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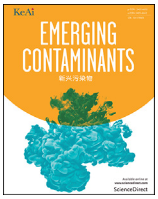

Legacy and emerging organic pollutants in indoor and outdoor environments in Africa: Contamination levels, health risks, and analytical techniques

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ABSTRACT

The World Health Organisation reported that over 20 % of the global disease burden and deaths are caused by environmental factors such as air and dust pollution. Legacy organic pollutants (LOPs) and emerging organic pollutants (EOPs) are among the biggest contributors to this burden. Herein we reviewed literature published for the period 2000–2023 on selected LOPs and EOPs in indoor dust and outdoor air in Africa to understand the occurrence, associated human health risks and common analytical techniques used to identify and quantify these contaminants. The findings revealed generally lower contamination levels for most pollutants than that in most regions outside Africa except for PCBs in outdoor air and OCPs in indoor settled dust. Apart from a few studies for polychlorinated biphenyls (PCBs), most studies reported hazard index (HI) values less than 1 for outdoor air and indoor dust exposure, which suggested negligible non-carcinogenic risks in children and adults. Gas Chromatography and mass spectrometry were the most frequently used analytical techniques for both air and dust probably due to their sensitivity and selectivity in detecting chemical traces at the pg m^{-3} and ng g^{-1} levels. Although the current contamination levels are generally low, their synergistic and cumulative effects may in the long run affect the environment and human health. There is also a paucity of data on most contaminants, especially for PCNs, and PFASs in both outdoor air and dust. Coordinated efforts are needed to limit the trade, importation, and disposal of products containing LOPs and EOPs in Africa.

1. Introduction

Air pollution is one of the major global environmental health risks affecting both living organisms and the environment [1]. The World Health Organisation (WHO) estimated that air pollution kills seven million people annually [2]. Air pollution and related mortalities are highly driven by the presence of legacy (LOPs) and emerging (EOPs) contaminants [3] in indoor environments which have become the subject of intensive scientific research globally [4].

LOPs are chemicals whose characteristics, toxicity mechanisms, and health effects have been investigated and are known [5]. LOPs such as organochlorine pesticides (OCPs), and polychlorinated

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biphenyls (PCBs) are ubiquitous in the environment and they are characteristically persistent, toxic, and have a bioaccumulation potential [6]. Pesticides are substances or combinations of substances intended to prevent, eradicate, or control pests such as fungi, rodents, insects, or undesired plant species posing harm to crops during production and storage [7]. PCBs are a group of synthetic organic chemicals that have been widely used in industrial and commercial applications owing to their chemical stability and insulating properties until they were banned because of their persistence in the environment and associated human health risks and ecological threats [8]. Polychlorinated naphthalenes (PCNs) are a class of compounds formed by replacing one to eight hydrogen atoms with chlorine atom (s) in naphthalene giving rise to 75 chlorinated compounds usually referred to as congeners. In 2015, the Stockholm Convention on persistent organic pollutants (POPs) banned PCNs due to their persistent, hazardous, and bioaccumulation nature [9]. EOPs, on the other hand, are either new chemical substances or those that have existed but whose toxicological profiles, health risks and contamination dynamics are not well understood [10] include per- and poly-fluoroalkyl substances (PFASs), current use pesticides (CUPs), novel brominated flame retardants (NBFRs), and organophosphorus esters (OPEs). PFASs, which are often referred to as “forever chemicals” belong to a wide class of synthetic chemicals that have attracted scientific and policy scrutiny due to their persistence in the environment and potential health impacts. They generally consist of a carbon backbone with fluorine atoms saturating most carbon atoms and at least one functional group, such as carboxylic acid, sulfonic acid, or amine. Due to their strong C-F bonds, PFASs can withstand extreme temperatures and resist degradation, thus, they have widespread household and industrial applications as additives in firefighting foams, adhesives, domestic cookware and food packaging [11].

OPEs are a novel group of semi-volatile organic compounds that are frequently added to various materials used in indoor and outdoor environments to delay the spread of flames [12]. NBFRs are also flame retardants that are either newly or recently observed in the environment or new on the market. NBFRs came into use as replacements for the polybrominated diphenyl ethers (PBDEs) which were banned and added to the list of POPs by the Stockholm Convention in 2009 [13]. Together, EOPs have become a concern to environmental scientists because they represent significant and often unknown ecological and human health risks [14]. Once introduced into the environment, EOPs and LOPs accumulate in dust due to conditions such as the absence of sunlight, precipitation, high temperatures, and microbial activity, which make them persist indoors [15]. Secondly, the chemical characteristics of LOPs and EOPs facilitate their binding affinity for particles and their ability to adsorb onto surfaces such as dust, increasing their environmental distribution. Pyrethroid pesticides, for instance, have low vapour pressures and high partition coefficients for octanol/water (Kow) and water/organic carbon (Koc), which make it easier for them to bind to the dust particle matter and partition into lipids and organic matter [16]. Therefore, understanding the distribution of LOPs and EOPs in indoor and outdoor environments is fundamental in evaluating the associated health risks caused by human exposure to these chemicals [10].

The increasing urban population in Africa has created sizeable markets for consumer goods like textile fabrics, indoor furniture coatings, paper products, cookware, semiconductors, automobiles and other electronics. The consumer goods market is projected to reach US$ 2.1 trillion by 2025 [17] which could exacerbate exposure to the pollutants in these components/items, that are known sources of EOPs [18]. For instance, a significant portion of imported consumer products contain electronic waste (e-waste), a major source of EOPs [19]. Moreover, it is estimated that more than 80 % of the e-waste from developed countries meant for recycling is usually illegally shipped to low- and middle-income countries, particularly those in Africa [20,21]. In 2019, the total amount of e-waste locally produced and imported in Africa was estimated to be between 3.4 and 5.8 metric tonnes, with Nigeria, Ghana, Tanzania, Kenya, Senegal, and Egypt being the main recipients [22]. Moreover, rapid urbanization in different parts of Africa could also potentially contribute to the environmental burden of EOPs and LOPs at levels that could significantly impact human health as well as the environment [23]. Indeed, some studies in Africa have reported the occurrence of LOPs and EOPs in different environmental matrices such as water, sediments, and aquatic organisms [24–26]. Previous reviews have synthesized published literature on the occurrence of these LOPs and EOPs in different matrices in Africa [27–32], and globally [33–35]. However, these reviews did not pay special attention to contamination in indoor dust and atmospheric compartments. The atmosphere plays an essential role in the dispersal of EOPs since it acts as a major global and local diffusion pathway and greatly contributes to the worldwide cycling of volatile and semi-volatile pollutants [36]. Precisely, less attention has been paid on the occurrence of LOPs and EOPs in indoor environments. The indoor environment is particularly important due to the high percentage of time spent indoors [37], and the consequential impact of indoor pollution on human health [38]. As such, there is a pressing need for increased research on the occurrence and impact of EOPs and LOPs in indoor environments.

This study aimed to synthesize the existing literature on the contamination levels of LOPs and EOPs in indoor settled dust and outdoor air in Africa, and to evaluate the associated health risks to humans. Common analytical techniques used for the identification and quantification of selected LOPs and EOPs in both indoor dust and outdoor air were also reviewed. Additionally, the review provided recommendations for future research and policy development regarding the management of these contaminants in indoor dust and outdoor air across Africa.

2. Materials and methods

Following PRISMA guidelines [39], a literature search was performed to identify studies that investigated the occurrence of selected LOPs and EOPs in indoor settled dust, and outdoor air in Africa. The search was performed in Google Scholar, Scopus, Science Direct, Springer Link, and PubMed using the terms “emerging organic pollutants”, OR “emerging organic contaminants”, “legacy organic pollutants”, “indoor dust”, and “outdoor air”. Names of specific pollutants such as “per- and poly-fluoroalkyl substances (PFASs)”, “novel flame retardants”, “organophosphate flame retardants”, “polychlorinated naphthalenes (PCNs)”, “legacy pesticides”, “Organochlorine pesticides”, and “current-use pesticides” were also used in search engines along with: “Africa”, “air”, “dust”, and/or the specific names of individual countries of Africa.

Only records published as journal articles from 2000 to 2023, and in English language were included in this review. A total of 42 studies were identified. Studies from other continents were included in this study for comparison purposes. Units were harmonized across studies to enhance comparison.

3. Results

3.1. Occurrence of selected organic pollutants in outdoor air samples in Africa

The summary of the concentrations of selected LOPs and EOPs in African outdoor air is presented in Table 1.

Table 1
Concentrations of selected pollutants in outdoor air in Africa (pg m^{-3}).

Compounds	Country	Sampling year	Median range (median)	Mean range (mean)	References
OCPs	Malawi	1997–1998	31–257	NA	[40]
OCPs	Ghana	2010	NA	[e](156) & ** (153)	[41]
OCPs	Togo, Nigeria, Benin, Cameroon	2012	NA	[a]23–2718 (441) **91–1880 (403)	[42]
OCPs	South Africa	2004–2005	(12)	NA	[43]
OCPs	Uganda	2017	0.34–25	NA	[44]
OCPs	Botswana, Ghana and South Africa	2004–2005	BDL-564	NA	[45]
OCPs	Ghana	2008	BDL-750	NA	[46]
OCPs	South Africa	2017–2018	(34)	NA	[47]
OCPs	Uganda	1999–2010	NA	22.8–130	[48]
CUPs	Uganda	2008–2010	NA	3.5–93.5	[48]
CUPs	South Africa	2018	0.181-25,000	NA	[49]
CUPS	Uganda	2017	8.7–14	9.3–15	[44]
CUPS	South Africa	2017–2018	(700)	NA	[47]
PFAs	South Africa	2005	BDL-2.5	NA	[50]
PFA$_S$ (neutral)	Botswana	2006–2011	BDL-7.1	NA	[51]
PCBs	South Africa	2004–2005	BDL-128	(128)	[52]
PCBs	Mali, Cape Verde,	2001–2008	BDL-52 (10)	(14)	[53]
PCBs	Ivory coast, Gambia, Sierra Leon, Ghana, Cape Verde	2007	BDL-300	6–99	[54]
PCBs	South Africa	2008	(104)	(132)	[55]
PCBs	Uganda	1999–2010	(26.9)	14.8–462	[56]
PCBs	Ghana	2010	NA	0.28-4.64	[57]
PCBs	Nigeria	2019–2020	85-460 (300)	(160)	[58]
PCBs	Botswana, Ghana, and South Africa	2008–2019	0.5–37.7	NA	[59]
PCBs	Ghana, Malawi, South Africa	2004–2005	BDL-252	NA	[45]
PCBs	DRC, Ethiopia, Ghana, Kenya, Mali, Mauritius, Zambia, Uganda, Nigeria, Senegal, Togo,		NA	(84)	[60]
PCBs	Tunisia	2015–2016	(2.1)	NA	[61]
OPEs	Egypt	2014	2.0–16	NA	[62]
OPEs	Uganda	2017	(408) & (688)	NA	[44]
OPEs	Tunisia	2015–2016	100–1060	NA	[63]
NBFR	Uganda	2008–2010	(1.77)	0.58-4.56	[64]
NBFR	Nigeria	2019–2020	34–900	NA	[58]
NBFR	Egypt	2014	0.20–13	NA	[65]

[a] NA-not available; BDL-Below detection limit, *Hexaclorocyclohexane, **Dichloro-Diphenyl-Trichloroethane,[e] Endosulfans.

3.1.1. *Pesticides (OCPs and CUPs)*

In West Africa, authors reported 19 OCP residues in atmospheric samples in Accra, Ghana, with concentrations ranging from below detection limit (BDL) up to 564 pg m^{-3} [45]. Endosulfan had the highest annual geometric mean concentrations in air samples (82 pg m^{-3}) while dieldrin, heptachlor, heptachlor epoxide, chlordane, and α- and γ-hexachlorocyclohexane (HCH) were among the OCPs with higher detection frequencies. The data from the study showed strong correlations between the levels of OCPs and estimates of global emissions, with developed and industrialized regions having the highest concentrations.

In Ghana, the urban (East Legon site), sub-urban (Kwabenya), and rural (Lake Bosumtwi) air samples were analyzed for 12 OCPs [46]. The concentrations of individual OCPs were up to 750 pg m^{-3}. Concentrations of endosulfans were higher at both urban and semi-urban sites while HCH was dominant in the samples from the rural site. Both urban and semi-urban sites had the lowest concentrations of α- and β-chlordane, while the rural site had the lowest concentrations of dichlorodiphenyltrichloroethane (DDT) and its metabolites. The authors attributed the high concentrations to intense use of pesticides in agricultural activities whilst the devolatilization of previously used pesticides from contaminated soils was thought to be a potential secondary source.

In a related study, Hogarh et al. [41] determined the concentrations of 25 OCPs in outdoor air samples at 13 sites across Ghana. The reported mean concentrations of DDTs and endosulfans were 156 and 153 pg m^{-3}, respectively. The results showed that mirex had the lowest concentration (0.2 pg m^{-3}). Distinct chemical signatures were established, with Northern Ghana dominated by heptachlor and chlordane, while southern regions were dominated by HCH, DDTs, and drins (endrin, dieldrin, and aldrin). The observed chemical signatures suggested fresh application of DDT, endosulfans and heptachlor possibly from illegal imports of these pesticides. However, the low concentrations of hexachlorobenzene (HCB) in both agricultural and non-agricultural sites indicated combustion as a probable source and not from technical HCB. A multi-national study by Isogai et al. [42] reported the atmospheric concentrations of 25 OCPs in Togo, Benin, Nigeria, and Cameroon. HCHs and DDTs had mean values of 441 pg m^{-3} (range: 23–2718 pg m^{-3}) and 403 pg m^{-3} (range: 91–1880 pg m^{-3}), respectively, and accounted for the highest burdens of atmospheric OCPs in the investigated samples. The OCP concentrations in rural sites were comparable to urban sites in different countries, which suggested that atmospheric OCP concentrations in some West African countries were equilibrating between rural and urban areas. The ratio of cis-to trans-chlordane (1.9) and a high percentage of aldrin compared to other drins in the samples suggested recent use of chlordane and aldrin, while the ratio of p, p'-DDT/DDE (<1) suggested past use of DDTs. Seasonal variations in the levels of these chemicals were also observed with the levels of OCPs (HCB, HCH, endosulfans, heptachlor, and DDT) higher in the rainy season than dry season, which coincided with the planting phase by the farmers.

In Southern Africa, the earliest study documented the occurrence of 16 OCPs including HCHs, DDT and its metabolites, chlordane, and endosulfans in air samples from Senga Bay of the Great Lake Malawi area in Malawi [40]. Total concentrations of OCPs varied from 31 to 257 pg m^{-3} in Senga Bay. The chemical signatures of DDT and its metabolites indicated recent application of DDT. In atmospheric samples from Durban, South Africa, the authors also reported mean levels of p, p'-DDT, and p, p'-DDD as 12 and 42 pg m^{-3}, respectively. Other legacy pesticides like lindane (mean: 133 pg m^{-3}), HCB (mean:4.5 pg m^{-3}), aldrin (mean:10.1 pg m^{-3}), α-chlordane (mean:10.9 pg m^{-3}), and γ-chlordane (mean: 9.5 pg m^{-3}) were also reported [43]. Suggested sources of the pesticides included fresh application of OCPs in agriculture and malaria-control programs, volatilization from soil residues, and emissions from abandoned pesticide storage sites. Another study in De Aar and Kalahari, South Africa reported the predominance of endosulfan-α across all seasons and mean concentrations of 13 OCPs ranged from BDL up to 325 pg m^{-3} [45].

A study by Veludo et al. [47] investigated 27 OCPs in the air samples taken from three agricultural sites: Grabouw (pome fruits), Hex River Valley (table grapes) and Piketberg (wheat) in Western Cape, South Africa. Eight OCPs including chlordane, DDT and its metabolites, endosulfan sulphate, lindane and mirex were detected in all samples investigated. p, p'-DDE had the highest concentration (median: 140 pg m^{-3}). The ratio for *trans-*/*cis*-chlordane ranged from 1.20 to 13.44, suggesting ongoing application of technical chlordane in the region. The ratios of $(DDE + DDD)/\sum DDT$ and endosulfan-α/endosulfan-β were >1, which suggested contamination from past use. In the same study, CUPS in outdoor air were investigated and carbaryl and chlorpyrifos were detected in all the samples analyzed. Chlorpyrifos was the most predominant among the 16 CUPs (median concentration 700 pg m^{-3}). The concentrations of several OCPs and CUPs varied among areas and seasons (spring and summer). Higher concentrations were generally observed during the most active spraying season (spring), remaining steady throughout the warm summer months whereas the concentrations of CUPs did not differ significantly throughout the study sites. The pesticides that showed higher levels during the rainy season could be the ones potentially being used in agriculture. The other dominant CUPs were carbaryl, tebuconazole, and terbuthylazine. A strong preference for the particulate phase was observed which was attributed to the limitations in mass transport kinetics following pesticide application. The sources of all the CUPs and OCPs were agricultural applications. Pozo et al. [45] also reported concentrations of various OCPs in the atmosphere of De Aar and Kalahari, South Africa (BDL to 325 pg m^{-3}). Though the concentrations of most pesticides were BDL in various seasons, endosulfan-α was consistently detected in most seasons and was the most dominant.

In East Africa, 14 CUPs and 21 OCPs were detected in atmospheric samples from Kibale National Park, Uganda [44]. Median concentrations of the pollutants were 0.34 pg m^{-3} for α -chlordane, 25 pg m^{-3} for HCB, 9.3 pg m^{-3} for chlorpyrifos, and 0.44 pg m^{-3} for cypermethrin. Chlorpyrifos was the most abundant among CUPs and was found in all the samples followed by cypermethrin. The pesticide levels were attributed to agricultural activities around the protected forest. In another study by Arinaitwe et al. [48], 25 OCPs and 6 CUPs were studied in the atmospheric environment of Lake Victoria, Uganda over two different periods between 1999-2004 and 2008–2010. The most predominant CUP was chlorpyrifos with mean concentrations of 93.5, 26.1, and 3.54 ng m^{-3} for Entebbe 2008, 2009, and 2010 sample sets, respectively. The most detected OCPs were endosulfans, DDTs, HCHs whose average total concentrations ranged from 12.3 to 282, 22.8–130, and 3.72–81.8 pg m^{-3}, respectively. Other OCPs detected were HCB, pentachlorobenzene (PeCB), and dieldrin. The most prevalent pesticides in the air samples were in the order; ΣHCHs $>$ ΣDDTs $>$ Endos $>$ chlorothalonil and chlorpyrifos. The major sources from the backward air trajectory analysis showed transboundary and local emission sources of these chemicals.

In general, the concentrations of OCPs reported in outdoor air samples from Africa ranged from BDL to 2718 pg m^{-3}. The most dominant OCPs in outdoor air were endosulfan, DDT, and HCH [40,46,66], while chlorpyrifos dominated the profile of CUPs [44,49]. Interestingly, the levels in Africa were comparable to those in Toronto, Canada (3.2-2,700 pg m^{-3}) [67] and in Bursa, Turkey (63.7–799 pg m^{-3}) [68]. The dominant sources of the pollutants were from agricultural use, malaria control programs, and emissions from abandoned storage sites.

3.1.2. *Per- and polyfluoroalkyl substances (PFASs)*

To date, two studies have investigated the levels of PFASs in African outdoor air. In South Africa, 7 PFAS congeners, including fluorotelomer alcohols (FTOHs), N-alkylated fluorooctane sulfonamides, and sulfonamidoethanols (FOSAs/FOSEs) were quantified [50]. 8:2 FTOH was the dominant compound at a maximum concentration of 14 pg m^{-3}. In the above study, levels of ionic PFASs were approximately two orders of magnitude lower than those of neutral PFASs; maximum concentrations in the samples were 2.5 and 2.0 pg m^{-3} for perfluorooctane sulfonic acid (PFOS), and perfluorooctanoic acid (PFOA), respectively. In Southern Africa (Botswana), Gawor et al. [69] analyzed PFASs in outdoor air in the Maun and Okavango Delta. 6:2 FTOH, 8:2 FTOH, 10:2 FTOH, methyl perfluorooctane sulfonamide (MeFOSA), ethyl perfluorooctane sulfonamide (EtFOSA), and methyl perfluorooctane sulfonamidoethanol (MeFOSE) were detected at the concentrations of <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}, respectively. A positive correlation between PFAS levels and sample site proximity to highly populated areas was observed. In addition, urban locations had higher levels of PFAS than rural ones, suggesting the impact of urbanization on PFAS levels.

Overall, in Africa, levels of neutral PFASs (FTOHs, NeMEFOSE, and EtFOSA) in outdoor air were higher than those of ionic PFASs (e.g., PFOS and PFOA). This is probably because neutral or precursor PFASs are more volatile than ionic PFASs and have distinct gas/particle partitioning, which is more common in outdoor air [70]. PFAS levels in outdoor air in Africa are lower than those reported across other regions, such as in North-Western Europe (up to 818 pg m^{-3}) [71], Dhaka, Bangladesh (up to 70,000 pg m^{-3}) [51] and Toronto, Canada (up to 248 pg m^{-3}) [72]. However, due to the few studies, the sources of PFASs in the atmospheric samples could not be confirmed.

3.1.3. *Polychlorinated biphenyls (PCBs)*

Eleven studies have reported the distribution and concentration of PCBs in the outdoor air across different regions in Africa. To date, three studies on PCBs in the atmosphere have been conducted mostly in West Africa. In Mali, air samples from the Sahara were found to contain Σ_6PCBs ranging from BDL to 52 pg m^{-3} (mean = 14, median = 10 pg m^{-3}) [53]. Compared to downwind sites, PCB congeners (PCB-101, -118, −138, −153, −183, and −187) were found at higher concentrations in dust source locations. Potential sources of PCBs were electrical equipment such as transformers, plastic manufacturing and open burning of waste, including contaminated soils and vegetation [53]. In Ghana, atmospheric PCB levels ($\sum_{190}$ PCBs) varied from 280 pg m^{-3} in Kumasi to 4,640 pg m^{-3} in Accra [57]. These levels were attributed to electronic waste recycling and uncontrolled open burning of waste products containing PCBs. In Nigeria, Akinrinade et al. [58] reported $\sum_8$PCBs in outdoor air at sites within the metropolitan area of Lagos ranging from 85 to 460 pg m^{-3}. The non-Arochlor PCB-11 (range, 49–220 pg m^{-3}; median, 100 pg m^{-3}) was the most dominant congener, contributing 47 % to the $\sum_8$PCBs. The authors suggested leakages from electrical transformers and proximity to activities like textile factories that produce and use dyes as potential sources of PCBs.

Elsewhere in Africa, 82 PCB congeners were investigated in air samples from KwaZulu-Natal, South Africa. Total PCB concentrations averaged at 128 pg m^{-3}, with the highest levels recorded in winter. The dominant PCBs were PCB-33, -118, −138, and −105. The authors attributed the pollutant levels to a combination of local, regional, and global sources [52]. In East Africa, seasonal variations in the concentration of air samples were investigated at two sites, Kakira and Entebbe in Uganda [56]. Σ_7PCBs ranged from 154 to 462 and 26.7−226 pg m^{-3} for Kakira in 1999−2000 and 2003−2004, respectively. For Entebbe, ΣPCBs were 27.0−186, 46.8−174, 19.2−128, 45.8−237, and 65.6−244 pg m^{-3} in 2003, 2004, 2008, 2009, and 2010, respectively. PCB-31 and PCB-28 were the most dominant in air samples. In Northern Africa, Castro-Jiménez et al. [61] reported average levels of $\sum_{18}$PCBs from the western Mediterranean basin of Tunisia as 2.1 pg m^{-3} [61]. The authors reported PCB-180 as the most dominant congener. The Northwest Mediterranean Sea was reported as a potential secondary source of PCBs.

Several multi-country studies of PCBs in outdoor air samples have also been conducted across Africa. 29 PCBs congeners were investigated in outdoor air from four West African countries (Sierra Leone, Ghana, Gambia and Ivory Coast) [54]. The concentrations of $\sum_{29}$ PCB ranged from 8.2 to 360 pg m^{-3}. The dominant PCBs were PCB-28, -52, −101, −118, −138, −153, and −180. The authors attributed the sources of PCBs to the illegal dumping of PCB-containing waste, release via volatilization, uncontrolled burning, and old ships. Another study reported 7 PCB congeners (PCB- 28, -52, −101, −118, −153, −138, and −180) in air from 26 sites in 15 African countries [55]. High PCB median levels were found in passive air samples from Dakar, Senegal (104,000 pg per sample), with a maximum of 132,000 pg per sample, for the $\sum_7$ indicator PCB congeners. However, sites such as Cairo, Egypt (41,600 pg per sample), the three industrial sites in Kenya (36,000 pg per sample), and Kinshasa, Democratic Republic of Congo (DRC) (30,000 pg per sample) also had elevated concentrations. High-chlorinated congeners, especially PCB-153 were found in higher concentrations than the more volatile congeners. The authors reported agricultural activities, and car emissions as possible sources of PCBs.

Another study reported high concentrations of $\sum_7$PCBs in outdoor air ranging from 8 to 2,074 pg m^{-3} in eleven African countries (Democratic Republic of Congo; Ethiopia, Ghana, Kenya, Mali, Mauritius, Nigeria, Senegal, Togo, Uganda and Zambia) [60]. The low-chlorinated PCB congeners (PCB-28 and -52) were dominant, followed by the highly chlorinated congeners (PCB-101, -118, −138, −153, and −180). Possible sources identified were electronic waste, exported to Africa from industrialized countries. Pozo et al. [45] investigated PCB levels in Accra, Ghana and at two sites in South Africa (De Aar and Kalahari). The levels ranged from BDL to 252 pg m^{-3}, with a geometric mean of 26 pg m^{-3}. High PCB levels were found in the vicinity of cities that are known to be close to emission sources. Most recently, a study covering nine African countries (Congo, Ghana, Mali, Ethiopia, Kenya, Mauritius, Morocco, Nigeria, and Sudan) reported the $\sum_6$ PCBs of 0.5−37.7 pg m^{-3}, with PCB-52 and -28 as the most dominant congeners [59]. The dominance of the congeners was attributed to their high vapour pressures which facilitate their volatilization compared to higher molecular weight congeners, especially in hotter climates [58]. This study identified the main sources of PCB emissions in Africa as uncontrolled imports, substandard recycling, and open burning of waste, especially electronic waste (e-waste). Volatilization from landfills and other waste, as well as leakages from old electrical transformers, were found to be additional possible sources.

There were wide variations in the levels of PCBs in Africa depending on the region with the highest concentrations reported in Ghana [57], Senegal [55], and the lowest in Tunisia [61], and Mali [53]. Various sources reported from different studies included illegal dumping of PCB-containing waste [54], open burning of waste [53,57,59], volatilization from contaminated soils, vegetation and landfills [53,59], electronic recycling [57], leakages from electrical transformers [58,59], textile files that produce and use dyes [58], agricultural activities and car emissions [55]. Overall, PCB levels in outdoor air in Africa were in the same range of data as those reported from other parts of the world, such as the eco-industrial park in China Σ_9PCBs (39.9−537 pg m^{-3}) [73] and North Rhine-Westphalia, Germany, Σ_3PCBs (300−1500 pg m^{-3}) [74]. However, the levels were higher than those reported in outdoor air samples from the Aliaga region, Turkey, Σ_{35}PCBs (349−94,363 pg m^{-3}) [75] and e-waste recycling sites of Qingyuan, China, Σ_{171}PCBs (7825−76,330 pg m^{-3}) [76].

3.1.4. *Organophosphate esters (OPEs)*

Due to their emerging nature, research on the occurrence of OPEs in outdoor air in Africa is in its infancy, with only three studies reported to date. In Tunisia, particle-bound OPEs (Σ_9OPEs) were reported with their concentrations varying from 100 to 1,060 pg m^{-3} in air samples from Bizerte [63]. Tris (2-chloroisopropyl) phosphates (TCiPPs), 2-Ethylhexyl di-Phenyl phosphate (EHDPP) and tri-isobutyl phosphate (TiBP) had the highest median concentrations of 110, 100, and 85 pg m^{-3}, respectively. In Egypt, 12 OPEs in outdoor air of apartments, cars, and working spaces were between 2.0 and 16 pg m^{-3}, with Tris (1, 3-dichloro-2-propyl) phosphate (TDCPP) as the dominant compound in the samples [62]. In Uganda (East Africa), a study carried out at two sites in Kibale National Park found median concentrations of Σ_{19}OPE at 408 and 688 pg m^{-3} [44]. Tri-n-butyl phosphate (TNBP), Tris (2-chloroethyl) phosphate (TCEP), and TCIPP were the OPE congeners with detection frequencies greater than 75 %. Open burning of plastics and other items containing flame retardants were suggested as potential sources. Generally, in Africa, TCIPP and TDCPP appear to be the most prevalent OPEs in outdoor air. The levels of OPEs in Africa were lower than those reported in Turkey (529 -19,139 pg m^{-3}) [12], Canada (BDL-7,630 pg m^{-3}) [67], and in sites in America, Europe, Latin America and Asia (464-15,100 pg m^{-3}) [77]. This observation is corroborated by a global study of OPEs in outdoor air from other sites including De Aar (South Africa). In the study, Σ_9OPEs ranged from 30 to 350 pg m^{-3} with the most dominant pollutants being TCPP, TCEP, TPP, and TBEP. De Aar had lower concentrations compared to other global sites involved in the study [78].

3.1.5. *Novel brominated flame retardants (NBFRs)*

Four studies have investigated NBFRs in outdoor air samples in Africa. In Eastern Africa, Arinaitwe et al. [64] reported the occurrence of 12 NBFRs in air samples around Lake Victoria, Uganda. The authors reported an increase in Bis (2-ethylhexyl) tetrabromophthalate (BEH-TEBP) between 2009 and 2010, with mean concentrations of 3.39 pg m^{-3} (detected in 16 % of the samples) in 2009, and 18.2 pg m^{-3} (detected in 88 % of the samples) in 2010. The concentrations of BEH-TEBP were attributed to increasing industrialization in the Lake Victoria watershed. In West Africa, Akinrinade et al. [58] investigated NBFRs in the air samples from a metropolitan area of Lagos, Nigeria, and found that the concentrations Σ_7NBFRs ranged from 34 to 900 pg m^{-3} with a mean of 330 pg m^{-3}. Decabromodiphenyl ethane (DBDPE) was the most dominant congener (<37−890 pg m^{-3}). The sources were reported to be local due to increased population, urbanization and industrialization. In North Africa, Khairy & Lohmann [65] determined 9 NBFRs in outdoor air in Egypt and reported levels ranging from 0.2 to 13 pg m^{-3}. The sources of NBFRs were not identified in this study.

In a global study that included South Africa (De Aar), hexabromobenzene (HBB), ethyl-1-hexyl-2,3,4,5-tetrabromobenzoate (EH-TBB) were the NBFRs that were detected with the highest frequencies and varying concentrations (0.11 and < 0.8 pg m^{-3}, respectively) [78].

Generally, NBFRs in Africa appear to be positively associated with the levels of urbanization and industrialization in the different areas. The levels of NBFR in outdoor air in Africa were lower compared to those reported in the different cities in China (Beijing, Tianjin, and Guangzhou, and large/medium cities (Qingdao, Xiamen, Kunming, Mianyang, and Nanchang) (9.56-1,350 pg m^{-3}) [79], and (Beijing, Shanghai, Guangzhou, Wuhan, Guiyang, Shijiazhuang, Lanzhou, Jinan, Chengdu, and Zhengzhou) (1–1428 pg m^{-3}) [80].

3.1.6. *Polychlorinated naphthalenes (PCNs)*

PCNs remain one of the most understudied pollutants in Africa. In Southern Africa, a study of PCN levels on a ship transiting to South Africa revealed mean $\sum_{13}$PCN concentrations that varied from 0.3 to 86 pg m^{-3}. PCN-24 was the most dominant compound; contributing 46.89 % of the $\sum$PCNs reported. The mean levels of PCNs measured across the Southern Hemisphere seas, including those around Africa (0.3–5 pg m^{-3}) were among the lowest ever recorded globally [81]. In West Africa, PCNs were determined at 13 sites in the atmospheric environment of Ghana. The authors reported $\sum_{63}$PCNs with mean levels of 49 pg m^{-3} (range 7–95 pg m^{-3}) and the main congeners were Tri-CNs and Tetra-CNs. Generally, the PCN levels in outdoor air in Africa were in the same range as those reported in India and Pakistan (29 and 7.7 pg m^{-3}, respectively) [82] but lower than for other reports from China (mean, 48 pg m^{-3}) [83] and 7.2 pg m^{-3} [84]. The major sources of PCNs in Africa were identified as open burning of waste, industrial emissions, and the harbor environment [85].

3.2. *Occurrence of selected pollutants in indoor dust from Africa*

The summary of the concentrations of the selected LOPs and EOPs in African indoor-settled dust is presented in Table 2.

3.2.1. *Pesticides (OCPs and CUPs)*

A few studies have investigated pesticide occurrence in indoor dust in Africa. In Southern Africa, the occurrence of pesticides in indoor dust from households and schools has been investigated [88]. 10 CUPs were detected in more than half of the 54 samples analyzed (median, 4.38 ng g^{-1} for tebuconazole to 365 ng g^{-1} for chlorpyrifos). The mean concentrations of pesticides in ng g^{-1} were diazinon (122), terbuthylazine (9.30), carbaryl (1020), carbendazim (16.3), chlorpyrifos (1250) and tebuconazole (13.6). In South Africa, DDT (mean, 1,200,000 ng m^{-2}) was reported in homes that had used DDT for indoor residual spray (IRS) to control malaria [86]. In a related study, DDT was quantified in indoor house dust from South Africa, with a mean level of 1,100 ng m^{-2}, and its metabolite o, p'-DDE was quantified with the highest concentration during the spraying period [101]. The findings from the studies sparked worries about the possible negative health consequences of DDT IRS on residents living in the environment that had been sprayed. In Northern Africa, Abdallah et al. [66] investigated 5 OCPs and 10 CUPs in indoor dust in Egypt and found high levels of pyrethroid compounds in indoor dust samples, with mean levels of 4750, 2690, and 15,210 ng g^{-1} for permethrin, cypermethrin, and cyhalothrin, respectively. Cyhalothrin was the dominant CUP (maximum concentration, 123,000 ng g^{-1}). OCPs were detected in most samples with endrin having the highest concentration. The mean levels of the OCPs were 10.42, 36, 33.68, and 1260 ng g^{-1} for DDT, HCH, endosulfan, and endrin,

Table 2
Concentrations of selected pollutants in indoor dust in Africa (ng g^{-1}).

Compounds	Indoor environment	Country	Sampling year	Median range (median)	Mean range (mean)	References
OCPs	# Indoor dust	Egypt	2008–2009	NA	6.9–1260.4	[66]
CUPs	$^{\#}$ Indoor dust	Egypt	2008–2009	NA	32.7–15,206.9	[66]
OCPs	$^{\#}$ Indoor dust	South Africa	2008	BDL-910,000[a]	NA	[86]
OCPs	$^{\#}$ Indoor dust	South Africa	2012	BDL-51.2[a]	NA	[87]
CUP	$^{\#}$ Indoor dust	South Africa	2018	4.38–365	NA	[88]
PFASs	$^{\#, \beta, \&\#\#}$ Indoor dust	Egypt	2013	1.3–69	1.09–55.2	[89]
PCBs	$^{\#}$, $^{\beta\ \&\ \delta}$ Indoor dust	South Africa	2012	NA	# (891), β (923), δ (1880)	[90]
PCBs	$^{\#\#}$, $^{\#}$, $^{\&\ \beta}$ Indoor dust	Nigeria	2014	0.08–34	NA	[91]
PCBs	^{P} Indoor dust	Nigeria	2015	21-2,200	NA	[92]
PCBs	$^{\gamma}$ Indoor	Nigeria	2017	96.6-3,949	NA	[93]
PCBs	$^{\#}$ Indoor dust	Nigeria	2014 & 2020	3.6–61(18)	22	[94].
OPEs	$^{\#}$, $^{\#\#}$, $^{\beta\ \&\ \alpha}$ Indoor dust	Egypt	2012–2013	NA	$^{\#}$ (310), $^{\#\#}$ (1011), α (2167) β (450)	[95]
OPEs	$^{\#, \beta, \delta, \&\#\#}$ Indoor dust	South Africa	2012	$^{\#}$(22,940), β (26,930), δ (19,565), $^{\#\#}$ (49,010)	NA	[96]
OPEs (10)	$^{\#}$ Indoor dust	South Africa	2018	NA	^{F} (44,800), ^{v} (19,800)	[97]
OPEs	$^{\#, \beta, \#\#}$ Indoor dust	Egypt	2014	$^{\#\#}$ (692), $^{\beta}$ (359), # (243)	$^{\#}$ 122–1117, β 189–1138, $^{\#\#}$ 272-1849	[62]
OPEs	$^{\#}$, $^{\#\#}$, $^{\beta\ \&\ ʒ}$ Indoor dust	Nigeria	2021	NA	$^{\#\#}$ (295), β (231), $^{\#}$ (277) & ʒ (127)	[98]
NBFRs	$^{\#, \beta\ \&\#\#}$ Indoor dust	Egypt	2013	$^{\#}$ (8.3), β (28.9), $^{\#\#}$ (49.9)	NA	[99]
NBFRs	$^{\#}$ Indoor dust	Egypt	2014	$^{\#}$0.60–10, $^{\#}$0.5–2.0	NA	[65]
NBFRs	$^{\lambda}$ Indoor dust	South Africa	2016	BDL-1402	NA	[100]
NBFRs	$^{\#}$Indoor dust	South Africa	NA	BDL-24,800	NA	[97]
NBFRS	$^{\#}$ Indoor dust	Nigeria	2014 & 2020	32–2,600 (320)	NA	[94]

[a] Concentration in ng m^{-2}; NA – not available, Homes/apartments (#), Elevated surface dust in home (#*), Floor dust in homes (#**), cars/buses (##), offices/work places (β), electronic repair workshops (γ), computer laboratories (δ), public microenvironments (PMES) (α), End of life equipment (λ),medical centers (ʒ) and power stations (P), *EH-TBB, **BEH- TEBP.

respectively. In general, the levels of OCPs quantified in the indoor dust samples from Africa were in the same range as those reported elsewhere, including Nepal (median, 87 ng g^{-1}) [102], and Singapore (median, 17.3 ng g^{-1}) [103]. For CUPs, the concentrations in indoor dust from Africa were higher than those from Argentina and Europe [104]. Although Africa is not known to produce these chemicals, a few studies conducted on indoor dust samples report high levels of both OCPs and CUPs. Therefore, further research is required to fully understand the levels, seasonal variations, and trends, of OCPs and CUPs in Africa.

3.2.2. Per- and polyfluoroalkyl substances (PFASs)

The current study found only one report on PFASs in indoor dust samples in Africa. The average concentration of 18 PFASs in indoor (home and car) dust in Cairo, Egypt varied from 1.3 to 69 ng g^{-1}, with FTOHs being the most dominant PFAS [89]. PFOA and PFOS were detected at median levels of 2.40 and 0.29 ng g^{-1}, respectively [89]. PFASs levels in Egypt were lower than those reported globally. For example, $\sum$PFASs quantified in house dust from Korea varied from 29.9 to 97.6 ng g^{-1}, with a dominance of PFOS, 8:2 FTOH, NEtFOSE and PFOA [105]. In Finland, the highest medians for PFOA, polyfluoroalkyl phosphoric acid esters (PAPs) and 8:2 FTOHs in some children's bedrooms were 5.26, 53.9 and 45.7 ng g^{-1}, respectively [106]. Noteworthy, for Africa, no reports on the occurrence of emerging PFASs such as per- and poly-fluoroalkyl ether carboxylic acids (GenX and ADONA) that have recently been adopted as PFOA replacements were found.

3.2.3. Polychlorinated biphenyls (PCBs)

In West Africa, PCBs have been investigated in indoor dust samples from Nigeria. The authors reported PCB levels ranging from 0.08 to 34 ng g^{-1} for 6 congeners (PCB-28, -52, −101, −153, −138 and −180) in dust from 16 cars, 12 homes and 18 offices in Lagos [91]. The concentrations of all target PCBs in office dust significantly exceeded those in car dust. PCB-180 had the highest concentration and was the most dominant in all the microenvironments studied while PCB-28 and -101 had the lowest concentration in homes. A related study reported the concentrations of Σ_7PCB in indoor dust from several power stations in Nigeria ranging from 21 to 2,200 ng g^{-1}, with median PCB concentrations ranging from 3.8 ng g^{-1} for PCB-101 to 52 ng g^{-1} for PCB-180 [92]. The authors attributed PCB concentrations in dust to activities at power stations. High PCB levels (96.6–3,949 ng g^{-1}) were also reported in indoor dust from electronic repair workshops in southern Nigeria [93]. The authors reported that the PCBs could have emanated from the technical mixture of Aroclor 1252, 1254 and 1260. A related study in Nigeria, reported PCB concentrations in indoor household dust in the range of 3.8–61 ng g^{-1} (median, 18 ng g^{-1}). Among the PCBs targeted, PCB-138 had the highest concentration while PCB-11 and -28 were BDL [94].

In South Africa, the mean $\sum_3$PCBs was 235 ng g^{-1} in indoor dust from two e-waste recycling sites and a university electronic equipment repair workshop in Durban. The authors reported PCB-180 as the most dominant congener [107]. In another report, Abafe & Martincigh [90] found that the mean concentrations of 3 PCBs (PCB-28, -153, and −180) in indoor dust from homes, offices and university computer laboratories in Durban, South Africa were 891, 923, and 1880 ng g^{-1}, respectively. Overall, the levels of PCBs reported in indoor dust samples from different parts of Africa varied significantly across countries. High levels have been reported in indoor dust in South- and West Africa [92,96]. These levels are even higher than those obtained in other parts of the world such as in Pakistan (mean, 24.84 ng g^{-1}) [108], Iran (mean, 3.46 ng g^{-1}) [109] and Greece (mean, 6.29 ng g^{-1}) [110]. PCB-180 was noted in this study as the most frequently detected congener in dust. The highest PCB concentration in dust was 3,949 ng g^{-1} from Nigeria [93]. The sources of PCBs in Africa were mostly attributed to e-waste sites and illegal waste dumping.

3.2.4. Organophosphate esters (OPEs)

In North Africa, OPEs were first reported from Egypt in houses, cars, and public microenvironments (PME) by Abdallah & Covaci [95]. The levels of Σ_8OPEs were 1011, 310, 450, and 2,167 ng g^{-1} in cars, houses, offices, and PMEs, respectively. The dominant OPE was triphenyl phosphate (TPHP), with the highest average concentration of 386 ng g^{-1}. Other authors analyzed OPEs in indoor dust of apartments, cars, and working spaces in Egypt [62]. The most common OPE found in the study was TDCIPP, with levels ranging from 150 to 1,850 ng g^{-1} in indoor dust. The dominance of TDCIPP, TCIPP, tris (2-butoxy ethyl) phosphate (TBOEP), and TPHP were reported in all the samples but exhibited much indoor variability [62]. These results were confirmed later by other researchers who detected OPEs in indoor dust from some homes in Cairo, Egypt. OPE levels ranged from 7,300 to 99,500 ng g^{-1} with TBOEP and TDCIPP being the most abundant among non-halogenated and halogenated congeners, respectively [111].

In Southern Africa, 4 OPEs were detected in indoor dust from houses, cars, and university computer laboratories from Durban. The mean values of the $\sum$OPEs for homes, offices, university computer laboratories, and cars were 52053, 28187, 22162 and 143716 ng g^{-1}, respectively. The dominant OPEs were TDCIPP and TCEP [96]. The number of electronics, foams, and furniture in households correlated strongly with OPE levels, suggesting human exposure to OPEs through these household items. Another study reported the Σ_{10}OPEs as 44,800 and 19,800 ng g^{-1} in freshly collected floor dust and vacuum cleaner dust in houses in South Africa [97]. The most dominant OPE was TCIPP. Because South Africa does not have any OPE production facilities, the high concentrations of chlorinated OPEs observed in indoor dust suggested that imported consumer items are a key source of these chemicals [97].

In West Africa, the quantities, profiles, and human exposure to 8 OPEs in indoor dust from several microenvironments such as cars, houses, offices, and medical centers were documented in Nigeria [98]. The dominant OPEs were TBOEP, TCIPP, and EHDPP. The average concentrations of Σ_8OPEs in dust samples from cars/buses (295 ng g^{-1}), offices (231 ng g^{-1}), homes (277 ng g^{-1}) and medical centers (127 ng g^{-1}) did not differ significantly. The levels of OPEs in indoor dust from Africa ranged from BDL to 143,716 ng g^{-1}. The most frequently detected OPEs in indoor dust were TBOEP, TCIPP, TDCIPP, TPHP and TCEP. Generally, OPEs exhibited the highest levels in indoor dust compared to outdoor air samples in Africa. This is probably because OPEs emitted indoors tend to accumulate in indoor dust, resulting in higher concentrations than the more dynamic and diluted outdoor air. With the exception of Abafe & Martincigh [96] in South Africa, the OPEs in indoor dust in Africa were lower than those in other parts of the world, including Nepal (median, 13,000 ng g^{-1}) [102], Belgium, Italy, and Spain (12,800 ng $g^{-1)}$ [112] and Iraq (5,950 and 3,590 ng g^{-1} in elevated surface dust and floor dust, respectively) [113]. However, this conclusion needs to be drawn up with caution since the number of studies conducted in Africa is still very low.

3.2.5. Novel brominated flame retardants (NBFRs)

Three studies on NBFRs have been conducted on indoor dust in Africa. In West Africa, NBFRs were determined in households in Nigeria and $\sum_7$NBFRs were found to be in the range 32-2,600 ng g^{-1} (median, 320 ng g^{-1}). DBDPE was the most dominant NBFR, comprising 68 % of all NBFRs studied [94]. In Northern Africa, researchers quantified the $\sum_6$NBFRs in indoor dust from homes, workplaces, and cars in Egypt as 8.30, 28.9, and 49.9 ng g^{-1}, respectively [99]. EH-TBB and HBB were the most detected NBFRs in all the samples. In a related study, 9 NBFRs were analyzed in dust from workplaces, apartments, cars, and floors from Alexandria in Egypt [65]. The concentrations ranged from 0.60 to 10 ng g^{-1} and 0.50–2.0 ng g^{-1} in elevated fine dust and floor dust, respectively. Although the sources of NBFRs were not identified, the profiles of NBFR in floor dust closely matched those of outdoor dust, suggesting a probable carryover from the external environment [65]. In South Africa, indoor dust samples from discarded office furniture and equipment contained EH-TBB as the most frequently detected NBFR (found in 57 % of all the indoor dust samples examined). The concentrations of 2 NBFRs ranged from <LOD to 673 ng g^{-1}. NBFRs were also analyzed in freshly collected floor dust and vacuum cleaner dust. The median concentrations were 80 and 44 ng g^{-1} for BEH-TEBP, and 31 ng g^{-1} and 29 ng g^{-1} for EH-TBB, respectively. BEH-TEBP and EH-TBB contributed 19 % of the BFR profile in the dust collected from vacuum cleaner bags and 11 % in freshly collected dust [97].

In general, NBFRs have been accepted as alternatives to legacy brominated flame retardants such as PBDEs and HBCD in consumer products [100]. Their presence in indoor dust and outdoor air in Africa could be attributed to their wider acceptance for use following the ban imposed on penta-, octa- and deca-BDE formulation [114]. The differences in the concentration of NBFRs in indoor and outdoor air among African countries could be related to the differences in the levels of NBFRs in imported materials [65]. The available literature shows that EH-TBB, BEH-TEBP, and DBDPE were the most frequently detected NBFRs in indoor dust in Africa. This could probably be due to their use in a wide range of consumer goods, including electronics, textiles, and plastics which are commonly found indoors [115]. The levels of NBFRs in indoor dust from Africa were comparable to those reported elsewhere in the world, such as in Istanbul, Turkey (320-31,400 ng g^{-1}) [116] and Lativia (120-7,295 ng g^{-1}) [117].

4. Human health risk assessment due to exposure to OCPs, CUPs, PCBs, OPEs, and NBFRs

Previous studies have linked OCPs, PCBs, OPEs, and NBFRs to various health effects. For instance, exposure of mothers and fetuses to OPEs during pregnancy has been linked to thyroid-disrupting effects [118]. OCPs have been reported to affect the brain and nervous system [119]. Human exposure to NBFRs has been linked to induction of cytotoxicity in umbilical vein endothelial cells [120]. In addition, severe motor control and neurological effects have been reported in women exposed to PCBs during pregnancy, while in children, PCBs have been reported to cause short-term memory loss and lowered IQ [121]. In this review, the assessment of health risks for the different categories of pollutants was carried out for non-carcinogenic risks, based on daily intakes and reference doses. Non-carcinogenic effects include neurotoxicity, mutagenicity, reproductive toxicity, and developmental toxicity [122]. Equations (1)–(3) were used to calculate the chronic daily intake (CDI) for OCPs, CUPs, PCBs, OPEs, and NBFRs with indoor dust via ingestion, inhalation, and dermal exposure [123,124].

$$CDI_{\text{ingestion}} = \frac{C_i \times IR_{ing} \times ED \times EF \times CF}{AT \times BW} \tag{1}$$

$$CDI_{\text{inhalation}} = \frac{C_i \times IR_{inh} \times ED \times EF}{AT \times BW \times R_{PE}} \tag{2}$$

$$CDI_{\text{dermal contact}} = \frac{C_i \times ED \times EF \times SA \times AF \times ABS \times CF}{AT \times BW} \tag{3}$$

For air, the chronic daily intake via inhalation was calculated using equation (4) [125].

$$CDI_{\text{inhalation}} = \frac{C_i \times IR_{i-air} \times ED \times EF}{AT \times BW} \tag{4}$$

where C_i is the concentration of the individual analytes in indoor dust (ng g^{-1}); IR_{ing} is the ingestion rate of dust (50 and 20 mg day^{-1} for children and adults, respectively); IR_{inh} is the inhalation rate of dust (7.6 and 20 m^3 day^{-1} for children and adults, respectively); ED is the exposure duration (6 and 70 years for children and adults, respectively); EF is the exposure frequency (365 days year^{-1}); CF is the conversion factor (1 $\times 10^{-6}$ kg mg^{-1}); AT is the averaging time (2,190 and 25,550 days for children and adults, respectively); IR_{i-air} is the inhalation rate of air (7.6 and 20 m^3 day^{-1} for children and adults, respectively); BW is the body weight (15 and 70 kg for children and adults, respectively); R_{PE} is the particle emission rate (1.36 $\times 10^9$ m^3 kg^{-1}); SA is the surface area of the skin that contacts dust (1,600 and 6,700 cm^2 for children and adults, respectively); AF is the skin adherence factor (0.5 mg cm^{-2} for both children and adults) and *ABS* is the dermal absorption fraction of the reviewed pollutants (0.03 and 0.001 for children and adults, respectively).

To estimate the non-carcinogenic risk via ingestion, inhalation, and dermal contact, hazard quotients (HQs) and hazard indices (HIs) from various studies were assessed using Equation (5) [123].

$$HI = HQ_{ing} + HQ_{inh} + HQ_{derm} = \sum\nolimits_1^n \left(\frac{CDI_{\text{ing}} + CDI_{\text{inh}} + CDI_{\text{derm}}}{RfD_i} \right) \tag{5}$$

Where RfD_i is the reference dose, mg kg^{-1} day^{-1}. If HI $\leq$ 1, no adverse health effects exist; if HI > 1, possibly adverse health effects exist.

Correspondingly, for air, the HI was calculated according to Equation (6);

$$\text{HI} = \sum\nolimits_1^n \frac{CDI_{inh}}{RfD_i} \tag{6}$$

The reference doses for OCPs and CUPS were obtained from the USEPA Integrated Risk Information System [126]. For OPEs, reference doses (RfDs) were obtained from the work of Ali et al. [127], while oral RfD values for NBFRs-EHTBB, BEHTEBP, BTBPE, and DBDPE-are 20,000, 20,000, 243,000, and 333,333 ng kg^{-1} bw day^{-1}, respectively, according to Stapleton et al. [128], and Hardy et al. [129]. The RfD of PCBs is 2 $\times$ 10^{-5} mg kg^{-1} day^{-1} according to IRIS 1994 [126]. The reference doses for PFOA and PFOS are 333 and 25 ng kg^{-1} bw day $^{-1}$, respectively. Those of PFSA are the same as of PFOS, while those of PFCAS are the same as those of PFOA [130]. Compounds whose RfDs were unavailable were not included in the health risk assessment calculations. Certain studies that were reviewed in this article were not used in health risk assessment for several reasons, such as due to the difference in units (ng sample^{-1}) [42], data not disaggregated for Africa for studies that covered Africa and other continents [50], and/or absence of concentration values as mean or medians [44]. To the best of our knowledge there are no RfD values for PCNs and so their HIs were not calculated.

The results of indoor dust and outdoor air health risk assessments are presented in Table 3. For countries where more than one study was carried out for the same group of pollutants, the average value of HIs was considered. For indoor dust, HI values (adults and children) greater than 1 (inhalation, ingestion, and dermal exposure via dust) were obtained for PCBs in South Africa. HI (children) values greater than 1 (inhalation, ingestion and dermal exposure via dust) were obtained for PCBs in Nigeria and South Africa. This

Table 3
Hazard Index (HI) values of pollutants in indoor dust and outdoor air.

Pollutants	HI (adults)	HI (children)	Country	Reference
Indoor dust				
OCPs	1.40E-03	1.92E-01	Egypt	[66]
Pyrethroids	1.08E-03	1.48E-01	Egypt	
PCBs	**1.71E+00**	**3.63E+00**	South Africa	[90]
PCBs	3.19E-02	**4.23E+00**	Nigeria	[91–94]
OPEs	2.22E-03	2.71E-02	Egypt	[95,111]
OPEs	9.40E-03	1.15E-01	South Africa	[96]
OPEs	1.62E-04	1.97E-03	Nigeria	[98]
NBFRs	1.42E-07	1.73E-06	Egypt	[99]
NBFRs	3.61E-05	4.40E-04	South Africa	[97,100]
CUPs	6.09E-05	7.41E-04	South Africa	[88]
Outdoor air				
OCPs	2.99E-03	4.55E-04	Malawi	[40]
OCPs	3.30E-04	5.01E-05	Ghana	[41,46]
OCPs	2.42E-04	3.36E-05	South Africa	[43]
OCPs	1.66E-01	1.01E-01	South Africa	[45,47]
OCPs	1.73E-04	2.63E-05	Uganda	[48]
CUPs	7.91E-04	1.20E-04	South Africa	[47,49]
NBFRs	1.32E-10	2.00E-11	Uganda	[64]
NBFRs	2.22E-03	3.38E-04	Nigeria	[58]
PCBs	2.22E-03	3.38E-04	Nigeria	[58]
PCBs	7.04E-02	1.07E-02	Tunisia	[61]
PCBs	1.83E-06	2.79E-07	South Africa	[52]
PCBs	1.50E-03	2.28E-04	Ivory coast	[54]
PCBs	2.21E-06	3.36E-07	Gambia	[54]
PCBs	1.80E-04	2.74E-05	Sierra Leon	[54]
PCBs	3.25E-04	4.94E-05	Ghana	[54]
PCBs	1.77E-03	2.68E-04	Uganda	[56]
OPEs	1.68E-04	2.56E-05	Tunisia	[63]

Bold values are those that had HI > 1.

indicates that South African residents and Nigerian children were likely exposed to non-cancer risks of PCBs from contaminated indoor dust. For the rest of the pollutants, the HI values were one to five orders of magnitude lower than the acceptable risk level (HI = 1), suggesting negligible non-cancer risk from exposure to the pollutants in indoor dust. For outdoor air, HI values were all several orders of magnitude lower than the acceptable risk level (HI = 1), suggesting negligible non-cancer risk from exposure to the reviewed OCPs, OPEs, PCBs and NBFRs. The results were consistent with the previous data for most of our target pollutants [131–135].

Based on Table 3, the HIs for children were higher than those for adults (inhalation, ingestion and dermal exposure via dust) probably due to their frequent hand–to–mouth contact and a lower body weight [136]. Additionally, the HI (adults) via air inhalation was higher than the corresponding values for children for all the reviewed pollutants because of the higher inhalation rate of the adults than children (20 and 7.6 m^3 day^{-1}, respectively). The CDI values for ingestion and dermal contact revealed the two as significant exposure pathways to the reviewed pollutants (OCPs, PCBs, OPEs, and NBFRs) by children and they were 5 orders of magnitude higher than that of the inhalation pathway (Table S1). Children are more susceptible to the effects of chemicals through dermal exposure because the metabolic systems of children are still under development, and they undergo rapid neurodevelopment [137]. Additionally, children may inadvertently ingest soil, dust, or chemical residues from surfaces and/or objects they touch. Therefore, more attention should be paid to children because they are susceptible to chemical exposure. For adults, the ingestion exposure pathway to the reviewed pollutants (OCPs, PCBs, OPEs and NBFRs) had a total CDI value that was 2–3 orders of magnitude higher than those of dermal contact and inhalation pathways (Table S1). These results agree with previous studies [138].

5. Analytical techniques utilized in the studies

5.1. Sampling, extraction, and clean-up of outdoor air samples

In the reviewed studies, legacy (OCPs, PCNs, and PCBs) and emerging (CUPs, PFASs, NBFRs, and OPEs) organic pollutants were collected from the air matrix using either active air sampling (AAS) or passive air sampling (PAS). Active samplers can collect target compounds in air because they operate under controlled flow rates [139]. However, the use of active samplers is limited by several factors such as the breakthrough of the compounds at higher sampling volumes, stripping of particulate phase compounds from the filter, degradation within the sampler [140], and possible sorption of gaseous phase compounds on the filter surfaces [141]. Passive samplers simplify and reduce the costs involved in air sample collection, and they do not require electricity [142]. Passive sampling, however, is not suitable for monitoring short-term analyte concentration variations and has low enrichment efficiency, among other limitations [142].

High-volume active air samplers (HV AAS) were equipped with glass fibre filters (GFFs) or quartz fibre filters (QFFs) to trap the target pollutants in particulate phase and polyurethane foam (PUF) discs to trap the vapour phase pollutants. Degrendele et al. [49] used a medium-volume sampler equipped with a quartz microfiber filter to capture the particulate phase of the pesticides and a PUF/XAD/PUF sandwich to capture the vapour phase pesticides. For passive air sampling (PAS), polyurethane foam (PUF) was used to collect vapour phase analytes. Additionally, low-density polyethylene passive samplers (LDPE) sheets were used as passive air samplers to capture novel brominated flame retardants in the vapour phase [65].

The extraction and clean-up methods employed in this study are presented in Table 4. Briefly, the sampling media (PUFs and or G/QFFs) were spiked with appropriate internal surrogate standards and extracted using Soxhlet extraction [40,41,45,52,81] or accelerated solvent extraction (ASE) or sonication [44,62,64] or cold extraction [65] or orbital shaking [50]. The extraction solvents or solvent mixtures that were used by different studies include petroleum ether, acetone, *n*-hexane, dichloromethane (DCM), benzene and methanol.

Soxhlet extraction was performed at room temperature for 12–36 h [47,49]. However, it is time-consuming, taking approximately 6–24 h depending on the complexity of the matrix, while ASE takes only a few (15–30) minutes [143]. In addition, Soxhlet extraction processes mostly require high volumes of solvents that may be hazardous and costly. To decrease the quantity of organic chemical residues and increase the concentration factor, more efficient methods requiring little to no solvent such as solid-phase extraction, and solid-phase micro-extraction should be used whenever feasible [144,145] as well as to obtain results with a lower detection limit. Following the extraction, the extracts were generally reduced to incipient dryness using rotary evaporation or under gentle nitrogen flow and then resolubilized in suitable organic solvents depending on the instrumental analytical technique. Clean-up and sometimes fractionation was achieved using column chromatography packed with a range of sorbents such as pre-cleaned cotton wool, activated florisil, activated silica, activated carbon, and deactivated silica (Table 4). Elution of the target compounds involved whole or fractionation elution using appropriate solvents. In one of the studies, extracts were further passed through gel permeation chromatography (GPC) using 6 g of Biobeads SX 3 [81]. GPC is merited because it can remove many polymers, copolymers, proteins, and other high molecular weight matrix components [146].

Table 4
Analytical methods used for different pollutants in outdoor air samples.

Pollutant category	Sample collection	Extraction Solvent	Clean-up	Instrumental system	Column	Reference
Pesticides	HVAAS/PUF/GF	Soxhlet extraction (*n*-hexane/DCM)	Column chromatography; Florisil/silica gel	GC-ECD & GC-MS	DB-5 capillary	[40]
Pesticides	PUF/PAS	Soxhlet extraction (DCM)		GC-MS, EI		[45]
Pesticides	PUF-PAS	Soxhlet extraction (dichloromethane)	silica gel/florisil column	GC-MS/MS EI, SIM Splitless, 1 μ L	DB-5-MS fused silica	[45]
Pesticides	PUF-PAS	Soxhlet extraction (acetone)	columns –Celite, florisil and graphite column/ florisil column/graphite column (F2)	HRGC/HRMS		[41]
Pesticides	PUF-PAS	Soxhlet extraction (acetone)	columns –Celite, florisil and graphite column/ florisil column/graphite column (F2)	HRGC/HRMS		[42]
Pesticides	HVAAS-PUF	Soxhlet extraction (benzene/*n*-hexane 4:1 v/v)	4.5 M sulphuric acid	GC-MSD NCI, SIM	DB-5 fused silica	[52]
Pesticides	PUF-PAS	Soxhlet extraction	silica column/sodium sulphate/silica gel/acidic silica	GC-MS GC-APCI-MS/MS		[47]
Pesticides	MVAAS/PUF/XAD/QFF	Soxhlet extraction (methanol)	syringe filters	HPLC-MS ESI+	Phenomenex Luna C-18 end capped	[47]
Pesticides	PUF-PAS	ASE (*n*-hexane/acetone)	silica column	OCP-GC-ECD CU-GC-MS	DB 5	[44]
Pesticides	HVAAS/PUF/GFF	ASE (*n*-hexane: acetone)	10 % deactivated silicagel	GC-dualECD GC-μ ECD	HP-5, HP-5MS, HP-1MS	[48]
CUPs	HVAAS/PUF/GF	ASE (*n*-hexane: acetone)	10 % deactivated silicagel	GC-MSD	HP-5MS	[48]
PFASs	PAS- XAD and AAS-PXP	Soxhlet extraction ASE (1:1 v/v) acetone: *n*-hexane)	1.0 g of sodium sulphate-no other clean up	GC-MS PCI, SIM GC-MS	DB-WAX	[69]
PFASs	HVAAS/GFF (1/2) PUF/XAD GFF (1/2)	Orbital shaking (ethyl acetate) Cold column elution (ethyl acetate) ultrasonication (methanol)		GC/PCI-MS NCI, LC-ToF/MS for ionic analytes		[50]
PCNs	HVAAS-PUF/GFF	Soxhlet extraction (DCM)	chromatography column; alumina/silica gel/ sodium sulphate/(GPC)	GC-MS, EI, SIM		[81]
PCNs	PUF-PAS	Soxhlet extraction (acetone)	multilayer silica gel/activated carbon columns		DB-5 MS	[85]
NBFRs	HVAAS-PUF	ASE (*n*-hexane/acetone 7:3 v/v)	deactivated (10 %) silica gel	GC-MSD NCI, SIM Splitless	HP 5MS	[64]
NBFRs	PUF-PAS	ASE (*n*-hexane)	Column: 1 g 44 % H_2SO_4-impregnated silica topped anhydrous Na_2SO_4/(DMSO)	GC-QMS EI, SIM Splitless,1 μ L	Restek	[58]
NBFRs	LDPE sheets	Cold extraction	–	GC-MS/MS EI, MRM	–	[65]
OPEs	HVAAS-QFF	ASE (DCM)	Column chromatography; sodium sulphate/silica/ alumina	GC-MS EI, SIM	HP-5MS capillary	[62]
OPEs	PUF-PAS	ASE (*n*-hexane/acetone)	silica column	GC-MS	Rtx-OPPesticides2 capillary	[44]
PCBs	HVAAS-PUF/QFF	Soxhlet extraction (benzene/*n*-hexane 4:1 v/v)	4.5 M sulphuric acid	GC-MS NCI, SIM	DB-5 fused silica	[52]
PCBs	HVAAS/PUF/GFF	ASE (*n*-hexane/acetone)	deactivated (10 %) silica gel	GC-ECD	HP-5, HP-5MS, HP-5 and HP-1MS	[56]
PCBs	HVAAS/QFF	Soxhlet extraction (*n*-hexane/DCM 9:1 v/v)	multilayer silica gel column/florisil column	GC-HRMS EI, SIM Splitless, 1 μ L	DB-5MS (60 m × 0.25 mm × 0.25 μ m)	[61]
PCBs	HVAAS/GFF/QFF/PUF	ASE 35 % ethyl acetate in *n*-hexane	CarboPrep SPE column/Oasis (HLB) SPE column	GC-MSD EI/ECNI	DB-5MS	[53]
PCBs	PAS-PUF	Soxhlet extraction (DCM)	Column; silica gel	GC-NCI/MS GC-EI-MS/MS	DB 5MS fused silica	[55]
PCBs	PAS-PUF	Soxhlet extraction (*n*-hexane)	Column chromatography; alumina/silica gel/ silica/sodium sulphate/GPC	GC-MS EI, SIM	Restek XTI-5	[54]
PCBs	PUF-PAS	Soxhlet extraction (acetone)		HRGC/HRMS		[57]
PCBs	PUF-PAS	ASE	Column H_2SO4-impregnated silica, anhydrous sodium sulphate and DMSO	GC-MS Splitless, 1 μ L	Restek Rxi-5Sil MS	[57,58]

ASE-Accelerated Solvent Extraction, SIM-Select ion monitoring, MRM-Multiple reaction monitoring, GC-Gas chromatography, MS-Mass spectrometry, EI-Electron ionisation, NCI-Negative chemical ionisation, ECNI-Electron-Capture Negative Ionisation, ECD-Electron capture detector, HRGC-High resolution gas chromatography, HRMS-High resolution mass spectrometry, NA-not available**.**

LDPE samples were cold extracted twice with methylene chloride and *n*-hexane for 24 h each with no further cleanup [65]. Researchers in Tunisia also added a few grams of activated copper to ASE flanks to remove potential traces of sulphur [61].

5.2. Sampling, extraction, and clean-up of indoor dust samples

As shown in Table 5, vacuum cleaners were typically used to gather dust samples because this method collects a larger volume of dust compared to wipes or doormats. This larger sample size

Table 5
Analytical methods used in Africa for different pollutants in indoor dust samples.

Category	Sample collection	Extraction (solvent)	Cleanup	Instrumental system	Column	Reference
Pesticides	Vacuum cleaners	Ultrasonic bath and centrifugation (methanol)	Chromafil syringe filters	HPLC-MS ESI+	Phenomenex Luna C-18 end capped	[88]
Pesticides	Vacuum cleaner HVS3	(*n*-hexane/acetone)	florisil column	OCP- GC/EC/FP Pyrethroid-HP-5MS	DB-17	[66]
Pesticides	Vacuum cleaner	NA	NA	GC-ECD	NA	[86]
Pesticides	Acrylic cloth	NA	NA	GC-MS	NA	[87]
PFAs	Vacuum cleaners	Vortexing, sonication and centrifugation (DCM), Methanol	activated carbon	GC-PCI MS for neutral and HPLC-MS/MS for ionic	Zorbax XDB –C-18	[89]
NBFRs	Vacuum cleaner bags	Sonication and centrifugation (DCM)	activated carbon	GC-MS NCI, SIM	DB-5MS capillary	[99]
NBFRs	Vacuum cleaner plastic brushes and aluminium shovel	Sonication (*n*-hexane/acetone)	fractionated over silica gel	GC-MS/MS		[65]
NBFRs	Pre-cleaned glass wool (End of life electronic equipment)	Ultrasonication and centrifugation (DCM/toluene)	glass wool silica/pesticarb/ sodium sulphate	GC-MS	DB-5	[100]
NBFRs	Vacuum cleaners	ASE (*n*-hexane/acetone)	Wool/silica/florisil,/ sodium sulphate	GC-MS/MS ECNI	BPX capillary	[97]
OPEs	Vacuum cleaners	Ultrasonic extraction (*n*-hexane/ acetone 3:1 v/v)	Florisil cartridge	GC-MS EI, SIM	DB-5	[95]
OPEs	Vacuum cleaners	ASE (*n*-hexane/acetone 3:1 v/v)	glass wool/silica/ florisil/sodium sulphate	GC-MS/MS EI, SRM	BPX5 capillary column	[97]
OPEs	Vacuum cleaners and small plastic Brushes	Ultra-sonication (DCM)	silica gel	GC-MS/MS EI, MRM		[62]
OPEs	Vacuum cleaners	Vortex, sonication and centrifugation (DCM)	Florisil column	GC-MS	DB − 5	[111]
OPEs	Vacuum cleaners and paint brushes	Vortexing, sonication and centrifugation (*n*-hexane: acetone 3:1 v/v)	Hypersep Florisisl® cartridges	GC-MS		[98]
PCBs	Vacuum cleaner	Ultrasonication and vortexing (*n*-hexane/methanol)	Silica gel/Florisil column	GC-MSD SIM	Restek Rtx®-1614 fused silica (5 % diphenyl, 95 % dimethyl poly siloxane)	[96]
PCBs	Vacuum cleaner	ASE (*n*-hexane/DCM 1:9 v/v) Florisil and Hydromatrix	SPE cartridges/ acidified silica	GC-MSD NCI	VF5 MS	[91]
PCBs	Paint brush	Vortexing, ultrasonication and centrifugation (*n*-hexane: DCM 1:1 v/v)	concentrated H2SO4/florisil column DMSO	GC-MS SIM	Restek RXi-PAH	[92]
PCBs	Paint brushes	Ultrasonication (acetone/*n*-hexane)	multilayer silica gel/alumina column	GC-MS EI, SI M	DB 17	[93]

ASE-Accelerated Solvent Extraction, SIM-Select ion monitoring, MRM-Multiple reaction monitoring, GC-Gas chromatography, MS-Mass spectrometry, EI-Electron ionisation, NCI-Negative chemical ionisation, ECNI-Electron-Capture Negative Ionisation, ECD-Electron capture detector, HRGC-High resolution gas chromatography, HRMS-High resolution mass spectrometry, ESI + -Electrospray ionisation (positive mode), NA-not available.

provides a more comprehensive representation of the contaminants present in the environment [62,65,66,88,89,91,95–99,111].

These vacuum cleaners were usually equipped with quartz fibre filters using a stainless-steel inlet equipped with pre-separation mesh sieving particles up to 1 mm connected to a conventional vacuum cleaner. In one study, a high-volume surface sampler (HVS3) was used to measure surface dust before and after using the Hoover Self-Propelled Vacuum with embedded Dirt Finder (HSPF) [66]. Other studies used pre-cleaned glass wool [100], small plastic brushes and a small aluminium shovel for other indoor environments [62,65]. Paint brushes were also used for OPEs [98] and PCBs [92,93]. Prior to instrumental analysis, dust samples were spiked with appropriate standards and extracted by vortexing, sonication and centrifugation [89,92,96,98] or accelerated solvent extraction using a myriad of solvents or solvents mixtures such as methanol or *n*-hexane/acetone mixture, *n*-hexane, methanol, *n*-hexane/DCM mixture [58,91,97]. Cleanup procedures employed multilayer column chromatography [61,91,93] over which fractionation was possible whenever needed. The commonly used sorbents included florisil, alumina or silica gel (Table 4).

5.3. *Instrumental analysis of outdoor air and indoor dust samples*

In general, the target compounds in samples collected from atmospheric air and indoor dust were analyzed using gas chromatography (GC) coupled with mass spectrometry (MS) (Tables 4 and 5). MS, a powerful tool for structural identification, renders GC-MS hybrids very useful instruments for the analysis of a wide class of organic contaminants. OCPs from the matrices were analyzed using GC-ECD [40,44], GC-MS [44,45], GC-MS/MS [45,47], GC-MSD [43], HPLC-MS [47,49], GC-EC/FP [66] and HRGC-HRMS [41,42]. Neutral PFASs were analyzed by GC-MS [50,69,89] whereas the ionic counterparts were analyzed by HPLC-MS/MS [89]. Two studies on PCNs were found for atmospheric air samples and the analysis was done using either GC-MS [81] or HRGC-HRMS [85]. NBFRs were analyzed by GC-MSD [64], GC-MS [99,100], GC-MS/MS [65,97] and GC-QMS [58]. OPEs were mostly detected by GC-MS [44,62,95,98,111] though GC-MS/MS was also used in some studies [62,97]. PCBs were analyzed by GC-MS [52,55,92,93], GC-ECD [56], GC-MSD [53,91,96], GC-MS/MS [55] and HRGC-HRMS [57].

High-performance liquid chromatography coupled with tandem mass spectrometry operating in positive electrospray ionisation was used for the analysis of some of the target analytes (OCPs) and quantification done by isotope dilution [88] (Tables 4 and 5). Separation was done on various kinds of columns such as DB-5-MS, DB-5 capillary, Luna C-18, DB-WAX, HP 5MS, HP-5, with varying lengths of 30–60 m. Helium gas was used as a carrier gas at appropriate flow rates and the suitable GC-MS parameters were used to ensure quantitative separation of the analytes. Helium was commonly used for chromatographic analysis because it is safe and has a relatively wide optimum linear velocity range, although it is expensive. Other gases that could be used include argon, nitrogen, hydrogen and carbon dioxide [147]. Generally, GC-MS and liquid chromatography-mass spectrometry (LC-MS) were frequently employed in Africa for the detection of POPs as shown in Tables 4 and 5 Given the trace level concentrations of these pollutants in environmental samples, both GC-MS and LC-MS offer sensitivity and selectivity, making it possible to detect POPs. Furthermore, these instrumental techniques offer high specificity, making it possible to precisely identify and quantify individual POPs even in complex mixtures. The analytical capabilities of the techniques are likewise broad; GC-MS is particularly useful for VOC and SVOC analysis, whereas LC works well for non-volatile and thermally unstable chemicals [148,149]. The use of HRGC-HRMS and Time-of-Flight Mass Spectrometry (TOF-MS) techniques are emerging as powerful tools for fast and precise analysis and have been used in a few studies on the African continent as shown in Table 4.

6. Conclusions and future research directions

This review synthesized over two decades literature on the levels of LOPs and EOPs in indoor dust and outdoor air in Africa, their associated health risks as well as analytical techniques used for their analysis. The most common pollutants in Africa were Chlorpyrifos for CUPs in both indoor dust and outdoor air; FTOHs for PFASs in both indoor dust and outdoor air; tetra CNs in outdoor air for PCNs; endosulfans, DDT, and HCH (outdoor air) and DDT and endrin (indoor dust) for OCPs. PCBs in outdoor air were dominated by PCB-28, -52, −118, −138, and −153, while indoor dust was PCB-180. TCIPP dominated the air for OPEs, whereas TDCIPP and TBOEP dominated in dust. For NBFRs, HBB and EH-TBBB predominated in the air, whereas DBDPE and EH-TBB prevailed in dust. PCB levels were consistently high in outdoor air and dust, in most cases in the same range of data and even higher than studies elsewhere in the world.

Most HIs associated with exposure to these pollutants were below the threshold value of 1, indicating negligible non-carcinogenic risk in adults and children except for PCBs in Nigeria and South Africa. Studies on the selected pollutants generally remain scanty, specific pollutant groups such as OPEs, PFASs and PCNs, were minimally studied in Africa in the two matrices. For instance, for PCNs, the reviewed literature did not reveal any study on indoor dust in Africa. The main sources of pesticides in Africa have been determined to be agricultural use (both current and previous use), as well as by-products of specific processes like manufacturing or combustion, while the sources for PCBs and OPEs were mostly emissions from consumer goods, e-waste sites, open burning of waste, plastic and textile manufacturing and illegal dumping sites. PCNs sources were reported to be open burning of waste and industrial emissions. In terms of analytical techniques, advanced separation methods such as two-dimensional gas/liquid chromatography, the latest ionisation methods, and contemporary mass spectrometry methods like time-of-flight (TOF) and high-resolution mass spectrometry (HR-MS), which improve the detection limits, accuracy, and reproducibility, were not widely used.

Though the available data generally points to lower levels of these pollutants in indoor dust and outdoor air in Africa, there is a need to institute policies geared towards reducing the entry of products containing these pollutants into Africa, as most of them are not manufactured on the continent. Intergovernmental cooperation, awareness-raising, and the strengthening of policies and regulations related to the import, registration, distribution, manufacture, and disposal of products containing such pollutants could achieve this. There is need for creation and enforcement of policies for better management of LOPs and EOPs in Africa.

Future studies should put more emphasis on various indoor environments, such as cars, homes, hospitals, and offices, to better understand the presence of these pollutants and their potential health risks. Furthermore, studies on pollutant sources as well as gas-particle partitioning for the contaminants should be carried out to generate data that will help establish the most probable exposure pathways. There is also need for increased collaboration among African countries and scientists for data sharing and technology transfer.

CRediT authorship contribution statement

Godfrey Muhwezi: Writing – review & editing, Writing – original draft, Conceptualization. **Christine Kyarimpa:** Writing – review & editing, Supervision, Funding acquisition. **Ivan Gumula:** Writing – review & editing, Supervision. **Christine Betty Nagawa:** Writing – review & editing, Funding acquisition. **Solomon Omwoma:** Writing – review & editing, Funding acquisition. **Henry Matovu:** Writing – review & editing. **Julius Matsiko:** Writing – review & editing. **Silver Odongo:** Writing – review & editing. **Douglas Sifuna:** Writing – review & editing. **Ivan Špánik:** Writing – review & editing. **Charles Drago Kato:** Writing – review & editing. **Patrick Ssebugere:** Writing – review & editing, Supervision, Funding acquisition.

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.

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Appendix A. Supplementary data

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