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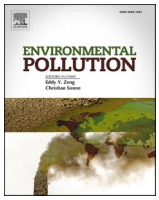

Review

Review on distribution, fate, and management of potentially toxic elements in incinerated medical wastes☆

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A B S T R A C T

Medical wastes include all solid and liquid wastes that are produced during the treatment, diagnosis, and immunisation of animals and humans. A significant proportion of medical waste is infectious, hazardous, radioactive, and contains potentially toxic elements (PTEs) (i.e., heavy metal (loids)). PTEs, including arsenic (As), cadmium (Cd), lead (Pb) and mercury (Hg), are mostly present in plastic, syringes, rubber, adhesive plaster, battery wastes of medical facilities in elemental form, as well as oxides, chlorides, and sulfates. Incineration and sterilisation are the most common technologies adopted for the safe management and disposal of medical wastes, which are primarily aimed at eliminating deadly pathogens. The ash materials derived from the incineration of hazardous medical wastes are generally disposed of in landfills after the solidification/stabilisation (S/S) process. In contrast, the ash materials derived from nonhazardous wastes are applied to the soil as a source of nutrients and soil amendment. The release of PTEs from medical waste ash material from landfill sites and soil application can result in ecotoxicity. The present study is a review paper that aims to critically review the dynamisms of PTEs in various environmental media after medical waste disposal, the environmental and health implications of their poor management, and the common misconceptions regarding medical waste.

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1. Introduction

Several analogous phrases have been applied in the published literature to define medical waste, such as 'health care waste' and 'hospital waste'. Medical waste poses a major global health care issue(WHO, 2018a). Hospitals and other healthcare-related activities and facilities, such as nursing homes, blood banks, autopsy and research centres, mortuaries, and laboratories, generate medical waste (Das et al., 2021). Approximately 75–85% of these medical wastes are regarded as nonhazardous, whereas 15–25% can be toxic, infectious or radioactive (Padmanabhan and Barik, 2019). The potential risks of medical wastes are diverse. They may contain infectious materials such as blood and tissues that can transmit diseases to working staff, other patients, and the public in general (Das et al., 2021; Borowy, 2020). Sharp objects, including needles and surgery knives, can cause injuries to people handling medical waste. Incineration of medical wastes can emit gaseous contaminants (dioxins, furans) and leave behind residual toxic elements in the environment (Rathna et al., 2018). Cytotoxic waste, including drugs used to treat cancer, rheumatoid arthritis, and multiple sclerosis, can exhibit teratogenic and/or carcinogenic and mutagenic properties. Radioactive waste derived mainly from cancer treatment can also impose short- and long-term health impacts (Miller et al., 2019).

Medical wastes contain a range of highly hazardous organic and inorganic compounds, including persistent organic pollutants (POPs) such as endocrine-disrupting chemicals (EDCs) (Rathna et al., 2018). Inorganic compounds of concern are mostly heavy metals and metalloids and are collectively termed 'potentially toxic elements' (PTEs) (Pourret and Hursthouse, 2019). The term 'Potentially toxic elements (PTEs)' in general includes metals and metalloids (metal (loids) that are both biologically essential [e.g., cobalt (Co), copper (Cu), chromium (Cr), manganese (Mn) and zinc (Zn)] and non-essential [e.g., Arsenic (As), cadmium (Cd), lead (Pb) and mercury (Hg)] elements. The essential elements (for the plant, animal or human nutrition) are required in low concentrations and hence are known as 'trace elements' or 'micro nutrients'. The non-essential elements are phytotoxic and/or zootoxic and are widely known as 'toxic elements'. Both groups are toxic to plants, animals and/or humans at extremely high concentrations (Bolan et al., 2014).

PTEs are mostly present in medical facilities' plastic, syringes, rubber, adhesive plaster, and battery wastes (Annapoorani, 2021). For example, the pigment of coloured plastics contains PTEs such as cadmium (Cd), cobalt (Co), chromium (Cr), lead (Pb), and copper (Cu) (Adama et al., 2016; Sridharan et al., 2022). Also, a larger quantity of Cr and zinc (Zn) in medical wastes come from syringes' disposal (Honest et al., 2020). PTEs in medical wastes typically occur in the form of metal (loid) elements, metal (loid) oxides, volatile metal (loid)ic sulfates and chlorides.

To maintain the occupational health and safety of health workers, patients, and the general public and reduce the environmental impacts from the unintentional exposure of biological and chemical hazards from medical wastes, including drug-resistant microorganisms (Zhou et al., 2022), safe and environmentally sound management approaches are needed. Medical wastes are generally treated through either sterilisation using autoclaves or incineration (Dharmaraj et al., 2021). Incineration is currently the most common practice and technology adopted for the safe management and disposal of medical waste. The incineration of medical waste primarily aims to eliminate deadly pathogens in the waste (Khobragade, 2019). However, incineration should also destroy bioactive organically-based pharmaceuticals. The incinerated ash materials from hazardous medical wastes are generally disposed of in landfills after the S/S process (Suryawan et al., 2019). The process of stabilisation is generally applied to reduce the toxicity and solubility of the potential pollutants, while in the process of solidification, a cementing material is applied to reduce the leachability of the contaminants by immobilising them (Bolan et al., 2022; Kumar et al., 2021; Bolan et al., 2014).

The incineration of medical wastes and the subsequent disposal of the incinerated ash affect the speciation, redistribution, mobility, and bioavailability of PTEs. For example, the incineration-induced oxidation of most PTEs, such as Pb and Cd, in medical wastes results in their precipitation as metal oxides, thereby decreasing their solubility and bioavailability (Liu et al., 2018). Oxidation of other PTEs, such as Cr and Sb, results in the alteration of their speciation (Cr to Cr(VI) and Sb to Sb (V)), thereby increasing their solubility, mobility and bioavailability (Bolan et al., 2022; Adama et al., 2016; Choppala et al., 2013). Several investigations have examined the distribution, leaching and plant uptake of PTEs derived from medical waste ash application and other sources to landfills and soils (Omeka et al., 2022; Kaur et al., 2022; Omeka and Egbueri, 2022; Kaur et al., 2019; Jamshidi-Zanjani et al., 2015; Zhao et al., 2010). The results indicated that the leaching and plant uptake varied amongst the PTEs and were primarily controlled by the oxidation state of the PTEs in the incinerated medical waste ash. For example, while the oxidised species of Cr [Cr(VI)] is more liable for leaching, that of As [As(V)] is less liable for leaching compared to their respective reduced species [Cr(III) and As(III)] (Mahimairaja et al., 2005; Choppala et al., 2013). Several reviews have examined the sources, treatment, and environmental and health impacts of medical wastes in general (Das et al., 2021; Chisholm et al., 2021; Singh et al., 2021; Torkashvand et al., 2020; Liu et al., 2018). However, there has not been a comprehensive review of the distribution, transformation, and bioavailability of PTEs derived from medical waste.

Therefore, the key objectives of the current review article are to comprehensively review: (i) PTEs sources in medical wastes, (ii) the management of medical wastes through incineration, (iii) the transformation of PTEs during incineration, (iv) the fate and environmental impacts of PTEs in the medical waste incinerated ash material, and (v) most common misconceptions encountered in the literature regarding PTEs in the medical waste. Specifically, we determine the (i) nature of medical wastes, (ii) sources of PTEs in medical wastes, (iii) transformation of PTEs during incineration of medical wastes, and (iv) environmental impacts of PTEs in medical waste ash material.

2. Methodology

Literature reviews play a critical role in understanding what has been reported on a subject or topic and identifying topics or questions requiring more investigation. There are a number of methods and techniques for synthesising quantitative (e.g., frequency analysis, meta-analysis) and qualitative (e.g., grounded theory, narrative analysis, meta-ethnography) evidence from literature data (Paré et al., 2015). While qualitative analysis is effective in undertaking narrative reviews, quantitative analysis is useful in undertaking a descriptive review. In this study, we undertook a quantitative analysis of literature data.

Relevant literature was searched through the databases of Web of Science, ScienceDirect, Scopus, Google Scholar, and other various sources by applying the keywords: 'Medical wastes', 'Infectious waste', 'Hazardous waste', 'radioactive waste', 'Incineration', 'Contaminants', 'Potentially toxic elements', 'Landfill', 'Leaching', 'Bioavailability', and 'Ecotoxicity'. This review aims to provide the current-state-of-the-art research on medical waste management and fill the knowledge gaps related to the environmental fate of PTEs in medical wastes.

This critical review will help develop a sustainable approach for handling PTE pollution of the environment caused by medical waste. This information is crucial for the development of strategies that limit the environmental toxicity of medical-waste-derived PTEs to humans and the ecosystem. This review provides a framework for identifying priority research areas to address the existing knowledge gaps about medical waste and its effective management. Furthermore, improved knowledge about medical-waste-derived PTE dynamics in liquid and solid matrices will benefit the medical sectors and industries by enhancing public awareness about medical waste disposal. Future research priorities in the sustainable management of contamination

resulting from medical wastes in these settings are also proposed.

3. Overview and toxic elements of medical waste

3.1. Characteristics of medical waste

The waste generated in hospitals, clinics, and health care institutions or waste streams contaminated by medical procedures that produce human secretion or human contact waste, including blood, body tissue and fluids, is considered medical waste. Medical waste can form part of the infectious pathways to animals and humans. A significant component of medical waste can be biologically toxic and corrosive in nature, and when improperly handled, it causes air, water, and soil pollution. It can also cause direct harm to humans through exposure or the release of toxins (Singh et al., 2021; Zhang et al., 2021). For example, Ephraim et al. (2013) examined the effect of burnt hospital wastes on the agricultural soil around a farm settlement near Obafemi Awolowo University Teaching Hospital Complex, Ile-Ife, South-West Nigeria. The presence of elevated concentrations of Pb (77.8–279.6 mg kg^{-1}) and Cl (102.2–167.2 mg kg^{-1}) was of serious health concern because of the intensive agricultural practices for food production in the neighbourhoods of the study sites. The composition of medical waste is a mixture of diverse types of medical discards, which varies greatly in different medical settings and locations of origin (Rani and Rampal, 2019). The quantity of medical waste in different hospitals and countries is very much correlated to socioeconomic factors such as the human development index, life expectancy, and health care expenditure per capita of the region and countries (Minoglou et al., 2017; Singh et al., 2021).

Technological advancements in social networks, transportation, and occupation have fuelled global economic growth resulting in the expansion of health care systems and a concomitant increase in demand for medical equipment and supplies. Thus, an increased standard of living in many countries has also played a key role in medical waste generation (Bloom et al., 2018; Singh et al., 2021). Fig. 1b shows the features of medical waste and the weight of medical waste generation (kg bed^{-1} day^{-1}) worldwide and reveals that medical waste generated by high-income countries is significantly greater than that of low- and middle-income nations. However, the environmental impact of medical

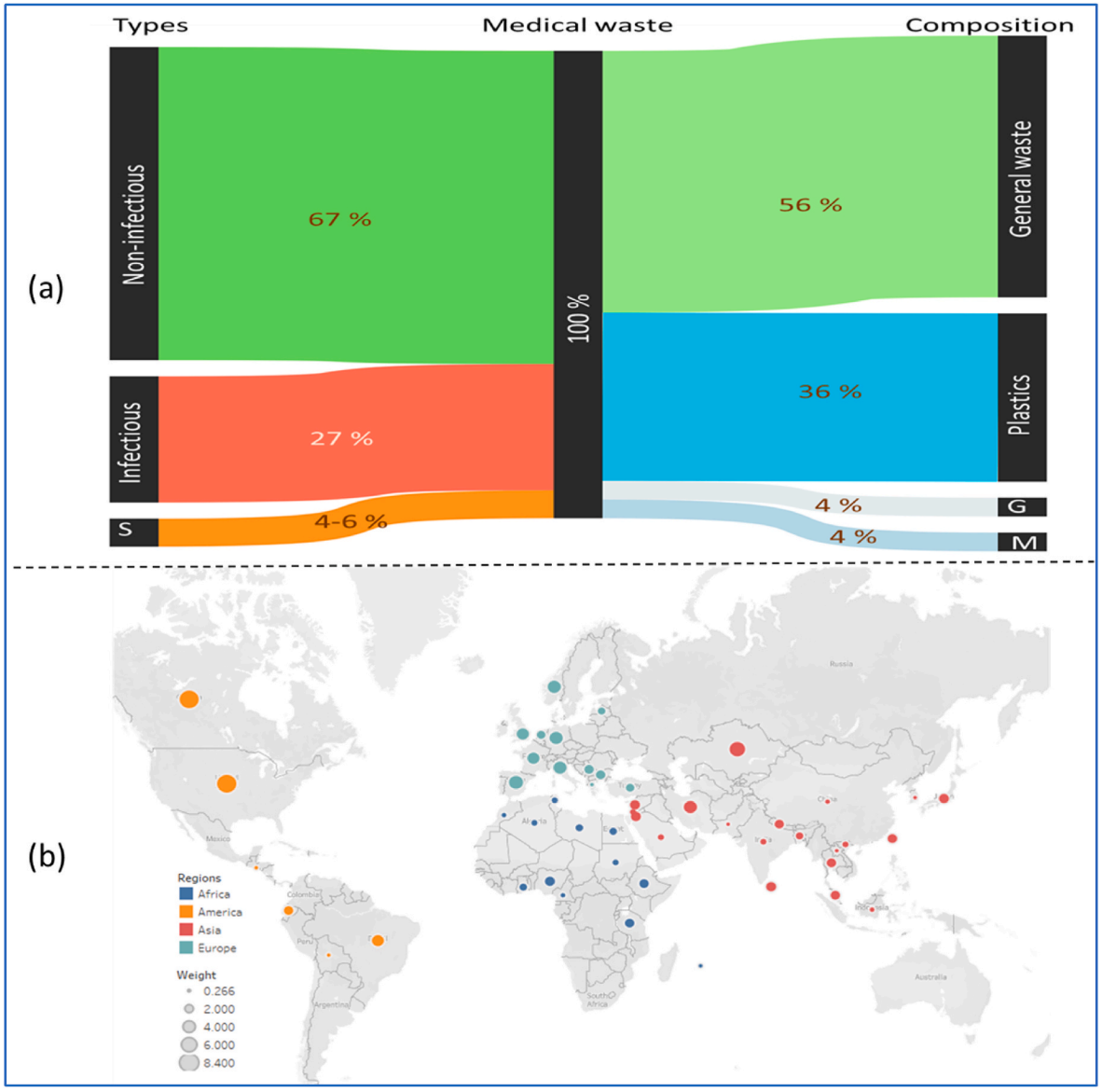

Fig. 1. Characteristics of medical waste and the weight of medical waste generated in different countries. (a) An average percent share of medical waste types and composition; (b) an average amount of medical waste generated in different countries (kg bed-1 day-1); S = sharps, G = glass, M = metals (Taghipour and Mosaferi, 2009).

waste may be worse in developing countries despite producing less waste, which is attributed to poor management of the generated waste (EPI, 2020; Singh et al., 2021).

Typically, medical waste is classified by the nature of the contamination, namely infectious waste, pathological waste, sharps waste, chemical waste, pharmaceutical waste, cytotoxic waste, radioactive waste, and general waste. Nonetheless, medical waste is categorically divided into three major types that include: 1) infectious or hazardous waste - waste containing pathological germs and toxins, 2) non-infectious or general waste - a mixture of discarded food and non-contaminated materials, and 3) sharps waste - waste containing metal and glass (Fig. 1a) (WHO, 2018b).

Countries vary significantly in disposal practices, and many countries have not established safe disposal practices. Medical waste in many municipalities is combined with general waste. As such, it is usually treated by open landfill relocation or incineration (Singh et al., 2020a; Singh et al., 2020b). A recent study based on the meta-analysis of medical and health care waste management practices in 78 countries has revealed that about 56% of medical waste contained general waste such as food, liquids, and paper (Singh et al., 2021). Plastic content is the second most abundant material, with about 36% of the total medical waste. Glass and metal waste contribute 4% each of the total waste generated globally (Singh et al., 2021).

3.2. Sources of potentially toxic elements and other pollutants

Medical waste disposal releases PTEs and other secondary pollutants, such as particulate matter, SO_2, NO_X, HCl, dioxins, and polycyclic aromatic hydrocarbons (PAHs), into the environment during their disposal or treatment (Liu et al., 2018). Incineration and engineered sanitary landfills are the most common treatment techniques for medical waste management worldwide. Other techniques that have been developed in recent years include pyrolysis, chemical disinfection, high-pressure steam sterilisation, plasma disinfection, and electromagnetic wave disinfection (Capoor and Bhowmik, 2017; Diaz et al., 2005; Xie et al., 2009).

Most secondary pollutants, such as PTEs and organic chemical compounds, are released from the stack gases of incinerators to the atmosphere during the incineration process. During the incineration process, some are transformed into volatile metallic vapours and enter the flue gas. The remaining majority settle down at the bottom as ash (Li et al., 2017; Liu et al., 2018; Wu et al., 2014), which contains a significant part of PTEs, except for the readily volatile PTEs such as Cd and Hg (Alvim-Ferraz and Afonso, 2003; Krivanek, 1996; Singh and Prakash, 2007; Wheatley and Sadhra, 2004).

While PTEs are emitted from all kinds of incineration techniques, including controlled air, rotary kiln and liquid-injection furnaces, the emission quantity depends on temperature and retention time in the incineration compartment, as well as the control measures aimed at capturing emissions from incinerators. The most commonly known PTEs associated with medical waste are Pb, Cd, Hg, Cr, arsenic (As), nickel (Ni), Zn, and barium (Ba) (Anastasiadou et al., 2012; Wang et al., 2012; Zhao et al., 2010). Typically, most of the PTEs in medical waste result from sharps materials, including needles, expired medicines containing PTEs, and discarded medical equipment, including waste batteries, plastics, rubbers, and adhesive plasters (Chen et al., 2013; Duan et al., 2008; Padmanabhan and Barik, 2019; Zhao et al., 2009).

PTEs in medical waste are typically present in the form of metal oxides (e.g., lead oxide), metal elements (e.g., copper element), volatile metallic chlorides (e.g., cadmium chloride), and sulfates (e.g., copper sulphate). The content of PTEs in the bottom ash of incinerated medical waste and the average values are shown in Tables 1 and 2. PTE concentrations vary considerably depending on different incineration techniques (Table 1, regarding bottom ash) and the type of ash produced (Table 2, regarding bottom or fly ash).

The average chemical composition of incinerated fly ash of medical waste and municipal solid waste (MSW) is shown in Table 3. The content of chlorines in medical waste comes from the high consumption of chlorinated plastics. Chlorine under high temperatures, especially during the incineration process, complexes with the PTEs to form PTE chlorides in the flue, and these PTE chlorides are effective catalysts in the formation of dioxins (Chen et al., 2013; Gunes et al., 2015; Hsu et al., 2016; Yoon et al., 2017; Wang et al., 2003). Dioxin is a human carcinogen associated with a range of adverse health effects, and its persistence in the environment has recently attracted considerable attention (Kanan and Samara, 2018; Marinkovic et al., 2010; Sany et al., 2015).

4. Environmental and health impacts of medical wastes

4.1. Health risks

Health care waste contains potential toxins and microorganisms that can be a source of transmission of diseases and contamination of hazardous chemicals if managed improperly and can infect people in health

Table 1
The concentration of potentially toxic elements (PTEs) in incinerated medical waste and an average value in mg kg^{-1}.

Incinerator type	Country/Location	Ag	As	Cd	Cr	Cu	Fe	Hg	Mn	Pb	Zn	Reference
De Montfort	Ghana	28.38	–	7.54	99.30	–	–	0.88	–	143.8.00	16418.00	Adama et al. (2016)
Pyrolysis	Thailand/Nakhon Ratchasima	82.00	–	–	–	–	78531.00	–	–	191.00	4678.00	Racho and Jindal (2004)
Diesel burner operated incinerator	Tanzania/Dares Salaam	–	–	0.01	0.23	67.40	380.90	–	51.60	0.43	80.32	Honest et al. (2020)
Hovel incinerator	Kuwait/Al-Sabah, Ibn Sina	–	–	29.00	1050.00	–	–	–	–	3029.00	–	Abdulla et al. (2001)
Not known	Taiwan	–	–	0.60	1.70	–	–	–	–	52.50	–	Kuo et al. (1999)
Not known	USA/New York	–	143.20	6.10	99.60	–	–	0.06	–	504.70	–	Thompson et al. (1995)
Not known	USA/New York	–	3.20	3.50	86.90	–	–	–	–	145.80	–	Thompson et al. (1996)
Not known	Greece	–	–	5.90	84.00	1100.00	2010.00	–	1.03	2050.00	5650.00	Valavanidis et al. (2008)
Engineered incinerator	Northwest region of Cameroon	–	–	–	530.00	80.00	4900.00	–	100.00	10.00	2870.00	Mochungong (2011)
Open fire pit	Northwest region of Cameroon	–	–	–	130.00	60.00	57250.00	–	180.00	90.00	5390.00	Mochungong (2011)
	Average value (mg kg^{-1})	55.20	73.20	7.50	231.30	326.90	16135.00	0.50	93.70	621.80	5847.00	

Table 2
The total content of trace elements in incineration waste (bottom or fly ash).

Material	Country	Ag	As	Ba	Cd	Co [a]	Cr	Cu	Hg	Mo	Ni	Pb	Sb	Se	Sn [a]	Zn	Reference
BA-I	China	24.24	24.24	2090	ND	36.34	895.37	1160			667.31	0.33	ND		368	8430	Zhao et al. (2010)
BA-II		21.43	21.43	1690	ND	49.87	515.19	1450			500.49	0.07	ND		406	12,700	
BA-III		27.36	27.36	2080	ND	35.77	916.50	1260			519.32	0.24	ND		375	13,720	
BA	Jordan		84	9900	ND	28	3900	2385			1248	3250				3534	Allawzi et al. (2018)
BA-I	Marocco		92.45		0.75		316.07				138.35	1043				3638	Bakkali et al. (2013)
BA-II			48.19		3.81		185.15				31.14	862.60				8236	
FA-I	Greece				3.54		14.7	9.76			22.1	39.6				27.0	Gerassimidou and Komilis (2015)
FA-II					1.73		14.9	22.9			10.2	18.8				92.6	
BA-I	Romania		1.53		2.67		24.4	224			131	1823				1174	Kim et al. (2019)
FA-I			24.9		81		2692	3432			575	10,218				16,500	
BA-I			1.65		3.38		243	435			70.8	529				2570	
FA-I			32.6		130		3860	2950			682	15,639				28,630	
BA	Greece		23.9	13,826	0.40	30.4	8700	19,500	0.01	69	3948	82.5	50.9		53	526	Kougemitrou et al. (2011)
FA			16.9	540	29.40	0.6	100	2300	0.32	9.1	41.1	1560	73.2		303	5137	
FA	Malaysia		4.0		6.9		150		ND			6100					Rahim et al. (2015)
BA			ND		ND		130		ND			340					
	Pakistan				0.05–1.04		0.25–1.03	0.19–19.11			0.39–3.15	0.59–12.08				1.3–13.6	Sabiha-Javied et al. (2008)
	Japan		28.89	54.38	13.56	3.86	12.65	320.22	6.18	7.91	19.96	439.74			1514	1831.5	Sukandar et al. (2009)
	China				32.2			1255	18.48		755.95	1255				794.22	Tan and Xiao (2012)
FA	Greece			100	ND		178	2397			198	1				8234	Valavanidis et al. (2008)
BA				–	5.9		84	1100			62	2050				5650	
FA [b]	Greece						42	765				938				3000	Vavva et al. (2020)
FA	China				60.3			400			227.1	2219				24,252	Li et al. (2020)
FA			103	1260	126		19	1880			154	3160					
BA	Greece	0.8	11	3840	3.2	34		1287	1.3	12.3	124	18	4.7			52.7	Papamarkou et al. (2018)
FA			170		230		60	1700			40	3540				78,690	Liu et al. (2018)
FA	Greece	4.2	12.7	102	3.3	28		138.2	1.8	27	22.4	135.5	13.6			1103	Tsakalou et al. (2018)
BA	Thailand	327.9										765.3				18,711	Racho and Jindal (2004)
BA [c]	China	24.2	24.2	5100	10.2	49.9	895	2300				2100	197		406	30,700	Zhao et al. (2009)
FA [c]		222	237	2900	635	238	264	2900				5400	230		1660	121,000	
FA	Turkey	140	20	5710			810	2840			150	320	250		240	2030	Akyıldız et al. (2017)
Ash [b]	Poland			200	100	100		300			100	200				5800	Sobiecka et al. (2014)
Sil. FA	Greece		140		11	32	97	71	7		41	34				142	Giergiczny and Krol (2008)
Fl. FA			183		16	67	63	136	8		48	35				209	

BA = Bottom ash; FA = Fly ash;. Sil. FA = siliceous fly ash; Fl. FA = fluidal fly ash.
ND: Non-detected.
[a] Elements not covered in the 2003/33/EC Directive.
[b] ("Establishing criteria and procedures for the acceptance of waste at landfills pursuant to Article 16 of and Annex II to Directive, 1999/31/EC").
[c] Obtained from 14 samples (BA) and from 8 samples (FA)—the maximum reported values are cited here.

Table 3
Average chemical composition of incinerated fly ash of medical waste and municipal solid waste (MSW) in (%).

Incinerated bottom ash	Medical waste	Medical waste	Medical waste	Average	MSW	MSW	MSW	Average
Al_2O_3	2.85	10.11	6.9	6.6	16.7	9.2	28.1	18.0
CaO	24.42	5.37	38.5	22.8	1.06	19.39	0.5	7.0
Cl	20.43	17.07	30.7	22.7	ND	0.44	ND	0.4
F	2.59	0.75	ND	1.7	ND	ND	ND	
Fe_2O_3	1.78	1.49	1.1	1.5	1.96	4.93	2.1	3.0
K_2O	2.8	1.64	3.3	2.6	3.94	0.43	1	1.8
MgO	1.8	3.48	2.3	2.5	ND	0.41	1	0.7
Na_2O	15.2	22.05	1.6	13.0	2.42	0.24	0.5	1.1
SiO_2	17.13	9.06	8	11.4	73.1	55.37	62.3	63.6
SO_3	6.37	1.03	1.6	3.0	ND	1.53	0.4	1.0
TiO_2	1.34	ND	3.2	2.3	0.35	ND	ND	0.4
References	Liu et al. (2013)	Wang et al. (2017)	Akyıldız et al. (2017)		Hanif et al. (2017)	Singh and Khosla (2017)	Aliabdo et al. (2016)	

The chemical composition of ash materials is often expressed as oxides of respective elements; ND = Not detected.

care settings. According to the WHO report, about 15% of the total generated health care waste is considered hazardous, and the remaining 85% is general or nonhazardous waste (WHO, 2018b). The generation of hazardous medical waste in high-income countries accounts for an average of up to 0.5 kg hospital bed^{-1} day^{-1}, while in low-income countries, it is up to 0.2 kg hospital bed^{-1} day^{-1} (WHO, 2018a). However, contrary to recommendations from the WHO, health care waste segregation is not practised in many low and middle-income countries. Hence, the toxins from medical waste, discarded from hospitals and sometimes from household and quarantine centres, get mixed with other nonhazardous waste for disposal via landfill or incineration (Singh et al., 2020b).

Potential risks to human health are associated with unsound management of medical waste, including exposure to pathogens, sharps-inflicted injuries, chemical exposure to medicines, especially the disposal of expired medicines, thermal injuries through open burning, and radiation exposure. Many studies on health risks from health care waste have shown that nearly 5.2 million people lose their lives annually due to the unsafe disposal of medical waste worldwide (WHO, 2018a). Additionally, in many developing countries, millions of workers involved in medical waste management may become infected with many infectious diseases, such as HIV, hepatitis B, and hepatitis C infections, due to the contagion of the waste (Singh et al., 2021). In 2015, a joint WHO/UNICEF assessment found that around 58% of sampled facilities from 24 countries had adequate systems in place for the safe disposal of health care waste WHO (2015). However, the waste handlers are at immediate risk of needle-stick injuries and exposure to toxic or infectious materials. For example, although injections with contaminated needles in low- and middle-income countries have reduced substantially in recent years, unsafe injections were still considered responsible for as many as 33,800 new HIV infections, 1.7 million hepatitis B infections and 315,000 hepatitis C infections (Pépin et al., 2014). In addition, the release of PTEs and other pollutants from medical waste presents indirect health risks associated with those pollutants, for example, lead poisoning.

The negative impacts of untreated health care waste are mostly inflicted on the vulnerable population living in poor socioeconomic conditions, that account for almost 24% of the urban population in developing countries (UN, 2020). A Health Care Without Harm (HCWH) report revealed that delinquent management and improper disposal and treatment of medical waste not only create risks to environmental assets and complications to public health but also impact human rights due to the little attention paid by international communities (HCWH, 2011).

4.2. Environmental impact

Waste management in the health care industry, combined with other health industry emissions, is estimated to be 4.4% of net global greenhouse gas emissions (Karliner et al., 2019). Unsound management of this medical waste could be a potential risk to the environment through the release of contaminants into the environment. For example, landfilling untreated medical waste (non-disinfected waste) could seriously contaminate environmental water bodies, including sources of drinking water, if those landfills are not appropriately designed (Babanyara et al., 2013). Poorly managed landfills can become a public health concern resulting from soil and water contamination (Kenny and Priyadarshini, 2021). Furthermore, landfill leachate and emission of dangerous gases, including volatile organic compounds and, particularly, benzene, toluene, ethylbenzene and xylene isomers (collectively called BTEX), can be harmful to human health (Lakhouit et al., 2014; Xu et al., 2018). Chemical disinfectants used to treat medical waste, such as sodium hypochlorite and chlorine, could pose severe indoor or outdoor contamination risks if the chemical substances are handled, stored, and disposed of improperly (Gola et al., 2019; Wang et al., 2020).

Incineration of medical waste and MSW are the third largest source of dioxin emissions into the air (EPA, 2020). However, incineration of medical waste containing or treated with chlorine can generate dioxins and furans, which are human carcinogens and have been associated with a range of adverse health effects (Kanan and Samara, 2018). Research has demonstrated that both traditional and advanced incinerators can contribute to the contamination of soil and vegetation with dioxins and PTEs. Populations residing near incinerators are potentially exposed to chemicals, including dioxins and PTEs, through inhalation of contaminated air or by consuming contaminated agricultural produce and by direct dermal contact with contaminated soil (Verma and Srivastava, 2000). For example, significantly increased levels of dioxins have been found in the tissues of residents near to incinerators in the UK, Spain and Japan, most likely as a result of such exposure (Gautam et al., 2010). Several studies have reported elevated levels of dioxins/dioxin congeners in the body tissues of individuals employed at both traditional and advanced incinerators, which has been attributed to direct exposure to contaminated ashes in the workplace.

Incineration of medical waste is one of the most commonly used treatment methods globally. In many developing countries, due to the high cost of abatement devices/treatments that capture the pollutants such as dioxins and PTEs after the burning of the waste, the medical waste is openly burnt or incinerated without safety control devices, which results in toxic air pollution (Vilella, 2012). Many studies conducted during the COVID-19 pandemic found that poor air quality is possibly correlated to a higher fatality rate ranging from 8% to 21.4% in the U.S. and the Netherlands (Wu et al., 2020).

An adequate disinfection process could play an important role in an effective and safe medical waste disposal system, especially in developing countries which practise unsafe incineration and open burning. The combination of chemical disinfection and incineration could play a vital role in the safe disposal of rapidly growing health care waste.

However, chemical infection using chlorine-based disinfectants followed by incineration may result in the release of dioxins. Hence, care must be taken in choosing the appropriate disinfectants. Medical waste should be managed comprehensively using existing infectious and health care waste management protocols recommended by national and international guidelines and manuals (UN, 2020). Proper segregation, packaging, and storage of potentially contaminated materials in the double bag for at least 72 h, adequate waste collection bins, and training and capacity building of the staff involved in healthcare waste management are essential for properly handling medical waste.

4.3. Increased challenges due to the Covid-19 pandemic outbreak

In 2020, the severe acute respiratory syndrome coronavirus 2 (COVID-19) pandemic spread worldwide. Millions of people have been infected since then, including some of the world's most isolated communities living in isolated islands where health care waste management does not exist. Almost 90% of the infected cases of COVID-19 were in urban areas, considered the epicentre of the COVID-19 spread (United Nations Human Settlements Programme (UNHSP), 2020). Inadequate health care waste management in urban areas during the virus outbreak has been reported in many countries. Hence, the pandemic resulted in an increased risk of disease transmission either directly through open-dumped health care waste or indirectly through food-chain contamination (Singh et al., 2020a; You et al., 2020). For instance, it has been calculated that inappropriate disposal of medical waste may have caused up to 30% of hepatitis B, 13% of hepatitis C, and 0.3% of HIV rates to be directly transmitted from patients to healthcare professionals (Singh et al., 2020a). Studies from Pakistan, Greece, Brazil, Iran, and India demonstrate that pathogens in contaminated wastes are directly responsible for a higher-than-average prevalence of viral infection in solid waste collectors (Mallapur, 2020). Likewise, exposure to PTEs indirectly through the food chain (soil-plant-animal-human or soil-plant-human) can cause chronic diseases, such as cancer, in a population (Bolan et al., 2022). Due to their high chemical stability, poor biodegradation (decomposition) capability, and high toxicity potential, PTEs have a prolonged residence time in the food chain and the environment (Omeka and Igwe, 2021).

The global health care waste management system, which already poses a risk to billions of people from occupational, public health, or environmental perspective, is now under unprecedented burdens due to the surge in COVID-19-related healthcare waste generation (Fig. 2A) (Singh et al., 2021). Biomedical waste generation has exceeded waste management capacity in many cities, particularly in middle- and low-income countries. For example, in Dhaka, the capital of Bangladesh, biomedical waste has increased nearly 70 times, from approximately 3 tons per day before the pandemic to 206 tons per day in 2020 (Rahman et al., 2020). In Delhi, infectious medical waste has increased by nearly 14 times compared to pre-pandemic levels, from approximately 25 tons per day in May to 371 tons per day in June 2020 (Reddy, 2020). In East-Asian cities, including Bangkok, Manila, Jakarta, Ha Noi, Kuala Lumpur, and Wuhan, biomedical waste has increased by approximately six times since the COVID-19 outbreak (ADB, 2020). The increased biomedical waste means increased risk to its exposure through PTEs and other pollutants.

Additionally, the transmissions of the COVID-19 virus are not solely through interaction with infected persons but also can spread rapidly via discarded PPEs, including face masks and other discarded protective materials. Studies conducted in 26 countries belonging to the World Health South-East Asia Region Organization (SEARO) group show that only 58% of sampled facilities had adequate disposal for health care waste (Fig. 2B) (WHO, 2015). The SEARO region countries, including Bangladesh, Bhutan, India, Nepal, Sri Lanka, and Timor-Leste, showed only 44% of facilities having a system for the safe management of health care waste.

5. Management of medical wastes

5.1. Medical wastes treatment and disposal methods

Improper medical waste management may risk health care staff, exposed populations, and the natural environment. There is a large body of literature on the issues and ways of achieving efficient medical waste management (Abdulla et al., 2008; Lee et al., 2004; Patil and Shekdar, 2001; Jang et al., 2006). The techniques employed by hospitals in northern Jordan for managing medical waste were examined by Abdulla

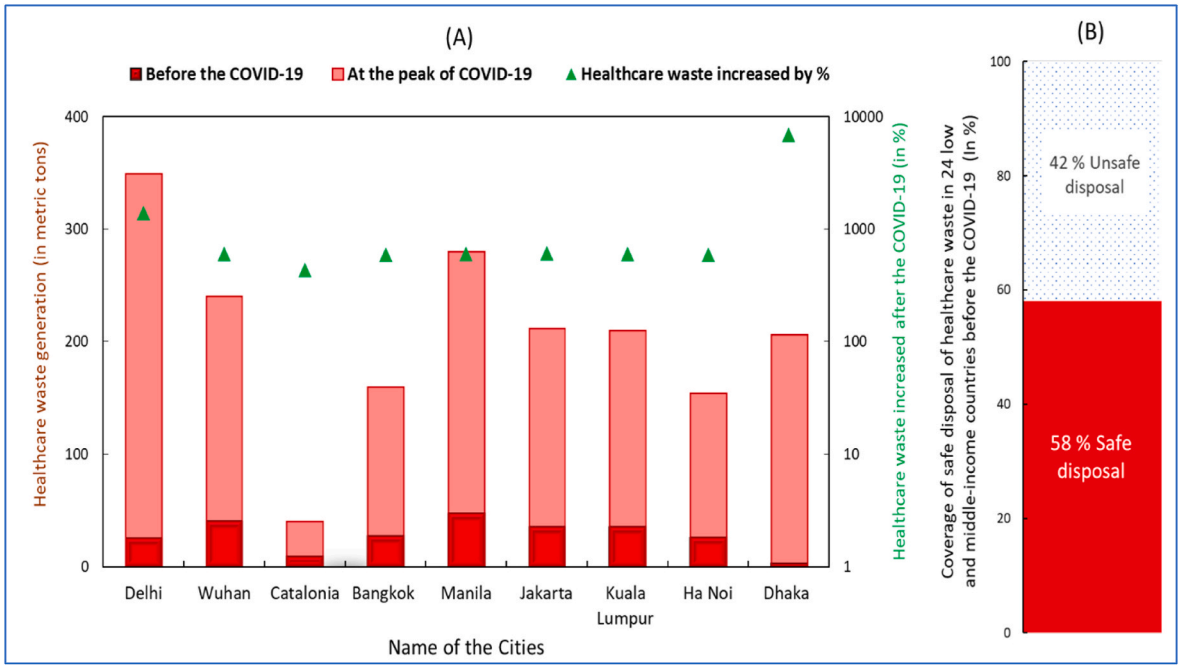

Fig. 2. Global medical waste generation and disposal scenario amid COVID-19 outbreak. (A) The medical waste generation before COVID-19 and at the peak of COVID-19 in the cities in 2020. (B) Coverage of safe disposal of healthcare waste from 24 low and middle-income countries.

et al. (2008). This study found that incineration was the most popular method of treating solid medical waste, although source segregation and treatment of liquid medical waste are still lacking (Abdulla et al., 2008). Previously, Lee et al. (2004) studied and reported that on-site integrated microwave and incineration is the most cost-effective method of regulated medical waste treatment in comparison to microwave and incineration alone. Patil and Shekdar (2001) studied and provided a detailed discussion on medical waste generation and management practices in India. The waste-management strategy for medical waste proposed by their study involved institutional arrangements, suitable technology, operational planning, financial management, and the development of suitable staff training programs. Similarly, Jang et al. (2006) studied and provided an overview of the management practices of medical waste in Korea. Medical waste incineration is identified as the most preferred disposal method. Nevertheless, the main problems in the future are thought to be waste minimisation and recycling and control of harmful air emissions during incineration.

The disposal and treatment of medical waste depend on waste characteristics, volume, costs associated with treatment, and environmental and safety factors (Eslami et al., 2017). Medical waste treatment and disposal processes are categorised into three main processes: thermal, chemical, and irradiation (Fig. 3). High temperature is used in thermal processes to reduce the waste volume and weight by converting it to inorganic and incombustible matter. This process treats waste that cannot be recycled, reused, or disposed of in a landfill site (Xu et al., 2020). Chemical treatment is used to kill or inactivate pathogens in medical waste. For chemical treatment, disinfection is more commonly practised than sterilisation. Chemical disinfection is used for treating liquid waste, such as urine, faecal matter, blood, or hospital sewage (Kenny and Priyadarshini, 2021). In some cases, Gamma radiation is used to irradiate medical waste. However, irradiation involves stringent regulatory requirements and approvals. The associated costs and the strict regulatory protocols hinder its usage (Kenny and Priyadarshini, 2021; Xu et al., 2020).

The incineration technologies are the most advantageous due to the significant reduction in the overall volume and weight of medical waste. Incineration is also suitable for all kinds of medical waste and can be implemented at a very large scale. However, the chemical disinfection process only requires modest and convenient instruments and operation, involving a small one-time investment and lower operational costs. A comparison of different processes, their usage, technical specifications, advantages, and limitations are detailed in Table 4.

5.2. Critical elements in medical waste treatment

Medical waste management issues are intensifying due to the lack of public awareness, proper waste disposal practices, and financial resources (Birpınar et al., 2009; Marinković et al., 2008). Some of the critical elements in improving the issues of medical waste management are shown in Fig. 4. Establishing rules and regulations for waste separation and disposal in the national or institutional framework would lead to improved waste management practices (Bokhoree et al., 2014; Taghipour and Mosaferi, 2009). Lack of awareness can be minimised through proper training of all the concerned personnel. Appropriate training is essential to develop a cognisance related to medical waste and its adverse impacts on the health of individuals and the environment (Bokhoree et al., 2014). Allocation of funds/resources to such training programmes would improve medical waste management efficiency. Establishing a database, including classifying and quantifying the generated medical waste, is essential for successful waste management. It will allow the efficient allocation of resources for managing medical waste (Walkinshaw, 2011).

The practices that reduce the volume of waste produced should be promoted. It is essential to segregate infectious and non-infectious medical waste before treating it. A colour-coded system can be used to separate medical waste properly. Such segregation practices will deliver a clean waste stream that can be effortlessly, safely, and economically managed (Yong et al., 2009). Medical waste storage areas should be secured to prevent flies, dogs, and rodents from spreading various diseases (Debita et al., 2014). Investing in eco-friendly, sustainable, and cost-effective health care waste treatment and disposal technologies such as microwave sanitation, gasification, and pyrolysis can reduce environmental hazards (Chisholm et al., 2021; Coker et al., 2009). However, these treatment technologies are applied with different waste management goals, and some of these waste treatment technologies (e.g., microwave) may alter the infection risk of medical wastes but not the PTE risk. Although, treating and discarding medical waste on-site is a desirable option, particularly in an emergency. According to a recent study, setting up on-site temporary thermal treatment facilities could be an effective way to handle the enormous growth in medical waste (You et al., 2020). However, the most significant hurdles are the strict limitations on air pollution emissions of on-site treatment facilities and the

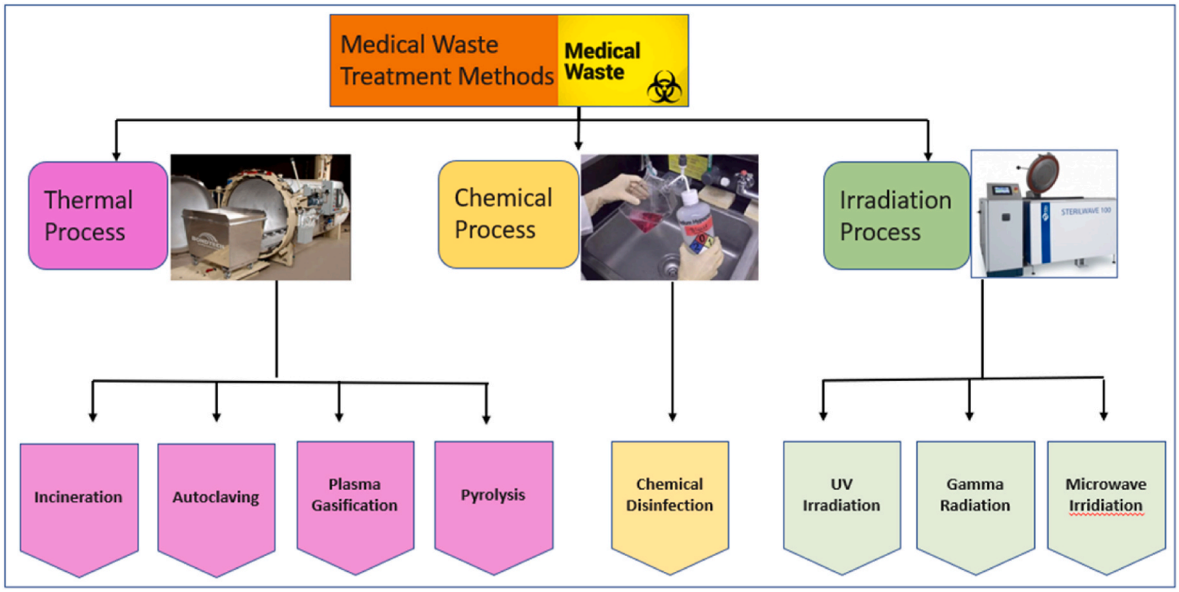

Fig. 3. Schematic representation of various medical waste treatment methods.

Table 4
Comparison of medical waste treatment processes.

Treatment method	Waste category	Technical specifications	Advantages	Limitations	References
Thermal Processes					
Incineration	Infectious waste Pathological waste	Operate in a furnace at 900 °C - 1200 °C	● Reduce the volume of materials ● Destroy pathogens and hazardous organic substances ● Reduce transport costs	● Emit trace amounts of human carcinogenic by-products (polychlorinated di-oxins and furan) ● Bioaccumulation in the environment. ● A complete purification system is required for exhaust gas ● Bottom slag and fly ash are hazardous	(Abdulla et al., 2008; Jang et al., 2006; Xu et al., 2020; Zafar, 2019)
Autoclaving	Cytotoxic, pathological or various other toxic chemical wastes, infected work clothes, dressings, syringes, microbial growth media	Low-heat thermal process, pressurized steam at a temperature around 121 °C	● Non-emission of toxic gases ● Low operational cost ● Fewer risks of residues ● Good disinfection efficiency ● Treatment for a diverse range of wastes	● Chances of air pollution ● Odour generation ● Cannot handle formaldehyde, phenol, mercury, and other substances	(Jang et al., 2006; Kenny and Priyadarshini, 2021; Shannon and Woolridge, 2011; Thakur and Katoch, 2012; Windfeld and Brooks, 2015; Xu et al., 2020; Zafar, 2019)
Hydroclaving	Cytotoxic, pathological or several other toxic chemical wastes	Waste is subject to indirect heating by applying steam	● Weight and volume reductions are up -to 70% and 85% ● Absence of harmful emissions ● No chemical use	● Required specialized and technical skills to operate ● Similar limitations to autoclaving	(Babanyara et al., 2013; JessMathew et al., 2017; Kenny and Priyadarshini, 2021)
Gas sterilisation	Infectious waste	Waste is kept in an evacuated tight-air chamber, and applying sterilisation agent (formaldehyde or ethylene oxide)	● Low investment, ● Low operating cost	● Required specialized and technical skills to operate	Windfeld and Brooks (2015)
Plasma gasification	Infectious waste, Pathological waste, organic waste	An electric current is supplied to produce a glow plasma discharge, which can reach a temperature of 1200–3000 °C and finally, waste is quickly heated and dehydrated	● High reduction in volume ● High treatment efficiency ● Appy to diverse forms of wastes ● No harmful emission ● The heat energy can be recycled and reused	● Required specialized and technical skills to operate ● Very high establishment and operation cost ● Poor system stability ● The consistency needs to be tested and upgraded	JessMathew et al. (2017); Kenny and Priyadarshini (2021); Xu et al. (2020); Zafar (2019)
High-temperature pyrolysis incineration method	Infectious waste and pathological waste	The organic fraction of medical waste treated at 600–900 °C under controlled or oxygen-free condition	● Does not require pretreatment ● Operating cost is much lower ● Significant reduction in flue gas	● Required specialized and technical skills to operate	JessMathew et al. (2017); Kenny and Priyadarshini (2021); Xu et al. (2020)
Thermal inactivation	Liquid clinical wastes	A chamber is pre-heated to a specific, intense temperature	● Operating cost is much lower	● Required specialized and technical skills to operate	Windfeld and Brooks (2015)
Chemical Processes					
Chemical Disinfection	Hospital sewage, urine, stools, or blood, medical wastes which are not sterilized by wetting or heating	Use Sodium hypochlorite (bleach), dissolved chlorine dioxide, dry inorganic chemicals, and peracetic acid	● Modest and suitable process ● Decent deodorization effect ● Swift disinfection method with a one-time investment ● Low operating cost	● Dry waste has higher requirements on the crushing system and the pH value monitoring (automation level) of the operation process ● Generation of liquid and gaseous pollutants during the wet waste treatment operation ● Potential for dioxin generation following incineration	Kenny and Priyadarshini (2021); Shareefdeen (2012); Thakur and Katoch, 2012; Xu et al. (2020)

(*continued on next page*)

Table 4 (*continued*)

Treatment method	Waste category	Technical specifications	Advantages	Limitations	References
Irradiative Processes					
UV irradiation	Infectious waste Pathological waste	UV light used	● Low cost as no requirement of chemicals ● Absence of harmful air emissions	● Appropriate materials, equipment and training of personnel are required	Kenny and Priyadarshini (2021); Sharma and Sharma (2019)
Cobalt-60	Infectious waste Pathological waste	Produces gamma rays	● The high penetrative effect, no requirement for chemicals ● Absence of harmful air emissions	● High processing cost Requires highly skilled personnel to manage the gamma irradiation	Kenny and Priyadarshini (2021); Sharma and Sharma (2019); Xu et al. (2020)
Microwave processes	Infectious waste Pathological waste	Microbial inactivation due to the thermal effect of the electromagnetic radiation spectrum (300-300,000 MHz	● Reduced volume of waste, ● No requirement for chemicals	● Processing cost is high ● Low weight reduction ● Requires the use of noisy shredders ● Produce foul odours	Akter (2000); Kenny and Priyadarshini (2021); Windfeld and Brooks (2015); Xu et al. (2020)

huge installation price (Taghipour et al., 2014). In addition, the thermal processing of medical waste has a number of drawbacks, including high energy requirements, bottom ash emission, and fly ash generation (Cai and Du, 2021).

Recently, thermal plasma processes have been gaining attention and are considered a cost-effective alternative to the thermal treatment of medical waste. Applying thermal plasma technology to treat medical waste results in lower energy consumption and energy and material recovery, which could lead to commercialisation feasibility (Munir et al., 2019). High process temperatures, a decrease in the need for off-gas treatment, and the ability to have a reduced installation area and cost are all benefits of thermal plasma technology. Therefore, the on-site and off-site treatment of medical waste is thus a promising solution (Cai and Du, 2021). Furthermore, to minimise the problems associated with the improper transportation and management of medical waste, a fixed timetable for transport should be prepared, and superior-quality, safe packaging containers should be used to transport medical waste (Abdulla et al., 2008; Yong et al., 2009).

6. Occurrence of potentially toxic elements in the vicinity of incineration plants

Lands outside hospital waste incineration plants are likely to be affected by emitted PTEs. However, studies that report data from such cases are extremely scarce. A notable reason may be that safety protocols, intense site monitoring and law enforcement of such areas make any leakage reporting very unlikely. However, some research has shown that if fumes are emitted, PTE enrichment in soils of the vicinity is indeed likely. For example, in Brazil, an area at a distance of up to 500 m from a hospital waste incineration plant was studied from 2000 to 2003 (Segura-Munoz et al., 2004). The plant was deactivated in 2002. It was found that Cd, Mn, and Pb decreased after the incineration plant deactivation. However, the measured elements' concentrations were unacceptably high for Cd and Mn. This was due to the fact that the initial emitted concentrations of Mn and Cd were considerably high, and this resulted in their long-time persistence in soil. In Ghana, a similar study was conducted by Adama et al. (2016) in an area outside a medical waste incineration facility. Soils were reported to be highly elevated with a variety of PTEs (Hg = 0.88; Zn = 16,418; Cd = 7.54 mg kg^{-1}), with a clear decreasing trend with distance. The pollution load index also was higher than 20 at a distance of 20 m from the facility. It decreased to less than unity at 120 m, a trend showing the significant enrichment caused by facility-emitted PTEs.

7. Transformation of potentially toxic elements in medical waste ash

The incineration of wastes, including medical wastes, has been extensively practised globally. An inefficient incineration process or the incineration of inappropriate wastes leads to the emission of pollutants into the atmosphere. It can also generate high ash residue. Incineration of medical waste having high PTE content, especially Pb, Hg, and Cd, can lead to the release of these PTEs in the environmental matrices. Table 2 shows the total content of trace elements in medical incineration waste (bottom or fly ash). The suitability of the incineration wastes is evaluated with leaching test protocols; various such protocols have been developed in various major countries or organizations and are shown in Table 5.

7.1. Stabilisation of potentially toxic elements in medical waste ash

Most solid waste treatments, including medical waste ash, aim to decrease the leachability of PTEs so that the wastes may have PTE concentration in the leachates below the maximum limits. The procedures are either chemical, where some binding agents are applied so that wastes containing PTEs are chemically stabilised, or physical/mechanical, where a solidification agent is applied. An overview of these methods is discussed below.

Chemical stabilisation methods, as the name suggests, use chemical agents. Phosphoric acid is one of the most widely used chemical binding agents. For example, Sukandar et al. (2009) used acidic (pH 1–2; Ashnite R303) and alkaline (pH 9.3–10.3; Ashnite R201) phosphoric acids for waste stabilisation. The binding mechanisms identified by the authors were as follows: with the application of Ashnite R201, Pb-hydroxyapatite precipitate is formed, leading to the removal of Pb from the soil solution. Also, Cu, Cd, Zn, Mn and Ni can be precipitated at high pH in the presence of phosphorus as phosphate-metal-hydroxy complexes. With Ashnite R303, chloropyromorphite $Pb_5(PO_4)_3Cl$ is likely formed, a highly insoluble complex that keeps Pb out of the solution. Likewise, Vavva et al. (2020) treated incinerated medical waste ash with 7% of phosphoric acid and found that Pb in the leachate decreased significantly from 4.59 to 0.30–0.35 mg L^{-1}, while there was no effect of phosphoric acid treatment on the leaching of other elements, including Cd and Cu.

Another chemical stabilisation process was reported by Jin et al. (2010) using alkaline agents. In this study, 20% w/w of NaOH and Na_2CO_3 were mixed with medical ash. The retention mechanism was

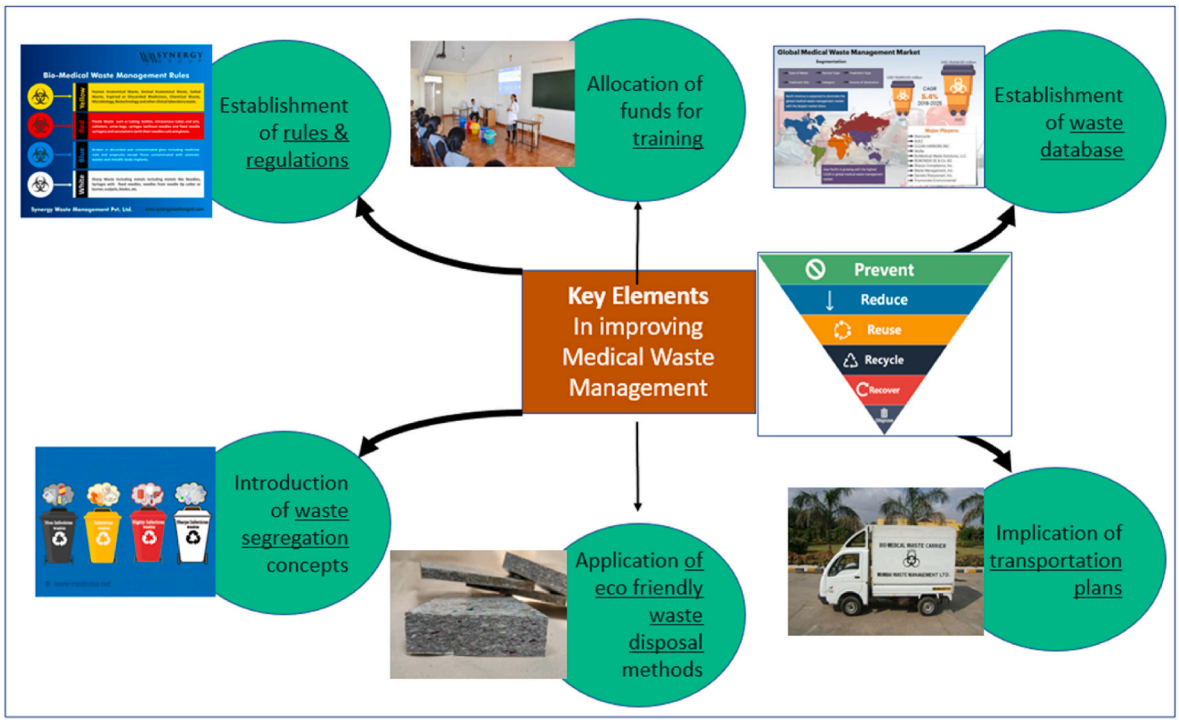

Fig. 4. Key elements for achieving a successful medical waste management system.

presumably the formation of insoluble metal hydroxides and metal carbonates. The authors found that As decreased by 51%, Pb by 58%, Sr by 97%, and Zn by 87% by the addition of NaOH, and 66% (As), 58% (Pb), 84% (Sr), and 82% (Zn) by addition of Na_2CO_3, compared to the untreated ash. It was also reported that the stabilisation failed for Cr, Cu, Cd, and Ni, the regulated metals, and Co and vanadium (V), the non-regulated metals.

As for the mechanical or physical stabilisation methods, they include melting at high temperatures (Tan and Xiao, 2012), vitrification (Tsakalou et al., 2018), clinkerization (Papamarkou et al., 2018), and S/S (Ababneh et al., 2020). However, those can be cost-prohibitive or difficult to implement on a large scale.

7.2. Release of potentially toxic elements from medical waste incineration fly ash (MWIFA)

Some treatment methods aim to liberate PTEs from the wastes so that they may be cleaned before applying a formal leaching protocol. In this way, leaching reduces PTEs to acceptable concentrations. The literature mainly includes reports on the use of ethylenediaminetetraacetic acid (EDTA) as a releasing agent of PTEs from MWIFA. For example, in the study by Li et al. (2020), a proper circulating system was established to remediate PTEs in MWIFA. MWIFA-column studies were accomplished to remediate Cd, Zn, Pb, Ni, and Cu from MWIFA by applying EDTA disodium (Na_2-EDTA). Column studies were performed to investigate the remediation efficiency of zero-valent iron (Fe (0)) on the five PTEs from MWIFA washing water. The Toxicity Characteristic Leaching Procedure (TCLP) test was opted to assess PTEs toxicity of MWIFA residues generated after waste ashes were treated with EDTA at a range of molarities of 0–0.2 mol L^{-1}. After being washed with 0.2 mol L^{-1} Na_2-EDTA solution, the ashes having PTEs (Cu = 6067.2 ± 422.3, Pb = 2219.7 ± 20.6, Zn = 24252.5 ± 416.4, Cd = 60.3 ± 12.1, and Ni = 227.1 ± 17.8 mg kg^{-1}) were subjected to leaching with the TCLP protocol: PTEs were decreased to Cu = 58.4 ± 2.0, Pb = 2.81 ± 0.14, Zn = 64.3 ± 4.0, Cd =

Table 5
Selected medical incineration waste leaching protocols.

Protocol	Country/ Organization	Liquid-to-solid ratio	Leaching solution	Hours of shaking	Particle size class	Leachate filtered through	Reference
DIN 38414	NA	10	d. H_2O	24 h	"Powder"	0.45 μm	Bakkali et al. (2013)
EN-14429	Germany	10	HNO_3 (pH unspecified)	48 h	<1 mm	0.45 μm	Xu et al. (2018)
EN-12457	European Union	2 (for EN-12457-1) 10 (for EN-12457-2)	d. H_2O	24 h	<10 mm	0.45 μm	EC (2003)
TCLP-1311	USEPA	20	Acetic acid diluted at 5.7 mL/L, pH 2.88	18 h	<10 mm	0.45 μm	Xu et al. (2018)
TCLP-1313	USEPA	10	HNO_3/KOH (pH 3, 5, 7, 9, and 11)	18 h	<10 mm	2 μm	Patel and Devatha (2019)
PN-Z-15-009	Poland	10	d. H_2O	4 h shaking/16 h undisturbed/4 h shaking	<1 mm	0.45 μm	Gielar and Helios-Rybicka (2013)
JLT-13	Japan	10	d. H_2O	6 h	Non specified	1 μm	Sukandar et al. (2009)
HJ/T299	China	10	Dilute H_2SO_4/HNO_3 to pH 3.20	18	<10 mm	0.45 μm	Xu et al. (2018)

NA: Non-applicable.

0.156 ± 0.005, and Ni = 0.381 ± 0.006 mg L^{-1}. The effluents from the EDTA treatment were driven through a Fe (0) column so that the ash-liberated PTEs may be stabilised. Indeed, they were efficiently reduced by 99.7% (Cu), 91.6% (Pb), 91.6% (Zn), 75.4% (Cd), and 75.7% (Ni). Also, the EDTA solution was reused for further ash treatment up to 4 times. The extraction efficiency of Pb and Cd (removal efficiency at different cycles divided by removal efficiency of new Na_2-EDTA) declined toward 80%. Similar work has been conducted by Ababneh et al. (2020). They applied EDTA and $Na_2S.9H_2O$ as releasing agents at a 10:1 solution-to-solid ratio with 0.2 M EDTA. The treatment decreased PTEs from the ashes by 100% (for Cd), 74% (Pb), and 86% (Zn). $Na_2S.9H_2O$ was much less effective, as it only decreased Pb by 37% and Zn by 7%, while it completely failed to decrease Cd concentration.

8. Three frequently encountered misconceptions

8.1. Only some medical waste incinerated residues are hazardous

The source of this misconception is the fact that the European Union has a different approach to that of the United States Environmental Protection Agency (USEPA) in America. Medicine-derived wastes are categorised only as "hazardous wastes" in the European Union legislation irrespective of their content in 91/689/EC Directive (EC, 1991)—no other characterization is allowed. It needs clarification in the sense that there is a widespread misconception in the literature that medicinal waste becomes hazardous only if some PTEs are found at high concentrations, a concept accepted by USEPA. For example, there are works that categorise MWIFA as "nonhazardous" or even as "inert," according to findings concerning their PTE leachability. According to the 1999/31/EC Directive, p. L182/3 (EC, 1999), "inert" is a waste if it "does not experience any substantial physical, chemical and biological alterations". Thus, no medical waste can be under this category, even with minimal PTE content.

Along these lines, Kougemitrou et al. (2011) claimed that the studied MWI ash was "inert" according to the European leaching protocol because leached concentrations of PTEs were low. Also, Tsakalou et al. (2018) and Papamarkou et al. (2018) compared their leaching rate of PTEs with "inert" and "hazardous" waste limits, while Ababneh et al. (2020) compared their leaching with limits for nonhazardous waste, implying that the waste is nonhazardous. In some other cases, the misconception may well be a matter of imprecise use of terms, as in the distinction between "hazardous" and "medical" wastes (Racho and Jindal, 2004), as well as "dangerous" and "hazardous" (Bakkali et al., 2013). Hazardous wastes are permitted to be disposed of in landfills, similar to nonhazardous waste if leaching tests satisfy specific criteria of maximum limits. These are listed in the 2003/33/EC Directive (EC, 2003) and are shown in Table 6. However, it is not incorrect to claim MWI nonhazardous if decided based on the United States Environmental Protection Agency (USEPA) guidelines (CFR, 2020). It states that the waste becomes hazardous only if the regulated PTEs [Pb, Ni, Hg, Cu, Cr, Cd, Ba, As, Ag, and selenium (Se)] overcome their limit values as noted by Zhao et al. (2010) and Xie and Zhu (2013). Thus, care must be exercised when comparisons of results are made against the European legislation (where no waste can be anything other than "hazardous") or against the USEPA legislation (where wastes may be hazardous or nonhazardous depending on content).

8.2. Landfilling is based on total PTEs in medical waste incinerated residues

The landfilling option for hazardous wastes such as MWI is decided based on total (or extractable) PTEs concentrations in the MWI wastes, which is a misconception. However, the fact is that landfilling option is decided based on the PTEs' leachability potential. Therefore, it is important to note that the crucial factor for deciding where waste ash is to be disposed of is not based on the actual content of PTEs (total or extractable in any form, Table 6). The misunderstanding comes from the fact that total PTEs content is the decisive factor in all other environmental matrices (soil, water and foodstuff) or wastes. In other words, *in contrast to other matrices, the leachability of metals (measured in mg L^{-1}) is used to determine the suitability of MWI waste of disposal in a landfill, while for other wastes, total elemental concentration (measured in mg kg^{-1}) is used to determine the suitability of waste for disposal in a landfill. This difference causes various studies to compare wrong limits.* This misunderstanding also derives from the fact that the 33/2003 Directive (EC, 2003) reports the PTEs limits in units of dry solid substance (in mg PTE leached kg^{-1} ash). Thus, this misunderstanding may lead to the wrong notion that the decisive matter is total content.

Along these lines, Li et al. (2020) evaluated the medical wastes based on their total PTEs content and compared it with total PTEs background concentrations in soil, falsely implying an association between the two. Also, in Sabiha-Javied et al. (2008), no leaching was performed, but only the total contents of PTEs were determined. Subsequently, the measured concentrations of Pb, Cr, and Cd (as mg kg^{-1} dry waste) were compared to the USEPA limits of these metals. However, these limits are derived from leaching measurements (not total) and are expressed in mg L^{-1} of leachate (not mg kg^{-1}). Such misunderstanding is not necessarily characteristic of medical waste alone, but it certainly derives from the fact that leaching limits are expressed in mg kg^{-1}. Nevertheless, *it is important to point out that total concentrations (in mg kg^{-1}) and leachate concentrations (in mg kg^{-1}) are not the same and should never be directly compared, regardless of whether the conversion from mg L^{-1} to mg kg^{-1} has been made.* Moreover, Ababneh et al. (2020) digested fly ash treated with EDTA and $Na_2S.9H_2O$ as metal-releasing agents and measured Cd, Pb and Zn in mg kg^{-1} of ash, concentrations which were then compared against the EC (2003) limits—misuse probably caused due to the misconception explained below.

8.3. Expression of PTE concentration in liquid or solid waste

The third misconception of PTEs' concentration in medical waste corresponds to their expressions in solid or liquid units. Mostly the PTE concentrations are expressed in the liquid state (i.e., the leachates), but as stated earlier, limit values in the 33/2003 Directive (EC, 2003) are expressed in mg of PTEs per kg of dry substance. Thus, at first, it is not clear whether this is to be measured in the dried leachate or whether the concentration of PTEs in mg L^{-1} of leachate must be calculated back to the original source waste mass. Also, although medicinal wastes are in solid form, there may be a misunderstanding that they may also exist in liquid form. Indeed, this misunderstanding has led most of the works to report their results in ppm of liquid (mg L^{-1}, i.e., directly in the eluted leachate), although such values cannot be directly compared to the legal limit EU values. This entanglement seems to be the reason for the misunderstanding, as seen in Vavva et al. (2020). In order to compare their findings (leaching conducted at liquid/solid (L/S) ratios of 7, 10 and 12) to the EU limits as per EC (2003), they expressed these limits in mg L^{-1}. However, if the association between concentration units per liquid vs per solid was only a matter of appropriate conversion, then the EU PTE limits at L/S = 2 (also expressed in mg kg^{-1}) should always be five times higher than those at L/S = 10.

The comparisons even become impossible if the leaching protocols are not based on the ratios of L/S = 2 and L/S = 10. For example, Allawzi et al. (2018) applied an in-house leaching protocol with L/S ranging from 17 to 83 and still compared their findings with the USEPA limits. Similar comparisons with the USEPA limits were attempted by Tan and Xiao (2012), although they conducted leaching at L/S ranging from 10 to 90. Similarly, Rajor et al. (2012) used an L/S ratio of 16. Also, Kim et al. (2019) applied the TCLP (TCLP, 1996) protocol of L/S = 20, but comparisons were made with the European Union limits of EC (2003).

This misconception has led some works to claim compliance with the EU limits, although this claim is false if wrong L/S ratios are employed (typically L/S = 20). Thus, comparisons of results derived from TCLP vs

Table 6
PTE content in medical waste (bottom and fly ash) from leaching tests. The values are compared to the limit values of leaching of "hazardous waste" (which include "pharmaceuticals, medicines and veterinary compounds," as per 91/689/EEC) at L/S (liquid-to-solid) = 2 L kg^{-1}, at L/S = 10 L kg^{-1} as per the 2003/33/EC Directive. Limit values are categorised in Criteria A ("for hazardous waste acceptable at landfills for non-hazardous waste) and in Criteria B ("for hazardous waste acceptable at landfills for hazardous waste) (units in mg kg^{-1} dry substance) of "hazardous waste."

Protocol	L/S	Comments	Ag	As	Ba	Cd	Cr	Cu	Hg	Mo	Ni	Pb	Sb	Se	Zn	Reference
[Units: mg kg^{-1}]																
DIN 38414 (1987)	10	Hospital I		0.07		0.04	0.03				0.04	0.06	0.47	0.92		Bakkali et al. (2013)
		Hospital I		0.05		0.04	0.27				1.22	0.12	0.04	1.75		
EN-14429 (2005)	10	pH 4				2.8		8			21	36			20	Gerassimidou and Komilis (2015)
		pH 10				0.2		0.1			1	4			18	
TCLP	20	BA 1		0.04		0.06	0.23	1.56			1.09	8.90			21.8	Kim et al. (2019)
		FA 1		0.92		0.38	65.1	74.3			21.7	372			527	
		BA 2		0.06		0.09	0.45	3.21			0.82	5.43			25.6	
		FA 2		0.84		0.57	82.6	66.8			35.3	421			970	
TCLP 1311	10					7.08	120.6	50.2	0.36		6.34	112				Tan and Xiao (2012)
	90					16.74	1085	225.9	4.62		570.6	439				
EN-14429	10	Ash				9.17	10.43	22.46			10.15	18.34			440.9	Sobiecka et al. (2014)
		Ash + cement 10%				4.52	31.94	33.45			18.72	26.43			131.7	
		Ash + cement 60%				7.10	19.99	27.34			13.96	21.94			303.5	
EN-12457	10	Ash				2.48	3.30	2.71				5.70			3.20	Giergiczny and Krol (2008)
Criteria A [a]	**2**			**0.4**	**30**	**0.6**	**4**	**25**	**0.05**	**5**	**5**	**5**	**0.2**	**0.3**	**25**	
Criteria A [a]	**10**			**2**	**100**	**1**	**10**	**50**	**0.2**	**10**	**10**	**10**	**0.7**	**0.5**	**50**	
Criteria B [b]	**2**			**6**	**100**	**3**	**25**	**50**	**0.5**	**20**	**20**	**25**	**2**	**4**	**90**	
Criteria B [b]	**10**			**25**	**300**	**5**	**70**	**100**	**2**	**30**	**40**	**50**	**5**	**7**	**200**	
[Units: mg L^{-1}]																
TCLP	20	BA-I		ND	0.52	ND	3.90	4.80			2.30				0.54	Zhao et al. (2010)
		BA-II		ND	1.30	ND	0.01	1.40			0.29				0.02	
		BA-III		ND	1.06	ND	4.13	1.38			0.67				0.43	
Own protocol	83	Bulk sample		ND		ND	0.9	0.04		0.06	ND	0.18		0.05	0.013	Allawzi et al. (2018)
	50	Bulk sample		ND		ND	1.0	0.06		0.08	ND	0.26		0.08	0.010	
	17	Bulk sample		ND		ND	1.5	0.21		0.18	ND	0.18		0.25	ND	
	50	180–300 μm		ND		ND	1.2	0.13		0.11	ND	0.025		0.011	0.011	
	50	300–500 μm		ND		ND	1.5	0.16		0.10	ND	0.009		0.012	0.001	
	50	500–1000 μm		ND		ND	1.4	0.18		0.10	ND	0.013		0.014	0.010	
	50	1000–2000 μm		ND		ND	0.4	0.12		0.06	ND	0.003		0.008	ND	
PN-Z-15 009	10	Ash 1						12.4				1.72			134	Gielar and Helios-Rybicka (2013)
		Ash 2						13.9				1.92			134	
TCLP 1311	50	BASH				0.01	0.19	10.56			0.05	96.92			5.24	Kougemitrou et al. (2011)
	50	FASH				0.01	0.42	1.91			0.05	0.16			19.75	
	50	BASH				0.01	0.03	0.02			0.05	0.10			0.02	
	50	FASH				0.01	0.36	1.39			0.05	89.12			14.18	
TCLP 1313	10			8		0.1		0.25	3		0.3	3.5		0.5		Patel and Devatha (2019)
TCLP	16	pH 3	1.25	3.14	35.1	0.02	3.59	42.9	ND		27.5	2.34		0.01		Rajor et al. (2012)
		pH 5	0.09	0.08	0.09	ND	1.46	36.2	ND		9.19	1.17		ND		
JLT-13	10	FA				59.3	160.4	152.6				582.0			76.3	Sukandar et al. (2009)
		FA+10%S803				ND	ND	66.4				44.4			41.5	
		FA+10%R303				Nd	15.4	55.8				115.9			18.5	
		FA+10%R201				ND	18.1	48.6				52.8			25.6	
TCLP 1311	20	FA				0.57	12.07	7.31	0.15		6.34	0.60				Tan and Xiao (2012)
		FA melted at 1080 °C				0.45	ND	1.39	ND		0.23	ND				
DIN EN-12457-2	10	FA			167	ND	22	557			20	1870			3370	Valavanidis et al. (2008)
		BA			69	ND	3	39			ND	1			36	
EN-12457-2	10	FA		ND	3.45	ND	0.14	0.07	ND	ND	ND	4.59	ND	ND	0.33	Vavva et al. (2020)
		FA+7%Ph.Ac		ND	2.6	ND	0.05	ND	ND	ND	ND	0.34	ND	ND	0.22	
		FA+10%Ph.Ac		ND	1.3	ND	0.04	ND	ND	ND	ND	0.30	ND	ND	0.24	
		FA+12%Ph.Ac		ND	1.5	ND	0.04	ND	ND	ND	ND	0.35	ND	ND	0.95	
TCLP 1311	20	FA				0.75		132			1.1	19			700	Li et al. (2020)

(*continued on next page*)

Table 6 (*continued*)

Protocol	L/S	Comments	Ag	As	Ba	Cd	Cr	Cu	Hg	Mo	Ni	Pb	Sb	Se	Zn	Reference
		FA+0.2EDTA				0.18		50			0.4	3			80	
TCLP	20	FA		0.3		6	0.05	82			7	143			2217	Xie and Zhu (2013)
TCLP	20	BA		2.2	25.2	0.04	4.25	53.70		14.73	18.54	7.25			27.3	Papamarkou et al. (2018)
		+vitrification 20%		ND	0.08	ND	0.01	0.11		0.04	0.05	0.02			0.09	
		+clinkerization		0.03	0.31	0.01	0.04	0.02		0.01	0.05	0.01			0.102	
TCLP	20	FA		3.4	125.7	0.17	8.07	41.75		24.3	1.25	532.5			125.3	Tsakalou et al. (2018)
		+vitrification		ND	0.34	ND	0.03	0.10		0.06	0.3	0.83			0.41	
USEPA	16	Ash	3.65									4.26			13.27	Racho and Jindal (2004)
TCLP [c]	20	BA	ND	2.2	2.2	0.65	3.9	4.8			2.3	1.8			873	Zhao et al. (2009)
		FA	0.39	0.38	0.38	49.7	0.44	1.89			2.2	156			1775	
USEPA	20	BA				ND	29.83	38.97			ND				13.26	Akyıldız et al. (2017)
		+50% S/S[d]				ND	5.13	ND			ND				0.65	
HJ/T299	10	Ash		0.05		1.97	0.07	0.15			0.29				8.43	Jin et al. (2010)
		+NaOH		0.05		0.02	0.02	0.03			0.02				0.38	
		+Na_2CO_3		0.20		0.07	0.02	0.16			0.02				2.29	
TCLP	20	FA				ND	0.35					6.72			19.8	Ababneh et al. (2020)
		+EDTA				ND	0.19					2.92			6.8	
TCLP	20	FA			2.53	ND					ND	0.01			0.15	Tzanakos et al. (2014)
		BA			1.80	ND					ND	6.02			11.8	
EN-12457	10	Sil. FA [e]		ND		0.001	0.148	0.005	0.3			0.072			0.054	Giergiczny and Krol (2008)
		Fl. FA [e]		ND		0.002	0.035	0.004	0.1			0.030			0.011	
TCLP [f]	20	FA			1840	17.1	85.5	1030			76.2	5216			13,200	Anastasiadou et al. (2012)
		+cement 50/50			959	5.6	70.7	550.7			54.6	524			1783	
		BA			2439	0.6	128	1550			625.8	5			119.1	
		+cement 50/50			1215	0.4	49.5	362.1			76.3	3.5			5.6	
Criteria A [g]	**2**			**0.2**	**15**	**0.3**	**2**	**12.5**	**0.025**	**2.5**	**2.5**	**2.5**	**0.1**	**0.15**	**12.5**	
Criteria A [g]	**10**			**0.2**	**10**	**0.1**	**1**	**5**	**0.02**	**1**	**1**	**1**	**0.07**	**0.05**	**5**	
Criteria B [g]	**2**			**1.5**	**50**	**1.5**	**12.5**	**25**	**0.25**	**10**	**10**	**12.5**	**1**	**2**	**45**	
Criteria B [g]	**10**			**2.5**	**30**	**0.5**	**7**	**10**	**0.2**	**3**	**4**	**5**	**0.5**	**0.7**	**20**	
USEPA TCLP [h]	**20**			**5**	**100**	**1**	**5**	**100**	**0.2**		**100**	**5**		**1**		

ND. Not determined.

[a] Additional limits include chloride (for $L/S_2 = 10{,}000$ and $L/S_{10} = 15{,}000$ mg kg^{-1}); fluoride ($L/S_2 = 60$ and $L/S_{10} = 150$ mg kg^{-1}); sulphate ($L/S_2 = 10{,}000$ and $L/S_{10} = 20{,}000$ mg kg^{-1}); dissolved organic C ($L/S_2 = 380$ and $L/S_{10} = 800$ mg kg^{-1}); TDS ($L/S_2 = 40{,}000$ and $L/S_{10} = 60{,}000$ mg kg^{-1}); total organic C = 5%; pH = 6.

[b] . Additional limits include chloride (for $L/S_2 = 17{,}000$ and $L/S_{10} = 25{,}000$ mg kg^{-1}); fluoride ($L/S_2 = 200$ and $L/S_{10} = 500$ mg kg^{-1}); sulphate ($L/S_2 = 25{,}000$ and $L/S_{10} = 50{,}000$ mg kg^{-1}); dissolved organic C ($L/S_2 = 480$ and $L/S_{10} = 1000$ mg kg^{-1}); TDS ($L/S_2 = 70{,}000$ and $L/S_{10} = 100{,}000$ mg kg^{-1}); loss on ignition = 10%; total organic C = 6%; pH = 6.

[c] .Maximum values of those reported in the cited paper.

[d] .S/S: Solidification/Stabilisation.

[e] .Siliceous FA; fluidal FA.

[f] Units in μg L^{-1}.

[g] . The values of the limits as reported by the 2003/33/EC Directive are hereby given in units of mg L^{-1} of the leachate for better comparison; they are calculated from the actual Directive limit values divided by 2 for L/S = 2 and by 10 for L/S = 10.

[h] As cited in CFR (2020).

the EU limits are impossible, although some works have wrongly attempted them. For example, Gielar and Helios-Rybicka (2013) made comparisons of leaching according to the Polish regulations with findings of other works reporting data from various other leaching protocols—but such comparisons are not feasible. In fact, the only viable way of deciding whether a certain waste complies with the EU legal requirements so that it may be disposed of to either a "nonhazardous waste landfill" or to a "hazardous waste landfill" is for a leaching experiment to be performed at L/S = 10 or at L/S = 2.

There are also some cases where leaching data are expressed in units that make any further comparisons very difficult, as in the case of Xie and Zhu (2013). They conducted leaching according to TCLP (USEPA), EN-12457 (European Union) and HJ/T (China) and expressed data as a "percentage of the extraction ratio" rather than in any form of concentration units. Also, although some works choose to perform leaching with well-established protocols, they seem to make unnecessary variations that make further comparisons impossible, such as the case with Kougemitrou et al. (2011). The latter used TCLP and EN-12457 at L/S = 50.

9. Conclusions and future research needs

Globally, medical waste management needs to be prioritised. Poor management and/or mismanagement of medical waste leads to various environmental and health-related issues, such as pollution of surface water and groundwater by untreated medical waste disposal in landfills and the spread of various diseases by viruses and microorganisms. Incineration is one of the most commonly practised solutions to reduce the massive volume of medical waste globally. However, the incineration of medical waste generates gaseous pollutants and ash as a new type of waste. Medical waste incineration generates solid residues, which release PTEs, organic compounds, and inorganic salts in the environmental matrices. Incineration is unable to control the PTEs levels, and PTEs are often released into the environmental matrices, along with ash, in more concentrated and hazardous forms. Physical or chemical stabilisation of medical waste is practised to prevent the release of PTEs into the environment. Chemical stabilisation has shown promising results in cost-effectively reducing the leaching of PTEs from medical waste. Released PTEs from the leachate of medical waste can be treated *ex-situ*. There are also widespread misconceptions about the toxicity of hazardous medical waste related to PTEs. However, under the current legal definition, no medical waste can be considered 'inert', even with minimal PTE content.

Given the current knowledge on the management of medical wastes and the fate of PTEs in medical waste incineration ash, the following research areas could be pursued.

- Effect of segregation of medical wastes on the distribution of PTEs in the ash material. For example, electronic and some plastic medical wastes are enriched with PTEs, which can be segregated to monitor the distribution of PTEs in the incinerated ash material.
- The transformation and release of PTEs during the incineration of medical wastes. While most of the PTEs are oxidised to their respective oxides during the incineration, some of the PTEs, including Cd and Hg, get volatilized, which needs to be monitored.
- Effect of solidification/stabilisation (S/S) of medical waste on the release of PTEs. The solidified/stabilised PTEs are subject to dissolution and solubilization when the ash materials are added to the soil. The release of PTEs and their subsequent leaching to groundwater sources need to be monitored.
- The plant uptake of PTEs during the soil application of medical waste ash. The uptake of PTEs by plants results in their transfer to animal and human food chains, which must be examined under various soil and climatic conditions.
- Development of a safe and sound environmentally friendly treatment technology to mitigate environmental and health impacts of PTEs in hazardous medical wastes.

CRediT author statement

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Declaration of competing interest

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

Data availability

No data was used for the research described in the article.

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