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RESEARCH

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# Low presence of potentially toxic elements in Singapore urban garden soils

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## Abstract

**Background:** Urban agriculture is potentially an important piece of the food security puzzle for a rapidly growing urban world population. Community gardening is also promoted as a safe and viable form of exercise for aging populations in crowded settings where opportunities to participate in other action activities may be limited. Knowledge of potential site-specific health risks to environmental contaminants is important in dialogues promoting urban farming.

**Methods:** We assess the pseudo-total concentrations of selected potentially toxic elements (PTEs) in the soils of community gardens, public parks, and woodlands in the tropical urban island nation of Singapore. We compare concentrations of cadmium, copper, lead, and zinc with amalgamated risk guidelines to form a baseline understanding of the level of contamination in these spaces. We also perform providence tracking with lead isotopes to identify potential sources of contaminants.

**Results:** All pseudo-total concentrations of Cd, Cu, Pb, and Zn in the soil were below threshold concentrations considered to represent substantial risk. Further, PTE concentrations in gardens were largely equivalent to those found in community parks and woodlands, but the geographical distribution varied. Provenance tracking with Pb isotopes indicated Pb in gardens was both anthropogenic and natural, but spatially variable. The lack of strong spatial clustering of areas with the highest PTE concentrations was inconsistent with a common point source of contamination. However, the correlation between Cu and Zn suggest a common source for these elements, such as road/trafficking or atmospheric deposition.

**Conclusion:** We find limited risk of urban gardeners to exposure to Cd, Cu, Pb, and Zn—elements that are commonly abundant in urban settings with dense transportation networks and substantial industrial activities. The low levels of PTEs are encouraging for the promotion of urban farming for food production and leisure in this dense urban setting. However, as concentrations were low, we did not assess bioavailability and bioaccessibility of the PTEs. These assessments would need to be determined in cases of with higher levels of contamination to provide a more thorough consideration of actual human risk.

**Keywords:** Urban agriculture, Green spaces, Parks, Woodlands, Environmental risk assessment, Heavy metals

## Background

### Urban gardening risks

Urban agriculture activities are increasingly being incorporated into the planning of cities worldwide (Taguchi and Santini 2019; Edmondson et al. 2020; Langemeyer et al. 2021), with urban hubs such as Hanoi, Havana, New York, and Singapore being examples of productivism (Hou 2017; Martin-Moreau and Ménascé 2020). Embedded within UN Sustainable Development Goal 111.3 is the importance of agriculture as part of sustainable cities

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(Mansfield and Mendes 2013; McClintock et al. 2018): "countries should aim to work to enhance inclusive and sustainable urbanization for participatory, integrated, and sustainable human settlement planning and management". Nearly a decade ago, Orsini et al. (2013) recognized that the food security benefit of urban agriculture was evidenced by 100–200 million people providing fresh horticultural goods to city markets. Arguably urban agriculture in developing countries plays a critical role in food availability and affordability, but there is need for improved understanding potential risks and limitations (De Bon et al. 2010; Zezza and Tasciotti 2010; Hamilton et al. 2014). While urban agriculture may not achieve food self-sufficiency per se, it is an alternative way of "feeding citizens differently" (Martin-Moreau and Ménascé 2020). Urban gardening is also a practical means of exercise, leisure, therapy, and perhaps supplementary income, in crowded urban spaces for all citizens (Orsini et al. 2013; Kalantari et al. 2018; Rogge et al. 2018; Winkler et al. 2019). Regardless of economic setting, it is increasingly important to preform foundational work addressing the potential realm of risks that await the growing wave of urban farmers, many of which may be advanced in age, and therefore in some cases, susceptible to the adverse effects of contaminated urban soils (Beckie and Bogdan 2010; Hamilton et al. 2014; Yang and Na 2017).

Pollution research worldwide has revealed the potential for people engaged in outdoor urban activities to be exposed to various PTEs that originate from past and present anthropogenic sources such as transportation networks, energy production systems, mining and industrial activities, and improper handling of waste (McLaughlin et al. 2000; Mitchell et al. 2014; Jean-Soro et al. 2015; Spliethoff et al. 2016; Bechet et al. 2018; López et al. 2019). Elevated PTE concentrations in urban soils are often legacies of past activities conducted before a garden was established (Kessler 2013; Wortman and Lovell 2013). While some elements such as copper and zinc are essential in very small amounts for normal health, they may be toxic when high bioavailable fractions are encountered (De Vries et al. 2007).

Exposure to high concentrations of PTEs can cause a variety of health problems including dizziness, nausea, cramping, coughing, diarrhea, and vomiting (Jaishankar et al. 2014; Gautam et al. 2016). Long-term and repeated ingestion may induce element-specific maladies such as hypertension and weakening of bone density (Cd), renal dysfunction (Pb, Cd), anemia (Zn), and other afflictions (ATSDR 2015; Satarug and Moore 2004; Swaddiwudhipong et al. 2011; Cabral et al. 2015). The potential for adverse health risks stemming from a seemingly healthy and sustainable activity is one aspect of the "dark side of urban gardening" (Kessler 2011; Mancebo 2016)—however, others conclude the risk is sometimes more perceived than real (Brown et al. 2016; Lupolt et al. 2021). Nevertheless, assessing the potential risk is beneficial to promoting healthy outdoor activities.

#### Human urban gardening exposure chain

Community risk to exposure to potentially toxic elements (PTEs) while working in urban gardens has been a topic of concern for more than half a century (Chaney et al. 1984). Lead in the USA during the 1960s and 1970s, for example, was found in very high concentrations owing to contamination from a variety of sources including automotive exhaust, lead-based paint degradation, smelter emissions, sewage sludge, pesticides, and element-enriched manure fertilizers (Preer and Rosen 1977; Spittler and Feder 1979). Similarly, Purves (1966) found urban garden soils in Edinburgh and Dundee to be elevated in copper and boron, probably as a result of atmospheric deposition of soot, which was also applied as a garden soil amendment. Warren et al. (1971) later reported elevated concentrations of Cd, Cu, Pb, and Zn to be a growing concern for vegetable gardening in urban and industrial areas of Canada. Over the years, dozens of studies have examined the levels of these and other potentially harmful elements in urban and peri-urban soils worldwide (Table 1). The high element concentrations in the soils of many of the latter studies was often linked to legacy activities (e.g., Pb from paint; Cu, Pb, Zn from smelters and mines), trafficking phenomena (e.g., vehicle emissions, vehicle and tire wear), deposition following the release from a variety of urban and industrial sources, irrigation with wastewater, inter alia (Table 1; Additional file 1: Table S1).

These prior works have demonstrated that the "human urban exposure chain" to PTEs in the soil is a complex issue involving behavioral, biochemical, developmental, geophysical, physiological, and societal variables (cf. Clark et al. 2006; Schram-Bijkerk et al. 2018). To frame our research, which is a baseline assessment of Cd, Cu, Pb, and Zn in the soils of urban gardens in Singapore, we portray this exposure chain as a complex dynamic system. In the following narrative, the italicized terms are key variables in a simplified causal loop diagram shown in Fig. 1.

Element concentrations in the soil are influenced by several factors: mineralogy and geochemical composition of the parent rock, organic element content, particle size distribution, soil horizonation, drainage, vegetation, management, and anthropogenic inputs (Kabata-Pendias 1993; Wuana and Okieiman 2011). The elevated presence of a PTE in an urban soil could be naturally part of the background *geogenic signature* or associated with

**Table 1** Ranges of Cd, Cu, Pb, and Zn concentrations reported for surface soils in (peri)urban garden soils worldwide (mg/kg; total/pseudo-total concentrations)

| Location; garden type | Cd | Cu | Pb | Zn | Issues mentioned | References |
|---|---|---|---|---|---|---|
| Belo Horizonte, MG, Brazil; urban gardens | 0.11–0.20[a] | 16–42[a] | 11–30[a] | ND | Proximity to potential sources | Dala-Paula et al. (2018) |
| Jos, Nigeria; cultivated soils | 0.08–0.19 | 1–4 | 3–5 | 8–13 | Urban ash (deposition, soil amendment) | Pasquini and Alexander (2004) |
| Nanjing, China; peri-urban ag soils | 0.08–0.15[a] | 21–30[a] | 31–33[a] | 53–71[a] | Deposition; industrial effluent | Huang et al. (2006) |
| Salamanca, Spain; urban gardens | 0.07–0.33 | ND | 10–26 | ND | Fertilizers (Cd); trafficking (Pb) | Sanchez-Camazano et al. (1994) |
| Visakhapatnam, India; semi-urban | ND | 11–31[a] | 15–48[a] | 43–50[a] | Proximity to industry | Srinivas et al. (2009) |
| Alice, S Africa; home garden soils | 0.10–0.80[a] | 5–8[a] | 5–14[a] | 18–53[a] | Minor contamination | Bvenura and Afolayan (2012) |
| Toronto; community gardens | 0.27–0.40 | 22–33 | 19–96 | ND | Trafficking processes | Wiseman et al. (2013) |
| Guangzhou, China; urban veg gardens | 0.02–1.9 | 6–25 | 25–55 | 57–195 | Atmospheric deposition | Xiong et al. (2016) |
| Namyangju, Korea; urban field near road | bdl | 15–30 | 14–18 | 61–77 | Inconclusive; some road effects | Kim et al. (2017) |
| Thessaloniki, Greece; Peri-urban farm plots | 0.16–0.26 | 18–40 | 16–37 | 3–124 | Proximity to industry was NOT an issue | Vousta et al. (1996) |
| Wuxi, China; peri-urban ag soils | 0.14–0.17[a] | 35–37[a] | 38–42[a] | 104–112[a] | Deposition; industrial effluent | Huang et al. (2006) |
| Rostock, Germany; urban garden soils | 0.05–0.60 | 1–59 | 3–57 | 3–95 | Diffuse; highest in city center | Kahle (2000) |
| Sakai, Osaka; (peri)urban veg farms | ~0.25–0.90[a] | ~10–30[a] | ~10–35[a] | ~30–125[a] | Redistribution of legacy metal; trafficking | Komai and Yamamoto (1982) |
| Skarżysko-Kamienna, Poland; allot. gardens | 0.30–5.8 | 3–15 | 7–33 | 2–99 | Inconclusive; probably industrial | Świercz and Zajęcka (2018) |
| Kabul, Afghanistan; urban gardens | 0.15–0.20 | 50–66 | 25–33 | 113–184 | Traffic, war, contaminated irrigation water | Safi and Buerkert (2011) |
| New Brunswick, CA; residential gardens | ~0.02–0.05 | ~18–34 | ~27–115 | ~77–145 | Industrial legacy; long-term deposition | Pilgrim and Schroeder (1997) |
| Baoding, China; sewage-irrigated soils | 0.17–0.40 | 26–44 | 27–58 | 113–250 | Sewage irrigation water | Xue et al. (2012) |
| Szeged (Bakto), Hungary; urban gardens | 0.27–2.86 | 19–580 | 5–61 | 33–199 | Traffic; pesticides | Szolnoki et al. (2013) |
| Jiangsu cities (China); peri-urban ag soils | ND | 38–45[a] | 88–120[a] | 45–47[a] | Industrial, urban processes | Hao et al. (2009) |
| Manitoba province, CA; urban ag soils | 0.8–1.7 | 16–37 | 24–43 | 62–110 | Road phenomena most likely | Mills & Zwarich (1975) |
| Beijing, China; gardens and green spaces | 0.14–0.24[a] | 28–50[a] | 30–74[a] | 78–118[a] | Age, traffic processes, urban sources | Beavington (1975) |
| Torun, Poland; botanical garden topsoil | bdl | bdl–30 | bdl–148 | 6–142 | Natural soil genesis | Charzyński et al. (2018) |
| Cowra (Au); commercial/residential gardens | 0.01–0.80 | 1–96 | 7–12 | 9–336 | No main pollution sources | Kachenko and Singh (2006) |
| Beijing (Tongzhou); irrigated soils | 0.41–1.71 | 22–49 | 48–53 | 136–176 | Waste-water irrigation | Khan et al. (2008) |
| Varnasi city, India; suburban garden soils | 0.90–5.65 | 3–30 | 9–25 | 22–208 | Waste-water irrigation | Sharma et al. (2006) |
| Koszalin, Poland; allotment soils | 0.04–0.53[a] | 8–34[a] | 8–57[a] | 78–220[a] | Natural signature; garden management | Bielicka-Giełdoń et al. (2013) |
| Singapore; urban gardens | bdl–3.2 | 10–77 | bdl–37 | 71–164 | Proximity to industry/roads & industry; diffuse urban signal | This study |
| Mumbai, India; railway gardens | 0.1–1.1 | 60–231 | 3–7 | 15–111 | Irrigation water; pesticides; railway signal | Vazhacharickala et al. (2013) |

**Table 1** (continued)

| Location; garden type | Cd | Cu | Pb | Zn | Issues mentioned | References |
|---|---|---|---|---|---|---|
| Ho Chi Minh City; urban ag lands | ND | 8–70 | bdl–27 | bdl–272 | Amendments (fert/pest); urban waste | Tran et al. (2020) |
| Vancouver, Ca; community garden | ND | 36–63 | 22–82 | 51–133 | Legacy deposition, roads, industry | Oka et al. (2014) |
| Kano City, Nigeria; urban veg gardens | 2.3–4.0 | 5–14 | 2–17 | 17–233 | Irrigation wastewater | Abdu et al. (2011) |
| Oregon (Corvallis, Portland); food gardens | bdl–0.57 | 25–109 | 0.2–347 | ND | Not determined; compost concerns | Nelson (2018) |
| Seoul, Korea; rooftop gardens | 0.9–1.9 | 6–118 | 6–21 | 51–268 | Deposition (local and long-distance) | Kim et al. (2015a, b) |
| Shangdong towns (China); garden soils | 0.03–0.99 | 1–160 | 9–27 | 25–223 | Trafficking; industry; waste irrigation | Liu et al. (2011) |
| Zurich; urban gardens | ND | 16–34 | 19–1076 | ND | Legacy (esp. Pb); deposition | Tresch et al. (2018) |
| Nanjing, China; peri-urban ag soils | 0.19–0.58 | 22–58 | 21–150 | 76–181 | Fertilizers; deposition; industry emissions | Hu et al. (2018) |
| Guelph, Ontario, CA; Community gardens | 0.38–1.84 | ND | 16–151 | 97–503 | Low-income area; legacy | Montaño-López and Biswas (2021) |
| Kaduna, Nigeria; river bank gardens | 0.07–0.95 | 11–36 | 26–154 | 53–218 | Deposition; vehicular emissions | Agbenin et al. (2009) |
| Lisbon area; urban allotment soils | ND | 10–62 | 1–110 | 33–146 | Uncertain; roads/airport likely | Leitão et al. (2018) |
| Nanjing; urban/suburban gardens | ND | 23–72 | 35–84 | 100–283 | Decreases from urban to rural | Fang et al. (2011) |
| Kunming, China; peri-urban garden | 0.53–0.80 | 87–208 | 42–62 | 77–148 | Geogenic; manure inputs; uncertain | Zu et al. (2014) |
| Bragança, Portugal; peri-urban gardens | $0.9 \pm 0.2^a$ | $128 \pm 28^a$ | $56 \pm 78^a$ | $89 \pm 28^a$ | Organic farming; new city sources | Arrobas et al. (2017) |
| Dar es Salaam, Tanzania; urban ag | $0.15–0.26^a$ | $4–8^a$ | $9–33^a$ | $13–57^a$ | Uncertain; contaminated irrigation water | Kibassa et al. (2013) |
| Delhi, India; peri-urban soils | $2.28–2.51^a$ | $24–77^a$ | $21–59^a$ | $73–175^a$ | Wastewater irrigation, fertilizer, deposition | Bhatia et al. (2015) |
| Kayseri, Turkey; urban veg gardens | $1–48-2.11^a$ | $44–68^a$ | $67–108^a$ | $128–179^a$ | Not determined | Tokalioglu et al. (2006) |
| Danang, Vietnam; urban ag/ garden soils | 0.1–0.7 | 30–495 | 1–5 | 53–562 | Air-borne pollutants; ag amendments | Thuy et al. (2000) |
| Johannesburg, SA; school veg gardens | ND | bdl–80 | 27–100 | 91–427 | Proximity to pollution sources | Kootbodien et al. (2012) |
| Hanoi, Vietnam; urban farm soils | 3.9–4.5 | 195–230 | 125–145 | 187–263 | Contaminated irrigation water | Nguyen et al. (2010) |
| Harare, Zimbabwe, metropolitan farms | 1.0–3.4 | 21–94 | 17–59 | 42–228 | Irrigation with wastewater | Mapanda et al. (2005) |
| Hyderabad, Pakistan; unclear | $1.6–4.3^a$ | 11–32 | 21–67 | 105–209 | Wastewater irrigation | Jamali et al. (2007) |
| Kano City, Nigeria; garden soil | $1.4–9.6^a$ | ND | $16–91^a$ | 77–246 | Inconclusive; irrigation water suspected | Egwu and Agbenin (2013) |
| Tacoma, Washington, USA; urban gardens | $0.30–0.80^a$ | $32–118^a$ | $38–110^a$ | $100–219^a$ | Uncertain; past management effects | McIvor et al. (2012) |
| Melbourne; urban garden soil | 0.12–1.04 | ND | 13–773 | ND | House age; distance to roads | Kandic et al. (2019) |
| Lisbon, Portugal; urban allotment gardens | bdl | 5–87 | 23–245 | 35–208 | Industry; traffic phenomena | Bechet et al. (2018) |
| Hue City, Vietnam; peri-urban farms | 0.03–0.44 | 3–39 | 4–25 | 12–212 | Irrigation water; flood disposition | Pham et al. (2021) |
| Ottawa, CA; garden soils | 0.11–0.75 | 6–42 | 16–547 | 50–380 | Age; limited industrial activity | Rasmussen et al. (2001) |
| Hangzhou; urban–rural vegetable soils | 0.10–0.64 | 14–78 | 20–87 | 59–265 | Agrichemicals; vehicles exhaust; industry | Chen et al. (2008) |
| Xianyang (China); suburban garden soil | 0.2–2.45 | 6–50 | 24–132 | 40–526 | Plastic, fertilizers, traffic, industry | Wang et al. (2018) |

**Table 1** (continued)

| Location; garden type | Cd | Cu | Pb | Zn | Issues mentioned | References |
|---|---|---|---|---|---|---|
| Hong Kong area; market garden soils | 0.25–2.17 | 5–67 | 24–113 | 20–232 | Urbanization and traffic volume | Wong (1996) |
| Plock city, Poland; allotment gardens | 0.41 [2.12]+ | 9 [38]+ | 9 [24]+ | 313 [2947]+ | Dust fall (Industry, transportation) | Mikula and Indeka (1997) |
| Perth, Australia; former market gardens | 0.02–0.66 | 2–68 | 3–174 | 6–304 | Construction; roads; waste disposal | Rate (2021) |
| Wuxi, China; peri-urban ag soils | bdl–0.14 | 27–152 | 17–162 | 30–445 | Trafficking, industry, manufacturing | Zhao et al. (2007) |
| Connecticut; community gardens | bdl | 20–75[a] | 22–450[a] | 60–238[a] | Not identified specifically | Stilwell et al. (2008a) |
| Wollongong, Australia; home gardens | 1.99 ± 0.3 | 847 ± 114 | 13.3 ± 3.1 | 229 ± 17 | Proximity to Cu smelter | Beavington (1975) |
| Taiwan; urban park soils++ | bdl–3.33 | 9–117 | 8–57 | 49–379 | Industry sources | Jien et al. (2011) |
| Guangdong, China; vegetable gardens | 0.26–1.17 | 210–450 | 73–134 | 92–142 | E-waste processing | Luo et al. (2011) |
| Kaduna, Nigeria; urban field gardens | ND | 18–85 | 160–200 | 156–433 | Wastewater irrigation; waste disposal | Akpa and Agbenin (2012) |
| Naples, Italy; urban gardens | ND | ~20–85 | ~75–325 | ~50–450 | Traffic, Industry, ag amendments | Imperato et al. (2003) |
| Vienna, Austria; buried pots in gardens | ~0.10–0.75 | ND | ~5–165 | 74–2600 | Great variability reported | Ziss et al. (2021) |
| Southampton (UK); urban horticulture soils | 0.14–0.80 | 19–86 | 34–488 | 60–485 | Geology; mixed (indust, petrol, burning) | Crispo et al. (2021) |
| Adelaide, Australia; urban ag sites | 0.01–0.38 | 25–183 | 30–268 | < 1–662 | Legacy (industry) | Salomon et al. (2020) |
| Vancouver, CA; urban gardens/farms | bdl | 35–127[a] | 69–208[a] | 77–887[a] | Not explored | Thomas (2013) |
| Nantes, France; urban allotment gardens | 0.1–17.7 | 13–117 | 19–229 | 26–272 | Industry; traffic phenomena | Bechet et al. (2018) |
| Bottrop, Germany; variety of urban gardens | ~1.8–2.7 | ~15–145 | ~50–155 | ~125–295 | Inconclusive | Burghardt and Schneider (2018) |
| Zagreb, Croatia; urban/suburban ag | 0.25–3.85 | 4–183 | 1–139 | 15–277 | Multiple anthropogenic processes | Romic and Romic (2003) |
| Harrisburg, PA (USA); potential gardens | ND | 11–46[a] | 26–402[a] | 30–216[a] | Legacy Pb; road phenomena | Kessler et al. (2013) |
| Madrid, Spain; community gardens | ND | 11–59 | 15–598 | 72–309 | Landuse history; proximity to urban center | Izquierdo et al. (2015) |
| Île-de-France, France; market gardens | 0.55–0.99 | 81–156 | 229–436 | 231–367 | Legacy (industry, sludge disposal) | Barbillon et al. (2019) |
| Cotorro, Havana; peri-urban gardens | ND | 50–945 | 21–731 | 91–7739 | Wind direction (smelter) | Díaz Rizo et al. (2015) |
| Torino, Italy; gardens and parks | ND | 40–293 | 17–565 | 75–550 | Diffuse pollution | Hursthouse et al. (2004) |
| Baton Rouge, LA, USA; peri-urban | ND | 6–189 | 11–967 | 32–1031 | Industry, construction, traffic | Weindorf et al. (2012) |
| Sheffield, UK; gardens/allotments | ND | ~45–180 | ~70–600 | ~185–750 | Varies by land use; e legacy metals | Weber et al. (2019) |
| Gyongyosoroszim, Hungary; veg gardens | 0.22–13.60 | ND | 21–694 | 97–2050 | Flood water from old Zn/Pb mine | Sipter et al. (2008) |
| Hangzhou, China; suburb veg gardens | ND | 4–144 | 14–98 | 76–302 | Organic manuring | Wuzhong et al. (2004) |
| Porto, Portugal; urban ag/farm soils | 0.1–1.1 | 10–248 | 27–415 | 92–465 | Airport; oil refinery; roads | Cruz et al. (2014) |
| Madrid; parks and gardens; other | ND | 18–194 | 31–463 | 82–514 | Waste sludge; fertilizers; roads; other | De Miguel et al. (1998) |
| Nates, France; community garden | ND | 16–151[a] | 28–5785[a] | 32–193[a] | Anthropogenic and geogenic | Jean-Soro et al. (2015) |

**Table 1** (continued)

| Location; garden type | Cd | Cu | Pb | Zn | Issues mentioned | References |
|---|---|---|---|---|---|---|
| Nova Scotia; community/yard gardens | ND | 11–108 | 10–767 | 42–350 | Various potential anthropogenic sources | Heidary-Monfared (2011) |
| Washington DC; urban farm/garden | ND | 13–159 | 16–869 | 32–607 | Legacy and urban/trafficking sources | Long (2010) |
| Liverpool (UK); urban horticulture soils | 0.19–2.2 | 27–166 | 76–514 | 94–497 | Geology; mixed (industry, petrol, burning) | Crispo et al. (2021) |
| Leicester (UK); urban horticulture soils | 0.25–4.8 | 27–110 | 76–454 | 127–614 | Geology; mixed (industry, petrol, burning) | Crispo et al. (2021) |
| Columbus and Cleveland; established gardens | 0.93–4.15 | 23–101 | 31–553 | 97–645 | Legacy (Pb paint); prior activities | Kaiser et al. (2015) |
| Ghent, Belgium; urban gardens | 0.16–4.91 | 11–107 | 55–385 | 39–1325 | Road phenomena; general industry signal | Folens et al. (2017) |
| Hong Kong; urban parks | 0.02–5.89 | 5–190 | 5–404 | 39–435 | Age; amendments (fertilizers), road dust | Li et al. (2001) |
| Sevilla, Spain; parks/ag/gardens | 0.18–4.85 | 14–198 | 23–725 | 39–326 | Trafficking; organic amendments | Ruiz-Cortes et al. (2005) |
| Newcastle (UK); urban horticulture soils | 0.43–1.1 | 25–246 | 82–501 | 185–629 | Geology; mixed (industry, petrol, burning) | Crispo et al. (2021) |
| Montreal, CA; urban gardens | 0.4–1.8[a] | 15–400[a] | 150–400[a] | 210–300[a] | Compost; highly variable | Murray et al. (2011) |
| Buffalo & NYC, NY; urban farms/gardens | BDL–3.6 | 40–83 | 18–3620 | 136–1060 | Deposition; non-uniform contamination | McBride et al. (2014), Cai et al. (2016) |
| Wales & England; urban soils only | 0.4–1.9 | 4–75 | 14–680 | 6–744 | Trafficking, industry, mining, other | Davies (1978) |
| Dublin area; garden soils | > 3–5 | 11–145 | 39–540 | 117–940 | Wind direction from urban sources | Fleming and Parle (1977) |
| Sevilla, Spain; allotments and parks | ND | 14–698 | 14–1080 | 38–611 | Vehicular sources | Hursthouse et al. (2004) |
| Leeds (UK); urban horticulture soils | 0.24–2.5 | 33–372 | 104–1682 | 139–503 | Geology; mixed (industry, petrol, burning) | Crispo et al. (2021) |
| Milton Keynes (UK); urban horticulture soils | 0.21–6.5 | 19–228 | 29–3943 | 89–413 | Geology; mixed (industry, petrol, burning) | Crispo et al. (2021) |
| Bangkok, Thailand; general urban soils++ | 0.05–2.53 | 5–283 | 12–269 | 3–814 | Soil type; urban processes; roads | Wilcke et al. (1998) |
| Breschia, Italy; home gardens | bdl–4.00 | 28–410 | 36–207 | 58–845 | Proximity to industry | Ferri et al. (2015) |
| Connecticut; comm and res gardens | bdl–3.5 | bdl–900 | bdl–3490 | 15–520 | Not identified specifically | Stilwell et al. (2008b) |
| Los Angeles; urban gardens | 0.11–4.27 | ND | 18–1720 | ND | Proximity to roads; age of neighborhood | Clarke et al. (2015) |
| Edinburgh (UK); urban horticulture soils | 0.30–2.4 | 33–103 | 84–1229 | 138–996 | Geology; mixed (industry, petrol, burning) | Crispo et al. (2021) |
| Nottingham (UK); urban horticulture soils | 0.54–4.6 | 42–127 | 124–1019 | 235–862 | Geology; mixed (industry, petrol, burning) | Crispo et al. (2021) |
| Quezon City, Philippines; cropped soils | ND | 84–784 | 300–6550 | bdl–1817 | Urban processes; landfill legacy | Navarrete et al. (2017) |
| Melbourne (AU); residential veg gardens | ND | bdl–323 | bdl–3341 | bdl–3003 | Age (e.g., for lead); road/rail proximity | Laidlaw et al. (2018) |
| Mexico City; garden, park & other soils | ND | 15–398 | 5–452 | 36–1641 | Trafficking/roads; Industrial sources | Morton-Bermea et al. (2009) |
| Sydney (AU); urban veg gardens | ND | 33–717 | 14–3080 | 50–2020 | Inner city has high lead legacy | Rouillon et al. (2017) |
| Wroclaw, Poland; allotment gardens | ND | 13–595 | 13–659 | 38–2103 | Proximity to pollution sources | Kabala et al. (2009) |
| Chicago; various urban gardens/farms | ND | 9–19[a] | 34–449 | 38–69[a] | Not substantiated | Witzling et al. (2010) |
| Christchurch; urban gardens | 0.33–10.7 | 12–118 | 23–2615 | 68–799 | Age (pre-1950 neighborhoods) | Ashrafzadeh et al. (2018) |

**Table 1** (continued)

| Location; garden type | Cd | Cu | Pb | Zn | Issues mentioned | References |
|---|---|---|---|---|---|---|
| Cardiff (UK); urban horticulture soils | 0.15–4.3 | 10–216 | 30–2149 | 46–1213 | Geology; mixed (industry, petrol, burning) | Crispo et al. (2021) |
| Bristol (UK); urban horticulture soils | 0.46–3.3 | 14–752 | 88–1211 | 115–960 | Geology; mixed (industry, petrol, burning) | Crispo et al. (2021) |
| Copenhagen, Denmark; urban gardens | ~0.3–3.7[a] | ~10–450[a] | ~10–600[a] | ~40–1260[a] | Inconclusive; urban signal | Warming et al. (2015) |
| Gijón (Asturias, Spain); peri-urban soils | 0.4–2.3 | 6–244 | 33–585 | 64–1047 | Coal combustion; unidentified sources | Boente et al. (2017) |
| Marseille, France; veg gardens | 0.16–8.86 | 14–473 | 20–566 | 43–1020 | Chemical fertilizers, fungicides | Zhong et al. (2022) |
| New York (USA); community gardens | bdl–3.1 | 6–598 | 11–2455 | 21–2317 | Legacy Pb; long-term urban processes | Mitchell et al. (2014) |
| Baltimore, MA, (USA); urban gardens | 0.02–13.65 | 1–97 | 1–10,900 | < 1–4880 | Legacy (inner city); urban activities | Mielke et al. (1983) |
| Paris area (France); Ag + urban gardens | 0.90–15.0 | ND | 42–854 | 100–1292 | Deposition legacy (mining, smelting) | Pelfrêne et al. (2012) |
| Baltimore (USA); urban farms/gardens | ND | ~20–300 | ~7–1000 | ~50–600 | Not determined; produce was safe | Lupolt et al. (2021) |
| Brooklyn, NY; urban gardens | ND | 53–328[a] | 66–1025[a] | 150–1474[a] | Legacy urban pollutants | Egendorf et al. (2018) |
| San Francisco (USA); garden soils | bdl–8.14 | 2–2630 | 7–2040 | 33–707 | Home age | Gorospe (2012) |
| Washington DC; urban gardens | 0.2–6.6 | 8–240 | 44–5300 | 74–2600 | Proximity to homes (esp. Pb); variable | Preer et al. (1984) |
| Noyelles-Godault (France); kitchen gardens | 2.8–21.1 | ND | 190–1425 | 199–2429 | Former mining/smelter area | Douay et al. (2013) |
| Paris area (France); kitchen garden soils | 17.2–23.9[a] | ND | 387–1708[a] | 2350–2871[a] | Garden amendments; ag practices | Waterlot et al. (2011) |
| Kampala City, Uganda; peri-urban ag | 0.12–21.0 | 26–1690 | 10–653 | 49–3270 | Inappropriate waste disposal | Nabulo et al. (2012) |
| Boolaroo (Australia); residential veg garden | 0.01–27.49 | 3–223 | 12–1626 | 45–4014 | Former Pb–Zn smelter area | Kachenko and Singh (2006) |
| Port Kemba (Australia); residential veg garden | 0.01–7.01 | 65–1032 | 14–430 | 87–1947 | Former Cu smelter area | Kachenko and Singh (2006) |
| Palmerton, PA (USA); garden soils | 18–113 | 41–595 | 90–370[a] | 1420–13,100 | Proximity to Zn smelter | Chaney et al. (1988) |
| Glasgow, Scotland; allotments and parks | ND | 52–484 | 250–7051 | 164–1004 | Diffuse pollution | Hursthouse et al. (2004) |
| Hong Kong; roadside gardens/parks | ND | 13–553 | 42–1034 | 173–11,316 | Road pollution | Tam et al. (1987) |
| New York (USA); urban gardens | 0.1–11.0 | 5–1286 | 3–8912 | 35–2352 | Legacy (lead; incineration); pesticides | Cheng et al. (2015) |
| New Orleans, LA (USA); urban ag | 0.25–8.80 | 2–200 | 1–9540 | 18–7330 | Legacy (Pb); current road/industry | Moller et al. (2018) |
| Newcastle, Australia; private spaces | 3.0–9.0 | 44–1040 | 42–11,600 | 140–8570 | Legacy Cu; smelting (slag waste) | Harvey et al. (2017) |
| Baia Mare, Romania, (peri) urban gardens | 1.2–13.8 | 171–2130 | 797–10,200 | 376–2130 | Legacy and deposition (mining, smelter) | Mihali et al. (2012) |
| Dalnegorsk, Russia; garden/other soils | 7–26 | 29–805 | 213–5621 | 593–5438 | Mining, smelter, industry | Von Braun et al. (2002) |
| London Boroughs (UK); garden-veg plot soils | bdl–68 | 8–2320 | 28–13,700 | 34–2780 | Home age | Culbard et al. (1988) |

[a] Pertains to a range of means or a mean ± standard deviation; otherwise, entries are ranges (min–max)

~Values are approximated from a graph

+Signals that values are means and maximums

++Not an agriculture study, but in close proximity to Singapore

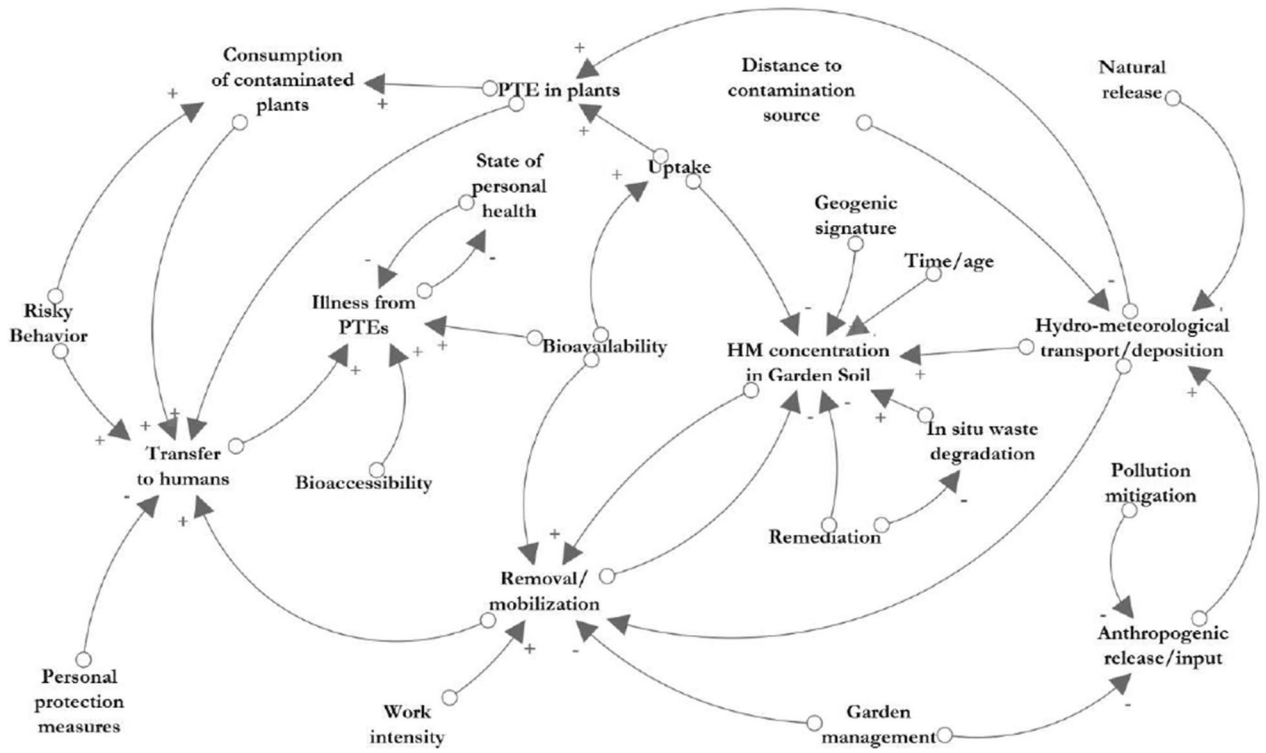


**Fig. 1** Causal loop diagram representation of the human urban exposure chain for heavy metal risk associated with urban gardens. Within this conceptual system model, arrows indicate the negative (−) or positive (+) effect on a particular variable that results from an increase in the extent/impact/amount of another variable (open circle) representing processes, actions, or phenomena

enhancements caused by *natural processes/sources* such as volcanic eruptions, groundwater seepage, and fluvial transport (Briffa et al. 2020; Müller et al. 2020). Often, however, enrichment occurs when contaminants are *transported* to and *deposited* at a site from *anthropogenic sources* by atmospheric (related to wind and dry deposition) and hydrologic (e.g., rainfall, surface runoff, waste water irrigation) dispersal pathways, as well as the degradation of discarded waste or applied chemicals, including compost, fertilizers and disease/pest control agents that are applied in *garden management* (Kelly et al. 1996; Holt 2000; Wuana and Okiemen 2011; Su et al. 2022).

While contamination may be acute, being associated with only a few pollution events (e.g., accidental spill/release; flooding), it is often chronic, stemming from an accumulation over longer periods time (He et al. 2013; Ashraf et al. 2014). Thus, *time* or *age* are critical factors. For example, the soils associated with old homes/neighborhoods often have elevated concentrations of some elements (inner city legacy), for example Pb from paint used to protect wooden and metal construction materials (Culbard et al. 1988; Li et al. 2001; Mitchell et al. 2014; Clarke et al. 2015; Rouillon et al. 2017; Kandic et al. 2019; Montaño-López and Biswas 2021). Gardens built on the former sites of long-running smelters, mines, factories, and waste dumps often are enriched in selected elements, representing a deposition legacy long after disuse (Kachenko and Singh 2006; Nabulo et al. 2012; Pelfrêne et al. 2012; Douay et al. 2013). Garden soils may become enriched over time from chronic atmospheric deposition (Kargar et al. 2013; Laidlaw et al. 2018). Thus, a new fill soil may not reflect the long-term pattern of contamination. Long-term application of metal-based amendments can also elevate PTEs in garden soils.

Gardens that are close in *distance* to mines, power generation plants, incineration facilities, factories, and manufacturing centers tend to receive higher pollution loads, but long-range dispersal is possible (Chaney et al. 1988; Marx and McGowan 2010; Hall et al. 2021). Gardens located in low-income neighborhoods are often in close *distance* to polluting activities (Mielke et al. 1983; Kaiser et al. 2015; Oguntade et al. 2020; Montaño-López and Biswas 2021). Roads and various phenomena related to trafficking (density/usage; road, vehicle and tire wear; emissions, etc.) are documented contributors to the PTEs found in roadside gardens (Tam et al. 1987; Hursthouse et al. 2004; Morton-Bermea et al. 2009; Folens et al. 2017). Airports, trains, and shipping are also potential

sources of PTEs in urban gardens (Vazhacharickal et al. 2013; Cruz et al. 2014; Laidlaw et al. 2018; Leitão et al. 2018). As shown in the Table 1, other common sources of elevated PTE presence in garden soils are the application of contaminated sludge or irrigation water (Jamali et al. 2007; Abdu et al. 2011; Xue et al. 2012; Vazhacharickal et al. 2013) and element-based garden agrochemicals (Waterlot et al. 2011; Tran et al. 2020; Zhong et al. 2022). Other types of interesting sources of PTEs reported in Table 1 include plastics (Wang et al. 2018), E-waste processes (Luo et al. 2011), and war activities (Safi and Buerkert 2011).

Children are generally considered of greatest risk from exposure to particular PTEs because of two issues: (a) their *state of health*, which is generally characterized as having an undeveloped immune system; and (b) *their risky behavior*, which involves hand-to-mouth and object-to-mouth actions (Zeng et al. 2016; Różański et al. 2021). Elderly gardeners may also be at risk if their *state of health* has been compromised or they have experienced long-term exposure to particular elements (Sun and Gu 2008; Alghamdi et al. 2022). Nevertheless, the risk can apply to anyone working at contaminated sites who experiences prolonged dermal exposure (increased by lack of personal protection equipment), *ingests* high concentrations of elements through inhalation of dust during *intense work* or through hand-to-mouth transfer, or by *consuming contaminated plants* (Zahran et al. 2013; Antisari et al. 2015; Hough 2007; Tong et al. 2020).

Important is that the degree of toxicity experienced from exposure to PTEs in the contaminated media is not simply related to the total concentration of the element involved, but importantly, the *bioavailability* and *bioaccessability* of the element (Chapman et al. 2003; Janssen et al. 2003). Bioavailability refers to the fraction of PTEs that is released from a mineral matrix into a liquid form that can be taken up by plants or released into the gastrointestinal tract upon digestion (Chen et al. 2020; Petruzzelli et al. 2020); and bioaccessability is the proportion that is available to induce a potentially toxic effect within the body. Bioavailability is determined by the characteristics of the element, soil, and biological organisms (Smith and Huyck 1999; Kim et al. 2015a, b; Bidar et al. 2020). Important soil properties include pH, organic matter, texture, cation exchange capacity, redox potential, oxide/hydroxide presence, and time (Petruzzelli et al. 2020). In performing risk assessments of element *toxicity*, bioavailability and bioaccessibility must be determined to avoid over-estimating the safety by using psuedo-total element concentrations alone (Izquierdo et al. 2015).

Psuedo-total concentrations are however useful for initial risk assessments that compare concentrations with national guidelines—although these guidelines vary greatly worldwide (Additional file 1: Table S2). If a garden soil, and/or the produce grown in it, contain elevated levels of a particular element, *remediation* actions may reduce concentrations to safe levels, for example by replacing the soil or constructing raised beds with safe media (Clark et al. 2008; Rai et al. 2019; Paltseva et al. 2020). Further, risk can be reduced through *protection measures* that reduce the longevity/intensity of exposure, limits extensive disruption of the soil, and/or restrict access to those at high risk (Briffa et al. 2020; Lupolt et al. 2021). Once concern advances to the stage where remediation is considered, for example if the concentration exceeds an intervention threshold, bioavailability and bioaccessibility studies would be highly appropriate to assess risk in greater detail. However, a viable risk may be present to some individuals in cases of repeated exposure (for example consumption of particular crops) for concentrations lower than established thresholds.

The indicated interactions in the simplified model shown in Fig. 1 affect the final outcome of the complex system, which is toxicity or illness associated with exposure to particular PTEs in gardens. Again, potential illness largely stems from ingestion/inhalation/contact with high levels of various elements in the soil and the *consumption* of plants with high levels of PTEs, but in consideration with their bioavailability/bioaccessibility (Rai et al. 2019; Briffa et al. 2020). One relevant feedback in the system is associated with the state of *health*, in the sense that a variety of types of element toxicity reduces one's health, yet various aspects of an individual's health may provide some degree of resilience (Monachese et al. 2012; Mitra et al. 2022). Nevertheless, most avenues for reducing the negative health effect are *behavior* risk reduction, site *remediation*, and appropriate garden *management* and *mitigation* actions (Mitchell et al. 2014).

Fundamentally, understanding the underlying *soil and elemental properties* is an important process in governing many of the processes of the human exposure chain depicted in Fig. 1 (Selim and Amacher 1996; Violante et al. 2010; Li et al. 2022). Thus, the baseline information required in investigating this issue is knowledge of the PTE concentrations in garden soils in context with the behavior of the at-risk group—this type of assessment aligns with the scope of our work and preliminary investigation (Goh 2018). Even in areas where health risks have not been reported, new and site-specific knowledge is needed to make informed decisions regarding potential risks to exposure (Chaney et al. 1984; Kim et al. 2014; Cooper et al. 2020). Once obtained, this information helps guide additional follow-on work to fully assess

health risks to PTE exposure in contaminated urban garden soils (cf. EPA 2007).

### Objectives

In this study we were concerned with assessing the presence of PTEs in the soils of community gardens in highly developed Singapore where urban gardening in housing estates is being promoted to encourage a growing, aging community to be active and self-sufficient (Tan and Neo 2009). Community gardens are also anticipated to supplement and diversify local food sources in the city (Chandran 2019). To that end, the nation aims to double the number of community gardens and allotments from 1500 to 3000 plots by 2030 (Begum 2020). The growing prevalence of gardens within public housing estates in Singapore that are visited frequently by onsite residents potentially represent an uncertain health risk because the gardens are inherently located in a dense matrix of industrial and residential land-uses with high road densities (Ng et al. 2006; Joshi and Balasubramanian 2010; Yuen et al. 2012). Further, other green landscapes such as public parks and accessible woodlands are also frequently visited locations of potential exposure to PTEs (Han et al. 2017).

We focus on the presence of four PTEs (Cd, Cu, Pb, Zn) that are often associated with contamination in urban garden soils worldwide (Xia et al. 2011; Kaiser et al. 2015; Table 1). These elements are also among the most studied in past assessments of urban contamination set in Singapore (discussed below). Lead has a legacy presence in Singapore soils despite most leaded petrol being phased out in Southeast Asia in the 1990s (Lee et al. 2014), as well as the regulation of Pb in other important sources such as paint (Kessler 2014). Our focus is on gardens constructed on soils; soil-less, raised bed, rooftop, and hydroponic systems are not considered. We also examine key soil properties (pH, Total Organic Carbon (TOC), texture) that may influence the mobility of elements in the soil (Zwolak et al. 2019). Absent in our assessment with respect to the exposure model (Fig. 1, "Human urban gardening exposure chain" section) are the determination of bioavailability, bioaccessibility, and PTE presence in vegetables. In foreshadowing our results, we did not perform these advanced experiments because pseudo-total element concentrations in soil were not alarmingly high. Finally, we use Pb to track the potential anthropogenic pollution sources of element contaminants in the Singapore urban soils (Lee et al. 2021).

## Study area

### Singapore geography

Singapore has a tropical equatorial climate (mean annual temperature = 27 °C; total annual rainfall = 2300 mm) with two monsoon seasons and short dry periods. The geology of Singapore is somewhat diverse, with major formations including igneous rocks such as granite and norite, various sedimentary rocks (sandstone, mudstone), isolated metamorphic rocks, Quaternary sediments, as well as marine clay deposits (Sharma et al. 1999). Soils are generally acidic (pH 3.7–6.2), have low cation exchange capacities (maximum mean value: 100 mmolc/kg), and have low P concentrations (mean values $\leq$ 0.28 g/kg) irrespective of geology (Leitgeb et al. 2019). Soils developed on sedimentary rocks and sediments tend to have higher concentrations of soil organic carbon, total P, base forming elements, exchangeable base cations and moderate levels of some PTEs, for example Cr, Cu, Ni, Pb, and Zn (Leitgeb et al. 2019). Nguyen et al. (2019, 2020) determined that relatively high concentrations of PTEs may be associated with particular types of geological material (e.g., norite) and native soils.

Singapore's land-use is characterized by a dense matrix of urban areas with some lands dedicated to green spaces (Fig. 2). These areas differ somewhat in the degree of human disturbance. Woodland areas are made of (replanted) secondary forests located in either nature reserves or urban parks (Gaw and Richards 2021; Ramchunder and Ziegler 2021). Some have been disturbed by old village settlements, military activity, road building, and other activities requiring forest clearance followed by subsequent reforestation. However, these woodlands are not currently being disturbed due to strict land-use regulation. Open parks, in contrast, are grassland fields with few trees, but may be surrounded by substantial forested areas in some locations. Although they are considered open green spaces, soils tend to be compacted, both from construction and subsequent recreational activities where trampling occurs (Henderson 2013).

While some of the woodland soils may be in situ, the surfaces on all parks and urban gardens contain substantial fill material that was likely either imported from several quarries that were once operational in Singapore, or from other reclamation and sand mining projects (cf. Kog 2015). Thus, soils in gardens likely do not correspond with the underlying geology. Gardens are often built on marginal pockets of land sandwiched between housing blocks. Most of the construction of modern Singapore, including the spaces we investigated, took place in the last 50 years (Guo 2016). One garden we studied was built by top-filling over an old asphalt parking lot; another was built over a concrete basketball

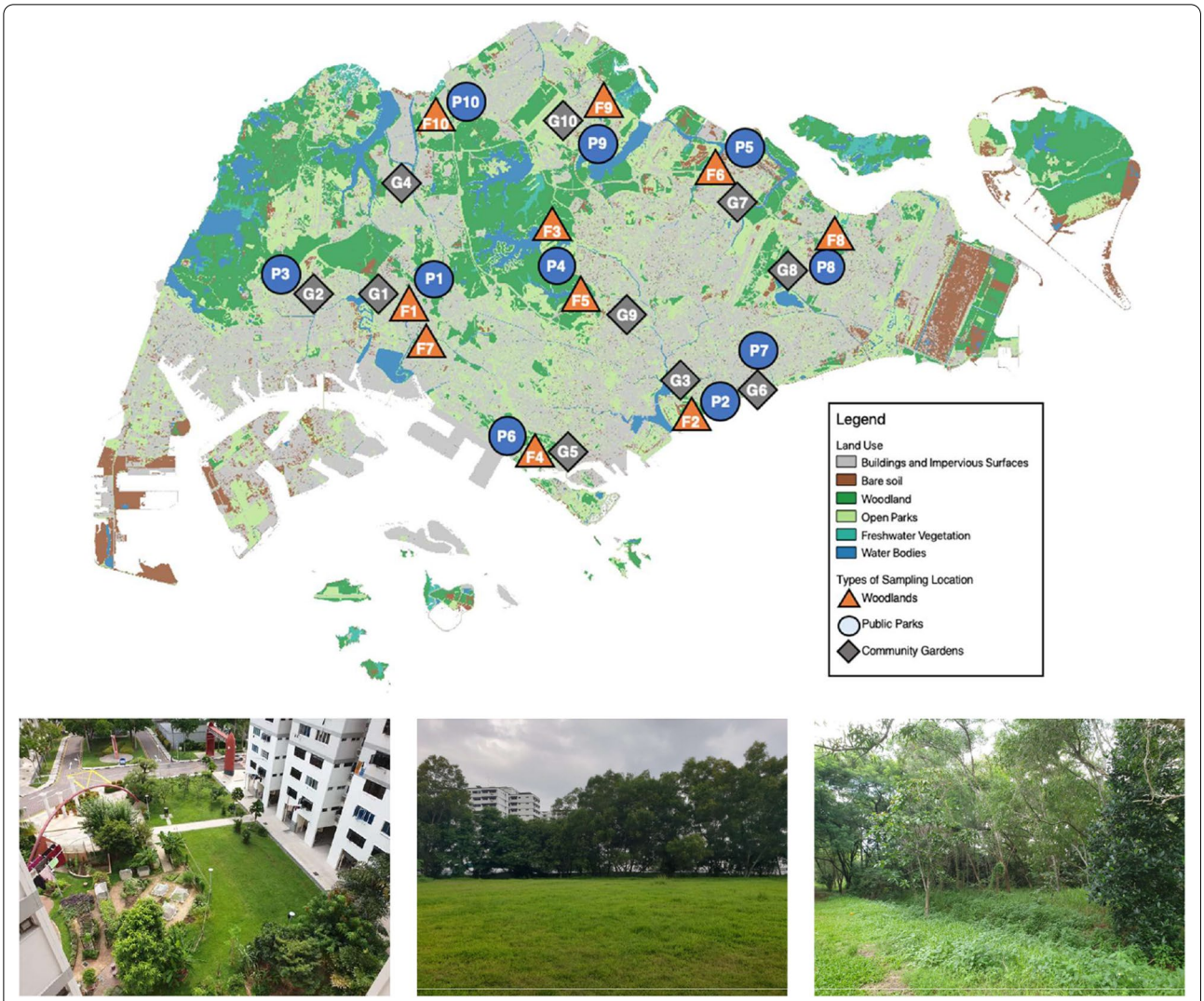


**Fig. 2** The map shows major land-cover/land-use types in Singapore, with sample locations shown as diamonds (community gardens), circles (public parks) and triangles (woodlands). (Adapted from a base map by Gaw et al. 2019). Below the maps are representative images for gardens (left), parks (center), and wooded areas (right)

court. These fill soils tend to be deficient in important plant nutrients, which necessitates the addition of fertilizers or construction of raised beds using garden soils purchased from local manufacturers. Store-bought soils and inorganic fertilizers may originate from Singapore or be imported from overseas. Gardeners also apply mulches and other organic materials (eggshells, coffee grounds, compost) to enrich soil nutrients. These additions may modify the amount of PTEs present in the garden. Garden soil disturbances are mainly related to reworking of the soil with hand tools.

## Singapore urban pollution

Several studies have examined selected PTEs in a variety of Singaporean soils/sediments, including forests (Nguyen et al. 2018, 2020; Leitgeb et al. 2019; Wang et al. 2022), landfills (Patra et al. 2017), mangroves (Cuong et al. 2005), water bodies (Sin et al. 1991; Chen et al. 1996, 2016; Nguyen et al. 2018), marine sediments (Rahman et al. 1979; Goh and Chou 1997; Wood et al. 1997; Cuong et al. 2008), military lands (Nguyen et al. 2019), road dust (Joshi and Balasubramanian 2010; Joshi et al. 2009; Yuen et al. 2012); fly ash (Tan et al. 1997; Wu and Ting 2006); and various types of urban lands (Zhou et al. 1997; Ng et al. 2006). The studies of Chen (1999), Zhou et al. (1997), and Wang et al. (2022)

**Table 2** Pseudo-total concentrations of Cd, Cu, Pb, and Zn found in a variety of soils (if not specified) and other media (e.g., sediment, ash) in Singapore

| Ref. | Site | Cd | Cu | Pb | Zn |
|---|---|---|---|---|---|
| d | Forest; Nee Soon, upper Catchment [medians] | bdl | 5 ± 5 | 13 ± 6 | 23 ± 6 |
| d | Forest; Nee Soon, lower Catchment [medians] | bdl | 12 ± 4 | 18 ± 7 | 33 ± 11 |
| b | Golf course (3 locations) | NA | 12–14 | 20–47 | 52–73 |
| d | Disturbed lands, Nee Soon Catchment [medians] | bdl | 16 ± 6 | 24 ± 9 | 43 ± 15 |
| e | Forest soils (variable geology) | bdl–0.03 | 7–20 | 9–17 | 13–61 |
| f | Park soils (4 locations)[a] | 0.05–0.08 | 5–11 | 14–21 | 11–24 |
| p | Park soils (3 locations)[a] | 0.46–1.66 | 6–14 | 14–37 | 45–71 |
| f | Airport/industrial area soils (2 locations)[a] | 0.06–0.08 | 7–16 | 20–26 | 13–31 |
| i | Reservoir susp. sediment (Kranji & MacRitchie) | NA | 15–16 | 20–43 | 49–73 |
| g | Residential soils (7 sites island-wide) | 0.45–1.68 | 3–90 | 6–70 | 9–93 |
| g | Reservoir sediments (3 sites) | 0.79–1.73 | 8–61 | 15–65 | 1–68 |
| d | Military lands, Nee Soon Catchment [medians] | bdl | 44 ± 34 | 56 ± 37 | 77 ± 49 |
| b | Forest; Nature Reserve (MacRitchie) | NA | 6–19 | 9–38 | 29–123 |
| b | Residential area (7 sites in Bishan) | NA | 13–57 | 16–46 | 70–151 |
| i | Reservoir sediments (MacRitchie) | NA | 13–21 | 19–64 | 75–133 |
| q | Mangrove sediment (S. Buloh & S. Khatib Bonsu) | 0.18–0.27 | 7–32 | 12–32 | 51–120 |
| g | Field soils (7 sites island-wide) | 0.45–1.42 | 1–105 | 5–89 | 1–150 |
| a | Parks (Island wide; 10 locations) | bdl | 9–70 | bdl–61 | 33–190 |
| h | MacRitchie Reservoir sediments (9-m core) | ~ bdl–0.40 | ~ 1–370 | ~ 1–55 | ~ 1–135 |
| a | Gardens (10 locations island-wide) | bdl–3.2 | 10–76 | bdl–37 | 71–164 |
| a | Woodlands (10 locations island-wide) | bdl–6.4 | 11–50 | bdl–33 | 19–146 |
| c | Forest; Bukit Gombak [medians] | 5.76 ± 5.16 | 36 ± 24 | 14 ± 6 | 100 ± 27 |
| o | Johore Strait marine sediments | 0.11–0.36 | 11–93 | 27–70 | 69–231 |
| i | Reservoir sediment (Kranji) | NA | 21–29 | 35–82 | 128–367 |
| r | Estuary sediment (Punggol) | bdl–1.37 | 4–439 | 1–157 | NA |
| s | Coastal sediments (island wide) | bdl–1.6 | 1–1781 | 1–82 | 95–281 |
| t | River sediment (Singapore river) | ~ 0.1–1.0 | ~ 15–95 | ~ 40–240 | ~ 100–600 |
| b | Industrial (Jurong West; 7 sites) | NA | 12–194 | 29–167 | 111–837 |
| b | Road side soils (heavy traffic area) | NA | 30–68 | 27–1981 | 124–938 |
| k | Road sediment; residential [means] | 0.71 ± 0.30 | 247 ± 54 | 69 ± 26 | 275 ± 55 |
| k | Road sediment; commercial [means] | 0.30 ± 0.10 | 98 ± 76 | 111 ± 15 | 620 ± 186 |
| p | Industrial zone soils (8 locations)[a] | 0.66–1.63 | 16–258 | 10–242 | 40–708 |
| l | Suspended sediment; residential road runoff | 0.89–3.03 | 11–74 | 5–206 | 16–941 |
| l | Suspended sediment; industrial road runoff | 2.13–8.59 | 30–671 | 25–182 | 112–2006 |
| g | Industrial soils (3 sites) | 1.11–3.21 | 1–485 | 51–235 | 15–3594 |
| j | Residential road sediment | NA | 203–1294 | 135–395 | 704–2105 |
| k | Road sediment; industrial [means] | 2.41 ± 1.0 | 9069 ± 3742 | 338 ± 56 | 1696 ± 446 |
| j | Road sediment (industrial areas) | NA | 251–4271 | 144–744 | 836–11,960 |
| n | Fly ash from municipal incinerator | 95 | 570 | 2000 | 6288 |
| m | Fly ash, Tuas solid waste incinerator [means] | 130 | 987 | 2120 | 8072 |

Listed informally in order of increasing level of enrichment

Most values are ranges, means ± standard deviation, or medians ± MAD; [a]indicates a range of means/medians; ~ signifies the values were estimated from a graph. ND is not determined; bdl is below detection limit

References:

(a) This study; see "Methods and materials" section for details on the analysis methods

(b) Zhou et al. (1997); HCL–$HNO_3$–HF digestion and a flame atomic absorption spectrophotometer for high concentrations and a graphite furnace atomic absorption spectrophotometer for low concentrations

(c) Nguyen et al. (2020); microwave-assisted digestion methods EPA 3051a (USEPA 2017) followed by ICP-OES (Perkin Elmer 8300)

(d) Nguyen et al. (2018); four-acid digestion protocol (MA300 method by Bureau Veritas Labs (Vancouver, Canada) and ICP-MS (Perkin Elmer ELAN 9000)

**Table 2** (continued)

(e) Leitgeb et al. (2019); pseudo-total element concentrations were determined by digesting soil samples with aqua regia using a microwave oven (MARS 6, CEM, Kamp-Lintfort, Germany) and by analysing the solution using ICP-OES (Optima 8300, Perkin Elmer, Waltham, MA)

(f) Ng et al. (2006), samples digested using microwave oven (Milestone, Ethos D, Monroe, C.T., USA) with a mixture of 9 ml $HNO_3$ and 3 ml of HF in a closed vent medium pressure vessel with a ramp to 180 °C at 600 W for 10 min and held at 180 °C, 600 W for 10 min

(g) Chen (1999), mixed acid method (1 ml HF (49%) added to 14 ml 3:1 HCL (36%): $HNO_3$ (68%) added by 20-min microwave digestion; analysis on ICP-AES

(h) Chen et al. (2016); leached with ultrapure grade 1.75 mol/l $HNO_3$–3 mol/L HCl in an ultrasonic bath, followed by analysis using a quadrupole inductively-coupled plasma mass spectrometry (Q-ICP-MS, VG PlasmaQuad 2+)

(i) Chen et al. (1970); $HNO_3$ digestion in a microwave followed by analysis with ICP Perkin-Elmer Plasma 400 Emission Spectrometer

(j) Yuen et al. (2012); digestion with strong acids $HNO_3$–$H_2O_2$–HF; analysis on a Thermo Scientific X Series2 Quadrupole ICP-MS

(k)Joshi et al. (2009); digestion with a combination of strong acids ($HNO_3$–$H_2O_2$–HF) in a microwave digester, and analyzed using inductively coupled plasma–mass spectrometry (ICP-MS, Perkin Elmer Elan 6100)

(l) Joshi and Balasubramanian (2010); digestion of the filter paper containing suspended solids in a microwave digester with a combination of strong acids $HNO_3$–$H_2O_2$–HF, followed by analysis on ICP-MS

(m) Tan et al. (1997); sequential extraction; analysis on a Perkin-Elmer ICP Plasma Emission Spectrophotometer

(n) Wu and Ting (2006); total digestion according to US EPA SW 846 Method 3050B using ICP-OES (Perkin-Elmer Optima 3000V)

(o) Wood et al. (1997); digestion with a mixture of $HNO_3$, $HClO_4$, and HF; analysis by atomic absorption spectrometry (AAS)

(p) Wang et al. (2022); means (3 reps) for three parks and eight industrial areas. ICP-OES; microwave-assisted digestion (details uncertain)

(q) Cuong et al. (2005); micro-wave assisted HNO3-HF digestion; Perkin-Elmer AAnalyst 600 GFAAS

(r) Nayar et al. (2004); digestion with suprapure $HNO_3$; Perkin-Elmer AAnalyst 600 GFAAS

(s) Goh and Chou (1997); digestion with suprapure $HNO_3$; analysis on Hitachi Polarised AAS

(t) Sin et al. (1991); digestion not specified; AAS/FASS

are particularly informative to our study because they show loading of Cu, Pb, and Zn in soils are higher in the proximity to industrial areas compared with natural or residential areas (Table 2). Chen (1999) found very high maximum values of Cu (485 mg/kg) and Zn (3594 mg/kg) in industrial soils. In the most recent study, Wang et al. (2022) found maximum concentrations of Cu and Zn were an order of magnitude higher in soils adjacent to industrial activities, compared with parks.

The main sources of high PTE concentrations in Singapore soils are likely vehicular phenomena and industrial activities (Zhou et al. 1997; Wood et al. 1997). For example, Yuen et al. (2012) reported roads in or near industrial areas had higher levels of PTEs in surficial dust compared with residential areas (Table 2). Joshi et al. (2009) reported very high concentrations of Cu ($9069 \pm 3742$ mg/kg) and Zn ($1696 \pm 446$ mg/kg) for road sediment in industrial areas. Wood et al. (1997) reported that elevated Pb and Zn levels in sediments of the Johor Straits along the Malaysia-Singapore causeway likely resulted from gasoline and tire wear, respectively. Fly ash from incinerators in Singapore have very high concentrations of PTSs (Table 2): Cu (570–087 mg/kg), Pb (2000–2150 mg/kg), and Zn (6288–8072 mg/kg). Potential contamination sources in Singapore are indicated in the map in Additional file 3: Fig. S2.

A prior study of airborne element-bearing aerosols in Singapore showed an enrichment of a variety of PTEs (Pb, Zn, Cd, V, Ni, Cr, and Cu) that originate from several sources including oil-fired power plants, element processing industry, land reclamation, construction activities, municipal solid waste incinerators, and traffic emissions (Balasubramanian and Qian 2004). Ng et al. (2006) determined that element accumulations in lichens across Singapore were the result of atmospheric deposition, with peak concentrations of Cu (45 mg/kg), Pb (17 mg/kg), and Zn (84 mg/kg) found in locations associated with heavy petroleum refinement, shipping industries, and road traffic, respectively. Rivellini et al. (2020) found evidence that trace elements in aerosols may originate from shipping emissions and the oil refinery industry. Once airborne, these contaminants can be deposited in the environment during rain events and via dry dust deposition (López et al. 2019).

Hu and Balasubramanian (2003) determined that the removal of the trace elements from the atmosphere by precipitation was influenced by the rainfall volume as well as pH. Further, the magnitude of mean annual wet deposition fluxes of combustion-generated elements (V, Ni, and Cu) were high relative to other locations because of abundant rainfall. The southwest monsoonal winds may cause slight increases in PTE concentrations of airborne particles due to the presence of industrial estates and petrochemical plants in the southwestern region of Singapore (Balasubramanian et al. 2003; Rivellini et al. 2020). Another potential source for the occasionally elevated concentrations of Cu and Zn found in residential areas of Singapore is the degradation of buildings materials.

## Methods and materials

### Sampling and analyses

We adopt an exploratory survey of PTEs in soils across the breadth of the island of Singapore to eliminate spatial bias that may be related to proximity to probable sources of element pollution. In May–June 2018 we collected 300-g samples from 10 sites in the following three land uses (Fig. 2): woodland; parks; and community gardens (raw data reported in Additional file 1: Table S2).

Soil pH was measured in a soil suspension with 1:5 soil:water ratio using a calibrated 9157BNUMD RossUltra pH/ATC Triode with AgCl–Hg electrolytes (Thermo Fisher Scientific, Waltham, USA). Soil particle size was determined using the Mastersizer 2000 (Malvern Instruments, Malvern, UK) via the laser obstruction method. Total organic carbon was determined by combusting 0.03 g of ground soil that was first pre-wetted, fumigated for 6 h to remove inorganic carbonate, and then oven-dried. The samples were combusted at 950 °C in the Vario TOC Cube (Elementar Analysensysteme GmbH, Langenselbold, Germany).

Soil element psuedo-total concentrations were determine using the PerkinElmer Optima 8300 Inductively-Coupled Plasma Optical Emission Spectrometer (ICP-OES). Prior to analysis we used a microwave-assisted acid approach for digestion (US EPA Method 3051A17). First, 0.5 g of the finely ground material was placed in digestion vessels before being reacted with 9 ml of Nitric acid (65%) and 3 ml of Hydrochloric acid (37%) in a fume hood for 15 min. The samples were then heated in a microwave digester, ramping up to 175 °C within 10 min, then maintaining that temperature for another 15 min. Multi-elemental standard solutions of 0.5 mg/kg, 1 mg/kg, 5 mg/kg and 10 mg/kg in 2% $HNO_3$ were used to calibrate the ICP-OES instrument.

**Table 3** Quality assurance/quality control information for Al, Cd, Cu, Pb, and Zn determinations for one standard reference material (USGS 2711a)

| | Cd | Cu | Pb | Zn |
|---|---|---|---|---|
| Certified value (mg/kg) | 54.1 ± 0.5 | 140 ± 2 | 1400 ± 10 | 414 ± 11 |
| Analyzed value (mg/kg) | 58.6 ± 0.8 | 190.6 ± 8.6 | 1462 ± 13 | 418.4 ± 4.7 |
| Recovery (%) | 108 | 136 | 104 | 101 |
| Bias (%) | 8.3 | 36.1 | 4.4 | 1.1 |
| Precision | 0.01 | 0.05 | 0.01 | 0.01 |

Certified value is the concentration reported by the SRM manufacturer; analyzed is the mean ± Stdev determined by ICP-OES (this study); recovery = analyzed value/certified value x 100%; bias = ((analyzed value − certified value)/certified value) × 100; precision = Stdev/mean (ie., coefficient of variation of analyzed values)

We used the USGS 2711a Certified Reference Material to perform quality control and quality assurance of both the accuracy and repeatability of PTE measurements (Table 3). The percentage recovery of PTEs in this research (% Lab recovery) is generally higher than the recovery rates from the USGS Contract Laboratory Study Programme (CLSP). We found higher recoveries compared to those obtained in the USGS CLSP, which mainly used the weaker EPA Method 200.7 (US EPA 2001) for soil digestion via hotplate reflux (Mackey et al. 2010). Microwave-assisted digestion results in higher recoveries because the high temperatures and pressures achieved in microwave vessels allows for a more complete digestion of the soil's PTEs (Chen and Ma 1998, 2001). Closed microwave vessels also reduce the potential loss of trace elements through volatilization, which is a feature in open hotplate digestion (Link et al. 1998).

The accuracy and bias of Cd, Pb, and Zn are acceptable; however, the over estimation of Cu (34%) is noted and considered in our interpretation of the results. We deemed that ICP-OES and di-acid digestion approach was acceptable for the expected range of total concentrations that would indicate potential risk (Olesik 2020). Over the years, a variety of methods have been used to determine pseudo concentrations of PTEs in garden soils (summarized in Additional file 1: Table S1).

Finally, we determined lead isotope ratios in five samples to investigate the origin of Pb at selected sites (Komárek et al. 2008). In doing so we assume that Pb is a representative indicator of general pollutant dispersal pathways (Wiederhold 2015; Harvey et al. 2017; Carrasco et al. 2018; Crispo et al. 2021). We analyzed the soil from five sites, one woodland (#4) and two each for parks (#1 and 7) and gardens (#2 and 10), to track likely sources of Pb in the three community spaces (see locations on Fig. 2). The Pb isotope values were determined by the Environmental Chemistry and Microbiology Unit at the Charles Darwin University in Darwin, Australia. The samples were selected because they were of relatively high concentrations. The finely-ground samples underwent acid digestion with $HNO_3$ and $HClO_4$. Isotope values were determined using an Inductively-Coupled Plasma Mass Spectrometer.

### Calculations and statistical handling

As the number of soil samples collected at each site were few (n = 10), we used nonparametric statistics for data inference, including the Spearman rank correlation test, the Kruskill Wallis (KW) for multiple treatment comparisons, and the Mann–Whitney U (MW-U) as a post-hoc test. Differences were considered significant at $\alpha = 0.05$ (P-values < 0.05). We used an Enrichment

Factor (EF) to assess the anthropogenic enrichment of PTEs in the soils of different land covers (Chester and Stoner 1973). Although the validity of EF values is debated (Reimann and Garrett 2005), we find them useful for indicating sites of potentially high contamination (Szolnoki et al. 2013). In the calculation of EF, we used Al as a normalizing element:

$$EF_M = \frac{\left(\frac{X_M}{X_{Al}}\right)}{\left(\frac{BG_M}{BG_{Al}}\right)}, \quad (1)$$

where $X_M$ and $X_{Al}$ are the concentrations of an element M and Al in the sample, respectively; and $BG_M$ and $BG_{Al}$ are their calculated background values. For background values we used those from a prior study conducted in the forested Nee Soon Forest Catchment (Nguyen et al. 2019): Al (72,900 mg/kg); Cd (0.2 mg/kg); Cu (5 mg/kg); Pb (13 mg/kg); and Zn (23 mg/kg).

### Soil element concentration risk guidelines

Finally, as Singapore does not have published guidelines pertaining to health risks associated with concentrations of PTEs in soils, we determined three thresholds based on a review of guidelines for nearly 50 countries/regions worldwide (Table 4): (1) Threshold risk concentration (TRC), probable risk concentration (PRC), and elevated risk concentrations (ERC). The values for these three thresholds for Cd, Cu, Pb and Zn (Table 4) are determined as the 20th, 50th (median), and 80th percentile of the values summarized globally (Additional file 1: Table S3). The ERC values are of approximately the same magnitude as those employed by Wang et al. (2022) as "intervention" values. We caution that while these guidelines are useful for examining potential levels of risk in our assessment, they may not be appropriate for policy/management applications without a more critical review.

In Tables 1 and 2 we list studies based on a first-order ranking of level of contamination. For each study we calculate a composite score that is determined by awarding 1 point if the maximum value of an element exceeds the TRC; 2 points if the PRC is exceeded; and 3 points if the ERC is exceeded. Four points are assigned if the maximum is tenfold the ERC (twofold for Cd). For studies, not reporting maximum values we use the mean (median) + stdev (MAD) as the upper value. If an element was not determined (ND) we adjust the ranking to align with level of contamination relative to other studies. These rankings are made to facilitate discussion in this paper, not to compare cities/locations directly (see below).

## Results

### Pseudo-total concentrations

The main difference between PTE presence in soils in gardens, parks, and woodlands was in concentrations of

**Table 4** Number of samples exceeding threshold risk concentrations (TRC), potential risk concentration (PRC), and elevated risk concentrations (ERC) for four metals and three treatments

| **TRC, PRC, ERC concentrations[a]** | | | | | |
|---|---|---|---|---|---|
| **Level[a]** | **Unit** | **Cd** | **Cu** | **Pb** | **Zn** |
| Background | mg/kg | 0.2 | 5 | 13 | 23 |
| TRC | mg/kg | 0.5 | 46 | 57 | 100 |
| PRC | mg/kg | 2 | 100 | 140 | 200 |
| ERC | mg/kg | 10 | 200 | 400 | 700 |
| **Number of samples exceeding TRC, PRC, ERC thresholds** | | | | | |
| **Threshold** | **Land cover** | **Cd** | **Cu** | **Pb** | **Zn** |
| TRC | Garden | 3 | 5 | 0 | 7 |
| PRC | Garden | 2 | 0 | 0 | 0 |
| ERC | Garden | 0 | 0 | 0 | 0 |
| TRC | Park | 0 | 2 | 1 | 1 |
| PRC | Park | 0 | 0 | 0 | 0 |
| ERC | Park | 0 | 0 | 0 | 0 |
| TRC | Woodland | 1 | 2 | 0 | 1 |
| PRC | Woodland | 1 | 0 | 0 | 0 |
| ERC | Woodland | 0 | 0 | 0 | 0 |

[a] The pseudo-total concentration indicators of level of risk are determined as the 20th (TRC), 50th (PRC), and 80th (ERC) percentile of the ranked concentrations from values reported for numerous countries worldwide (summarized in Additional file 1: Table S2). In the bottom panel, n = 10 for each land cover

**Table 5** Summary statistics for PTE pseudo-total concentrations and soil properties for the three land covers

| Element | Units | Gardens | Parks | Woodlands |
|---|---|---|---|---|
| Al | mg/kg | 99,070 ± 10,130 [61412–123700] | 118,850 ± 18,070 [96980–143440] | 110,590 ± 32,970 [40832–156400] |
| Cd | mg/kg | bdl [bdl–3.2] | bdl [bdl–bdl] | bdl [bdl–6.4] |
| Cu* | mg/kg | 42.5 ± 11.5 (b) [10.0–76.6] | 26.4 ± 6.2 (a) [9.2–69.6] | 17.6 ± 5.5 (a) [10.6–49.6] |
| Pb | mg/kg | bdl [bdl–36.6] | bdl [bdl–60.8] | bdl [bdl–32.6] |
| Zn* | mg/kg | 124.8 ± 33.8 (b) [70.8–164.2] | 49.8 ± 11.2 (a) [32.6–189.6] | 51.5 ± 17.4 (a) [19.2–146] |
| TOC | % | 3.4 ± 1.1 [1.3–5.9] | 3.7 ± 1.1 [1.1–5.5] | 4.5 ± 0.8 [1.7–9.0] |
| pH | | 6.9 ± 0.6 (c) [5.8–7.8] | 6.0 ± 0.5 (b) [4.8–8.0] | 5.0 ± 0.7 (a) [4.0–8.6] |
| Clay | % | 7.8 ± 0.5 [5.8–9.7] | 6.9 ± 0.5 [5.8–8.0] | 7.8 ± 0.5 [5.4–9.6] |
| Silt | % | 55.1 ± 2.1 (a) [51.5–62.2] | 67.0 ± 4.1 (c) [52.9–74.9] | 62.2 ± 6.5 (b) [41.9–78.5] |
| Sand | % | 35.5 ± 1.1 (b) [27.1–39.7] | 24.4 ± 1.7 (a) [17.0–38.8] | 29.6 ± 5.5 (b) [12.0–47.8] |

Values reported are medians ± standard absolute deviations (n = 10 samples). Ranges are reported in brackets. bdl refers to below detection limit, which are 1.62 (Cd), 5.82 (Cu), 12.6 (Pb), and 3.54 (Zn). Values in rows with different letters in "()" are significantly different, based on post-hoc testing (KW, followed by MW-U; α = 0.05)

*The differences among landcovers is significant at α = 0.5 (KW Test), with garden values being significantly higher than the other treatments (KW; α = 0.05). Values in rows with different letters in "( )" are significantly different, based on post-hoc testing (MW-U; α = 0.05)

Cu and Zn (Table 5). Firstly, detectable concentrations of Cu and Zn were found in all 30 samples. Garden soils had statistically higher concentrations of Cu (42.5 ± 11.5; [10.0–76.6] mg/kg Cu) than soils in parks (26.4 ± 6.2; [9.2–69.6] mg/kg Cu) and woodlands (17.6 ± 5.5; [10.6–49.6] mg/kg Cu). The values reported are medians ± median absolute deviations; the range is in brackets; and n = 10 for all three treatments. Zinc concentrations in gardens (124.8 ± 33.8 [70.8–164.2] mg/kg Zn) were also statistically ($\boldsymbol{\alpha} = 0.05$; KW) higher than in park soils (49.8 ± 11.2; [32.6–189.6] mg/kg Zn) and woodland soils (51.5 ± 17.4; [19.2–146] mg/kg Zn).

Unlike Cu and Zn, Cd was rarely detectable in any sample (detection limit = 1.6 mg/kg Cd): three garden soils had Cd concentrations ranging from 1.7 to 3.2 mg/kg Cd; one woodland soil sample had a relatively high value of 6.4 mg/kg Cd (Table 5). No detectable Cd was found in park soils. Similarly, Pb was also rarely present in concentrations above our (high) detection limit of 12.6 mg/kg (n = 5, 5, and 4 in gardens, parks, and woodlands, respectively). Detectable Pb concentrations ranged from 13 to 61 mg/kg Pb across the three treatments (Table 5). The maximum concentrations found in gardens, parks, and woodlands were about 36, 61, and 33 mg/kg Pb, respectively. The generally low PTE concentrations we found are generally comparable to those reported by Chen (1999), Zhou et al. (1997), Ng et al. (2006), Nguyen et al. (2018, 2020), and Wang et al. (2022) for a variety of urban soils in Singapore.

### PTE enrichment

In comparison with elemental background concentrations from soils in the forested Nee Soon Catchment, the Cu and Zn values found in soils of the three land uses investigated are considered "enriched" to some degree. For example, the median enrichment factor ($EF_M$; Eq. 1) for Cu in gardens was 7; median $EF_M$ values were 3 and 4 in parks and woodlands, respectively. Simply, these values indicate the medians of the samples measured in gardens, parks, and woodlands are all three- to sevenfold higher than the background concentration of 5 mg/kg Cu. Enrichment factors for Zn ranged from 5 in garden soils to 2 for both park and woodland soils (background = 22 mg/kg Zn). Here, enriched does not necessarily indicate high risk. It may simply indicate an anthropogenic contribution; alternatively, it may simply the particular mineralogical composition of the soil—which again may be imported or natural for that particular location.

### Correlations

The garden soils were slightly sandier (median of 36%) than those in parks or woodlands (24 to 30%; Additional file 3: Table S4). Copper and Zn concentrations were negatively associated with clay content in garden soils ($\rho = -0.663$ and $-0.725$, respectively; $\boldsymbol{\alpha} < 0.01$; Additional file 3: Table S4). In the woodlands, Cu concentrations were positively correlated with silt percentage (0.674), and negatively correlated with the sand fraction ($-0.609$; $\boldsymbol{\alpha} < 0.01$). This same textural association was present to a lesser degree for Zn in woodland soils: silt ($\rho = 0.371$; $\boldsymbol{\alpha} < 0.05$); and sand ($-0.255$; $\boldsymbol{\alpha} < 0.05$). The pH of garden soils ranged from 5.8 to 7.8, with a neutral median value (6.9 ± 0.6; Additional file 3: Table S4). In comparison, the soils of the other two land uses were more acidic (median pH was 5 in the woodland soils; 6 in park soils) and displayed larger ranges: 4.8–8.0 (parks); 4.0–8.6

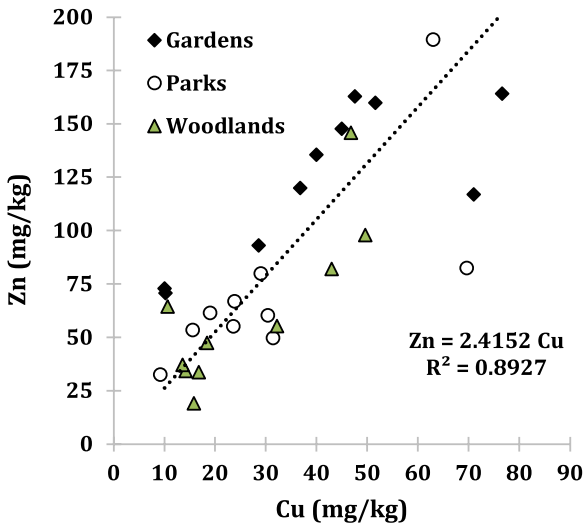


**Fig. 3** Relationship between copper (Cu) and zinc (Zn) in garden, park, and woodland soils in Singapore. The Spearman rank correlation coefficient is $\rho = 0.78$ (p-value < 0.001)

(woodlands). The only significant correlation involving pH was a weak correlation with Cu concentration in garden soils (0.387; $\alpha < 0.01$; Additional file 3: Table S4).

None of the soils had particularly high TOC, with median percentages ranging from 3.4 to 4.5 (Table 5). The highest value was found at a woodland site (9%); low values < 2% were found among all three treatments. TOC concentrations were positively correlated with Cu and Zn in garden soils ($\rho = 0.761$ and 0.667; $\alpha < 0.01$; respectively), as well as park soils (0.447 and 0.667, but with different levels of significance). Collectively, these significant correlations between element concentrations and soil properties provide some indication of elements being bound to the organic matter or particular texture fractions in soils, but without a consistent pattern across land uses. Unexpected is the negative association with clay content (Additional file 3: Table S4).

With regard to the PTEs, the correlation between Cu and Zn was significant ($\rho = 0.78$; $\alpha < 0.001$; Fig. 3). Correlation analysis involving Cd and Pb was hindered by so many samples having concentrations below detection limits.

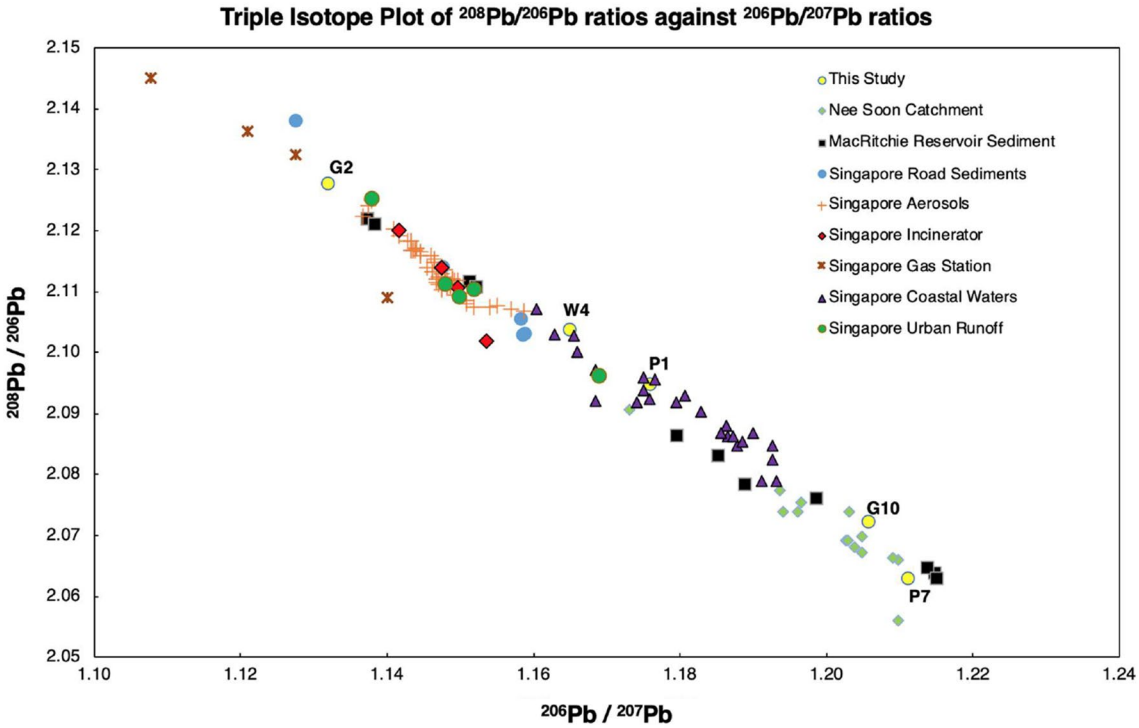


**Fig. 4** Triple Pb isotope plot of Pb sources in Singapore, which includes roadside sediments in blue circles (Yuen et al. 2012), aerosols in orange crosses (Lee et al. 2014; Chen et al. 2016), incinerator ash in red diamonds (Chen et al. 2016), gas station soil residuals in brown asterisks, Singapore coastal water in purple triangles and urban runoff in green circles (Carrasco et al. 2018), MacRitchie reservoir sediments (Chen et al. 2016) and Nee Soon Catchment soil samples (unpublished data)

### Lead isotope ratios

Chen et al. (2016) suggested that due to the close proximity of industrial estates with incinerator plants, Pb pollution in Singapore usually comes from multiple sources including industrial activities, incineration, the past use of leaded petrol, and the weathering of parent rock material. The Pb isotopes for garden site G2 were similar to those found in urban runoff from petrol stations and shipyards (Fig. 4). This result suggests that even though Pb was banned from petrol in 1998 (Chen et al. 2016) the legacy of Pb contamination has persisted, probably because of the low solubility (Sutherland and Tolosa 2001; Yuen et al. 2012). Abrasive blasting of element surfaces in shipyards during repair and construction works are another potential source of Pb aerosols. Considering the down-wind location of G2 from the Tuas shipyards, the aerial transport of Pb particles to the site is plausible. Wang et al. (2022) reported high concentrations of selected elements in the plant foliage in this area—although lead was not elevated. Again, potential sources of pollution are shown in the map in Additional file 3: Fig. S2.

Sample W4 is similar to the roadside sediments collected from Tuas and Senoko Industrial Estates and the incinerator from Keppel Seghers (Fig. 3). Again, incinerator ash from Singapore contains high concentrations of Pb (≥ 2000 mg/kg; Table 2). The presence of Pb at W4 could therefore be due to the upwind (S and SW winds) location of the Keppel Seghers incinerator and the airborne transport of Pb in fly ash from industrial activity in Tuas and West Coast, followed by deposition (Hu and Balasubramanian 2003). Industrial emissions of Pb also include plastics production and element processing plants (Ragaini et al. 1977; Siddiqui et al. 2009), which are found at the Tuas (southwest) and Senoko industrial estate in the north.

The Pb isotope ratio at park site P1 is similar to that of coastal waters of the Central Business District, Kranji and Sembawang (Fig. 3). Rahman et al. (1979) state that element accumulation in Singapore waters and marine sediments must come from natural sources or be introduced from land- and marine-based activities, including leakage from landfills, oil/fuel spills, and/or runoff from shipyards, wood mills, motor works shops, and small industries. Lead concentrations in marine sediments at seven locations ranged from 20 to 28 mg/kg Pb (Rahman et al. 1979). The Pb ratios for P1 likely reflect the mixing of natural Pb and that associated with land-based runoff; however, this inference has uncertainty.

Two of the soil samples tested did not contain an anthropogenic Pb signature (Fig. 3). Isotopic ratios for sites P7 and G10 are close to those found at the base of the MacRitchie Reservoir sediment core, which is believed to reflect naturally occurring Pb in Singapore that was transported with eroding material from the forested headwaters. The G10 signature is most similar to those from soils in the relatively undisturbed Nee Soon Catchment on the island.

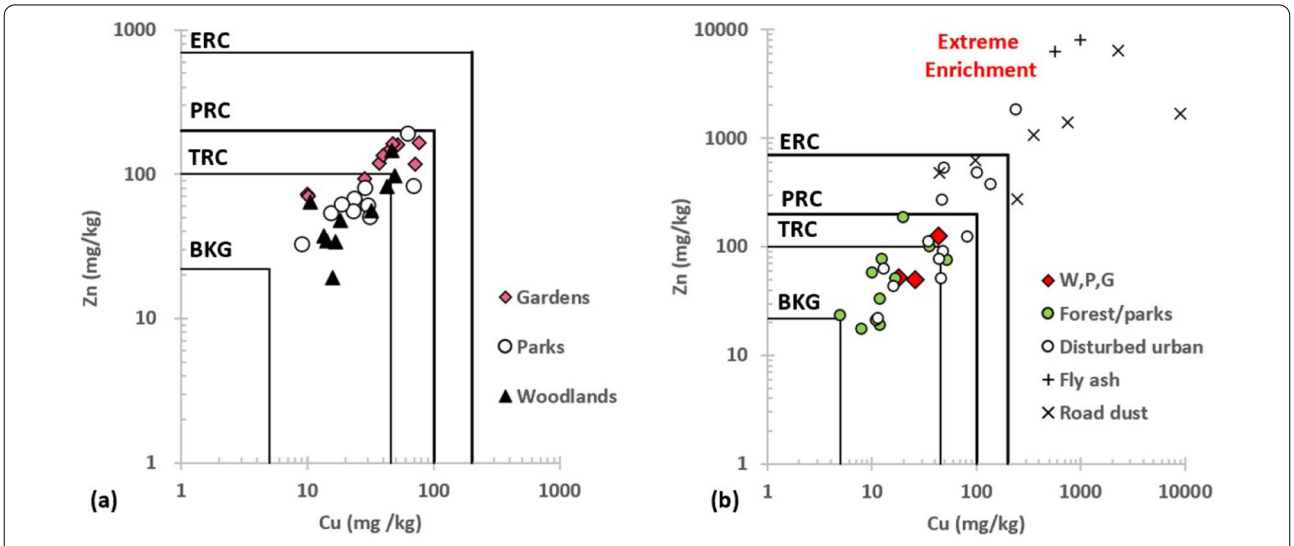


**Fig. 5** Plots of copper (Cu) versus zinc (Zn) concentrations for: (a) garden (G), park (P) and woodland (W) sites investigated in this study (n = 10 samples each); and (b) various soils, road dust, and fly ash in Singapore (from Table 1), along with median values from this study. Background (BKG), Threshold Risk Concentrations (TRC), Potential Risk Concentrations (PRC), and Elevated Risk Concentrations (ERC) are listed in Table 4 (top panel). In panel a, all thirty values for each land use type are shown. In panel b, element concentrations beyond the ERC thresholds are considered to have a substantial anthropogenic enrichment (as in the case of fly ash and most road dust samples)

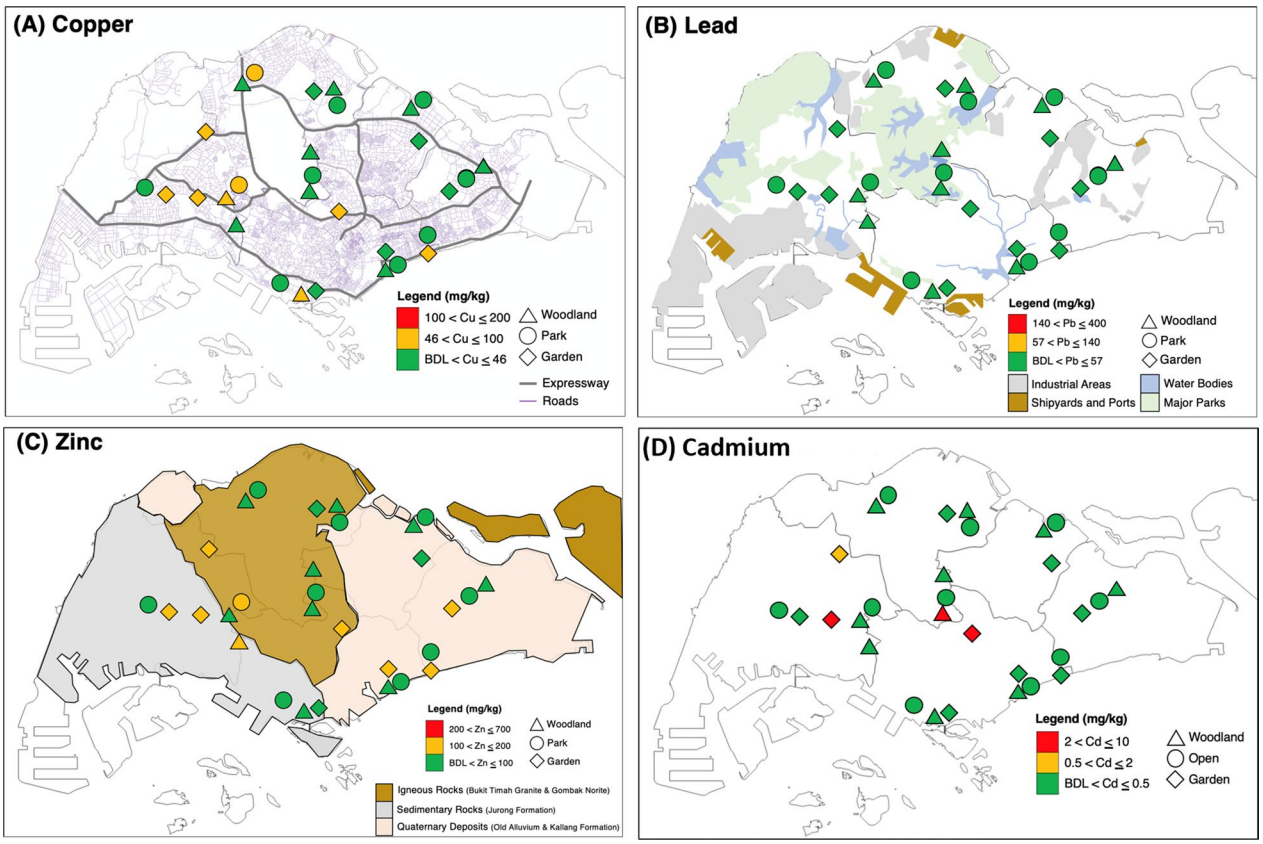


**Fig. 6** Concentrations of Cu (**A**), Pb (**B**), Zn (**C**) and Cd (**D**) at garden (diamond), park (circle), and woodland (triangle) sample sites in Singapore. Concentrations are presented relative to levels of risk (top of Table 4). The background in each map varies to allow comparison of the study sites with the road network (**A**), major parks, industrial areas, water bodies and shipyards (**B**) and large geological formations (**C**). The base maps in **A**–**C** do not necessarily reflect an impact on the element that is mapped

## Discussion

### Spatial patterns

The observed median values of Cu (18–30 mg/kg) and Zn (50–52) for the three treatments are at the high end of the range of central tendency values (reported means, median, midpoints of ranges) associated with soils in forests and natural spaces determined by others for Singapore (Table 2; Fig. 5). The median concentrations for garden soils were slightly higher (43 and 125 mg/kg Cu and Zn, respectively), and therefore, resembled the lower values associated with disturbed soils in Singapore (Table 2). Comparison with reservoir sediment values supports the idea of some degree of anthropogenic enrichment above the natural geochemical signal on the island. With respect to geography, the highest Cu concentrations had no obvious pattern, while Zn concentrations tended to be higher in the western and south-eastern region of Singapore (Fig. 6). Higher Pb concentrations were found in the northern part of Singapore, although a more obvious relation was the occurrence of detectable concentrations in park soils (n = 3) versus garden (1) and woodland soils (1). Again, most Cd concentrations were low and had no strong geographic pattern.

Elevated concentrations of elements in samples from the western and northern area of Singapore were somewhat expected because of the close proximity to industrial element processing, wood combustion, and waste incineration facilities, which are shown in gray shading in Fig. 6b (Nriagu 1979; Wang et al. 2022). Prior analysis of emissions from two incineration plants in the southwest of Singapore showed that high levels of Cu and Zn attached onto particulate matter (Tan et al. 1997), which may lead to elevated concentrations via deposition. We also expected some enrichment in this area because of the role of the southwest monsoonal winds blowing over industrial estates and petrochemical plants (Balasubramanian et al. 2003). Of note, five of the western-most sites had concentrations of Cu and Zn that were higher than their respective TRCs (Fig. 6d; Table 3).

The great variability in Pb isotopic ratios (Fig. 4) indicates different potential sources, both natural and anthropogenic (Fig. 4). The absence of common sources

for lead aligns with the general lack of spatial pattering in the concentrations (Fig. 6). Relevant is that many of the samples enriched in Cu and Zn found in prior studies were often associated with industrial activities in similar locations (Zhou et al. 1997; Chen 1999; Wang et al. 2022). Both our results and those of Wang et al. (2022) suggest that the footprint of many potential pollution sources is relatively small—and therefore contribute to the "spotty" nature of enrichment across the island.

#### Singapore urban garden soil contamination context

The apparent combination of several activities/processes that contribute to the variable enrichment of particular PTEs makes Singapore different from several studies conducted in very contaminated urban environments (Table 1). Briefly, the sites in Table 1 are ordered informally according to general contamination level of four elements. The studies at the top of the table have concentrations that rarely exceed the conservative risk metrics we used herein (Table 4); the locations at the bottom of the table have concentrations for multiple PTE that exceed the ERC concentration indicating elevated risk. Singapore sits at position 27 on this list of 142 entries (Table 1). Again, one should be careful in making literal distinctions among sites/studies owing to differences in methods, timing, and rationales for each study. For example, many early studies were conducted in areas where contamination from known activities was.

Generally, soil PTE concentrations in gardens worldwide tend to be affected by legacy contamination from the following (Table 1): leaded paint and various construction materials used in old neighborhoods (e.g., Baltimore, Christchurch, London, New York, New Orleans, San Francisco, Washington DC), smelting/mining (Baia Mare, Dalnegorsk, Gijon, Palmerton, Paris area) and unsafe waste disposal/usage in gardens as irrigation or fertilizer (Kampala, Paris, Marseille, New York, New Castle). The proximity to heavily used roads may also be an important source of Cd, Cu, Pb, and Zn in gardens, for example, as is the case in Bristol, Cardiff, Hong Kong and New Orleans (Table 1). Older roads may have high levels of Pb stemming from past leaded petrol use.

Noticeable in Singapore is the generally low PTE concentrations in garden, park, and woodlands soils, compared with road-associated sediments and fly-ash (Table 2). As concentrations of Cu and Zn decrease dramatically with increasing distance from roads (Chen et al. 2010; Kim et al. 2017), the moderate PTE concentrations we found are plausible because many of these spaces not often located directly adjacent to roads. Gardens are often ensconced within housing complexes and occasionally sheltered by buildings and trees. Enriched values in garden soils may also be result from degradation of building materials associated with the residential units (Chaney et al. 1984; Lim et al. 2021).

Many authors in the reviewed studies refer to general urban atmospheric deposition of elements in dense cities with much industrial activity and traffic congestion. This type of "new urban deposition" phenomena is likely an apropos description explaining the variations we found in Singapore. In support, Balasubramanian and Qian (2004) reported that the enrichment of aerosols in Singapore was more than 1000 times for Cd, Pb, and Zn; 100 times for Cu. Deposition is almost certainly superimposed on the signatures of the garden soil and may have a legacy signature associated with the age of the building complex. Management activities might be partly responsible for our finding of enriched Cu and Zn concentrations despite the coarse nature of the garden soils, as reported worldwide (Table 1).

For example, the high TOC percentages we found in gardens, which are maintained by application of composted dead organic matter containing element-fixing humic acid, potentially contribute to the enrichment of Cu and Zn in surface soils (Veeken et al. 2000; Amir et al. 2006; Sánchez-Monedero et al. 2002). In addition, NPK fertilizers may facilitate the leaching of Cu and Zn from the topsoil (Young 2013; Bolan and Duraisamy 2003; Loneragen and Webb 1993; Carbonell et al. 2011; Rajneesh et al. 2017; Warman and Cooper 2000). Further, Cu- and Zn-based fertilizers often contain insoluble compounds (e.g., $CuSO_4$, ZnO, and $ZnSO_4$) that potentially elevate Cu and Zn concentrations (Tsang et al. 2014; Jones and Belling 1967). Garden patrons often remedy acidity by applying Ca-rich fertilizers including eggshells to raise the soil pH (Ok et al. 2011), which may in turn facilitate the precipitation of Cu and Zn compounds in the topsoil (Degryse et al. 2006; Kabata-Pendias 2011; McBride 1994). Fertilization may also elevate concentrations in park soils. Focused research would be needed to understand these processes and relationships more thoroughly.

#### Environmental risk of gardening

In general, comparison of the pseudo-total concentrations of PTEs with risk thresholds do not indicate substantial risk. In the Singaporean context where gardening is an activity that is promoted as beneficial for the growing, aging community to be active and self-sufficient, our analysis indicates limited risk of exposure to contaminated soil. Most Singapore gardeners only spend a few hours per day in their gardens (Goh 2018); and therefore, inhalation and exposure rates are very low (Additional file 3: Table S5). For completeness, however, those concerned about health risks would

also need to consider all exposure routes, including the consumption of produce grown in gardens (visualized in Fig. 1). Others report that consuming the products of community gardens comprise about 60% of overall PTE exposure because some garden produce can bioaccumulate elements via root intake or foliar uptake (Spliethoff et al. 2016; Finster et al. 2004). Plants can potentially accumulate up to 10 times the concentration of Cd and a third of the concentrations of Cu, Zn and Pb in the soil (Kabata-Pendias 2011; Byrne et al. 1976). Dalenberg and van Driel (1990) state that atmospheric deposition is a significant source of Pb for field crops, accounting for 73–95% of total Pb found in plants.

Gardener activities such as applying fertilizers, pesticides, mulching, or soil amendment may influence the presence of elements in the soil (Mortvedt 1996; Atafar et al. 2010), but our analyses did not reveal a consistent association between element concentrations and soil variables in Singapore urban gardens. Nor did we identify sources of substantial contamination. Even for such cases of low levels of contamination, the presence of pollution sources (e.g., industry, road networks) necessitates that urban garden activities should be monitored from time to time to safeguard and inform the public of the level of risk, even if none (cf. Chaney et al. 1984; Kim et al. 2014). Patrons of parks and gardens should be aware that the presence of elements in the soil are a function of background residual signatures and potential enrichment via atmospheric deposition, surface runoff deposition, and application of soil amendments.

Finally, with respect to adverse health risk from consuming elements, bioavailability/inaccessibility should be considered to assess the potential for a toxicity effect if concentrations in soils and in farmed produce are high (Intawongse and Dean 2008). Again, we did not perform this assessment because of low pseudo-total PTE concentrations. We assume that the Singapore context is analogous to that of reported by Izquierdo et al. (2015) in Madrid with slightly higher levels of PTE contamination. They concluded that only in the worst-case scenario of children playing in urban gardens and eating produce grown from them, would risk exceed limits of acceptability. This assessment is in agreement with those of Sialelli et al. (2011) for urban soils in Torino (Italy), and Sipter et al. (2008) studying contaminated vegetables in Gyongyosoroszi (Hungary), but not of Bielicka-Giełdoń et al. (2013) in Koszalin (Poland) for conservative guidelines.

### Limitations

We recognize that our elemental recovery methods were not sensitive enough to determine many low-level concentrations associated with Cd and Pb samples. While this issue did not affect our ability to assess environmental risk to these elements, it did hinder our attempt to identify associations between all elements and soil physical factors. Further the recovery of Cu from the certified reference material was high, indicating our Cu determinations are potentially elevated. However, we did not determine the observed pseudo-total concentrations of Cu to be unrealistically high. With regard to the methods employed, others working in Singapore have had better luck deriving total concentrations using the techniques employing strong or mixed acids and analysis with an ICP-MS, particularly for low concentration elements such as cadmium (Yuen et al. 2012; Nguyen et al. 2018).

We are also somewhat uncertain of how the observed associations in soil pH, TOC, and texture play a role in element retention in garden soils of Singapore. For example, the negative correlation between elements (Cu and Zn) and clay content is inconsistent with most reports of higher element concentrations in clayey soils due to the greater presence of negatively-charged sites for element cation adsorption (Scokart et al. 1983; Quenea et al. 2009; Huang et al. 2014; Thomas and Lavkulich 2015). Nevertheless, some studies have reported greater PTE adsorption on coarse sediments in roadside dust and soils (Varrica et al. 2003; Wang et al. 2006; Acosta et al. 2011).

Finally, we were also limited by the inability to source the location of the soil material used in the construction of the gardens and parks. Doing so would have allowed us to fully understand element enrichment over the background concentrations that have taken place since construction of the gardens. Related, we also do not know the age of the gardens, with respect to element accumulation time.

## Conclusions

While provenance tracking using lead isotopes indicated a range of potential sources for PTEs entering urban gardens, measured pseudo-total concentrations of Cd, Cu, Pb, or Zn did not indicate substantial health risk to those practicing urban gardening in Singapore. Further, estimated intake rates determined in a preliminary study (Goh 2018; Additional file 2: Table S5) were much below guideline specifications of maximum allowable doses, although the method to estimate these rates was coarse. Given the low concentrations of PTEs in the garden soils, we did not expand the analysis to determine element concentrations in produce grown in the gardens or the bioavailability/bioaccessability of the elements if ingested or inhaled. However, when combined with findings

from similar studies performing these assessments worldwide, the observed low pseudo-total concentrations observed leads us to conclude that urban gardening activities represent a viable and safe form of leisure, with the potential to produce food for consumption in Singapore. Risk would increase if vegetables in very contaminated soils were consumed, particularly by small children. These positive results add to the discussions on the viability and sustainability of urban agriculture in most urban settings—in agreement with Brown et al. (2016) and Lupolt et al. (2021) that the benefits often outweigh the risks. With respect to urban contamination in general, the growing work in Singapore, summarized in Table 5, supports the notion of a highly heterogenic signal of PTEs, whereby high concentrations associated with particular point sources, especially roads and industrial activities, are mapped onto a variable natural signal that has probably been also been enriched diffusely by the deposition of PTEs over time.

#### Abbreviations

Al: Aluminum; Cd: Cadmium; Cu: Copper; EF: Enrichment factor; EFm: Median enrichment factor; ERC: Elevated risk concentration; KW: Kruskal–Wallis test; MAD: Median absolute deviation; MW-U: Mann–Whitney U test; Pb: Lead; PTE: Potentially toxic element; PRC: Potential risk concentration; $\rho$: Spearman rank correlation coefficient; TRC: Threshold risk concentration; TOC: Total organic carbon; Zn: Zinc.

## Supplementary Information

The online version contains supplementary material available at https://doi.org/10.1186/s43170-022-00126-2.

**Additional file 1: Table S1.** Ranges of Cd, Cu, Pb, and Zn concentrations reported for surface soils in (peri)urban garden soils worldwide (mg/kg; total/pseudo-total concentrations). **Table S2.** Raw data. **Table S3.** Summary of various health guidelines and threshold risk criteria for Cd, Cu, Pb, and Zn used worldwide.

**Additional file 2: Table S4.** Spearman rank correlation coefficients (rho values) indicating associations between Cu, and Zn and soil clay fraction, silt fraction, sand fraction, total organic carbon (TOC), and pH. **Table S5.** Estimated exposure concentrations of copper, zinc, lead and cadmium to community gardeners compared to maximum allowable contaminants guidelines set out by Singapore AVA, WHO, and USA ATSDR. **Figure S1.** Relationship between Cu and Zn. The Spearman correlation coefficient is $\rho = 0.78$ ($\alpha < 0.001$). There are no meaningful correlations between other elements pairs. Shown in the figure is the linear regression equation and coefficient of determination ($R^2$).

**Additional file 3: Figure S2.** Map of Singapore showing the locations of many of the places mentioned in the discussions, including potential sources of contamination.

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#### Author contributions

GTA: as part of thesis work, conducted all phases of sampling, analysis, interpretation, and writing; ADZ: supervised the research, contributed to data analysis, and writing of the final manuscript; SJR: contributed to interpretation of the findings and writing of final draft. All authors read and approved the final manuscript.

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#### Availability of data and materials

All new data are available in the supplement and at the following site: www.adziegler.com/data/Singapore_gardens_2020.

## Declarations

#### Ethics approval and consent to participate

Not applicable. We did not do research on human subjects.

#### Consent for publication

Not applicable.

#### Competing interests

The authors declare that they have no competing interests.

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