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Urban soil and human health: a review

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Summary

Rapid industrialization and urbanization during recent decades are having dramatic effects on urban soil properties and lead to large discharges of pollutants, which inevitably affect the health of the soil, ecosystems and human populations. This paper provides a systematic review of the relations between urban soil and human health. First, it summarizes the organic and inorganic pollutants in urban soil and their potential risks to human health. Second, the relations between urban greenbelt land, soil microbial diversity and human health are also explored. Third, we propose that future research should focus on the integration of assessments of health risks with exposure pathways and site characteristics. Bioavailability-based risk assessment frameworks for pollutants in urban soil can elucidate the complicated relations between urban soil, pollutant exposure and human health in cities. Finally, management of urban soil and policy should be strengthened in the future to maintain its sustainable development and utilization. More effort should be directed to understanding the relations between soil microbial diversity, green space and human health in cities.

Highlights

- Evidence indicates the importance of urban soil in maintaining human health.
- Pollutants, green space and microbial biodiversity have been systematically summarized.
- Urban vegetation and antibiotic resistance genes in urban soil have implications in human health.
- Bioavailability of pollutants and antibiotic resistance genes should be considered for human risk assessment.

Introduction

The urban environment represents socio-ecological systems and the most complex mosaic of land cover and multiple land use of any landscape (Andersson, 2006). Edmondson *et al.* (2012) reported that urban soil represents about 3% only of the global terrestrial surface, but 54% of the world's population now live in urban areas according to the 2014 Revision of World Urbanization Prospects (United Nations, 2014). The urban population is likely to increase by 2.5 billion by 2050, accounting for 66% of the global population (United Nations, 2014). The soil, an important component of the Earth that provides vital functions for human society (Amundson *et al.,* 2015), has been studied in mainly rural and semi-rural areas, and its importance is often overlooked or underestimated in urban environments (Hazelton & Murphy, 2006).

Oliver & Gregory (2015) observed both direct and indirect effects of the soil on human health. Humans are directly exposed to contaminants in soil through skin contact, inhalation, ingestion and consumption of plants grown in soil, whereas indirect effects include interactions between soil, the soil microbial community, vegetation and the nutritional value of foods and human health. Water and air directly influence our health because of direct contact through ingestion and inhalation, but the connection between human health and soil is often more complex and thus is not well understood. There are some comprehensive reviews on soil and human health (Oliver, 1997; Abrahams, 2002; Pepper, 2013;

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Brevik & Sauer, 2015; Oliver & Gregory, 2015; Steffan *et al.,* 2018), but there is little focus on urban soil in these. With the rapid economic development and urbanization, urban soil plays an increasingly important role in environmental and human health. In this review, we focus on the relations between urban soil and human health. First, we examine the ecosystem functions of urban soil, and second, the interactions between urban soil contamination, vegetation and soil microbial diversity and human health are investigated. Finally, soil management and policy related to human health are also summarized.

The characteristics of urban soil

Compared with traditional soil science, urban soil science is a relatively young scientific subject and has attracted research interests since the mid-1970s only, especially in the USA, Germany and Russia (Norra & Stüben, 2003). Urban soil, described as 'anthropic soil', has been disturbed profoundly by human activities through the mixing, importing and exporting of materials (De Kimpe & Morel, 2000), and is often characterized by contamination, compaction and soil sealing, as well as deposition, and removal or mixing of natural substrates. Soil in the urban environment tends to be very disturbed because of surrounding human activities and might even be exogenous (i.e. transported from elsewhere) (Craul, 1999; Bullock & Gregory, 2009). Urban soil is widely distributed in parks, along roads, sports fields, urban rivers, peri-urban areas, landfill sites and mining areas, and near to buildings, industries and transport facilities. Urbanization modifies natural ecosystems to human-dominated systems by anthropogenic activities. It is well known that urbanization brings various advantages, such as better healthcare, sanitation and public transport (Zhu *et al.,* 2011, 2017b), but it changes the natural landscape in the human residential environment. Rapid expansion of urban areas transforms more and more natural and agricultural soils into urban soil and poses risks to the ecosystems and to humans within and around cities (Luo *et al.,* 2012c). For example, vehicle emissions, coal combustion, building, waste disposal and incineration, metallurgy and paint use in urban areas have put considerable pressure on the urban environment (Wong *et al.,* 2006). Urbanization will inevitably result in physical changes to the urban environment. Therefore, those large changes make the physical, chemical and biological properties of the urban environment quite different from the natural one.

The most obvious change from natural to urban soil is the structural change with urban soil use, which can affect the diversity of soil functions. A direct effect of urbanization is soil sealing (impervious surfaces), which will reduce most of the soil functions, such as production, pollution attenuation, hydrological cycling and energy balance. Large quantities of pollutants can be produced and transferred from the urban environment into urban and peri-urban soils (Simon, 2008; Zhu *et al.,* 2017b). Although soil is inherently capable of buffering against perturbation, overloading the system in the urban environment can pose a threat to urban soil functions (Luo *et al.,* 2012c). The urban ecosystem, urban and peri-urban soils are being linked increasingly to human health and well-being. It is therefore important that soil scientists should be actively involved in investigating urban soil in terms of both human well-being and urban ecosystem health.

Urban soil contamination and human health

Rapid urbanization leads to intensive anthropogenic activities and consumption of resources and energy in urban areas (Pouyat *et al*., 2007; Luo *et al*., 2012c). Emissions in urban areas come from transport (fossil fuel combustion, attrition of parts and tyres, petrol and engine oil leaks), coal combustion (power plants and heating), industrial activities (mining, metallurgy and chemical engineering) and building, and waste disposal and incineration contaminate the soil and ecosystems (Cachada *et al.,* 2012; Luo *et al.,* 2012c). Atmospheric deposition, effluents, solid wastes and soil pollution in the industrial areas are much more serious than in rural areas because of the large intensity of discharge. Urban soil mainly occurs in urban green spaces such as parks and gardens and is a repository for contaminants, and it is in such places that people have more direct contact with soil (Luo *et al.,* 2012c). People are also exposed to contaminants through the food chain from urban and peri-urban agriculture (Wortman & Lovell, 2013; Zhu *et al.,* 2017b). Therefore, more attention should be paid to the soil of peri-urban areas (Figure 1b) and urban parks (Figure 1c). Urban soil pollution can act as a secondary source of pollutants, for example by air transport of volatile substances and particles by wind or water runoff. Currently, the study of urban soil pollution focuses on three aspects: (i) source, status and fate of pollutants, (ii) effects of pollution on the ecosystem and human health and risk assessments, and (iii) urban soil use and remediation. Here, we consider how organic and inorganic pollutants and their distribution and bioavailability in urban soil affect human health and assess the risks to health.

Organic and inorganic pollutants in urban soil

Organic pollutants in urban soil have been focused on in research in environmental science in recent decades because of their complex structure and harmful effects on the environment and human beings (Cachada *et al*., 2012). Amongst the important classes of organic chemicals, the four main categories of persistent organic pollutants (POPs) have attracted the most attention: (i) byproducts of combustion or industrial processes (i.e. polychlorinated dibenzo–p–dioxins and polychlorinated dibenzofurans (PCDD/Fs) and polycyclic aromatic hydrocarbons (PAHs)), (ii) families of chlorinated aromatics, including polychlorinated biphenyls (PCBs), PCDD/Fs, dioxins and organochlorine pesticides (OCPs) such as dichloro–diphenyl–trichloroethane (DDT) and hexachlorohexanes (HCHs) and hexachlorobenzene (HCB) and (iii) emerging contaminants such as brominated flame retardants (BFRs) and polybrominated diphenyl ethers (PBDEs) (Figure 2).

In addition to organic pollutants, inorganic pollutants of heavy metals have been of concern in urban soil for some time. Urban soil is an important sink for heavy metals because they are non-degradable and difficult to remove, and can persist for long

Figure 1 Contaminated urban and peri-urban soil in a rapidly urbanized Yangtze Delta city, China: (a) contaminated urban soil next to a steel manufacturing site, (b) vegetables grown in contaminated peri-urban soil adjacent to one of the largest coal-burning power plants in China, (c) urban park used for exercise and recreation activities for urban residents and (d) hand-to-mouth exposure from urban soil for children in an urban park. Photographed by Gang Li.

periods. Toxic elements including lead (Pb), cadmium (Cd), chromium (Cr), mercury (Hg) and arsenic (As), as well as essential elements of copper (Cu), zinc (Zn), cobalt (Co), manganese (Mn) and nickel (Ni) that become toxic at large concentrations, are the main heavy metal pollutants of urban soil. In an urban environment, heavy metals can be discharged from numerous anthropogenic sources, which include traffic emissions, industrial activities, municipal waste disposal, corrosion of construction or building materials, coal power generating plants, and mining and smelting operations.

Distribution of organic pollutants. Organic pollutants are ubiquitous in urban soil, but here we focus on PAHs, PCBs, OCPs and emerging pollutants such as phthalic acid esters (PAEs) and BFRs. The PAHs are a family of compounds found in the environment because of the extensive use of fossil fuels and incomplete combustion processes (Shen *et al*., 2013). They are strongly lipophilic substances and reside preferentially in soil for years to decades because of their low volatility, especially the compounds with higher molecular weights (Jones & de Voogt, 1999). Many studies have investigated the distribution of PAHs in urban soils around the world (Table 1). The concentrations of PAHs in urban soil of different cities vary greatly. Developing countries in Asia such as China have the largest concentrations, especially in the larger and more developed cities such as Beijing and Shanghai (Table 1). There are also large concentrations of PAHs in western countries, including Portugal, the UK, Slovenia and Spain, in the larger and more industrialized cities such as Lisbon, Portugal, and Glasgow, UK, in contrast to smaller and less industrialized cities such as Viseu, Portugal, and Tarragona, Spain (Table 1). The main sources of PAHs include motor vehicle exhaust, industrial activities, coal and biomass combustion, and atmospheric deposition of long-range transported PAHs. The sources can be different in a single area and vary in concentration because long-range

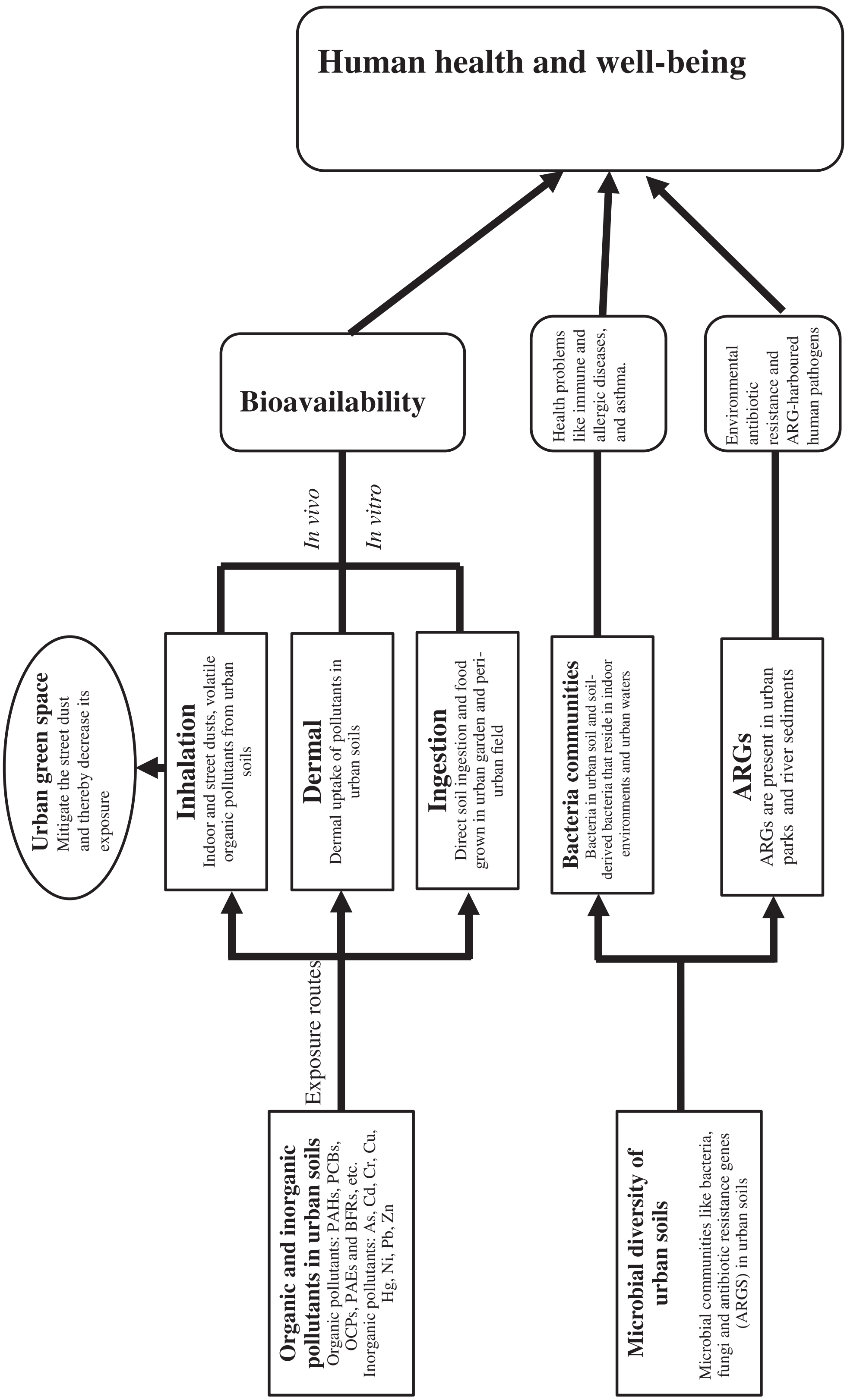

Figure 2 Characterization of pollutants, urban green space and microbial diversity in urban soil. PAHs, polycyclic aromatic hydrocarbons; PCBs, polychlorinated biphenyls; OCPs, organochlorine pesticides; PAEs, phthalic acid esters; BFRs, brominated flame retardants.

Table 1 Organic pollutants in urban and peri-urban soils

Location	Number of urban surface soils sampled	Depths / cm	Statistics	∑PAHs / ng g^{-1}	∑PCBs / ng g^{-1}	OCPs / ng g^{-1}	Other contaminants / ng g^{-1}	Reference
Bangkok, Thailand	30	0–5 and 5–10	Range	17–3260[a]	0.30–6.17[b]	–	–	Bandowe *et al*. (2014)
			Mean	508[a]	1.63[b]			
Beijing, China	10	0–20	Range	–	–	21–276[c]	–	Wang *et al*. (2008)
			Mean			77[c]		
Beijing, China	118	0–20	Range	–	–	1200–2 848 000[d]	–	Yang *et al*. (2009)
			Mean			216 140[d]		
Beijing, China	233	0–10	Range	93–13 141	ND–37[e]	–	–	Peng *et al*. (2011) and Wu *et al*. (2011)
			Mean	1228	12[e]			
Beijing, China	25	0–20	Range	126–365 926	47–3883[f]	2.4–933[g]		
			Mean	24 307	680[f]	69[g]	–	Yuan *et al*. (2014)
Beijing, China	30	0–5	Range	336–27 825				
			Mean	3917	–	–	–	Tang *et al*. (2005)
Beijing, China	30	5–30	Range	467–5470	–	–	–	Li *et al*. (2006b)
			Mean	1637				
Beijing, China	47	0–20	Range		–	–	20–2900[h]	Cheng *et al*. (2015a)
			Median	–			3808[h]	
Beijing, China	127	0–20	Range	–	–	–	1.9–3142[i]	Xia *et al*. (2011b)
			Mean				1140[i]	
Boston, Providence, Springfield, USA	60	0–6	Range	2.3–167[j]	–	–	–	Bradley *et al*. (1994)
			Mean	18[j]				
Bratislava, Slovakia	61	0–10	Range	45–12 151	–	–	–	Hiller *et al*. (2015)
			Mean	2065				
Glasgow, UK	20	0–10	Range	1487–51 822[a]	4.5–78[k]	–	–	Cachada *et al*. (2009) and Morillo *et al*. (2007)
			Mean	11 930[a]	–			
Guangzhou, China	40	0–20	Range	–	–	–	1950–33 600[l]	Zeng *et al*. (2008)
Mean							6820[l]	
Guangzhou, China	37	0–20	Range	–	–	–	1670–322 000[l]	Zeng *et al*. (2009)
			Mean				–	
Hongkong, China	138	0–5	Range	ND–19 500	–	–	–	–
			Mean					
Hongkong, China	58	0–10	Range	–	1.6–9.9[m]	–	–	Zhang *et al*. (2007)
			Mean		4.8[m]			
Harbin, China	17	0–20	Range	202–3256	0.3–6.17[b]	–	–	Ma *et al*. (2009)
			Mean	837	1.63[b]			
Kurukshetra city, India	13	–	Range	19–2538	3.3–35[n]	–	–	Kumar *et al*. (2013)
			Mean	632	12[n]			
Lisbon, Portugal	51	0–10	Range	6.3–22 670	0.18–34[o]	–	–	Cachada *et al*. (2012)
			Mean	1544	7.0[o]			
Ljubljana, Slovenia	21	0–10	Range	218–4488[a]	2.8–48[k]	–	–	Cachada *et al*. (2009) and Morillo *et al*. (2007)
			Mean	989[a]	–			
London, UK	76	5–20	Range	4–67	1–750[p]	–	–	Vane *et al*. (2014)
			Mean	18	22[p]			
Nanjing, China	139	0–5	Range	58.6–18 000	–	–	–	Wang *et al*. (2015a)
			Mean	3330				
Ningbo, China	90	0–20	Range	–	–	–	ND–103[q]	Tang *et al*. (2014)
			Median				23[q]	
							ND–79[r]	
							9.2[r]	
							ND–16.4[s]	
							0.95[s]	
Romania (eight cities)	26	0–5	Range	–	–	2.8–89.5[t]	–	Covaci *et al*. (2001)
			Mean			26.1[t]		
						9–187[u]		
						62.5[u]		
Shanghai, China	154	0–20	Range	18.8–6320	–	–	–	Wang *et al*. (2015d)
			Mean	807				
Shanghai, China	54	5–10	Range	62–31 900[e]	–	–	–	Liu *et al*. (2010)
			Mean	1700[e]				

Table 1 Continued

Location	Number of urban surface soils sampled	Depths / cm	Statistics	∑PAHs / ng g^{-1}	∑PCBs / ng g^{-1}	OCPs / ng g^{-1}	Other contaminants / ng g^{-1}	Reference
Shanghai, China	55	0–15	Range	442–19 700[v]	232–1132[m]	–	–	Jiang *et al*. (2009a, 2011)
			Mean	3780[v]	3057[m]			
Taiyuan, China	15	0–10	Range	–	–	5.1–120[w]		Fu *et al*. (2009)
			Mean			–		
Torino, Italy	20	0–10	Range	148–23 500[a]	1.8–172[k]	–	–	Cachada *et al*. (2009) and Morillo *et al*. (2007)
			Mean	1990[a]	–			
Tarragona, Spain	27	0–3	Range	42–1472	0.19–10[p]	–	–	Nadal *et al*. (2007)
			Mean	438	4.4[p]			
Uppsala, Sweden	20	0–10	Range	–	2.3–77[k]	–	–	Cachada *et al*. (2009) and Morillo *et al*. (2007)
			Mean		–			
Viseu, Portugal	14	0–10	Range	6.0–790	0.08–15[o]	–	–	Cachada *et al*. (2012)
			Mean	169	4.6[o]			
Xianyang, China	59	0–25	Range	–	–	–	128.6–10 288[x]	Wang *et al*. (2015c)
			Mean				638[x]	
Yinchuan, China	12	0–5	Range	–	–	0.306–74.2[t]	–	Wang *et al*. (2009)
			Mean			7.98[t]		
						0.410–1068[u]		
						92.1[u]		

[a]Σ15PAHs, polycyclic aromatic hydrocarbons. [b]Σ44PCBs, polychlorinated biphenyls. [c]Σ15OCPs, organochlorine pesticides. [d]Σ8OCPs. [e]Σ18PCBs. [f]Σ125PCBs. [g]Σ23OCPs. [h]Σ13PAEs, phthalate esters. [i]Σ5PAEs. [j]Σ17PAHs. [k]Σ19PCBs. [l]Σ16PAEs. [m]Σ74PCBs. [n]Σ28PCBs. [o]Σ21PCBs. [p]Σ7indicatorsPCBs. [q]Σ4HBCDs, hexabromocyclododecane. [r]TBBPA, tetrabromobisphenolA. [s]TBC, Tris–(2,3–dibromopropyl)isocyanurate. [t]ΣHCH, hexachlorocyclohexane isomers. [u]ΣDDT, dichloro–diphenyl–trichloroethane. [v]Σ22PAHs. [w]Σ21OCPs. [x]Σ6PAEs. ND, no detection.

transport of airborne substances produces a concentration gradient, which can expand downwind into surrounding rural areas. For example, ∑16PAHs concentrations in the city of Nanjing, China, had a decreasing trend along a city centre–suburban–rural gradient. Among the different functional areas in cities, larger concentrations of PAHs often occur more along roadsides and industrial areas than in parks, commercial and residential districts. Moreover, many factors, such as vegetation cover, soil organic matter content and levels of soil microbial activity, can affect the concentrations of PAHs in urban soil (Tang *et al*., 2005; Morillo *et al*., 2007; Peng *et al*., 2012).

Polychlorinated biphenyls are not present naturally in the environment and are discharged into soil by anthropogenic activities such as waste combustion, landfill, inadequate disposal of transformers and capacitors, and accidental spills (Ma *et al*., 2009; Vane *et al*., 2014). They have been used widely as dielectrics in capacitors and transformers, and as plasticizers in paints and joint sealants, since the 1930s. Similar to PAHs, PCBs also tend to remain in the soil for long periods, ranging from years to decades because of their long half-life and recalcitrance (Jones & de Voogt, 1999); their production was banned in 1977. There have been many studies about PCB concentrations in urban soil worldwide (Table 1). The concentrations of PCBs in urban soils in China, especially in large cities like Beijing and Shanghai, are tens to hundreds of times larger than those measured in Europe. In large and highly industrialized European cities, such as Torino, Italy, and Glasgow, UK, larger concentrations have been observed than in smaller cities (Table 1). As for PAHs, there is a relatively strong positive correlation between the PCB congeners and soil organic matter in urban soil.

Organochlorine pesticides (OCPs) have been used widely for several decades; they mainly include hexachlorocyclohexane isomers (α–HCH, β–HCH, γ–HCH, δ–HCH, ΣHCH as the total) and dichlorodiphenyltrichloroethane and metabolites (*p*,*p*'–DDT, *p*,*p*'–DDE and *p*,*p*'–DDD, ΣDDT as the total). Around the world, ca. 10 million tons of HCHs were released to the environment between 1948 and 1997; China consumed the largest amount of HCH, accounting for almost half of the total global use. Most studies on OCPs focus on agricultural soil because of their application in agricultural settings (Li *et al*., 2006a; Jiang *et al*., 2009b; Yang *et al*., 2012; Lupi *et al*., 2016). There are few studies on OCPs in urban soil, such as in parks, school playing fields or contaminated sites (Li *et al*., 2008; Wang *et al*., 2008; Zehra *et al*., 2015). The OCPs in urban park soils that have been investigated in big cities of China had large ΣHCH and ΣDDT concentrations (Li *et al*., 2008; Wang *et al*., 2009). Soil samples collected from college campuses showed that DDTs accounted for 93.7% of total OCPs, followed by HCHs (2.25%), indicating that OCPs could pose a potential health risk to humans (Wang *et al*., 2008). The OCPs in soil from rural sites in China were significantly larger than those in urban soil in Romania. For example, OCPs in agricultural soil in China ranged from 386.5 to 4689.4 ng g^{-1} compared to 2.8–89.5 ng g^{-1} in urban soil in Romania (Li *et al*., 2006a, Table 1). However, the concentrations in contaminated urban sites (Yang *et al*., 2009) can be two or three orders of magnitude larger than those found in the soil of parks and schools (Yuan *et al*., 2014, Table 1). The OCPs in soil can be transferred into the atmosphere by evaporation or in the form of dust, and be transported for long distances (Bozlaker *et al.,* 2009); this can expose humans to these organic pollutants through inhalation some distance away. Yang *et al*. (2009) found that both dermal uptake and inhalation exposure were two of the main routes of exposure to OCPs in urban soil for people in Beijing, China, and the exposure could cause considerable risk to human health, with problems such as neurological damage, endocrine disorders and hypertension.

In addition to POPs, emerging organic contaminants have been of considerable concern, in particular phthalate esters (PAEs) and brominated flame retardants (BFRs) (Clarke & Smith, 2011). Research on emerging organic pollutants in urban soil has increased (Zeng *et al*., 2008; Xia *et al*., 2011b; Tang *et al*., 2014; Lyche *et al*., 2015). In comparison to other cities, relatively large concentrations of PAEs were found in urban and suburban soils from Guangzhou and Beijing, China, with average concentrations of 3808 and 6820 ng g^{-1}, respectively (Zeng *et al.,* 2008; Cheng *et al.,* 2015a). It has been shown that the concentrations decrease from the city centre to the suburbs (Zeng *et al*., 2008; Xia *et al*., 2011b; Cheng *et al*., 2015a) because PAEs are derived mainly from municipal solid waste leachates, discarded plastic degradation, and chemical and building materials produced in urban areas (Xia *et al*., 2011b).

Brominated flame retardants (BFRs) are brominated organic chemical compounds including polybrominated diphenyl ethers (PBDEs), tetrabromobisphenol A (TBBPA) and hexabromocyclododecane (HBCD) and Tris–(2,3–dibromopropyl) isocyanurate (TBC). Urban soil samples collected from heavily industrialized and urbanized cities often have large concentrations of HBCDs, TBBPA and TBC, especially at waste and industrial sites (Tang *et al*., 2014). This reflects that BRFs are widely used in electronic circuits, building materials and textiles.

Distribution of heavy metals in urban soil. Many studies have investigated the distribution of heavy metals in urban soil (Table 2); they have shown that pollution occurs to different degrees and with considerable spatial heterogeneity. Sampling of urban soil needs to take into account the horizontal and vertical distribution of heavy metals. The spatial variation in density and distribution of pollution relates to different functional zones in cities. In general, areas with industry and traffic have the largest heavy metal concentrations, followed by residential, commercial and administrative areas, whereas recreational and scenic areas have relatively small concentrations (Luo *et al*., 2012c; Cai *et al*., 2013). Concentrations of heavy metals, like organic pollutants, are larger in the centre of cities and decrease radially to peri-urban areas, and closely align with traffic intensity and density of point sources (Wei & Yang, 2010; Luo *et al*., 2012c). There are three categories of horizontal distribution of heavy metal pollution: (i) point pollution like industrial and mining areas, (ii) belt pollution such as road transport and (iii) non-point pollution with regional deposition of dust or particles discharged by fuel combustion and industries. These types of pollution usually exist together. In addition to the horizontal distribution of heavy metals in topsoil, they can be distributed vertically, with the largest concentrations in the top layer of urban soil (Imperato *et al*., 2003; Luo *et al*., 2012b).

Heavy metal concentrations in urban soil have been summarized in previous studies. Table 2 indicates heavy metal contamination of urban soil in typical cities around the world. The concentrations of different metals in urban soil vary widely, generally in the order of Zn > Cr,Pb,Cu > Ni > As> > Cd,Hg (Table 2). The background values are also in the same order, suggesting they are of geogenic origin, such as soil parent materials. The exception is Pb, which is mainly from anthropogenic activities (Laidlaw & Filippelli, 2008). The origin of heavy metals is an important factor to consider with metal concentrations in urban soil. The mean concentrations of heavy metals in urban soil, in particular As, Cd, Cu, Pb and Zn, were distinctly larger than the corresponding background values, indicating that anthropogenic activities have had an important effect on their concentrations. Metal concentrations were large in industrial and urbanized cities where metals were deposited from the emissions of traffic, power plants and industrial processes (Cheng *et al*., 2014; Zhao *et al*., 2014). Traffic emission can contribute to large heavy metal concentrations in road dusts. Concentrations of Cd, Cu, Pb, Zn and Ni in road dusts depended on the traffic volume and were greatest close to the main highways. The major sources of heavy metals in road dusts varied greatly, with Ni and Cu mainly from exhaust emissions and brake abrasion and Zn from vehicle emissions and wear of tyres (Duong & Lee, 2011); they can enter urban soil through dry and wet deposition. Soil samples collected from roadside fields had large Cu, Pb and Zn concentrations, which indicated that atmospheric deposition from traffic was the main source of these metals (Imperato *et al*., 2003; Martin *et al*., 2015). Coal-burning power plants are the most important sources of anthropogenic heavy metal emissions, such as As, Cr, Hg, Ni, Pb and Zn (Stalikas *et al*., 1997; Mandal & Sengupta, 2006; Tian *et al*., 2015; Martin & Nanos, 2016). Chromium and Ni concentrations were enriched in surface soil close to coal-burning power plants (Stalikas *et al*., 1997). Similar results have been observed for other heavy metals such as As, Cu, Pb, Zn (Mandal & Sengupta, 2006) and Hg (Martin & Nanos, 2016). Airborne fly ash from the combustion of coal generally contains large concentrations of heavy metals that can precipitate to surface soil through wet and dry deposition. Industrial processes like steel production and electroplating can markedly increase heavy metals in urban soil and street dusts in urban areas (Banerjee, 2003). Larger concentrations of Cr, Ni and Cu occurred in street dusts in the urban industrial area where electroplating and rolling were carried out in Delhi, India, than Pb and Cd (Banerjee, 2003; Xiao *et al*., 2015) because it is the main industry. It can release large amounts of Cr, Ni and Cu into the environment during the production processes. In contrast, areas of Cd, Cu, Pb and Zn contamination have been found in urban soils affected by the steel industry (Xiao *et al*., 2015).

Table 2 Heavy metal concentrations (mg kg^{-1}) in urban and peri-urban soils

Location	Number of sampled urban surface soils	Depths / cm	Statistics	As	Cd	Cr	Cu	Hg	Ni	Pb	Zn	Reference
Aberdeen, UK	80	0–10	Range				–			–	–	Paterson *et al.* (1996)
			Mean				24			94.4	58.4	
Baltimore, USA	122	0–10	Range		0.003–3.12	8.3–794	6.41–216		4.85–336	4.07–5650	5.61–1110	Yesilonis *et al.* (2008)
			Mean		1.1	72	45		27	231	141	
Bangkok, Thailand	30	0–5	Range		0.05–2.53	4.3–57.4	5.1–283		4.1–52.1	12.1–269.3	3–814	Wilcke *et al.* (1998)
			Mean		0.29	26	42		25	48	118	
Beijing, China	30	0–5	Range				24.1–458		6.10–37.2	25.5–208	25.7–197	Chen *et al.* (2005)
			Mean				71.2		22.2	66.2	87.6	
Chicago, USA	57	0–15	Range		–	–	–	–	–	–	–	Cannon & Horton (2009)
			Mean		20	71	150	0.64	36	395	307	
Fuzhou, China	179	1–20	Range	1.39–42.2	0.03–1.65	7.70–116.5	3.6–265	0.020–6.24	1.70–33.1	22.8–1072	24.8–1073	Chen *et al.* (2008)
			Mean	8.28	0.3	40.1	39.4	0.77	16.0	89.8	158.6	
Ghaziabad, India	42	0–20	Range		0.2–4	356–2300	30–444		206–403	28–341	42–477	Chabukdhara & Nema 2013
			Mean		1.1	807	295		279	225	373	
Guanzhou, China	40	0–10	Range		0.028–2.41		6.47–394		6.17–77.6	18.5–348	10.1–610	Lu *et al.* (2007)
			Mean		0.5		62.6		25.7	109	117	
Guangzhou, China	28	0–5	Range		0.08–0.26	22.4–100.7	26.7–120.2		6.9–40.7	31.9–272.6	61.6–112.3	Gu *et al.* (2016)
			Mean		0.2	53.1	63.7		25.4	110.6	91.5	
Hongkong, China	236	0–15	Range		0.11–1.36	2.56–51.4	1.3–277		0.24–19.9	7.53–496	23.0–930	Lee *et al.* (2006)
			Mean		0.36	17.8	16.2		4.08	88.1	103	
Islamabad, Pakistan	834	0–23	Range		0–8.6		1.1–41.6		2.7–219.8	69–973	101.6–3255.9	Ali & Malik (2011)
			Mean		3.5		18		91	208	1643	
London, UK	35	0–15	Range				–			–	–	Thornton (1991)
			Mean				73			294	183	
Madrid, Spain	36	0–20	Range			4.67–57.19	11.30–171.6		4.89–15.03	15–598	76–309	Izquierdo *et al.* (2015)
			Mean			16.93	36.6		9.02	98	139	
Mexico City, Mexico	135	0–2	Range			50–265	15–398		20–146	5–452	36–1641	Morton-Bermea *et al.* (2009)
			Mean			117	101		40	140	307	
Moscow, Russia	36	0–10	Range		–	–	–		–	–	–	Plyaskina & Ladonin (2009)
			Mean		2.0	79	59		19	37	208	

Table 2 Continued

Location	Number of sampled urban surface soils	Depths / cm	Statistics	As	Cd	Cr	Cu	Hg	Ni	Pb	Zn	Reference
New York, USA	475	0–20	Range	0.9–76	0.1–11	4–262	5–1286		2–333	3–8912[a]	35–2352	Cheng *et al.* (2015b)
			Mean	12	1.6	55	110		32	600	327	
Novi Sad, Serbia	121	0–10	Range	2.1–11.1		10.6–50.6	4.4–459.2		10.2–74.2	8.9–999.1	46.2–193.8	Mihailović *et al.* (2015)
			Mean	6.5		28	38.8		28.7	82.3	100.3	
Palermo, Italy	70	0–10	Range			48.1–139.7	12.2–89.4			36.3–472.6	57.7–851.6	Manta *et al.* (2002)
			Mean			84.7	66.1			107.3	162.6	
Siena, Italy	30	0–20	Range		0.05–0.48		20.4–82.1			16.3–178	57.2–207	Nannoni & Protano (2016
			Mean		–		–			–	–	
Stockholm, Sweden	45	0–25	Range			6–145	7.3–153		2.1–21.3	2.4–444	18–408	Linde *et al.* (2007)
			Mean			35	30		17.4	30	144	
Talcahuano, Chile	140	0–10	Range				4.6–113			0.3–201	20.8–347	Tume *et al.* (2008)
			Mean				27.6			19.8	91.7	
Vigo, Spain	36	0–20	Range			33.5–195.3	22.6–208.2		11.5–60	34.4–259.1	59.0–234.4	Rodríguez-Seijo *et al.* (2017)
			Mean			68.6	66.1		32.0	96.3	149.03	
World soils			Median	6.0	0.35	70	30		50	35	90	Adriano (2001)
Continental crust			Mean	4.8	0.09	92	28		47	17	67	Rudnick & Gao (2003)
China soil quality standard[b]				30	0.3		100	0.50	50	300	250	CEPA (1995)
Dutch soil guidelines			Target value	29	0.8	100	36	0.3	35	85	140	Luo *et al.* (2012c)

[a] $n = 1652$.
[b] The environmental Quality Standard for Soils (GB 15618-1995) established by the China Ministry of Environmental Protection is intended for agricultural soil.
[c] Soil remediation values proposed by the Dutch Ministry of Housing, Spatial Planning and Environment (Luo *et al.*, 2012c).

Bioavailability of organic pollutants and heavy metals

The total amount of pollutants in urban soil determined by traditional chemical methods cannot provide a reliable assessment of the risks to human health. Even if we assume that 100% is bioavailable to human beings, a fraction only will be available and absorbed into the human body. Therefore, traditional methods inevitably overestimate the risk of soil pollution to humans (Alexander, 2000; Semple *et al*., 2004; Lal, 2015). Improving the assessment of bioavailability of pollutants in urban soil is fundamental for a better understanding of the health risks associated with exposure from urban soil. Thus, *in vivo* bioassay and *in vitro* chemical methods to measure bioavailability of organic pollutants and heavy metals have been developed in recent years (Denys *et al*., 2012; Bradham *et al*., 2014; Juhasz *et al*., 2014; Li *et al*., 2016).

The *in vivo* bioassays are more accurate than chemical methods. However, this method is still challenging because of the complexity of the metabolism, species and extraction *in vivo* of pollutants. *In vivo* studies with animal models have been used for predicting bioavailability of organic pollutants and heavy metals (Juhasz *et al.,* 2007; Denys *et al.,* 2012). For example, Juhasz *et al*. (2014) determined PAH bioavailability in creosote-contaminated soil with a mouse model and found that the absolute bioavailability was more than 65% regardless of the molecular weight of the PAHs. The study also used an *in vitr*o surrogate assay and observed 2000 times less bioavailability than with the *in vivo* one. This indicates that different methods can produce very different results and affect the perceived magnitude of the risk. Arsenic bioavailability with an *in vivo* swine assay varied from 11% to 75%, with generally small bioavailability in the soil with naturally elevated As concentrations. The largest bioavailability was in the soil collected in railway corridors, indicating that bioavailability is site specific (Juhasz *et al.,* 2007). In comparison to contaminated soil, natural soil has more buffering ability to heavy metals, which can make heavy metals strongly bound and associated with organic and inorganic ligands that can stabilize the heavy metals, and these chelated metal complexes have less bioavailability. A study on lead by Laidlaw & Filippelli (2008) indicated that anthropogenic Pb generally exists in very bioavailable carbonate and Fe and Mn hydroxide fractions, whereas that in natural soil is in residual or non-bioavailable fractions.

In vitro methods are a rapid, cheap and ethical approach that has been used widely for bioavailability assessments. Examples include simulation of the human intestinal microbial ecosystem (SHIME), the physiologically based extraction system (PBET), the Fed Organic Estimation human Simulation Test (FOREhST), the *in vitro* gastrointestinal method (IVG), the Dutch National Institute for Public Health and the Environment (RIVM), the Solubility Bioaccessibility Research Consortium (SBRC) assay and the Deutsches Institute Normunge.V. (DIN) (Cai *et al*., 2013; Cui *et al*., 2013; Juhasz *et al*., 2016). These *in vitro* methods have attempted to assess the bioavailability of organic pollutants (Tao *et al*., 2009; Siciliano *et al*., 2010) and heavy metals to humans. *In vitro* simulation methods can overcome the shortcomings of high costs, ethical constraints and time-consuming limitations associated with *in vivo* animal models. Many factors, however, such as soil particle size and total organic carbon (TOC) can affect the bioavailability assay with *in vitro* methods (Cachada *et al*., 2016). Consequently, urban soil from different sites has variable bioavailability (Juhasz *et al*., 2007; Semenzin *et al*., 2007; Ruby *et al.,* 2016), which should be integrated into the assessment of health risk to reduce the uncertainty. In addition, the accuracy of bioavailability of pollutants by the cheap and rapid *in vitro* methods can be similar to that achieved by *in vivo* models. Nevertheless, the *in vitro* methods still need to be validated by *in vivo* approaches, albeit fewer, in real scenarios. Combining both methods provides a more accurate assessment of the risks to human health from pollutants. Therefore, the *in vivo–in vitro* predictive model is often adopted to mimic the real-life situation (Juhasz *et al.,* 2014; Li *et al.,* 2014).

Human health risks of pollutants in urban soil

People living in cities cannot avoid exposure to urban soil; therefore, it is important to assess the potential risks to human health. In general, assessments of the risk to human health include two different procedures: the assessment of exposure and of the hazard (Swartjes, 1999). The exposure to pollutants from urban soil is mainly by two pathways: the soil–human pathway and soil–plant–human pathway. The soil–human pathway is mainly through soil ingestion and inhalation, which is controlled by the rates of intake of soil and dust, pollutant concentrations in soil and dust, body weight, duration of exposure and the bioavailability factor in the human body (Swartjes, 1999; USEPA, 2011). The soil–plant–human pathway relates to urban agriculture that produces food on small plots such as urban gardens in cities and small market gardens (Kessler, 2013; Wortman & Lovell, 2013). The assessment of hazard is based on dose–response data to estimate the health risk by incorporating an assessment of exposure (USEPA, 2006).

Accurate measurement of bioavailability is of vital importance to assess any risk to human health. Total concentrations are typically used to assess the health risks posed by pollutants (Cachada *et al*., 2012; Du *et al*., 2013; Kumar *et al.,* 2013). Urban soil pollution was often assessed by total concentrations rather than bioavailable values; therefore, health risk was inevitably overestimated, leading to increased costs of remediation (Luo *et al*., 2012a; Cachada *et al*., 2016). More recently, there has been wide assessment of bioavailability-based health risks from oral ingestion of organic pollutants or heavy metals in urban street dusts, school playgrounds, contaminated sites (Figure 1a), park soil (Figure 1c) and urban gardens (Reis *et al*., 2014; Izquierdo *et al*., 2015; Khan *et al*., 2016; Zhong & Jiang, 2017). Bioavailability adjustments have also been applied in assessments of health risks under the regulatory framework in some countries, including Australia and New Zealand (Ng *et al.,* 2010), Canada (Cachada *et al.,* 2009) and the USA (USEPA, 2007).

When the health risk of PAH-contaminated urban soil was evaluated by conventional methods, cancer risks greater than 10^{-4} were observed (USEPA, 2001), whereas the bioavailable-based cancer risk was within an acceptable range (10^{-6}–10^{-4}) for seven out of

18 soils (Juhasz *et al.,* 2016). Similar results have been found in sites contaminated by car dismantling, which showed a moderate potential risk of cancer for total PAHs, but it was small for bioavailable concentrations (Man *et al*., 2013). Similar trends are evident for heavy metals in urban soil, with the risk of cancer reduced considerably on average for the *in vitro* bioavailability methods (Luo *et al*., 2012b; Li *et al*., 2014; Kastury *et al*., 2017; Zhong & Jiang, 2017). More emphasis should be directed to research on the risks to health for children because they are more sensitive to pollutants and are at greater risk than adults (Gaspar *et al*., 2014; Zhong & Jiang, 2017). Children have more exposure per unit of body weight and there is more hand-to-mouth behaviour than with adults (Figure 1d); therefore, they are more vulnerable to pollutant exposure (Swartjes, 1999; Kessler, 2013). Ko *et al*. (2007) observed a significant correlation between children's hand-to-mouth frequency and lead ingestion in an epidemiological study. Lead poisoning from exposure through dust inhalation and soil ingestion is a typical issue for children and has been widely investigated (Thornton *et al*., 1990; Lanphear & Roghmann, 1997; Ryan *et al*., 2004; Laidlaw & Filippelli, 2008). Lead exposure occurs mainly by ingestion and it is absorbed in the intestine (Mielke & Reagan, 1998). About 50% of ingested Pb finds its way into tissues of children, compared with 5% for adults, because of their less-developed gastrointestinal system (Laidlaw & Filippelli, 2008). Lead exposure in children can cause health problems such as mental retardation, learning disorders, attention-deficit and hyperactivity disorders, and so on (Nigg *et al.,* 2008). Epidemiological studies have indicated a strong relation between lead-polluted indoor dust and blood lead concentrations in children (Lanphear & Roghmann, 1997; Lanphear *et al*., 1998). In view of children's sensitivity and vulnerability, action at the national level is required to prevent and reduce prenatal and childhood exposures to lead (Bellinger *et al.,* 2017).

Urban gardens for food production are of vital importance to urban health because they produce food locally, but they also facilitate exposure to contaminated soil (Kessler, 2013; Wortman & Lovell, 2013; Luo *et al*., 2015). In addition to direct soil ingestion (Izquierdo *et al*., 2015; Guney & Zagury, 2016; Spliethoff *et al*., 2016), there are potential associated health risks from vegetables and crops grown on contaminated sites because they might assimilate contaminants in greater concentrations than those produced in less polluted environments (Sipter *et al*., 2008; Kessler, 2013; McBride *et al*., 2014). This is especially likely with soil of contaminated brownfield sites (Defoe *et al*., 2014). Urban gardens can expose children to Pb through soil ingestion and the food chain. A survey of 140 gardens demonstrated that intake of homegrown produce contributed 3% only of total daily exposure of lead in children, whereas ingestion of fine soil particles accounted for 82% of the daily exposure (Clark *et al*., 2008). Over 60 and 10% of spinach grown in urban gardens near to ferroalloy plant activity exceeded by two- to three-fold the acceptable European maximum Pb and Cd concentrations, respectively (Ferri *et al*., 2015). In contrast, the potential risk of Cd, Cu, Ni and Zn exposure from the consumption of home-produced vegetables has been assessed in the UK; the risk assessment indicated that food cultivated on 92% of urban gardens in the UK presented minimal risk to the average person (Hough *et al*., 2004). However, the more sensitive members of the population, such as children, the elderly and the infirm, were subject to greater risk and more attention should be paid to them.

Many factors, such as the extrapolation of bioavailability from large to small doses in risk assessment, from animal models to human beings, from one single study focused on one specific site, exposure routes, and so on, can affect the uncertainty of assessment of human health risk (Keller *et al.,* 2002; Naidu *et al*., 2015). More emphasis should be placed on site-specific considerations, which should be incorporated into assessments of health risk (Sipter *et al.,* 2008; Ollson *et al.,* 2014; Zhong & Jiang, 2017) because background concentrations of pollutants, their bioavailability and pollutant types are site specific and can strongly affect the level of risk. The use of site-specific data on the bioavailability of contaminants in urban soils has been encouraged when available; for example, in the recently released National Environmental Protection Measure of Australia (NEPC, 2013).The incorporation of estimates of site-specific bioavailability into risk assessments might reduce uncertainty in determining contaminant risk at a given site and minimize remediation costs. More epidemiological studies should be carried out to investigate the relations between exposure to a given pollutant and actual health effects, especially long-term exposure to small doses and effects of exposure of children and other sensitive groups.

Urban green space and human health

One of the functions of urban soil is to support urban green space, which includes parks, squares, gardens, green corridors, wetlands, plant nurseries, and so on. Although urban green space is critical to urban life, rapid urban development is having an unprecedented effect on it. Future urban soil research should also include how soil management can improve and optimize urban green space. Different vegetation types might require particular soil conditions; thus, urban green space design should consider soil properties that could be amended to suit specific vegetation types. Remediation of contaminated sites in urban environments should consider combining risk mitigation with the creation of green spaces. For example, Obrycki *et al*. (2017) evaluated three management options for an urban site contaminated with Pb and PAHs. They found that the soil capping placed over the site reduced surface soil Pb from up to 5149 to 12.4 $mg\,Pb\,kg^{-1}$ and benzo(a) pyrene content from up to 5.50 to $0.99 \pm 0.41\,mg\,kg^{-1}$, which reduced human exposure to these contaminants markedly. Furthermore, urban waste management can also play a role in improving green space, such as the production of biochar from green waste and the use of biochar to improve soil physiochemical properties (e.g. nutrients and soil structure) to improve the quality of vegetation in green spaces.

Microbial diversity of urban soil and human health

The 'biodiversity hypothesis' proposes that richness in microbial diversity in the environment protects humans from allergic and autoimmune diseases (von Hertzen & Haahtela, 2006). Reduced contact between humans and natural environmental biodiversity, including environmental microbiota, might decrease the commensal microbiota of humans, leading to the increased likelihood of immune dysfunction and disease (Hanski *et al*., 2012). The microbial diversity encountered in the environment comes mostly from the soil, plants and animals. Almost all species (including humans), obtain their microbiota from soil (Viinanen *et al*., 2005). They have been living on the planet for 3.7 billion years (Papineau, 2010). Soil-derived bacteria are ubiquitous and bacteria commonly found in soil and water are the most abundant in outdoor air (Kembel *et al*., 2012; Dunn *et al*., 2013). Modern buildings in the urban environment, quite different from the natural environment, are constructed with mainly synthetic materials, plastics and concrete. Timber, although used in buildings, is treated with adhesives or biocides. Therefore, these modern structures do not become colonized easily with bacterial communities. These built environments are the primary habitat of humans, and the microbial communities within buildings have been linked to human health through immune regulation and proliferation of human pathogens. Some studies have shown that the diversity of both fungal and bacterial exposure might play a role in protective effects (Ege *et al*., 2011; Heederik & von Mutius, 2012). It has already been shown that the diversity and abundance of microorganisms in urban environments can be reduced (Alenius *et al*., 2009).

Bacterial communities in the near-surface atmosphere of suburban areas had the smallest relative abundance of soil bacteria compared with the corresponding near-surface atmosphere from agricultural fields and forests (Bowers *et al*., 2011). It was estimated that humans in industrialized countries, especially urban populations, spend more than 90% of their lives in buildings (Custovic *et al*., 1994; Kelley & Gilbert, 2013). The exposure to microbes, particularly in soil, has been dramatically reduced with increased urbanization (von Hertzen & Haahtela, 2006). Reduced microbial biodiversity might also cause a public health problem such as asthma and other atopic diseases (von Hertzen & Haahtela, 2006). For example, the proportion of farmers among the population has decreased considerably from 17.3% in 1970 to 4.9% in 2000 in Finland because of urbanization. During the same time, the occurrence of allergic rhinitis increased from 0.1 to 8.9% (von Hertzen & Haahtela, 2006). The proportion of the population that has a connection with soil has decreased with the increased use of asphalt (10-fold in three decades) and reduction in outdoor activities. Both reduced personal exposure to microorganisms because of sedentary lifestyles and decreased microbial concentrations in urban areas increase the risk of atopy (Brisbon *et al*., 2005; von Hertzen & Haahtela, 2006).

Within cities, biological communities are usually radically altered in terms of species composition, abundance, richness and evenness. The biodiversity of some taxonomic groups is well documented for plants, birds and arthropods in temperate cities (McKinney, 2008; Luck & Smallbone, 2010; Raupp *et al*., 2010). Microbial diversity of urban soil is affected substantially by human-induced disturbance through a variety of abiotic and biotic changes in land use and land cover, such as movements of soil and construction of buildings. To understand the changes in soil microbial diversity in cities, the effects of urbanization on their abundance, diversity and species richness are reviewed here. We searched the Web of Science (ISI databases) for words ‘urban, soil, biodiversity’ in abstracts, and 342 published articles were identified. Each paper was examined with a focus on the microbial biodiversity related to urbanization or within the urban area. However, there were very few studies on microbial diversity of urban soil compared with the many on natural soil (Cousins *et al*., 2003; Hall *et al*., 2009), and also few on the effects of urbanization on microbial diversity (Bowers *et al*., 2011; Kembel *et al*., 2012).

Urban environments are characterized by localized increases in ambient air temperature known as the urban heat island effect (Oke, 1989). The temperatures at the centre of Baltimore in the USA were 5 to 10°C warmer than in suburban residential areas (Brazel *et al*., 2000). The increased air temperature also caused an increase in soil temperatures (Pouyat *et al*., 2007). Mean air temperature in the urban area increased by 2.1°C and soil temperature increased by 0.7°C compared with corresponding rural sites. The difference in soil temperature could affect soil microbial activity considerably (Carreiro *et al*., 2009).

Urbanization has been shown to have a marked effect on soil properties and distribution of nutrient elements, causing large concentrations of nitrogen and phosphorus in the soil (Hobbie *et al*., 2017). The introduction of buildings and impervious surfaces alters the allocation and accumulation of nutrients in soil (Noe & Hupp, 2005). Carbon, nitrogen and phosphorus concentrations were enhanced in permeable sites (Noe & Hupp, 2005; Pouyat *et al*., 2006; Pickett & Cadenasso, 2009). However impervious areas prevent the exchange of gas, water and nutrients between the soil and other environmental compartments, resulting in small soil microbial biomass C and N compared with forest and bare lands (Yang *et al*., 2001; Zhao *et al*., 2012). The abundance of predatory and omnivorous nematodes tended to be small in urban soils (Pavao-Zuckerman & Coleman, 2007). Trace elements, organic matter (Santorufo *et al*., 2012) and pH (Zhao *et al*., 2013) have also been shown to affect composition of the invertebrate community (Santorufo *et al*., 2012), soil microbial biomass and microbial functional diversity (Zhao *et al*., 2013), and to decrease arbuscular mycorrhizal fungal colonization and its effect on community composition (Egerton-Warburton & Allen, 2000; Beall & Fox, 2009; Xu *et al*., 2014).

The immunomodulatory role of saprophytic bacteria in the soil is being increasingly recognized (von Hertzen *et al*., 2011; Rook, 2013), and disruption of the relation between humans and soil might have important consequences as a result of urbanization (von Hertzen & Haahtela, 2006; von Hertzen *et al*., 2011; Rook, 2013). People living in apartments have been shown to have less personal exposure to microorganisms, partly because of decreased outdoor activities, than those in houses (Horner *et al*., 2004). Furthermore, individuals living in urban environments have less diversity of bacteria on their skin than those from rural areas, which might have a negative effect on their immune system (Hanski *et al*., 2012).

Several studies have shown that exposure to soil microorganisms reduces the prevalence of allergic diseases (Matricardi & Bonini, 2000; Rook, 2010; Hanski *et al*., 2012; Brevik & Sauer, 2015). The degree of greenness in urban environments such as forest and agricultural land within 2–5 km of homes was inversely and significantly associated with the risk of atopic sensitization in children (Ruokolainen *et al*., 2015). The disconnection between humans and soil could have important consequences (von Hertzen & Haahtela, 2006; Rook, 2013; Ruokolainen *et al*., 2015) and deserves more attention.

Another important issue is the transport of soil microorganisms into the aquatic environment (potential sources of drinking water). Soil microorganisms can be transported from soil by runoff and leaching into lakes, rivers or ground water. Microorganisms on soil particles are also transported into such waters by erosion. Water from the above sources is often used for drinking by rural people, frequently without any chemical or other treatment. For urban populations, however, drinking water is usually treated to remove contaminants and microbes, thus reducing the contact with soil microorganisms and exposure to pathogens. Some studies have suggested that consumption of river water in rural areas possibly provides more protection against atopy than consumption of treated water in urban areas (Cooper *et al*., 2004; Haileamlak *et al*., 2005). Almost 70% of the world's population will be living in urban environments by 2050 (Luo *et al*., 2012c); therefore, this might lead to more allergic diseases because of reduced contact between humans and soil microbiota.

Antibiotic resistance genes (ARGs) are emerging contaminants and posing a potential human health risk worldwide (Wang *et al*., 2014, 2015b; Chen *et al*., 2016; Steffan *et al*., 2018; Zhu *et al*., 2017c). Soil has been regarded as a rich source of ARGs because of the large microbial community and diversity of antibiotic-producing microbes in soil (Su *et al*., 2015). The ARGs in urban soil can be enriched by human activities. For example, irrigation with reclaimed water in urban environments is becoming popular because of rapid urbanization and water shortages (Wu *et al*., 2010; Zhu *et al*., 2013; Chen *et al*., 2014). Antibiotics and heavy metals, which exist in reclaimed water or sewage sludge, could increase antibiotic resistance in the environment (Pei *et al*., 2006; Pruden *et al*., 2013; Su *et al*., 2014; Xie *et al*., 2018), and also relatively small concentrations of heavy metals might be sufficient to induce bacterial antibiotic resistance through co-selection (Mulder *et al*., 2011). This is because diverse genes in bacteria for resistance to heavy metals are often clustered next to multiple antibiotic resistance genes on the same genetic element (Zhu *et al*., 2017a). Co-selection for resistance genes can occur when heavy metal is introduced for which resistance to that agent and other resistance genes are genetically clustered (Zhu *et al*., 2017a). The use of reclaimed water for irrigation in urban environments has been found to result in the enrichment of ARGs in soil (Zhu *et al*., 2013; Wang *et al*., 2015b) and urban river sediments (Lu *et al*., 2010; Chen *et al*., 2014). There is increasing concern about the use of reclaimed water in urban parklands because human pathogens such as *Enterococcus faecium* might potentially harbour some ARGs (Arias & Murray, 2009) and pose a direct health risk to the urban population (Wang *et al*., 2015b).

Before urbanization, humans lived in close contact with soil, either directly or indirectly through food, water and air. Natural exposure to microbes, particularly in soil, has been considerably reduced as a consequence. This disconnection between humans and soil microorganisms affects human health by reducing microbial biodiversity of human bodies and more effort is needed in future to understand the relations between soil microbes and human health.

Urban soil management and policy

Urban soils are key components of urban ecosystems; therefore, their successful management is important to achieve urban sustainability. One of the main tasks of soil management is to maintain soil quality to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation (Doran & Zeiss, 2000). Soil erosion, chemical contamination, physical degradation and loss of biodiversity are important types of soil degradation in the general environment (Chen *et al*., 2002; Lal, 2015). Because of rapid urbanization, urban soil is also becoming degraded by chemical contamination (Zhang *et al*., 2003; Luo *et al*., 2012c; Cachada *et al*., 2016), which can be a threat to human health through ingestion of soil particles. Contamination of urban soil should be considered during urban management practices. Land-use planning can be effective in preventing humans from exposure to potential contaminants, especially in urban agriculture (Barton, 2009; Lovell, 2010). In general, contaminated land can be allocated for industrial and commercial uses that have less stringent standards, and clean or less contaminated sites should be designated for food production in urban agriculture and children's playgrounds. Technical standards and risk-based approaches (including risk characterization and risk assessment) for managing contaminated land should be developed for better re-use of the contaminated land. In 2014, China released the new national technical standards for monitoring, risk assessment and remediation of contaminated land in urban areas (Chiang & Gu, 2015). However, remediation of soil contamination is costly (Ren *et al*., 2015); therefore, detailed soil monitoring and land-use planning might provide cost-effective strategies to reduce the risk to human health from contamination. Nevertheless, management options will vary between types of contamination and countries, and the contaminated sites should be remediated to reach the standards to qualify for further uses. Different urban land uses, including industrial, residential, transport, green space and urban gardens, can lead to variable human exposure to contaminants (Patz *et al*., 2004; Xia *et al*., 2011a). Therefore, the use made of land should consider any risk to human health. Microbial diversity is an important indicator of soil health, and has been shown recently to be altered during urbanization (Hall *et al*., 2009; Zhao *et al*., 2013). The relation between soil microbial diversity and human health is not well established, but increasing evidence suggests there are implications for human health through complex interactions between human and environmental microbiomes, and loss of soil biodiversity can have a substantial negative effect on human health globally (Wall *et al.,* 2015).

Establishment of a soil policy is another way to maintain soil quality and prevent its degradation. There are many soil policies, such as the Global Soil Partnership (GSP) launched in 2012 by the United Nations Food and Agriculture Organization, but so far there are no specific policies for managing urban soils. These policy initiatives are related to both agricultural, natural and urban soils; therefore, further work must be carried out to develop a soil policy framework oriented to urban soils because they have unique characteristics and ecosystem services.

Conclusion

Urban soils are a vital component of the urban ecosystem and are closely related to human health. The problems identified with urban soils are traditional organic and inorganic pollutants and ARGs. Relations between pollutants, ARGs and human health have been well established, but there remains much to do. Previous studies on pollutants in urban soils focused on their distribution, bioavailability evaluation and health risks, which indicated that inhalation, dermal contact and ingestion of soil and food grown in urban soil are the main exposure routes of pollutants. However, human beings are seldom exposed to a single pollutant in urban soils, but more often to multiple chemicals simultaneously. Integrating bioavailability pollutants in urban soils and multi-chemical exposure into the framework of health risk assessment can improve our understanding of their holistic risks. The health effects of long-term low-dose exposure to pollutants through urban soil directly or indirectly are not well known. Therefore, epidemiological studies should be carried out to elucidate the long-term dose exposure of pollutants in urban soils. In addition to traditional chemical pollutants, future studies should also consider emerging contaminants, such as ARGs, and the relations between urban green spaces and human health, although some indirect effects have been observed. Effective urban soil management and policy should also be established to ensure human health in the context of global rapid urbanization.

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