addition, the only studies available have focused on eggs of wild birds in South Africa, hence, the status of PFASs in birds' eggs in other African countries cannot be ascertained. Furthermore, although Vaccher et al. (2020) reported that PFASs were below detectable limits in chicken eggs in West Africa, it is not clear if PFASs exist in eggs of human dietary importance (such as those of chicken) in other countries in Africa, as is the case elsewhere outside Africa (Jian et al., 2017). This issue is of particular concern because, in Africa, chicken eggs have shown detectable levels of other organic pollutants such as PCDD/Fs and dioxin-like PCBs (Ssebugere et al., 2019), PBDEs (Brits et al., 2016) and organochlorine pesticides (Polder et al., 2016), and positive correlations have been established between consumption of chicken eggs and human body burdens of organic pollutants in Uganda (Matovu et al., 2019). These studies suggest that edible birds' eggs could be a possible dietary exposure route of humans to organic pollutants. More monitoring in this area needs to be done in order to ascertain the potential exposure of humans to PFASs through edible eggs in other African countries.

4.10. PFASs in human samples

Human samples such as blood serum can be used as a marker to assess human exposure to organic pollutants. Analysis of placenta samples and cord blood serum, and breastmilk can specifically provide information on contaminant transfer to infants in the prenatal period and postnatal periods, respectively (Matovu et al., 2020; M. H. B. Müller et al., 2019). To date, no study has been carried out on levels of PFASs in breastmilk or placenta samples while only two studies have reported the levels of PFASs in blood serum in Africa. Hanssen et al. (2010) determined PFASs in maternal serum and cord blood of participants belonging to the black ethnicity in South Africa. In terms of median concentrations, PFOA was the dominant compound in cord blood (median:1.3 ng mL^{-1}), followed by PFOS (0.7 ng mL^{-1}) and PFHxS (0.3 ng mL^{-1}).

The median cord blood levels of PFASs in this study were lower than those reported in more industrialized countries such as Japan (PFOS: 5.2, PFOA: 1.4 ng mL^{-1}) (Miura et al., 2018), Norway (PFOS: 13, PFOA: 2.2 ng mL^{-1}) (Starling et al., 2014) and China (PFOA: 6.96, PFOS: 2.48 ng mL^{-1}) (Wang et al., 2016). These studies highlight that prenatal exposure to PFASs is lower in African populations. However, exposure to PFASs might be influenced by maternal living conditions such as diet, as seen in other countries. For instance, Wang et al. (2016) showed that in China, cord blood levels of PFOS, PFNA and PFDA had a positive correlation with maternal fish consumption. More research is warranted in Africa.

Hanssen et al. (2010) reported that the levels of PFASs in maternal blood were higher for PFOS (median; 1.6 ng mL^{-1}), than PFOA (1.3 ng mL^{-1}) and PFHxS (0.5 ng mL^{-1}). Further to this, higher PFAS concentrations in urban and semi-urban communities were reported compared to the rural settings for both maternal and cord blood sera. In another study, Müller et al. (2019) reported lower median levels of six PFASs (PFOA, PFNA, PFDA, PFUdA, PFHxS and PFOS) in maternal serum samples from Tanzania. ∑PFASs ranged from 0.18 to 1.54 ng mL^{-1} with a median value of 1.18 ng mL^{-1}. PFOS (median; 0.5 ng mL^{-1}) dominated the profile of the pollutants followed by PFOA (0.21 ng mL^{-1}), PFNA (0.17 ng mL^{-1}) and PFDA (0.14 ng mL^{-1}). However, the median maternal serum PFOS concentrations in Tanzania and South Africa were lower than those reported in the plasma of mothers from the industrial city of Norilsk (Arctic Russia) (median;11.0 ng mL^{-1}) (Hanssen, 2013), France (3.065 ng mL^{-1}) (Cariou et al., 2015) and Japan (5.2 ng mL^{-1}) (Itoh et al., 2016). The low PFOS levels in maternal blood samples from Africa can be attributed to the phase out of the compound in the early 2000s and to the low historical exposure to the compound in South Africa (Hanssen et al., 2010).

For PFOA, the maternal serum concentrations reported by Müller et al. were lower while those of Hanssen et al. (2010) were in the same range of data as those reported in Beijing, China (1.15 ng mL^{-1}) (Yang et al., 2016) but lower than those reported in Ohio, USA (3.10 ng mL^{-1}) (Kato et al., 2014). The concentrations of PFOA in human blood by Yang et al. (2016) were attributed to the industrial discharge of PFOA from manufacturing plants and the use of a number of consumer products containing PFOA. The latter could be the case for Africa since the compounds are not known to be manufactured anywhere on the continent (Müller et al., 2019). Therefore, regional differences in PFAS exposure could be due to differences in living habits of populations, although the exact sources and pathways of exposure are still unclear. More studies regarding the levels and determinants of PFASs exposure in humans need to be carried out in Africa.

5. Future research directions and implications for stakeholders in Africa

Reviewed literature showed that analysis of PFASs on the continent have been carried out in laboratories only in Kenya and South Africa, while for the rest of the studies, analysis was carried out by collaborators in Asia, Europe and USA. This is possibly due to limited logistical support and as well as technical problems facing African institutions. For instance, PFASs have limited volatility and form unstable derivatives which limits the use of gas chromatography/mass spectrometry (GC-MS) as an analytical technique (Macheka-Tendenguwo et al., 2018). In addition, PFASs lack chromophores hence ultraviolet–visible (UV-VIS)-based methods cannot be used to detect them. However, many laboratories in Africa have GC/MS and UV-VIS as the most common analytical equipment, since the laboratories lack adequate research funds to acquire advanced instrumentation like ultra-performance liquid chromatography/tandem mass spectrometry (UPLC-MS). Besides, expertise in operating advanced equipment is still limited

(Gwenzi & Chaukura, 2018). These factors make analysis of PFASs challenging in many laboratories in Africa.

The prevalence of PFASs in the African environment is exacerbated by poor governance, weak legislation, as well as illegal and uncontrolled importation of products (such as carpets and food packaging materials) containing these compounds from the developed world. Therefore, the relevant role-players in Africa (academics, policy-makers and industrialists) need to advocate for, strengthen and implement existing policies, and to develop capacities in African laboratories including sufficient quality control/quality assurance measures to be able to handle PFASs and related compounds on routine basis for example, to remove background contamination by laboratory materials such as PTFE. It is also time for environmental management authorities and trade ministries in Africa to document the PFAS containing products and dosages imported into the different African countries. In creating these profiles, African governments should consider engaging commercial research laboratories to do some of the analysis since these laboratories are generally better equipped than laboratories in public institutions. In addition, the protocols for dissemination of findings should be improved. For instance, documents must be made freely accessible to environmental campaign organisations, trade organisations and the general public to increase their awareness of the prevalence and impact of organic pollutants, such as PFASs, in the environment. Such community engagement will improve environmental health literacy and help to influence waste management practices. Consequently, it will help to reduce human and environmental exposure to PFASs and other pollutants.

African researchers and governments also need to prioritize the limited resources for testing in areas close to the hotspot areas identified as PFAS sources such as WWTPs, municipal ponds and landfills, and industrialized areas in the different countries. In addition, regional collaborations between the well facilitated laboratories in South Africa and those in other African countries are encouraged. Profiling the PFAS levels in these areas from time to time will help in evaluating the contribution of each source to the total PFAS load in the environment, as well as the health risks/effects induced by these compounds, and to develop research capacity in many more African countries.

Despite the scarcity of systematic monitoring studies on PFAS levels in the environment of Africa in many countries, reviewed data shows that the exposure of the general population is highly variable between different geographical areas and even within countries, in some cases a result of social-economic strata in the populations. For instance, results by Hanssen et al. (2010) indicated that urban communities in South Africa are more susceptible to PFASs exposure than the rural communities since they (urban communities) tend to have more access to modern consumer products such as stain resistant furniture, carpets, clothing and cosmetics which contain PFASs. Therefore, continuous monitoring to understand the origin, spatial and temporal trends, as well as the fate of these chemicals in environmental and human matrices from Africa needs to be done.

Furthermore, most of the studies reviewed in this paper are exposure-based studies, focused on determining the nature and levels of legacy PFASs in the different human and environmental media. Only very few have studies have gone further to investigate the possible ecological and human health risks associated with exposure to the pollutants. Therefore, there is no evidence to plausibly link human chronic or acute responses to exposure to PFASs in Africa. For instance, studies have suggested that respiratory infections are a leading cause of child mortality in sub-Saharan Africa (Källander et al., 2005). A case in point here is Uganda where at least 27% of infants are diagnosed with respiratory infections (Källander et al., 2008). Whether or not these infections are related to PFAS exposure needs investigation. Therefore, in addition to determining the nature and levels of PFASs in Africa, researchers should evaluate the potential ecological and human health risk and dose-effect relationships arising from exposure to PFASs at the reported levels of contamination.

Further still, research into the current state of new PFASs such as 2-[(1,1,2,2,3,3,4,4,5,5,6,6,6-tridecafluorohexyl)oxy]-1,1,2,2-tetrafluoroethane sulfonic acid potassium salt (F-53), dodecafluoro-2-methylpentan-3-one (3M™ Novec™ 1230), perfluoro[(2-ethyloxy-ethoxy)acetic acid] ammonium salt, or 3H-perfluoro-3-[(3-methoxy-propoxy)propanoic acid] ammonium salt (ADONA) which are currently used as alternatives to PFOA and PFOS should be done in Africa and across the world (Wang et al., 2019). This is because little is known about the health risks they pose in humans, yet they are already present in human samples (Fromme et al., 2017).

In addition, due to the ever changing and increasing amounts and recipes of PFASs, including the already known and the yet-to-be identified fluorinated compounds, researchers in Africa should consider adopting established bulk methods to measure total organic fluorine since determining individual compounds (as is the case currently) may underestimate the actual contamination levels. Such methods include combustion ion chromatography (Miyake et al., 2007), particle-induced gamma ray emission spectroscopy and fluorine nuclear magnetic resonance spectroscopy (McDonough et al., 2019). Due to the lack of facilitation in laboratories in Africa, adopting these methods may not be easy. However, where possible, it is achievable through the collaborations that researchers in less developed African countries have established with researchers in South Africa, as well as other developed countries in Asia, Europe and USA.

5. Conclusions

This study reviewed literature on PFASs in human and environmental matrices from Africa. However, due to lack of data from many countries, the results presented here must be interpreted with caution as they may not represent the actual state of affairs regarding PFASs in Africa. In addition, the levels of PFASs are mainly in the range of nanograms for most samples from Africa. In effect, the actual levels of PFASs in Africa may be of a wider range (especially for drinking water) than those reported by researchers because the instruments used have higher detection limits. Lower instrumental detection limits of picograms or even upper femtogram levels can be achieved by instrumental techniques such as UPLC (Gebbink et al., 2017). Furthermore, although the reported pollutant levels in the majority of matrices are still low compared to more developed regions in the world, the results provide evidence for decision-makers to establish priorities to reduce any further entry of the PFASs into the environment of Africa, because the effect of long-term exposure to these chemicals by humans and the aquatic ecosystem cannot be ascertained (Guelfo et al., 2018).

The reported emission sources of PFASs were mainly point sources such as solid waste landfills, effluents from industries and WWTPs, and diffuse sources like surface runoff. Evaluation of global bans on PFASs could not be done due to lack of literature reporting on temporal trends. This necessitates that adequate emission control strategies be implemented by scientists, governments, purchasing organisations, retailers, and consumers to stop possible emission of PFASs into the environment of Africa. In addition, more research into the levels and sources of the pollutants in the different human and environmental media in many countries in Africa should be carried out. This will help in the evaluation of possible health risks and the effectiveness of global bans on PFASs.

Author Contribution Statement

Patrick Ssebugere: Conceptualization, Methodology, Formal analysis, Investigation; Methodology, Writing - Original Draft and Funding acquisition. **Mika Sillanpää**: Writing - Review and Editing, Supervision, Funding acquisition. **Henry Matovu**: Conceptualization, Writing - Review and Editing. **Zhanyun Wang**: Conceptualization, Writing - Review and Editing. **Karl-Werner Schramm**: Writing - Review and Editing, Supervision, Funding acquisition. **Solomon Omwoma**: Conceptualization, Writing - Review and Editing. **William Wanasolo**: Conceptualization, Writing - Review and Editing. **Emily Chelangat Ngeno**: Conceptualization, Writing - Review and Editing. **Silver Odongo**: Conceptualization, Writing - Review and Editing.

Conflict of interest

The authors declare no conflict of interest.

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Table 1: Comparison of PFOA and PFOS levels (ng g^{-1} dw) in sludge samples from WWTPs around the world

Country	Number of WWTPs	Year of sampling	PFOA	PFOS	Major contributor of PFAS	Reference
Africa						
Kenya	9	2013	0.117-0.673	<0.02-0.683	Discharges from hospitals	Chirikona et al. (2015)
Nigeria	10	2012	0.01-0.416	<0.01-0.54	Hospital sewage	Sindiku et al. (2013)
North America						
USA (New York)	3	2007	8-20	32-77	Domestic sewage	Yoo et al. (2009)
USA (San Francisco Bay)	9		<6-29.4	14-2610	Domestic sewage	Higgins et al. (2005)
USA (Georgia)	1	2005	7-130	<2.5-77	Domestic and commercial discharges	Loganathan et al. (2007)

USA (Kentucky)	1	2005	8.3-219	8.2-993	Domestic and commercial discharges	Loganathan et al. (2007)
Asia						
China	16	2004	<0.8-4,780	<0.5-5,383	Municipal discharges	Guo et al. (2008)
Singapore	2	2006-2007	13.1-702	5-69	Industrial discharges	Yu et al. (2009)
Europe						
Switzerland	20	2008	<1.5-20	20-600	Municipal discharges	Sun et al. (2011)
Spain (Catalonia, Aragón, and Navarra)	12	2011	<0.1-0.69	<0.12-4.45	Industrial discharges	Gomez-Canela et al. (2012)
Germany (Bravia)	3	2011	0.59-1.21	2.08-4.45	Industrial discharges	Gomez-Canela et al. (2012)
Australia						
Australia	19	2017	<LOD-	<LOD-	Industrial	Coggan et al.

	25	90	discharges	(2019)

<LOQ = not quantifiable

Table 2: Levels of PFAS in surface and pore water (ng L^{-1}) from different water bodies in Africa

Country	**Year of sampling**	**Sampling area**	**PFOA (range/mean)**	**PFOS (range/mean)**	**∑PFAS (range/mean)**	**Dominant PFAS**	**Notable source of water contamination**	**Reference**
Surface water								
Kenya	2006-2007	Lake Victoria Gulf	<0.4-11.7	<0.4-2.53		PFOA	Urban and industrial wastewater	Orata et al. (2009)
	2015	River Sosiani, Eldoret	1.6 (sparsely populated areas); 8.8 (densely populated areas)		58.8 (sparsely populated areas); 109.4 (densely populated areas)	PFDoDA, PFNA, PFDA	Domestic and industrial discharges	Shafique et al. (2017)
Ethiopia	2014	Lake Tana	<0.28-0.69	<0.05-0.22	0.073-5.6 (2.9)	PFBA, PFHxA	Wastewater from Bahir Dar	Ahrens et al. (2016)
Uganda	2015	Lake Victoria and Nakivubo	2.4	1.6	8.5-12 (Nakivubo Channel); 1.0-2.5	PFBS	Urbanization and industrial activities in	Dalahmeh et al. (2018)

		Channel			(Lake Victoria)		the lake catchment area	
Nigeria	2014	Seven rivers across the country		1.7-16.2		/	Discharges from industrial and urban run-off	Ololade (2014)
	2016	Seven rivers across the country	0.8-2.8 (1.7)	3.9-10.1 (6.8)		PFOS	Discharges from industrial and domestic wastewater; agricultural run-off	Olalade et al. (2018)
South Africa	2014	Plankenburg River	12.8-62.6	<0.06-12.4	62.3-186.4	PFHpA, PFBA, PFOA	Domestic and industrial discharges containing electronic waste, fire-fighting chemicals and petrochemic	Fagbayigbo et al. (2018)

					als	
2014	Vaal River	0.6-4.6	0.4-35.7	PFOS	Mining industries, and wastewater treatment plants in the river basin	Groffen et al. (2018)
2011	Diep, Western cape	1.7-314	<LOD- 183	PFOA	Wastewater treatment plants; run-off from municipal landfills	Mudumbi et al. (2013)
2011	Salt River	0.7-390	<LOD-47	PFOA	Wastewater treatment plants, run-off from municipal landfills	Mudumbi et al. (2013)
2011	Eeste River	3.4-146	<LOD-23	PFOA	Wastewater treatment plants; run-off from municipal	Mudumbi et al. (2013)

							landfills	
	2012	Olifants River	<LOQ	<LOQ	<LOQ	/	/	Verhaert et al. (2017)
	2016	uMvoti Estuary	416-1089	<14.6		PFOA	Pulp and paper mill, urbanization, agriculture, sewage treatment plants	Fauconier et al. (2019)
	2016	aMakikulu Estuary	142-310	<14.6-54.2		PFOA	Agriculture, atmospheric deposition	Fauconier et al. (2019)
Ghana	2015	Pra and Kakum Rivers	86.2-321 (Pra River), 1.78-301 (Kakum River)	95.4-277 (Pra River), 77.2-163 (Kakum River)	398 (Pra River), 281 (Kakum River)	PFOA, PFOS	Poor disposal of PFAS-containing products	Essumang et al. (2017)
Pore water								
Nigeria	2016	Several rivers across the country	4.7-11.1 (1.7)	10.9-20.4 (6.8)		PFOS	/	Ololade et al. (2018)

<LOD = below quantifiable limits

Table 3: Levels of PFASs in suspended solids (ng g^{-1} dw) and sediments (ng g^{-1}) in water bodies in Africa

Country	Year of sampling	Sampling area	PFOA (range/mean)	PFOS (range/mean)	∑PFAS (range/mean)	Reference
Suspended solids						
South Africa	2011	Suspended solids from 3 major rivers in Western cape	16 (Diep River) 14 (Salt River) 28 (Eerste River)	<LOD (Diep River) 5 (Salt River) 26 (Eeste River)		Mudumbi et al. (2013)
Sediments						
South Africa	2011	Western Cape rivers	10.7-772 (Diep River) 38.6-187 (Salt River) 15.2-193 (Eerste	2.53-121 (Diep River) <LOD-19.9 (Salt River) 0.72-75.1 (Eerste		Mudumbi et al. (2014a)

			River)	River)		
South Africa	2014	Plankenburg River	0.14-0.33	<0.02-0.7	1.49-2.12	Fagbayigbo et al. (2018)
	2014	Vaal River	<LOQ	<LOQ-2.36	<LOQ-2.36	Groffen et al. (2018)
	2016	uMvoti Estuary	0.26-0.91	0.05-0.99		Fauconier et al. (2019)
	2016	aMakikulu Estuary	0.83-1.73	0.05		Fauconier et al. (2019)
Ethiopia	2014	Lake Tana	/	<0.01-0.089	0.22-0.55 (0.30)	Ahrens et al. (2016)
Kenya	2008	Winam Gulf, Lake Victoria	<1-24.1	<1-4.0		Orata et al. (2011)
	2008	Rivers in Lake Victoria	1.4-99.1	<1-57.5		Orata et al. (2011)

		basin			
Nigeria	2014	Seven rivers across the country		<1.64-10.3	Olalade (2014)
	2016	Seven rivers across the country	0.9-4.6 (2.2)	3.6-10.8 (7.1)	Olalade et al. (2018)

<LOQ = not quantifiable; <LOD = below detectable limits

Table 4: Levels of the different PFASs (ng g^{-1} ww) in fish samples in Africa compared to those reported elsewhere

Sampling area	Year of sampling	Fish species	N	PFOS		PFOA		PFNA		PFDA		PFUdA		PFDoA		Ref.
				M	L	M	L	M	L	M	L	M	L	M	L	
Africa																
Winam Gulf, Lake Victoria, Kenya	2007	*Oreochromis niloticus*	25	0.9-12.4	1.5-23.7											Orata et al. (2008)
	2007	*Lates niloticus*	25	0.9-10.5	1.4-35.7											Orata et al. (2008)
Lake Tana, Ethiopia	2014	*O. niloticus*		<0.03-0.04				<0.03-0.47		<0.05-1.2		<0.01-1.3		<0.01-0.16		Ahrens et al. (2016)
unguja, Pemba, Mtwara, Tanzania	2016	Milk fish	47	<0.42		<0.42		<0.42		<0.42		<0.42		<0.42		Mwakalapa et al. (2018)
Kalk Bay harbor, SA	2017	Several	24			<0.02-13.9*	<0.02-61*	15.9-58.5*	23.1-61*	28.3-78.9*	20.1-159*	<0.02*	<0.02*			Ojemaye and Petrik (2019)
Olifants river Basin, SA	2012	Several	51	0.15-2.7	0.99-2.7	0.11-0.42	0.21-0.61	<LOQ-0.14	<LOQ-0.35							Varhaert et al. (2017)
Vaal River, SA	2014	Several	33	<0.12-	<0.12-	<0.07-	<0.07 -	<0.54	<0.54-	<0.71-	<0.71-	<0.95	<0.95-	<0.07-	<0.07-	Groffen et al. (2018)

				45.7	461	0.4	2.3		1.7	1.9	8.7		4.7	0.4	1.9	
Loskop Dam, Mpumalanga, SA	2014-2016	*Oreochromis mossambicus*	25	0.32 - 1.65		<0.01-0.04		<0.004-0.07		<0.01-0.213				0.014-0.078		Bangma et al. (2017)
uMvoti Estuary, SA	2016	Several	/	0.18-0.97	7.29-28	0.12 - 0.58	0.17-1.01			<0.71	0.85-2.34			0.15 -0.4	0.77-1.37	Fauconier et al. (2019)
aMatikulu Estuary, SA	2016	Several	/	0.09-2.25	1.49-12.1	0.09 - 0.67	0.54-2.96			<0.71	<0.71-1.9			<0.06-0.2	<0.07-0.61	Fauconier et al. (2019)
Other areas																
Inland lakes, New York, USA	2001-2003	Several	66		9-315		<1.5-7.7									Sinclair et al. (2006)
Lake Vattern, Sweden	2001	Several	25	0.97-23.1		<0.1 - 0.25		0.08 - 0.71				0.12-0.89		<0.08-0.53		Berger et al. (2009)
Baltic Sea, Sweden	2001	Several	25	0.47-3.34		<0.1 - 0.39		<0.08-0.47				<0.08-0.61		<0.08-0.15		Berger et al. (2009)
Lake Varese, Italy	/	*Perca fluviatilis*	10	5.4-17.2		<0.5										Squadrone et al. (2015)
Norwegian harbours	2008-2009	*Gadus morhua*	200		1.5-21.8		<1.8-2.9						<1.5-3.7		<2.4-3.1	Valdersnes et al. (2017)
Etobicoke Creek,	2005	Several	37		219 -	<LOQ			<LOQ		9.52-				24.7-	Oakes et al. (2010)

Canada					350						13.5				33.1	
Lake Taihu, China	2010	Several	52	2.26-20.96		0.38-2.47		0.42-2.86		1.63-10.3		1.27-9.41		0.61-2.62		Xu et al. (2014)
Several rivers, Korea	/	Crucian carp	69		0.18-145		ND-0.33		ND-0.86		0.06-3.48				ND-2.08	Lam et al. (2014)
	/	Mandarin fish	20		1.61-115		0.09-0.33		0.38-5.78		0.38-5.78				0.92-3.17	Lam et al. (2014)
Pearl River Delta region, China	2011-2012	Several	141	<LOQ-79	0.95-1500	<LOQ	<LOQ-4.3	<LOQ-0.15	<LOQ-2.0	<LOQ-5.1	<LOQ-14	<LOQ-2.4	<LOQ-57	<LOQ-0.81	<LOQ-16	Pan et al. (2014)

N= number of samples analysed; /= not available; <LOQ = below quantifiable limits (LOQs are not reported); ND = not detected (Detection limits are not reported); M= muscle tissue; L = liver tissue; * mean ng g^{-1} dw values reported

Declaration of competing interest

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

Graphical abstract

Highlights

- Limited data is available due to limited capacity to monitor PFASs in Africa
- PFASs are present at low levels in human and environmental samples in Africa
- No biomagnification of PFASs could be confirmed in aquatic systems in Africa
- Poor waste management, WWTPs and consumer products are major sources of PFAS
- Policy makers need to prioritize resources for research on PFASs in hotspot areas