# Highlights

- Safe management of hazardous medical wastes is a major global challenge

- A significant proportion of medical waste is infectious, hazardous and radioactive

- Potentially toxic elements (PTEs) in medical waste occur mostly as oxides

- Incinerated medical waste ash application can release PTEs to environment

- Chemical stabilisation is effective in preventing PTEs release into the environment

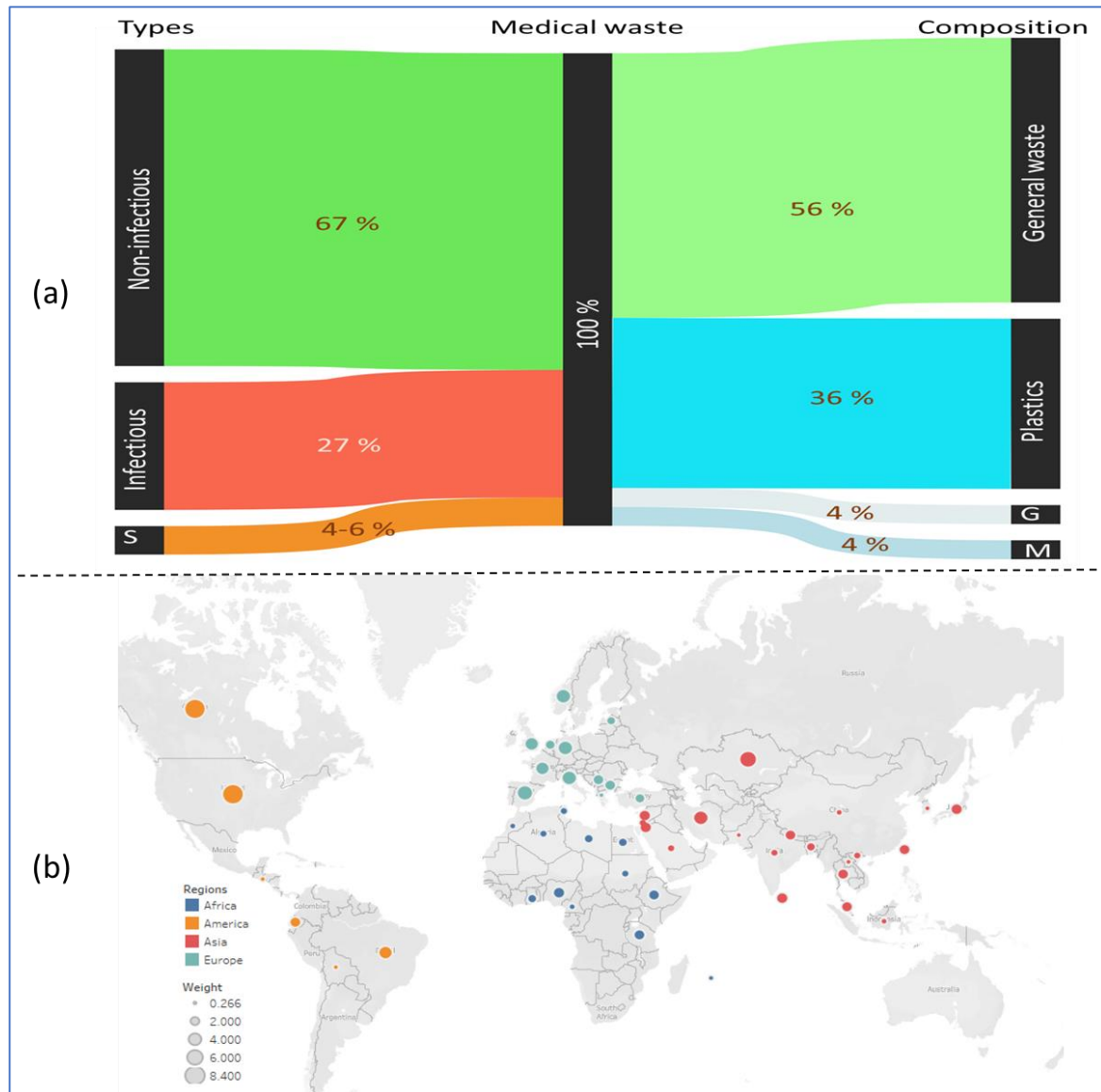Graphical Abstract
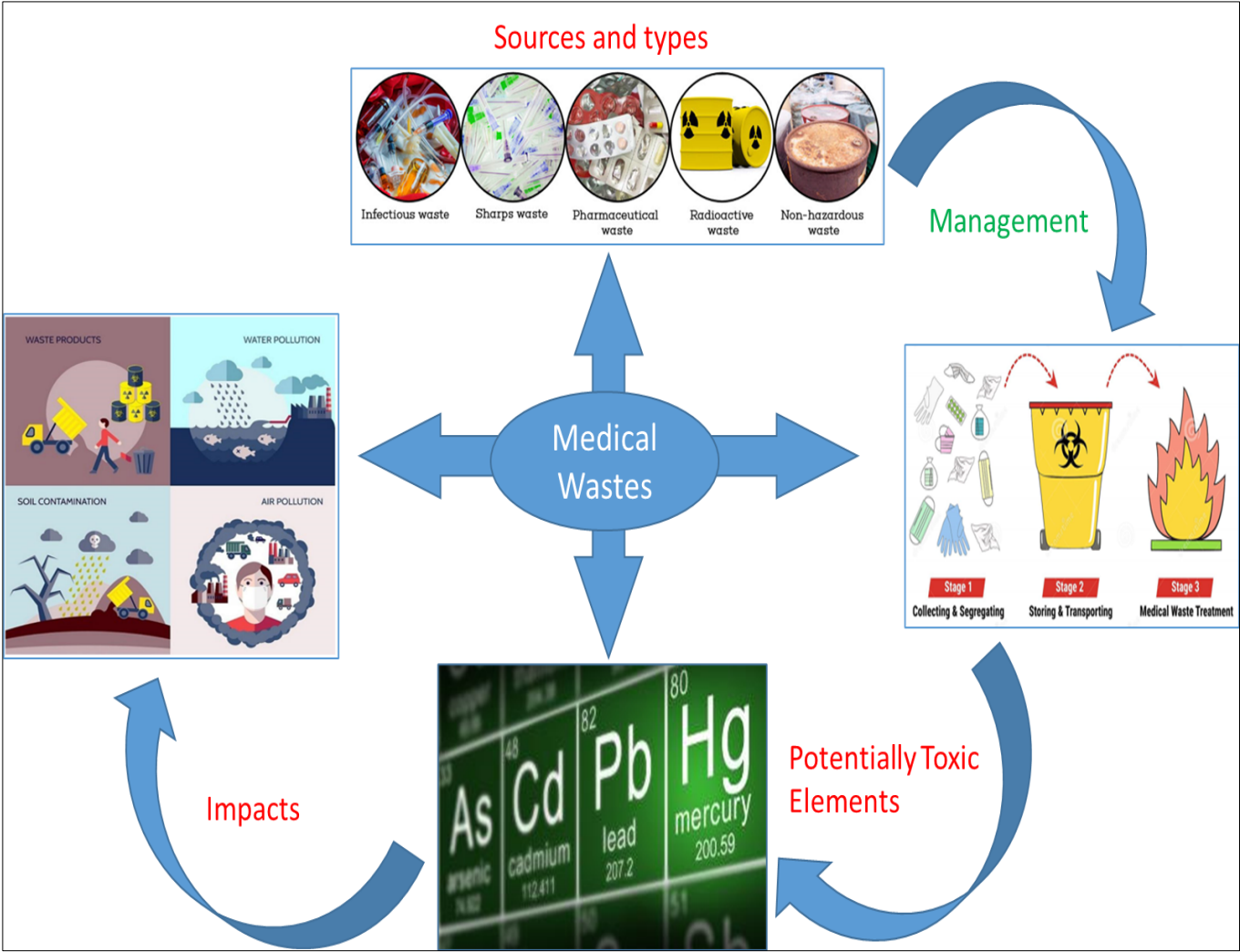
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Figure 2. Global medical waste generation and disposal scenario amid COVID-19 outbreak. (A) the medical waste generation before COVID-19 and at the peak of COVID-19 in the cities in 2020. (B) coverage of safe disposal of healthcare waste from 24 low and middle-income countries.
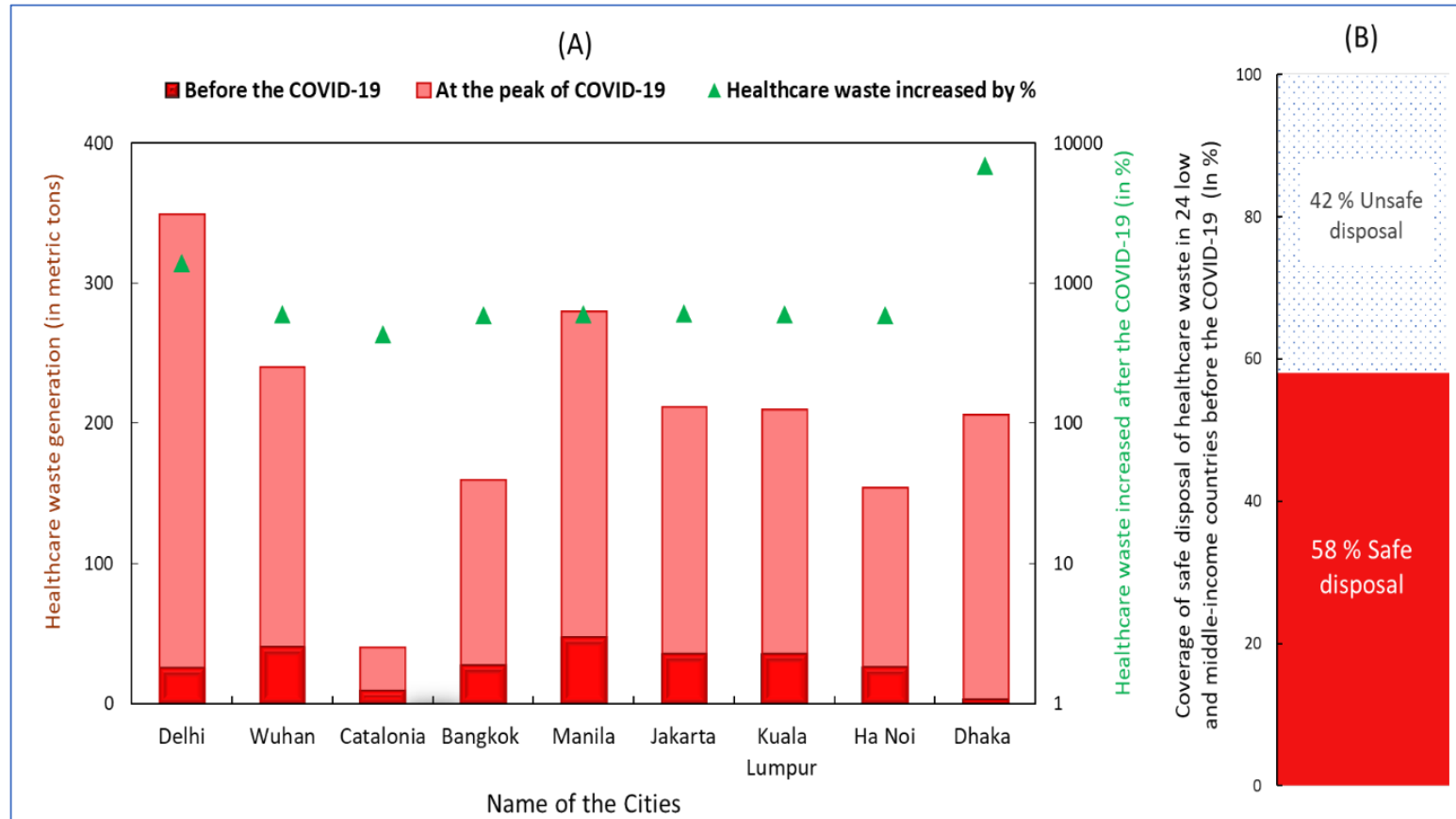
Figure 3. Schematic representation of various medical waste treatment methods.

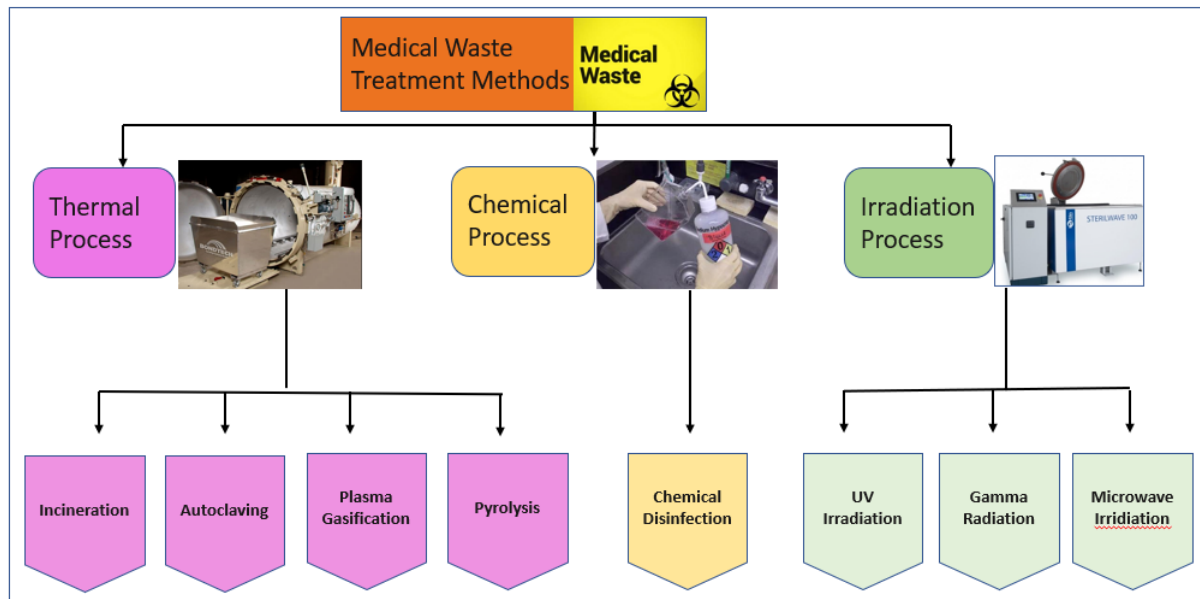**Figure 4.** Key elements for achieving a successful medical waste management system.

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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)

Source	Location											Reference
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	12700	
BA-III		27.36	27.36	2080	ND	35.77	916.50	1260			519.32	0.24	ND		375	13720	
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	Gerasimidou 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	10218				16500	
BA-I			1.65		3.38		243	435			70.8	529				2570	
FA-I			32.6		130		3860	2950			682	15639				28630	
BA	Greece		23.9	13826	0.40	30.4	8700	19500	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)

	Country															Reference
	Japan		28.89	54.38	13.56	3.86	12.65	320.22	6.18	7.91	19.96	439.74		1514	1831.5	Sukandas 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			24252	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 [c]			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 [d]	China	24.2	24.2	5,100	10.2	49.9	895	2,300				2,100	197	406	30,700	Zhao et al. (2009)
FA [d]		222	237	2,900	635	238	264	2,900				5,400	230	1660	121,000	
FA	Turkey	140	20	5,710			810	2,840			150	320	250	240	2030	Akyildiz et al. (2017)
Ash [b]	Poland				200	100	100	300			100	200			5,800	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

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");

b. As oxides;

c. obtained from 6 papers;

d. obtained from 14 samples (BA) and from 8 samples (FA)—the maximum reported values are cited here;

e

ND: Non-detected.

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

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 - 1,200 °C	• Reduce the volume of materials • Destroy pathogens and hazardous organic substances • Reduce transport costs	• Emit trace amounts of human carcinogenic by-products (polychlorinated dioxins 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 Katocha, 2012; Windfeld and Brooks, 2015; Xu et al., 2020; Zafar, 2019)
Hydroclaving	Cytotoxic, pathological or several other	Waste is subject to indirect heating by applying steam	• Weight and volume reductions are up -to 70% and 85%	• Required specialized and technical skills to operate	(Babanyara et al., 2013; JessMathew et al., 2017; Kenny

			• Absence of harmful emissions • No chemical use	• Similar limitations to autoclaving	and Priyadarshini, 2021)
Gas sterilization	Infectious waste	Waste is kept in an evacuated tight-air chamber, and applying sterilization 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 ℃ 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 Katocha, 2012; Xu et al., 2020
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 |

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-Gmouh (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

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 categorized 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	Gerasimidou 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+cement10%				4.52	31.94	33.45			18.72	26.43			131.7	
		Ash+cement60%				7.10	19.99	27.34			13.96	21.94			303.5	

Method		Sample														Reference
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		

Method	L/S	Sample														Reference
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 ^{0}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)
		FA+0.2EDTA				0.18		50			0.4	3			80	
TCLP	20	FA	0.3			6	0.05	82			7	143			2,217	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)
		+vitrification20%	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. (2008)
		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	Akyildiz 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$_2$CO$_3$		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	5,216			13,200	Anastasiadou et al. (2012)
		+cement 50/50			959	5.6	70.7	550.7		54.6	524			1783	
		BA			2,439	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**		

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/Stabilization.

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).

ND. Not determined.

1   **Review on distribution, fate, and management of potentially toxic elements in incinerated**

2   **medical wastes**

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Abstract

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.

Keywords: Medical waste; Potentially toxic elements (PTEs); Incineration; Organic pollutants; Medical waste management

1. Introduction

Several analogous phrases have been applied in the published literature to define medical waste, such as 'health care waste' and 'hospital waste', and medical waste poses a major global health care issue (WHO, 2020). 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

92 regarded as nonhazardous, whereas 15-25% can be toxic, infectious or radioactive (Padmanabhan

93 and Barik, 2019). The potential risks of medical wastes are diverse. They may contain infectious

94 materials such as blood and tissues that can transmit diseases to working staff, other patients, and

95 the public in general (Das et al., 2021; Borowy, 2020). Sharp objects, including needles and

96 surgery knives, can cause injuries to people handling medical waste. Incineration of medical

97 wastes can emit gaseous contaminants (dioxins, furans) and leave behind residual toxic elements

98 in the environment (Rathna et al., 2018). Cytotoxic waste, including drugs used to treat cancer,

99 rheumatoid arthritis, and multiple sclerosis, can exhibit teratogenic and/or carcinogenic and

100 mutagenic properties. Radioactive waste derived mainly from cancer treatment can also impose

101 short- and long-term health impacts (Miller et al., 2019).

102 Medical wastes contain a range of highly hazardous organic and inorganic compounds,

103 including persistent organic pollutants (POPs) such as endocrine disturbing chemicals (EDCs)

104 (Rathna et al., 2018). Inorganic compounds of concern are mostly heavy metals and metalloids

105 and are collectively termed 'potentially toxic elements' (PTEs) (Pourret and Hursthouse, 2019).

106 The term 'Potentially toxic elements (PTEs)' in general includes metals and metalloids

107 (metal(loids) that are both biologically essential [e.g., cobalt (Co), copper (Cu), chromium (Cr),

108 manganese (Mn) and zinc (Zn)] and non-essential [e.g., Arsenic (As), cadmium (Cd), lead (Pb)

109 and mercury (Hg)] elements. The essential elements (for plant, animal or human nutrition) are

110 required in low concentrations and hence are known as 'trace elements' or 'micro nutrients'. The

111 non-essential elements are phytotoxic and/or zootoxic and are widely known as 'toxic elements'.

112 Both groups are toxic to plants, animals and/or humans at extremely high concentrations (Bolan

113 et al., 2014).

114 PTEs are mostly present in medical facilities' plastic, syringes, rubber, adhesive plaster,

115 and battery wastes (Annapoorani, 2021). For example, the pigment of coloured plastics contains

116 PTEs such as cadmium (Cd), cobalt (Co), chromium (Cr), lead (Pb), and copper (Cu) (Adama et

117 al., 2016; Sridharan et al., 2022). Also, a larger quantity of Cr and zinc (Zn) in medical wastes

118 come from syringes' disposal (Honest et al., 2020). PTEs in medical wastes typically occur in the

119 form of metal(loid) elements, metal(loid) oxides, volatile metal(loid)ic sulfates and chlorides.

120   To maintain the occupational health and safety of health workers, patients, and the general

121 public and reduce the environmental impacts from the unintentional exposure of biological and

122 chemical hazards from medical wastes, including drug-resistant microorganisms (Zhou et al.,

123 2022), safe and environmentally sound management approaches are needed. Medical wastes are

124 generally treated through either sterilisation using autoclaves or incineration (Dharmaraj et al.,

125 2021). Incineration is currently the most common practice and technology adopted for the safe

126 management and disposal of medical waste. The incineration of medical waste primarily aims to

127 eliminate deadly pathogens in the waste (Khobragade, 2019). However, incineration should also

128 destroy bioactive organically-based pharmaceuticals. The incinerated ash materials from

129 hazardous medical wastes are generally disposed of in landfills after the S/S process (Suryawan et

130 al., 2019). The process of stabilisation is generally applied to reduce the toxicity and solubility of

131 the potential pollutants, while in the process of solidification, a cementing material is applied to

132 reduce the leachability of the contaminants by immobilising them (Bolan et al., 2022; Kumar et

133 al., 2021; Bolan et al., 2014).

134   The incineration of medical wastes and the subsequent disposal of the incinerated ash affect

135 the speciation, redistribution, mobility, and bioavailability of PTEs. For example, the incineration-

136 induced oxidation of most PTEs, such as Pb and Cd, in medical wastes results in their precipitation

137 as metal oxides, thereby decreasing their solubility and bioavailability (Liu et al., 2018). Oxidation

138 of other PTEs, such as Cr and Sb, results in the alteration of their speciation (Cr to Cr(VI) and Sb

139 to Sb(V)), thereby increasing their solubility, mobility and bioavailability (Bolan et al., 2022;

140 Adama et al., 2016; Choppala et al., 2013). Several investigations have examined the distribution,

141 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., 2004; 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,

166   quantitative analysis is useful in undertaking descriptive review. In this study, we undertook a

167   quantitative analysis of literature data.

168   Relevant literature was searched through the databases of Web of Science, ScienceDirect,

169   Scopus, Google Scholar, and other various sources by applying the keywords: 'Medical wastes',

170   'Infectious waste', 'Hazardous waste', 'radioactive waste', 'Incineration', 'Contaminants',

171   'Potentially toxic elements', 'Landfill', 'Leaching', 'Bioavailability', and 'Ecotoxicity'. This

172   review aims to provide the-current-state-of-the-art research on medical waste management and fill

173   the knowledge gaps related to the environmental fate of PTEs in medical wastes.

174   This critical review will help develop a sustainable approach for handling PTE pollution

175   of the environment caused by medical waste. This information is crucial for the development of

176   strategies that limit the environmental toxicity of medical-waste-derived PTEs to humans and the

177   ecosystem. This review provides a framework for identifying priority research areas to address the

178   existing knowledge gaps about medical waste and its effective management. Furthermore,

179   improved knowledge about medical-waste-derived PTE dynamics in liquid and solid matrices will

180   benefit the medical sectors and industries by enhancing public awareness about medical waste

181   disposal. Future research priorities in the sustainable management of contamination resulting from

182   medical wastes in these settings are also proposed.

183   **3. Overview and toxic elements of medical waste**

184   *3.1 Characteristics of medical waste*

185   The waste generated in hospitals, clinics, and health care institutions or waste streams

186   contaminated by medical procedures that produce human secretion or human contact waste,

187   including blood, body tissue and fluids, is considered medical waste. Medical waste can form part

188   of the infectious pathways to animals and humans. A significant component of medical waste can

189   be biologically toxic and corrosive in nature, and when improperly handled, it causes air, water,

190   and soil pollution. It can also cause direct harm to humans through exposure or the release of toxins

191   (Singh et al., 2021; Zhang et al., 2021). For example, Ephraim et al. (2013) examined the effect of

192   burnt hospital wastes on the agricultural soil around a farm settlement near Obafemi Awolowo

193   University Teaching Hospital Complex, Ile-Ife, South-West Nigeria. The presence of elevated

194   concentrations of Pb (77.8 - 279.6 mg kg^{-1}) and Cl (102.2 - 167.2 mg kg^{-1}) was of serious health

195   concern because of the intensive agricultural practices for food production in the neighbourhoods

196   of the study sites. The composition of medical waste is a mixture of diverse types of medical

197   discards, which varies greatly in different medical settings and locations of origin (Rani and

198   Rampal, 2019). The quantity of medical waste in different hospitals and countries is very much

199   correlated to socioeconomic factors such as the human development index, life expectancy, and

200   health care expenditure per capita of the region and countries (Minoglou et al., 2017; Singh et al.,

201   2021).

202   Technological advancements in social networks, transportation, and occupation have

203   fuelled global economic growth resulting in the expansion of health care systems and a

204   concomitant increase in demand for medical equipment and supplies. Thus, an increased standard

205   of living in many countries has also played a key role in medical waste generation (Bloom et al.,

206   2018; Singh et al., 2021). **Figure 1b** shows the features of medical waste and the weight of medical

207   waste generation (kg bed^{-1} day^{-1}) worldwide and reveals that medical waste generated by high-

208   income countries is significantly greater than that of low- and middle-income nations. However,

209   the environmental impact of medical waste may be worse in developing countries despite

210   producing less waste, which is attributed to poor management of the generated waste (EPI, 2020;

211   Singh et al., 2021).

212   Typically, medical waste is classified by the nature of the contamination, namely infectious

213   waste, pathological waste, sharps waste, chemical waste, pharmaceutical waste, cytotoxic waste,

214   radioactive waste, and general waste. Nonetheless, medical waste is categorically divided into

215   three major types that include: 1) infectious or hazardous waste - waste containing pathological

8

216   germs and toxins, 2) non-infectious or general waste - a mixture of discarded food and non-

217   contaminated materials, and 3) sharps waste - waste containing metal and glass (**Figure 1a**)

218   (WHO, 2018b).

219   Countries vary significantly in disposal practices, and many countries have not established

220   safe disposal practices. Medical waste in many municipalities is combined with general waste. As

221   such, it is usually treated by open landfill relocation or incineration (Singh et al., 2020a; Singh et

222   al., 2020b). A recent study based on the meta-analysis of medical and health care waste

223   management practices in 78 countries has revealed that about 56% of medical waste contained

224   general waste such as food, liquids, and paper (Singh et al., 2021). Plastic content is the second

225   most abundant material, with about 36% of the total medical waste. Glass and metal waste

226   contribute 4% each of the total waste generated globally (Singh et al., 2021).

227   *3.2 Sources of potentially toxic elements and other pollutants*

228   Medical waste disposal releases PTEs and other secondary pollutants, such as particulate

229   matter, SO_2, NO_X, HCl, dioxins, and polycyclic aromatic hydrocarbons (PAHs), into the

230   environment during their disposal or treatment (Liu et al., 2018). Incineration and engineered

231   sanitary landfills are the most common treatment techniques for medical waste management

232   worldwide. Other techniques that have been developed in recent years include pyrolysis, chemical

233   disinfection, high-pressure steam sterilisation, plasma disinfection, and electromagnetic wave

234   disinfection (Capoor and Bhowmik, 2017; Diaz et al., 2005; Xie et al., 2009).

235   Most secondary pollutants, such as PTEs and organic chemical compounds, are released

236   from the stack gases of incinerators to the atmosphere during the incineration process. During the

237   incineration process, some are transformed into volatile metallic vapours and enter the flue gas.

238   The remaining majority settle down at the bottom as ash (Li et al., 2017; Liu et al., 2018; Wu et

239   al., 2014), which contains a significant part of PTEs, except for the readily volatile PTEs such as

240 Cd and Hg (Alvim-Ferraz and Afonso, 2003; Krivanek III, 1996; Singh and Prakash, 2007;

241 Wheatley and Sadhra, 2004).

242 While PTEs are emitted from all kinds of incineration techniques, including controlled air,

243 rotary kiln and liquid-injection furnaces, the emission quantity depends on temperature and

244 retention time in the incineration compartment, as well as the control measures aimed at capturing

245 emissions from incinerators. The most commonly known PTEs associated with medical waste are

246 Pb, Cd, Hg, Cr, arsenic (As), nickel (Ni), Zn, and barium (Ba) (Anastasiadou et al., 2012; Wang

247 et al., 2012; Zhao et al., 2010). Typically, most of the PTEs in medical waste result from sharps

248 materials, including needles, expired medicines containing PTEs, and discarded medical

249 equipment, including waste batteries, plastics, rubbers, and adhesive plasters (Chen et al., 2013;

250 Duan et al., 2008; Padmanabhan and Barik, 2019; Zhao et al., 2009).

251 PTEs in medical waste are typically present in the form of metal oxides (e.g., lead oxide),

252 metal elements (e.g., copper element), volatile metallic chlorides (e.g., cadmium chloride), and

253 sulfates (e.g., copper sulphate). The content of PTEs in the bottom ash of incinerated medical waste

254 and the average values are shown in **Tables 1 and 2**. PTE concentrations vary considerably

255 depending on different incineration techniques (Table 1, regarding bottom ash) and the type of ash

256 produced (Table 2, regarding bottom or fly ash).

257 The average chemical composition of incinerated fly ash of medical waste and municipal

258 solid waste (MSW) is shown in **Table 3**. The content of chlorines in medical waste comes from

259 the high consumption of chlorinated plastics. Chlorine under high temperatures, especially during

260 the incineration process, complexes with the PTEs to form PTE chlorides in the flue, and these

261 PTE chlorides are effective catalysts in the formation of dioxins (Chen et al., 2013; Gunes et al.,

262 2015; Hsu et al., 2016; Yoon et al., 2017; Wang et al., 2003). Dioxin is a human carcinogen

263 associated with a range of adverse health effects, and its persistence in the environment has

264 recently attracted considerable attention (Kanan and Samara, 2018; Marinkovic et al., 2010; Sany

265 et al., 2015).

4. Environmental and health impacts of medical wastes

4.1 Health risks

268   Health care waste contains potential toxins and microorganisms that can be a source of

269 transmission of diseases and contamination of hazardous chemicals if managed improperly and

270 can infect people in health care settings. According to the WHO report, about 15% of the total

271 generated health care waste is considered hazardous, and the remaining 85% is general or

272 nonhazardous waste (WHO, 2018b). The generation of hazardous medical waste in high-income

273 countries accounts for an average of up to 0.5 kg hospital bed^{-1} day^{-1}, while in low-income

274 countries, it is up to 0.2 kg hospital bed^{-1} day^{-1} (WHO, 2018a). However, contrary to

275 recommendations from the WHO, health care waste segregation is not practised in many low and

276 middle-income countries. Hence, the toxins from medical waste, discarded from hospitals and

277 sometimes from household and quarantine centres, get mixed with other nonhazardous waste for

278 disposal via landfill or incineration (Singh et al., 2020b).

279   Potential risks to human health are associated with unsound management of medical waste,

280 including exposure to pathogens, sharps-inflicted injuries, chemical exposure to medicines,

281 especially the disposal of expired medicines, thermal injuries through open burning, and radiation

282 exposure. Many studies on health risks from health care waste have shown that nearly 5.2 million

283 people lose their lives annually due to the unsafe disposal of medical waste worldwide (WHO,

284 2018a). Additionally, in many developing countries, millions of workers involved in medical

285 waste management may become infected with many infectious diseases, such as HIV, hepatitis B,

286 and hepatitis C infections, due to the contagion of the waste (Singh et al., 2021). In 2015, a joint

287 WHO/UNICEF assessment found that around 58% of sampled facilities from 24 countries had

288 adequate systems in place for the safe disposal of health care waste WHO/UNICEF (2015).

11

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 as 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).

338   An adequate disinfection process could play an important role in an effective and safe

339   medical waste disposal system, especially in developing countries which practise unsafe

340   incineration and open burning. The combination of chemical disinfection and incineration could

341   play a vital role in the safe disposal of rapidly growing health care waste. However, chemical

342   infection using chlorine-based disinfectants followed by incineration may result in the release of

343   dioxins. Hence, care must be taken in choosing the appropriate disinfectants. Medical waste should

344   be managed comprehensively using existing infectious and health care waste management

345   protocols recommended by national and international guidelines and manuals (UN, 2020). Proper

346   segregation, packaging, and storage of potentially contaminated materials in the double bag for at

347   least 72 hours, adequate waste collection bins, and training and capacity building of the staff

348   involved in healthcare waste management are essential for properly handling medical waste.

349   *4.3 Increased Challenges due to the Covid-19 pandemic outbreak*

350   In 2020, the severe acute respiratory syndrome coronavirus 2 (COVID-19) pandemic spread

351   worldwide. Millions of people have been infected since then, including some of the world's most

352   isolated communities living in isolated islands where health care waste management does not exist.

353   Almost 90% of the infected cases of COVID-19 were in urban areas, considered the epicentre of

354   the COVID-19 spread (United Nations Human Settlements Programme (UNHSP), 2020).

355   Inadequate health care waste management in urban areas during the virus outbreak has been

356   reported in many countries. Hence, the pandemic resulted in an increased risk of disease

357   transmission either directly through open-dumped health care waste or indirectly through food-

358   chain contamination (Singh et al., 2020a; You et al., 2020). For instance, it has been calculated

359   that inappropriate disposal of medical waste may have caused up to 30% of hepatitis B, 13% of

360   hepatitis C, and 0.3% of HIV rates to be directly transmitted from patients to healthcare

361   professionals (Singh et al., 2020a). Studies from Pakistan, Greece, Brazil, Iran, and India

362   demonstrate that pathogens in contaminated wastes are directly responsible for a higher-than-

363 average prevalence of viral infection in solid waste collectors (Mallapur, 2020). Likewise,

364 exposure to PTEs indirectly through the food chain (soil-plant-animal-human or soil-plant-human)

365 can cause chronic diseases, such as cancer, in a population (Bolan et al., 2022). Due to their high

366 chemical stability, poor biodegradation (decomposition) capability, and high toxicity potential,

367 PTEs have a prolonged residence time in the food chain and the environment (Omeka and Igwe,

368 2021).

369   The global health care waste management system, which already poses a risk to billions of

370 people from occupational, public health, or environmental perspective, is now under

371 unprecedented burdens due to the surge in COVID-19-related healthcare waste generation (**Figure**

372 **2A**) (Singh et al., 2021). Biomedical waste generation has exceeded waste management capacity

373 in many cities, particularly in the middle- and low-income countries. For example, in Dhaka, the

374 capital of Bangladesh, biomedical waste has increased nearly 70 times, from approximately 3 tons

375 per day before the pandemic to 206 tons per day in 2020 (Rahman et al., 2020). In Delhi, infectious

376 medical waste has increased by nearly 14 times compared to pre-pandemic levels, from

377 approximately 25 tons per day in May to 371 tons per day in June 2020 (Reddy, 2020). In East-

378 Asian cities, including Bangkok, Manila, Jakarta, Ha Noi, Kuala Lumpur, and Wuhan, biomedical

379 waste has increased by approximately six times since the COVID-19 outbreak (ADB, 2020). The

380 increased biomedical waste means increased risk to its exposure through PTEs and other

381 pollutants.

382   Additionally, the transmissions of the COVID-19 virus are not solely through interaction

383 with infected persons but also can spread rapidly via discarded PPEs, including face masks and

384 other discarded protective materials. Studies conducted in 26 countries belonging to the World

385 Health South-East Asia Region Organisation (SEARO) group show that only 58% of sampled

386 facilities had adequate disposal for health care waste (**Figure 2B**) (WHO, 2015). The SEARO

387 region countries, including Bangladesh, Bhutan, India, Nepal, Sri Lanka, and Timor-Leste, showed

388 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

391 Improper medical waste management may risk health care staff, exposed populations, and the

392 natural environment. There is a large body of literature on the issues and ways of achieving

393 efficient medical waste management (Abdulla et al., 2008; Lee et al., 2004; Patil and Shekdar,

394 2001; Jang et al., 2006). The techniques employed by hospitals in northern Jordan for managing

395 medical waste were examined by Abdulla et al. (2008). This study found that incineration was the

396 most popular method of treating solid medical waste, although source segregation and treatment

397 of liquid medical waste are still lacking (Abdulla et al., 2008). Previously, Lee et al. (2004) studied

398 and reported that on-site integrated microwave and incineration is the most cost-effective method

399 of regulated medical waste treatment in comparison to microwave and incineration alone. Patil

400 and Shekdar (2001) studied and provided a detail discussion on medical waste generation and

401 management practices in India. The waste-management strategy for medical waste proposed by

402 their study involved institutional arrangements, suitable technology, operational planning,

403 financial management, and the development of suitable staff training programs. Similarly, Jang et

404 al. (2006) studied and provided an overview of the management practices of medical waste in

405 Korea. Medical waste incineration is identified as the most preferred disposal method.

406 Nevertheless, the main problems in the future are thought to be waste minimisation and recycling

407 and control of harmful air emissions during incineration.

408 The disposal and treatment of medical waste depend on waste characteristics, volume, costs

409 associated with treatment, and environmental and safety factors (Eslami et al., 2017). Medical

410 waste treatment and disposal processes are categorised into three main processes: thermal,

411 chemical, and irradiation (**Figure 3**). High temperature is used in thermal processes to reduce the

412 waste volume and weight by converting it to inorganic and incombustible matter. This process

413 treats waste that cannot be recycled, reused, or disposed of in a landfill site (Xu et al., 2020).

414 Chemical treatment is used to kill or inactivate pathogens in medical waste. For chemical

415 treatment, disinfection is more commonly practised than sterilisation. Chemical disinfection is

416 used for treating liquid waste, such as urine, faecal matter, blood, or hospital sewage (Kenny and

417 Priyadarshini, 2021). In some cases, Gamma radiation is used to irradiate medical waste. However,

418 irradiation involves stringent regulatory requirements and approvals. The associated costs and the

419 strict regulatory protocols hinder its usage (Kenny and Priyadarshini, 2021; Xu et al., 2020).

420 The incineration technologies are the most advantageous due to the significant reduction

421 in the overall volume and weight of medical waste. Incineration is also suitable for all kinds of

422 medical waste and can be implemented at a very large scale. However, the chemical disinfection

423 process only requires modest and convenient instruments and operation, involving a small one-

424 time investment and lower operational costs. A comparison of different processes, their usage,

425 technical specifications, advantages, and limitations are detailed in **Table 4**.

426 *5.2 Critical elements in medical waste treatment*

427 Medical waste management issues are intensifying due to the lack of public awareness,

428 proper waste disposal practices, and financial resources (Birpınar et al., 2009; Marinković et al.,

429 2008). Some of the critical elements in improving the issues of medical waste management are

430 shown in **Figure 4.** Establishing rules and regulations for waste separation and disposal in the

431 national or institutional framework would lead to improved waste management practices

432 (Bokhoree et al., 2014; Taghipour and Mosaferi, 2009). Lack of awareness can be minimised

433 through proper training of all the concerned personnel. Appropriate training is essential to develop

434 a cognisance related to medical waste and its adverse impacts on the health of individuals and the

435 environment (Bokhoree et al., 2014). Allocation of funds/resources to such training programmes

436 would improve medical waste management efficiency. Establishing a database, including

17

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 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

462 temperatures, a decrease in the need for off-gas treatment, and the ability to have a reduced

463 installation area and cost are all benefits of thermal plasma technology. Therefore, the on-site and

464 off-site treatment of medical waste is thus a promising solution (Cai and Du, 2021). Furthermore,

465 to minimise the problems associated with the improper transportation and management of medical

466 waste, a fixed timetable for transport should be prepared, and superior-quality, safe packaging

467 containers should be used to transport medical waste (Abdulla et al., 2008; Yong et al., 2009).

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

469 Lands outside hospital waste incineration plants are likely to be affected by emitted PTEs.

470 However, studies that report data from such cases are extremely scarce. A notable reason may be

471 that safety protocols, intense site monitoring and law enforcement of such areas make any leakage

472 reporting very unlikely. However, some research has shown that if fumes are emitted, PTE

473 enrichment in soils of the vicinity is indeed likely. For example, in Brazil, an area at a distance of

474 up to 500 m from a hospital waste incineration plant was studied from 2000 to 2003 (Segura-

475 Munoz et al. 2004). The plant was deactivated in 2002. It was found that Cd, Mn, and Pb decreased

476 after the incineration plant deactivation. However, the measured elements' concentrations were

477 unacceptably high for Cd and Mn. This was due to the fact that the initial emitted concentrations

478 of Mn and Cd were considerably high, and this resulted in their long-time persistence in soil. In

479 Ghana, a similar study was conducted by Adama et al. (2016) in an area outside a medical waste

480 incineration facility. Soils were reported to be highly elevated with a variety of PTEs (Hg=0.88;

481 Zn=16,418; Cd=7.54mg kg^{-1}), with a clear decreasing trend with distance. The pollution load index

482 also was higher than 20 at a distance of 20 m from the facility. It decreased to less than unity at

483 120 m, a trend showing the significant enrichment caused by facility-emitted PTEs.

484 **7. Transformation of potentially toxic elements in medical waste ash**

485 The incineration of wastes, including medical wastes, has been extensively practised

486 globally. An inefficient incineration process or the incineration of inappropriate wastes leads to

487   the emission of pollutants into the atmosphere. It can also generate high ash residue. Incineration

488   of medical waste having high PTE content, especially Pb, Hg, and Cd, can lead to the release of

489   these PTEs in the environmental matrices. **Table 2** shows the total content of trace elements in

490   medical incineration waste (bottom or fly ash). The suitability of the incineration wastes is

491   evaluated with leaching test protocols; various such protocols have been developed in various

492   major countries or organizations and are shown in **Table 5**.

493   *7.1 Stabilisation of potentially toxic elements in medical waste ash*

494   Most solid waste treatments, including medical waste ash, aim to decrease the leachability

495   of PTEs so that the wastes may have PTE concentration in the leachates below the maximum

496   limits. The procedures are either chemical, where some binding agents are applied so that wastes

497   containing PTEs are chemically stabilised, or physical/mechanical, where a solidification agent is

498   applied. An overview of these methods is discussed below.

499   Chemical stabilisation methods, as the name suggests, use chemical agents. Phosphoric

500   acid is one of the most widely used chemical binding agents. For example, Sukandar et al. (2009)

501   used acidic (pH 1-2; Ashnite R303) and alkaline (pH 9.3-10.3; Ashnite R201) phosphoric acids

502   for waste stabilisation. The binding mechanisms identified by the authors were as follows: with

503   the application of Ashnite R201, Pb-hydroxyapatite precipitate is formed, leading to the removal

504   of Pb from the soil solution. Also, Cu, Cd, Zn, Mn and Ni can be precipitated at high pH in the

505   presence of phosphorus as phosphate-metal-hydroxy complexes. With Ashnite R303,

506   chloropyromorphite $Pb_5(PO_4)_3Cl$ is likely formed, a highly insoluble complex that keeps Pb out of

507   the solution. Likewise, Vavva et al. (2020) treated incinerated medical waste ash with 7% of

508   phosphoric acid and found that Pb in the leachate decreased significantly from 4.59 to 0.30-0.35mg

509   L^{-1}, while there was no effect of phosphoric acid treatment on the leaching of other elements,

510   including Cd and Cu.

511   Another chemical stabilisation process was reported by Jin et al. (2010) using alkaline

512   agents. In this study, 20% w/w of NaOH and Na_2CO_3 were mixed with medical ash. The retention

513   mechanism was presumably the formation of insoluble metal hydroxides and metal carbonates.

514   The authors found that As decreased by 51%, Pb by 58%, Sr by 97%, and Zn by 87% by the

515   addition of NaOH, and 66% (As), 58% (Pb), 84% (Sr), and 82% (Zn) by addition of Na_2CO_3,

516   compared to the untreated ash. It was also reported that the stabilisation failed for Cr, Cu, Cd, and

517   Ni, the regulated metals, and Co and vanadium (V), the non-regulated metals.

518   As for the mechanical or physical stabilisation methods, they include melting at high

519   temperatures (Tan and Xiao, 2012), vitrification (Tsakalou et al., 2018), clinkerization

520   (Papamarkou et al., 2018), and S/S (Ababneh et al., 2020). However, those can be cost-prohibitive

521   or difficult to implement on a large scale.

522   *7.2 Release of potentially toxic elements from medical waste incineration fly ash (MWIFA)*

523   Some treatment methods aim to liberate PTEs from the wastes so that they may be cleaned

524   before applying a formal leaching protocol. In this way, leaching reduces PTEs to acceptable

525   concentrations. The literature mainly includes reports on the use of ethylenediaminetetraacetic acid

526   (EDTA) as a releasing agent of PTEs from MWIFA. For example, in the study by Li et al. (2020),

527   a proper circulating system was established to remediate PTEs in MWIFA. MWIFA-column

528   studies were accomplished to remediate Cd, Zn, Pb, Ni, and Cu from MWIFA by applying EDTA

529   disodium (Na_2-EDTA). Column studies were performed to investigate the remediation efficiency

530   of zero-valent iron (Fe(0)) on the five PTEs from MWIFA washing water. The Toxicity

531   Characteristic Leaching Procedure (TCLP) test was opted to assess PTEs toxicity of MWIFA

532   residues generated after waste ashes were treated with EDTA at a range of molarities of 0–0.2 mol

533   L^{-1}. After being washed with 0.2 mol L^{-1} Na_2-EDTA solution, the ashes having PTEs

534   (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

535   kg^{-1}) were subjected to leaching with the TCLP protocol: PTEs were decreased to Cu=58.4 ± 2.0,

536 Pb=2.81 ± 0.14, Zn=64.3 ± 4.0, Cd=0.156 ± 0.005, and Ni=0.381 ± 0.006mg L^{-1}. The effluents

537 from the EDTA treatment were driven through a Fe(0) column so that the ash-liberated PTEs may

538 be stabilised. Indeed, they were efficiently reduced by 99.7% (Cu), 91.6% (Pb), 91.6% (Zn), 75.4%

539 (Cd), and 75.7% (Ni). Also, the EDTA solution was reused for further ash treatment up to 4 times.

540 The extraction efficiency of Pb and Cd (removal efficiency at different cycles divided by removal

541 efficiency of new Na_2-EDTA) declined toward 80%. Similar work has been conducted by Ababneh

542 et al. (2020). They applied EDTA and $Na_2S.9H_2O$ as releasing agents at a 10:1 solution-to-solid

543 ratio with 0.2 M EDTA. The treatment decreased PTEs from the ashes by 100% (for Cd), 74%

544 (Pb), and 86% (Zn). $Na_2S.9H_2O$ was much less effective, as it only decreased Pb by 37% and Zn

545 by 7%, while it completely failed to decrease Cd concentration.

546 **8. Three frequently encountered misconceptions**

547 *8.1 Only some medical waste incinerated residues are hazardous*

548 The source of this misconception is the fact that the European Union has a different

549 approach to that of the United States Environmental Protection Agency (USEPA) in America.

550 Medicine-derived wastes are categorised only as "hazardous wastes" in the European Union

551 legislation irrespective of their content in 91/689/EC Directive (EC, 1991)—no other

552 characterization is allowed. It needs clarification in the sense that there is a widespread

553 misconception in the literature that medicinal waste becomes hazardous only if some PTEs are

554 found at high concentrations, a concept accepted by USEPA. For example, there are works that

555 categorise MWIFA as "nonhazardous" or even as "inert," according to findings concerning their

556 PTE leachability. According to the 1999/31/EC Directive, p. L182/3 (EC, 1999), "inert" is a waste

557 if it "does not experience any substantial physical, chemical and biological alterations". Thus, no

558 medical waste can be under this category, even with minimal PTE content.

559 Along these lines, Kougemitrou et al. (2011) claimed that the studied MWI ash was "inert"

560 according to the European leaching protocol because leached concentrations of PTEs were low.

22

561 Also, Tsakalou et al. (2018) and Papamarkou et al. (2018) compared their leaching rate of PTEs

562 with "inert" and "hazardous" waste limits, while Ababneh et al. (2020) compared their leaching

563 with limits for nonhazardous waste, implying that the waste is nonhazardous. In some other cases,

564 the misconception may well be a matter of imprecise use of terms, as in the distinction between

565 "hazardous" and "medical" wastes (Racho and Jindal, 2004), as well as "dangerous" and

566 "hazardous" (Bakkali et al., 2013). Hazardous wastes are permitted to be disposed of in landfills,

567 similar to nonhazardous waste if leaching tests satisfy specific criteria of maximum limits. These

568 are listed in the 2003/33/EC Directive (EC, 2003) and are shown in **Table 6**. However, it is not

569 incorrect to claim MWI nonhazardous if decided based on the United States Environmental

570 Protection Agency (USEPA) guidelines (CFR, 2020). It states that the waste becomes hazardous

571 only if the regulated PTEs [Pb, Ni, Hg, Cu, Cr, Cd, Ba, As, Ag, and selenium (Se)] overcome their

572 limit values as noted by Zhao et al. (2010) and Xie and Zhu (2013). Thus, care must be exercised

573 when comparisons of results are made against the European legislation (where no waste can be

574 anything other than "hazardous") or against the USEPA legislation (where wastes may be

575 hazardous or nonhazardous depending on content).

576 *8.2 Landfilling is based on total PTEs in medical waste incinerated residues*

577 The landfilling option for hazardous wastes such as MWI is decided based on total (or

578 extractable) PTEs concentrations in the MWI wastes, which is a misconception. However, the fact

579 is that landfilling option is decided based on the PTEs' leachability potential. Therefore, it is

580 important to note that the crucial factor for deciding where waste ash is to be disposed of is not

581 based on the actual content of PTEs (total or extractable in any form, **Table 6**). The

582 misunderstanding comes from the fact that total PTEs content is the decisive factor in all other

583 environmental matrices (soil, water and foodstuff) or wastes. In other words, in contrast to other

584 matrices, the leachability of metals (measured in mg L^{-1}) is used to determine the suitability of

585 MWI waste of disposal in a landfill, while for other wastes, total elemental concentration

23

586   (measured in mg kg^{-1}) is used to determine the suitability of waste for disposal in a landfill. This

587   difference causes various studies to compare wrong limits. This misunderstanding also derives

588   from the fact that the 33/2003 Directive (EC, 2003) reports the PTEs limits in units of dry solid

589   substance (in mg PTE leached kg^{-1} ash). Thus, this misunderstanding may lead to the wrong notion

590   that the decisive matter is total content.

591   Along these lines, Li et al. (2020) evaluated the medical wastes based on their total PTEs

592   content and compared it with total PTEs background concentrations in soil, falsely implying an

593   association between the two. Also, in Sabiha-Javied et al. (2008), no leaching was performed, but

594   only the total contents of PTEs were determined. Subsequently, the measured concentrations of

595   Pb, Cr, and Cd (as mg kg^{-1} dry waste) were compared to the USEPA limits of these metals.

596   However, these limits are derived from leaching measurements (not total) and are expressed in mg

597   L^{-1} of leachate (not mg kg^{-1}). Such misunderstanding is not necessarily characteristic of medical

598   waste alone, but it certainly derives from the fact that leaching limits are expressed in mg kg^{-1}.

599   Nevertheless, it is important to point out that total concentrations (in mg kg^{-1}) and leachate

600   concentrations (in mg kg^{-1}) are not the same and should never be directly compared, regardless of

601   whether the conversion from mg L^{-1} to mg kg^{-1} has been made. Moreover, Ababneh et al. (2020)

602   digested fly ash treated with EDTA and Na$_2$S.9H$_2$O as metal-releasing agents and measured Cd,

603   Pb and Zn in mg kg^{-1} of ash, concentrations which were then compared against the EC (2003)

604   limits—misuse probably caused due to the misconception explained below.

605   **8.3 *Expression of PTE concentration in liquid or solid waste***

606   The third misconception of PTEs' concentration in medical waste corresponds to their

607   expressions in solid or liquid units. Mostly the PTE concentrations are expressed in the liquid state

608   (i.e., the leachates), but as stated earlier, limit values in the 33/2003 Directive (EC, 2003) are

609   expressed in mg of PTEs per kg of dry substance. Thus, at first, it is not clear whether this is to be

610   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 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

635 either a "nonhazardous waste landfill" or to a "hazardous waste landfill" is for a leaching

636 experiment to be performed at L/S=10 or at L/S=2.

637 There are also some cases where leaching data are expressed in units that make any further

638 comparisons very difficult, as in the case of Xie and Zhu (2013). They conducted leaching

639 according to TCLP (USEPA), EN-12457 (European Union) and HJ/T (China) and expressed data

640 as a "percentage of the extraction ratio" rather than in any form of concentration units. Also,

641 although some works choose to perform leaching with well-established protocols, they seem to

642 make unnecessary variations that make further comparisons impossible, such as the case with

643 Kougemitrou et al. (2011). The latter used TCLP and EN-12457 at L/S=50.

644 **9**. **Conclusions and future research needs**

645 Globally, medical waste management needs to be prioritised. Poor management and/or

646 mismanagement of medical waste leads to various environmental and health-related issues, such

647 as pollution of surface water and groundwater by untreated medical waste disposal in landfills and

648 the spread of various diseases by viruses and microorganisms. Incineration is one of the most

649 commonly practised solutions to reduce the massive volume of medical waste globally. However,

650 the incineration of medical waste generates gaseous pollutants and ash as a new type of waste.

651 Medical waste incineration generates solid residues, which release PTEs, organic compounds, and

652 inorganic salts in the environmental matrices. Incineration is unable to control the PTEs levels,

653 and PTEs are often released into the environmental matrices, along with ash, in more concentrated

654 and hazardous forms. Physical or chemical stabilisation of medical waste is practised to prevent

655 the release of PTEs into the environment. Chemical stabilisation has shown promising results in

656 cost-effectively reducing the leaching of PTEs from medical waste. Released PTEs from the

657 leachate of medical waste can be treated *ex-situ*. There are also widespread misconceptions about

658 the toxicity of hazardous medical waste related to PTEs. However, under the current legal

659 definition, no medical waste can be considered 'inert', even with minimal PTE content.

660 Given the current knowledge on the management of medical wastes and the fate of PTEs in

661 medical waste incineration ash, the following research areas could be pursued:

662 - Effect of segregation of medical wastes on the distribution of PTEs in the ash material. For

663   example, electronic and some plastic medical wastes are enriched with PTEs, which can

664   be segregated to monitor the distribution of PTEs in the incinerated ash material.

665 - The transformation and release of PTEs during the incineration of medical wastes. While

666   most of the PTEs are oxidised to their respective oxides during the incineration, some of

667   the PTEs, including Cd and Hg, get volatilized, which needs to be monitored.

668 - Effect of solidification/stabilisation (S/S) of medical waste on the release of PTEs. The

669   solidified/stabilized PTEs are subject to dissolution and solubilization when the ash

670   materials are added to the soil. The release of PTEs and their subsequent leaching to

671   groundwater sources need to be monitored.

672 - The plant uptake of PTEs during the soil application of medical waste ash. The uptake of

673   PTEs by plants results in their transfer to animal and human food chains, which must be

674   examined under various soil and climatic conditions.

675 - Development of a safe and sound environmentally friendly treatment technology to

676   mitigate environmental and health impacts of PTEs in hazardous medical wastes.

677

678 **Acknowledgement**

682

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