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REVIEW PAPER

Urban areas as emitters of potentially toxic elements to urban and suburban forest ecosystems in Poland: a review*

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Abstract

Heavy metals, metalloids, and many trace elements are harmful to living organisms when they accumulate above a certain level of concentration. These potentially toxic elements (PTEs) are released in large quantities in urban areas due to domestic heating, heavy traffic, industrial emissions, and municipal and industrial waste disposal. A portion of PTEs is transported to neighboring ecosystems, including urban and suburban forests, which serve as important barriers and sinks for pollutants. The impact of PTEs extends to both abiotic and biotic ecosystem components, as well as to human populations that benefit from forest ecosystems. This paper aims to characterize Polish urban areas as sources of PTEs and the channels through which PTEs flow from urban areas to urban and suburban forest ecosystems, with a focus on the existing national literature. The potential consequences of pollution of urban and suburban forests by PTEs are also discuss. Published Polish studies do not allow for a full qualitative and quantitative assessment of PTEs entering forest ecosystems from urban areas via different channels (air, water, human-related channels). National monitoring and scientific investigations have focused primarily on Al, As, Cd, Cu, Fe, Hg, Mn, Ni, Pb and Zn. Many other PTEs have been surveyed less frequently and irregularly. The transfer of PTEs via water and human-related channels is not extensively addressed in the national literature. A more comprehensive approach to this topic is needed in future studies.

Keywords: heavy metals, trace elements, urban ecosystems, forest ecosystems, element flow, pollution

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INTRODUCTION

Potentially toxic elements (PTEs) is an umbrella term commonly used to define a group of elements that occur naturally in the environment but are harmful to living organisms when accumulated above certain levels (Pourret et al. 2021). They include heavy metals, metalloids, and numerous trace elements. The list of elements with potential toxicity to organisms is quite long, and the ones most commonly referred to are: aluminum (Al), antimony (Sb), arsenic (As), barium (Ba), cadmium (Cd), chromium (Cr), cobalt (Co), copper (Cu), gold (Au), iron (Fe), lead (Pb), manganese (Mn), mercury (Hg), molybdenum (Mo), nickel (Ni), selenium (Se), silver (Ag), thallium (Tl), tin (Sn), tungsten (W), uranium (U), vanadium (V) and zinc (Zn) – Nawar et al. (2020). Some of them, i.e. Co, Cu, Ni, Fe, Mn, Zn, Mo, and Se, are essential for living organisms but can cause health problems if consumed above the safe limit (Munir et al. 2022). Others, such as As, Cd, Hg, and Pb, are not vital to biological functions and can be hazardous even at low concentrations (Gruszecka-Kosowska 2019). These elements are listed among the top seven toxic substances on the Substance Priority List of the Agency for Toxic Substances and Disease Registry (ATSDR 2022). In recent decades, numerous human activities, including mineral resource exploitation, metal processing and smelting, industrial emissions[1], fertilizer and pesticide use, wastewater irrigation, and transportation, have led to a significant increase in the accumulation of PTEs in the environment (Pan et al. 2018). Among anthropogenic pollutants, they are of great significance not only for their toxicity but also for their persistence and limited options for neutralization (Pourret et al. 2021).

Currently, 56.9% of the global population lives in urban areas (WorldBank 2022), and it is projected that along with the population growth, the proportion of urban population will increase, reaching 68.4% in 2050 (StatisticsTime 2024). This implies that global urban areas will also experience rapid expansion (Chen et al. 2020). Europe is considered the third most urbanized region in the world (75%) (StatisticsTime 2024), and in Poland 60% of the population is classified as urban (WorldBank 2022).

Urban areas are defined in various ways, from different points of view (e.g., geography, statistics, administration, spatial structure) and for different purposes, e.g., public health, land planning (Breckenkamp et al. 2017), often by population size or density, building density or dependence on non-agricultural economic activities (industry, trade, services) – Dijkstra et al. (2021). National definitions differ substantially, and different levels of urbanization are distinguished (Breckenkamp et al. 2017, Dijkstra et al. 2021). In general, the term 'urban area' refers to cities, towns, and suburbs, and

[1] emission – the total rate at which a solid, liquid or gaseous pollutant is emitted into the atmosphere from a given source; usually expressed as mass per unit time (IUPAC Compendium of Chemical Terminology https://goldbook.iupac.org/)

includes the city itself, as well as the surrounding areas. The EU-OECD introduced the concept of 'functional urban area', which includes a city and its commuting zones (Dijkstra et al. 2019). Karim et al. (2014) defined 'urban ecosystem' as a complex composite of both natural (local climate, geology, and the geographical characteristics) and anthropogenic (population and settlement patterns, use and misuse of resources, and people's behavior governed by their socio-economic conditions) factors. For the purposes of this paper, we have adopted a very broad understanding of the term 'urban area', as a unit that is the antonym of rural area, so an area (not necessarily referring to the boundaries of administrative units) with evident transformations of buildings and infrastructure, accompanying non-agricultural economic activities or resulting from the adoption of urban style (patterns) of living and working (Markowski, Marszał 2006). In urban areas defined as above, a wide range of anthropogenic activities, including domestic heating, heavy traffic, industrial emissions, municipal and industrial waste disposal, results in PTEs loading on the environment (Pan et al. 2018). A large amount of PTEs remain locally, i.e., in the urban area where they are generated and where they cause health problems once they enter human bodies through different pathways (Bartkowiak 2022). However, some part of PTEs is also exported to neighbouring ecosystems. Thus, the urban area is both a source and a sink of pollution. Naturally, urban issues, including pollution, differ between the city center, surrounding dense neighborhoods, adjacent suburbs, and the rural areas inside a commuting zone (Dijkstra et al. 2021). However, when treating urban areas as sources of pollution, it seems justified to consider them as one entity.

The process of urbanization exerts a significant pressure on the environment, with profound, multifaceted impacts that are manifested on local, regional, and global scales (Bai et al. 2017). Pollution of individual environmental compartments is one of the chief issues which, alongside the generation of such pollution by urban areas, is becoming a growing concern as it outpaces our abilities to implement control measures (Grimm et al. 2008).

Forests within urban and suburban areas are important pollutant barriers and sinks (O'Brien et al. 2022). In Poland, forests located within the administrative boundaries of cities and within 10 km from the administrative boundaries of cities with more than 50,000 inhabitants are defined as 'protection forests' (ForestAct 2024). In addition to their protective roles, they perform a multitude of other functions, including recreational, leisure, educational, health, and landscape ones (O'Brien et al. 2022). Currently in Poland, urban and suburban forests cover 632,410 ha managed by the State Forests or owned by municipalities, which together account for 6.7% of the total forest land in the country (RSL 2021). National and regional data on forest land and cover are presented in Table 1. There is no strong relationship between the share of urban and suburban forests in voivodships and the total forest cover of these units.

Table 1

Forest land and urban and suburban forests in Poland by voivodships in 2020 (RSL 2021)

Voivodships	Forest land (ha)	Forest cover (%)	Urban and suburban forests*		
			area (ha)	share (%) in	
				forest land	voivodship area
Poland	9,464,200	29.6	632,410	6.7	2.0
Dolnośląskie (DS)	611,800	29.9	59,202	9.7	3.0
Kujawsko-Pomorskie (KP)	432,200	23.5	70,682	16.4	3.9
Lubelskie (LU)	596,000	23.4	20,207	3.4	0.8
Lubuskie (LB)	710,300	49.3	56,924	8.0	4.1
Łódzkie (LD)	397,500	21.4	43,722	11.0	2.4
Małopolskie (MA)	439,300	28.6	16,584	3.8	1.1
Mazowieckie (MZ)	845,300	23.4	43,239	5.1	1.2
Opolskie (OP)	257,900	26.7	17,245	6.7	1.8
Podkarpackie (PK)	691,800	38.2	32,972	4.8	1.9
Podlaskie (PD)	634,800	31.0	13,605	2.1	0.7
Pomorskie (PM)	685,000	36.4	57,227	8.4	3.1
Śląskie (SL)	405,800	32.1	26,540	6.5	2.2
Świętokrzyskie (SK)	338,000	28.3	31,591	9.4	2.7
Warmińsko-Mazurskie (WM)	787,100	31.7	31,140	4.0	1.3
Wielkopolskie (WP)	788,800	25.8	66,589	8.4	2.2
Zachodniopomorskie (ZP)	842,600	35.8	44,943	5.3	2.0

* total managed by State Forests and owned by municipalities

Forests are regarded as peculiar filters for pollution, with the capacity to not only absorb but also neutralize toxic substances, including PTEs (Yaneva et al. 2022). However, this does not imply that such absorption is without consequences for forest ecosystems. PTEs exert an impact on both abiotic and biotic ecosystem components, as well as on people who benefit from the forest (Kwapuliński et al. 2003). Permanent immission[1] of PTEs may lead to exceeding the level of the chemo-ecological capacity of the forest (understood as this amount of all elements with physiological or toxic properties, which ensures the maintenance of the existing characteristic composition of flora and fauna species) and disturb the ecosystem's homeostasis.

The objective of this review is to characterize Polish urban areas as sources of PTEs and the channels through which PTEs flow from urban areas to urban and suburban forest ecosystems, with a focus on the existing national literature. Furthermore, the article discusses the potential consequences of pollution of urban and suburban forests by PTEs.

[1] immission – the transfer of pollutants from the atmosphere to a 'receptor' (IUPAC Compendium of Chemical Terminology https://goldbook.iupac.org/)

SOURCES AND CARRIERS

In the urban space, in addition to the groups of buildings with various functions and forms and communication systems, typically there are also numerous production and processing enterprises (Gut et al. 2019). Concentration of industries in urban agglomerations or in their immediate vicinity is common in many Polish regions (e.g., Kraków agglomeration, Upper-Silesian agglomeration). Frequently, mining plants and dumps and waste landfills are sited within urban areas (Gut et al. 2019). In such cases, it is hard to separate strictly urban from industrial pollution. Hence, PTEs in urbanized areas may originate from a variety of anthropogenic sources, the main ones being energy generation and transmission, metallurgical activities (mining, smelting, metal finishing, and others), microelectronics manufacture, and waste disposal (Bradl 2005). Additional sources mentioned in the literature include: corrosion of metal structures (including galvanized roofs and fences), releases from asphalt and concrete structures, paints and other decorative materials, fertilizers, composts and pesticides used in urban gardens, contaminants from the former land use (e.g., agricultural or horticultural), bonfires, accidental fires and wartime bombing residues (Alloway 2013, Aryal et al. 2016). PTEs can be released in gaseous, particulate, aqueous, or solid form (Bradl 2005), and emanate from point (e.g., industrial plants), line (e.g., transport vehicles), and area emitters (e.g., emissions related to residential heating) – Kaszewski (2020). The main carriers of PTEs in urban areas from which they can enter forest ecosystems are polluted urban air, street dust, and various types of waste and trash.

Urban air

Pacyna et al. (2007) identified the following major urban source categories for PTEs emissions to the atmosphere in Europe: fossil fuel combustion in utility boilers (power plants), industrial, residential, and commercial boilers; iron and steel production (in which heavy metals are used as additives); cement production; non-ferrous metal manufacturing; waste incineration; gasoline combustion (only for Pb); and other sources (including various uses of metals). PTEs are part of the particulate matter (PM) emitted into the atmosphere, which refers to solid particles and liquid droplets with the diameter from a few nanometers to 100 micrometers. These particles can remain suspended in the air for a long time (Ali et al. 2019). Due to international (CLRTAP 1979, EMEP 1979) and national obligations (EP-EC 2016), the annual release of major PTEs into the atmosphere is published by the National Centre for Emissions Management (KOBiZE; supervised by the Minister of Climate and Environment) as part of the reports on national emissions of greenhouse gases and other substances. Cd, Hg, and Pb are reportable, while to date As, Cr, Zn, Cu, and Ni, are reported on a voluntary basis.

According to the National Pollutant Emission Balance (KOBiZE 2023) for Poland, the current annual emissions of PTEs to the atmosphere are

15.0, 11.0, 48.1, 380, 8.5, 77.4, 280, and 562 tons of As, Cd, Cr, Cu, Hg, Ni, Pb, and Zn, respectively. It is worth emphasizing that over the past 30 years (reference year 1990), the country has seen a decrease in the amounts of As, Cd, Cr, Hg, Ni, Pb, and Zn emitted into the atmosphere by 89.6, 9.1, 14.7, 42.0, 61.2, 48.6, and 29.2%, respectively, and an increase in Cu by 76.8%. Currently, the dominant source of As, Cd, Cr, and Pb emissions are industrial processes, which account for 36, 39, 27, and 55% of total As, Cd, Cr, and Pb emissions, respectively. The largest source of Hg and Ni is fuel combustion in energy industries: 60% and 43%, respectively. As and Zn emissions from the latter sector are also considerably high (35% and 19%, respectively). Combustion of fuels in the transport sector is the source of the largest emission of Cu (69%). It comes mainly from tribological processes in road transport. The prevailing amount of the above-mentioned pollutants is generated in areas of urban-industrial agglomerations. Detailed information on emission volumes by sectors for urban areas is not published regularly.

Measurements of As, Cd, Ni, and Pb content in PM_{10} dust (particulate matter 10 μm or less in diameter) in urban areas are carried out by the Chief Inspectorate for Environmental Protection within air quality monitoring, and the results (mainly as annual average concentration) are published in annual regional/voivodeship reports (GIOS 2024) and an online database (GIOS-BDP 2024). According to the national Environmental Protection Act (EnvProtAct 2001), in each voivodeship three zones are distinguished in air quality assessments: (i) agglomeration with a population of more than 250 thousand (e.g., Warsaw, Wrocław, Kraków, and 9 others), (ii) city (not being an agglomeration) with more than 100 thousand inhabitants (e.g., Olsztyn, Opole, Radom, and 15 more), and (iii) the remaining area of the voivodeship, not included in (i) or (ii) zone. Within zone (iii), three types of areas are additionally specified in which measurement stations are located: urban (cities or towns with fewer than 100 thousand inhabitants), suburban and non-urban ones. Data from stations located in the latter areas represent the background pollution for Poland. The concentrations of Hg (total gaseous mercury, TGM) in the air are reported only from non-urban areas, as measurements are made only in selected regional background measurement stations. The range of annual Hg(TGM) concentrations at these sites between 2010 and 2020, i.e. since the start of monitoring, were 1.074-2.388 ng m^{-3} (GIOS-BDP 2024), while the global background air pollution was defined as 1.5-2.0 ng m^{-3} (GIOS 2021*b*). Values above 2.0 ng m^{-3} were recorded only in 2010 and 2011 at a background station in Dolnośląskie Voivodship. Considering the sources of Hg emissions and the fact that Poland is the country with the highest Hg emissions among the European Union countries, it is obvious that air in urban areas is contaminated with this element to a greater extent. To date, no standards have been set for Hg(TGM) concentrations (GIOS 2021*b*).

Ranges of daily and annual concentrations of As, Cd, Ni, and Pb in the air of urban areas in different categories are presented in Table 2. For As,

Cd and Ni, the highest values were recorded in urban areas with fewer than 100 thousand inhabitants, while for Pb – in cities (100-250 thousand inhabitants). Exceedances of target values were recorded most frequently for As, and were related to the industrialization of the region (voivodeship) rather than to the type of urban area. In Dolnośląskie Voivodeship, exceedances of the As target level in the air were noted even in non-urban areas.

Table 2

Ranges of PTE concentrations in airborne particulate matter (PM_{10}) in Poland in 2000-2020, including urban areas (ng m^{-3}), based on the database of the Chief Inspectorate for Environmental Protection of Poland (GIOS-BDP 2024)

Area type	As	Cd	Ni	Pb
Range of daily averages recorded				
Agglomerations	0.0 - 90	0.0 - 51.0	0.0 - 155	0.0 - 1004
Cities	0.0 - 161	0.0 - 60.9	0.0 - 109	0.0 - 4790
Urban#	0.0 - 342	0.0 - 87.2	0.0 - 164	0.0 - 2164
Suburban	0.0 - 16	0.0 - 5.6	0.0 - 18.1	0.0 - 2530
Non-urban	0.0 - 119	0.0 - 47.4	0.0 - 50.0	0.0 - 489
Range of annual averages recorded				
Agglomerations	0.07 - 9.1	0.11 - 2.6	0.005 - 12.1	1.87 - 206
Cities	0.0 - 18.0	0.00 - 4.6	0.0 - 13.0	1.90 - 664
Urban	0.09 - 30.2	0.04 - 7.1	0.006 - 28.1	1.43 - 245
Suburban	0.0 - 2.3	0.0 - 1.1	0.0 - 6.7	3.16 - 162
Non-urban	0.17 - 6.3	0.06 - 2.5	0.25 - 5.5	1.64 - 69
Limit* / target** †	6**	5**	20**	500*
Recorded limit/target exceedances				
Agglomerations	KP: 2010	n.r.	n.r.	n.r.
Cities	DS: 2004-2008, 2011-2018	n.r.	n.r.	DS: 2003
	LB: 2013	n.r.	n.r.	n.r.
Urban	DS: 2006, 2013-2020	KP: 2008	DS: 2006	n.r.
	KP: 2009, 2010	PK: 2008	n.r.	n.r.
	LB: 2011-2013, 2017	DS: 2003, 2004	n.r.	n.r.
	SL: 2007	n.r.	n.r.	n.r.
Suburban	n.r.	n.r.	n.r.	n.r.
Non-urban areas	DS: 2002, 2003	n.r.	n.r.	n.r.

cities or towns with fewer than 100 thousand inhabitants, † according to ME-PL 2012, n.r. – not recorded

Information concerning the presence and concentrations of other PTEs in the air of urban areas in Poland can be found in numerous scientific articles. Most of the studies were inspired by a concern for the health of urban dwellers; however, the results can provide an informative basis for assessing potential threats to adjacent ecosystems. Selected examples are presented in Table 3. Understandably, most attention has long been paid to urban areas in highly industrialized regions, such as Upper Silesia and Kraków regions (Wyszyńska et al. 1970, Rogula-Kozłowska et al. 2013, Rachwał et al.

Table 3

PTEs detected in the air of different urban areas in Poland (ng m^{-3}) – selected examples

City	Sampling site	Sampling period	PM fraction	Element and content (average, average ± SD, or range)	Analytical technique	References
Gdańsk (PM)	urban, n.s.	heating# 2010	$PM_{2.5}$	As (1.0); Cd (1.3); Ni (1.0); Pb (32.2); *n* = 6	ETAAS; $FAAS_{(Pb)}$	(Rogula-Kozłowska et al. 2014)
		non-heating† 2010		As (0.4); Cd (0.3); Ni (1.8); Pb (12.5); n = 6		
Katowice (SL)	urban, n.s.	heating 2010	$PM_{2.5}$	As (3.0); Cd (2.1); Ni (2.6); Pb (65.3); *n* = 6		
		non-heating 2010		As (1.4); Cd (1.2); Ni (2.8); Pb (33.8); *n* = 6		
Katowice (SL)	urban, n.s.	Jan-Dec 2007	$PM_{2.5}$	Ag (2.8±2.1); Al (137±81.6); As (2.1±4.0); Ba (4.2±4.3); Br (9.1±8.1); Cd (2.3±2.2); Co (0.4±0.4); Cr (1.8±1.3); Cu (8.2±3.6); Fe (157±69.1); Mn (8.8±6.9); Mo (2.9±2.5); Ni (0.4±0.9); Pb (35.1±21.0); Rb (0.3±0.5); Sb (5.6±3.8); Se (1.0±1.2); Sr (3.0±2.5); Te (1.6±2.4); Ti (4.2±5.6); V (0.5±0.8); Zn (90.3±49.9); *n* = 39	EDXRF	(Rogula-Kozłowska et al. 2013)
Kraków (MA)	residential	1992-1999 (daily, once a weak)	PM_{10}	Cd (0.2-13.0); Cr (0.20-106); Cu (6.0-1615); Fe (6.0-5635); Pb (6.0-434); Zn (14.0-1443); *n* = n.s.	AAS	(Szafraniec, Jedrychowski 2001)
	industrial			Cd (0.3-43.0); Cr (1.0-154); Cu (21.0-520); Fe (8.0-35 200); Pb (16.0-739); Zn (15.0-1935); *n* = n.s.		
	roads			Cd (0.4-10.4); Cr (0.6-148); Cu (25.0-417); Fe (14.1-14 435); Pb (21.0-1147); Zn (34.0-1122); *n* = n.s.		
Kraków (MA)	urban, n.s.	2 Mar - 14 May 2015	$PM_{2.5}$	Br (15.0±13.0); Cr (2.7±2.3); Cu (6.5±6.1); Fe (172±200); Mn (6.3±1.1); Ni (0.7±0.6); Pb (28.0±21.0); Rb (1.9±0.7); Sr (1.1±0.6); Ti (21.0±8.0); Zn (76.0±62.0); *n* = 48	EDXRF	(Samek et al. 2017)
		9 Mar - 22 May 2016		Br (11.4±4.3); Cr (2.5±0.3); Cu (4.3±2.0); Fe (123±56.0); Mn (9.8±3.2); Ni (3.7±0.8); Pb (20.9±9.9); Rb (1.4±0.3); Sr (0.4±0.2); Ti (11.3±1.1); Zn (69.0±25.0); *n* = 48		

cont. Table 3

Łódź (LD)	urban, n.s.	winter 2001 -2003	PM_{10}	Ag (0.6-2.3); As (0.5-0.8); Cd (0.4-1.0); Co (0.1-0.3); Cr (5.5-7.4); Hg (0.1-2.2); Mn (5.3-14.6); Ni (4.2-12.9); Pb (12.2-26.9); Sb (1.4-4.2); Th ($2 \cdot 10^{-5}$-0.1); U (0.03-0.14); V (0.4-2.2); Zn (69.4-90.0); n = n.s.	ICP-MS	(Krzemińska--Flowers et al. 2006)
		spring 2001 -2003		Ag (0.5-0.6); As (0.2-0.5); Cd (0.2-0.8); Co (0.1-0.3); Cr (4.3-5.6); Hg (0.03-0.06); Mn (3.5-11.1); Ni (2.2-7.1); Pb (3.8-23.6); Sb (0.7-4.0); Th (0.02-0.08); U (0.02-0.03); V (1.0-2.3); Zn (25.6-66.5); n = n.s.		
Radom (MZ)	urban, n.s.	spring 2016	PM_{10}	Cr (1.1); n = 3	AAS	(Molik et al. 2018)
		winter 2015/16		Cr (2.1); n = 7		
Warsaw (MZ)	urban back-ground	26 Nov - 25 Dec 2013	$PM_{2.5}$	Al (217±183); As (10.6±34.4); Br (6.4±9.6); Cd (9.6±22.2); Co (9.4±13.7); Cr 1.2±1.4); Cu (13.2±26.6); Fe (84.6±142); Mn (3.7±11.7); Ni (3.5±5.0); Pb (21.9±26.1); Sc (5.5±7.5); Se (12.7±30.5); Sr (4.1±3.5); Ti (1.7±2.9); V (15.1±32.7); Zn (16.5±20.2); n = 30	EDXRF	(Majewski, Rogula--Kozłowska 2016)
Wrocław (DS)	urban, n.s.	Dec 2009 - Mar 2010	PM_{10}	Al (195±121); As (4.9±3.2); Cr (4.5±1.1); Cu (48.0±9.0); Fe (422±186); Mn (28.0±8.0); Ni (2.2±0.7); Pb (83.0±22.0); Sr (3.9±1.3); Ti (37.0±20.0); Zn (235±102); n = n.s.	EDXRF	(Zwoździak et al. 2013)
		Apr - Sept 2010		Al (445±287); As (3.2±2.7); Cr (4.2±3.4); Cu (36.0±18.0); Fe (604±368); Mn (23.0±10.0); Ni (2.2±2.4); Pb (38.0±9.0); Sr (3.4±2.4); Ti (61.0±42.0); Zn (74.0±43.0); n = n.s.		
Zabrze (SL)	crossroads	Jan 2006	PM_{10}	Cd (77.0); Cr (983); Cu (590); Fe (28557); Mn (372); Ni (296); Pb (1683); n = 17	AAS	(Pastuszka et al. 2010)
	urban background			Cd (7.0); (Cr 50.0); Cu (45.0); Fe (1706); Mn (50.0); Ni (17.0); Pb (224); n = 17		

cont. Table 3

Zabrze (SL)	urban, n.s.	Jan - Mar 2009	$PM_{2.5}$	Ag (1.0); Al (0); As (9.6); Ba (<DL); Br (25.1); Cd (1.1); Co (<DL); Cr (1.3); Cu (8.9); Fe (180); Mn (21.6); Mo (2.7); Ni (0.4); Pb (49.7); Rb (0.03); Se (0.5); Sb (3.7); Sr (1.3); Te (0.5); Ti (0.5); V (<DL); Zn (184); n = 22	EDXRF	(Rogula--Kozłowska et al. 2012)
			$PM_{2.5-10}$	Ag (2.2); Al (400); As (<DL); Ba (1.6); Br (<DL); Cd (<DL); Co (<DL); Cr (0.7); Cu (4.2); Fe (290); Mn (7.9); Mo (2.1); Ni (0.8); Pb (4.3); Rb (0.4); Sb (<DL); Se (0.02); Sr (2.3); Te (0.3); Ti (13.5); V (<DL); Zn (17.8); n = 22		
Zabrze (SL)	urban, n.s.	24 Oct 2018 -1 Jan 2019	TSP	Al (9330±2190); As (2.7±1.0); Ba (22130±5040); Cd (0.8±0.7); Cr (7.0±4.8); Cu (13.0±8.5); Fe (243±168); Mn (14.5±16.9); Ni (6.3±6.0); Pb (32.2±27.0); Ti (13.1±7.3); Tl (0.4±0.4); V (0.7±0.3); Zn (17210±3830); n = 35	HR-ICP-MS	(Rachwał et al. 2020)

n.s. – not specified, # heating season – October-March, † off-heating season – April-September, PM – particulate matter, TSP – total suspended particles, <DL – below detection level, ETAAS – electrothermal atomic absorption spectroscopy, FAAS – flame atomic absorption spectrometry, EDXRF – energy dispersive X-ray fluorescence, AAS – atomic absorption spectrometry, ICP-MS – inductively coupled plasma mass spectrometry, HR-ICP-MS – high-resolution inductive coupled plasma-mass spectrometry

2020). Research has focused on a different range of elements, from one (Molik et al. 2018) to a dozen (Rogula-Kozłowska et al. 2013), different sites of urban areas, e.g., highways, crosswords (Pastuszka et al. 2010, Rogula--Kozłowska 2015), different time periods of the year, e.g. different seasons, months, heating and non-heating periods (Krzemińska-Flowers et al. 2006, Rogula-Kozłowska et al. 2012, 2014), and different PM fractions, e.g. PM_1, $PM_{2.5}$, PM_{10}, $PM_{1-2.5}$. Rogula-Kozłowska (2015) presented detailed chemical composition of thirteen PM fractions: from $PM_{0.03-0.06}$ to PM_{10-40}. Studies have been conducted in a single city or town (Szafraniec, Jedrychowski 2001, Majewski, Rogula-Kozłowska 2016) or in multiple cities simultaneously (Wyszyńska et al. 1970, Rogula-Kozłowska et al. 2014). Table 3 shows that in many cases a larger standard deviation (SD) than the average was observed, indicating a wide range of variability in the data over the study period. Varying weather conditions, especially wind directions, are considered to be one of the major causes of this variability (Rogula-Kozłowska et al. 2013).

Urban street dust

A large part of PTEs bound with airborne PM becomes deposited by dry (sedimentation under gravity) or wet deposition (falling with precipitation) (Nowak et al. 2014), and accumulates on impervious surfaces in urban areas (e.g., sidewalks, roadways), creating street/road dust with other pollutants (Zgłobicki et al. 2019, Skorbiłowicz et al. 2020). PTEs in street dust are bound to solid mineral or organic particles of various origins, including industrial and traffic-related ones. In addition to deposition, street/road dust solid particles containing PTEs are generated by any mechanical treatment of materials, including crushing, grinding, violent impact, handling, detonation, and decomposition of organic and inorganic materials, such as rocks, ores, and metals (Khan, Strand 2018). PTEs enter street dust from vehicle engine exhaust fumes, from fluids released from vehicles onto the street surface, and from abrasion of various vehicle parts (tires and brake rotors), road surfaces or overhead lines (used by streetcars, trolleybuses, and trains) – Zgłobicki, Telecka (2021). Deposited street/road dust can also be resuspended by wind force or other anthropogenic activities (Jancsek-Turóczi et al. 2013). Table 4 presents examples of studies demonstrating the role of street dust as a source of PTE pollution in urban environments and as a potential carrier of PTEs to forest ecosystems. While the latter aspect is evident, it is challenging to quantify explicitly. The studies were carried out in different cities, at different time points, and covered diverse ranges of determined elements. The most common elements in the lists of PTEs under investigation were Cd, Cr, Cu, Mn, Ni, Pb, and Zn, and their concentrations in street dust exhibited considerable temporal and spatial variability. In some studies, however, rare elements such as yttrium (Y), gallium (Ga), zirconium (Zr), and rubidium (Rb) were also found in the street dust (Gruszecka-Kosowska, Wdowin 2016). Using Lublin as an example (Table 4), it seems optimistic that many PTEs had lower street dust levels in 2018 than in 2013.

Urban watercourses

Stormwater and wastewater are potential sources of PTEs in urban watercourses (Sakson et al. 2018). Pollutants can be discharged into surface waters from a number of sources, including wastewater treatment plants, outfalls from drainage systems, combined storm overflows and surface runoff from impervious surfaces. The problem is exacerbated by system failures and during intense rainfall when collectors and sewage treatment plants reach their maximum capacity. According to European reports, the list of PTEs contained in stormwater is quite long (Eriksson et al. 2007), with Zn, Cu, Pb, and Cd being the most commonly studied elements.

As reported by Górska and Sikorski (2013), the concentrations of Cd, Cr, Cu, Ni, Pb, and Zn in the stormwater runoff flowing into a sewer system from the area of a catchment in the center of Kielce during a snowmelt period (February-March 2010) averaged 19, 41, 119, 100, 400 and 387 μg L^{-1},

Table 4

PTEs detected in street/road dust in different urban areas in Poland (mg kg^{-1}) – selected examples

City	Sampling period	Other information	Element and content	Analytical technique	References
Białystok (PD)	spring, 2018; dry weather	large congestion[#]	average±SD: Cr (9.1±3.1); Cu (16.4±14.9); Fe (2355±2246); Mn (68.6±32.6); Ni (5.2±2.7); Pb (11.4±15.2); Zn (69.9±40.2); *n* = 32	FAAS	(Skorbiłowicz et al. 2020)
		small congestion[†]	average±SD: Cr (7.7±2.3); Cu (11.5±9.7); Fe (1472±800); Mn (59.7±10.5); Ni (4.7±1.1); Pb (5.8±2.9); Zn (51.8±23.5); *n* = 26		
		parks and green areas	average±SD: Cr (7.4±1.8); Cu (10.1±8.0); Pb (3.7±1.3); Fe (1387±451); Mn (48.5±20.2); Ni (3.8±1.8); Zn (41.8±11.9); *n* = 11		
Katowice (SL)	Aug 2018	sunny days; particles < 2 mm	average: Al (22370); As (109); Ba (162); Cr (106); Mn (1619); Ni (34.0); Cu (175); Rb (9.0); Zn (2683); *n* = 8	ICP-MS	(Rybak et al. 2020)
Kraków (MA)	Nov 2014 - Jan 2015, Nov 2015 - Jan 2016	city center	average±SD: Al (6400±249); As (21.0±7.5); Ba (89.0±1.6); Be (0.5±0.1); Cd (1.2±0.3); Co (2.1±0.6); Cr (112±0.3); Cu (185±8.2); Fe (36200±228); Ga (3.7±1.0); Li (22.7±0.3); Mn (644±4.8); Ni (23.1±0.9); Pb (88.0±1.9); Rb (6.4±0.9); Sn (14.0±2.1); Sr (112±1.0); Ti (252±3.8); V (36±0.2); Y (1.8±0.1); Zn (10630±36.7); Zr (4.7±0.2); *n* = 8	ICP-MS	(Gruszecka--Kosowska, Wdowin 2016)
Lublin (LU)	Feb - Mar 2013	particles < 200 µm	average±SD: Co (5.5±0.4); Cr (52.8±9.0); Cu (65.7±28.5); Hg (0.03±0.02); Mn (379±23.7); Mo (3.8±0.5); Ni (26.8±4.8); Pb (23.3±8.9); V (7.9±4.9); Zn (202±64.7); *n* = 49	EDXRF, $AAS_{(Hg)}$	(Kiebała et al. 2015)
Lublin (LU)	Apr 2013	particles 63-200 µm	average±SD: Cd (5.1±1.7); Cr (84.4±23.3); Cu (60.1±69.2); Ni (16.2±3.9); Pb (43.5±16.4); Zn (239±94.6); *n* = 68	EDXRF	(Zgłobicki et al. 2019, Zgłobicki, Telecka 2021)
		particles < 63 µm	average±SD: Cd (6.3±1.5); Cr (109±18.6); Cu (115±60.5); Ni (21.4±4.2); Pb (62.0±21.8); Zn (364±118); *n* = 68		
	Apr 2018	particles 63-200 µm	average±SD: Cd (3.8±0.6); Cr (120±32.1); Cu (57.7±37.0); Ni (11.9±3.2); Pb (25.4±8.0); Zn (178±82.5); *n* = 68		
		particles < 63 µm	average±SD: Cd (5.5±0.9); Cr (112±43.8); Cu (121±82.3); Ni (17.1±3.6); Pb (46.6±14.1); Zn (296±104); *n* = 69		

cont. Table 4

Radom (MZ)	May - June 2015	high housing density; particles ≤ 63 μm	average±SD: Cr (53.5±7.9); Cu (239±54.0); Fe (20600±1700); Mn (565±63.0); Ni (50.0±16.0); Pb (88.0±15.0); Zn (618±79.0); *n* = 8	ICP-MS	(Trojanowska, Świetlik 2020)
Warsaw (MZ)	2000		average±SD: Cd (0.7±0.3); Cu (30.6±20.5); Fe (600±140); Mn (134±64.4); Ni (10.3±3.8); Pb (33.9±21.7); Zn (63.6±36.1); *n* = 42	AAS	(Kozanecka et al. 2003)
Warsaw (MZ)	summer, 2013, 2014		median: Cd (0.2); Co (1.8); Cr (20.3); Cu (184); Fe (6801); Mn (110); Ni (17.0); Pb (17.1); Zn (150); *n* = 23	ICP-MS	(Dytłow, Górka-Kostrubiec 2021)
Wrocław (DS)	Aug 2018	sunny days; particles < 2 mm	average: Al (5133); As (3.0); Ba (84.0); Cr (77.0); Cu (126); Mn (258); Ni (50.0); Rb (10.0); Zn (153); *n* = 9	ICP-MS	(Rybak et al. 2020)

[#] large congestion – 13,680-30,720 vehicles day^{-1}, [†] small congestion – 4,800-18,000 vehicles day^{-1}, FAAS – flame atomic absorption spectrometry, ICP-MS – inductively coupled plasma mass spectrometry, EDXRF – energy dispersive X-ray fluorescence, AAS – atomic absorption spectrometry

respectively. A study performed in Łódź in 2011-2013 (Sakson et al. 2018) showed that the concentrations of Cd, Cu, Pb, and Zn (median of event mean concentration) in outlets from separate storm sewers were 0.5, 60.0, 15.0, and 320 μg L^{-1}, and in combined sewer 0.2, 84.0, 35.0, and 345 μg L^{-1}, respectively.

PTEs that enter sewage systems through various routes end up in sewage sludge biomass after a number of wastewater treatment processes (Hubeny et al. 2021). According to Hubeny et al. (2021), the concentrations of As, Co, Cr, Ni, Pb, and Zn in raw wastewater from a wastewater treatment plant (WWTP) in Warmia and Mazury amounted to 2.0-4.0, <6.0, 9.0-13.0, <9.0, <15.0, and 247-249 μg L^{-1} and those from a WWTP of Silesia were 1.0-4.0, <6.0, 6.0-17.0, 11.0-17.0, 15.6-21.0, and 209-299 μg L^{-1}, respectively. These data show that the samples from a WWTP located in an industrialized region (Silesia) were characterized by slightly higher PTE contamination than those collected from a WWTP located in a less industrialized region (Warmia and Mazury). Insufficiently treated wastewater carries some of these pollutants to urban rivers.

Waste and trash

Sewage sludge is an inevitable byproduct of municipal wastewater treatment (Latosińska et al. 2021). High contents of PTEs in sewage sludge are mainly the result of the share of industrial wastewater (e.g., tanning, lacquering, metallurgy) in the total mass of municipal wastewater. In addition, heavy metals come from domestic sewage, surface runoff, and corroded pipes (Gawdzik 2012). Processes commonly used in wastewater treatment

plants do not guarantee the removal of PTEs from sewage sludge (Tytła 2019). The choice of a sewage sludge management method depends on the sludge quantity and properties (Latosińska et al. 2021). For a sewage treatment plant operator, landfilling (storage within the ground or on the surface, e.g. on a landfill site), mostly in a pre-stabilized, low-mineralized form, with high organic content, is one of the most cost-effective and simple methods of sewage sludge management (Przydatek, Wota 2020). This method was legal in Poland until 1st January 2016. Since 2016, landfilling of unprocessed sewage sludge (sewage sludge with gross calorific value exceeding 6 MJ kg^{-1} of dry matter) has been prohibited (ME-PL 2015). Despite this, 6.8% of the total sewage sludge generated in 2022 was still landfilled, and sewage sludge accumulated on landfills by wastewater treatment plants reached 6,120 tons of dry mass at end of that year (GUS 2023*b*). PTEs occur in sewage sludge in a variety of forms, some of which are mobile and can infiltrate from the sludge into the ground environment, including watercourses (Gawdzik 2012).

A similar problem is generated by municipal solid waste and its landfill sites. In Poland, 39.1% municipal solid waste collected in 2022 was designated for landfilling (GUS 2023*b*). Municipal solid waste is a mixture of different organic and inorganic materials, which may contain a certain portion of PTEs (Tałałaj 2015). High PTE concentrations can be found in food waste (Cu, Cr, Pb, Zn), plastics (Cd, Cu, Pb, Ni, Zn), coal cinders, glass (Cd, Cr, Ni, Zn), dust (Cu, Cr, Ni), and textiles (Cu, Pb, Ni). The estimated ranges of Cd, Cr, Cu, Hg, Pb, and Zn in municipal solid waste in Poland were 0.5-4.0; 3.0-200; 24.0-156; 0.03-1.4; 8.0-209; 47.0-1,073 mg kg^{-1}, respectively (Tałałaj 2015). Used vehicle tires and their storage represent another problem. No relevant data were found in the Polish literature, but according to Shakya et al. (2006), the contents of Cd, Cr, Fe, Pb, and Zn in tires range within 0.02-27.1, 0.14-1.18, 17.8-381, 0.96-458 μg g^{-1}, and 3.95-8.21 mg g^{-1} respectively, depending on the tire type and quality.

As a result of waste materials decomposing and rainwater percolating through waste layers, leachate containing PTEs can be transported from the landfill to the environment, including groundwater and watercourses. Leachate is produced during the exploitation of the landfill and also after its closure (Gworek et al. 2016). Although new landfill facilities are being built according to strict rules that guarantee the least possible environmental impact (ME-PL 2013), a significant problem is posed by old landfills that were built in the 1960s, without proper ground preparation, often in mineral workings or natural depressions (Porowska 2019). Illegal landfills in an urban area are rather unlikely but may occur in nearby forests (Bielinis et al. 2022).

Incineration of sewage sludge and wastes is an important way for their management, which results in the generation of increasing quantities of ashes. In recent years, the amount of sludge being thermally treated has been increasing in particular (GUS 2023*b*). Sewage sludge ash (SSA) and munici-

pal solid waste ash (MSWA), which are also categorized as waste, may contain PTEs and require further management (Smol et al. 2020). Despite their potential as a secondary raw material, they are still largely landfilled (GUS 2023*b*). The risk of PTE leaching from stored waste ashes into the environment should also be considered. Table 5 shows examples of PTE levels in leachates from various wastes. According to Kasina et al. (2021), the leachability of elements is strongly dependent on the mineral composition of the waste product (the concentration and composition of soluble phase), the chemical composition (the mobility of hazardous elements and their affinity to soluble minerals) and the pH of the material. Even though the leachability of many PTEs is low (Table 5), and their concentrations in the leachates enable the classification of waste as inert, they may drastically influence the surface water quality.

Table 5

Levels of PTEs in leachates from various wastes in Poland – selected examples

Leachates from	Location	Unit	Element and concentration range	Analytical technique	References
SS-MF	MWTP, Śląskie	(mg kg^{-1})	Cd (<QL-3.9±0.4); Cr (<QL-1.9±0.2); Cu (<QL-9.4±0.3); Hg (<QL); Ni (1.5±0.1-41.6±2.4); Pb (<QL); Zn (193±18.0-1259±58.0); n = 3	ICP-OES; CV-$AAS_{(Hg)}$	(Tytła 2020)
MWL	Dolnośląskie	(μg L^{-1})	Cd (0.5-13.0); Cr (0.5-590); Cu (20.0-6670); Fe (19.0-38730); Mn (20.0-14480); Ni (0.5-530); Pb (0.5-440); Zn (123-4820); n = 7	GFAAS	(Wdowczyk, Szymańska-Pulikowska 2020)
MWL	Podlaskie	(μg L^{-1})	Cd (7.0-23.0); Cr^{VI} (114-452); Cu (47.0-114); Hg (1.0-7.0); Pb (45.0-740); Zn (259-442); n = 27	ICP-OES; $AAS_{(Hg)}$	(Tałałaj et al. 2016)
MSSA	no region information	(μg L^{-1})	Al (107-3024); As (2.0-13.6); Cd (0.04-1.2); Co (0.1-0.2); Cr (0.2-0.3); Cu (0.2-2.6); Hg (0.1-2.0); Mn (5.6-26.0); Mo (2.0-952); Ni (0.3-1.1); Pb (0.4); Se (<20.0); Ti (0.01-0.6); Zn (<2.0); n = n.s.	ICP-MS	(Kępys et al. 2021)
MSSA	no region information	(μg L^{-1})	Ag (20.0-30.0); Al (555-1780); As (10.0-20.0); Ba (70.0-170); Be (10.0); Bi (20.0-40.0); Cd (1.0-2.0); Ce (0-2.0); Co (0-0.1); Cr (40.0-70.0); Cs (3.0-6.0); Cu (20.0-30.0); Dy (0.1); Eu (0.1); Er (0.1); Fe (800-1000); Ga (3.0-5.0); Gd (0.1); Hf (0.1); Hg (20.0-30.0); Ho (0.1); La (0.1-0.4); Lu (0.1); Mn (10.0); Mo (690-980); Nb (0.4-0.7); Nd (0.2-3.9); Ni (20.0-40.0); Pb (0-120); Pr (0.1); Rb (70.0-120); Sb (10.0-30.0); Sc (80.0-100); Se (90.0-290); Sm (0.1); Sn (10.0); Sr (409-840); Ta (0.1); Tb (0.1); Th (0.1); Ti (10.0); Tl (0.5-1.6); Tm (0.1); U (0.1-0.2); V (180-460); W (90.0-140); Y (0.2-0.4); Yb (0.1); Zn (40.0-210); Zr (1.0-3.0); n = 5	ICP-MS	(Kasina et al. 2021)

cont. Table 5

MSWA	no region information	(μg L^{-1})	Ag (40.0-800); Al (400-74700); As (10.0-100); Ba (50.0-6500); Be (20.0-400); Bi (60.0-1000); Cd (2.0-40.0); Ce (1.0-25.0); Co (1.0-20.0); Cr (100-5020); Cs (1.0-360); Cu (50.0-800); Dy (0.2-4.0); Er (0.2-4.0); Eu (0.2-4.0); Fe (2000-404000); Ga (2.0-51.0); Gd (0.2-4.0); Hf (0.2-4.0).0; Hg (40.0-800); Ho (0.2-4.0); La (0.2-5.2); Lu (0.2-4.0); Mn (20.0-400); Mo (40.0-480); Nb (1.0-20.0); Nd (0.2-33.3); Ni (60.0-1000); Pb (20.0-61500); Pr (0.2-4.0); Rb (30.0-7180); Sb (0-40.0); Sc (200-4000); Se (40.0-800); Sm (0-2.0); Sn (20.0-400); Sr (460-11400); Ta (0.2-4.0); Tb (0.2-4.0); Th (0.2-4.0); Ti (20.0-0,400); Tl (0.2-39.7); Tm (0.2-4.0); U (0.2-4.0); V (20.0-400); W (10.0-80.0); Y (0.6-10.0); Yb (0.2-4.0); Zn (100-11400); Zr (0-280); n = 11		

SS-MF – sewage sludge-mobile fraction, MWTP – Municipal Wastewater Treatment Plants, MWL – municipal waste landfill, MSSA – municipal sewage sludge ash, MSWA – municipal solid waste ash, <QL – below limit of quantification, ICP-OES – inductively coupled plasma optical spectrometry, CV-AAS – cold vapor atomic absorption spectrometry, GFAAS – graphite furnace atomic absorption spectrometry, AAS – atomic absorption spectrometry, ICP-MS – inductively coupled plasma mass spectrometry, n.s. – not specified

FLOW CHANNELS

The main pathways by which PTEs are transported from urban areas to forest ecosystems are:

- air channel – moving air currents (wind) and deposition (dry or wet) of dust, aerosol sized particles and gaseous forms of metals, e.g. Hg (Alloway 2013);
- water channel – direct runoff from urban areas during heavy rainfall and watercourses (rivers and streams) and their floodings (Alloway 2013);
- human-related channel – direct input from people as a result of leisure-related or other activities (Malinowska, Szumacher 2013).

Air channel

The air channel is the most important and best quantitatively studied pathway by which PTEs enter forest ecosystems. According to current knowledge, PTEs in fine dust emitted or formed in the atmosphere can be transported over long distances – hundreds or even thousands of kilometers from the source of emission (GIOS 2021*a*), particularly Hg (Wilkening et al. 2000). Movement of PTEs depends on the circulation of air masses and the speed and direction of the wind (Janusz, Nadziakiewicz 2002). Apart from these, there are other factors which influence the distribution and movement of the

pollutants, such as the partition coefficient, polarity, vapor pressure and molecular stability (Briffa et al. 2020). The forest canopy acts as a barrier and receptor for various air-borne pollutants (Szopka et al. 2013), and it captures a significant portion of them (Małek, Wężyk 2000). The amount and nature of pollutants received/retained by a forest is influenced not only by the amount of emissions from the source, but also by the forest ecosystem itself, including the forest's size, and the type and distribution of vegetation (especially trees) (Manecki et al. 1984). Air flow is twice as strong above as inside the forest and hence air pollutants accumulate more at the top of the forest (Reczyńska-Dutka et al. 1984). The PTEs that are transported via air are retained in the forest ecosystem primarily through the process of atmospheric deposition. It can take the form of wet or dry deposition. Either falling by gravity or with precipitation, PTEs can end up directly on the forest floor and in the soil, or they are first retained on plant surfaces (mainly trees and shrubs) and only later are washed to the floor by rain or melting snow (IBL 2021). Some of the PTEs settled on leaves may be absorbed by plants. Some may also remain suspended in the forest air (Mirosławski et al. 1992). Forest causes a change in wind velocity, which affects the fall of pollutants, which is greater at the eastern and western margins of the forest than in the center (Reczyńska-Dutka et al. 1984). The relationship between the concentration of PTEs and the distance from the emission source has not been proven, although a decrease in the fall of PTEs, especially Cu, Cd, and Ni, was observed in the forest center.

Transport and input of PTEs through the air channel were studied and quantitatively assessed in 1976-1980 in the Niepołomice Forest complex (11,000 hectares), which is situated in Southern Poland, 20 km east of Kraków (Manecki et al. 1984). At that time, the forest was also under strong pressure from local industrial plants (mainly the Lenin Steelworks at that time) and more distant ones (the Upper Silesia industrial region). It was determined in the above study that during an average year every square kilometer of the Niepołomice Forest received about 3000, 125, 31, 18, 6, and 1.5 kg of Fe, Zn, Pb, Cu, Ni, and Cd, respectively. About 45% of this amount represented soluble components, posing the gravest danger to living organisms.

In 1999, the input of some PTEs into a mountain forest ecosystem of the Dupniański stream catchment (Istebna village) in the Silesian Beskid region was determined (Małek, Wężyk 2000). Polluted air moves into the Silesian Beskid from Katowice and Kraków (~100-120 km from Istebna), as well as from Ostrava (the Czech Republic, ~60 km from Istebna), which results in excessive load of these PTEs with atmospheric precipitations. Annual bulk deposition (combined dry and wet) of Al, Cd, Cr, Cu, Fe, Mn, Pb, and Zn was 35.3, 2.07, 3.56, 15.0, 89.7, 14.3, 21.5, and 285 mg m^{-2}, respectively. It was also shown that while rain is the dominant source of pollution, snow during thaw can introduce quite large amounts of PTEs into ecosystems in a short

period of time, affecting not only vegetation but also the quality of water in the stream.

Since 1994, studies on the chemical composition of atmospheric precipitation and the amount of some PTEs (Al, Cd, Cu, Fe, Mn, Pb, Zn) deposited with them on Polish forest areas have been carried out as part of intensive monitoring of forest health condition (http://www.gios.gov.pl/monlas/monitoring.html). The national forest monitoring program is a part of the International Co-operative Programme on Assessment and Monitoring of Air Pollution Effects on Forests (ICP-Forests) – Kowalska et al. (2016). Twelve measurement points/plots have been designated across the country in different forest ecosystems (districts) – IBL (2021). These points/plots are located at relatively short distances from urban areas of different sizes (Table 6). PTE deposits provided with precipitation in open space and throughfall and stemflow deposition have been assessed. Precipitation reaching soils under tree canopies differs from atmospheric precipitation as the tree crown layer participates in processes that lead to the enrichment (e.g. leaching from plant tissues) or depletion (e.g., element sorption in tree crowns) of compounds in precipitation (Kowalska et al. 2016).

Information on forest area loadings of As, Cd, Hg, and Ni, deposited from the air has been provided by assessment reports performed at national regional background stations that operate under the State Environmental Monitoring (Table 7). The aforementioned stations are situated in forest complexes in close proximity to small towns, in areas that have experienced relatively low levels of human impact. The first such report was released in 2012 with data for 2010 and 2011, and subsequent reports were published periodically on an annual basis. Data from Tables 6 and 7 mostly show a tendency for PTE loads to decrease at the measurement points, but there is little optimism about Mn, Ni and As, whose deposition at some locations has increased in the last decade.

Apart from the aforementioned national monitoring programs, scientific studies on the qualitative and quantitative assessment of PTEs entering forest ecosystems through the air channel have not been very frequent in Poland. Atmospheric bulk deposition of As, Cd, Cr, Cu, Ni, Pb, and Zn in the forest Jeziory, situated about 30 km southwest of the Poznań Agglomeration, was studied between April 2013 and October 2014 (Siudek, Frankowski 2017). The mean bulk deposition fluxes of these elements found in the cited study were 465, 7.3, 132, 1904, 19.2, 161, and 6655 μg m^{-2}, respectively. Higher deposition of trace elements was observed in winter. The vast majority of PTEs tested showed higher concentrations in mixed precipitation samples, followed by rain and snow. A large part of the PTEs deposited in Jeziory was attributed to the emissions resulting from various industrial processes/heating of dwellings in the Poznań Agglomeration, but also driven by wind from distant regions of Europe (polluted regions of the Czech Republic, Germany, France).

Table 6

Annual deposit of PTEs provided with precipitation in open space and throughfall deposition in the points of national intensive monitoring of forests in 2010 and 2020 (mg m^{-2}), based on IBL (2021)

Forest district	Location (potential urban impact source)	Year	Al	Cd	Cu	Fe	Mn	Pb	Zn
Deposit with precipitation in open space									
Białowieża (7/1#, PD)	10 km from a small town (Hajnówka)	2010	7.2	0.2	8.7	5.6	2.4	0.8	29.0
		2020	3.1	0.1	1.8	3.1	4.9	0.6	11.2
Bircza (108/4, PK)	dense forest complex, low population density	2010	11.0	0.2	10.5	8.8	6.9	1.6	33.8
		2020	3.1	0.1	2.1	3.8	4.8	1.3	16.2
Chojnów (1/17, MZ)	25 km from the center of a large urban agglomeration (Warsaw)	2010	6.4	0.1	7.9	6.8	3.7	1.3	32.3
		2020	2.2	0.1	2.4	3.3	2.7	0.6	11.3
Gdańsk (71/15, PM)	10 km of a large urban agglomeration (Gdańsk-Gdynia-Sopot)	2010	9.5	0.7	8.2	10.8	10.3	1.7	37.9
		2020	3.0	0.1	1.5	3.5	5.8	0.8	10.9
Krotoszyn (44/9, WP)	11 km of a medium-sized city (Krotoszyn)	2010	7.2	0.1	7.1	6.0	4.6	1.6	25.7
		2020	2.3	0.0	1.5	2.0	4.4	0.7	9.6
Krucz (60/8, WP)	50 km of a large city (Poznań)	2010	5.2	0.1	5.9	4.9	6.4	1.1	27.6
		2020	1.6	0.1	1.0	1.9	4.1	0.7	08.4
Łąck (53/6, MZ)	6 km from a medium-sized city (Płock)	2010	8.0	0.1	8.7	7.8	15.1	1.3	29.7
		2020	2.2	0.0	1.9	2.6	12.5	0.8	11.5
Piwniczna† (40/8, LP)	dense forest complex, low population density	2020	2.9	0.1	1.6	3.3	4.9	1.3	12.9
Strzałowo (83/7, WM)	approx. 20-30 km from the largest tourist towns (Mrągowo, Ruciane Nida)	2010	4.9	0.1	7.3	6.9	2.9	0.9	28.0
		2020	3.9	0.1	1.3	3.1	6.3	1.0	9.6
Suwałki (105/1, PD)	20 km of a medium-sized city (Suwałki)	2010	9.0	0.1	7.5	7.8	5.1	1.0	24.8
		2020	2.0	0.1	1.3	2.1	2.9	1.1	9.3
Szklarska Poręba (DS)	35 km E of a brown coal mine and fossil-fuel power station	2010	14.7	0.3	13.7	14.1	4.0	3.2	58.8
		2020	4.6	0.1	2.6	6.2	3.4	2.3	21.4
Throughfall deposition (+stemflow)									
Range for all plots and tree stands		2020	5.7(+0.7) - 24.2	0.0 - 0.1	1.5(+0.1) - 4.0	4.9(+0.8) - 17.9	18.9 - 69.0	0.5 - 2.4	8.0(+0.7) - 19.0

point locations available at: http://www.gios.gov.pl/monlas/images/spo_03.png, † data only for 2020

Table 7

Annual deposition of PTEs at regional background stations in 2010 (2011) and 2020 (μg m^{-2}), calculated based on (GIOS 2021*a*, *b*)

Station	Location[#]	Year	As	Cd	Hg	Ni
Diabla Góra (WM), PL0005R[†]	Borecka Forest (Puszcza Borecka), ~20 km from a medium-size tourist town (Giżycko)	2010	173.4	25.2	7.488	141.3
		2020	150.1	14.6	6.645	289.1
Osieczów (DS), PL0505A	Lower Silesian Forest (Bory Dolnośląskie), ~10 km from a small town (Bolesławiec)	2010	674.9	55.1	2.369	610.3
		2020	838.1	29.3	1.610	874.7
Zielonka (KP), PL0077A	Tuchola Forest (Bory Tucholskie), ~10 km from a small town (Tuchola)	2011	87.6	15.0	16.474	225.9
		2020	102.5	11.0	6.291	208.6

[#] station location available at: https://powietrze.gios.gov.pl/pjp/maps/measuringstation?lang=en&woj= (station type: Specialized EU regional background stations), [†] international code of the station

In 1998-2000, the inputs of selected PTEs to the forest floor due to urban pollution from Kraków were estimated based on litter fall and its chemical composition (Sawicka-Kapusta et al. 2003). Small forests located at different distances from the Kraków conurbation were included in the study. The annual input of Cd, Cu, Fe, Pb, and Zn was 84-382, 778-2768, 22-263, 406-3717, and 0-38 μg m^{-2}, respectively. Input of metals to the forest floor tended to decrease with increasing distance from the center of Kraków, and the higher values was observed in the eastern transect, close to a steel works.

As can be seen from the above review, the studies on the input of PTEs into forest ecosystems have focused on few elements, especially those considered most hazardous. However, the list of PTEs determined by Kotowski et al. (2020) in precipitation in Kraków included as many as 36 elements (Ag, Al, As, B, Ba, Be, Bi, Br, Cd, Co, Cr, Cs, Cu, Ga, Fe, Hg, I, Li, Mn, Mo, Ni, Pb, Sb, Se, Sn, Sr, Rb, Te, Ti, Tl, U, V, W, Y, Zn, Zr). Potentially, all these elements may reach forest ecosystems via air channel.

Water channel

Urban areas are to a great extent covered with impervious surfaces such as roofs, streets, sidewalks, and parking lots (Eriksson et al. 2007). During rainfall, stormwater flows over these surfaces into urban watercourses (streams and rivers) or is directed into storm drains and then discharged into watercourses. It transports PTEs from street dust and PTEs from wet deposition in the urban area (Bąk et al. 2012). Landfill leachates can enhance this load (Czarnocki, Paluszkiewicz 2014). Urban rivers often receive inadequately treated or accidentally discharged wastewater containing PTEs (Hubeny et al. 2021). Intense precipitation or post-winter snowmelt can result in drainage system inefficiencies, allowing surface runoff to flow directly into nearby forests, where it can infiltrate the soil or enter forest water bodies (Eriksson et al. 2007, Górska et al. 2013). The waters of streams and

rivers flowing out of urban areas can bring PTEs into the forests through which they flow. Such PTE-polluted water (PTEs in dissolved fractions and suspended matter) can be taken up by riparian plants of the forest ecosystem. It is also used by forest animals. River floods in forest ecosystems can also input PTEs transported from cities into forest soils (Górska et al. 2013).

Although this channel of PTEs flow from urban areas to forest ecosystems is self-explanatory, information regarding its quantitative estimation has not been found in available Polish scientific sources. The potential of this flow channel can be inferred indirectly from the amount of PTEs in municipal wastewater, stormwater, and landfill leachates (see section **Urban watercourses**). Hubeny et al. (2021) reported that the evacuation of treated wastewater did not increase As, Co, Cr, Ni, Pb, and Zn concentrations in river water in sites downstream from the effluent discharge point relative to sites upstream from the effluent discharge point both in the Warmia and Mazury Region and in Silesia Region. However, the cited authors emphasize that treated wastewater released into the natural environment can pose a continuous threat of its contamination with PTEs.

Human-related channel

This channel includes both unintentional and intentional transmission of PTEs into the forest. PTEs are unintentionally transported along with street dust deposited on the shoes and wheels of bicycles and baby carriages of people resting in the forest, as well as on the fur of animals accompanying people. These amounts are rather small, but they depend on the intensity of recreational traffic. However, no relevant quantitative estimates were found. The highest recreational traffic rate is observed in the vicinity of water reservoirs; it also changes periodically depending on the season and occurrence of fruits of the forest ground cover (LP-NO 2015). A bigger problem may be generated by cars entering or parking on forest edges. Moreover, various forms of recreation in forest areas are unfortunately accompanied by people leaving there various types of trash, most often food and drink packaging. For example, a single used beverage can (31.6 g) represents 27.7, 0.726, 1.07, 0.054, 0.31, and 0.057 g of Al, Fe, Mg, Ni, Sn, and Zn, respectively (Rabah 2003).

Illegal dumping in forested environments is a separate issue, and dumps usually occur near urban areas (Bielinis et al. 2022). According to the Central Statistical Office of Poland, as of the end of 2022 there were 2,217 illegal dumping sites in Poland (GUS 2023*a*). In forest wild dumps, a variety of waste materials can be found, including debris and ceramic waste, glass, plastics, metals, textiles, tires, furniture, used electrical and electronic equipment, organic waste from households, as well as grass cuttings and garden residues (Breza-Boruta et al. 2016). The release of pollutants (including PTEs) from such an 'open dump' is easy and uncontrolled. As a result of rainwater infiltration, they can migrate from the dumped waste down

the soil profile and contaminate soil, surface waters, and groundwaters. The studies by Bartkowiak and Lemanowicz (2014) and Breza-Boruta et al. (2016) show that the contents of Cd, Cr, Cu, Ni, Pb, and Zn in soil under wild dumps in forests were elevated, but did not exceed acceptable limits. This situation may worsen over time (if the dump is not detected and eliminated) as individual wastes gradually decompose (Król 2016). Importantly, illegal dumps can be also uncontrolled sources of Hg in the air and soil, and its concentrations in these media can locally surpass maximum allowable levels (Falkowska et al. 2013). Some examples of the content of PTEs in selected types of waste are presented in the section **Waste and trash**.

IMPACT

The retention of chemical pollutants, including PTEs, places urban and suburban forests at risk of the consequences of this function. Over time, the impact of urban emissions, particularly when combined with those from industries, results in the accumulation of PTEs in forest ecosystems. The forest soils are the main reservoir for such elements (Lasota et al. 2020), and there is a concern that they may act as an 'ecological time bomb' (Magiera et al. 2007). Large accumulations of PTEs can disrupt the homeostasis of the soil environment by reducing microbial biomass and activity, affecting the structure and diversity of soil microbial communities, the density and diversity of meso- and macrofauna, and the decomposition of organic matter (Sawicka-Kapusta et al. 2003, Chodak et al. 2013, Chrzan 2018, Borgulat et al. 2021). High concentrations of PTEs can even affect mycorrhizal fungi, commonly known as biota that mitigate metal toxicity in plants by sequestering large amounts of PTEs and other mechanisms (Rusinowski et al. 2019).

PTEs impede the nutrient cycling in forests by reducing the availability of essential nutrients or by hindering plants' ability to access and transport water (Wegiel et al. 2018). Very low soil pH usually favors the release of PTEs into the soil environment and, through the soil, directly into the forest plants and underground water system (Magiera et al. 2007). The uptake of PTEs by plants proceeds through the root system, but also through the leaf blades (Ociepa-Kubicka, Ociepa 2012). Plants have developed a series of defense strategies to cope with the toxicity of harmful elements (Riyazuddin et al. 2022) and the toxic effects of PTEs on most plants often occur only when soil pollution is high (Ociepa-Kubicka and Ociepa 2012). Individual plant species, their ecotypes, or populations differ in their level of tolerance to the stress of excess PTEs in the environment (Riyazuddin et al. 2022). However, an increasing concentration of some PTEs in plant cells, tissues and organs causes molecular, structural and metabolic disorders at these levels of biological organization that result in morphological changes in most plants (Stroinski 2002, Siwek 2008). These changes are manifested by a reduction in the size and weight of the entire plant or its organs (Stroinski

2002). High concentrations of PTEs can be lethal to less tolerant plant species, leading to a reduction in species diversity and changes in the species structure of plant communities (Stefanowicz et al. 2020). The extreme effect can be the death of the entire forest flora, which basically means the destruction of the entire ecosystem (Tumanyan et al. 2020). In Poland, examples of forest stand die-back have already been recorded, probably caused by the interaction of the critical load of heavy metals in the soil and drought, which is an additional stress factor in heavily contaminated sites (Pająk, Pietrzykowski 2021).

Plants or their dead parts are a source of food for animals, through which heavy metals enter the higher levels of the food chains, with the added problem of the biomagnification process (Szynkowska et al. 2018). It is obvious that contact with PTEs for both animals and humans implies a serious risk. The most common route of entry for PTEs into human and animal organisms is through the digestive tract. However, they can also enter through inhalation or absorption through the skin (Ociepa-Kubicka, Ociepa 2012). It is well known that toxic elements cause serious impacts on human and animal health, including reduced growth and development, cancer, organ damage, nervous system damage, and in extreme cases, death (Briffa et al. 2020, Bartkowiak 2022). The health effects of regular consumption of products containing even small amounts of these elements can take many years to manifest themselves. This risk is especially true for people who regularly consume mushrooms and berries from urban and suburban forests (Strzyszcz, Rachwał 2012). From a clinical point of view, a very important problem is the high availability of PTEs that are inhaled by anyone being in the forest (Mirosławski et al. 1992). Greater absorption via this pathway is favored by the phenomenon of secondary dusting of a forest contaminated with PTEs (Musielińska et al. 2018).

SUMMARY AND CONCLUSIONS

The urban areas of Poland, in conjunction with the industrial complexes situated within their boundaries, represent a significant source of potentially toxic elements (PTEs). A multitude of anthropogenic activities, including domestic heating, heavy traffic, industrial emissions, municipal and industrial waste disposal, occurring within these areas results in the loading of PTEs on the environment. PTEs are partially retained within the urban ecosystem, but a certain portion is transported to adjacent ecosystems, including urban and suburban forests. PTEs present in airborne dust, dust deposited on various urban surfaces, and in various types of waste and trash are transported to these ecosystems through air, water, and human-related channels. Urban and suburban forests serve as important pollutant barriers and sinks, but they also perform a multitude of other ecosystem service functions.

Studies on PTEs in the environment have been documented in a large number of articles. However, they do not allow for a full qualitative and

quantitative assessment of PTEs entering urban and suburban forest ecosystems from urban areas via relevant channels. National monitoring, conducted at designated points around the country, focuses on regular quantitative studies of emissions, air pollution, and deposition of selected PTEs (Al, As, Cd, Cu, Fe, Hg, Mn, Ni, Pb and Zn) considered most dangerous to living organisms. Many other PTEs are surveyed less frequently and irregularly. The transfer of PTEs via water and human-related channels is not extensively addressed in the national literature.

Given the increasing awareness of the hazards posed by PTEs and of the multiple roles played by urban and suburban forests, as well as the improved analytical techniques and an expanded array of elements detected, it appears that the issue of PTE transfer from urban areas to forest ecosystems, as addressed in the article, has the chance for a more accurate and holistic approach in future studies.

Author contributions

M.J. – conceptualization, methodology, visualization, writing – original draft preparation, M.K.K. – conceptualization, methodology, writing – review & editing. All authors have read and agreed to the published version of the manuscript.

Conflicts of interest

The authors ensure that they have neither professional nor financial connections related to the manuscript sent to the Editorial Board. The funders had no role in the design of the study; in the collection, analyses, or interpretation of data; in the writing of the manuscript, or in the decision to publish the results.

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