Waste incineration and the environment

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Arnika is a Czech non-profit organisation that has been connecting people working for a better environment since 2001. Our mission is to protect nature and a healthy environment for future generations at home and around the world. We have long been working for less waste and hazardous substances, healthy rivers and diverse nature, and the right of citizens to make decisions about the environment.

Arnika's "Don't Burn, Recycle!" campaign was launched at the end of 2011 in response to repeated efforts to build massive waste incinerators in the Czech Republic. Through the campaign, Arnika calls on the public and politicians to actively oppose this trend and focus their attention and support on waste prevention, recycling, composting and take-back and recovery systems. Find out more at https://arnika.org/kampan-nespaluj-recykluj.

Nikola Jelínek provides expert support in EIA processes related to waste incineration. At Arnika, she provides expert support for Czech and international projects, including environmental sampling.

Jindřich Petrlík is currently engaged in international campaigns and projects as part of Arnika's work for the International Pollutants Elimination Network (IPEN). He has made a significant contribution to the preparation of key guideline documents of the Stockholm Convention on Persistent Organic Pollutants and of the Minamata Convention on Mercury. He also led successful campaigns for introduction of Ozone Layer Protection law and Pollutant Release and Tranfer Register in the Czech legislation. His primary objective is to reduce the environmental pollution with dioxins.

Sarah Ožanová is the coordinator of "Don't Burn, Recycle!" project in Arnika, where she helps local citizens and associations navigate EIA processes related to waste incineration. She also worked to change national legislation towards greater sustainability and a truly circular economy.

Centre de Recherche et d'Education pour le Développement (CREPD) is a Cameroon-based non-profit NGO created since 2004 dedicated to bridge the gap between science and action in Cameroon and sub-Saharan Africa and to promote sustainable development. The organization was granted its legal status in 2005. CREPD has a consultative status with UNEP and UNEA and serving as the Francophone Africa Hub of the International Pollutants Elimination Network (IPEN).

Gilbert Kuepouo is graduated in Geology/petrology in 1994 and he obtained his Ph.D. degree in geochemistry at the University of Kobe, Japan in 2004. He is currently the Coordinator/Executive Director of CREPD, Co-Chair of the IPEN Heavy Metal Working Group, Member of the Dioxins Working Group of IPEN, Co-Ordinator of the International Pollutants Elimination Network (IPEN) Hub for Francophone Africa. Dr. KUEPOUO has extensive experience in SAICM issues and in the negotiations of Basel, Rotterdam and Stockholm conventions during their COPs meetings, and is very instrumental in their implementation nationally and regionally.

The International Pollutants Elimination Network (IPEN)

FOR A TOXICS-FREE FUTURE

IPEN is a global network forging a healthier world where people and the environment are no longer harmed by the production, use, and disposal of toxic chemicals. Over 600 public interest NGOs in more than 120 countries, largely low- and middle-income nations, comprise IPEN and work to strengthen global and national chemicals and waste policies, contribute to ground-breaking research, and build a global movement for a toxics-free future.

Lee Bell is a Technical and Policy Advisor to IPEN specialising in POP waste, incineration, non-combustion technologies and hazardous waste management.

Centre for Environment Justice and Development (CEJAD), is a public interest Non-Governmental Organization in Kenya. CEJAD works to promote sound management of chemicals and waste in order to protect the environment and human health, especially vulnerable populations. CEJAD is an accredited NGO to UNEP and undertakes advocacy programs seeking to eliminate exposure to toxic chemicals by both humans and the environment.

CEJAD's mission is to promote sound chemicals and waste management in order to protect the environment and human health, especially vulnerable populations. Its programs include plastic and waste management, elimination of lead in paint, elimination of highly hazardous pesticieds and mercury.

CEJAD's activities empower communities to make informed choices towards chemical use and disposal. We also use the data to build stronger cases for policy and regulatory reforms. We form international collaborations to address local challenges to chemical safety that are embedded in international trade inequalities and global policy frameworks. At the same time, we form and support local grassroots coalitions that enable learning safer alternatives to chemicals and build demands for national policy reforms.

Griffins Ochieng is the Co-founder and the Executive Director of CEJAD. Griffins Ochieng holds a Bachelor's Degree in Environmental Science from Kenyatta University, Nairobi, Kenya. At CEJAD, he also serves as the Programme's Coordinator, managing a portfolio of various projects related to chemicals and waste management including heavy metals such as mercury and lead, Plastics and Waste Management, Persistent Organic Pollutants (POPs) elimination among others.

Toxics Free Australia (TFA) is a not for profit, civil society network working towards pollution reduction, protection of environmental health and environmental justice for all. As the Australian focal point for the International Pollution Elimination Network (IPEN), we are committed to creating a Toxics-Free Future and working towards the full implementation of the Stockholm Convention on Persistent Organic Pollutants and other global chemical conventions such as the Minamata Convention (Mercury), the Basel Convention (hazardous waste exports), the Rotterdam Convention (chemicals and pesticides) and the new Global Treaty on Plastic. Toxics Free Australia envisions a world where the food we eat, the products we use and the waste we generate is free from toxic substances and materials that can harm our health and environment. A key campaign of TFA is Zero Waste Australia which promotes Zero Waste and Circular Economy policies and an end to waste incineration.

Jane Bremmer is the Chair of TFA and campaign coordinator for the Zero Waste Australia campaign, providing advocacy, referral and support to communities across Australia facing environmental health and justice impacts from industrial and chemical pollution sources.

https://www.toxicsfreeaustralia.org.au/

Contents

List of abbreviations

AMESA – Automated Measuring System for Real–Time Recording of Dioxins and Furan Emissions
APC – Air Pollution Control
ARIA - Analysis, Research and Information on Accidents - French database catalogues incidents or accidents that were, or could have been, deleterious to human health, public safety or the environment
BA – bottom ash
BaP – Benzo[a]pyrene
BARPI - General Directorate for Risk Prevention, the BARPI (French acronym for Bureau for Analysis of Industrial Risks and Pollutions)
BAT – Best Available Technologies
BAT–AEL – Best Available Techniques–Associated Emission Levels
BCD – Base Catalysed Decomposition
BEQ – Bio-toxic equivalent (equivalent to TEQ for bioassay analysis)
BFR, BFRs – Brominated Flame Retardants
BPA – Bisphenol A
BREF – BAT Reference document(BAT and BREF refer to directives and guidelines established by the European Union)
BTBPE – 1,2-Bis(2,4,6-tribromophenoxy)ethane
BTEX – Benzene, Toluene, Ethylbenzene, and Xylenes (a group of toxic VOCs)
CBzs – Chlorinated Benzenes
CDC – Catalytic Dechlorination using Copper catalysis
CEJAD – Centre for Environment Justice and Development
CEWEP – Confederation of European Waste-to-Energy Plants
CHD – Catalytic Hydrogenation
CO – Carbon Monoxide
CREPD – Centre de Recherche et d'Education pour le Développement
CSOs – Civil society organizations
CR – Cancer Risk
DBDPE – Decabromodiphenyl ethan
DDT – 1,1'-(2,2,2-Trichloroethane-1,1-diyl)bis(4-chlorobenzene)
DR CALUX – Dioxin–Responsive Chemically Activated LUciferase Gene Expression
DRE – Destruction and Removal Efficiency
dm – dry matter
DE – Destruction Efficiency
dl PCB – dioxin-like Polychlorinated biphenyls
EC – European Commission
ED – Exposure Duration
EEB – European Environmental Bureau
EFSA – European Food Safety Authority
EIA – Environmental Impact Assessment
EPR – Extended Producer Responsibility
EU – European Union
EU–ETS – European Union Emissions Trading System
EUROSTAT– Statistical Office of the European Union
FA – fly ash
GAIA – Global Alliance for Incinerator Alternatives
GPCR – Gas Phase Chemical Reduction
GWP - Global Warming Potential
H_2S – Hydrogen Sulfide
HWI – Hazardous waste incineration (incinerator)
HBCD – Hexabromocyclododecane
HCB – Hexachlorobenzene
HCBD – Hexachlorobutadiene
HCH – Hexachlorocyclohexane
HCl– Hydrochloric Acid
HDPE – High–Density Polyethylene
HI – Hazard Index
HP – Hazardous Properties
IARC – International Agency for Research on Cancer
I–TEQ – International Toxicity Equivalency Factor
IED – Industrial Emissions Directive
IMPEL - French Ministry of Sustainable Development
IPEN – International Pollutants Elimination Network

ITD – Indirect Thermal Desorption
IWI – Industrial waste incineration (incinerator)
LDPE – Low-density polyethylene
LPCL – Low POPs Content Level
MedWI – Medical waste incineration (incinerator)
MFA– Ministry of Foreign Affairs
MoEES - Ministry of the Environment, Energy and the Sea (in France)
MSWI – Municipal solid waste incineration (incinerator)
NA – Not Analysed/Available
NaClO – Sodium Hypochlorite
NATO – North Atlantic Treaty Organization
nBFR/nBFRs – novel Brominated Flame Retardants
NGO – Non–Governmental Organization
NH_3 – Ammonia
N_2O – Nitrous Oxide
NO_x – Nitrogen Oxides
OBIND – Octabromo-1,3,3-trimethyl-1-phenylindane
OCDD – Octachlorodibenzo–p–dioxin
OCDF – Octachlorodibenzofuran
OTNOC – Operation other Than Normal Operating Conditions
PA – Polyamide
PAH – Polyaromatic hydrocarbons
PBDD/Fs – Polybrominated dibenzo-p-dioxins and dibenzofurans
PBDE – Polybrominated diphenyl ethers
PBT – Pentabromotoluene
PBB – Polybrominated biphenyls
PCDD/F – Polychlorinated Dibenzo-p-Dioxins and dibenzofurans
PCDT – Polychlorinated dibenzo-p-thiophenes
PCN – Polychlorinated naphthalenes
PCTA – Polychlorinated thianthrenes
PeCB – Pentachlorobenzene
PE – Polyethylene
PET – Polyethylene terephthalate
PFAS – Per- and PolyFluoroAlkyl Substances
PFOA – Perfluorooctanoic acid
PFOS – Perfluorooctanesulfonic acid
PM – Particulate Matter
PMMA – Poly(methyl methacrylate)
POPs – Persistent Organic Pollutants
POP RC– Persistent Organic Pollutants Review Committee
PP – Polypropylene
PPF Group – A global investment company.
PS - Polystyrene
PTFE - Polytetrafluoroethylene (Teflon)
PU - Polyurethane
PVC - Polyvinyl chloride
PXDD/F - Polyhalogenated dibenzo-p-dioxins and dibenzofurans; contain halogen atoms in various combinations
P2P - Plastic to plastic (Plastic 2/to Plastic)
P2F - Production of fuel from plastic (Plastic 2/to Fuel)
REACH – EU Regulation on Registration, Evaluation, Authorization and Restriction of Chemicals
RDF – Refuse-Derived Fuel
SCR – Selective Catalytic Reduction
SHSWTC - The Swan Hills Solid Waste Treatment Centre
SNCR – Selective Non-Catalytic Reduction
SO_x – Sulfur Oxides
SOS - Safe Operating Space
SR – Alkali Metal Reduction – Sodium Reduction
SCWO – Supercritical Water Oxidation
TEQ – Toxic Equivalent
TFA – Toxics Free Australia
TOC – Total Organic Carbon
TVOC – Total Volatile Organic Carbons
TWI – Tolerable Weekly Intake
UNEP – United Nations Environment Programme
UK – United Kingdom
USA – United States of America
UPOPs – Unintentional POPs produced as by-products
US EPA – U.S. Environmental Protection Agency
VOCs –Volatile Organic Compounds
WHO – World Health Organization
WtE – Waste-to-Energy
y – year
ZWE – Zero Waste Europe

1. Introduction

Waste management represents a profoundly significant and difficult challenge in today's world. Globally, Municipal Solid Waste (MSW) generation is expected to grow from 2.3 billion tonnes in 2023 to 3.8 billion tonnes by 2050 (UNEP & ISWA, 2024).

The waste we generate reflects our entire materials production systems. As such we can see that the plastics and petrochemical industry are heavily embedded in the materials, products and packaging we use and discard in our modern world. This brings inherent problems for humanity and our planet as we rapidly move away from natural cycles of decomposition and regeneration, towards a waste stream that is dominated by synthetic chemicals and materials that do not degrade and cause serious pollution and harm. Indeed, waste is inherently tied to the triple planetary crisis of climate change, pollution and biodiversity loss. The choices we make today about how to manage our waste will define our future.

In many countries, waste is currently largely disposed of in landfills. Landfills have become places where waste pollution accumulates and it is clear that we need to address this problem and improve our waste management. Waste incineration is often promoted as the ideal solution to the waste problem, providing us with electricity or heat as a bonus. But is it really the ideal solution? Both landfill and incineration offer us a way to dispose of our waste, but do we have other options besides these two? We know that landfill can cause serious environmental and human health impacts, but what about incinerators? Can we really burn our waste safely and destroy these finite resources indefinitely? These are the questions that frontline communities who live with incinerators or who are facing such projects, often ask, in many cases without answers from authorities.

Arnika is very often asked for advice on how to assess the environmental impact of proposed waste incineration plants. This suggests that there is a lack of support for civil society to defend their environmental health and justice. This report aims to provide essential and technical information for any community facing the threat of a waste incinerator. For easy navigation you can find a final summary of the key issues in Chapter 12.

The study is primarily intended to serve civil society organizations (CSOs), impacted citizens and communities, state and local governments. While this report has been written with a Czech Republic focus following the original country specific version (Jelinek et al., 2023), the technical references and key issues identified here are internationally applicable. Indeed, many countries rely on the European Best Available Technique (BAT) guidelines (European Commission, 2019) as the basis for their own country specific industrial regulation standards to justify approving incinerator projects. Yet this report highlights the significant failures of these guidelines as experienced by the Czech Republic and many other European countries. The material reality of the adverse impacts of waste incineration on those communities living close to such facilities, is underscored by this report. As the global south faces a concerted push to establish waste incineration widely, particularly in the Southeast Asian region, where there is little experience with such technologies and industrial regulatory oversight is not assured, the protection of the environment and human health subsequently faces many serious threats.

The potential environmental and human health impacts of waste incinerators can be both complex and severe. While waste incineration may offers some energy benefits, it is important to carefully consider its overall impacts and seek more sustainable alternatives. This study focuses on a detailed analysis of these impacts and aims to contribute to a greater awareness and understanding of these cross-sector and inter-related issues, while protecting our environment and health. In doing so, we have drawn on our long-term experience and engagement with experts in the field to provide you a robust and effective report to assist the public, municipalities and state administrations. Waste and its generation, represents our entire material production systems and reflects back to us those design failures that we must address if we want to truly develop a safe and toxic free Circular Economy. This report adds to the growing body of evidence that waste incineration essentially undermines more sustainable Zero Waste policies and the goal of a Circular Economy.

2. What is a waste incineration?

A waste incinerator is a technology that is used for the combustion of waste materials. The goal of waste incineration is to reduce the volume of waste and destroy, minimize, or concentrate hazardous components it contains. If the waste incinerator also produces energy in the form of electricity or heat, it is referred to as a Waste to Energy (WtE) facility or plant. This distinction does not have a major impact on the generation of emissions or solid residues from waste incineration, so with a few exceptions we will continue to make no distinction between waste incinerators and WtE[1] incineration plants. Other plants that burn waste together with other fuels, called "co-incineration", are cement or lime plants (see Photo 2.2). These are not called incinerators, but co-incinerators. Their technology differs significantly from that of waste incinerators, as they are primarily designed to produce cement or lime. Sometimes waste is co-incinerated in other facilities, such as coal-fired power plants and pulp and paper mills.

There are different types of incinerators, for example: grate incinerators, fluidized bed incinerators or rotary kilns (see Figures 2.1 – 2.3). Fluidized bed incinerators are more commonly used to incinerate homogeneous material (sludge, biomass), where 3 to 4 times more fly ash is produced (Stockholm Convention, 2008). The third type - rotary kilns - are suitable for less homogeneous materials, both solid and liquid waste, as they ensure better mixing of the waste.

In general, for incineration, the waste must have a calorific value above 5 $MJ.kg^{-1}$ (Vejvoda et al., 2018),[2] an ash content of less than 60 %, a moisture content of less than 50 % and a volatile combustible content of more than 25 %. Incineration takes place at a temperature of about 850°C, for hazardous waste at 900 to 1,200°C (Stockholm Convention, 2008).

Incineration means treating waste with more than the stoichiometric[3] amount of oxygen needed to oxidize the substances present. The excess air in incinerators is between 1.5 and 2.5 times the stoichiometric oxygen content. According to (Neuwahl et al., 2019), thermal

1 Biogas production can also be considered a form of WtE, as it involves the recovery of energy from organic waste materials through anaerobic digestion, but in this study WtE stands for the process of generating energy (usually electricity or heat) from the combustion of waste materials.

2 In the Czech Republic, according to Act No. 541/2020 Sb., waste with a calorific value higher than 6.5 MJ.kg-1 may not be landfilled after 2030.

3 The substance's complete oxidation requires a specific amount of oxygen, known as the stoichiometric quantity.

Figure 2.1: *Schematic of a rotary kiln incineration system. (Source: Jiang et al., 2019)*

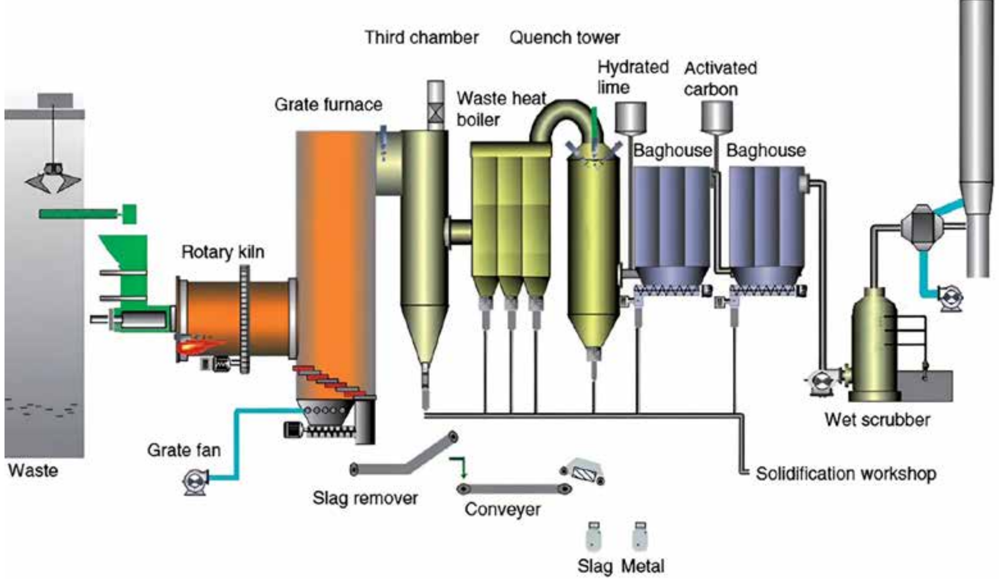
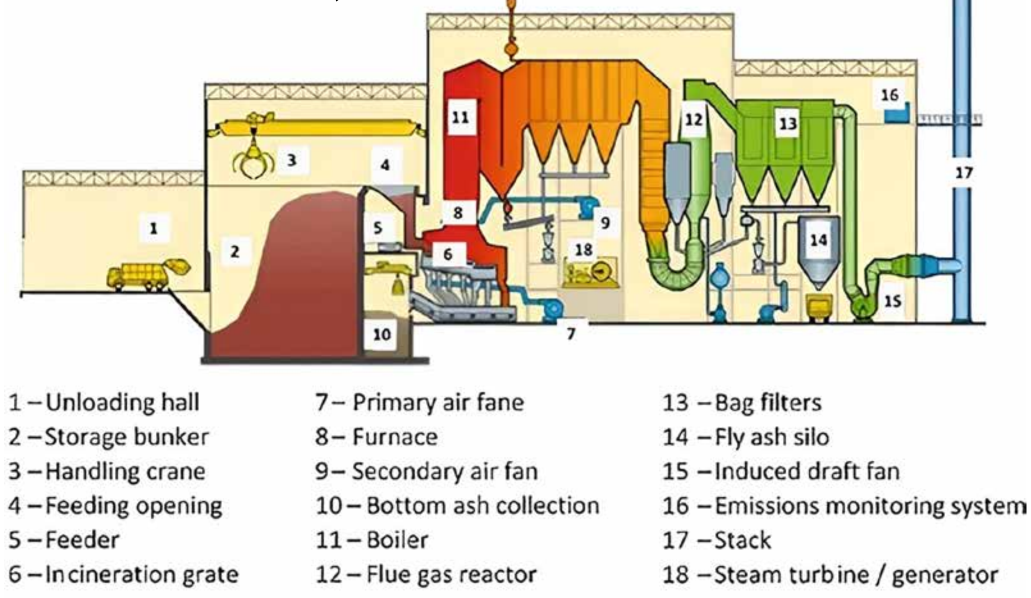


Figure 2.2: *Schematic of a grate incineration system. (Source: Bandarra & Quina, 2021)*



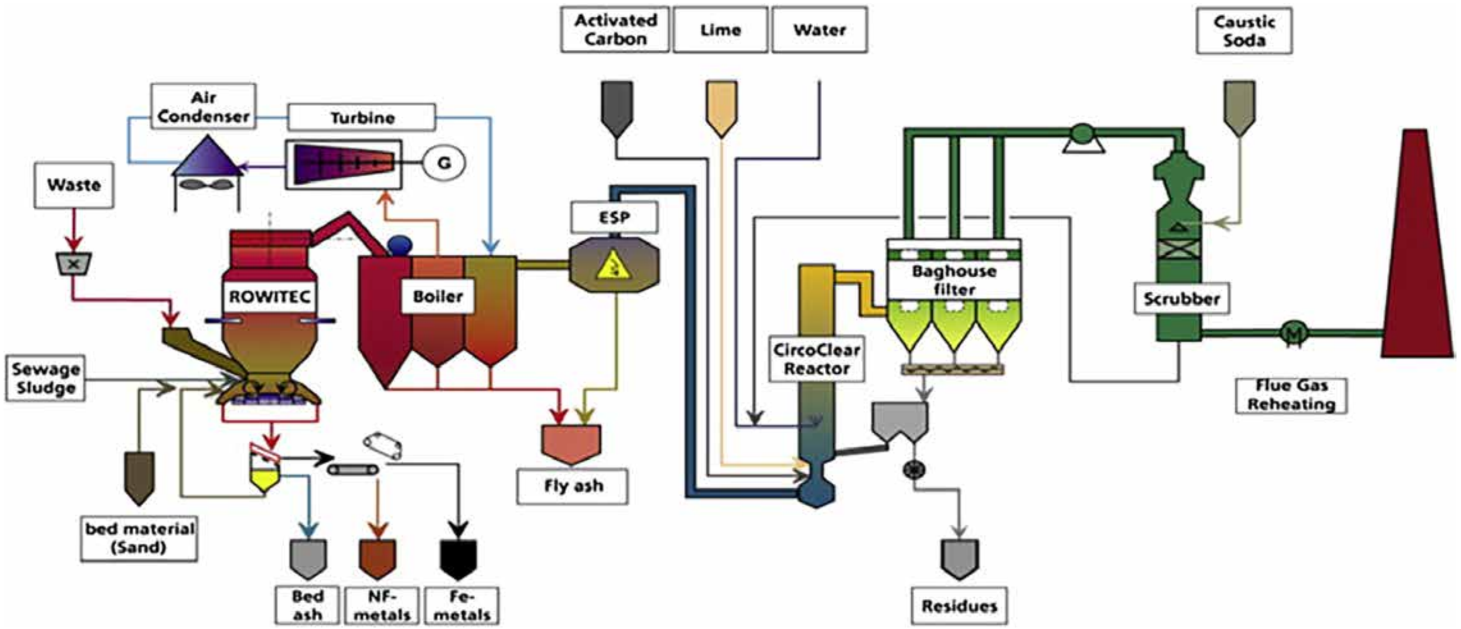


Figure 2.3*: Schematic of a fluidized bed incineration system. (Source: Van Caneghem et al., 2012)*

Photo 2.1: *Municipal Waste Incinerator (ZEVO) Termizo, a.s. Liberec, as it looked in 2021. Photo taken from a drone by Marek Jehlička (skyworker.cz).*

treatment of waste also includes gasification or pyrolysis of waste. These differ from incineration (which is essentially oxidation) in that they take place in the presence of less (gasification) or no (pyrolysis) added oxygen, i.e. at less than the stoichiometric oxygen content (in a reducing environment) (see Chapter 2.4). Still, both pyrolysis and gasification technologies of municipal solid waste that create outputs such as fuels, are classified as waste incineration technologies by both the EU (European Parliament and Council, 2010) and US (US EPA, 2008).

2.1 Municipal solid waste incineration (MSWI)

The incineration of municipal solid waste (MSW) requires pre-treatment because it is an extremely heterogeneous material.[4] In particular, this involves homogenization (usually shredding and mixing) and separation of inert materials[5] (such as rocks, concrete and brick rubble). In Europe, grate incinerators are most commonly used to incinerate MSW. Incinerators that burn municipal waste tend to have larger capacities than those that focus on burning industrial or medical waste. MSWI is commonly accompanied by the recovery of energy (in the form of steam and/or the generation of electricity).

Possible alternatives to incineration of municipal waste incineration are listed in Chapter 8.1 (Municipal Waste).

2.2 Hazardous waste incineration (HWI)

Incineration is one of the *most common technologies used* for dealing with waste that is classified as hazardous or is a waste mixture that contains hazardous materials (e.g. has hazardous properties HP1 to HP15 according to Annex III of EU Directive 2008/98/EC on waste). Due to their hazardous properties, these wastes also require special treatment for transport, storage, etc., which may also apply to the residues resulting from *all types of incineration technologies*.

Hazardous waste incineration also occurs in the cement industry via the use of rotary kilns in cement plants. (i.e. spent solvents). Kiln temperatures are typically in the range 900°C-1,200°C when incinerating hazardous waste

4 Heterogeneous materials are composed of particles of varying sizes and types, while homogeneous materials consist of particles that are similar in size and character.

5 Inert materials do not undergo chemical reactions or combustion-oxidation.

Photo 2.2: *The Cemex cement plant in Prachovice incinerates mainly plastic waste, some of which is imported from abroad. See also photos 10.1 and 10.2 from preparation of the plastic waste for this cement kiln. Photo: Jan Losenický, Arnika.*

and include a 30 to 60 minutes residency time inside the rotary kiln drum itself. Similar to MSW incineration, the organic materials are destroyed, the volume is reduced, and the pollutants are concentrated into the ash. However, higher volumes of particulates are usually generated in the process requiring expensive additional air pollution control equipment. Energy recovery is less common than with MSW incineration.

Hazardous waste may be incinerated in commercial incinerators, which typically process a variety of waste streams, or in dedicated or captive incinerators, which are typically part of industrial facilities and process the waste generated there.

Possible alternatives to incineration of hazardous waste are listed in Chapters 8.2 (Hazardous waste), Chapter 8.3.6 (Handling mercury-containing waste), Chapter 8.3.7 (Waste containing POPs) and Chapter 3.3.1 (Processing Waste Containing POPs).

2.3 Medical waste incineration (MedWI)

Medical waste *includes* all the waste generated by healthcare facilities, medical laboratories and biomedical facilities, as well as waste from minor sources. The bulk of healthcare waste is produced by hospitals. It is universally accepted as a potential danger to human health and the environment if it is not managed in an environmentally safe manner.

The incineration of medical waste (infectious healthcare waste, biological healthcare waste, and sharps) in dedicated incinerators is done to minimize chemical, biological, and physical risks and to reduce the volume of waste as a step prior to environmentally sound landfilling.

Medical waste incinerators tend to have smaller capacities and therefore the application of best available techniques (BAT) may be challenging for them. However, if medical waste is not incinerated according to BAT, it may result in the release of dangerous persistent organic pollutants such as PCDD/Fs to air.

Possible alternatives to incineration of medical waste are listed in Chapter 8.3, noting that up to 90 % of hospital waste is similar in composition to conventional municipal waste.

2.4 Gasification and Pyrolysis

Gasification and pyrolysis are "alternative" thermal waste treatment technologies (i.e. incineration) that operate in a reduced or oxygen free environment, to convert waste into gaseous, liquid, and/or solid products that can then be used as chemical feedstocks or burned to recover energy. When involving plastic feedstocks only, they are often referred to as thermal depolymerization processes. These processes are most suitable for polymers composed of a limited number of elements - carbon and hydrogen (PP, PS, PE) or additional oxygen (PMMA). The process can be carried out in a

Figure 2.4 *Gasification diagram. (Source: Zafar, 2009)*

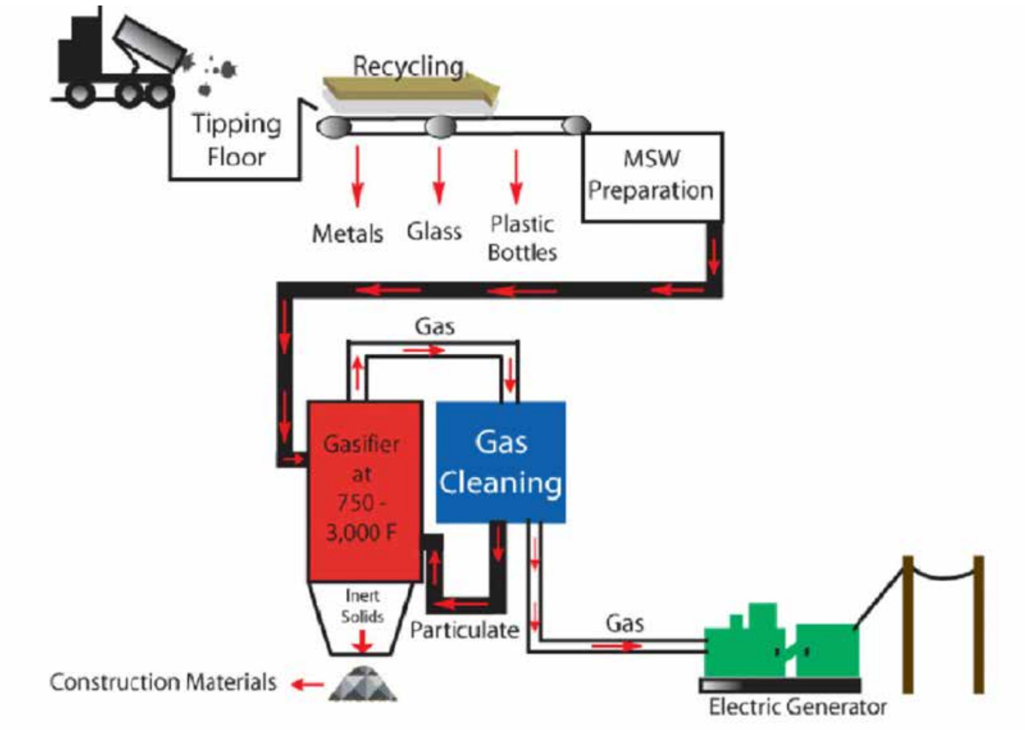


Photo 2.3: *Pyrolysis in Burgau, according to Gleis (2012) expensive in construction and operation. Photo: LfU Bayern.*

controlled way (to form monomers) or as a cracking[6] thermal depolymerization, which produces a mixture of different molecules, resulting in a product similar to a petroleum fraction. This is because the splitting of molecules occurs at random positions (and therefore cannot be controlled); (ZWE, 2019a). However, the application of these systems is low compared to combustion, and operational problems have been reported in some plants[7] (Gleis, 2012; Stockholm Convention, 2008). Controlled depolymerization technologies such as those employed by multi-national chemical corporation - Arkema - processes PMMA ("plexiglass") at 450°C with high yields (ZWE, 2019a).

6 Cracking refers to the processing of crude oil, in which longer-chain hydrocarbons are broken down into shorter-chain hydrocarbons.

7 In Chapters 7.3 and 9.2 we describe the case of the pyrolysis unit in Hamm, Germany.

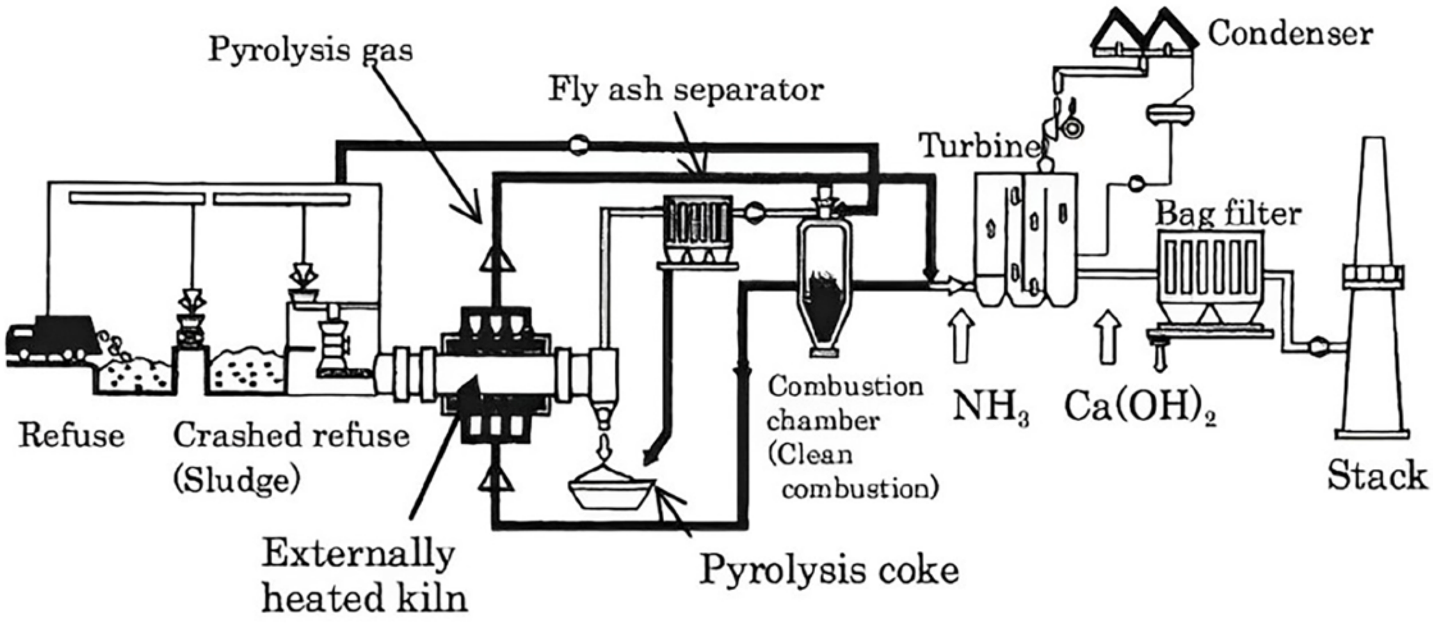


Fig. 2.5 *Pyrolysis diagram. (Source: Weber & Sakurai, 2001)*

2.4.1 Gasification

Gasification generally refers to the heating of mixed materials under limited oxygen conditions (ZWE, 2019a). The product of gasification is mostly pyrolysis gas, which consists mainly of hydrogen and carbon monoxide. It also contains carbon dioxide, water, methane or higher hydrocarbons, ammonia, sulfane, and inert nitrogen (Němcová, 2017). The reactions take place at temperatures between 500 °C and 1,400 °C, the pressure in the reactor is atmospheric or higher. The reaction produces vitrified slag (at high gasification temperature). Compared to combustion, the gasification process produces fewer gaseous products and more CO than CO_2 (Chang & Pires, 2015).

2.4.2 Pyrolysis

During pyrolysis and in the absence of (added) oxygen or other oxidizing agents, substances are thermally decomposed into low molecular weight substances and a solid residue is formed. (Cornelissen et al., 2009). Pyrolysis generates pyrolysis gas, liquids and solid residue (Carrier et al., 2011; Chen & He, 2011). A distinction is made between fast pyrolysis, which takes place at temperatures between 500 °C and 1,000 °C (with a residence time in the reactor of a few seconds), and slow pyrolysis, which takes place at temperatures between 400 °C and 600 °C (with a residence time in the reactor of several hours); (Malaťák & Jevic, 2017). Due to the rapid heating, pre-treatment of the waste into a more homogenous and finer fraction, sometimes even into a Refuse Derived Fuel (RDF), is necessary.

It is claimed that it is possible to pyrolyse feedstock without any limitations (even mixed or contaminated types of plastics). However, the reality is that some oxygenated resins contribute to the formation of more coke, or in the case of PVC, to the formation of HCl, which can lead to corrosion

Photo 2.4: *Fires can easily occur in pyrolysis plants. In February 2019, a tire pyrolysis unit caught fire in Nederweert, the Netherlands. (Source: Scott, 2019).*

of the equipment. There is a lack of information on contaminants in the outputs. A constant composition of input plastics is required for stable operation. Typically, similar units are focused on PE or PP, but this process is very challenging for PET, nylon or PVC. It is a highly energy intensive process that has been shown to produce PAHs or dioxins (ZWE, 2019a; Bell et al., 2023b).

Slow pyrolysis produces all phases - liquid, gas and solid - in approximately equal proportions. For example, 43 to 85 % by weight of carbon is found in the solid residue after pyrolysis (of plastics); (Němcová, 2017). Pyrolysis oil is a mixture of several hundred substances in which phenolic compounds, organic compounds, furfural and its derivatives are significantly represented (de Wild et al., 2009; Li et al., 2010; Sınağ et al., 2011). It can be used in combined heat and power plants, but its acidity and viscosity are too high for conventional diesel engines, and it is unstable (Jílková et al., 2012). The oil produced does not meet the cracking requirements, so BASF's Chemcycling unit, for example, has to dilute it with conventional petrochemical feedstocks to meet the required specifications (Koyuncu et al., 2021), undermining claims that this technology delivers recycling outcomes.

In addition, pyrolysis results in a large loss of carbon, only about half of which is converted to oil. As early as 1995, pyrolysis was found to be the largest contributor (among chemical recycling[8] processes) to global warming and photochemical ozone creation (ZWE, 2019a). At the same time, it generates the largest amount of solid waste after landfilling, which causes other negative environmental impacts (Hegyi et al., 2021; Mølgaard, 1995). Fires are also common in the pyrolysis units. They spread quickly due to the handling of plastic waste (Hegyi et al., 2021; Hutková, 2016; Scott, 2019; ZWE, 2019a).

2.4.3 Plasma gasification

A special case of gasification is plasma gasification. This process takes place at high temperature (1,250 to 3,500 °C) in the presence of plasma, an ionized gas consisting of a mixture of electrons, ions and neutral particles. The heat source is one or more plasma torches that create an electric arc. It is a process that can be applied to waste with minimal need for pre-treatment (Arena, 2012), such as single-use plastics. The organic components of the waste are converted to gas. The inorganic components are converted to vitrified slag after cooling (Young, 2010). The composition of the gas produced depends on the composition of the

8 „Chemical recovery" should be used instead of „chemical recycling" as per (Koyuncu et al., 2021) because the amount of recycled product in the recycled material is minimal.

raw material entering the plant. The use of plasma gasification directly for waste treatment is very energy intensive, therefore it is used more for the purification of the gas produced by gasification (Němcová, 2017).

There was Westinghouse Plasma Corp. plasma waste gasification plant proposal to be built in Czech Republic (Kašpar et al., 2019) in 2019, Westinghouse tried the same in Barbados in 2014. (Cheeseman, 2014). The project ended in failure (Dean, 2016), in particular because there was not enough waste on the island to fill the overcapacity of the planned project. In 2016, Westinghouse Air Products' giant municipal waste incinerator project on Teesside in the United Kingdom also ended in failure (Clay, 2016; Simkins, 2016). It failed because of the inability to deal with the problem of corrosion in the waste gasification process unit itself due to acid vapors.[9] The problem was that the technology was not working at the capacity of 360,000 tons of waste being incinerated per year.

2.5 Chemical recycling

Gasification and Pyrolysis technologies are often also referred to as Chemical Recycling, Plastic Recycling and Advanced Recycling technologies, especially when the feedstock is solely plastic. These technologies use a combination of heat, pressure, low oxygen level, catalysts, and solvents to break down plastics[10] into fuels (P2F) or "building blocks" for new plastics (P2P); (ZWE, 2019a). The processes covered by the term chemical recycling can be divided into thermochemical (pyrolysis and gasification can be included here), chemical-solvent-based and, less commonly, enzymolysis (see Table 2.1). Plastics (such as PET) are also used for mechanical recycling,[11] leaving hard to recycle or contaminated plastics for chemical recycling and placing the technology in direct competition with material recycling (ZWE, 2019a) for viable feedstocks. Meanwhile, mechanical recycling is more cost effective, ecologically sound and requires less energy input (Tabrizi et al., 2022). The final emissions associated with the operation of a chemical recycling facility are also significantly influenced by the energy source of the facility, as well as the feedstock and can result in significant generation of halogenated chemicals and persistent organic pollutants (Bell et al., 2023b ZWE, 2019a).

Solvolysis is a process based on making the polymer that forms the plastic soluble in a selected solvent (ZWE, 2019a). The polymer is separated from the undissolved additives by filtration or extraction (the further treatment of the additives is not very clear). However, the polymer may contain residual additives, other impurities or solvent. Solvolysis works with single type plastics (PVC, PS, PP, PE), so the quality of the input polymer is critical to the quality of the output. Similar to mechanical recycling, there is a reduction in the average chain length of the polymer, which reduces its quality. The VinylLoop unit, which processed 10,000 tons of softened PVC per year, was closed after 16 years of operation (in 2018) because it was not economically feasible to remove phthalates[12] from the

9 Phil Whitehurst, GMB's construction manager, is reported to have told ENDS that the plasma gasification equipment installed at TV1 had „eroded the walls of the gasification plant through a combination of heat and acids". According to ENDS, the tests created „large holes" in the ceramic lining of the gasifier stack. Although some parts were taken from the second project (to repair TV1), this attempt was unsuccessful. In a statement issued in early April, the GMB union blamed an „incompetent company" for the closure of the plant.

10 Chemical recycling refers mainly to plastics.

11 Mechanical (or material) recycling means that waste material is collected, sorted, cleaned, cut into smaller pieces, melted and re-granulated. This process does not alter the polymeric structure or composition of the original material.

12 Phthalates are a group of substances derived from phthalic acid that are commonly used as plasticizers. They are most commonly used in plasticized PVC, such as in the production of synthetic linoleum.

Table 2.1: *Classification of chemical recycling processes. (Source: Rollinson et al., 2020)*

Chemical Recycling					
Solvent-Based		**Thermochemical**		**Enzymolysis**	
Dissolution by	**Solvolysis**	**Pyrolysis**	**Gasification**		
Dichloromethane	Alcoholysis (Glycolysis, Methanolysis)	Thermal cracking	Steam	In vivo	In vitro
Methylethyl ketone	Hydrolysis	Thermal depolymeriza-tion	In the presence of oxygen/air		
Tetrahydrofuran	Ammonolysis / Aminolysis	Catalytic cracking	In the presence of a catalyst		
Xylol	Other solvents	Hydrocracking	In the presence of hydrogen		
Other solvents					

output polymer (ZWE, 2019a). Another example is the PolystyreneLoop plant, which uses the CreaSolv® process to decontaminate polystyrene containing HBCD. This is described in more detail in Chapter 8.3.7.1. The Polystyvert unit works in a similar way.

Dissolution (see Table 2.1) or “chemical depolymerization” refers to processes aimed at forming monomers (hence depolymerization) from polymers using solvents (ZWE, 2019a) such as methanol, alcohol or other solvents. It is usually used for polymers formed by polycondensation. For some plastics, it is not worth returning to the monomer phase, but only splitting the polymer into shorter chains (dimers or oligomers), which is done in the presence of a catalyst and the application of heat. Since the monomer is polymerized again, it does not have the same problem as solvolysis, since the polymerization repairs the chain damage and downcycling[13] does not occur. As with solvolysis, a highly specific feedstock is required, and PET, PA or PU appear to be possible plastics for this application. The resulting plastic quality can be high, but information on yield, residual by-products or catalyst handling is lacking (ZWE, 2019a).

Most chemical depolymerization projects focus on PET processed by glycolysis, hydrolysis or methanolysis (high temperature, high pressure). Glycolysis is used by the Italian company Garbo in its ChemPET process or by the Dutch company Ioniqa.

In 2019, there were no commercial-scale chemical recycling units in operation and, conversely, those that were in operation did not provide access to data on the technologies used. As a result, the studies that have been produced on them (often by the companies themselves) cannot be (critically) evaluated, as they often focus on presenting positive facts from the operation of the facility (Rollinson et al., 2020). The data also lack information on the toxicity and ecotoxicity of the outputs produced (Tabrizi et al., 2022).

The oft-repeated claim that chemical recycling results in lower CO_2 emissions is only true for LDPE and compared to incineration (Tabrizi et al., 2022). The gasification route, on the other hand, leads to higher emissions of most of the monitored parameters in air emissions (CO_2, CO, dust, NO_x, SO_2) and has a higher acidification potential compared to olefin[14] production from crude oil or shale gas. It is also associated with

13 Downcycle means to recycle (something) in such a way that the resulting product is of a lower value than the original item : to create an object of lesser value from (a discarded object of higher value).

14 An olefin is a type of unsaturated hydrocarbon that contains one double bond between carbon atoms in an open chain. It is also known as alkene.

the presence of phthalates, bisphenol A (BPA), polybrominated diphenyl ethers (PBDEs), other toxic brominated compounds, polyaromatic hydrocarbons (PAHs), and mutagens,[15] carcinogens, and respiratory and nervous system agents (Rollinson et al., 2020; Tabrizi et al., 2022). Because the resulting oil may contain high levels of these substances, it requires further purification or dilution.

The report 'Chemical Recycling: A Dangerous Deception' (Bell et al., 2023b) argues that chemical recycling is not a solution to the plastic crisis, despite being labeled as "advanced" and touted by the plastic industry as a significant way to reduce global plastic pollution.

Chemical recycling, despite being explored for decades, remains an ineffective solution to the plastic pollution crisis due to its continued failure to produce viable outcomes. The process involves using plastics made with toxic chemicals, which not only persist through the recycling process but can also lead to the formation of new toxic substances via cross-contamination and heating, posing significant health and environmental risks. Moreover, chemical recycling is marked by inefficiency and high energy consumption, contributing to climate change with potential environmental impacts far exceeding those of producing virgin plastic, sometimes by up to a hundredfold. This method generates minimal amounts of usable products alongside considerable quantities of toxic waste, which is typically incinerated or landfilled, further exacerbating pollution issues. Facilities dedicated to chemical recycling are also sources of toxic emissions and hazardous waste, and they carry risks of fires and explosions. There's a lack of evidence supporting the efficacy of chemical recycling in processing mixed plastic waste, and it competes for feedstock with conventional recycling methods. Considering that the output of chemical recycling is often burned, it undermines the very definition of recycling. The relaxation of regulations surrounding chemical recycling compromises public health, advocating for the redirection of public funds towards genuinely sustainable solutions rather than supporting such a problematic process. Case studies in the United States of America show that chemical recycling processes produce insignificant amounts of recycled plastics from plastic waste. Instead, they often produce low-quality fossil fuels for burning.

Chemical recycling is an expensive and risky investment that draws public funds away from truly renewable and sustainable projects. It also harms the environment and human health, and threatens already overburdened environmental justice communities. The industry labels it as successful and 'green' with little to no accountability, while keeping the cost and impact on public health, environment, and managing plastic waste a secret. Although each facility takes a different approach, failure is constant.

Regardless of the technology used, the chemical recycling process cannot be considered recycling if the resulting products are incinerated, which is the case for most outputs from these facilities today. The EU is on a path to phase out fossil fuels and the plastic fuels produced by chemical recycling are a continuation of the release of CO_2 emissions into the environment. The use of P2P does not reduce the demand for virgin plastics, which must continue to be produced; at the same time, the need for plastics for these devices justifies their production and overuse. The real solution is to reduce the production and consumption of plastics, replace single-use plastics, detoxify[16] them, simplify formulations, and design business models for the efficient use of plastics (ZWE, 2019a).

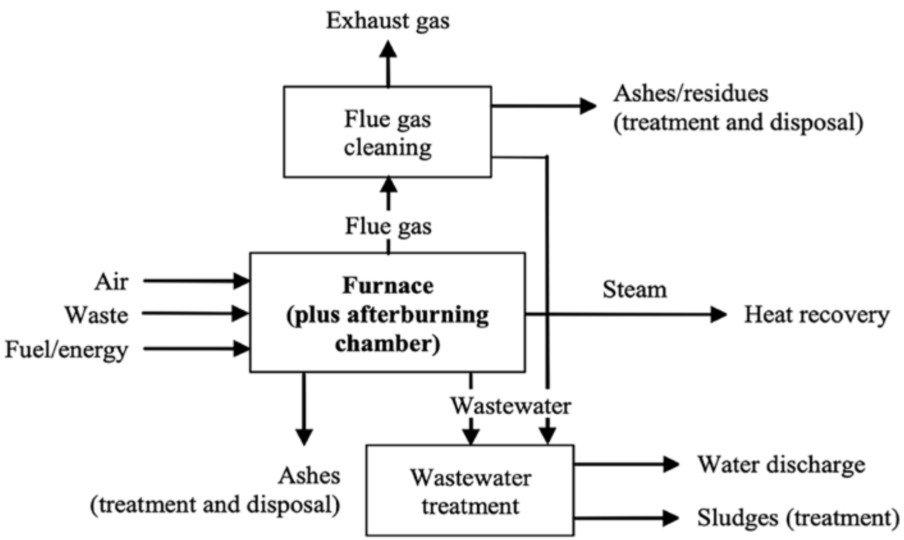
15 Mutagens are substances that can cause mutations or alter an organism's genetic information.

16 Detoxification is the process of removing toxic substances from a material.

3. Environmental Impacts of Incinerators

This chapter focuses on the environmental impacts of the substances and materials produced or released by the incinerator. These substances may be emitted to the air, discharged to the water, or (for the most part) end up as solid residues. Selected (toxic) substances or groups of substances are dealt with in Chapter 5.

Figure 3.1: *Simplified diagram of an incinerator.*
(Source: Stockholm Convention on POPs, 2019)



All incinerator projects, regardless of their location, should be subject to a full Environmental Impact Assessment. This requirement is enshrined in the Guidelines adopted by the Basel Convention (Basel Convention, 2022).

Of the waste entering the incinerator, about one-third of the original mass ends up in the form of solid residues, mainly bottom ash (slag, cinder). The flue gas cleaning process produces hazardous waste residues, which represents about 2.5 % of the initial mass (EA, 2020) and is referred to as fly ash. In addition, the incinerator releases substances in air emissions. The third pathway is emissions to water, in the case of incinerators with wet scrubbers. A simplified overview of the inputs and outputs to/from the incinerator is shown in the diagram in Figure 3.1.

3.1 Air emissions

Dust particles, inorganic substances (sulfur and nitrogen oxides, inorganic chlorine, bromine and fluorine such as HCl, HBr and HF, carbon monoxide, carbon dioxide and a variety of metals), organic substances (chlorinated dioxins, brominated dioxins, polyhalogenated dioxins, per- and polyfluorinated substances, dioxin-like polychlorinated biphenyls, polyaromatic hydrocarbons and others) expressed as total organic carbon (TOC) or total volatile organic compounds (TVOC) are emitted by incinerators to the air. A study (Jay & Stieglitz, 1995) identified about 250 organic compounds in the TOC of combustion emissions (see Chapter 5.2). Among the substances identified in the emissions were a number of carcinogenic and hazardous substances (i.e. mutagenic or toxic to reproduction, persistent, bio-accumulative and toxic). Despite the limitations of this single study, these compounds, as well as others, are not routinely measured in incinerator emissions. Figure 5.12 lists some of the substances found. As early as 1992, flue gases were found to contain mutagenic substances (Ma et al., 1992).

According to Marziali et al. (2024) emissions from waste incinerators can be divided into two categories – primary emissions, which refer to the main stack(s) of a waste incinerator (the stack that releases flue gas coming from the combustion chamber) and are treated by the air pollution control (APC) technology installed; and secondary emissions which refer to the venting of silos containing ashes recovered by the APC system, and the discharge of the air from waste storage compartments or other indoor environments (see Chapter 3.1.1). Secondary emissions have also been recently estimated to account for up to 29 % of the total dust and 10% of the organic carbon releases (Schiavon et al., 2020) which makes their contribution not negligible and should always be part of the Environmental Impact Assessment.

Waste incineration facilities are not a "cutting-edge technology" or "equipment with safe filters," as is sometimes claimed because they are associated with numerous negative impacts, as discussed in Chapter 3.5.3. This is related to the fact that proponents of incineration plants often respond to pollution-related questions by claiming that "emissions into the air are under control in the latest generation of 'most modern' waste incineration plants." Behind their claims lie three unsubstantiated assumptions: Firstly, that there are acceptable emission levels for all pollutants emitted by incineration plants (see Chapters 3.1.6 and 5.1.1.1); secondly, that incineration plant emissions into the air are now precisely measured (see Chapter 3.5.3 and 3.1.1.1); and thirdly, that if emissions are measured, they fall within limits currently defined as "acceptable" (see 3.5.5, etc.).

Indeed, the history of waste incineration regulation in Europe demonstrates that over time the regulation of incineration has had to change and account for the serious air pollution (i.e. dioxins) that waste incinerators released for many years with impunity. In 2006 the EU Commission amended the EU Waste Incineration Directive's Best Available Technology Reference document (EU BREF et al 2006) to address the fact that the industry had been releasing dangerous levels of dioxins. Incinerators now require Advanced Pollution Control units which represent a significant and major cost to new and existing operations to control (but not eliminate) their toxic and hazardous air emissions and residues. While these new regulations are often quoted by the industry to expand and develop in many other countries (particularly in the global south) as evidence that they are now safe and non-polluting, by utilizing these 'modern high technology' advanced air pollution control units, the following chapters in this report reflect a different reality.

3.1.1 Air emission limits applied for waste incinerators in EU

The European Council's 1996 directive on Integrated Pollution Prevention led to the introduction of BAT. In the EU (OECD 2020), BAT are defined as "most effective and advanced stage in the development of activities and their methods of operation, indicating the practical suitability of particular techniques for providing the basis for emission limit values and other permit conditions designed to prevent and, where this is not practicable, to reduce emissions and the impact on the environment as a whole." BAT and BAT associated emission levels (BAT-AEL) can be found in reference documents known as BREFs (Best Reference Documents). The original BREF on waste incineration was adopted by the European Commission in 2006, last updated version was adopted in 2019.

Table 3.1: *BAT-AEL for emissions to air according to the EU Commission Decision 2019/2010 on Best Available Techniques for Waste Incineration (European Commission, 2019).*

Parameter	BAT-AEL ($mg.Nm^{-3}$)		Averaging period
Dust	<2-5[a)]		Daily average
Cd+Tl	0.005–0.02		Average over the sampling period
Sb+As+Pb+Cr+Co+Cu+Mn+Ni+V	0.01–0.3		Average over the sampling period
	New Plant	**Existing Plant**	
HCl	< 2–6[b)]	< 2–8[b)]	Daily average
HF	< 1	< 1	Daily average or average over the sampling period
SO_2	5–30	5–40	Daily average
NO_x	50–120[c)]	50–150[c) d)]	Daily average
CO	10-50	10-50	Daily average
NH_3	2-10[c)]	2-10[c) e)]	Daily average
TVOC	3-10	3-10	Daily average
Hg[f)]	< 5–20[g)]	< 5–20[g)]	Daily average or average over the sampling period
	1–10	1–10	Long-term sampling period

Parameter	ng $I\text{-}TEQ.Nm^{-3}$		
	New Plant	**Existing Plant**	
PCDD/F[h)]	< 0.01–0.04	< 0.01–0.06	Average over the sampling period
	< 0.01–0.06	< 0.01–0.08	Long-term sampling period[i)]
	ng $WHO\text{-}TEQ.Nm^{-3}$		
PCDD/F + dioxin-like PCBs[h)]	< 0.01–0.06	< 0.01–0.08	Average over the sampling period
	< 0.01–0.08	< 0.01–0.1	Long-term sampling period[i)]

a) For existing plants dedicated to the incineration of hazardous waste and for which a bag filter is not applicable, the higher end of the BAT-AEL range is 7 $mg.Nm^{-3}$.
b) The lower end of the BAT-AEL range can be achieved when using a wet scrubber; the higher end of the range may be associated with the use of dry sorbent injection.
c) The lower end of the BAT-AEL range can be achieved when using SCR. The lower end of the BAT-AEL range may not be achievable when incinerating waste with a high nitrogen content (e.g. residues from the production of organic nitrogen compounds).
d) The higher end of the BAT-AEL range is 180 $mg.Nm^{-3}$ where SCR is not applicable.
e) For existing plants fitted with SNCR without wet abatement techniques, the higher end of the BAT-AEL range is 15 $mg.Nm^{-3}$.
f) Either the BAT-AEL for daily average or average over the sampling period or the BAT-AEL for long-term sampling period applies. The BAT-AEL for long-term sampling may apply in the case of plants incinerating waste with a proven low and stable mercury content (e.g. mono-streams of waste of a controlled composition).
g) The lower end of the BAT-AEL ranges may be achieved when:
- *incinerating wastes with a proven low and stable mercury content (e.g. mono-streams of waste of a controlled composition), or*
- *using specific techniques to prevent or reduce the occurrence of mercury peak emissions while incinerating non-hazardous waste. The higher end of the BAT-AEL ranges may be associated with the use of dry sorbent injection*

h) Either the BAT-AEL for PCDD/F or the BAT-AEL for PCDD/F + dioxin-like PCBs applies.
i) The BAT-AEL does not apply if the emission levels are proven to be sufficiently stable.

Any waste incineration plant built in the European Union must use the best available techniques for waste incineration in order to avoid excessive emissions. The use of BAT implies emission limit values (BAT-AELs), which Member States have to consider as the maximum possible and which can be reduced (tightened) if necessary at a national level. Their overview in Table 3.1 is based on the EU Commission Decision 2019/2010 on Best Available Techniques for Waste Incineration (European Commission, 2019). Some substances are monitored but do not have a BAT-AEL, such as N_2O, PBDD/F or benzo[a]pyrene. NO_x, NH_3, CO, SO_2, HCl, HF, Hg and dust (PM) are supposed to be monitored continuously with some exceptions. Frequency of monitoring of other pollutants is different.

3.1.1.1 Long-term sampling of dioxins and mercury

In the case of dioxins and mercury (as well as some other pollutants), monitoring frequency can be on a one-off basis or on a long-term basis (semi-continuous or continuous monitoring). Although there is a "possible" choice between both options in EU BAT, in several cases, the short-term, one-time measurement option is often chosen, even though it has been shown that one-time measurements may not be informative at all for combustion plants (ENDS, 2006). Some operators of incinerators use this method to avoid stricter monitoring requirements. Especially for dioxins, incinerators are identified as minimally controlled and/or uncontrolled dioxin sources due to insufficient monitoring (Cheruiyot et al., 2016).

3.1.1.1.1 Long-term sampling of dioxins

In Belgium, one-time measurements for dioxins have shown that emissions have been underestimated, while by comparison, a semi-continuous AMESA[17] system demonstrated that these same emissions were detected at levels several times higher (De Fré & Wevers, 1998; Reinmann, 2011). This is why Belgium decided to legislate for the semi-continuous measurement of dioxin emissions from municipal waste incinerators – to check that incinerators comply with the limits set by legislation. Later, hazardous waste incinerators and cement plants were also included. Sweden and France also perform semi-continuous measurements of dioxins. The Netherlands has also experienced a similar issue with the difference between one-time and semi-continuous emissions measurements (Arkenbout, 2018). According to BAT, monitoring dioxins twice a year is permissible, if the dioxin emissions are proven to be stable.

The European Environment Bureau (EEB) recommends long-term dioxin monitoring for waste incineration plants (EEB, 2019) and states that: *"Stable emission levels cannot be determined through periodic measurements taken every six months, such as short-term sampling periods lasting 6-8 hours as required by the IED. Instead, a monthly monitoring frequency using long-term sampling must be established in all cases. Only if these measurements indicate stable emissions can the authority authorize a less rigorous monitoring regime. The competent authority should request monthly monitoring for one year via long-term sampling to assess whether the PCDD/F emission levels are stable enough. This procedure may be repeated every 5 years."* Claims that such monitoring regimes are an unfair financial burden to industry are disputed by the Land and Environmental Court and by the Swedish Supreme Court *"Long-term sampling is 'best possible technique' and with 10 eurocent.t^{-1} is economically viable"*.

On the websites of individual companies planning to build Waste-to-Energy (WtE) facilities in the Czech Republic or abroad, one can find information about the energy recovery from waste. Similarly, CEWEP, the Confederation of European Waste-to-Energy Plants, claims on its website that these strict monitoring regulations deliver incinerator emission outputs that demonstrate that dioxins found in areas around the incineration

17 More about AMESA is available at https://www.envea.global/product/amesa-d/. Other instruments besides AMESA can also measure dioxins semi-continuously – for example GT90 Dioxin+

plant are not related to emissions from the facility. They assert that incineration plants adhere to some of the strictest limits for dioxins (representing less than 0.2 % of industrial dioxin emissions) and that the emission profiles are very similar during periodic and continuous emissions monitoring. However, this contradicts what is evident from this chapter. According to the European Commission (2022) waste incinerators were responsible for 19 % of dioxin emissions into the air in the EU-28 in 2015, see Figure 5.2.

3.1.1.1.2 Long-term sampling of mercury

Mercury is also a concern in this situation. While efforts are made to minimize mercury entering the facility at the incinerator entrance, it is not always successful. To detect sudden changes in mercury emissions, incinerators should continuously measure mercury (European Commission, 2019). However, even this continuous measurement can be avoided if the incinerator demonstrates low and stable mercury content. Mercury peaks can occur unexpectedly, as demonstrated by a German incinerator in Frankfurt's Sindlingen district that burned only sludge (a mono-stream of waste with "assumed" to have stable composition). Even though only sludge enters the incinerator, peaks in mercury concentration occurred. According to EEB (EEB, 2019): *"It is important to monitor and control mercury levels in all waste disposal facilities. The Sindlingen plant even exceeded the 25 µg.m^3 level while burning sewage sludge. Discontinuous mercury monitoring is ineffective in detecting mercury peaks."* To demonstrate a low and stable concentration of mercury in emissions, data must be supplied to permit authorities, as is the case with dioxins.

3.1.2 Air emission limits for waste incineration in USA

In the US, the New Source Performance Standards (NSPS) under the Clean Air Act and emission guidelines (EG) for large municipal waste combustion (MWC) units, is currently underway (and is repeated every

Table 3.2: *Current and proposed emission limits for large municipal waste combustion units according to US EPA. (Source: EPA, 2024)*

Pollutant	Units of measure (at 7 % O_2) for 1 dscm[18]/ ppmdv[19]	Current limit for existing source (2006)	Proposed limit for existing source RDF/S	New source according current limit (2006)	Proposed limit for new source MB/RC
Cd	µg/dscm	35	1.5	10	1.1
Pb	µg/dscm	400	56	140	13
PM	mg/dscm	25	7.4	20	4.9
Hg	mg/dscm	50	12	50	6.1
PCDD/F	ng/dscm	30/35[b]	7.2	13	1.8
HCl	ppmdv	29	13	25	7.8
SO_2	ppmdv	29	20	30	14
NO_x	ppmdv	180-250[ac]	110[a]	150[f]	50[f]
CO	ppmdv	50-250[d]	100-250*	50-150[g]	16-100**

** MB/WW, MB/RC, RDF/S, RDF/SS, RDF/FBC ** MB/WW, MB/RC, RDF/S*

a) NO_x limit based on the 110 ppm (24-hour) NO_x limit being finalized under National Ambient Air Quality Standards (NAAQS). Units equipped with SCR devices will be subject to their currently permitted limit of 50 ppm.

b) 30 ng/dscm for fabric filter equipped MWC units and 35 ng/dscm for electrostatic precipitator-equipped MWC units.

c) Range in limits based on combustor type. MB/WW (205); RDF (250); MB/RC (210); RDF/FBC (180).

d) Range in limits based on combustor type. MB/WW (100); MB/RC (250); RDF/S (200); RDF/SS (250); RDF/FBC (200); modular starved air or modular excess air (50).

e) Reevaluated MACT floor limit for MB/WW (100) and RDF/SS (250) was less stringent than current limit, so is not proposed to change.

f) NO_x limit based on 50 ppm (24 hour) permitted limit for units currently equipped with SCR control devices.

g) Range in limits based on combustor type. MB/WW (100); RDF/S (150); Modular starved air or modular excess air (50).

18 dscm – dry standard cubic meter.

19 ppmdv – parts per million on a dry volume basis.

Photos 3.1 and 3.2: *At public hearings, people are being convinced that nothing more than water vapor or carbon dioxide will come out of the chimney of the future incinerator. But how can we believe that when continuous emission sampling has not been tested in most incinerators, as it has, for example, in Harlingen (see Chapter 3.5.3)? Illustrative photos from SYCTOM WI in Paris, Jane Bremmer, December 2015.*

five years). In Table 3.2, you can see the current and proposed limits for existing facilities in the first and second columns with limits, and the current and proposed limits for new sources in the third and fourth columns (EPA, 2024).

3.1.3 Emission limits, case study: Czech Republic

Air emissions are usually the focus of most attention in the Environmental Impact Assessment of incinerators, especially for substances for which emission limits are set. In the Czech Republic, the level of emission limits is adjusted by Regulation No. 415/2012 Sb. Limit values can be differentiated for waste incineration plants, cement kilns co-incinerating waste, cement rotary kilns co-incinerating waste or other plants performing thermal treatment of waste. The specific emission limits for combustion stationary sources are related to the total rated thermal input and to normal conditions (273 K, 101.32 kPa), converted to dry gas. For solid fuels, they are related to the calculated reference oxygen content (11 % by volume).

As can be seen from the emission limit values, the tightening of the BAT conclusions for waste incineration has not yet been reflected in the national emission limit values. For example, only short-term measurements of dioxins and metals are sufficient.

Table 3.3: *Emission limits for pollutants according to Regulation No. 415/2012 Sb. (Source: Ministry of the Environment of the Czech Republic ČR, 2012)*

Pollutant	(mg.m-3)
Emission limits for pollutants detected primarily by continuous measurement (daily average)	
Particulate matter (similar to dust)	10
TOC (not TVOC)	10
HCl	10
HF	1
SO2	50
NOx	400 or 200
CO	50
NH3	
Emission limits for pollutants detected primarily by single measurement	
Hg and its compounds	0.05
Cd+Tl and their compounds	0.05
Sb+As+Pb+Cr+Co+Cu+Mn+Ni+V and their compounds	0.5
PCDD/F	0.1 ng TEQ.m-3

3.1.4 Mercury

In addition to dioxins, incinerators also release other harmful substances into the air, such as mercury. Chapter 5.3.6 focuses on the impact of mercury on human health. Most mercury emitted from incinerators is usually captured by a combination of activated carbon and fabric filters. Mercury, along with dioxins or other organic substances, can condense on the surface of activated carbon.

Photo 3.3: *Mercury accumulates in fish in the environment, which is a major exposure route for humans. Photo: Jindřich Petrlík, Arnika.*

3.1.5 Other Metals

In addition to dioxin and mercury, heavy metals are released into the air, like lead, cadmium, arsenic, chromium, or beryllium. Chapter 5.3 delves more into heavy metals and their impact on health.

3.1.6 Particulate Matter

Solid particles, also described as Particulate Matter (PM) emitted into the air from incinerators can vary in size, shape, and surface area. These particles reflect the heterogenous nature of waste as a fuel feedstock for incinerators, containing numerous individual, cumulative and synergistic substances. They can carry other pollutants like metals or organic

Photo 3.4: *Today, we associate the decline of spruce monocultures mainly with bark beetles, but their damage (weakening) is primarily caused by acid precipitation (rain and fog), contributed by emissions from incinerators. Dead forest in the Jizera Mountains, Czechia. Photo: Lovecz, Public domain, via Wikimedia Commons.*

substances. The smaller these particles, the deeper they can penetrate into the respiratory system. Particles smaller than 2.5 μm can reach the lung's air sacs. Such particles are associated with asthma, reduced lung function, respiratory issues, heart disorders, and increased mortality (Vohra et al., 2021).

WHO's recommended air quality standards are 15 μg.m^{-3} for PM_{10} annually and 5 μg.m^{-3} for $PM_{2.5}$. These recommendations stem from epidemiological studies, unlike the derived economic and technological measures used in the EU BAT for waste incineration. Further details are in Chapter 5.4.

3.1.7 Gases

A range of acidic gases (HCl, HF, HBr, or SO_x) are emitted from incinerators and can corrode incineration plant equipment (refer to Chapters 7 and 9.2), worsen respiratory problems, and contribute to acid rain formation. Nitrogen oxides (NO_x) are difficult to remove directly from flue gases as they are chemically neutral, contributing to the formation of photochemical smog.

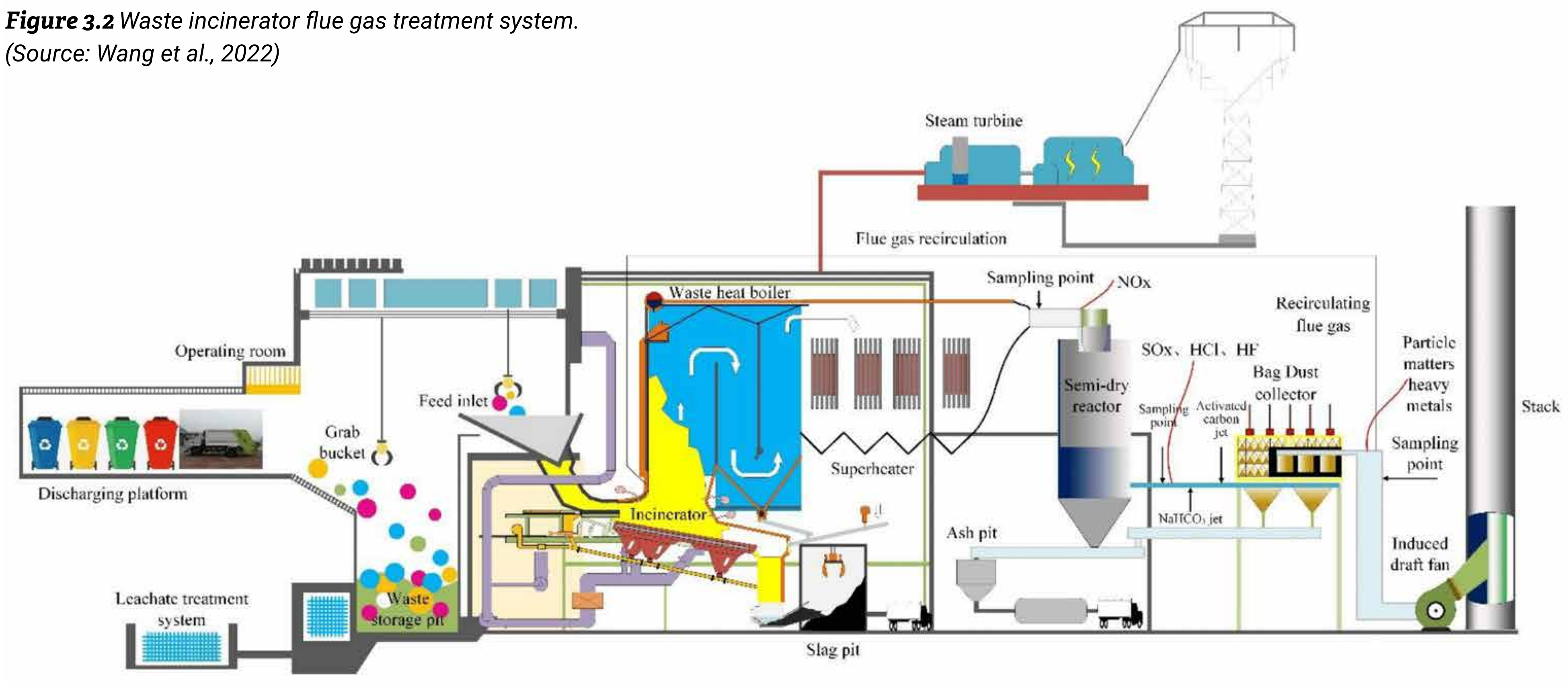
3.1.8 Flue Gas Cleaning

The composition and quantity of emissions from waste incinerators largely depends on the types and quantities of the waste feedstock, the combustion conditions, and the design, and operational parameters of the flue gas cleaning system. Generally, gases are directed to a post-combustion chamber where basic combustion conditions must be met i.e. minimum flue gas temperature of 850°C or 1,100°C (depending on chlorine content in the fuel), minimum oxygen concentration (6 % by volume), and minimum flue gas residence time (2 seconds). Adhering to these conditions ensures compliance with emission limits for carbon monoxide and organic substances in the flue gases.

Sufficiently high temperatures in the post-combustion chamber are essential for initiating and maintaining equipment operation. Thus, if the temperature drops during the combustion process, the waste feed into the facility must be halted. An auxiliary burner, activated automatically during temperature decline, restores the temperature. Other emission limits are achieved through chemical or physicochemical flue gas treatment.

Flue gas treatment principles remain consistent between waste-to-energy facilities and waste incinerators, differing mainly in gas volumes (significantly higher in waste-to-energy facilities due to increased capacity). These processes are quite costly, for example expenditure on flue gas

Figure 3.2 *Waste incinerator flue gas treatment system.*
(Source: Wang et al., 2022)



cleaning technologies constitute 30 % to 50 % of the initial investment in the entire facility (Vejvoda et al., 2018). Specific investment costs for a new MSWI installation according to chosen flue-gas cleaning can be seen in Table 3.4.

Flue gas treatment systems for waste incinerators are very similar to those used by other fossil fuel energy generation plants (for dust, sulfur dioxide, or nitrogen oxides). The concentration of substances in flue gases exiting the post-combustion chamber depends on the type of waste being incinerated. (Vejvoda et al., 2018). According to waste incineration plant designers these emissions are likely to be:

- Solid pollutants (dust) range up to 4 g.m^{-3}.
- Heavy metals as gases are in units of mg.m^{-3}.
- Gaseous pollutants include:
- Hydrogen chloride (600–1,500 mg.m^{-3})
- Hydrogen fluoride (3–300 mg.m^{-3})
- Sulfur dioxide (200–500 mg.m^{-3})

- Nitrogen oxides (200–500 $mg.m^{-3}$)
- Unburned hydrocarbons (in units of $mg.m^{-3}$)

Waste incinerators usually require flue gases exiting the chimney to maintain temperatures of at least 100–110°C.

3.1.8.1 Dust (particulate matter) removal

Dust separation from flue gases typically initiates the waste gas cleaning process and its removal is essential for all incinerator operations. By removing solid particles, the concentration of condensing substances on them (such as metals, metalloids or organic compounds incl. dioxins) decreases.

Electrostatic precipitators and fabric filters are commonly used. Less frequently used are multicyclones or Venturi scrubbers (a type of wet scrubber). Cyclones and multicyclones are less efficient and should only be used as a prefiltering step prior to the removal of coarser particles from the flue gas treatment system. Pre–separation of coarse particles will decrease the amount of fly ash contaminated with high loads of persistent organic pollutants. Electrostatic precipitators offer low flow resistance for gas streams and can handle gases up to 350°C; however, meeting the emission limit of 10 $mg.m^{-3}$ is challenging, yet they may be considered in situations where waste composition varies rapidly (e.g. hazardous waste incinerator).

Fabric filters are more commonly used to meet the emission limits and for handling gas temperatures below 250°C. To minimize formation of dioxins and other chemicals listed in Annex C of Stockholm Convention, both (electrostatic precipitators and fabric filters) should be operated below 200 °C. Fabric filters are usually made of expanded polytetrafluoroethylene (PTFE[20]), which captures solid particles on its surface, forming a filtration cake and when coupled with semi-dry sorbent injection (spray drying), it provides additional filtration and reactive surface. Fabric filters, including the filtration cake, impose significant gas flow resistance, making the process demanding for flue gas fans. Pressure- drop across fabric filters should be monitored to ensure filter cake is in place and bags are not leaking or getting wet. Additionally, fabric filters cannot handle moist flue gases, therefore gas streams must be maintained above the dew point (130-140 °C) and are more expensive than electrostatic precipitators, requiring higher investment and maintenance.

Venturi scrubbers are used when separating submicron particles (sized in tenths of micrometers) but require effluent treatment and are usually employed following dedusting.

Fixed or moving-bed adsorption can be employed as a next step (aimed at mercury, metals and metalloids as well as organic compounds incl. PCDD/F) and act as an effective polishing filter for dust. Double filtration

Table 3.4 *Specific investment costs for a new MSWI installation related to the annual capacity and some types of Flue Gas Cleaning (FGC) in Germany (European Commission, 2019)*

Type of flue-gas cleaning	Specific investment costs (EUR.t^{-1} waste input.y^{-1})			
	100 kt.a^{-1}	200 kt.a^{-1}	300 kt.a^{-1}	600 kt.a^{-1}
Dry	670	532	442	347
Dry-plus	745	596	501	394
Dry plus with residue processing	902	701	587	457

20 PTFE is more commonly known as Teflon.

(filters in series) can routinely achieve collection efficiencies for dust at or below 1 $mg.m^{-3}$. Flue gas polishing may have greatest utility at large installations and in further cleaning of gas streams prior to selective catalytic reactions.

3.1.8.2 Acid gas removal

Boiler sorbent injection is used for partial abatement of acid gas emissions upstream of other techniques. Magnesium- or calcium-based absorbents are injected at a high temperature in the boiler post-combustion area, to achieve partial abatement of acid gases. The technique is highly effective for the removal of SO_x and HF and provides additional benefits in terms of flattening emission peaks.

Pre-dedusting of the gas stream may be necessary to prevent clogging in the next steps.

In order to reduce channeled peak emissions of HCl, HF and SO_2 to air from the incineration of waste while limiting the consumption of reagents and the volume of residues generated from dry sorbent injection and semi-wet absorbers, the automated reagent dosage, and/or recirculation of reagents, can be used.

Acid gas removal can be divided into:

- dry systems
- wet systems
- semi-wet (or semi-dry) systems

Dry systems capture HCl, HF, and SO_2 on solid sorbents in a fluidized or moving bed reactor through separation on a fabric filter ($NaHCO_3$, CaO). This process generates a considerable amount of waste product (see Table 3.6). With regards to acid gas removal, dry scrubbing systems cannot reach the efficiency of wet or semi-wet (spray dry) systems without significantly increasing the amount of reagent/sorbent. Increased reagent use adds to the volume of fly ash. Sodium-based sorbents are generally more efficient and cheaper than calcium-based systems (Jurczyk et al., 2016).

Wet systems involve absorbing acidic gases with alkali-reacting solutions or suspensions (Vejvoda et al., 2018). In this process, the pH of the scrubber water is a function of removal efficiency. Significant portions of HCl and HF are captured in water, reducing the pH to 0.5–1.0. The remaining HCl, HF, and most of the SO_2 are separated in the second step, at a pH of 6–7. Milk of lime or limestone ($CaCO_3$) can be used as a neutralizing agent, resulting in water insoluble residues that contain sulfates, carbonates and fluorides. Using Sodium Hydroxide (NaOH) can avoid this problem since the products of this type of neutralization are soluble. However it can lead to the accumulation of $CaCO_3$ inside the scrubber (Wang et al., 2023). Solid particles in the scrubber water may also cause interaction with PCDD/F. This can influence the reliability of the relationship between results obtained from the periodic stack gas monitoring, the gas monitoring and the plants destruction performance. Additionally, NO_x is separated, which is formed from the nitrogen present in the waste and the oxidation of atmospheric nitrogen in the combustion air. High-temperature NO_x does not form above 850°C. The main wet technologies according to EU BAT are:

- jet scrubbers,
- venturi scrubbers,
- spray scrubbers, and
- packed tower scrubbers

This process generates wastewater requiring the removal of dissolved substances (using FeOH3 and trimercaptotriazine) and pH adjustment (with lime). Using $FeOH_3$ creates a sediment that captures heavy metals

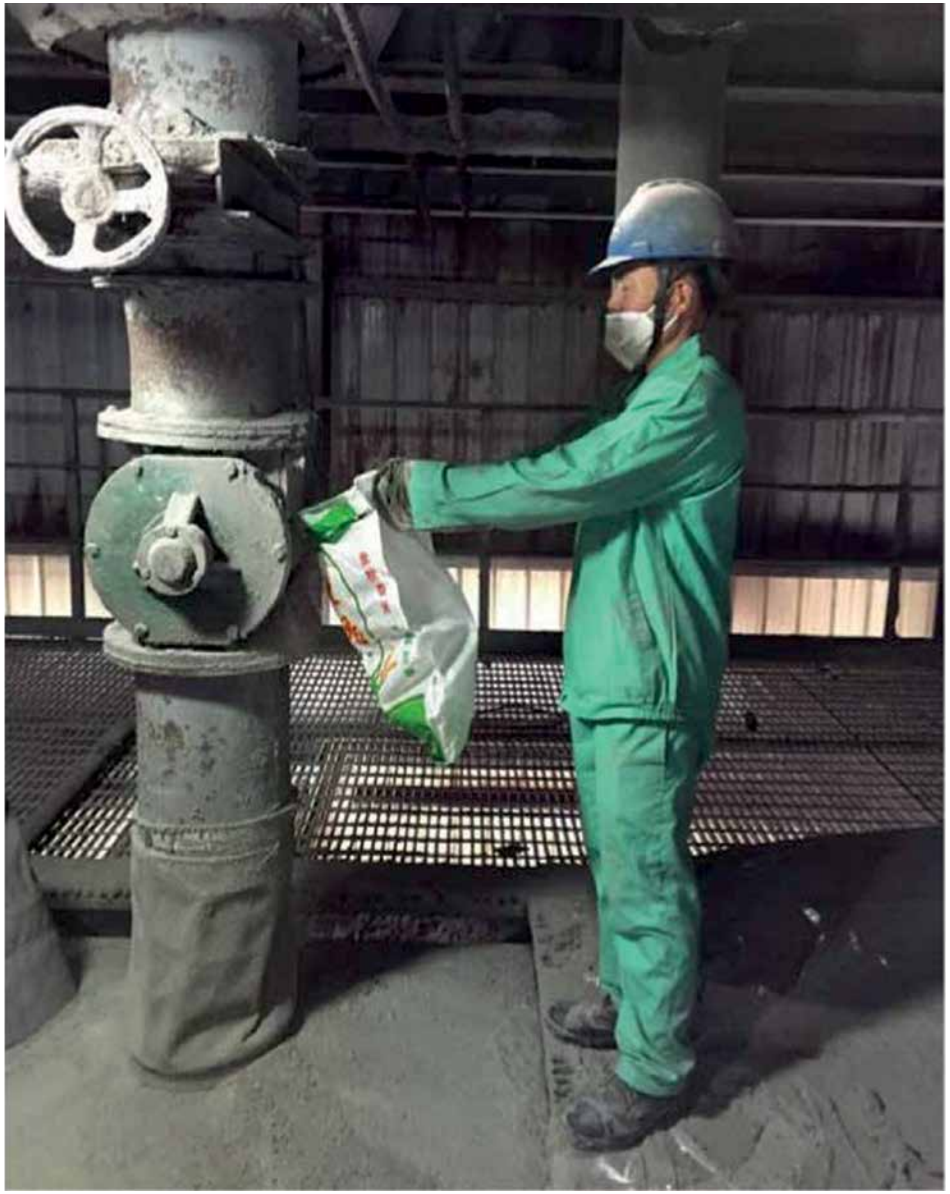


Photo 3.5: *Fly ash sample collection as a byproduct of flue gas cleaning in a Chinese incinerator. (Source: Tang et al., 2016)*

and other impurities on its surface. The sludge is separated using a pressure filter, and the water can either evaporate back into the flue gas or be discharged into the sewer.

Semi-wet scrubbing involves spraying a suspension into a spray dryer. When the suspension (or solution) comes into contact with acid gases, the water evaporates and the reaction product is dry. It needs to be separated, for instance, in a fabric filter or by electrostatic precipitator, and is treated as hazardous waste. Reaction products can be collected along with residual fly ash at the boiler exit. Evaporation of water is also the reason why this process cools the flue gases. To prevent condensation and corrosion of filter bags, a temperature above 130°C – 140 °C is required. To remove chlorinated dioxins, activated carbon is dosed before the fabric filter (it is also effective in removing mercury). When using lime, an excessive amount of it is added, generating a larger amount of waste product (see Table 3.6); (Stockholm Convention on POPs, 2019). When using $Ca(OH)_2$ or NaOH for scrubbing, the flue gases must be cooled before entering the scrubber. $Ca(OH)_2$ reacts with acidic components at temperatures of 120 – 180 °C. Spray dry scrubbing systems typically achieve 93 % SO_2 and 98 % HCl control.

Main semi-wet technologies are:

- injecting sorbent into the boiler,
- circulating fluidized bed (CFB) dry purifier,
- inline sorbent injection (DSI), and
- atomizer, spray dryer absorber (SDA)

For solid products, see Table 3.6.

3.1.8.3 Nitrogen oxides (NOx) removal techniques

Selective non-catalytic reduction (SNCR) is used to remove nitrogen oxides using ammonia or urea. One of these substances is sprayed into the incineration plant's post-combustion chamber at temperatures of 800–1,000°C.

Selective catalytic reduction (SCR) is usually employed after dedusting and acid gas removal and differs by using a catalyst (V_2O_5 + MoO_3 on TiO2 or Al_2O_3) at much lower temperatures of 300–350°C (Vejvoda et al., 2018) or to 250–400°C (Stockholm Convention on POPs, 2019). Unlike SNCR, SCR results in less remaining ammonia in the flue gases and aids in reducing the volume of chlorinated dioxins and other gas phase chemicals listed in Annex C with an efficiency of 98–99.9 % (Neuwahl et al., 2019) in the flue gases, although it's an expensive and energy intensive process primarily used in larger facilities.

3.1.8.4 Reduction of organic compounds including PCDD/F and PCBs

Reduction of organic compounds starts with the optimization of the incineration process and with control of the waste feed and can continue with on-line and off-line boiler cleaning to reduce the dust residence time and accumulation in the boiler. When shutting down the boiler, rapid flue gas cooling from a temperature of 400°C to 250°C to prevent de novo synthesis of PCDD/F, is recommended. This can be achieved by appropriate design of the boiler and/or with the use of a quench system. The latter option limits the amount of energy that can be recovered from the flue gas and is used in particular for the incineration of hazardous waste with a high halogen content (Stockholm Convention on POPs, 2019).

For dioxin removal from flue gases in waste incinerators, sorption methods using activated carbon materials or catalytic processes are utilized. The use of carbon-impregnated materials, activated carbon, or coke in scrubber packing materials, can achieve a 70% reduction in PCDD/PCDF across the scrubber (Stockholm Convention on POPs, 2019) but this may not be reflected in overall releases. Apart from the previously mentioned SCR process, a fabric filter made of expanded polytetrafluoroethylene containing an internal catalytic layer can be employed to reduce dioxin content in flue gases. The entire process occurs at temperatures between 180–260°C, and with an input concentration of 10 ng TEQ.m3, it is possible to achieve 0.1 ng TEQ.m^{-3}. The estimated lifespan of the filter is a minimum of 5 years.

The most common way to remove dioxins by adsorption is through activated carbon. Granular adsorbents can be dosed onto fixed beds (smaller facilities), moving beds (larger facilities), or before a fabric filter (powdered adsorbent). Alongside dioxins, some heavy metals (Hg, Cd) are also captured. However, the use of activated carbon directly promotes the formation of additional dioxins, potentially leading to a 30% increase in dioxin levels (Chang & Lin, 2001).

Carbon sorbent in a wet scrubber: PCDD/F and PCBs are adsorbed by carbon sorbent added to the wet scrubber, either in the scrubbing liquor or in the form of impregnated packing elements. The technique is used for the removal of PCDD/F in general, and also to prevent and/or reduce the re-emission of PCDD/F accumulated in the scrubber (the so-called memory effect) occurring especially during shutdown and start-up periods (Stockholm Convention on POPs, 2019).

The amount of waste generated by flue gas cleaning (per ton of waste) is summarized in Table 3.6 (in Chapter 3.3).

3.1.8.5 VOC removal

Marziali et al. (2024) also mention volatile organic compounds (VOC) removal, which can be abated by different techniques, especially by the optimization of the combustion process and the waste feed, dry sorbent injection, adsorption and SCR with alternatives (Corbasson et al., 2022). These alternatives, for example, membrane for the separation of VOCs (Gan et al., 2023) or non-thermal plasma (Martini et al., 2019) are accompanied by unwanted by-products (Schiavon et al., 2017).

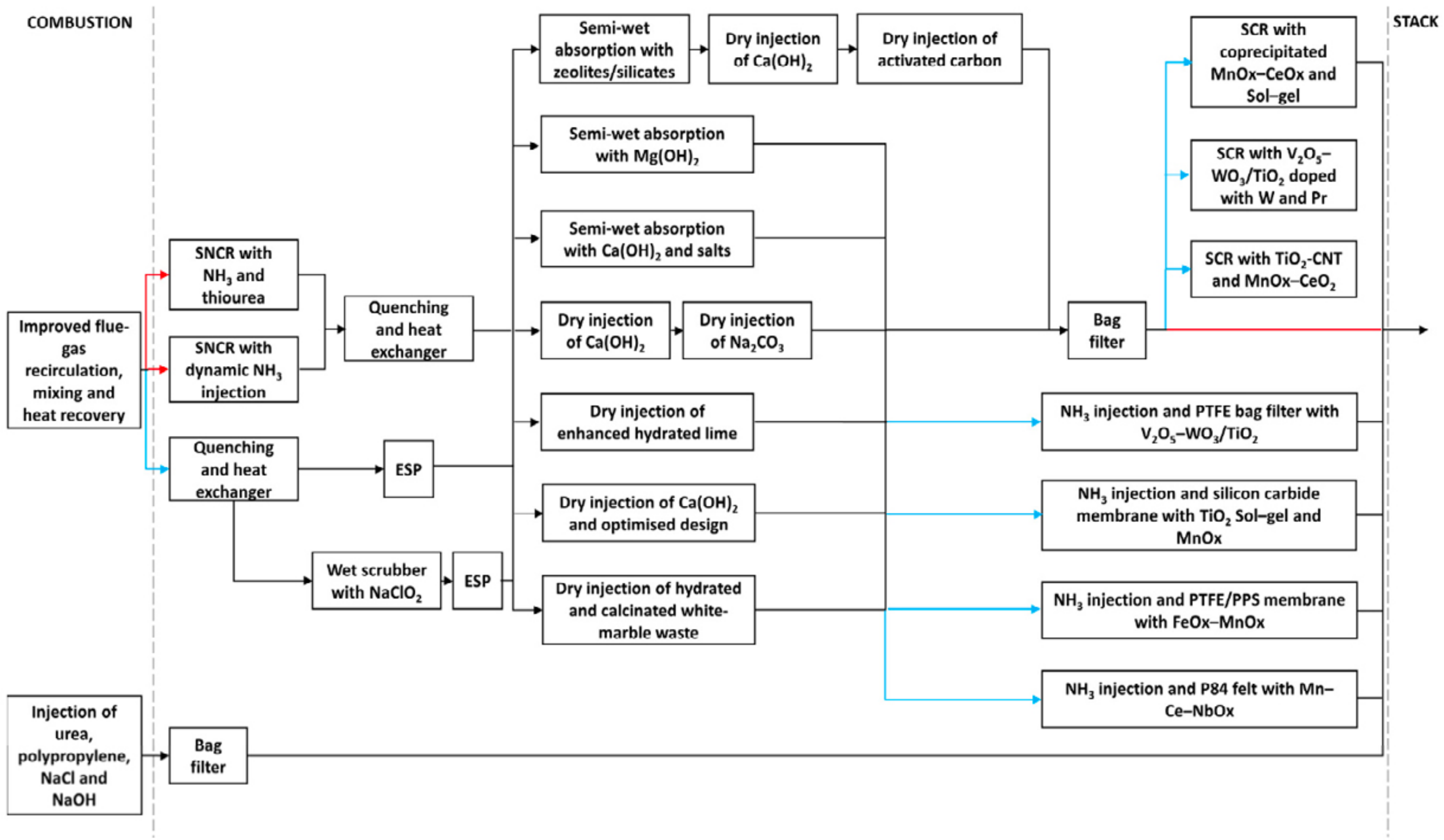


Figure 3.3: *Proposed configurations for the implementation of the novel techniques reviewed for the abatement of gaseous pollutants from waste incineration plants. Paths specifically intended for configurations with SNCR and SCR are represented in red and blue respectively. Source: Schiavon et al., 2024*

3.1.9 Transportation Emissions

Most planned and existing waste to energy incinerator facilities rely on road transport (rail transport being a rare exception) for bringing waste feedstocks and auxiliary substances and chemicals into the facility and for the transport of bottom and fly ash out of the facility. In addition to the negative impacts on air quality that incinerators generate, the associated transport emissions of exhaust gases and noise from accompanying traffic are also significant. Mainly, emissions of particulate matter ($PM_{2.5}$, PM_{10}), benzene, benzo[a]pyrene, carbon monoxide, nitrogen oxides, and carbon dioxide are observed. Benzene, benzo[a]pyrene, and particulate matter (PM) are classified as Group 1 carcinogens by IARC.[21]

21 The IARC classifies chemicals, physical agents and work processes according to hazard into five groups. Group 1 are proven carcinogens, Group 2 are potential carcinogens (2A – probable carcinogens, 2B – possible carcinogens), Group 3 are substances not evaluable due to lack of scientific evidence and Group 4 includes probable non-carcinogens.

3.1.10 Fugitive Emissions

In addition to the common emissions measured continuously or periodically, there are also what we call 'fugitive emissions'—these aren't released through the chimney but through other pathways. These could include dust emissions with associated compounds, or volatile organic substances, often related to odors. Leakage might occur during container loading/unloading, from storage spaces, conveyor systems, due to poor building sealing or exhaust system failure, or during handling, storage, and transportation of solid residues.

Fugitive emissions were likely responsible for environmental contamination and the presence of PCBs and dioxins in chicken eggs around the Panteg area incinerator in the United Kingdom (Lovett et al., 1998). A similar case was reported in the vicinity of a municipal waste incinerator in Wuhan, China. High concentrations of chlorinated and brominated dioxins were found in eggs from local chicken farming, likely originating from fugitive emissions from fly ash stored in the incinerator yard (see Photo 3.7).

Photo 3.6: *Waste is supposed to be transported to the incinerator in the Mělník, Czech republic power plant by roads, which understandably displeased the local residents. Photo: Jindřich Petrlík, Arnika.*

Photo 3.7: *Ash stored in the yard of a municipal waste incinerator in Wuhan, China—likely a source of fugitive dioxin emissions. (Source: Zhang et al., 2015)*

Photo 3.8: *A worker from the waste incineration plant (WtE) in Geiselbullach, Germany, showing children ash and slag from the incinerator stored in the yard. Fugitive emissions rise from the smoking ash. Photo: HEJ Support, Germany.*

3.2 Emissions to Water

Besides dioxins, waste effluents may contain metals including mercury, inorganic salts, and other organic substances (such as phenols) (Neuwahl et al., 2019).

3.2.1 Waste Incineration Wastewater Treatment

Emissions of chlorinated dioxins into water occur only if wet systems are used for flue gas cleaning. Modern wastewater treatment plants include steps such as neutralization, precipitation, flocculation, and active carbon filters, which remove organic substances from wastewater; however, these methods do not destroy chlorinated dioxins. BAT-AELs for direct and indirect emissions to a receiving body are summarized in Table 3.5.

3.2.2 Emergency Water Leaks

Accidents, fires and other unforeseen events leading to the emergency leakage of toxic substances into aquatic environments from MSW and hazardous waste incinerators, can and do happen. For instance, mercury leaked from the Megawaste incinerator into the municipal sewage system in Prostějov in 2003 (MF Dnes & Jurčová, 2003), and in 2017, oil substances leaked from the largest Czech hazardous waste incinerator in Ostrava into the Odra River (ČTK, 2018a). The company operating the incinerator faced a hefty fine for this leak.

Toxicity tests undertaken in 2006 and 2007 discovered contamination in the water outfall after a fire at the El Dorado hazardous waste incinerator (see chapter 7.2.1.1.) Analytical laboratory tests discovered organophosphate-based surfactants as the most likely source of toxicity in the water (FTN Associates Ltd., 2007). Various PFAS's can be found commonly in fire-water runoffs (Bluteau et al., 2019). Fire-water runoffs after

Table 3.5: BAT-AELs for direct and indirect emissions to a receiving body. Source: European Commission (2019).

Parameter		Process	Unit	BAT-AEL – direct emissions[a]	BAT-AEL indirect emissions[a, c]
TSS		FGC Bottom ash treatment	mg.L^{-1}	10-30	-
TOC		FGC Bottom ash treatment	mg.L^{-1}	15-40	-
Metals and metalloids	As	FGC	mg.L^{-1}	0.01–0.05	0.01–0.05
Metals and metalloids	Cd	FGC	mg.L^{-1}	0.005-0.03	0.005-0.03
Metals and metalloids	Cr	FGC	mg.L^{-1}	0.01-0.1	0.01-0.1
Metals and metalloids	Cu	FGC	mg.L^{-1}	0.03-0.15	0.03-0.15
Metals and metalloids	Hg	FGC	mg.L^{-1}	0.001-0.01	0.001-0.01
Metals and metalloids	Ni	FGC	mg.L^{-1}	0.03-0.15	0.03-0.15
Metals and metalloids	Pb	FGC Bottom ash treatment	mg.L^{-1}	0.02-0.06	0.02-0.06
Metals and metalloids	Sb	FGC	mg.L^{-1}	0.02-0.9	0.02-0.9
Metals and metalloids	Tl		mg.L^{-1}	0.005-0.03	0.005-0.03
Metals and metalloids	Zn	FGC	mg.L^{-1}	0.01-0.5	0.01-0.5
Ammonium-nitrogen (NH_4-N)		Bottom ash treatment	mg.L^{-1}	10-30	-
Sulphate (SO_4^{2-})		Bottom ash treatment	mg.L^{-1}	400-1,000	-
PCDD/F		FGC	ng T-TEQ.L^{-1}	0.01-0.05	0.01-0.05

Note: The averaging periods are defined in the General considerations. The BAT-AELs may not apply if the downstream wastewater treatment plant is designed and equipped appropriately to abate the pollutants concerned, provided this does not lead to a higher level of pollution in the environment.

***Photo 3.9:** The inconspicuous Megawaste incinerator in Prostějov, heating surrounding greenhouses, became the source of a mercury leak in 2003. Photo: Jindřich Petrlík, Arnika.*

using fluorinated firefighting foams should therefore be contained and disposed of properly (Seow, 2013).

On December 22, 2021, the Cologne district government announced that firefighting water containing clothianidin had been discharged into the Rhine river during firefighting efforts at the Leverkusen Chempark hazardous waste incinerator, where a fatal explosion occurred on the 27th July 2021 (see chapter 7.2.1.2). In the weeks following the incident, the Rhine waterworks in the Netherlands detected clothianidin in drinking water extracted from the Rhine for the first time. Natural currents released the remaining liquids and firefighting water containing PFOS, into the Rhine but failed to inform the responsible International Commission for the Protection of the Rhine about the discharge of toxic substances.

3.3 Waste or Solid Residues from Waste Incineration

An overview of solid residues after waste incineration is summarized in Table 3.6. Mostly, incinerators generate bottom ash (200 to 350 kg per ton of burned waste), followed by residues from flue gas cleaning, which usually include fly ash (approximately 25 kg per ton of waste), or occasionally Sorbalite or other materials. Usually, residues from flue gas cleaning are collectively referred to as fly ash, unless it’s filtration cake. The smallest portion of waste incineration residues, but often similarly toxic to fly ash, is the boiler dust generated during the maintenance of the combustion chamber. Residues from flue gas cleaning can account for 2–5 % of the original weight of the burned waste (Petrlik, Bell et al., 2017; Sabbas et al., 2003). Overall, solid residues can occasionally reach up to 40 % of the original waste weight (EA, 2002; Petrlik, Bell et al., 2017). Burning liquid waste generates fewer solid residues (Petrlik & Ryder, 2005).

Photo 3.10: *“Backyard” of one of the French waste incineration plants with piles of residues from waste incineration. Photo: CNIID, France.*

Solid residues are contaminated with persistent organic pollutants (POPs) and heavy metals, whose concentration depends primarily on the technology used, input material, or the incineration facility’s operational methods. Ash is largely an inhomogeneous material—see Photos 3.11, 3.12 and Tables 3.7 and 3.8. However, data extracted from EU technical documents might not fully represent the extent of toxic substances and their concentrations. Substances captured and not released in air emissions end up in the flue gas cleaning system and thus in fly ash or filtration cake. The cleaner the air emissions, the higher the concentration of harmful substances in the residues from flue gas cleaning. These concentrations vary at each incineration plant, making it hard to estimate their exact flows.

Table 3.6: *Weight of solid residues and residues resulting from flue gas cleaning per ton of waste. (Source: BAT/BET SC, 2021)*

Solid residue / process	Weight in kilograms of dry matter per ton of waste
Slag / bottom ash (ash)	200–350
Boiler dust and boiler dedusting	20–40
Residues from flue gas cleaning without dust from filters	
Wet sorption	8–15
Semi-wet sorption	15–35
Dry sorption	7–45
Residues from flue gas cleaning including dust from filters	
Wet sorption	30–50
Semi-wet sorption	40–65
Dry sorption	32–80
Loaded activated carbon	0.5–1

Photo 3.11: *Ash and slag after incinerating municipal waste in a Copenhagen incineration plant. Photo: Erik Refner (information.dk)*

The most reliable method is to measure these concentrations in solid residues after waste incineration. Besides the substances listed in Table 3.7, waste incineration bottom ash also contains, for example, brominated dioxins, hexachlorobenzene (HCB), pentachlorobenzene (PeCB), polybrominated diphenyl ethers (PBDEs), or other POPs (Bell et al., 2023a; Lin et al., 2014; Petrlik et al., 2006). For more information on POPs concentrations, refer to Chapter 5.1.

Fly ash from hazardous waste incineration (see Photo 3.12) or from incinerators in tropical countries, where there's a lot of organic waste from food and tropical fruits and vegetables (see Photo 3.14), might look somewhat different. Additionally, bottom ash from small medical waste incinerators, mainly located in developing countries, is full of unburned sharp objects, glass, and sometimes unburned plastic residues (see Photos 3.15 and 3.16). At the WtE SAKO Brno (Czech Republic) incineration plant in 2004, they tracked the flows of heavy metals, polycyclic aromatic hydrocarbons, polychlorinated biphenyls, and chlorinated dioxins into individual residues after incineration, which they referred to as slag (bottom ash), end-product, fly ash, and solidified ash.[22] An overview of the

22 Fly ash, end-product and solidified ash are APC residues, which are referred to in this study for simplicity as fly ash.

Photo 3.12: *Bunker for bottom ash and slag in a Košice municipal waste incineration plant.*

measurement results is provided in the graph in Figure 3.4 (Bogdálek & Moskalík, 2008), indicating that most of the cadmium, mercury, and dioxins ended up in the residues from flue gas cleaning (fly ash, end-product, and solidified ash). Conversely, other heavy metals, polycyclic aromatic hydrocarbons, and polychlorinated biphenyls were more concentrated in the bottom ash (slag).

A new study examining the presence of PFAS in ash from waste incinerators in various countries (Czech Republic, Netherlands, Philippines, and Thailand) found the highest concentration in ash from a hazardous waste incinerator in the Philippines. PFAS was also found in ash used for road construction in Katwijk, Netherlands (see Photo 3.26

Table 3.7: *Concentration of selected groups of substances in some residues after waste incineration. Source: Stockholm Convention on POPs (2019)*

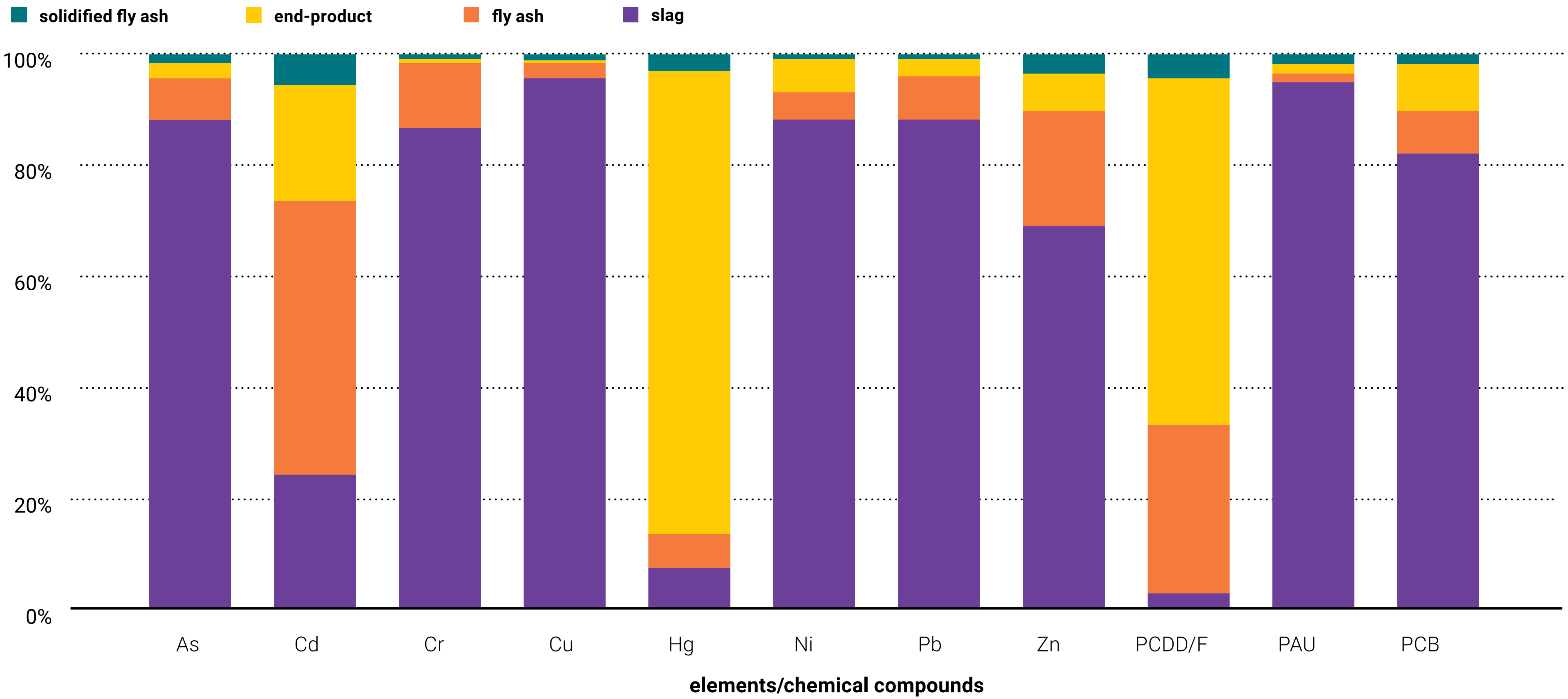
Group of substances	Bottom ash	Boiler ash	Fly Ash
Unit	$ng.kg^{-1}$	$ng.kg^{-1}$	$ng.kg^{-1}$
PCDD/PCDF (I-TEQ)	<1–10	20–500	200 – 10,000
PCBz[a]	<0.002–0.05	200,000-1,000,000	100,000-4,000,000
PCPh[b]	<0,002-0,005	20,000-500,000	50,000-10,000,000
PAH	<0.005–0.01	10,000 – 300,000	50,000 – 2,000,000

PCBz: polychlorinated benzenes
PCPh: polychlorinated phenols

Table 3.8: *Chemical composition of waste incineration bottom ash. (Source: Neuwahl et al., 2019).*

Element	Average (wt. %)	Element	Average (wt. %)	Element	Average (ppm)
SiO_2	*49.2*	P_2O_5	*0,91*	Cr	*648*
Fe_2O_3	*12*	MgO	*2.69*	Ni	*215*
CaO	*15.3*	Na_2O	*4.3*	Cu	*2,151*
K_2O	*1.05*	CO_2	*5.91*	Zn	*2,383*
TiO_2	*1.03*	Sulfates	*15.3*	Pb	*1,655*
MnO	*0.14*	Chlorides	*3.01*		
Al_2O_3	*8.5*				

Figure 3.4: *Graphical representation of the distribution of heavy metals, PCDD/F, PAHs, and PCBs in residues after the incineration of municipal waste in the WtE SAKO Brno (Czech Republic) based on analyses from 2004. (Source: Bogdálek & Moskalík, 2008).*



and Chapter 3.3.3.1) and in the water and sediment from the adjacent body of water (Jelinek et al., 2024).

Concentrations and congener profiles of seven chlorinated benzenes (CBzs) were analyzed in bottom and fly ash samples from a medical and a municipal waste incinerator in northern Vietnam, revealing higher levels in fly ash compared to bottom ash. The study found that higher chlorinated congeners were more abundant, but compositional profiles varied between ash types, incinerators, and sampling days, indicating complex formation processes (Nguyen et al., 2021).[23]

23 For more information on this topic see Chapter 5.2.

Photo 3.13: *Ash from the hazardous waste incinerator in Trmice. Photo: Jindřich Petrlík, Arnika.*

Photo 3.14: *Sampling of ash from the Taiwanese incinerators at Ancing Road in 2016 (Bell et al., 2023a). Photo: Jindřich Petrlík, Arnika.*

Photo 3.15: *Sample of ash from a closed incinerator at a hospital in Accra, sample taken in 2018. Photo: Martin Holzknecht, Arnika.*

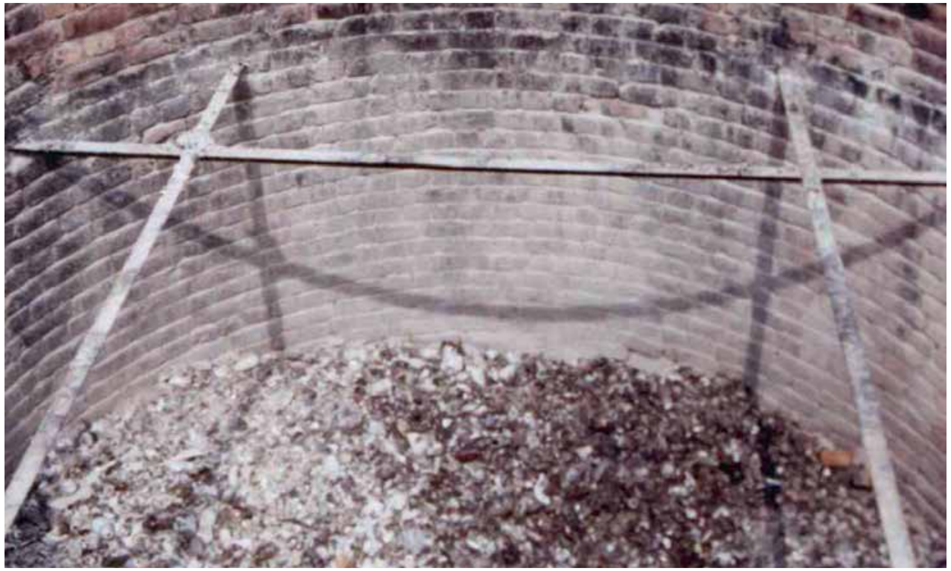


Photo 3.16: *View into a pit with ash from a medical waste incinerator in Lahore, Pakistan. (Source: Petrlik & Khwaja, 2006).*

Photo 3.17: *Waste mound in the rear part of the landfill in Benátky nad Jizerou, mostly composed of bottom ash from WtE Malešice (Prague). Photo: Marek Jehlička (skyworker.cz).*

3.3.1 Processing Waste Containing POPs

As Table 3.7 indicates, waste incineration causes some POPs to be transferred into the solid waste residues, making these wastes subject to regulation under the Stockholm Convention. The aim of this convention is to eliminate selected persistent organic pollutants, although it would be simplest if waste containing POPs were not generated at all. As such, the use and/or storage of waste incineration residues containing POP's is prohibited. This obligation applies to wastes above the limit named in Article 6 of the Stockholm Convention as the Low POPs Content Level (more about LPCL in Chapter 5.1.10), which translates literally as "low level of POPs content". Currently, this limit for dioxins and dioxin like PCBs is set relatively high, at 5,000 pg TEQ.g^{-1} in the European Union (European Parliament and Council of the EU, 2022). Globally, there are two options for dioxin limits (LPCL), namely 1,000 or 15,000 pg TEQ.g^{-1} (Basel Convention, 2023).

There are non-incineration technologies for POPs destruction that can decompose dioxins and other POPs in waste without generating new dioxins or POPs as by-products or causing their release. Mainly used in commercial scales are:

- Gas-phase chemical reduction (GPCR)
- Supercritical or subcritical water oxidation (SCWO)
- Base catalysed decomposition (BCD)
- Catalytic hydrogenation (CH)
- Reduction by alkali metals

The mechanical-chemical decomposition, known as the "ball milling method," is also showing promising development. However, not all of these methods are suitable for removing PCDD/F, dl PCB, HCB, or PeCB from fly ash and other solid residues after waste incineration. Catalytic hydrogenation and reduction by alkali metals have been particularly effective in decomposing PCBs used as transformer or hydraulic oils.

Photo 3.18: *Gas-phase chemical reduction (GPCR) technology used in Australia. (Source: Arnold, 2003)*

For some of these technologies, it is necessary to concentrate POPs into a smaller volume of material from wastes, for example, through indirect thermal desorption (Basel Convention, 2023; Bell, 2020; Petrlik, Bell et al., 2017).

Gas-phase chemical reduction (GPCR) requires contaminants to be in the gaseous phase (they must first be released from solid matrices). The process involves the thermochemical reduction of organic substances at 850°C. At this temperature and under low pressure, organic substances react primarily with hydrogen to form methane, HCl (if chlorine is present), and small amounts of low molecular weight hydrocarbons. HCl is neutralized by adding sodium hydroxide during the inlet gas cooling process or can be removed for further use. This method effectively decomposes DDT, HCB, PCB, and PCDD/F. It can be applied to all matrices containing POPs but does not remove present metals. Its advantage lies in its high efficiency and the absence of creating Unintentional Persistent Organic Pollutants (UPOPS) (Arnold, 2003).

Supercritical Water Oxidation (SCWO) takes place in a closed system using an oxidizing agent—oxygen, hydrogen peroxide, nitrites, or nitrates in water in either supercritical (374°C and 218 atm) or subcritical conditions. Under these conditions, substances become more soluble in water and are oxidized into CO_2, water, inorganic acids, and salts using the oxidizing agent. The destruction efficiency is generally higher than 99.99 % for organic substances (e.g., pesticides, PCDD/F, or flame retardants). The method is applicable to all POPs, suitable for aqueous and oily liquids, solvents, and solid particles smaller than 200 µm. Concentrated wastes need to be diluted (to 20 % w/w). In subcritical water oxidation, water at temperatures above 100°C remains in a liquid state, which has the potential to remove POPs from fly ash. This method has been used, for instance, to remove POPs from sediments (Weber et al., 2002). SCWO technology can also handle the decomposition of per- and polyfluoroalkyl substances (PFAS) in wastes (Austin et al., 2023).

Base catalysed decomposition (BCD) takes place in the presence of alkali metal hydroxide and a catalyst. It consists of two phases. BCD requires concentrating POPs into process oil and hence uses a pre-treatment unit. When heated above 300°C, the first phase involves thermal desorption of organic substances, and in the second phase, these substances react with a basic mixture (NaOH) at 236°C. This method is particularly suitable for PCB, PCDD/F, HCB, PeCB, and chlorinated pesticides (e.g., DDT, chlordane, HCH). This method can treat high-concentration POPs wastes or soils. For soils, it may need to be processed into smaller particles and may require pH and moisture adjustments (Petrlik, Bell et al., 2017). This system has been used in the Czech Republic

Photo 3.19: *Reactor for base catalysed decomposition (BCD) used in Spolana Neratovice, Czech Republic, for dioxin and organochlorine pesticide decomposition. Photo: Jindřich Petrlík, Arnika.*

at the Spolana Neratovice site for decomposing dioxins, organochlorine pesticides, and other POPs from a contaminated area (IPEN et al., 2003; Kubal et al., 2004).

The mechanical-chemical hydrodechlorination process (ball milling method) is carried out at low temperatures. Mitoma et al. (2011) used this process to effectively remove all traces of PCDD, PCDF, and PCB from municipal waste incineration fly ash. They found that the most suitable agent for decomposition was a mixture of metallic calcium and calcium oxide. A sample of fly ash with 5,200 pg TEQ.g^{-1} of dioxins and dl PCB was ground overnight in a ball milling at a speed of 400 revolutions per minute, resulting in complete detoxification (no traces of PCDD, PCDF, or PCB were detected).

This chapter mainly focuses on technologies capable of dealing with dioxins in waste from waste incineration, but non-incineration technologies for POPs decomposition in other wastes are covered in numerous studies containing detailed information on their utilization and effectiveness. They include alternatives to incinerating PCB and other POPs-containing waste (Bell, 2020; IPEN et al., 2003; McDowall, 2010; McDowall, 2007; US EPA, 2010; Weidlich et al., 2020).

3.3.2 Are residues from incinerators hazardous waste?

One of the important characteristics determining whether waste is considered hazardous or not is its potential to leach elements or groups of substances (see also Chapter 3.3.2.1). The hazardous properties of waste can be reduced by altering certain physical and chemical properties, particularly by converting it into a less soluble and less mobile product, while the physical nature of the waste may remain the same. Commonly used procedures include:

Photo 3.20: *This is what solidified fly ashes (mixed with cement) look like at the Yan Chao site in Taiwan (Bell et al., 2023a). Photo: Jindřich Petrlík, Arnika.*

- Solidification
- Encapsulation
- Cementation

Solidification, the conversion of liquid or loose waste into solid material, does not reduce the content of hazardous substances. During chemical fixation, small waste particles (molecules or atoms) react with components of the solidification medium or form a mixture with it. This leads to the "fixation" of waste in the mixture.

During encapsulation, the medium constrains and immobilizes the waste particles, isolating them from the environment (particles do not mix with the medium; the medium surrounds them). One method of immobilizing so-called final waste is vitrification. Enclosing heavy metals in a hard physical matrix significantly reduces their biological availability and the rate at which they can re-enter the environment. Disadvantages of encapsulation include costs and energy consumption.

Photo 3.21: *Pastor R. L. Gundy of Mount Sinai Missionary Baptist Church, who has been diagnosed with prostate cancer, lived in 2009 in Jacksonville, USA, in the vicinity of a waste incineration ash dump. According to the 1990 U.S.Census, more than 30 thousand inhabitants lived in the area of four sites contaminated with ash. (Sources: Morrison, 2009; Petrlik and Bell, 2017; US EPA ROD, 2006).*

Cementation fixes waste into a silicate matrix. Both organic and inorganic wastes can be fixed in bitumen (if they withstand the temperature of molten bitumen); (Kafka & Vošický, 1998).

Dioxins will be a concern for incinerator, or other authorities that intend to use bottom ash (slag) for road construction or embankments. Following France's example (French Republic, 2011), the Czech Republic has established a limit for dioxin content for such uses at 10 ng $I\text{-}TEQ.kg^{-1}$ of dry matter (Ministry of the Environment, 2021e). The same regulation sets limits for other indicators, such as heavy metals (Ministry of the Environment, 2021e).

Ash processing is still an emerging industry, primarily developing since the 1990s, where no two ash processing facilities are the same (Bunge, 2019). Processing usually occurs off-site by another commercial entity and often after transportation across regional or state borders (Arkenbout, 2019). This can happen, for example, after transportation from Switzerland to Germany (Petrlik & Ryder, 2005).

Practices like using fly ash (or mixtures of fly ash with bottom ash) containing pollutants in road construction or technical landfill security do not make sense in terms of the amount of money and effort spent to capture them from waste incinerator emissions if, as a result of this practice, they are released back into the environment (see, for example, 5.1.1.3.3).

3.3.2.1 Leachate tests' deficiencies

The Rollinson report (Rollinson, 2022) compared various testing methods for solid residues from waste incineration used in different countries, particularly in Europe. Results which varied depending on the chosen method, determined the potential for further use of the samples.

For instance, Glauser et al. (2021) discovered that the Dutch column test and the Swiss test produced similar results (statistically significant) for Cu and Cl-, but different results for Zn or Pb. Kalbe and Simon (2020) studied different methods on the same sample and found that Cd, Co, Ni, and Pb leached out of bottom ash in greater amounts in lysimetric and column tests than in batch tests. Conversely, Cl, Sb, and Sn showed the opposite trend, confirming the conclusions reached by Glauser et al. (2021). Kalbe and Simon (2020) argue that lysimetric tests produce results that are closer to the conditions in landfills due to the larger sample volume and direction of flow rate, but no European country currently uses this type of test. The choice of test method can lead to discrepancies in allowing certain countries to use incinerator bottom ash. For instance, the results of batch tests and Dutch leachate tests can differ by more than twofold (Rollinson, 2022).

Allam et al. (2019a) used a test that differs from the one used in the Netherlands but is common in other countries. They found that this procedure would cause the samples to exceed the Dutch legal limits for Cu, Cr, Mo, and Sb, as well as Cl- and SO42-. Therefore, the use of bottom ash as a building material would not be allowed in Netherlands. Allam et al. (2019b) tested the sequential leaching test, exposing the sample to different conditions that represent a worst-case scenario for bottom ash under natural conditions. However, no European country currently uses this test. The test revealed that Zn is highly mobile at low pH, while Cr, Cu, Ni, and Sb are highly mobile under oxidizing conditions. The authors suggest that toxic elements complex with humic substances, which can become mobile when organic matter is oxidized. These findings support the idea that sterile leaching tests do not reflect realistic conditions, as they do not account for the interaction with organic matter.

Different test methods can lead to varying results, affecting the possibility of meeting the limits for any further use. Other factors such as bottom ash weathering, pH, buffering, humic substances, and grain size may also play a role. Tests indicate that even removing finer fractions does not reduce toxicity (Kalbe and Simon, 2020; Vateva and Laner, 2020; Mantovani et al., 2021; Caviglia et al., 2019).

3.3.3 Where do residues from waste incineration end up?

Despite the Best Available Techniques (BAT) documents of the Stockholm Convention (for municipal waste incineration) recommending fly ash and residues to be handled separately and that fly ash should not be used in agriculture or similar areas, these recommendations are unfortunately often ignored. This is connected to the current, excessively lenient Low POPs Content Level (LPCL) limit – see Chapters 3.3.1 and 5.1.10 (Petrlik, Bell et al., 2017).

A global report produced in collaboration with Arnika, the International Pollutants Elimination Network (IPEN), and the National Toxics Network (Australia) released in 2017 mapped several hundred studies documenting numerous cases proving that waste containing POPs (fly ash and other residues from flue gas cleaning), even with dioxin concentrations below the interim Low POPs Content Level set at 15,000 pg TEQ.g^{-1}, can cause serious problems. Among these cases would fall the example of WtE Termizo Liberec (Czech Republic), from which 25,000 to 40,000 tons of fly ash and slag mixture were deposited in landfills designated for municipal waste. However, very few publicly accessible documents exist regarding this matter (Petrlik & Ryder, 2005). Arnika Association discovered this mixture in Czechia in the forest, below the Větrov landfill near Frýdlant and in the cycle path of the Jizera Mountains Protected Landscape Area, where it was used by the company Strabag (Petrlík et al., 2007).

Although there are technologies available for reducing POPs in waste, the generators of waste are not obliged to use these technologies due to the excessively high LPCL limit set in the European Union and individual states. Consequently, continuous contamination of the environment occurs.

For example, in the Czech Republic – information about the quantity and locations where these materials are used, is completely absent. In other words, there is no registry of these sites, and it seems that even the Ministry of the Environment of the Czech Republic has no overview of where bottom ash and fly ash from waste incineration end up (Arnika, 2019a).

Photo 3.22: *Fly ash from extensive Swedish incinerators ends up in an old limestone quarry on the Norwegian island of Langøya in the middle of the sea near Oslo. As noted by the photographer, the island's nature has suffered; the reserve on the eastern coast is slowly dying. (Source: Opie, 2015)*

The lack of recognition of the high dioxin content in fly ash leads to even worse practices in developing countries. In Gabon, for instance, the operator of a new hazardous waste incinerator encourages the use of fly ash to "enhance" soil properties (Dzonteu, 2020). Studies from Cameroon confirmed similar practices (Mochungong et al., 2012).

In scientific literature, many scientists discuss various uses of incineration fly ash, even in contradiction to the BAT Stockholm Convention guidelines. For example, Ferreira et al. (2003) established the basic division of fly ash use into:

Photo 3.23: *Residues from WtE Termizo Liberec (Czech Republic) were also used for surface treatments in the waste dump in Košťálov. Photo: Děti Země Liberec.*

Photo 3.24: *A mixture of fly ash and slag from SPRUK also ended up in the body of the cycle path in the Jizera Mountains Protected Landscape Area, funded by EU funds. After this fact was publicized in 2007, the sign indicating the subsidy disappeared. Photo: Jindřich Petrlík, Arnika.*

- Construction materials (cement, concrete, ceramics, glass, and glass ceramics)
- Geotechnical applications (road base layers, dams, etc.)
- Agricultural use (for soil improvement)
- Other (sorbent, sludge treatment, etc.).

If fly ash is pre-treated to minimize future contaminant leakage, it is usually in relation to metal and salt content, not organic substances, let alone dioxins (Petrlik, Bell et al., 2017). Subsequently, this pre-treated material is deposited in landfills (hazardous waste) or deep repositories, such as salt mines. Apart from this, there is a wide range of pre-treatment

Photo 3.25: *This photograph explains why there are such high concentrations of dioxins in poultry eggs and dust from roads in Bishop's Cleeve, UK. Photo: Public Interest Consultants, UK.*

methods that can only be described as diluting contaminants. It has been documented that fly ash incorporated into cement monoliths can be released and is dispersed by the wind (Wang et al., 2006). Thus, dust containing dioxins contaminates the vicinity of the repository or landfill. However, fly ash was also released directly from ash processing facilities, such as in the vicinity of Bishop's Cleeve in the United Kingdom, where eggs from domestic poultry were found with a concentration of 55 BEQ.g^{-1} of fat (Katima et al., 2018) determined by the DR CALUX method (see Chapter 5.1.1). For comparison, according to Directive 2013/711/EU, the limit for PCDD/F content is 1.7 pg BEQ.g^{-1} of fat or 3.3 pg BEQ.g^{-1} of fat for PCDD/F and dl PCB.

Similar to waste incinerators, POPs are also generated in other combustion facilities. Therefore, waste generated, for example, in metallurgy represents a serious risk of environmental contamination by these substances.

Substances present in incineration bottom or fly ash will not disappear in landfills or when used elsewhere. It is only a matter of time before they are released (landfills commonly leak) and contaminate both groundwater and soil. Consequently, environmental contamination leads to the entry of these substances into the food chain. See more in Chapter 4.2.

The following case study describes the situation in the Netherlands. A similar practice (using bottom ash in road construction) is briefly mentioned in a case study from Tallinn (Chapter 10.2.5). Blasenbauer et al. (2020) discussed the situation in various European countries in a scientific article.

3.3.3.1 Case study: Netherlands

According to Arkenbout (2019), the main methods of disposing of fly ash are: cement production, landfilling, storage in deep underground cavities, or immobilization. While most fly ash (up to 40 %) is used in cement production, the remainder is either stored in deep underground cavities or sent to landfills. In the Netherlands, landfilling is heavily taxed, making it economically advantageous for WtE facilities to deposit these residues into salt mines, such as Sonderhausen in Germany. In 2017, the Dutch government banned the export of these wastes abroad, but in 2019, the export was again permitted (Arkenbout, 2019).

In 2012, the Dutch Waste-to-Energy (WtE) industry reached an agreement with the Ministry of Infrastructure and Water Management to enhance the quality of incineration ash so that it could be used for "useful" applications without the need for insulation measures (Arkenbout, 2019; DWMA, 2016). Nevertheless, reliable data regarding PCDD/F, PAHs, or PFAS levels

Photo 3.26: *The use of bottom ash from municipal waste incinerators in Katwijk aan Zee disregards potential water contamination, as seen in a photograph from November 2021. Photo: Jindřich Petrlík, Arnika.*

Photo 3.27: *Incineration ash is widely used in the Netherlands, as visible in the terrain modifications in Katwijk aan Zee. Photo: Jindřich Petrlík, Arnika.*

found in ash from waste incinerators remains absent (Shen et al., 2010; Strandberg et al., 2021; Zhao et al., 2010). This indicates that outdated information about the toxicity of these materials is being relied upon by both industry and regulators. Before being used in "useful" applications (construction and road materials), the bottom ash is treated in facilities like Heros Sluiskil in the south of the Netherlands. Among the methods used in these facilities are metal extraction using magnetic devices or removal of large pieces of slag. After these basic mechanical treatments, the bottom ash is labelled as "suitable" for "useful" applications.

During road construction, bottom ash is often piled up, and these heaps should be covered to prevent any release of contained pollutants into the environment. However, it has been documented that merely wrapping the ash in plastic is not sufficient to prevent environmental contamination during rainfall (Arkenbout, 2019), as observed in photo documentation from Katwijk aan Zee (Photo 3.26). The Sluiskil plant expressed concerns about significant fluctuations in bottom ash quality and a general trend of declining bottom ash quality. The causes of this phenomenon are largely unknown. The Dutch Inspectorate for the Environment and Transport of the Ministry of Infrastructure and Water Management issued a report in September 2019, highlighting the risks of importing, producing, and using ash for the environment and human health. Millions of tons of ash are used in public constructions, roads (see Photos 3.26, 3.27, and 3.30), and waterworks. However, data on

Photo 3.28: *The presence of various unburned residues of metal, glass, or even plastic waste indicates that it is ash from waste incineration. Photo: Jindřich Petrlík, Arnika.*

Photo 3.30: *Photographs from Katwijk aan Zee taken at the end of 2021 confirm Ernst Worrell's statement (see Photo 3.29). Photo: Jindřich Petrlík, Arnika.*

Photo 3.29: *Professor Ernst Worrell from Utrecht University labeled Dutch roads as "linear landfills." Photo: https://hetgroenebrein.nl/wetenschapper/ernst-worell/.*

the quantity and location of this material are lacking, making it unclear and impossible to verify whether these sites meet all regulatory requirements (Arkenbout, 2019).

The Netherlands, with oversized incineration capacities and needing to import waste from abroad to fill them, subsequently faces a lack of places to deposit the bottom ash. Hence, it is extensively used in road embankments. Nobel laureate Ernst Worrell (Photo 3.29) termed Dutch roads "linear landfills" (Photo 3.26 and 3.30); (Göblová, 2021). In the Czech Republic, the new Waste Management Regulation 541/2020 Sb. also allows the use of bottom ash in road embankments and other surface engineering structures (Ministry of the Environment of the Czech Republic, 2021e).

3.4 Soil

In some studies, the term “emission to land” is used to refer to emissions into the soil, including transfers of toxic substances in waste. This can often be encountered, for instance, in older dioxin emission inventories from the UK, where dioxins ending up in waste were counted as emissions to soil, resulting in figures much higher than emissions into the air or water. However, in our study, chapters specifically dedicated to waste address the transfer of toxic substances from incineration residues. This does not mean, though, that there are no emissions of toxic substances into the soil. Firstly, this could be understood as the indirect transfer of toxic substances from incineration through rain and their deposition on the earth’s surface. Secondly, it could involve the application of waste incineration residues containing toxic substances, onto the earth’s surface followed by contamination of soil with these substances (by leaking and/or dust spread around). Sometimes, deep injections are also included in these, but these are rather emissions into deeper geological horizons, i.e., the geosphere, rather than the pedosphere.

In the Czech Republic, this might include cases where fly ash from incinerators have been used or are still being used for the remediation of old mining works (for example, near Žacléř in the Krkonoše Mountains; see Chapter 5.1.1.3.4). A study focused on the presence of toxic substances in the vicinity of the Hůrka area near Temelín, dealt with such utilization, where most of the prepared mixtures were used for the remediation of the lagoons remaining after uranium ore processing in Mydlovary.

Calculating the deposition of toxic substances originating from waste incineration as a primary source is very challenging, especially for the situation in the Czech Republic, where reported emissions are mixed with many other sources, making it difficult to isolate those from waste incineration residues. Nevertheless, some of these emissions can be

Photo 3.31: *In Hůrka near Temelín, fly ash is being processed among other things. Photo: mail.oakrupkovo.cz.*

partially traced, for instance, using a specific profile of dioxin congeners in collected samples (Chang et al., 2004; Petrlik et al., 2022). Similarly, sources of mercury emissions can be traced using specific isotopes (Du et al., 2018; Elizalde, 2017; Sherman et al., 2015). However, it is simpler and more effective to identify contaminant profiles in the soil pollution surrounding incinerators where there are few other sources of pollution, as was the case with a waste incineration plant in Iceland near Úlfsá (closed since 2011). High concentrations of dioxins were measured in cow’s milk in the vicinity, and even years later, increased concentrations of hexachlorobenzene and arsenic are still found in mussel meat (AMAP Assessment, 2016). These are consequences of the transfer from emissions and probably from solid residues produced by the incinerator in otherwise pristine nature in Iceland.

The following Table 3.9 demonstrates the transfer of dioxin contamination from their sources, which are mostly poorly stored waste from incinerators, through the soil into eggs of domestically raised poultry or birds (i.e. sample from Phuket); (Petrlik, 2011; Katima et al., 2018).

In general, it can be said that fly ash with dioxins at a concentration of 2,500 ng $TEQ.kg^{-1}$ can contaminate soil up to levels of tens or hundreds of ng $TEQ.kg^{-1}$ of dioxins, further leading to the accumulation of dioxins in chicken eggs at concentrations exceeding the European limit by more than twenty times. The severity of this problem was highlighted in a global study in 2017 (Petrlik, Bell et al., 2017).

Soil represents a significant medium in the transfer of pollution from incineration sites into food chains, hence it should be considered when assessing the health impacts of waste incineration. A recently published study, which monitored POPs in the vicinity of three European incinerators, was based on this premise (Arkenbout & Bouman, 2021). Eggs (from chickens) serve as sensitive bioindicators of persistent and bio-accumulative substances that bind in the fats, of chickens, when free-ranging, and are in direct contact with soil that enters their bodies through food intake. A chicken can ingest 11 to 30 grams of soil in a single day (Hoogenboom et al., 2006; Waegeneers et al., 2009).

Soil contamination with substances such as dioxins, due to inadequately controlled waste incineration processes, can lead to long-term pollution. This was demonstrated, for instance, in Lausanne, Switzerland (see Chapter 3.5.1).

Table 3.9: *Summary information on concentrations of chlorinated dioxins in TEQ and/or BEQ found at sites affected by fly ash and other dioxin-contaminated waste. (Source: Katima et al., 2018; Petrlik et al., 2019).*

	Year(s) of Sample Collection	Fly Ash (Waste)	Soil/ Sediment – Direct Impact	Soil/Sediment – Reference Value	Egg	Egg – Reference Value[1)]
Units		pg $TEQ.g^{-1}$ dry matter			pg $TEQ.g^{-1}$ fat	
Thailand (Phuket incinerator)	2010–2011	3,200–8,000	2,700**	NA	6.1*	0.08[a]
China (Wuhan incinerator)	2014–2015	779	NA	NA	12.2	0.2[b]
United Kingdom (Bishop's Cleeve)	2010–2011	2,500	6.5–11*	0.05–1.2	1.8; 21; 55*	0.2[c]
United Kingdom (Newcastle)	2000	20–9,500	7–292	NA	0.4–56	0.2[c]
Peru (Zapallal)	2010	50–12,000	5–11	0.05–1.2	3.4–4.4	0.12[d]
Taiwan	2005	NA	NA	NA	32.6	0.274[e]
Poland (chicken coop made of treated wood)	2015	3,922	16–47	0.1–0.8	12.5–29.3	0.44[f]
Ghana (Accra, medical waste incinerator)	2018	551	NA	2 ***	49	0.39 g

a) (Petrlik et al., 2017); b) (Petrlik 2015); c) (Pless-Mulloli et al., 2001a); d) (Swedish EPA 2011); e) (Hsu et al., 2010); f) (Piskorska-Pliszczynska et al., 2016); g) (Petrlik et al., 2019)
**BEQ (Total Dioxin-like Toxicity)*
*** data for sediment; NA – data not available*
**** dl PCB + PCDD/Fs (site in Accra); (Tue et al., 2016)*

Photo 3.32: *In the vicinity of a now closed hospital waste incinerator in Accra, there remains a pile of ash that became a source of dioxin contamination in domestically raised poultry (Petrlik et al., 2019). Photo: Martin Holzknecht, Arnika.*

3.5 Case Studies

Here, we present case studies where contamination of food chains (mostly eggs and poultry meat from domestic farms) occurred due to dioxin air pollution in the vicinity of incineration plants. A similar case study from Harlingen, the Netherlands, is presented in Chapter 3.5.3.

3.5.1 Lausanne (Switzerland)

In December 2020, significant soil concentrations of chlorinated dioxins were discovered in large parts of Lausanne, Switzerland, reaching up to 640 pg WHO-TEQ.g^{-1} dry weight (Vernez et al., 2023). For comparison, in the vicinity of the Czech site most contaminated with dioxins, old ecological burdens in Spolana Neratovice, 518.8 pg WHO-TEQ.g^{-1} dry weight were measured in sediment (Petrlik et al., 2006).

The most likely source of contamination in Lausanne was a former municipal waste incinerator. A three-stage multidisciplinary health risk assessment was conducted to determine potential exposure of the population to chlorinated dioxins and identify suitable preventive measures. Exposure scenarios based on the use of contaminated soil were created, followed by an evaluation of the toxicological risks of different scenarios (according to the consumption of food grown on contaminated soil). Subsequently, detailed geostatistical mapping of soil contamination by chlorinated dioxins was performed (see map in Figure 3.5). Three main scenarios were evaluated:

- Direct ingestion of soil by children in playgrounds
- Consumption of vegetables from private gardens by children and adults
- Consumption of food from livestock raised on contaminated soil.

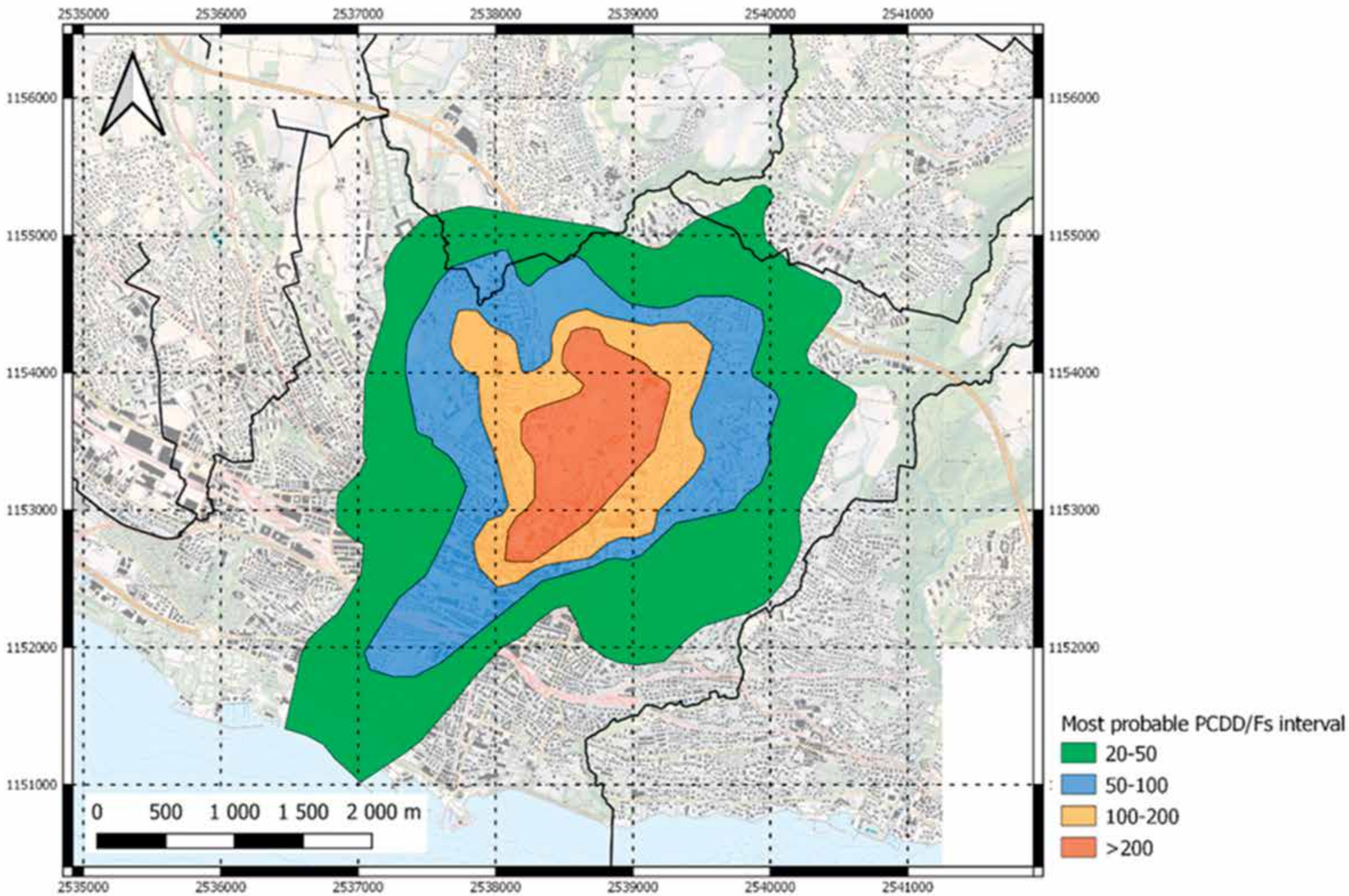


Figure 3.5: *Map of soil contamination by chlorinated dioxins in the vicinity of the former municipal waste incinerator in Lausanne. (Source: Vernez et al., 2023)*

The worst exposure scenario involved consuming eggs from private poultry, resulting in significantly higher concentrations of chlorinated dioxins in blood serum than would normally be expected. No relevant increases in serum concentrations were calculated for direct soil ingestion and consumption of vegetables, except for gourd vegetables. The combination of mapping and exposure scenario assessment led to targeted protective measures for soil users, especially regarding food consumption. The results also raised concerns about consumption of potentially hazardous products from animals raised on land with chlorinated dioxin concentrations only slightly above environmental background levels (Vernez et al., 2023).

3.5.2 Maincy (France)

Maincy is a small French village approximately 60 km south of Paris. It is situated near an old waste incinerator that had operated for more than 20 years from 1974. In 2002, it was shut down due to very high dioxin emission levels, which were more than 2,000 times higher than the then current European standard of 0.1 ng TEQ.m^{-3} (Pirard et al., 2005).

A study led by Belgian scientist Catherine Pirard found dioxin concentrations in soil ranging from 3.26 to 59.04 pg I-TEQ.g^{-1} dry weight, compared to dioxin concentrations in eggs ranging from 5.1 to 121.55 pg WHO-TEQ.g^{-1} fat (Pirard et al., 2005). The sum of dl PCB concentrations ranged from 0.78 to 2.80 pg I-TEQ.g^{-1} dry weight in soil and from 0.85 to 52.48 pg WHO-TEQ.g^{-1} fat in eggs. The initial study also measured concentrations in chicken abdominal fat tissues, ranging from 34.3 to 121.1 pg WHO TEQ.g^{-1} fat. These concentrations also exceeded the then-applicable EU standard of 2 pg WHO TEQ.g^{-1} fat. In the second study, the measured concentrations were higher than the range of 0.1 to 6 pg TEQ.g^{-1} dry weight typically reported for surface soil samples taken near modern and operating European incinerators, except for one study reporting soil concentrations near another very old incinerator. The concentrations of dioxins found in eggs and poultry tissue samples from domestic farms in Maincy were more than 15 times higher than the European limit at that time of 3 pg WHO-TEQ.g^{-1} fat (DiGangi & Petrlik, 2005).

3.5.3 Harlingen (The Netherlands)

Semicontinuous measurements at the incineration plant in Harlingen, which became operational in 2011 as a state-of-the-art[24] facility, revealed that actual dioxin emissions were higher than permitted by the incinerator and what was reported based on short-term measurements (Arkenbout & Petrlik, 2019). Long-term sampling using the AMESA system (envea, 2021; Reinmann, 2002; Wu et al., 2014) revealed fluctuations during the incinerator's start-ups after maintenance shutdowns. The incineration plant in Harlingen had an emission limit for dioxins ten times lower (0.01 ng TEQ.m^{-3}) than the usual standard (0.1 ng TEQ.m^{-3}). The difference is evident in the graph from the 2019 study (see Figure 3.6). Another graph in Figure 3.6 shows measured PFAS emissions, specifically perfluorooctanesulfonic acid (PFOS) and perfluorooctanoic acid (PFOA), from the same incineration plant.

Photo 3.33: *Instances when not just water vapor but colored or black smoke emitted from the Harlingen incinerator chimneys prompted local toxicology experts to actively monitor emissions. They revealed serious flaws in the measurement of dioxins and other substances (Arkenbout et al., 2018). Source: http://www.toxicowatch.org.*

24 The term state-of-the-art means using the latest technology.

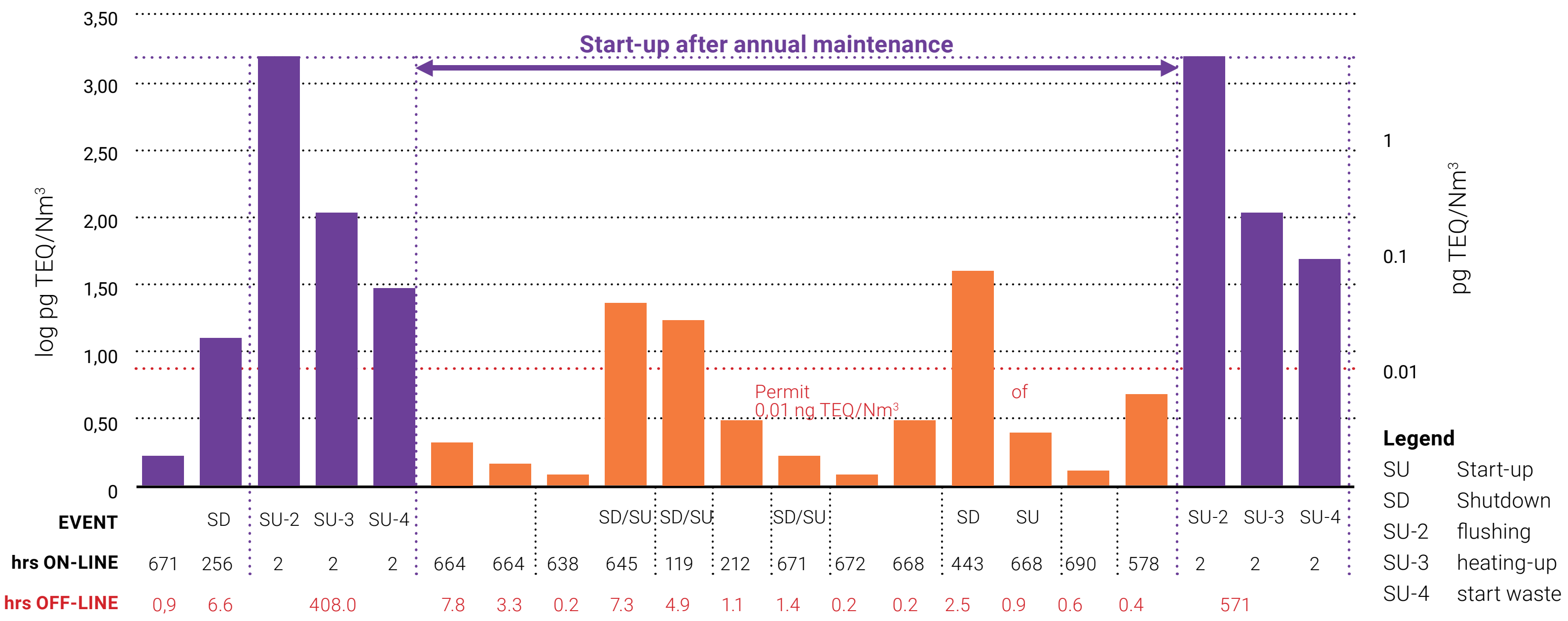


***Figure 3.6:** Graph showing the difference in dioxin emissions from the Harlingen incinerator during start-up situations after maintenance shutdowns. These values were only captured using semicontinuous sampling by the AMESA system. (Source: Arkenbout & Petrlik, 2019)*

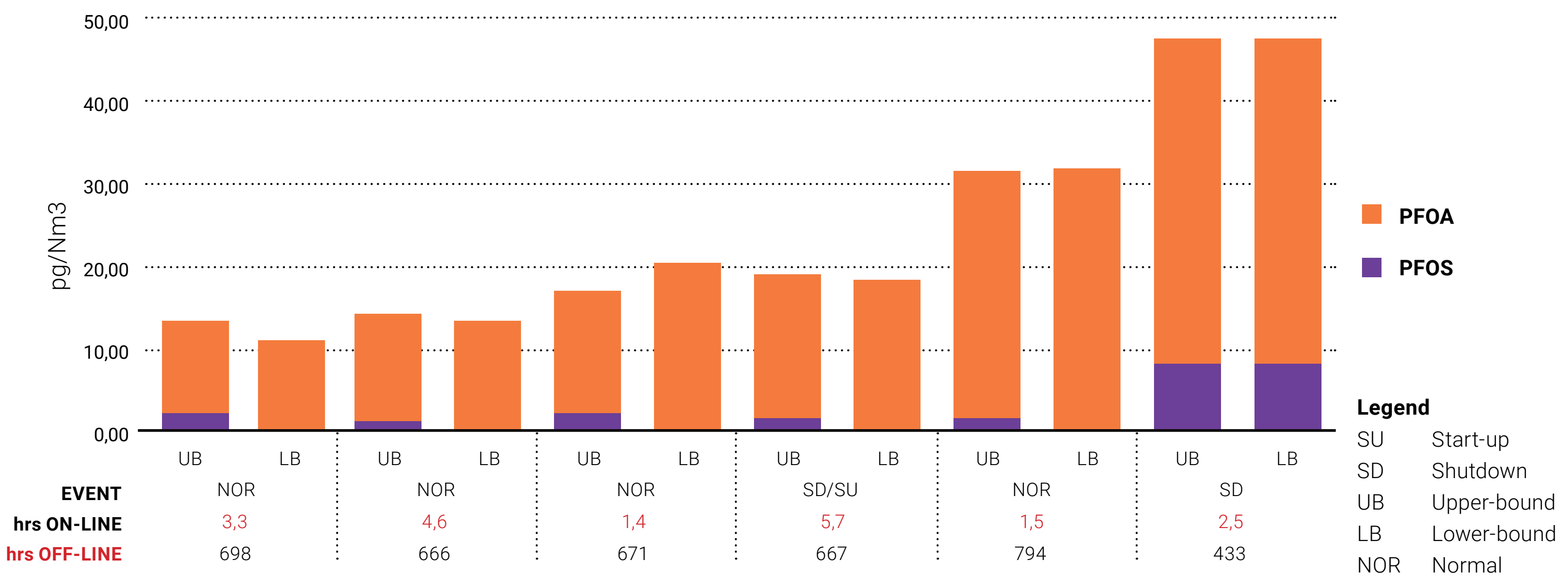


***Figure 3.7:** Graph showing measured concentrations of PFOS and PFOA (two substances from a wider range of PFAS not analyzed in this case) in emissions from the Harlingen incineration plant. (Source: Arkenbout & Petrlik, 2019)*

As part of the long-term monitoring around the Harlingen incinerator, eggs and grass in the vicinity were also examined. Dioxin congeners in both the eggs and grass matched those measured in the semicontinuous monitoring of waste incineration emissions, and a decrease in concentrations of these substances was observed with distance from the incinerator (Arkenbout & Esbensen, 2017).

Similar monitoring (eggs, moss, and needles) was conducted around three waste incineration plants in Europe—in Lithuania, Spain, and the Czech Republic (WtE Chotíkov)—but without available information on emissions from semicontinuous monitoring of the facilities themselves (Arkenbout & Bouman, 2021). The monitored incineration plants do not have semi-continuous emission measurement systems. The dioxin congeners found in eggs were compared to those measured in incineration emissions in the Netherlands (Arkenbout & Esbensen, 2017) and in China (Chen et al., 2017). The resulting PCDF:PCDD ratio, which was 1.7:1, generally indicates that their source could indeed be municipal waste incineration.

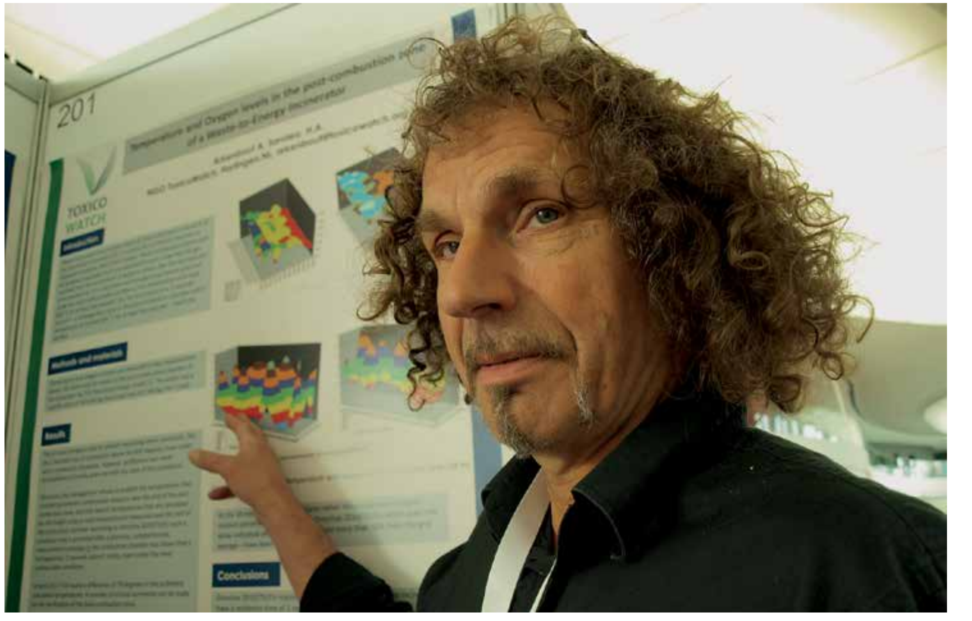


Photo 3.34: *Abel Arkenbout (Toxicowatch) presenting the results of monitoring toxic substance emissions from the Harlingen incinerator at the Dioxin 2018 conference in Krakow (Arkenbout et al., 2018). Photo: Jindřich Petrlík, Arnika.*

3.5.4 Small Medical Waste Incinerators

Open burning and incineration of medical waste (see chapter 2.3 for its definition) without adequate pollution control, exposes waste workers and the community to toxic contaminants in air emissions, bottom ash (Emmanuel, 2012), fly ash (Grochowalski, 1998; Nguyen et al., 2021) as described in Chapters 3 and 5 (Liu et al., 2023; Strandberg et al., 2021) and in other pollution residues (Petrlik and Ryder, 2005; UNEP and Stockholm Convention, 2013), with high levels of PCDD/Fs observed in residues from simple batch-type Medical Waste Incinerators (MedWIs); (UNEP and Stockholm Convention, 2013).

The contamination of food chains has also been observed in the vicinity of small MedWIs (Agarwal et al., 2005; Calonzo et al., 2005; Marcanikova et al., 2005) and is not limited to developing countries only (Skalsky et al., 2006). In developing countries it is the case that other wastes, not only medical wastes, are incinerated in MedWIs (Skalsky et al., 2006). Five sites (primarily small batch type of MedWI up to 3,000 tons of waste per year) were studied (Jelinek et al., 2023b) while ash, soot and free-range chicken eggs were sampled (DiGangi and Petrlik, 2005; Petrlik et al., 2021; Skalsky et al., 2006) and compared to previously studied MedWIs in Pakistan and Mozambique (Khwaja and Petrlik, 2006; Mochungong, 2011). In Accra (Ghana), only bottom ash was accessible for sampling after the MedWI ceased its operation. Visual documentation of some of the sampling sites are provided in Photos from 3.35 to 3.37.

Photos 3.35, 36 and 37: *Examples of small medical waste incinerators in Yaoundé, Cameroon (Photo: CREPD) Islamabad, Pakistan (Photo on the right side: Jindřich Petrlík, Arnika) and Kumasi, Ghana (Photo: Martin Holzknecht, Arnika).*

Table 3.10: *Comparison of PCDD/Fs levels in bottom ash samples from MedWIs. Classes according to Dioxin Toolkit classification (UNEP and Stockholm Convention, 2013).*

Country	Year	PCDD/Fs	Class MedWI	Country	Year	PCDD/Fs	Class MedWI
3 African countries	2011-2019	347-2151	Class 1 and 2	Germany (Gidarakos et al., 2009)	2009	1160-19710	Class 4
Pakistan	2005	38-2105	Class 1	Jordan (Arar et al., 2019)	2019	206-476	Class 2
Poland (Grochowalski, 1998)	1998	7800-43000	Class 2 and 3	Thailand (Fiedler, 2001)	2001	1390	Class 2
Vietnam (Pham et al., 2019)	2019	22.9-139	Class 3	Algeria (Yacine et al., 2018)	2018	1.68 - 878	Class 3

Measured levels of PCDD/Fs and dl PCBs in bottom ash samples and one soot sample from small MedWIs included in this study, are summarized in Table 3.10 and compared to available literature. Measured levels of PCDD/Fs are comparable to those from the waste incinerators of the same class in Thailand or Algeria, however levels measured in Polish MedWIs in the 1990's (Grochowalski, 1998) were of a magnitude higher. Also, levels of PCDD/Fs observed in bottom ashes from German MedWIs were much higher. According to the UNEP Dioxin Toolkit and its corresponding classes, it suggests that the major pathway of PCDD/Fs releases from small MedWIs, is to air (UNEP and Stockholm Convention, 2013).

Levels of PCDD/Fs, dl PCBs, and HCB in free-range chicken eggs from the vicinity of small MedWIs in African countries are summarized in Figure 3.8. They are compared with reference sample of eggs from a supermarket in Accra and with levels measured in free-range chicken

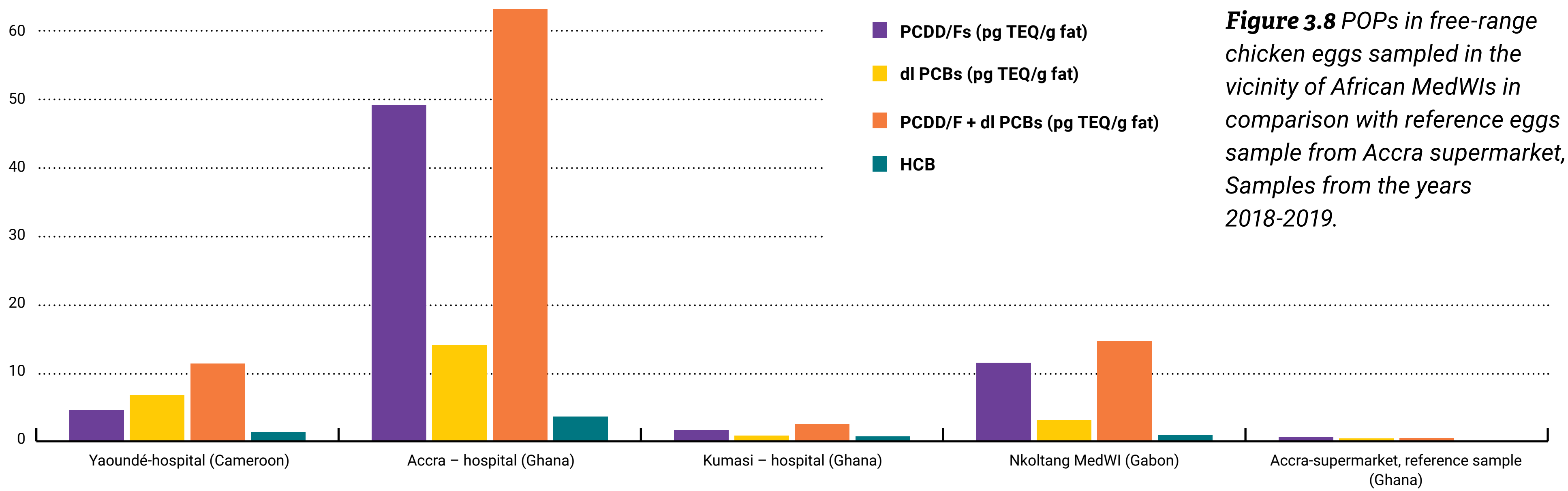
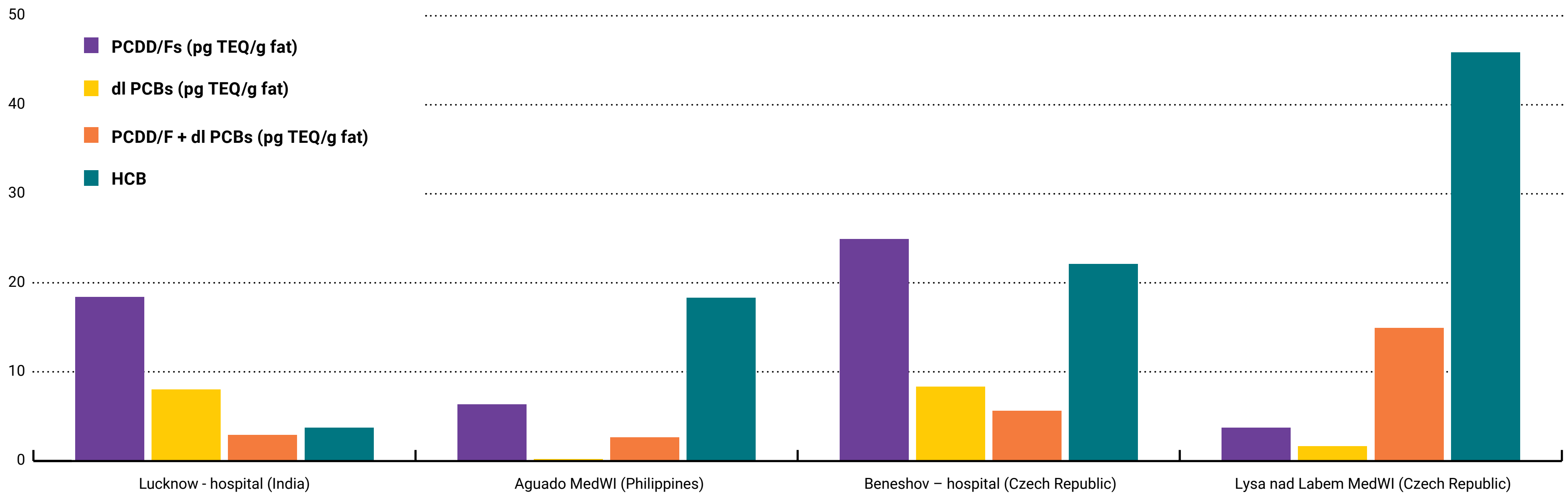


Figure 3.8 *POPs in free-range chicken eggs sampled in the vicinity of African MedWIs in comparison with reference eggs sample from Accra supermarket, Samples from the years 2018-2019.*

Figure 3.9 *POPs in free-range eggs from the vicinity of small MedWIs – samples from the years 2004–2005.*



eggs from the vicinity of small MedWIs collected in 2004 – 2005 which are summarized in Figure 3.9.

In comparison with the European limit for PCDD/Fs and dl PCBs in eggs as food (5 pg WHO-TEQ.g^{-1} fat); (European Commission, 2022a), with exemption of the eggs from Kumasi, samples collected in 2018-2019 exceeded this limit by more than 2–105-fold along with samples from 2004-2005. Sample from Accra (49 pg WHO-TEQ.g^{-1} fat) was 25th highest globally (Petrlik et al., 2022).

These results are related to how incineration residues are managed *(Mochungong et al., 2012)* and calls for setting strict limits for PCDD/Fs in wastes used on land surface, for untreated waste at levels of several tenths of pg TEQ.g^{-1} dw as the maximum (Petrlik et al., 2022; Swedish EPA, 2011). The substances in eggs are concentrated because hens raised as local food have free access to residues from waste incineration. Risks related to MedWIs on human health, including POPs releases, are underestimated (Mochungong, 2014).

4. Incinerators and the Planetary Ecosystem

In 2009, a study (Rockström et al., 2009) was published, identifying nine and quantifying seven planetary boundaries (Earth's boundaries) that must not be exceeded to prevent harmful or even catastrophic consequences for the world (such as ecosystem devastation, reduction of ecosystem services, and ecological disasters). Human activity during the Anthropocene period may result in a sudden (and sometimes irreversible) change in the planet's environment, which could lead to conditions less favorable for the continued existence of humanity. These boundaries include:

- Climate change
- Biosphere integrity
- Biogeochemical flows of phosphorus and nitrogen
- Novel entities[25] (chemical contamination)
- Land-system change
- Ocean acidification
- Freshwater change
- Stratospheric ozone depletion
- Atmospheric aerosol loading

25 The original study refers to "man-made chemicals" as novel entities, but also to plastics and heavy metals. What all these substances have in common is that they have been introduced into the environment by human activity. Dioxins and other substances produced as by-products of the incineration of waste fit perfectly into this definition.

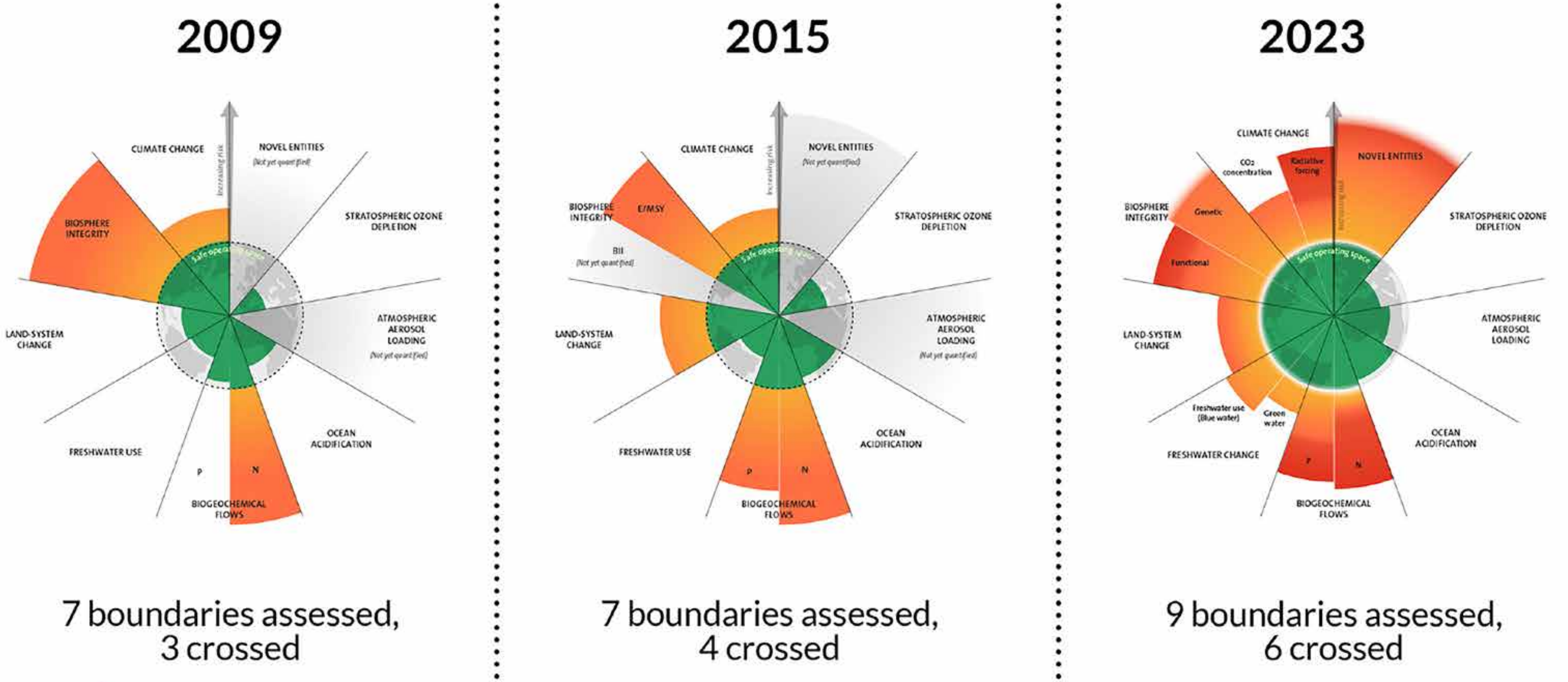
Indicators have been established for most of these nine planetary boundaries, and permissible limits have been proposed. However, the first five boundaries (Stockholm Resilience Centre, 2023) listed above have already exceeded the "safe" planetary limits (see Figure 4.1). For this reason, we will delve into several subchapters to examine the influence of incinerators on selected planetary boundaries. Planetary boundaries are closely linked to sustainable development or movement within environmentally safe planetary limits.

In the following chapters, we analyze those planetary boundaries significantly impacted by incinerators. However, a deeper analysis might reveal further connections, such as the impact on biogeochemical flows of phosphorus and nitrogen (Kopittke et al., 2021); (see Chapter 10.2.4 and the commentary by Professor Lars Stoumann Jensen).

4.1 Climate Change

Waste generation and its management represent a significant global challenge and threat to climate change. Municipal solid waste generation is predicted to grow from 2.3 billion tonnes in 2023 to 3.8 billion tonnes by 2050 (UNEP & ISWA, 2024) How countries manage and dispose of their waste,

Figure 4.1 *Planetary boundaries in 2009, 2015 and 2023; Licenced under CC BY-NC-ND 3.0 (Credit: Azote for Stockholm Resilience Centre, Stockholm University). (Source: Stockholm Resilience Centre, 2023)*



defines the quantities of climate (and other) pollution that is subsequently generated. The amount of waste incinerated in the European Union continues to rise (EUROSTAT, 2023). Carbon dioxide emissions resulting from waste incineration depend upon the carbon content in the waste, which can be of fossil or biogenic origin. Generally, burning one ton of waste releases approximately 0.7 to 1.7 tons of CO_2 into the atmosphere (Tangri, 2023; Vahk, 2019). Furthermore, recent studies have demonstrated that incinerators emit more greenhouse gases per unit of electricity produced (1707 g CO_2 eq. kWh^{-1}) than any other power source (range: 2.4 to 991.1g CO_2 eq. kWh^{-1}); (Tangri, 2023). Often proponents of waste to energy incinerators submit environmental impact assessment documentation that-commonly does not account for biogenic carbon (considering the potential for global warming, i.e., GWP=0). This allows the industry to claim that incinerating waste is better than landfilling waste due to the more potent climate impacts of methane (Marmier & Schosger, 2020) which is generated through the decomposition of organic wastes in landfill. However, this ignores a number of compelling facts which challenge this claim, such as the utilization of landfill gas extraction technologies increasingly being used and mandated for landfill operations and the improvements achieved through what is known as 'Zero Waste' practices that ensure organic wastes are separately collected and redirected to better outcomes such as composting and recycling. Studies have shown that landfill with full pre-treatment (i.e. organic waste removal, further waste segregation of recyclables and bio-stabilisation) outcompetes waste incineration in terms of climate pollution, toxic air pollution and associated health costs (Hogg, 2006).

Like many other developed countries, the average carbon intensity of the EU power grid is decreasing over time due to displacing fossil fuels with renewable energy sources (with very low or zero GWP). For instance, in

Photo 4.1: *Climate change caused by excessive CO_2 emissions and other greenhouse gases results in extreme weather events such as droughts and fires that erupted extensively in 2023, including on Greek islands. Photo: Creative Commons (from http://www.planetacestovani.cz).*

Photo 4.2: *Extreme weather events associated with climate change also include more frequent torrential rains and floods (Ústí nad Labem, 2006). Photo: Hana Kuncová, Arnika.*

2018, it was only 296 g CO_2 eq/kWh. Consequently, as the EU power grid decarbonises, waste incineration will have an increasingly negative impact on climate change in the future than it currently does (Vahk, 2019). In addition, incinerators are the most emissions-intensive form of generating electricity in the U.S. today (Tangri, 2023).

In documentation for incinerator construction plans within EIA processes, the amount of CO_2 emissions from burning carbon in waste (fossil origin) for energy production is commonly compared to emissions that would result from landfilling the same amount (but not necessarily the same type) of waste (CO_2 and CH_4) and by generating heat and energy by burning fossil fuels (especially coal). Methane emissions from landfills that take organic waste such as food and plant biomass, are not directly comparable to waste to energy incinerators that burn waste that has been source separated to remove such organic waste and recyclable materials. As mentioned above, pre-treated waste (i.e. full source separation and stabilization) entering landfills has been shown to emit less GHG's than incinerators with or without heat capture. Waste incinerators compare very badly to recycling, composting and waste prevention – see Figure 4.3 and Table 4.1. Comparing energy recovery from waste to

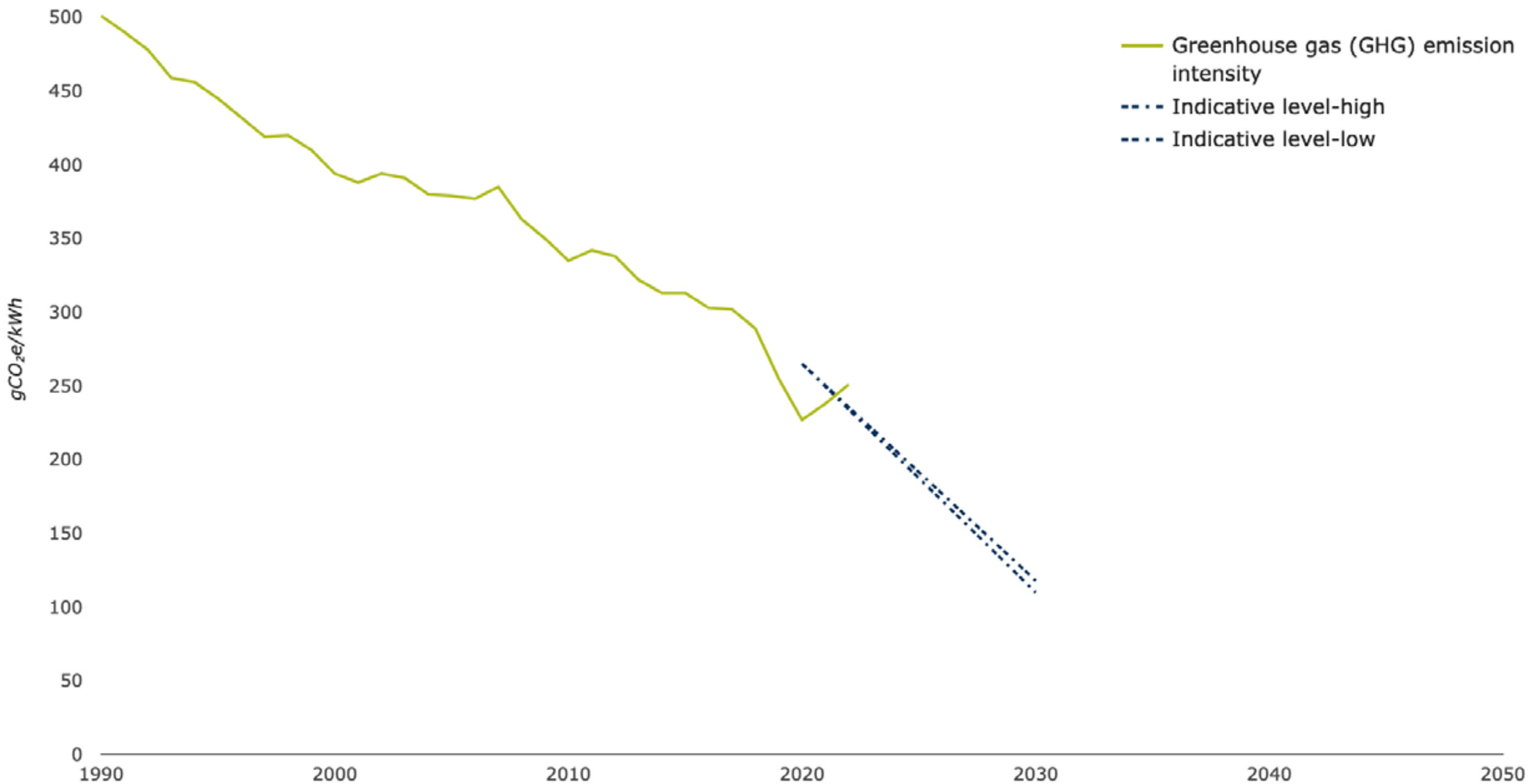


Figure 4.2: *Greenhouse gas emission intensity of electricity generation in EU. (Source: EEA, 2024)*

landfilling is a deliberate attempt to confuse the benefits of improved waste management and renewable energy, against a cynical business as usual approach to both, where resources continue to be wasted (i.e. landfilled) and energy remains entrenched in fossil fuels. It is clear that international authorities (UNEP, 2019) and countries in both the global north and global south are less concerned with such industrial inevitability arguments and are taking action to both reduce waste and implement more sustainable waste management practices (including removal of organic waste from landfill) and support renewable energy projects, exactly to address climate change.

Given the expectation that newly constructed waste-to-energy facilities will serve for approximately another 20-30 years, the construction of these facilities delays the transition to less carbon-intensive methods of energy production from renewable sources. Construction of waste incineration facilities also delays better resource recovery and waste management practices that reduce methane emissions and preserve waste resources and their quality for a Circular Economy. Supporting electricity production from waste would hinder the ambitious goal of reducing emissions in the energy sector, aligned with the Paris Agreement, genuinely striving to limit the rise of the average global temperature to below 1.5°C, unlike the construction of incinerators. Waste incineration facilities in the EU, including Waste-to-Energy Plants (WtE), will become part of the emissions trading system (EU ETS) starting from 2026 (Zanni, 2022). This could lead to increased waste disposal fees and higher prices for electricity and heat.

Photo 4.3: *WtE Malešice (Prague) shows increasing CO_2 emissions according to the data in IRZ (Arnika, 2022b). Photo: VitVit, Wikimedia Commons CC BY-SA 4.0*

According to a study by the international network of non-governmental organizations Friends of the Earth (2009), the European Union loses resources worth €5.25 billion annually simply by incinerating or landfilling them instead of recycling. If this material were recycled, then EU countries would save 148 million tons of CO_2 annually, equivalent to the emissions from around 47 million cars in a year. In the European Union (at the time of the study, including the United Kingdom), approximately 405 million tons of key recyclable materials are available in municipal waste, of which about 52 % is disposed of (landfilled or incinerated).

Figure 4.3: *Comparison of the impact on climate change (CO_2 emissions comparison) of various waste management methods excluding biogenic carbon (Source: Hogg & Ballinger, 2015).*

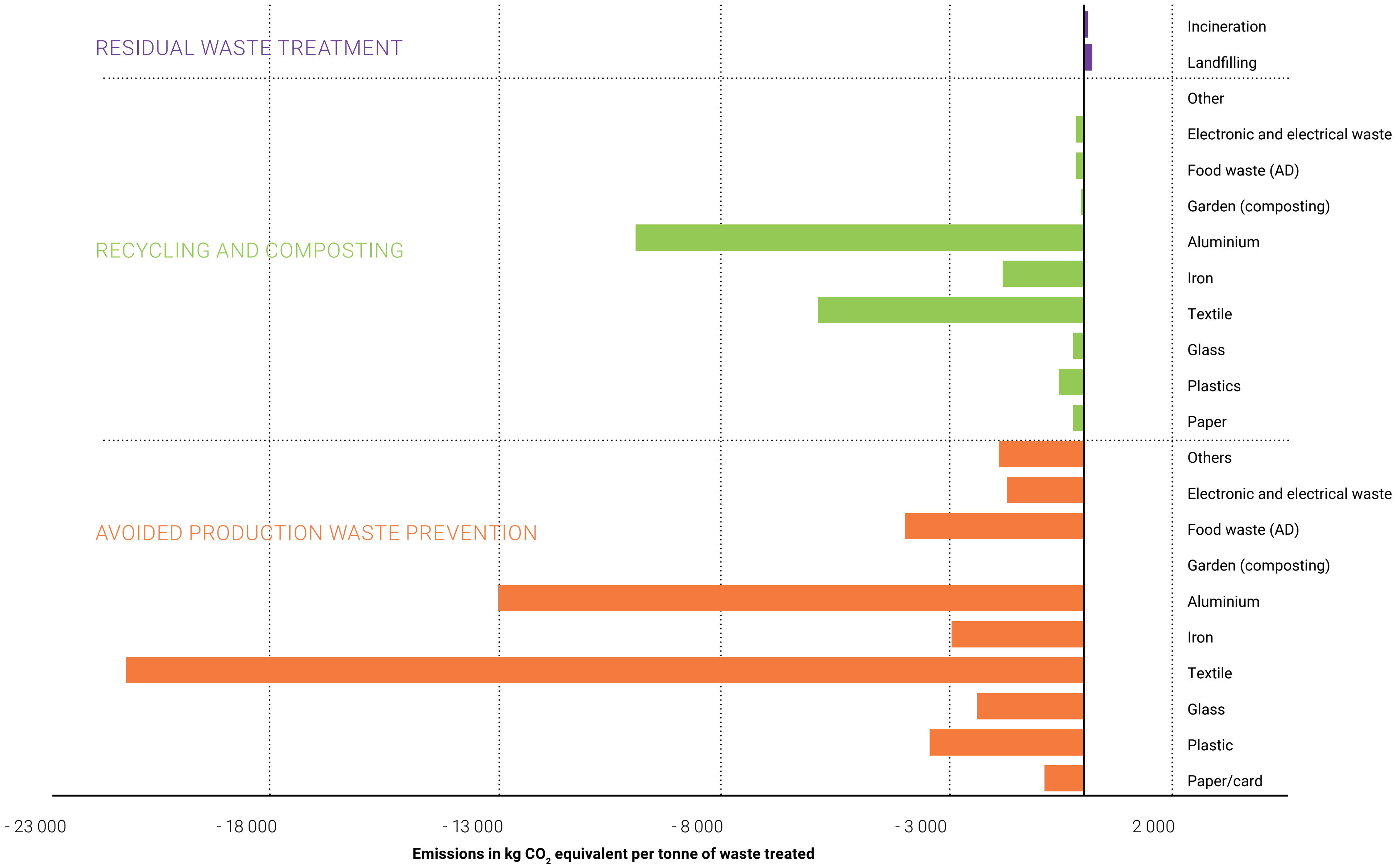


Table 4.1: *Comparison of greenhouse gas emission reduction using recycled inputs versus burning selected materials. (Source: Jofra, 2013)*

Material	Reduction in GHG emissions using recycled inputs instead of virgin raw materials (CO_2 eq.)	GHG reduction per ton of incinerated waste (CO_2 eq.)*
Glass	0.28	-0.02
Office Paper	2.85	0.48
Newspaper	2.78	0.56
Steel Cans	1.80	-0.02
PET	1.11	0.75
Copper Wire	4.89	-0.02
Aluminum Cans	8.89	-0.02

*The '-' sign signifies, conversely, the production of GHG (Greenhouse Gas).

Plastic waste is increasing globally and so too is interest in burning plastic waste for energy. By 2050, the transformation of plastic waste into energy (including incineration in WtE plants) is predicted to result in greater CO_2 emissions than burning fossil fuels, as suggested by a study by scientists from South Korea (Kwon et al., 2023). Their findings indicate that the conversion of plastic waste into energy should be perceived as a far greater issue than it currently is. Replacing coal, oil, or natural gas in waste incineration plants does not solve the problem of the share of CO_2 emissions from these plants. The numerous types of plastic waste to fuel and/or energy plants also known as Chemical Recycling or Advanced Recycling plants, that are being promoted as solutions to the global plastic waste crisis, are heavily contested (Bell et al., 2023b). To avoid further contributions to atmospheric warming, we must steer clear of the dead-end path led by WtE and similar technologies.

4.2 Chemical Pollution (Novel Entities)

A study mapping the global extent of chemical pollution on the planet was published in 2022 (Persson et al., 2022) with unequivocally negative outcomes. According to it, the safe operational space for so-called novel entities[26] of chemical pollution has been exceeded, with 'annual production and releases growing at a rate that exceeds the global capacity for monitoring and assessment.' A specific aspect highlighted is pollution by plastics. The study draws several conclusions, indicating an increase in the production of novel entities, their diversity, and global release over time, while safety assessment, regulation of substances, and countries' ability to enforce them cannot keep up with the speed of their introduction. This means that while handling chemical substances and waste may improve in one place, in others, regulations will not be followed or may not even exist, as novel entities will continue to be manufactured, used, and disposed of. This will perpetuate the release of novel entities into the environment, shifting from one place on Earth to another, thereby constituting a global problem. The duration of novel entities in the environment is also problematic. Even if the production of these substances were immediately halted, their release into the environment wouldn't cease, and the substances themselves (in many cases for a long time) would not disappear.

Incinerators are unequivocally a significant vector for introducing novel entities into the environment, as metals, organic, and inorganic substances are emitted into the air, released into wastewater, but primarily into solid residues. Incineration leads to the release of novel entities from matrices containing them, rather than their decomposition or destruction. Brominated

26 The study refers to „man-made chemicals" as novel entities, but also to plastics or heavy metals. What all these substances have in common is that they have been introduced into the environment by human activity. Dioxins and other substances produced as by-products of waste incineration fit this definition perfectly.

Photo 4.4: *Material from waste incineration (SPRUK) contains unburned plastic waste. Photo: Jindřich Petrlík, Arnika.*

Photo 4.5: *Photograph of the location where SPRUK ended up scattered in the forest. Photo: Jindřich Petrlík, Arnika.*

flame retardants or plastics, contributing to the formation of microplastics, are typical examples of substances already present in waste.

Despite initially seeming improbable, at high incineration temperatures, bottom ash and fly ash contain relatively high amounts of micro- and nanoplastics, which did not burn during the combustion process, partly due to the presence of flame retardants. Sampling and analysis of 31 ash samples from 16 modern waste incinerators operating under stable conditions revealed that complete destruction of plastics does not occur in these plants. Their concentration in the bottom ash ranged from 1.9 to 565 particles per kilogram, up to 102,000 particles per ton of incinerated waste (Yang et al., 2021). The largest share of plastic particles consisted of packaging and construction materials (PP and PS) containing BFRs. Currently, standardized methods to determine the content of microplastics in solid matrices or ash are not available, and therefore, there are no set limits for them in residues after waste incineration. Through bottom ash and fly ash, as well as through air emissions, these plastic particles continue to spread into the environment (Bhat et al., 2023; Yang et al., 2021). This was well-documented in a 2021 study and its graphic abstract (see Figure 4.4); (Shen et al., 2021). In the Czech Republic, for example, a mixture of bottom ash and fly ash (known as SPRUK) from the Termizo Liberec WtE incinerator was found, among other places, in a forest near Frýdlant (Arnika, 2019b), which contained unburned pieces of plastic waste (see Photos 4.4 and 4.5).

Furthermore, waste incineration also leads to the production of POPs as unintended byproducts, which contaminate the residues from waste incineration.

Dioxins in residues from waste incineration are a perfect example of how incinerators contribute to surpassing planetary boundaries concerning chemical pollution. According to data from 2008, the estimated total global emissions of dioxins were 101.4 kg TEQ annually. This estimate was based on summaries of national inventories of dioxin emissions and transfers (Fiedler, 2016). In 2021, a study presented at the international conference 'Dioxin 2021', estimated globally and with the highest probability, that the amount of dioxins transferring into the fly ash and residues from flue gas treatment in waste incinerator plants, to be between 14 and 15 kg TEQ of dioxins annually. In the respective study, an analysis was conducted on how this estimate was reached (Petrlik et al., 2022).

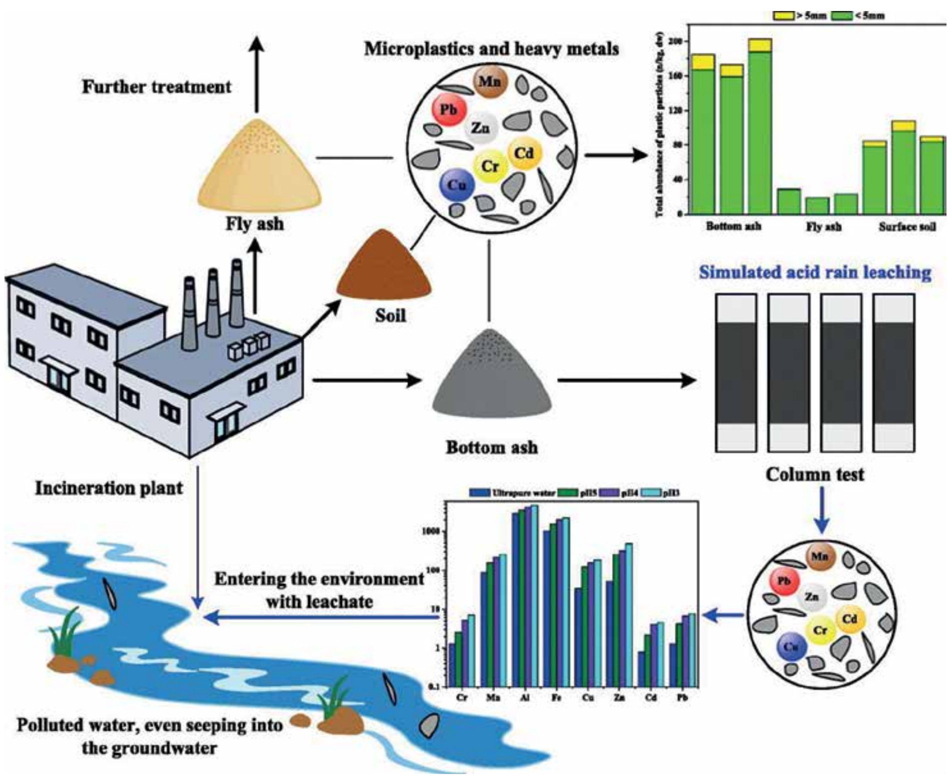


Figure 4.4: *A graphic abstract of the study focusing on tracking microplastics and heavy metals from waste incineration effectively documents their pathways. (Source: Shen et al., 2021)*

Based on the current and globally accepted LPCL limit (15 ng TEQ.g^{-1}), at least half of the 14 to 15 kg TEQ of PCDD/F[27] dioxins annually, remains in fly ash from waste incineration. It is likely even more.

The European Food Safety Authority (EFSA) tightened the value of the tolerable weekly intake (TWI) of dioxins and dl PCBs to 2 pg TEQ.kg^{-1} of body weight in 2018 (EFSA CONTAM, 2018). This means that a person weighing 70 kg should not ingest these substances in an amount exceeding 7,280 pg TEQ per year. For the entire population on Earth (7.7 billion people in 2019), this equates to a value of 56,056 g TEQ of dioxins per year.

Therefore only half of the quantity of dioxins, that remain annually in fly ash from waste incineration (and are expected to remain uncontrolled) corresponds to the annual tolerable dose for the human population of 133 planet Earths. If we use the TWI value of 14 pg TEQ.kg^{-1} body weight (previously used by WHO), then this quantity of dioxin corresponds to the

27 The shift to a stricter limit of 5 ng TEQ.g-1, which has occured in the EU, will not make much difference to the estimate of dioxins in waste (fly ash) left unchecked. While the 15 ng TEQ.g-1 limit was exceeded by only one of the 35 incinerators where dioxins in fly ash were measured in the EU, the 5 ng TEQ.g-1 limit was exceeded by four of the 35 incinerators, or 11% (Ramboll, 2019).

Photo 4.6: *Fly ash from incinerators is a major source of dioxins and escapes control due to lenient limits. Equipment for collecting and transporting fly ash from hazardous waste incinerator in Jihlava, as it looked in 2012. Photo: Matěj Man, Arnika.*

annual tolerable dose for the entire population of 19 planet Earths. Fortunately, not all dioxins end up in our food chain. But the scale of dioxin contamination evident here is a clear indication of novel entities such as dioxin surpassing planetary boundaries and yet this global contamination load is contributed by incinerators solely through the production of dioxins in fly ash.

4.3 Biodiversity

A recently released study (Sigmund et al., 2023) links biodiversity loss to the presence of anthropogenic chemical substances ubiquitous in the environment, posing a growing threat to biological diversity and ecosystems. Waste incineration contributes to biodiversity loss by releasing toxic substances of anthropogenic origin into the environment (emissions, residues from waste incineration, wastewater) and simultaneously producing those that would not exist without incinerators. POPs released from waste incinerators significantly contribute to biodiversity endangerment. Just as they affect human reproductive capabilities, these substances also threaten the capabilities of other animals.

Clear evidence of the negative impact of POPs on populations of wild animals has been monitored by humanity since at least the 1960s, including dioxins and PCBs. Research from the early 1960s in the Great Lakes watershed noted reproductive failures in minks fed fish from a farm with high concentrations of organochlorine compounds. Fetal deaths and abnormalities were associated with dl PCB and PCDD/F (Brunström et al., 2001; Giesy et al., 1994; Wren, 1991). A 30-year longitudinal study of Swedish otter populations (1968-1999) showed that increasing concentrations of PCBs in the environment correlated with reduced success in rearing offspring (Roos et al., 2001).

Some heavy metals and their compounds released from waste produced by incinerators, contribute to endangering aquatic organisms, for example. Ash from incinerators contains high concentrations of zinc or copper and its compounds, which are not as hazardous for humans as they are for aquatic organisms (Javed & Usmani, 2017; Sibiya et al., 2022), see, for example, Chapters 5.3.7 and 5.3.8. Uncontrolled handling of bottom ash from incinerators adds to the overall environmental burden of zinc and its compounds, as well as other heavy metals (see Figure 5.21).

One of the most bizarre cases of biodiversity endangerment is the contamination of a coral reef off Bermuda by depositing residues from waste incineration into an underwater landfill (see Figure 4.5). Bermuda constructed an incinerator in 1995. Contamination by chlorinated dioxins,

Photo 4.7: *The symbol of POPs endangering the animal kingdom first became the white-tailed eagle in the last century, whose ability to reproduce was threatened by DDT. Photo: Wikimedia Commons.*

Photo 4.8: *The European otter, with reduced success in rearing offspring due to increasing PCB concentrations in the environment. Photo: Bernard Landgraf, Wikimedia Commons.*

Photo 4.9: *Compounds of zinc, found in high concentrations in bottom ash and slag from municipal waste incinerators, can be toxic to fish. Illustrative photo of a pike – author: katdaned – Pike, CC BY 2.0, https://commons.wikimedia.org/w/index.php?curid=42498819*

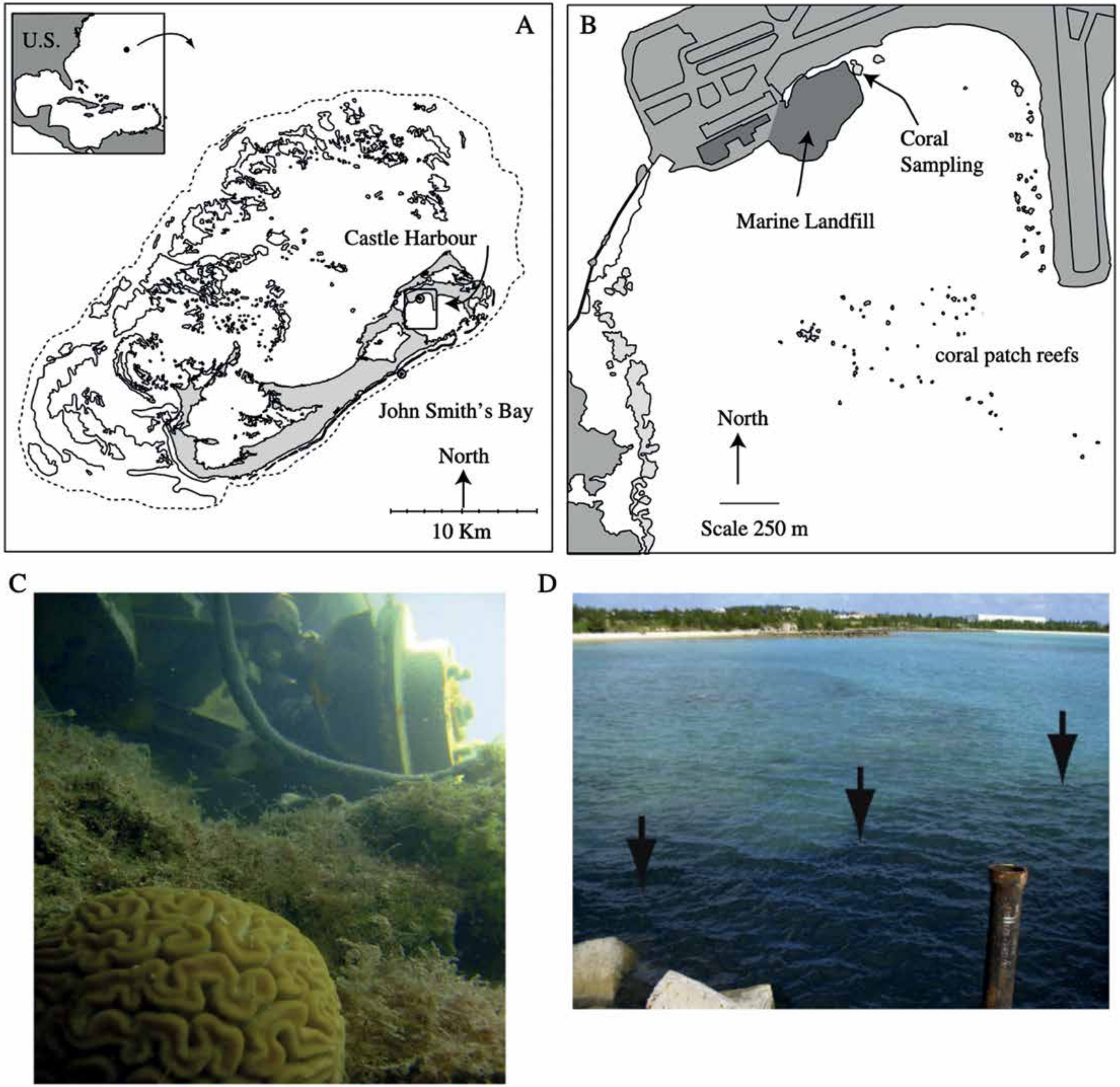


Figure 4.5: *(A) Map of Bermuda and its location in relation to the east of the United States, along with locations where coral drilling was carried out at Castle Harbour and John Smith's (marked as round points). The coral reef area is delineated by dashes. (B) Enlarged Castle Harbour and the location of the underwater landfill, coral collection sites, and mottled coral reefs within the harbor. (C) Photograph of a Diploria labyrinthiformis coral head approximately 20–30 cm in size from Castle Harbour against the backdrop of the underwater waste landfill. (D) Black water flowing from the landfill in Castle Harbour during low tide, when anoxic (oxygen-deprived) material is released from the landfill. (Source: Prouty et al., 2013)*

Photo 4.10: *Sites where waste containing POPs is incinerated, which can then enter food chains, pose a threat to local populations of endangered animal species. The photograph depicts a loggerhead sea turtle after laying eggs in a mangrove swamp adjacent to a cement plant incinerating waste with PFAS in Queensland, Australia (see Photo 5.21); (Limpus et al., 2013). Photo: Jindřich Petrlík, Arnika.*

polycyclic aromatic hydrocarbons, polychlorinated biphenyls, and heavy metals (zinc, lead, manganese, and mercury) was detected, while the disposal of ashes solidified with cement turned out to be one of the likely sources (Jones, 2011; Prouty et al., 2013).

4.4 Biogeochemical flows of phosphorus and nitrogen

Nitrogen (N) and phosphorus (P) are indispensable for life, playing pivotal roles in the intricate web of biochemical processes that sustain ecosystems. These essential elements undergo continuous cycles, transitioning between the abiotic realms of land, water, and air, and the living organisms that inhabit them (Krieger, 2022). Over millennia, natural ecosystems have finely tuned these cycles to maintain equilibrium. However, human activities have disrupted this delicate balance, resulting in excessive levels of N and P in circulation, while simultaneously triggering a concerning scarcity of phosphorus (Sandström et al., 2023).

Much like the N cycle, the P cycle has fallen into disarray. Phosphorus, crucial for the development and maintenance of vital bodily structures like teeth and bones, was historically sourced from guano, rich in phosphates, traded globally to enhance soil fertility. Today, raw phosphates are extracted from mines, processed into soluble fertilizers, or directly

Photo 4.11: *The final segment of the 98 km-long conveyor belt, transporting phosphates to the Port of Laâyoune. Photo by Pera Outdoor via Panoramio on Google Earth.*

Photos 4.12 and 4.13: *Another segment of the conveyor belt for transportation of phosphates and their mining in Western Sahara. (Source: Jain, 2021).*

applied to agricultural fields (Kalmykova et al., 2015). However, a significant portion of the phosphorus fertilizer applied to fields doesn't reach plants; instead, it leaches into water bodies, particularly during heavy rains, leading to harmful algal blooms and toxic cyanobacteria, rendering waterways inhospitable and, in severe cases, causing "dead zones" devoid of life in the oceans (Krieger, 2022).

Efforts to address P scarcity include initiatives to reclaim it from sewage sludge, with mandatory P recycling programs set to be implemented in large sewage treatment plants (Scholz & Wellmer, 2013). It's imperative to ensure that this recycling process doesn't inadvertently reintroduce pollutants such as heavy metals.

Based on the analysis of Carpenter and Bennett (2011), Steffen et al. (2015) proposed an additional regional-level P boundary, designed to avert widespread eutrophication of freshwater systems, at a flow of 6.2 metric tons P*/yr* from fertilizers (mined P) to erodible soils.

Moreover, it's essential to recognize the N and P are not the sole substances vulnerable to disruption by human activities. Consequently, future considerations regarding planetary nutrient flow boundaries may need to encompass a broader spectrum of substances (Krieger, 2022).

Disparities in nutrient usage, particularly phosphorus, peaked between 1950 and 1990 before plateauing. Notably, after 1990, high-input countries in Western Europe decreased their synthetic fertilizer usage, while lower-input regions in South America and Asia increased their use of activated nutrients. However, despite reductions, Western Europe still significantly exceeds sustainable P usage levels (Liu et al., 2008). This disparity if well visible in map-diagrams in Figures 4.6 and 4.7 (Sandström et al., 2023).

Sweden's potential for P self-sufficiency highlights the importance of reclaiming it from urban waste sources such as food waste, sewage sludge, and municipal solid waste incineration residues (European Commission, 2008). Phosphorus flow in municipal waste in Gothenburg, Sweden quite well shows how much of it is lost in waste incineration residues (Kalmykova et al., 2012).

Modern agriculture heavily relies on fertilizers for food, feed, fiber, and biofuels production. N and P are irreplaceable yet finite resources, derived predominantly from rock deposits. While physical scarcity seems remote, economic scarcity, driven by escalating prices, is a present reality. P management, once primarily concerned with mitigating aquatic pollution and eutrophication, has shifted focus to its strategic importance in agricultural production and food security (Antikainen et al., 2005; Sokka et al., 2004). Despite improvements in phosphorus recycling, primarily from wastewater (Sokka et al., 2004), significant losses still occur. Yet, during the period 1995–1999 in Finland for example, sewage sludge contained only about 2.4 % of the nitrogen and 11.8 % of the phosphorus applied to agricultural soils as synthetic fertilizers during the same time period (Antikainen et al., 2005; Sokka et al., 2004).

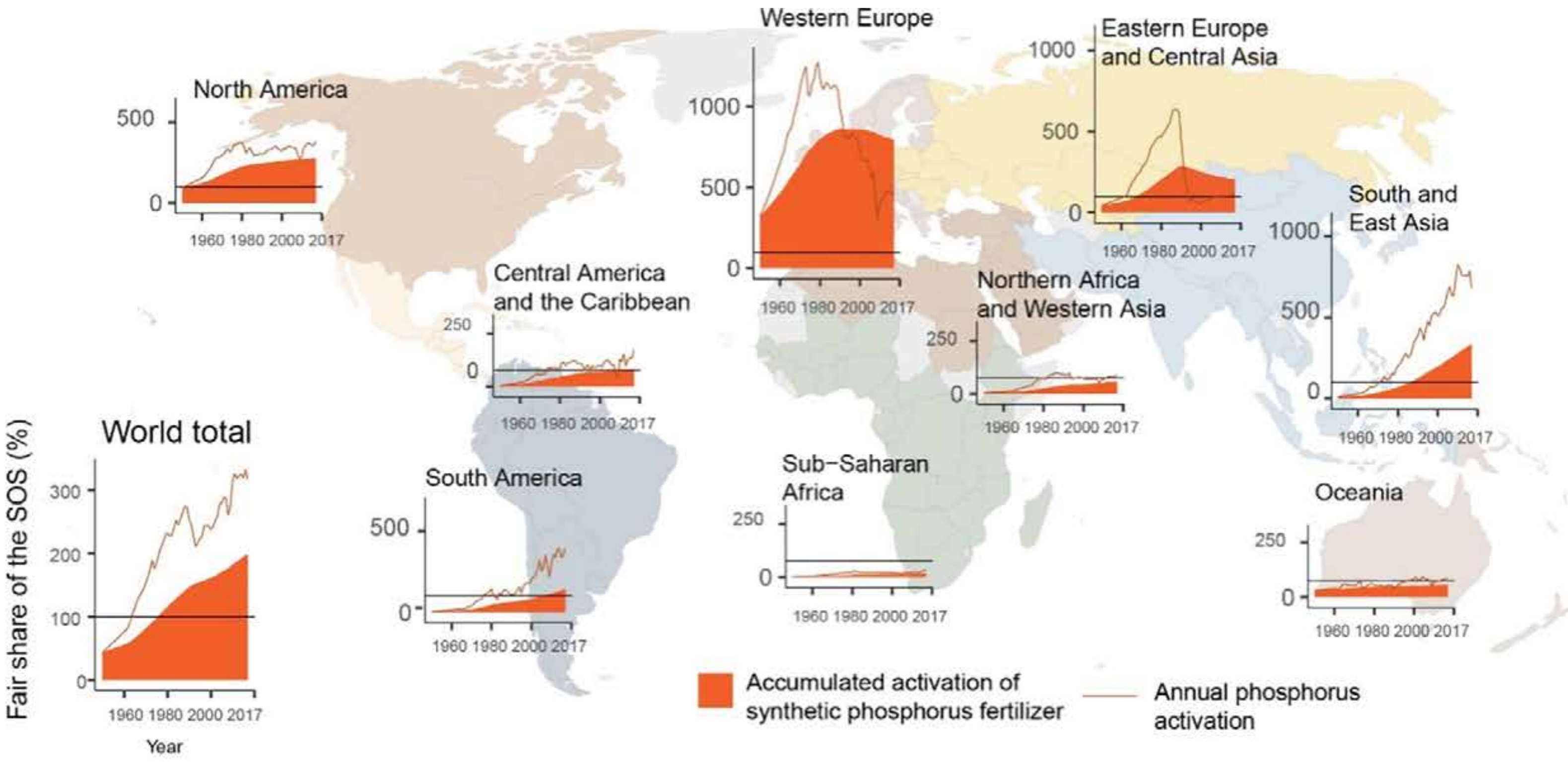
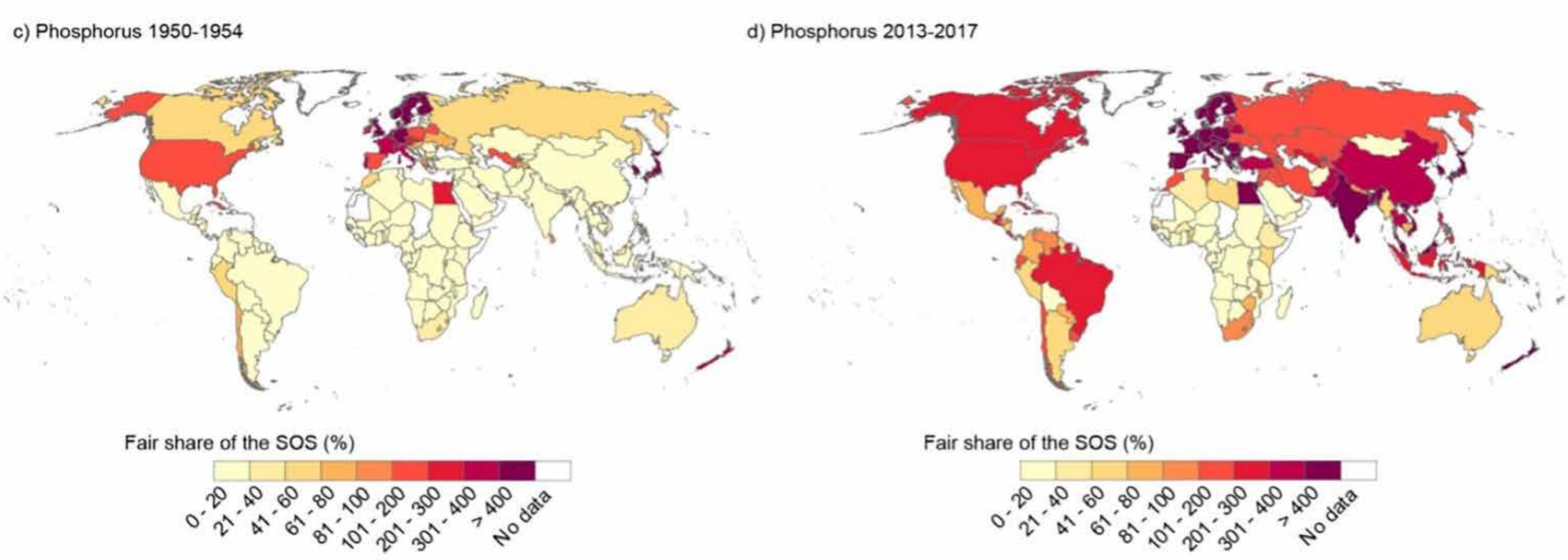


Figure 4.6: *Cumulative and annual use of activated phosphorus relative to regions' fair share of the safe operating space (SOS) from 1950 to 2017. The relative use of the fair share was calculated by dividing the cumulative sum of annual nutrient activation by cumulative sum of annual fair share of the SOS applying planetary boundary of 6.2 Tg P.yr^{-1}. Note that the plots show regional and global means weighted by agricultural area. (Source: Sandström et al., 2023)*

Figure 4.7: *Cumulative use of the countries' fair share of the safe operating space (SOS), c) phosphorus 1950–1954, and d) phosphorus 2013–2017. The use of the fair share was calculated by dividing the cumulative sum of annual nutrient activation by cumulative sum of annual fair share of the SOS. (Source: Sandström et al., 2023).*



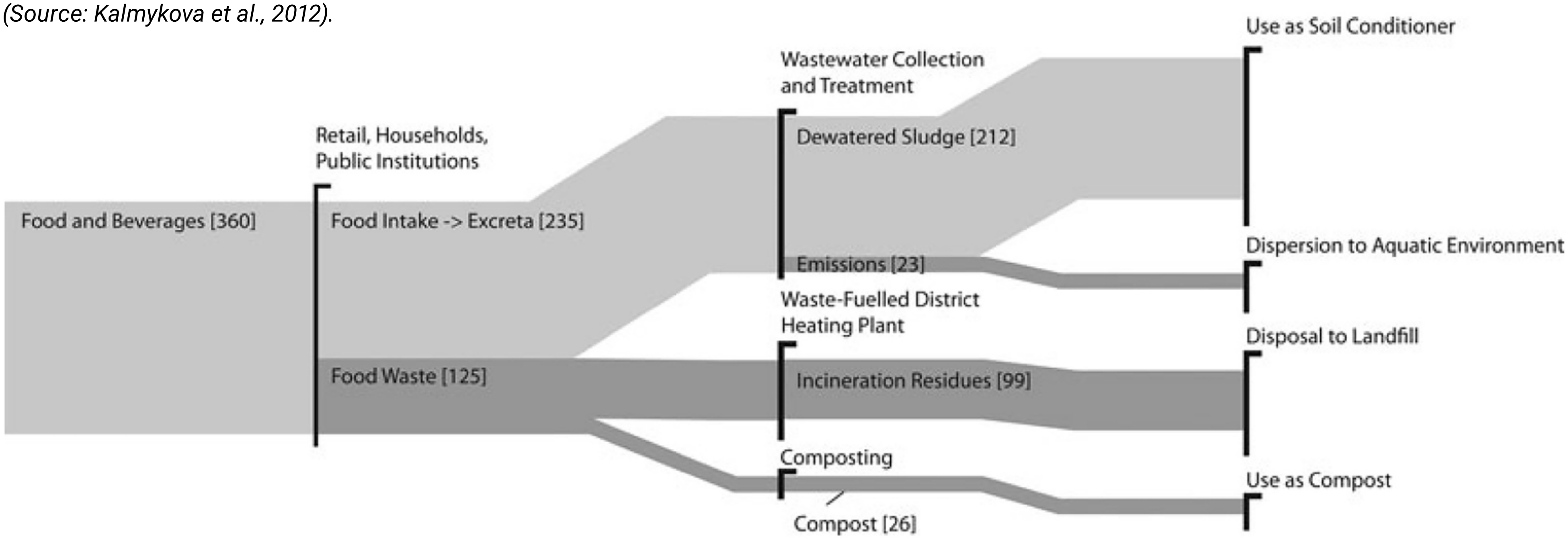
Efforts to mitigate phosphorus demand include recycling from secondary sources such as agricultural waste and human waste. While manure is largely recycled, sewage sludge and solid waste recycling rates remain low. Solid waste, often overlooked as a P source, presents a considerable challenge, compounded by inadequate organic waste recycling efforts (Borking, 2011).

"Every year, almost 10,000 tonnes of phosphorus is lost in the Danish incinerators," estimates Professor Lars Stoumann Jensen from the University of Copenhagen's Faculty of Biosciences (KU-LIFE). This roughly corresponds to the amount we import annually as feed phosphate.

We are slowly draining the soil of the nutrients that the plants depend on. Farmers compensate for this with, among other things, phosphorus fertiliser. The problem is just that the price of P has increased fivefold in recent years and that the very few mines in the world that contain raw phosphate will one day be exhausted (Borking, 2011).

As incineration of waste is increasingly common waste management option worldwide and the P, along with other elements, is expected to be stored in the incineration residues. A method for P recovery from the MSWI ash has recently been developed (Kalmykova & Fedje, 2013). However, it is

Figure 4.8: *Dispersion pathways for phosphorus contained in food products in Gothenburg, Sweden, in 2009, in tonnes of phosphorus per year (t P.yr^{-1}). (Source: Kalmykova et al., 2012).*



better to avoid incineration of MSW and recover P by composting sorted biowaste (see also chapters 9.1.3 and 10.2.4).

Composting the organic fraction of MSW is by comparison a more cost effective and efficient way to provide P to the agricultural sector and also comes with significant climate benefits compared to raw materials extraction and processing of P. Studies have demonstrated that a comparison of technologies for food organic waste processing is best achieved through Anaerobic digestion with incineration and landfill being the least effective option (Gao et al., 2017; Nordahl et al., 2020).

Further, the organic fraction of MSW streams (in most countries it is the highest volume of waste) that are lost to incineration, represents a wasted opportunity to compost, which provides significant climate and biosphere benefits through carbon sequestration, methane reduction and critical nutrient support for agriculture. Composting organic waste thus represents a significant way to address the triple planetary boundaries that our our unsustainable materials production systems and associated waste management systems are driving. Composting organic waste is the most transformative waste management decision any government can make right now (Walsh, 2022) and is recognised in progressive international projects such as the *4 per 1000 initiative*'.

5. Toxic Substances from Incinerators, their Flows, and Health Impacts

5.1 Persistent Organic Pollutants (POPs)

POPs are substances capable of persisting in the environment for long periods—under normal conditions, they degrade or break down very slowly, if at all. They are highly harmful, characterized by high toxicity, with some being carcinogenic, mutagenic, or teratogenic.[28] Due to their low degradation rate, they travel over long distances (by water, air and organisms) and accumulate in living organisms, multiplying their negative effects. Although they do not act as immediately lethal toxins, their gradual accumulation in the body causes a range of serious illnesses. Some of these substances have been and are still used in industry and agriculture, but often they are unintentionally produced—for example, through waste incineration or as chemical by-products.

The management of POPs and the emissions of those POPs generated as by-products in incinerators, are globally regulated by the Stockholm Convention, which was agreed upon in May 2001 and entered into force on May 17, 2004. Each contracting party must develop a national implementation plan (NIP) and regularly update it, to fulfil its obligations. Hazardous waste incinerators and cement plants are often used for the disposal of POPs-containing waste, although they do not represent an ideal method for breaking down these substances and, simultaneously, new POPs can be generated as unintentional by-products when burning waste such as PCB or PFOS. Unintentionally formed POPs (UPOPs) are formed by combustion processes even when burning waste not containing POPs as well as municipal and household waste which may contain end of life POP contaminated products. Dioxins are among the most well-known UPOPs. Dioxin and other UPOPs are listed in Appendix C of the Stockholm Convention. Incinerators and cement plants, along with metallurgical operations, are listed as significant sources of dioxins in the Stockholm Convention annexes (MFA, 2006).

Some of the newly listed POPs under the Stockholm Convention show how the world around us is changing and becoming increasingly complex, particularly concerning toxic substances. More substances with specific properties are being manufactured, complicating their breakdown later. The incineration of waste cannot be viewed without consideration of the increasing range, behavior and impact of chemicals being manufactured, which the act of incineration tends to widen, and not just regarding dioxins. For instance, published evidence suggests that incinerating highly complex PFAS can generate new forms of these substances through thermal degradation, which could be as toxic as their original forms. Incineration of waste

28 Teratogens are external factors that are capable of causing a developmental defect or that significantly increase the risk of such defect.

products containing brominated flame retardant POPs such as polybrominated diphenyl ethers (PBDEs) can lead to the formation of brominated dioxins in emissions and ash. Brominated dioxins have been found to have similar toxicity to chlorinated dioxins which are more commonly known but are largely unregulated. This may soon change, as Switzerland has proposed adding brominated dioxins to the Stockholm Convention list as part of a larger group of polyhalogenated dioxins (PXDD/Fs), (POP RC, 2024).

5.1.1 Chlorinated dioxins

The collective term "dioxins" refers to a group of structurally and chemically related polychlorinated dibenzo-p-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs). The term "dioxins" sometimes also includes 12 dioxin-like congeners of polychlorinated biphenyls (dl PCBs), which have similar toxic properties to dioxins and are assigned toxic equivalency values (TEQs) representing PCBs and are included in Appendix C of the Stockholm Convention (MFA, 2006). Dioxins are found worldwide in the environment and accumulate in the fatty tissues of organisms because they are lipophilic (attracted to fat), leading to their accumulation in food chains.

Over 90 % of human exposure to these substances occurs through food—mainly via meat and dairy products, fish, and crustaceans. The original source of the dioxins contaminating this food is from industrial emissions and releases. Short-term exposure to high concentrations of dioxins, usually in chemical plants or after chemical accidents unrelated to waste incineration, can result in skin lesions (chloracne), spotted skin darkening, and liver function changes. Long-term toxic effects include reproductive and developmental problems, including nervous system damage in children, immune system disruption, endocrine system disruption, and cancer (Anwer et al., 2016; Carpenter, 2013; Eskenazi et al., 2018; Giesy & Kannan, 1998). Once dioxins enter the body, they persist for a long time due to their chemical stability and the ability to be absorbed into fatty tissues, where they then accumulate. The half-life of dioxins in the human body is 7 to 11 years (WHO, 2016). Nursing women can eliminate them more quickly through breast milk (transferring dioxins to their offspring). Health effects of dioxins and dl-PCBs are summarized in the diagram at Figure 5.1.

The developing fetus is most sensitive to exposure to dioxins. Some individuals (or groups) may be exposed to higher levels of dioxins due to their diet (for example, heavy fish consumers in certain parts of the world) or their occupation (for example, workers in the chemical industry, waste incineration, or hazardous waste sites). Given the extent of environmental contamination with dioxins, efforts must be made to reduce it, most effectively directly at their sources. Dioxins are primarily by-products of industrial processes, especially chlorination chemistry, metal smelting, or incineration of chlorinated waste (MFA, 2006; UNEP & Stockholm Convention, 2013). They can also be produced to a lesser extent due to natural processes, such as volcanic eruptions and forest fires. All types of waste incinerators are major sources of dioxins, including waste co-incineration in cement kilns. An expert panel of the Stockholm Convention created the Dioxin Toolkit to calculate total dioxin emissions, setting emission factors for various processes and their products (UNEP & Stockholm Convention, 2013).

To determine the specific dioxin effect of a particular sample, the bioassay analysis called DR CALUX is used. The DR CALUX method determines the concentration of PCDD/Fs and dl PCBs expressed as bio-toxic equivalent BEQ (European Commission, 2012), which indicates how the sample behaves compared to the most toxic congener[29] of dioxins – 2,3,7,8-TCDD

29 Congeners are substances that are structurally similar. In the case of dioxins (PCDD/F), the differences lie in the location and number of chlorine atoms. Dioxins are a part of a group of 75 congeners of polychlorinated dibenzo-p-dioxins (PCDDs) and 135 congeners of polychlorinated dibenzofurans (PCDFs), of which 17 are toxicologically significant.

(Besselink et al., 2004a; Besselink et al., 2004b). It is described in the US EPA 4435 methodology. The extract of the sample is applied to a genetically modified cell line. Dioxins or dl PCBs (or other substances) activate the aryl hydrocarbon receptor, initiating the expression of the observed gene. The concentration of dioxins and similar substances in the sample is determined based on the response level. In comparison with chemical analysis, in this case, we don't have information about the concentration and present congeners, but rather information about the sample's effect on mammalian or human cells. The BEQ value can also be influenced by other substances with a dioxin effect, such as brominated dioxins, among others. In this form of sampling more dioxin-like activity can be detected than the standard laboratory analysis which targets specific regulated dioxins. In this way the full impact of all dioxin like chemicals in the sample can be assessed – not just the concentration level of a specific target chemical in the sample.

5.1.1.1 Air

Monitoring emissions of certain substances (including dioxins) from waste incinerators occurs usually only for a few hours per year (e.g. 6 - 12 hours), which might correspond to only a fraction of the total operational

***Figure 5.1:** Toxicity mechanism and health effects of PCDD/Fs and dl PCBs. (Source: Dai et al., 2020)*

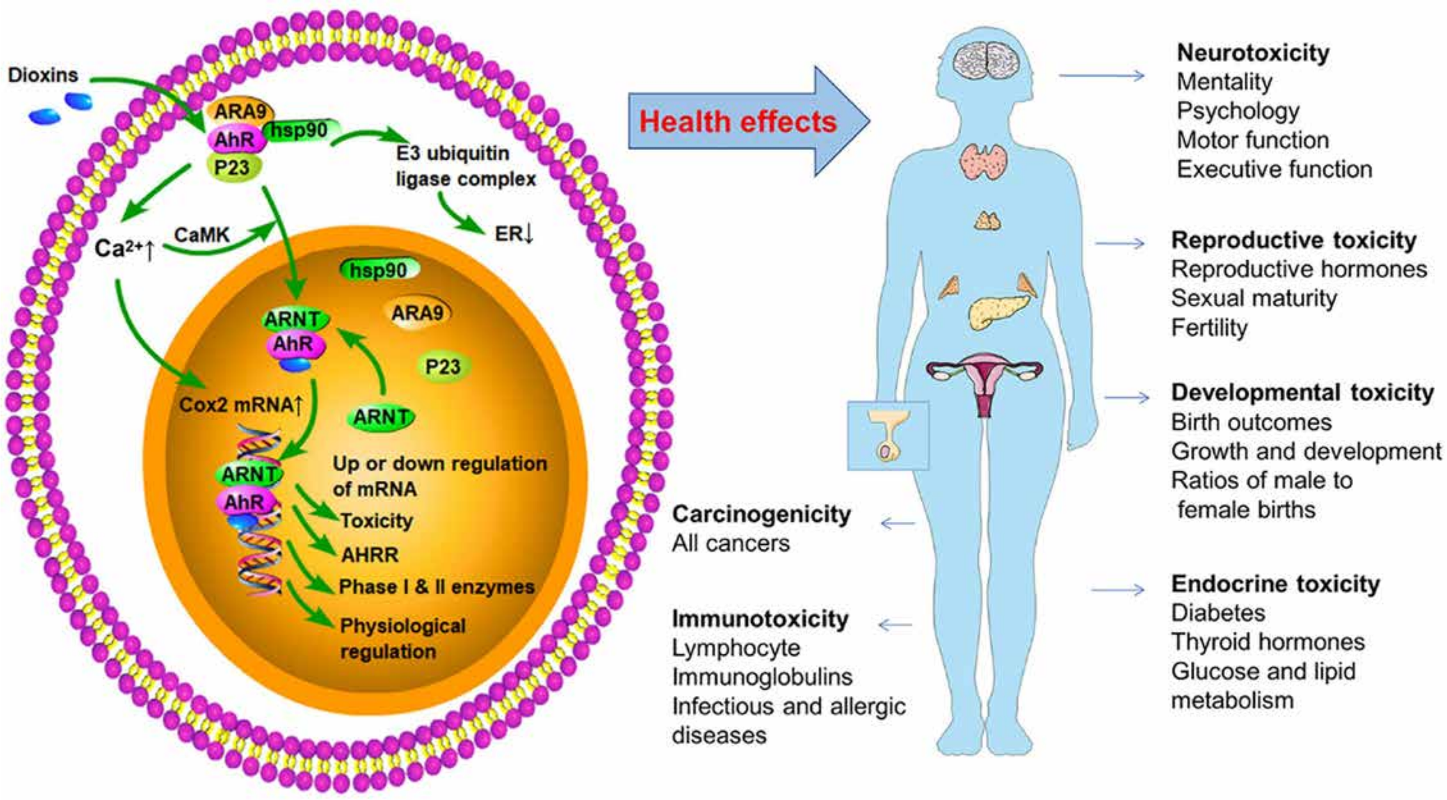
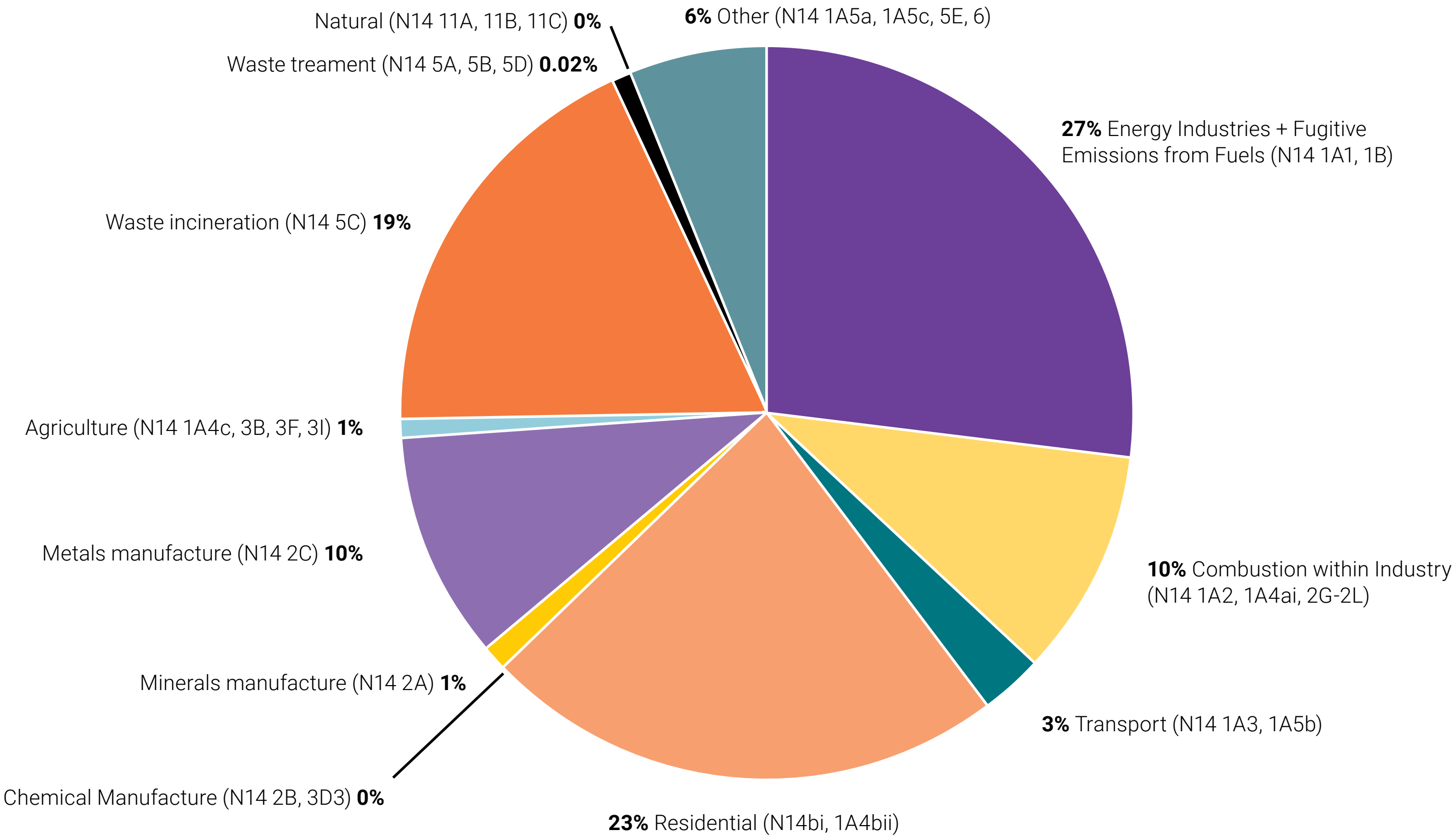


Figure 5.2: Sources of polychlorinated dioxin and furan emissions into the air in the EU-28 in 2015 (European Commission, 2022b)



time of the facility (typically around 8,000 hours per year). Additionally, this monitoring usually happens under stable operational conditions and not during start-ups or shut-downs of the main furnace, during which significantly higher emissions of dioxins might occur (for further details, see the text later in this chapter or in Chapter 3.1.1.1). Other situations where high dioxin emissions occur include power outages to the flu gas scrubbers such as electrostatic precipitators, wet scrubbers, and activated carbon injection units. It can also occur when the stacks are periodically 'cleaned' of soot and scale where large quantities of particulates are forced out of the stack. Another circumstance is filter bypass operations where pressure or temperature is too high for the filter systems and the emissions can be channelled through other pipes and released to atmosphere bypassing the air pollution control devices. Collectively these and other unstable incinerator operations are known by regulators as Other Than Normal Operating Conditions (OTNOC) and in most cases no monitoring of dioxins occurs during these situations where high dioxin releases are expected. In the EU, new regulations seek to address this loophole.[30]

30 https://zerowasteeurope.eu/press-release/long-awaited-revamp-of-industrial-emissions-directive-improves-dioxin-monitoring-in-incinerators/

The conclusions on best available techniques (European Commission, 2019) are part (an excerpt) of an extensive document, BREF, and are binding for permitting new sources of pollution in industry and agriculture in the European Union. In 2015, the BREF for waste incineration was in effect, demanding an emission limit for dioxin of 0.1 ng TEQ.m^{-3} (European Commission, 2005); current levels can be found in Chapter 3.1.1. Despite such a low limit, waste incinerators were responsible for 19 % of dioxin emissions into the air in the EU-28 in 2015 (European Commission, 2022b), see Figure 5.2. After the release of these substances into the air, they undergo transport and deposition in the nearby and/or distant vicinity of the incineration source.

Dioxin emissions from most modern waste incinerators, when implementing the best available techniques, range from 0.0008 to 0.05 ng I-TEQ.m^{-3} (Stockholm Convention, 2008). According to the currently applicable BREF document (Neuwahl et al., 2019), they range between the quantification limit and 0.24 ng I-TEQ.m^{-3}. Facilities equipped with a solid adsorption layer showed emission levels below 0.05 ng.m^{-3} (Neuwahl et al., 2019). However, poorly designed and operated facilities might have substantially higher emissions. Requirements for emission levels of PCDD/PCDF or PCDD/F with dl PCB are regulated by the latest conclusions on best available techniques for waste incineration from 2019, as shown in Table 5.1.

Whether waste incinerators meet the set limits for dioxins is usually checked by two short-term measurements of a few hours throughout the year. In the Czech Republic, this option is left to the decision of the authority issuing the so-called integrated permit, even though it has been clearly demonstrated that short-term measurements do not show the real emission levels of dioxins. They fail to capture critical periods during the start-up and shutdown of technology (De Fré & Wevers, 1998; Kriekouki et al., 2018), occurring annually in connection with necessary

Table 5.1: *Emission levels associated with best available techniques (BAT-AEL) for controlled emissions of TVOC, PCDD/F, and dl PCB from waste incineration into the air. (Source: European Commission, 2019)*

Parameter	Unit	BAT-AEL		Averaging Period
		New Facility	Existing Facility	
TVOC	mg.m^{-3}	< 3–10	< 3–10	Daily Average
PCDD/F*	ng I-TEQ.m^{-3}	< 0.01–0.04	< 0.01–0.06	Sample Collection Interval Average
		< 0.01–0.06	< 0.01–0.08	Long-Term Sample Collection Interval**
PCDD/F + dl PCB*	ng WHO-TEQ.m^{-3}	< 0.01–0.06	< 0.01–0.08	Sample Collection Interval Average
		< 0.01–0.08	< 0.01–0.1	Long-Term Sample Collection Interval**

** Either BAT-AEL for PCDD/F or BAT AEL for PCDD/F + dl PCB will be used.*
*** BAT-AEL won't be used if it's demonstrated that emission levels are sufficiently stable.*

maintenance of the technology. They also don't capture other failures in flue gas cleaning. In the case study of a modern incineration plant (WtE) in Harlingen, the Netherlands (see Chapter 3.5.3), we demonstrate how significant the difference in emissions of dioxins (and not only them) can be, when calculated based on one-time measurements (lasting for about 24 hours) compared to measurements using continuous emissions sampling (Cheruiyot et al., 2016; Reinmann, 2011). Similarly critical is the measurement of mercury emissions, for which continuous emission sampling can also be used. For both, see Chapter 3.1.1.1.

Photo 5.1: *WtE Termizo Liberec (Czech Republic) initially failed to meet the 0.1 ng TEQ.m-3 limit for dioxins, thus requiring the costly addition of a filter (see Chapter 9.1.1). Photo: Jindřich Petrlík.*

Photo 5.2: *Dioxins can spread through fugitive emissions, such as when handling residues from waste incineration, as depicted in this image from Harlingen, the Netherlands, or Photo 4.6 from a hazardous waste incineration plant in Jihlava, where noticeable dust leakage occurred during fly ash handling. Photo: ToxicoWatch. (Source: Arkenbout & Bouman, 2018).*

According to Reinmann (2011), a semi-continuous system for measuring dioxin emissions was used in Belgium between 1999 and 2000 for municipal waste incinerators and later for hazardous waste incinerators, cement kilns, and other facilities. The first country in the world to adopt these legislative requirements was France in 2010, which began semi-continuous monitoring of dioxins in municipal and hazardous waste incinerators (approximately 200 sources). Worldwide, about 450 to 500 sources are monitored in this way, of which 160 were monitored by the AMESA system as of 2011. Due to the interest in continuous monitoring of dioxins, the EN 1948-5 standard for continuous monitoring of PCDD/F and dl PCB was developed and started to be implemented.

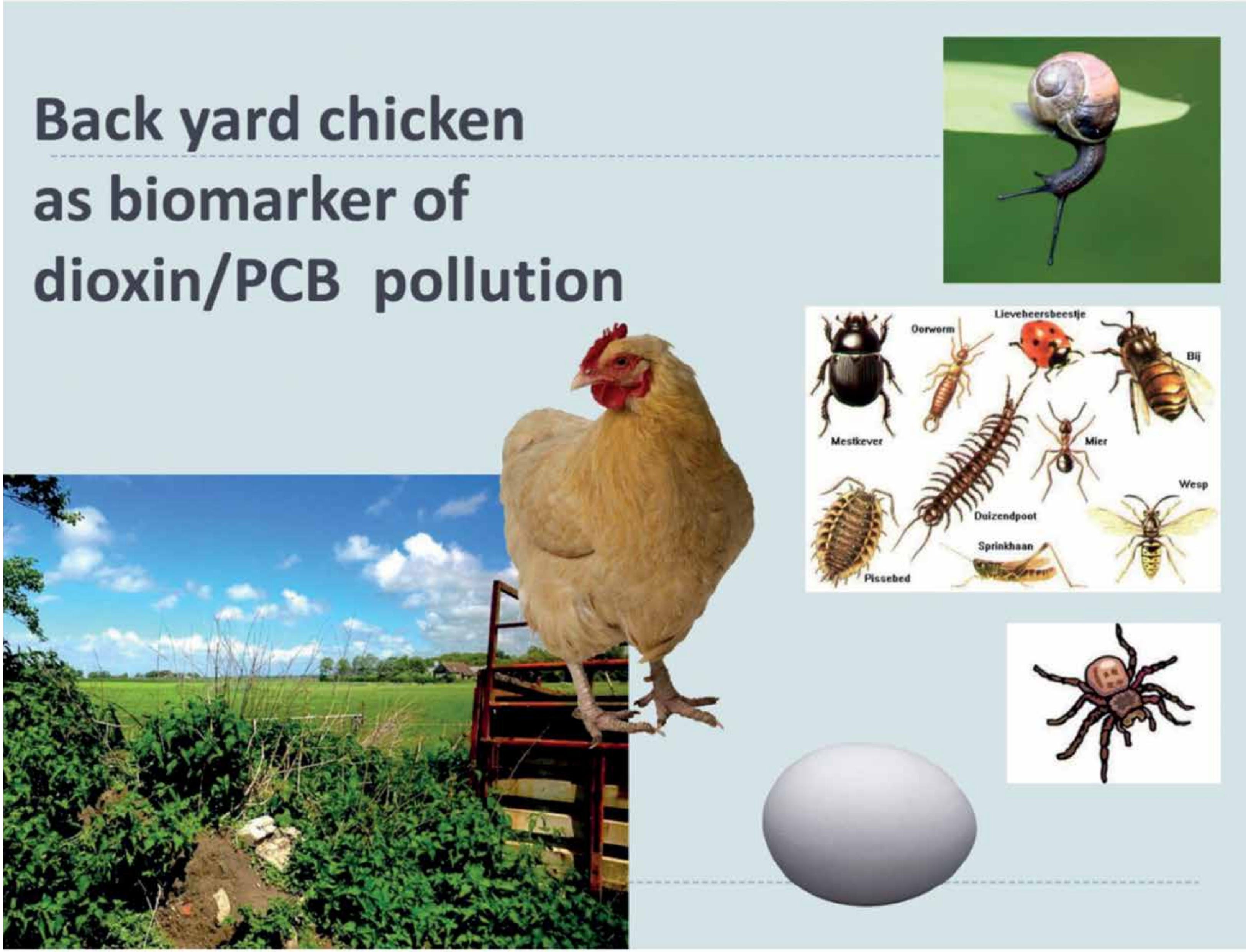


Figure 5.3: *Some studies have shown that dioxins and dl PCB in free-range chicken eggs exceed acceptable levels for food in the EU (PCDD/F 2.5 pg TEQ.g-1 fat in eggs) already at PCDD/F concentrations in soil of 2-4 ng TEQ.kg-1 dry matter. Chickens consume a lot of soil organisms, including soil itself, as well as a large amount of dust particles. Source of the infographic: (Arkenbout & Bouman, 2018).*

5.1.1.3 Solid residues from incineration

Before stricter limits and monitoring (Reinmann et al., 2011), typical municipal waste incinerators, with capacities of 50,000 to 200,000 tons of burned waste per year, under stable conditions, released up to 100 g TEQ PCDD/F annually into the air between 1950 and 1980 (Rappe et al., 1987). Such leaks contaminated soil with dioxins around waste incineration plants. For instance, in the soil around a hazardous waste incineration plant in the UK, concentrations reached up to 58 pg TEQ.g^{-1} (Holmes et al., 1995), in the soil around a waste incineration plant in the US, concentrations reached up to 450 pg TEQ.g^{-1} (Goovaerts et al., 2008), and in soil taken near a long-term-operated municipal waste incineration plant in Switzerland, concentrations reached up to 640 pg TEQ.g^{-1} (Vernez et al., 2023). Near a small medical waste incineration plant in Ostrava – Poruba (Czech Republic), concentrations of 19.7 pg TEQ.g^{-1} were measured in 2001 (Jech et al., 2001). For comparison, soils with dioxin concentrations between 1.6-14 pg TEQ.g^{-1} are considered contaminated areas in the Czech Republic due to industrial sources (Ivan Holoubek et al., 2003). For more case studies, see Chapter 3.5.

In the 1990s such high soil dioxin contamination from waste incineration plants led to the contamination of meat and milk from animals raised around these facilities (Goovaerts et al., 2008; Liem et al., 1991). This also happens in places with careless handling of residues from waste incineration (such as bottom ash and fly ash). We have discussed this issue in detail in the chapter on soil impacts (see Chapters 3.4 and 3.5), where specific cases are presented.

Some studies have shown that dioxins and dl PCB in free-range chicken eggs exceed acceptable levels for food in the EU (EU limit for PCDD/F 2.5 pg TEQ.g^{-1} fat in eggs) already at PCDD/F concentrations in soil of 2-4 ng TEQ.kg^{-1} dry matter (Hoogenboom et al., 2016; Kijlstra et al., 2007; Weber et al., 2019); see Figure 5.3.

Concentrations between 0.020-4.224 ng I-TEQ.kg^{-1} dry matter in soil led to concentrations of up to 56 pg WHO-TEQ.g^{-1} fat in eggs in Newcastle (Pless-Mulloli, 2003). Consumers of contaminated eggs, especially young children, can thus be easily exposed to doses of these substances that exceed the Tolerable Weekly Intake (TWI), meaning excessive exposure to dioxins.

Chapters 3.5 and case studies Lausanne (Chapter 3.5.1), Maincy (Chapter 3.5.2), Harlingen (Chapter 3.5.3) and small medical waste incinerators in general (Chapter 3.5.4) address critical cases of soil contamination and subsequently eggs from domestically raised chickens. A similar case of contamination of locally grown food in an area contaminated by hexachlorobenzene from hazardous waste incineration by a cement plant is discussed in the Wietersdorfer case study (see Chapter 5.1.5.1).

A Chinese study from the vicinity of a hazardous waste incinerator (HWI) found the largest PCDD/Fs increase value in soils predicted by integrating AERMOD and a reservoir model was very limited after 25 years (2.03 × 10^{-5} ng WHO-TEQ·kg^{-1}), indicating relatively minor impacts of HWI on surrounding soil. However, there was a noticeable impact on area downwind from the stack in short distance (e.g., within 0.5 km) that should be recognized (see Figure 5.4); (Lin et al., 2020).

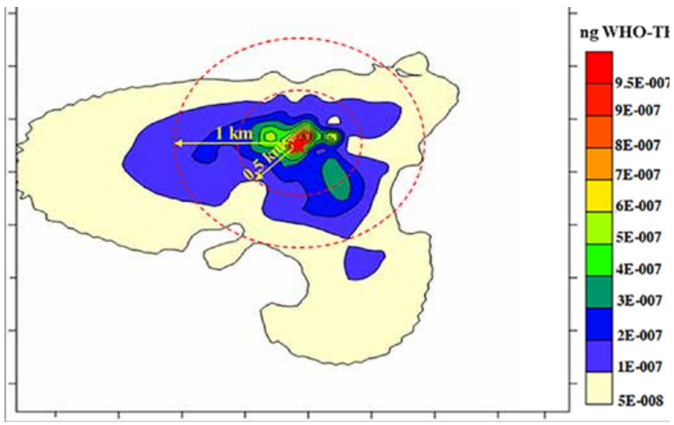


Figure 5.4: *Distribution of PCDD/Fs in soils in the vicinity of the Chinese HWI. (Source: Lin et al., 2020)*

An overview of dioxin releases into various media per ton of waste can be seen in Table 5.2. Substances listed in Annex C of the Stockholm Convention are released or transferred mainly through fly ash, bottom ash, and wastewater treatment sludge. Most of the generated dioxins end up in residues from flue gas treatment. The highest concentration found in fly ash was at a hazardous waste incineration plant in Medellin: 181,535.8 ng WHO-TEQ.kg^{-1} (Cobo et al., 2009). In the fly ash from a modern municipal waste incineration plant in recent years, a concentration of up to 23.9 ng TEQ.g^{-1} was recorded (Ramboll, 2019), and from older data, up to 28 ng TEQ.g^{-1} (Johnke & Stelzner, 1992). Therefore, it is crucial to ensure the safe disposal of these residues and, if the limit for POPs is exceeded, their treatment with non-incineration technologies leading to the destruction/decomposition of POPs (Stockholm Convention, 2008).

The prevailing waste industry opinion is that harmful substances successfully redirected into fly ash are adequately fixed within it, making the content of these substances in waste essentially unnecessary to consider.

Table 5.2: *Estimate of dioxin releases and transfers into various media for municipal waste incinerators. Source: Stockholm Convention (2008, p. 5)*

Medium	Formation per ton of waste	Unit (per ton of waste)	Average concentration	Unit	Quantity [mg I-TEQ.t^{-1} waste]
Ash	220	kg	46	ng I -TEQ.kg^{-1}	10.12
Fly ash	20	kg	2,950	ng I -TEQ.kg^{-1}	59
Filter cake	1	kg	4,000	ng I -TEQ.kg^{-1}	4
Wastewater	450	L	0.3	ng I -TEQ.L^{-1}	0.135
Air	5 000	m^3	0.02	ng I -TEQ.m^{-3}	0.1
Total					73.355

These views likely stem from older studies (Fischer et al., 1992; Hagenmaier et al., 1992; Ratti et al., 1986) on the behavior of dioxins in soil, supporting the original idea of strong dioxin fixation in fly ash and bottom ash. Rules for leaching tests concerning POPs, especially dioxins, are therefore not based on the (latest) scientific findings – see the next Chapter 5.1.1.3.1. Presumably, no one bothers about them when the limit for dioxins in waste (LPCL) is set so artificially high that fly ash complies with it anyway (see Chapter 5.1.10).

5.1.1.3.1 Pollutant Release and Transfer Register (PRTR) as Source of Information about Dioxin in Ash

Finding information about the quantity of dioxins and other POPs produced by individual waste incinerators and other sources is not easy, if not directly impossible. Determining their presence in residues after waste incineration is not a given, and unfortunately, reporting on dioxins in waste is not mandatory even in the European Pollutant Release and Transfer Register (E-PRTR), so it is up to the discretion of member states whether to introduce such an obligation or not. This has also been reflected in the number of EU member states reporting on dioxins in waste (see Table 5.4). In the Czech Republic, the PRTR was established before joining the EU, and therefore it includes the obligation of chemically specific reporting on substance transfers, including dioxins in waste. (Petrlik et al., 2018; Petrlik et al., 2023).

Among the largest producers of PCDD/Fs in waste in the Czech Republic, according to data in the Czech PRTR system called IRZ,[31] are metallurgy and waste incineration, as evident from the table below (see Table 5.3), with some transfers reported as recycling. Two studies by the IPEN network and Arnika addressed the issue of dioxins in ashes from waste

31 IRZ – Integrated Pollution Register (Integrovaný registr znečišťování)

incineration, highlighting potential dioxin leaks into the surrounding areas where these wastes are managed (Katima et al., 2018; Petrlik and Bell, 2017). Global studies monitoring dioxins and similar substances in free range eggs from domestic in various locations, including those where ash from waste incineration or their surroundings is handled, also document this concern (Jelinek et al., 2023a; Petrlik et al., 2022; Weber et al., 2015).

According to data reported to IRZ, nearly 120 g TEQ PCDD/Fs were transferred in waste between 2014 – 2021, several times more than the emissions into the air. However, not all data on PCDD/Fs transferred in waste is included in IRZ. For example, data on PCDD/Fs in waste from the chemical industry is missing (Bell et al., 2021).

The reported data about dioxins reveals an interesting aspect concerning the municipal waste incineration plant in Liberec (Petrlik et al., 2006).

Table 5.3: *Summary of information on dioxins transferred in waste in grams TEQ per year based on IRZ data. (Source: Ministry of the Environment of the Czech Republic, 2022a; Petrlik et al., 2023).*

Year	2014	2015	2016	2017	2018	2019	2020	2021	Average
Municipal Waste Incineration	14.77	7.42	8.39	28.99	13.87	18.08	8.11	8.74	13.63
Hazardous and Medical Waste Incineration	10.67	23.7	17.4	18.979	31.89	39.43	45.16	9.13	22.01
Metallurgy	25.8	48.6	37	199.25	171.43	129.78	106	70.7	83.37
Total	51.24	79.72	62.79	247.22	217.19	187.29	159.27	88.57	119.01

Despite its exemption from reporting obligations to the PRTR, the incinerator became a significant source of dioxins in waste when it temporarily halted its practice of converting waste into a product for European REACH regulation compliance in 2011. This revealed that 8.8 g TEQ of dioxins ended up in the incinerator's waste (Petrlik, 2013).

The European Food Safety Authority (EFSA) established a tolerable daily intake of 0.25 picograms of TEQ (toxic equivalent) per kilogram of body weight in 2018 (EFSA CONTAM, 2018). Extrapolating this to a yearly intake for 1.25 billion people, the tolerable amount is determined to be 8 grams of TEQ.[32] In the context of a city with 100,000 inhabitants, like Liberec, it would be considered harmful if just 0.008% of this specified amount (8.8 g TEQ.y^{-1}) of dioxins were to enter the food chain. This underscores the significance of closely monitoring and controlling the presence of dioxins in the food supply to safeguard the health of the population (Petrlik, 2023).

In 2016, reporting obligations to the IRZ were waived for small medical waste incinerators (MV ČR, 2016). However, data from previous years indicates that these incinerators were significant sources of dioxins in waste, a characteristic shared by small medical waste incinerators in general (Arar et al., 2019; Jelinek et al., 2023b; Khwaja and Petrlik 2006; Skalsky et al., 2006). See also the case study on small medical waste incinerators in chapter 3.5.4.

5.1.1.3.2 Data from the Reporting to the Stockholm Convention

The reporting of PCDD/F transfers in waste (emissions to residues) to the PRTR also relates to the ability of individual states to report the annual amount of PCDD/Fs ending up in waste each year, as evident from

32 This calculation is based on 0.25 pg TEQkg-1 of body weight per day as maximum tolerable daily intake (TDI), what means 6,387.5 pg TEQ and/or 6.3875 ng TEQ for 70 kg person per year.

Table 5.4: *Emissions of dioxins and furans (PCDD/Fs) to all vectors based on those reported to the EU and Stockholm Convention. (Source: European Commission, 2022b)*

Year	2015	2012	2013	2015	2015	2015	2012	2013	2015	2013	2014
Member State	AT	BE	HR	CZ	EE	FR	IE	NL	ES	SE	UK
Air	12%	99.8%	69%	12%	56%	89%	68%	84%	14%	14%	43%
Water	NR	0.2%	1%	NR	2%	11%	NR	16%	86%	NR	3%
Land	85%	NR	NR	29%	5%	NR	32%	NR	NR	NR	5%
Residue	NR	NR	27%	59%	15%	NR	NR	NR	NR	86%	32%
Product	2.4%	NR	2%	NR	22%	NR	NR	NR	NR	NR	17%

Table 5.4 on PCDD/F emissions to various environmental compartments. The data are derived from the EU's report on the implementation of the Stockholm Convention (European Commission, 2022b). National implementation plans (NIPs) of the Stockholm Convention represent another important source of information on the quantity of PCDD/Fs transferred in waste.

Austria (AT), Croatia (HR), Czechia (CZ), Estonia (EE), Ireland (IE), Sweden (SE), and the United Kingdom (UK) reported emission estimates for concentrations of PCDD/Fs within residues and/or land.[33] Broadly similar levels of emissions are quoted between air and land/residue in Austria, Czechia, and Sweden, with these estimates suggesting that residue is a much more significant emission vector than air (European Commission, 2022b).

The estimates quoted from the United Kingdom for the product vector, which in 2015 amounted to 145 g I-TEQ, relate to those waste materials from combustion processes re-used within the cements and aggregates industry (European Commission, 2022b).

Recent estimates of total PCDD/Fs releases globally were 101.4 kg TEQ year (Wang et al., 2016). It was also based on an earlier summary of PCDD/Fs national inventories (Fiedler, 2016). Out of 86 countries, 8 did not report about PCDD/Fs transfers in residues in these inventories, including Australia, Austria, Bulgaria, Finland, France, Germany, Russia and Switzerland (Fiedler, 2016).

In an effort to estimate the total amount of PCDD/Fs ending up in the ashes from incinerators globally, the outcome of which you will find in Chapter 4.2, we had to draw upon data on the amount disposed of in waste, as reported by various states in their NIPs. The result of this investigation is summarized in Table 5.5.

5.1.1.3.3 Deficiencies in Leaching Tests

Experiments involving rainwater washing have confirmed the leachability of dioxins from residues after waste incineration (Takeshita & Akimoto, 1991), further heightened by present surface-active substances (Sakai et al., 1997). Additional studies have shown that under conditions similar to those in landfills, the leachability of dioxins is higher (Kim & Lee, 2002) and is further increased by the presence of humic[34] substances. Also, using fir-extinguishing water leads to the release of dioxins from fly ash (Schramm et al., 1995).

33 Emissions to land stands mainly to releases from open burning or natural fires as explained in the EU report (European Commission, 2022b).

34 Humic substances are substances of natural origin, produced by the decomposition pf (mainly) plant residues.

Table 5.5. *PCDD/F transferred in wastes in g TEQ.y^{-1}.*
(Source: Petrlik & Bell, 2017), if not specified otherwise.

Country (source)	Argen-tina[I]	Brazil[II]	China[III]	Czechia[a)]	EU[b) IV]	Hungary[d) V]	India[e) VI]
HWI	27	20.72	186	20.7	61.8	11.53	3,965.8
MedWI	-	-	748.9		29.1	-	-
Year	2006	2014	2004	2015	2005	2006	2010
Country (source)	**Indo-nesia[f) VII]**	**Japan[VIII]**	**Kenya[g) IX]**	**Lithuania[X]**	**Nigeria[XI]**	**South Africa[XII]**	**USA[h) XIII]**
HWI	58	1,514	10.15	0.64	0	12.22	93 – 1,395
MedWI	-		-	0.5	15.851	-	NA
Year	2001	2018	2006	2004	2004	2012	2005

Notes:; a) Calculation based on 8 years reporting in PRTR, includes also MedWI; b) MedWI calculated for 10 EU member states only c) Industrial waste and sewage sludge incineration; EU + Switzerland and Norway; d) Calculated from data in Annex 6 (Ministry of Environment and Water, 2009); e) This figure is for all waste incineration plants in India (including MedWI), however there was only one WtE plant in operation in India with capacity 54,000 t.y^{-1} (Coenrady, 2013); f) Not very clear whether all comes from hazardous waste incinerators; g) Both HWI and MedWI h) Calculated by using Dioxin Toolkit (UNEP & Stockholm Convention, 2013); sources for individual countries data: I) (República Argentina, 2007); II) (Federative Republic of Brazil, 2015); III) (The People´s Republic of China, 2007); IV) (BiPRO, 2005); V) (Ministry of Environment and Water, 2009); VI) (Government of India, 2011); VII) (The Republic of Indonesia, 2008); VIII) (Government of Japan, 2020); IX) (EEC of SC, 2016; MENR, 2006); X) (MoE Republic of Lithuania, 2006); XI) (Federal Ministry of Environment, 2009); XII) (MWEA, 2012); XIII) (UNEP & Stockholm Convention, 2013; US EPA, 2016).

Photo 5.3: *When using a mixture of fly ash and bottom ash from waste incineration for surface adjustments on landfills, such as at the Větrov landfill near Frýdlant, Czech Republic (the mixture from WtE Termizo Liberec is used), the specific environmental conditions for leaching dioxins need to be taken into account. A drone photo taken in 2021: Marek Jehlička (skyworker.cz).*

Leaching tests are not sufficiently representative; their duration is too short, while the elements examined can remain mobile even after 6 years (Simon et al., 2021). Due to pH adjustment, they provide false results, making fly ash appear more stable than it actually is (Rollinson et al., 2022). Leaching tests do not account for substances that demonstrably influence environmental release such as acidic, anaerobic landfill environments or areas subject to acid rain or high tropical rainfall rates. Substances slightly soluble in water or substances attached to solid particles do not reach the aqueous leachate in corresponding concentrations causing measurement underestimations. Authorities usually only consider hazardous waste transferred to authorized personnel, but they do not concern themselves with how the waste is handled further down the chain of waste operators (Arnika, 2019a).

Photo 5.4: *Area of Jan Šverma Mine in Lampertice in the Podkrkonoší region, where fly ash was deposited along with other hazardous wastes. Photo from 2006, Jan Feřtek.*

5.1.1.3.4 Case Study - Jan Šverma Mine, Czech Republic

Under the Krkonoše Mountains, between the town of Žacléř and Lampertice, lies the oldest deep coal mine, Jan Šverma, which was closed in 1990. It is situated in a typical foothill area, traversed by the Lampertice stream, with a complex system of groundwater. According to experts from the GEMEC Union company (working on mine reclamation), water from the mine does not leak. However, locals who worked in the mine, do not believe this opinion and claim the situation is much more complex. It's common practice for these old mines to be filled with various materials to prevent surface landscape movement. According to records from environmental control authorities, residues from waste incineration were deposited in this mine up to 7,000 tons per year (Petrlik & Ryder, 2005). According to GEMEC Union, the technology used was safe, and leaching of toxic substances from materials stored in the mine did not reach groundwater. However, results of sediment tests from the Lampertice stream showed that at one location (under the discharge from the wastewater treatment plant in the mine area), the concentration of dioxins was ten times higher than the lowest detected value in the sampled area, which was detected upstream from the mine (specifically, the Lampertice stream "U Kirschů" inflow, which drains the southern part of the dump).

In the first half of 2004, Arnika association published the results of analyses of four trout samples from various locations in the Czech Republic, analyzed for various POPs. Among the analyzed substances, the trout from Lampertice showed the second-highest measured value of hexachlorobenzene in fish in the Czech Republic (462 $ng.g^{-1}$ fat); (Arnika, 2004b). This example shows that depositing a mixture of waste into a seemingly groundwater-isolated mine can lead to contamination of surface waters, specifically sediments or fish living in the affected area. Later sediment analyses also confirmed high concentrations of hexachlorobenzene in the vicinity of the mine (Arnika, 2011) and added PFAS to the list of toxic substances released from the mine (Lanková et al., 2011).

5.1.1.3.5 Case Study Newcastle

Between 1994 and 1999, 2,000 tons of a mixture of fly ash and bottom ash from the nearby municipal waste incineration plant in Byker were used on roads in Newcastle (Pless-Mulloli, 2003). Dioxin concentrations found in the fly ash ranged from 11 to 4,224 pg $I\text{-}TEQ.g^{-1}$ dry weight (Pless-Mulloli et al., 2001a). Watson (2001) even reported dioxin concentrations of up to 9,500 pg $I\text{-}TEQ.g^{-1}$ dry weight and stated that in this case, it was likely fly ash, not mixed with bottom ash.

Seventeen out of nineteen egg samples from areas where the fly ash was used, showed contamination levels significantly exceeding concentrations in eggs from chickens raised in poultry houses then purchased in supermarkets. The weighted average of all egg samples was 16.4 pg I-TEQ.g^{-1} fat. The weighted average of those samples exhibiting incineration plant-matching dioxin congener representation in egg samples was 22.2 pg I-TEQ.g^{-1} fat (Pless-Mulloli et al., 2001b) with a maximum of 56 pg I-TEQ.g^{-1} fat.

In Newcastle, waste with dioxin content less than a third of the LPCL limit for dioxins, as defined by the Basel Convention (15 ng TEQ.g^{-1} dry weight), was used in road reconstruction. Nevertheless, this resulted in contamination of poultry eggs, which on average exceeded the then-applicable EU limit for dioxins in eggs by 5.5 to 7 times. The soil in the investigated areas also exceeded limits for heavy metals, specifically arsenic, cadmium, copper, mercury, lead, and zinc (Pless-Mulloli et al., 2001b). Even though research in Newcastle, according to Watson (2001), did not include dl PCBs, it concluded that the consumption of eggs from domestic farms on fly and bottom ash-affected lands from the incineration plant, could have a significant impact on residents' health. Research conducted two years after the mixture was removed from the lands found substantial decreases in dioxin concentrations in eggs (Pless-Mulloli, 2003).

5.1.1.4 Wastewater

Generally, dioxin emissions from incinerators in wastewater range from 0.01 to 0.3 ng I-TEQ.L^{-1} (Stockholm Convention, 2008). According to BAT for waste incineration, BAT-AEL for both direct and indirect emissions to a receiving water body are 0.01-0.05 ng I-TEQ.L^{-1} (European Commission, 2019).

5.1.1.5 How Much Dioxin Does an Incinerator Break Down and Produce?

Persistent organic pollutants (POPs), including dioxins, enter the environment not only through the air but also in water or in waste (solid residues generated by waste incineration), which needs to be considered in their balances.

Photo 5.5: *A photograph from contemporary press captures the decontamination of the area affected by the use of fly and bottom ash from the Byker incineration plant in Newcastle. Photo: Public Interest Consultants Archive, UK.*

A frequently used argument from the municipal waste incinerator industry is to focus on the quantity of dioxin emissions released into the air while ignoring the amount of dioxins in solid residues after waste incineration. For instance, the director of WtE Malešice (Prague) in 2007 claimed: "The usual waste that comes to WtE contains around 50–60 ng of dioxins per kg of waste. Over the period 2000–2006, about 77 grams of dioxins were thus brought into the waste incinerator. The incinerator then emits 7 % of the total dioxin amount in the form of bottom ash. The concentration of dioxins in slag is 17 ng per kg of slag, which is lower than in some rocks, making it usable, for instance, in road construction. 81 % of dioxins remain in the fly ash, treated as hazardous waste, mixed with cement (i.e., solidified), and stored in hazardous waste landfills, never entering the free environment. During the waste incineration process itself, 11 % of dioxins vanish from the mass balance," (Mach, 2007). He used outdated data regarding the amount of dioxins in municipal waste, which could lead to the claim that the incinerator destroys dioxins rather than creates it.

A waste incinerator with a capacity of 300,000 tons of waste annually receives municipal waste with a dioxin concentration of 5 pg TEQ.g^{-1} waste. However, higher values are reported in the literature, for example, 50 pg TEQ.g^{-1}, which is a value derived from the composition of municipal waste in Germany in the 1980s (Wilken et al., 1992). However, this value does not correspond to the current situation. Another value - 37 pg TEQ.g^{-1} (BiPRO, 2005) is slightly lower but includes the previous data from Germany in the 1980s. Recent studies report values below 10 pg TEQ.g^{-1} (Abad et al., 2000) or 15 pg TEQ.g^{-1}. The Dioxin Toolkit (UNEP & Stockholm Convention, 2013) uses the value of 5 pg TEQ.g^{-1} (or 5 µg TEQ.t^{-1}), which we also chose for calculation as agreed upon by experts in the international panel of the Stockholm Convention representing governments, industries, and non-profit sectors. Therefore, for a model incinerator with a capacity of 300,000 tons of waste annually, we find 1.5 g TEQ in the waste per year.

The amount of dioxins emitted into the air by an incinerator of this capacity per year (for instance, WtE Malešice – Prague, with a similar capacity) is approximately 0.01 g TEQ year^{-1} (see Table 5.7). This value is based on short-term measurements and calculated according to operating hours and the amount of emissions. It does not represent a measurement over the entire year. During startup and shutdown of boilers, dioxin emissions can sometimes be as high as those in half a year of incinerator operation (Gass et al., 2002). However, measurements are not taken in such situations. To get closer to reality, we will continue using the value of 0.015 g TEQ.year^{-1}.

Photo 5.6: *Despite their unappealing appearance, approximately 1.5 g TEQ of dioxins enter an incinerator the size of the WtE Malešice (Prague) in municipal waste each year. Far more exits the incinerator.*
Photo: Jindřich Petrlík, Arnika.

Our hypothetical incinerator with a capacity of 300,000 tons of waste annually generates by incineration approximately 100,000 tons of solid residues, of which one-tenth is fly ash and the remaining nine-tenths is bottom ash. The concentration of dioxins in the fly ash ranges from 100 to 25,000 pg TEQ.g^{-1} according to the literature, while in WtE Malešice (Prague), equipped with relatively efficient filters, this value ranges from 300 to 2,200 pg TEQ.g^{-1} (Mach, 2017). If we use a concentration of 1,000 pg TEQ.g^{-1}, in one-tenth (in fly ash) of solid residues, we get 3 g TEQ.$year^{-1}$. Because approximately 10–30 % of all dioxins (Abad et al., 2000) produced in the incinerator end up in the bottom ash (on average, we'll consider 20 %), this means an additional 0.75 g TEQ of dioxins end up in the bottom ash annually. From Table 5.6, it is evident that we are dealing with the lower limit of the actual amount of dioxins transferred by the incinerator in waste.

Thus, 1.5 g TEQ enters the incinerator annually, while 0.015 g TEQ is released as air emissions, 3 g end up in the fly ash annually, and another

Table 5.6: *Amount of dioxins transferred in waste by municipal waste incinerators in the Czech Republic in g TEQ.year-1. (Source: Petrlik et al., 2018)*

Facility	2012	2013	2014	2015	2016	2017
WtE Pilsen	0	0	0	0	0.455	0.001
WtE Malešice (Prague)	13	8	11	4.56	5.7	26.75
WtE SAKO Brno	2.543	2.25	3.773	2.857	2.236	2.238
WtE TERMIZO Liberec	2.1	0	0	0	0	0
Total	17.64	10.25	14.77	7.42	8.39	28.99
Total hazardous waste incineration facilities	5.35	18.366	10.665	23.7	17.4	18.979

Photo 5.7: *The balance of dioxins in waste from hazardous waste incinerators is significantly impacted by the one located in Trmice in the Ústí nad Labem Region – this photograph shows a pile of medical waste awaiting incineration. Photo: Jindřich Petrlík. Arnika.*

0.75 g end up in the bottom ash annually. Overall, 3.765 g TEQ exit the incinerator annually. This is more than 2.5 times the amount of dioxins that entered the incinerator. As a result, a significant part (more than 99 % of dioxins) ends up in the solid residues of the incinerator, primarily in the fly ash. Therefore, a key piece of information is that an average waste incinerator does not destroy dioxins, although waste incinerators may present themselves as doing so (Info.cz, 2023; Mach, 2007; MHMP, 2013), but rather produces dioxins. Real data can be seen in Table 5.7.

5.1.1.6 Myths Associated with Dioxin Production in Incinerators

In Chapter 5.1.1.5, we debunked the argument that the incinerator "destroys" dioxins, as was written about WtE Malešice (Prague). Another argument encountered is that one incinerator produces in emissions as much dioxin as a whole community. Reports from the Czech Republic indicate that emissions of dioxins from domestic burning in one village are similar to emissions from a large incinerator (Horák & Hopan, 2009). As noted in Watson et al., 2012, *"These claims are rather misleading because emission factors relate only to emissions into the air, and in modern incinerators, much more dioxin is concentrated in residues from gas cleaning than is emitted into the air. Residues from waste incineration are often deposited in areas where sufficient environmental protection against the release of toxic substances from this material is not ensured. The Stockholm Convention on POPs relates to emissions into all components of the environment (including water or waste), and it is therefore important for its approach to be reflected in practice in the design of certain technologies and the creation of specific procedures aimed at reducing these emissions. If this problem is not understood correctly, it will likely lead to a unilateral focus on reducing air emissions, while other equally significant or even more significant flows of POPs will remain unresolved. At the same time, it will fail to focus on the elimination of precursors to the formation of these substances."*

Table 5.7: *Amount of dioxins transferred in waste for the year 2021 and released into the air at four Czech hazardous waste incinerators. Sources: Ministry of the Environment of the Czech Republic (2022) and ČHMÚ (2021)*

	Transferred in waste [g]	Air emissions [g]
WtE Malešice (Prague)	1.28	0.010
WtE SAKO Brno	4.845	0.0072
WtE Chotíkov	-	0.0028
WtE Termizo Liberec	2.61	0.0063

Note: WtE Chotíkov did not report any dioxins transferred in waste for the year 2021, but it was not verified whether it should have done so.

This argument is also discussed in Watson's commentary (2012). According to him, the reality is that three Czech waste-to-energy facilities (at that time, excluding WtE Chotíkov) produce as much dioxin as 120 to 270 communities with a total population of 76,000 to 176,000.

It is also not true that fireworks are a larger producer of dioxins than waste incinerators. It is a recurring, albeit erroneous, claim based on a scientific article published in 1999 (Lee et al., 1999), which concluded that millennial fireworks and bonfires produced 30 grams of dioxins. Over time, attention shifted away from campfires, although they were already mentioned in the aforementioned article as more significant source of dioxins than fireworks. However, emissions of dioxins from fires have been overestimated over time, and the UK National Atmospheric Emissions Inventory estimated emissions from all such fires in 2003 at 6.79 grams TEQ. An English expert on POPs who extensively investigated this case (Watson, 2009), for example, calculated that *"Incinerators would therefore produce at least 40 times more dioxins than campfires. It is important, of course, to emphasize 95 % plus of dioxins are in residues."*

Another argument is that a hundred years of operation of one waste incinerator corresponds to one landfill fire. By comparing annual dioxin emissions in outputs from the incinerator, we find that during one fire (in a landfill), 300 μg $TEQ.t^{-1}$ are released into the air and 10 μg $TEQ.t^{-1}$ of dioxins into the soil. When compared to the release of 73.355 μg $I\text{-}TEQ.t^{-1}$, one fire cannot correspond to a hundred years of operation of a waste incinerator.

In the appendix to the EIA documentation for the incinerator in Mělník (Czech republic), ČEZ Group, claimed that the impact of incinerators on the environment is negligible. Their theory was based on the assumption that the total dioxin emissions into the air in the Czech Republic from all sources were 740 grams TEQ. However, the actual total emissions are 25 times lower. The value of national dioxin emissions for the year 2016 was 29.26 grams TEQ (Arnika & Ekozahrada pod věží, 2018). Furthermore, ČEZ claimed that 82.5 % of dioxins are produced by natural processes, such as forest fires. However, if all forests in the Czech Republic were to burn, the emissions of dioxins would account for roughly three-fifths of the 740 grams TEQ (Arnika & Ekozahrada pod věží, 2018).

Zhang et al. (2012) also calculated the dioxin mass balance for MSWI (WtE plant). Their result demonstrated that the annual dioxin input value was around 5.38 g I-TEQ.y^{-1}, lower than the total output value (7.62 g I-TEQ.y^{-1}), signifying a positive dioxin balance of about 2.25 g I-TEQ.y^{-1}.

5.1.2 Brominated Dioxins (PBDD/F)

Polybrominated dibenzo-p-dioxins and dibenzofurans (PBDD/F), commonly referred to as brominated dioxins, exhibit similar properties to chlorinated dioxins (PCDD/F) (WHO, 1998). They are toxic to the immune system and the thyroid gland and are teratogenic (van den Berg et al., 2013). Some studies have also demonstrated a negative impact on intelligence (reduction), concentration ability (reduction), and behavior (hyperactivity in children). Negative effects have been proven on the thymus, liver, and body weight (van den Berg et al.,

Similar to chlorinated dioxins, brominated ones are produced as unintended byproducts in chemical processes, such as the manufacture of brominated flame retardants. This means they already enter incinerators in certain quantities in waste, but incinerating (even municipal) waste containing bromine and its compounds leads to the formation of additional brominated dioxins (Soderstrom & Marklund, 2002). "In addition to brominated dioxins, polyhalogenated dioxins (PXDD/Fs)35 - especially those combining both bromine and chlorine - can also form as unintentional byproducts. This group, along with brominated dioxins, has been proposed for addition to the Annex C to the Stockholm Convention list (POP RC, 2024).

***Photo 5.8:** Eggs from domestic poultry in the vicinity of the municipal waste incinerator in Wuhan, China, contained high concentrations of brominated dioxins.[35] Photo: Jindřich Petrlík, Arnika.*

35 There are 1550 brominated and chlorinated dibenzo-p-dioxin (PBCDD) and 3050 brominated and chlorinated dibenzofuran (PBCDF) congeners (POP RC, 2024; Yang et al., 2021). Limited historical data on the occurrence of PBDD/Fs and PBCDD/Fs have been recorded, most of which confirm emissions from incineration processes containing chlorine and bromine in the raw materials (Wang et al., 2023)

Photo 5.9: *Ash from municipal waste incinerators in southern Taiwan deposited near reservoirs used for fish farming and other food sources contains high concentrations of brominated dioxins. Photo: Jindřich Petrlík, Arnika.*

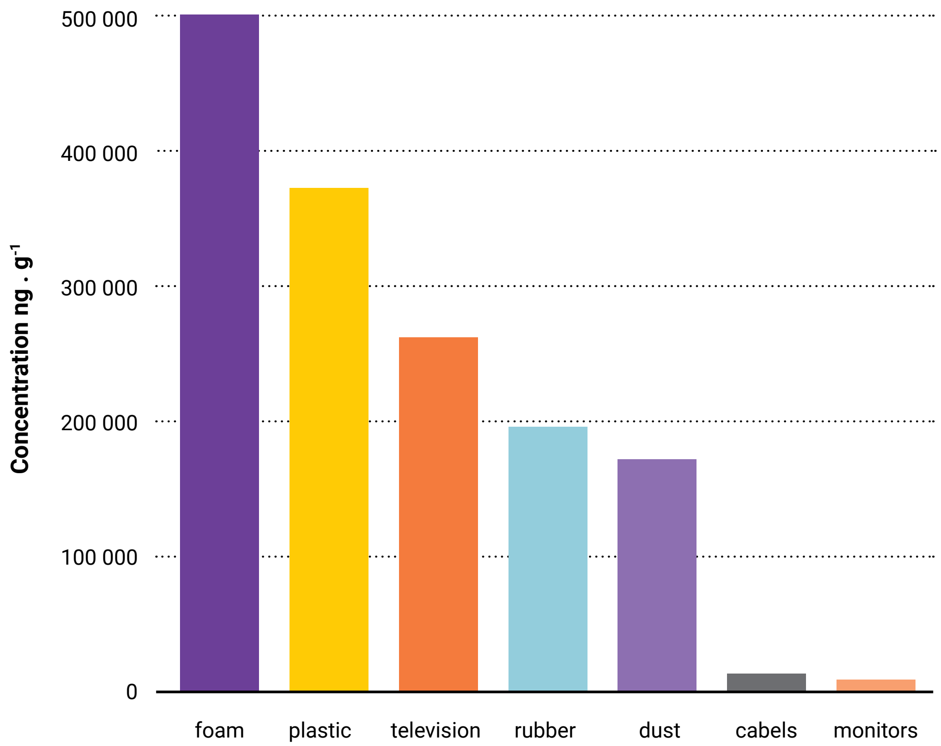
Photo 5.10: *Toys (mostly made of black plastic) contain high concentrations of brominated dioxins (Behnisch et al., 2023) and PBDEs (Møller et al., 2021) contributing to their formation during waste incineration. Photo: ALHEM, Serbia.*

Brominated dioxins are present in gaseous emissions from incinerators, as well as in fly ash, bottom ash, and other residues from gas cleaning (Chatkittikunwong & Creaser, 1994; Wang et al., 2010a). They have been detected in the air (M.-S. Wang, S.-J. Chen, K.-L. Huang et al., 2010a), soil (Song et al., 2022), and even in eggs from domestic poultry near incinerators and sites where fly ash and bottom ash from incinerators are handled (Teebthaisong et al., 2021; Weber et al., 2015). Unlike chlorinated dioxins, they tend to accumulate more in bottom ash than in fly ash (Bell et al., 2023a; Wang et al., 2010a; Wang et al., 2009). This also applies to brominated flame retardants, which do not completely decompose during incineration (Wang et al., 2010a; Wang et al., 2009; M.-S. Wang, S.-J. Chen, Y.-C. Lai et al., 2010b). Brominated dioxins significantly contribute to the overall dioxin toxicity in collected samples of eggs from domestic poultry (Petrlik et al., 2021). In China, there is a case of a municipal waste incinerator that was a source of contamination in its vicinity with brominated dioxins (Petrlik, 2015; Weber et al., 2015).

Relatively high concentrations of PBDD/F were found in samples of bottom ash from municipal waste incinerators illegally deposited near reservoirs along Ancing Road in southern Taiwan, resulting in high concentrations of brominated dioxins in the sediment of these reservoirs (Bell et al., 2023a).

European legislation mandates the measurement of PBDD/F in air emissions every six months in facilities incinerating waste containing brominated flame retardants or in facilities using continuous bromine injection (Evropská komise, 2019). The first of these conditions is essentially met by almost every waste incinerator because brominated flame retardants are found in a wide range of products ending up in both municipal and hazardous waste, including bulky furniture waste or toys and other items made from recycled plastics (DiGangi et al., 2011; Straková & Petrlík,

Figure 5.5: *Concentrations of PBDE in various materials from scrapyards and electronic waste. (Source: Nagyová, 2012).*



2017). Concentrations of PBDE in scrapyard waste and in electronic waste are summarized in the graph in Figure 5.5. Although this group of BFRs has been banned and a large part is estimated to have ended up in waste, with a peak in 2011 (Abbasi et al., 2019), their replacements are mostly brominated compounds. Based on the materials presented in the graph, at least part of them can be expected in mixed municipal waste.

Incinerators produce a much wider range of dioxin substances than just PCDD/Fs, dl PCBs or PBDD/Fs. Song et al. (2019) monitored PCDD/Fs, PBDD/Fs, and combined polybrominated/chlorinated dibenzo-p-dioxins and dibenzofurans (PBCDD/Fs) in emissions from waste incinerators and metallurgical plants in China. The contributions of PBDD/Fs and PBCDD/Fs to the total concentrations exceeded that of PCDD/Fs in some cases, such as in HWIs and secondary copper smelter (Song et al., 2019).

5.1.3 Polychlorinated Biphenyls (PCBs)

Polychlorinated biphenyls (PCBs) represent a group of substances that were both intentionally manufactured and continue to be generated as unintended byproducts in similar processes to dioxin formation, including waste incineration. This group comprises 209 congeners (blue-growth.org, 2018; Stockholm Convention, 2019).

PCBs were produced by numerous manufacturers throughout the world. The prominent producers include Monsanto (USA), Kanegafuchi Chemical (Japan) and Bayer Leverkusen (Germany); (Kimbrough and Jensen, 2012; Lang, 1992). In 1984 the total cumulative world's production was estimated to 1.2 million t, in Czechoslovakia 18,900 t were produced in Chemko Strážske (Slovakia) and production ceased in 1984. PCBs have been used for more than 60 years. In 1966 PCBs were first identified as pollutants (Lang, 1992). Japan ceased PCBs production in 1972, the first among countries producing PCBs. Most sources suggest that production ended in 1993 (in Russian Federation, former Soviet Union respectively); (UNEP Chemicals and Waste Branch, 2016).[36]

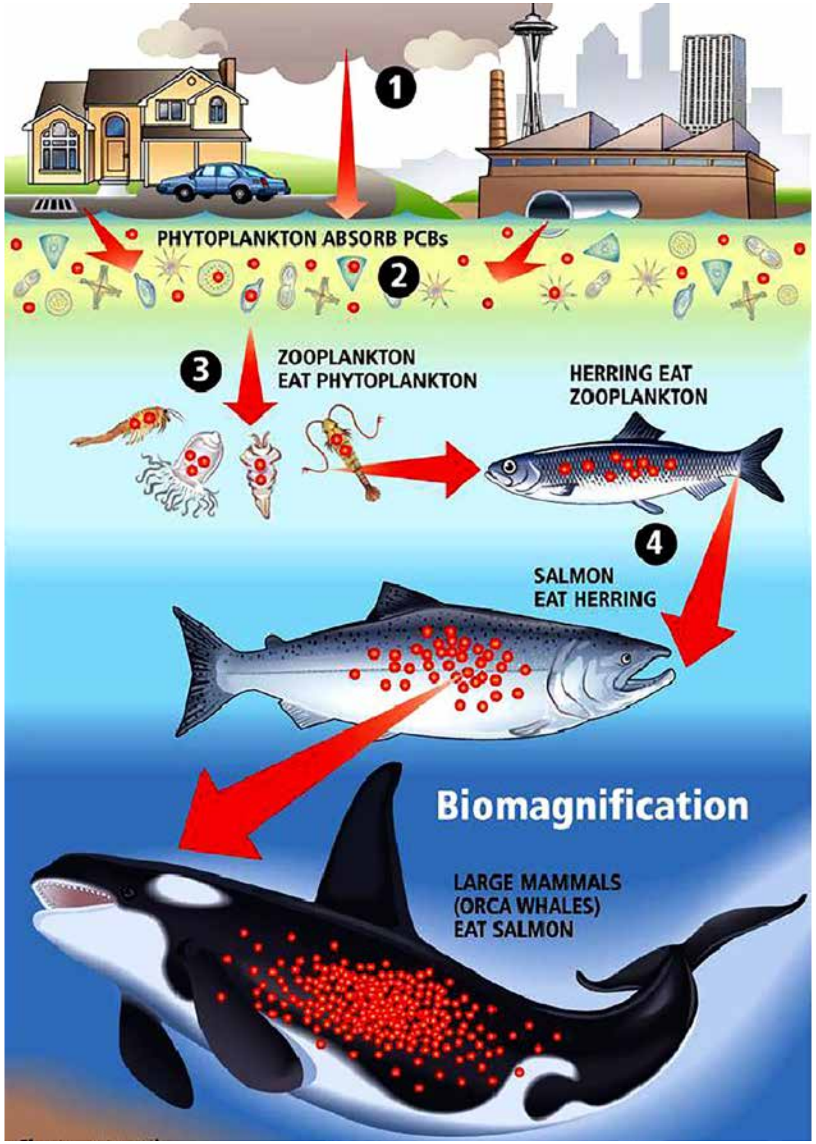
While few countries produced PCB, a number of countries imported PCB to produce transformers and capacitors or other liquids and equipment. It has been estimated that 48 % of the PCB production was used for transformer oil, 21 % for small capacitors, 10 % for other closed systems; 21 % open uses (caulking, paints etc). Thus, transformers usually represent the single largest source of PCB. More generally, electrical equipment can be considered as the main destination for PCB (UNEP Chemicals and Waste Branch, 2016).

PCBs production was halted, but they persist in various facilities, waste, contaminated buildings, and soils, thus polluting different environmental components, including fish, crabs, mussels, poultry and cattle meat, cow and camel milk (Amutova et al., 2021; Asante et al., 2010; Grechko et al., 2021a; Grechko et al., 2021b; Konuspayeva et al., 2011; Li et al., 2021; Mach and Petrlík, 2016; Mach et al., 2016; Malisch and Kotz, 2014; Ruus et al., 2006; Schaum et al., 2003; Weber et al., 2018). Even today, they can be found, for instance, in the form of old coatings on metal structures (ČIŽP, 2017) or even in applications such as paints and plasters inside houses (Grontmij/COWI, 2013; Ruus et al., 2006).

PCBs are toxic to fish, killing them at higher doses and causing spawning failures at lower doses. Research also links PCBs to reproductive failure (He et al., 2021) and suppression of the immune system in various wild animals, such as seals and mink (Stockholm Convention, 2019).

36 However, according to its NIP, the Democratic People's Republic of Korea (DPRK) continued producing PCB at least until 2006 (DPRK 2008; UNEP Chemicals and Waste Branch 2016).

Figure 5.6: *PCBs biomagnification in the marine food chain. Similar process of biomagnification and bioaccumulation in foodweb can be seen in terrestrial ecosystems. (Source: blue-growth.org, 2018)*



PCBs are among the most significant toxic pollutants in many countries and they were on the list of initial 12 POPs banned and regulated under the Stockholm Convention (Stockholm Convention, 2019). They damage the immune, hormonal, and reproductive systems (Carlson et al., 2023). Since 2016, they have been classified as proven human carcinogens (Group 1 according to IARC) (IARC, 2023). They are primarily associated with liver cancer. They cause delayed development in children and also negatively impact thyroid function.[37]

PCB-containing waste is incinerated in hazardous waste incinerators (e.g., in Ostrava, Czech Republic, Swan Hills, Canada or in Trédi WtE plant in Salaise-sur-Sanne, France) and cement kilns at high temperatures, although this poses a risk of dioxin formation. PCB congeners in emissions from incinerators are not commonly monitored.

Polychlorinated biphenyls (PCBs) can be found in higher concentrations in both fly ash (Arp et al., 2020; Ramesh Kumar et al., 2021; Shen et al., 2010) and bottom ash from waste incineration (Sakai et al., 2007; M.-S. Wang, S.-J. Chen, Y.-C. Lai, et al., 2010b). In monitoring at WtE SAKO Brno (Czech Republic) from 2004, most PCBs were accumulated in ash at up to 170 g annually (Bogdálek & Moskalík, 2008). From 80% to 99.9% of PCBs are bound to solid particles or dissolved organic carbon.

Key findings from the study conducted in Poland indicate a correlation between proximity to waste incinerators (MWI and industrial waste incinerators - IWI) and increased accumulation of PCBs in both soil and plants (Gabryszewska and Gworek, 2020). PCB congeners 52, 44, and 110 were prevalent in soils near MWI, while congeners 28, 52, and 110 dominated near IWI. Notably, PCBs tended to penetrate deeper soil layers near MWI, with the highest accumulation observed at 20-30 cm depth due to soil characteristics. Conversely, near IWI, PCBs remained predominantly in surface layers (0-5 cm). Wind direction had limited impact on PCB accumulation around MWI but influenced higher accumulation on the leeward side of IWI. Overall, the study highlights the significant influence

Photo 5.11 *In Boršice near Buchlovice, in the wine-growing region of southern Moravia, there were plans in the 1990s to build a hazardous waste incinerator for the disposal of meat and bone meal and other waste containing high concentrations of PCBs. Local people did not approve of this plan, and they made it clear during the visit of the then Prime Minister of the Czech government, Václav Klaus, in 1994.*
Photo: Zahrada Moravy, environmental association.

37 A recent review study focused on non-cancer PCB effects identified 637 mammalian toxicological studies evaluating endpoints in a variety of species exposed for different durations and at different life stages and 953 epidemiological studies conducted using diverse populations and methods (Carlson et al., 2023).

of proximity to waste incinerators on PCB levels in both soil and plants, with PCB accumulation in plants surpassing that in soils, particularly near MWI (Gabryszewska and Gworek, 2020).

PCBs leaked accidently from Swan Hills high temperature waste incinerator of hazardous waste in Alberta, Canada. More details about the case can be found in the following case study.

5.1.3.1 Case Study: Swan Hills – Incidents in POPs Waste Treatment Center

The Swan Hills Solid Waste Treatment Centre (SHSWTC) in Alberta, Canada, has been employing high-temperature waste incineration for the destruction of polychlorinated biphenyls (PCBs) since its inception in 1987 (Henton, 2015). Since it opened in 1987, the plant processed more than 295,000 metric tonnes of hazardous waste (Froese, 2021a). However, the facility's operational history has been marred by accidents, leading to the unintended release of PCBs and other toxic compounds into the environment (Guidotti, 2018).

At the time of the incident, SHSWTC was owned by the Province of Alberta and operated by Chem-Security, Ltd. The facility was sited on a relatively remote plateau in northern central Alberta in part to isolate it from population centers and in part because the community of Swan Hills invited it, as a means of economic diversification.

Central to the waste treatment process for organochlorines is a high-temperature incinerator complex designed to destroy organic materials contained in the liquid, solid and sludge waste received at the facility. This consisted, at the time, of a rotary kiln and two rocking kiln incinerators. Combustion byproducts are scrubbed to remove particulate matter and acidic gases prior to being discharged to the ambient air through the stack (Guidotti, 2018).

5.1.3.1.1 The incidents

The incident (actually two related incidents occurring close together) involved a breach in containment in the pyrolysis operation that vented through a defect in a heat exchange manifold and bypassed the scrubbing system.

On 16 October 1996, a malfunction in the incinerator at the SHSWTC resulted in a leak. The leak vented to the outside and released quantities of organochlorine compounds, including dioxins, furans and PCBs. The plant had by then treated 27 million kilo grams of hazardous waste, much of it contaminated with PCBs, during 1996 alone (Blais et al., 1998; Froese et al., 1998; Guidotti, 2018).

A health and risk assessment study was ordered by the government to determine the exposure to humans. That study, published in 1997, resulted

Figure 5.7: *The Swan Hills Treatment Centre is located about 250 kilometers north of Edmonton. Source: Google Earth*

Photo 5.12: *The Swan Hills Treatment Centre (SHSWTC) in 2021, a source of PCB and dioxin pollution in Alberta, Canada. (Source: Froese, 2021a).*

in advisories for consumption of wild game and fish taken within a 30 km radius of the treatment center.

While that study was still underway, an explosion occurred on July 21, 1997. Although the company operating the facility at the time said there was "minimal" chance of contamination, PCB levels measured around the plant were as much as 14 times above average.

Since 2001, the province has owned the facility, with a succession of companies contracted to operate it — currently Veolia, a French multinational. In 2009, another explosion and fire caused the facility to shut down for 10 months (Guidotti, 2018).

5.1.3.1.2 PCBs, and PCDD/Fs Levels in Biota and People

Studies conducted following the 1996 incident revealed elevated levels of PCBs, and PCDD/Fs in both wildlife and fish samples from the vicinity of the SHSWTC (Gabos et al., 2012). These findings underscored the persistent environmental contamination associated with the facility's operations, prompting ongoing monitoring and remediation efforts (Henton, 2015).

PCB levels in both snow and sediments were studied in the vicinity of the SHSWTC in 1998. PCB accumulation rates increased from about 3 μg.m^2.year^{-1} in the late 1980s to a maximum of 82 μg.m^2.year^{-1} in 1997. There was also a smaller increase in PCB accumulation rates in sediments after 1993 which is the year that the processing of PCB wastes at the SHSWTC was increased (Blais et al., 1998).

Despite attempts to mitigate contamination, levels of these contaminants remained notably higher than in reference areas, fueling ongoing health and environmental concerns. Sum of PCB levels between 1999 and 2010 were decreased as compared to the levels in liver samples ($p < 0.001$) and muscle samples ($p < 0.01$) in 1997, but the levels were still significantly higher than those in deer collected from the reference area in 1999 (Gabos et al., 2012).

The levels of PCBs, dioxins and furans were significantly elevated in both liver and muscle samples of deer from the study area as compared to the reference areas. The patterns of PCB congener distribution were different for deer samples from study and control areas. The majority of dioxin-like TEQ value was due to dioxin and furan concentrations in the liver. One PCB congener, 126, accounted for 97% of total dioxin-like TEQ in muscle from deer in Swan Hills.

The levels of PCBs, dioxins and furans were significantly elevated in the liver and muscle samples of fish taken from the lake nearest and downwind of the SHSWTC, specifically Chrystina Lake (immediately downwind); (Gabos et al., 1998).

Under normal circumstances, northern pike, a predator, would be expected to have higher contaminant concentrations than brook trout which feed on planktonic invertebrates. The lower contaminant values in pike from Roche Lake (933 km east of Swan Hills) and Chip (reference) Lake indicate very low contaminant background (Gabos et al., 2012).

Human Populations: No statistically significant difference in human serum levels of PCBs, PCDDs, and PCDFs based on current detection limits was observed for the 65 residents living within a 100 km radius of the SHSWTC compared to subjects living in an urban reference area. There was no statistically significant difference between local wild game and fish consumers and non-consumers. Different PCB congener patterns were observed in a small number of the workers at the SHSWTC (Guidotti, 2018).

5.1.3.1.3 Economic Considerations

For some residents in the area, the Swan Hills Treatment Centre represents more than just an environmental issue—it's a vital economic lifeline. Alberta Infrastructure told CBC News that the plant currently has 47 employees and operates four days a week. However, despite its economic contributions, the treatment center has proven to be a financial burden for the provincial government. Originally established through a partnership with private entities, the facility's profitability was compromised when the projected market for hazardous waste disposal failed to materialize. Legislative changes further impacted its financial viability, such as the 1993 exemption for hazardous oilfield wastes from mandatory disposal at Swan Hills (Lambert, 2024).

The infrastructure ministry noted in its 2022-23 annual report that a new policy to charge Alberta Health Services for processing waste contributed a "significant portion" of the $14.7 million in revenue generated by the facility that year.

That revenue, however, is outweighed by the centre's roughly $30 million in annual operating expenses. The high costs of operation and lack of significant revenue streams has weighed for decades on the province, which has spent hundreds of millions of dollars on the plant since its inception (Lambert, 2024).

What will remain, though, is the contaminated site of the Swan Hills Treatment Centre. Post-closure cleanup is a liability that has grown in estimates over the decades, from $20 million in 2000 to current estimate at $223 million. The economic strain on the province has been significant. Despite attempts to mitigate costs, the financial outlook remains bleak (Froese, 2021a; Lambert, 2024).

5.1.3.1.4 Sociological and Health Impacts

However, the economic benefits of the Swan Hills Treatment Centre are contrasted with significant sociological and health concerns. For many, particularly environmental advocates like Julie Asterisk, who worked organizations such as Keepers of the Water, the facility symbolizes an ongoing environmental hazard. With a track record of multiple unplanned releases of hazardous chemicals, including PCBs and dioxins, the center has raised serious health concerns for nearby communities. Health advisories, implemented following accidents in 1996 and 1997, underscore the ongoing risks to human health, particularly regarding the consumption of wild game and fish.

In 2015, Keepers of the Water filed a statement of concern objecting to the renewal of the SHTC's operating permit on the grounds of environmental contamination (Lambert, 2024). But Sucker Creek First Nation Chief Jim Badger said "Thursday the plant has contaminated the forest and poisoned traditional lands" (Henton, 2015). When the permit was renewed in 2019 despite those concerns, Keepers filed an appeal, which is still pending, and substantive issues regarding environmental contamination remain unaddressed. The impending closure of the facility, while a relief to some, does not erase the lingering questions about its impact on human health and the environment (Lambert, 2024).

As closure approaches, there is a pressing need for further investigation and transparency from the province to ensure the well-being of affected communities. "There's way more people with concerns than people who think it's great," says Jule Asterisk, an environmental advocate and longtime resident of the area to CBC News. Additionally, the remediation

Photo 5.13: *Jule Asterisk is an environmentalist and longtime resident of Slave Lake which is close to Swan Hills treatment center. She has worked with Keepers of the Water for many years. (Source: Lambert, 2024). Photo: Jule Asterisk's archive.*

Photo 5.14: *Indigenous chiefs and councillors walk during an honour song at a Healing Gathering in Driftpile on July 1. Leading the way are Driftpile Chief Dwayne Laboucan, left, and Sucker Creek Chief Jim Badger. (Source: Froese, 2021b).*

costs, estimated at $220 million, pose a significant economic challenge for the province, further highlighting the complex interplay between environmental, economic, and health considerations (Lambert, 2024).

In summary, the history of the SHSWTC highlights the intricate interplay between environmental protection, public health, and economic development.

5.1.4 Dioxin-Like Polychlorinated Biphenyls (dl PCB)

A few of the polychlorinated biphenyls possesses properties similar to dioxins, these 12 congeners are often measured together with chlorinated dioxins as dl PCB and expressed in toxic equivalents (TEQ) (van den Berg et al., 2006). This means they have similar effects on human health. In 2016, similar to intentionally manufactured PCBs, they were classified as proven carcinogens (Group 1 according to IARC) (IARC, 2023).

Generally, it is assumed that dl PCBs are formed by the same mechanisms as PCDD/Fs (Jansson et al., 2011; Lemieux et al., 2001) and can be expected in the same materials. Together with dioxins, they are more commonly found in fly ash than in bottom ash from waste incinerators (Bell et al., 2023a; Pan et al., 2013; Pekarek et al., 2001; Soong & Ling, 1996).

A study focused on PCB flow in waste incinerators concluded that dioxin-like PCBs in the input waste are destroyed during incineration and others are newly formed in the post combustion zone. The dl PCB fingerprints of boiler and fly ash from full scale waste incinerators correspond well to the fly ash fingerprint obtained in lab scale de novo synthesis experiments (Van Caneghem et al., 2014). This confirmed the same mechanisms as for PCDD/Fs. Creation of dl PCBs was also confirmed in flue gases from WI in Korea. The total TEQ concentrations of PCBs, calculated using WHO-TEF values, varied from 0.001 to 0.55 ng-TEQ.Nm^{-3} and from 0.001 to 8.29 ng-TEQ.Nm^{-3} in the industrial waste incinerators and municipal solid waste incinerators, respectively (Shin et al., 2006).

Dioxin-like PCBs contributed significantly to total TEQs in biological samples, including free range chicken eggs collected from the vicinity of municipal waste incinerators located in Lithuania, Czech Republic, Spain and France (Arkenbout and Bouman, 2021a; Arkenbout and Bouman, 2021b).

Czech operators and investors in incinerators from the Teplárenské sdružení (Association for the District Heating of the Czech Republic) ordered an assessment criticizing the claims of a study conducted by ToxicoWatch for Zero Waste Europe (Arkenbout and Bouman, 2021a) suggesting they are not based on any relevant evidence confirming that the WtE plant in Chotíkov (Pilsen) would be a source of contamination of eggs with POPs. This claim was supported primarily by the absence of measurements of POPs in the soils of the investigated sites (Holoubek, 2022). However, the discussion of the assessment published on Arnika’s website revealed that: *“The assessment does not provide any convincing reasons that would rule out a possible connection between WtE Chotíkov (Czech Republic) and POP contamination in eggs. However, the ToxicoWatch study points out some important aspects of emissions of pollutants in waste incinerator exhaust gases and WtE (e.g., possible increased emissions into the air during conditions other than normal operating conditions, emissions of pollutants without defined emission limits), which in any case deserve attention and further investigation”* (Dvorska, 2023a).

Nevertheless, the health authority for the Ile-de-France region, the Agence Régionale de Santé (ARS), has taken a different stance and has validated the results presented by the ToxicoWatch study (Arkenbout and Bouman, 2021b) through a new study conducted at 25 sites, including 14 in proximity to the three main waste incinerators in the region. The ARS has issued warnings to citizens regarding the consumption of free-range chicken eggs. Out of the 25 sites analyzed, 21 samples were found to exceed regulatory thresholds for dioxins, furans, and PCBs, and two were found to have particularly high levels of PCBs in eggs—up to 40-50 times the EU threshold (Rickerby, 2023; Southey, 2023).

5.1.5 Hexachlorobenzene (HCB), Pentachlorobenzene (PeCB), and Hexachlorobutadiene (HCBD)

Pentachlorobenzene (PeCB) and hexachlorobenzene (HCB) are primarily produced unintentionally during incineration, as well as in thermal and industrial processes. They are also formed as byproducts in the production of various chlorinated hydrocarbons or pesticides. They were previously intentionally manufactured as pesticides or technical substances (POP RC, 2008).

In high doses, HCB is lethal to some animals and adversely affects their reproduction at lower levels. Scientists have also found that, similar to other organochlorine compounds, HCB can pass through the placenta (Sala et al., 2001). Besides causing cancer, studies by Reed et al. (2007) revealed that HCB's effects on human health from exposure involve systemic damage to human organs (thyroid gland, liver, kidneys, bones, skin), blood cells, as well as the immune and endocrine systems. It also has teratogenic effects and disrupts the nervous system. PeCB is moderately toxic to humans, highly toxic to aquatic organisms, and can cause long-term adverse effects in the aquatic environment (POP RC, 2007b).

Hexachlorobutadiene (HCBD) is a byproduct in the production of the same chlorinated hydrocarbons as PeCB and HCB. It is also formed unintentionally in combustion processes of acetylene and chlorine. HCBD is highly toxic to aquatic organisms, causing kidney damage and cancer in animal studies and chromosomal aberrations in people exposed to it occupationally (Balmer et al., 2019; POP RC, 2012a). HCBD is toxic upon repeated or chronic exposure, even at low exposure levels (i.e., 0.2 $mg.kg^{-1}$). The target organ for toxicity is the kidneys; its biotransformation into reactive compounds leads to organ toxicity, genotoxicity, and carcinogenicity due to lifelong dietary exposure (POP RC, 2012a).

All three substances are generated as byproducts during waste incineration, although HCBD to a far lesser extent than HCB and PeCB. In the hazardous waste incinerator in Ostrava, Czech Republic, waste named 'hexa-residue' from the production of chlorinated solvents from chemical plant in Ústí nad Labem, Spolchemie, has been incinerated while

Photo 5.15: *High concentrations of hexachlorobenzene also appeared in eggs from the vicinity of the hazardous waste incinerator in Lysá nad Labem. The incinerator, among other issues, struggled with where to store received hazardous waste, often leaving it loosely stored in unsecured areas within its premises. The photograph depicts the situation from 2002. Photo: Jindřich Petrlík, Arnika.*

Table 5.8: *HCB concentrations in eggs from various locations around waste incinerators.*

Location	Type of Incinerator	Concentration [ng. g^{-1} fat]	Sampling Year	Source of Information
Wuhan (CN)	MSW	481 and 28.9	2014	(Petrlik, 2016)
Liberec (CZ)	MSW	250	2005	(DiGangi & Petrlik, 2005)
Lysá nad Labem (CZ)	HW	46.2	2005	(Skalsky et al., 2006)
Benešov (CZ)	MW	14.9	2004	(Skalsky et al., 2006)
Košice (SK)	MSW	10.7	2005	(Hegyi et al., 2005)
Šala (Duslo); (SK)	HW	8.64	2006	(Petrlík, 2006)
Shetpe (KA)	HW (?)	6.29		(Petrlik et al., 2016)
Aguado (FI)	HW	1.9	2005	(Calonzo et al., 2005)
Aguado (FI)	HW	3.6 a 4.6	2019	(Jindrich Petrlik et al., 2021)
Lucknow (IN)	MW	3.8	2005	(Agarwal et al., 2005)
Ústí nad Labem (CZ)	HW°	35.8	2005	(Petrlik et al., 2005)
Coatzacoalcos (ME)	HW°	34.5	2005	(Bejarano et al., 2005)
Izmit (TR)	HW	5.30	2005	(Yarman et al., 2005)
Accra (GH)	MW	3.63	2018	(Hogarh et al., 2019)
Kumasi (GH)	MW	0.76	2018	(Hogarh et al., 2019)

Explanations: MSW – Municipal Solid Waste, HW – Hazardous Waste, MW – Medical Waste, ° – incinerator located in the area of a chemical plant that may significantly contribute to HCB contamination; (?) in Shetpe, it was uncertain whether the cement plant incinerated hazardous waste or not; CN – China, SK – Slovakia, KZ – Kazakhstan, PH – Philippines, MX – Mexico, TR – Turkey, GH – Ghana.

Photo 5.16: *Hexachlorobenzene appeared in significant quantities in eggs from the vicinity of another hazardous waste incinerator located near the hospital in Benešov. Photo: Jindřich Petrlík, Arnika.*

containing HCB, PeCB, and HCBD. Using non-incineration technologies for their decomposition would likely be a more environmentally friendly approach.

A comparison of HCB levels in free-range poultry eggs in Asian countries revealed that the highest value (481 $ng.g^{-1}$ fat) was found near a large municipal waste incinerator in Wuhan, China (Dvorska et al., 2023b). Similarly, from the vicinity of WtE Termizo Liberec (Czech Republic), a mixed egg sample with a high concentration of HCB at 250 $ng.g^{-1}$ fat was identified (DiGangi & Petrlik, 2005). Relatively high concentrations of HCB were also found in eggs near some hazardous waste incinerators (see Table 5.8).

HCB and/or PeCB were also found in fly ash from waste incinerators in the Czech Republic (Mach, 2017; Petrlik et al., 2007), Sweden (Lundin and Marklund, 2007), China (Yu et al., 2020) or Taiwan (Bell et al., 2023a). HCBD either was not detected in concentrations above the limit of quantification (Mach, 2017) or it was measured at low levels (Bell et al., 2023a) in them.

There are also lower chlorinated benzenes (CBzs). We included them under "other organic substances" (see Chapter 5.2).

5.1.5.1 Case Study: Wietersdorfer Cement Plant (Carinthia, Austria)

The Wietersdorfer Cement Plant in the alpine valley in Carinthia incinerated slaked lime containing hexachlorobenzene (HCB) between 2011 and 2014 (Holub, 2017; Kundi, 2015). When an increase in HCB levels in food from the valley was observed in 2014, blood (and breast milk) was collected from 120 local residents. It was found that in 21 residents, blood concentrations exceeded newly established Austrian reference values (corresponding to ten-year-old German reference levels). The highest concentrations were found in those who primarily consumed local (valley-produced) food.

Photo 5.17: *Wietersdorfer Cement Plant, which contaminated the Carinthian valley with hexachlorobenzene. Photo: Wikimedia Commons CC-BY-SA-3.0.*

The contamination of the environment reflected increased HCB concentrations in those residents living near the cement plant. This was associated with concentrations in the air (up to 5.1 $ng.m^{-3}$) and in spruce needles (30–50 $ng.g^{-1}$). Based on these findings, a direct pathway from the air to the soil (up to 0.8 $ng.g^{-1}$) and through livestock feed and human food (especially milk, meat) could well explain the variability of HCB concentrations in the blood (Kundi, 2015). The cement plant had EMAS[38] certification, which it lost after these findings (Holub, 2017).

38 The Environmental Management and Audit Scheme (EMAS) is one of the ways in which an organisation can proceed to implement and Environmental Management System (EMS). It can be defined as a part of organisation's overall management system, the aim of which is to integrate environmental protection requirements into the overall strategy of the organisation and its day-to-day activities.

Photo 5.18: *Čížkovice Cement Plant is part of the Lafarge group; the photo is from 2005. Photo: Miaow Miaow under Wikimedia Commons license.*

The entire case is noteworthy in that periodic measurements of PCDD/F emissions were conducted, always staying below the emission limit of 0.1 ng TEQ.m-³. HCB emissions were not measured, even though the cement plant was disposing of waste containing it. It is assumed that the high HCB emissions were caused by the introduction of contaminated slaked lime into the raw meal mill (Stockholm Convention on POPs, 2019).

In the Czech Republic, waste with high POPs content is often incinerated in cement plants. These cement plants do not have specific emission limits set for incinerated POPs, so they are not monitored in emissions. For example, when the Čížkovice cement plant incinerated sludge from Ostrava lagoons containing PCB, it was only required to monitor PCB in emissions once every three years (KÚÚK, 2011).

5.1.6 Polycyclic Aromatic Hydrocarbons (PAHs)

Polycyclic Aromatic Hydrocarbons are a group of more than a hundred substances consisting solely of carbon and hydrogen in the form of benzene rings. PAHs are a common component of the environment. Except in rare cases, they are not deliberately manufactured but are present in various industrial products such as oil or asphalt. They naturally occur during the combustion of any organic matter (transport, power plants, industry, cigarette smoke, etc.). Some PAHs have a high bioaccumulation potential. Humans can be exposed to them through inhalation, ingestion, or skin contact. Exposure leads to skin and eye irritation. They damage kidney and liver tissues (ATSDR, 1995; Havel & Válek, 2010). Several PAHs are classified as known or probable human carcinogens (lung, digestive tract, or skin cancer); (IARC, 2023). In animals, negative effects on reproduction and offspring development have been observed. Among the most toxic is generally considered to be benzo[a]pyrene (BaP), which IARC has classified as a Group 1 known human carcinogen (IARC, 2023). In addition to non-halogenated PAHs, increased attention has recently been paid to halogenated (chlorinated or brominated) PAHs, which are generated, among other sources, by incinerating halogenated waste (Altarawneh & Altarawneh, 2022; Wang et al., 2003), some of which are considered more toxic than non-halogenated PAHs (Ohura, 2007) or dioxins (Jin et al., 2020).

Polycyclic aromatic hydrocarbons have been found in various concentrations in fly ash (Alawi & Al-Mikhi, 2016; Mininni et al., 2007; Till et al., 1997) as well as in bottom ash (Shen et al., 2010; Zhao et al., 2010) from waste incinerators. In monitoring at the WtE SAKO Brno (Czech Republic) in 2004, most PAHs were accumulated in fly ash in quantities exceeding 6.5 kg annually (Bogdálek & Moskalík, 2008), also see the graph in Figure 3.4.

Although PAHs are not among the commonly measured substances in air emissions, they are emitted by incinerators into the air (Hsu et al., 2021; Liu et al., 2010a; Mininni et al., 2007; Petrlík et al., 2007). France has set a limit for their content in bottom ash/slag, if used in surface engineering construction at 50 $mg.kg^{-1}$ (French Republic, 2011). In the Czech Republic, the limit for similar use stands at 1 $mg.kg^{-1}$ dry matter, but only for the sum of concentrations of four PAHs (Σ benzo[a]pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, indeno(1,2,3-cd)pyrene); (Ministry of the Environment of the Czech Republic, 2021e). Elevated concentrations of PAHs were found in sediments around a plant that processed (among other things) fly ash from waste incinerators into a waste mixture prepared for the reclamation of lagoons after uranium ore treatment in Mydlovary (Mach, 2017), although other waste could also have been their source. PAHs are also products of waste gasification, which pollute the final product (Rollinson, 2018).

Ranzi et al. (2013) found a dose-response trend for urinary and serum heavy metals and PAH in their study of 65 people living near or working in an incinerator and with 103 controls. Oh et al. (2005) found urinary PAH metabolites were 15 and 3.5 times higher in incineration workers compared to the controls ($p<0.05$).

5.1.7 Brominated Flame Retardants

Brominated Flame Retardants (BFRs) are an integral part of waste (Van Caneghem et al., 2010). They are added to potentially flammable materials to prevent or at least slow down their combustion. They are commonly found in plastics, textiles, or electrical equipment. In plastics, their concentration ranges between 1 to 15 % (Hennebert, 2020), reaching up to 33 % in extreme cases (Alaee et al., 2003). Despite the current ban on certain flame retardants in the European Union, it can be assumed that eventually all of them, including those already or potentially banned in the future, will appear in waste. Brominated flame retardants include PBDEs, HBCD, PBB, brominated bisphenols, and many others, including so-called new/alternative flame retardants (nBFRs) as described in Chapter 5.1.7.2. We will focus mainly on those that are important in terms of toxic substance flows during waste incineration or are more closely monitored.

5.1.7.1 Polybrominated Diphenyl Ethers (PBDE)

Polybrominated diphenyl ethers (PBDEs) belong to a group of brominated flame retardants that have been gradually added to the list in the Stockholm Convention for global elimination. PBDEs are additives mixed into plastic polymers and are not chemically bound to the material, hence they are released into the environment. They have adverse effects on reproductive health, as well as developmental and neurotoxic effects (POP RC, 2006, 2007a, 2014). DecaBDE and its degradation products can act as endocrine-disrupting chemicals (POP RC, 2014). PentaBDE is used in polyurethane foam for automotive upholstery and furniture, while Octa- and DecaBDE are mainly used in plastic covers for electronics. Example of the exposure pathways can be seen at diagram from study focused on workers from e-waste site in Kalasin Province, Thailand (see Figure 5.8).

When PBDEs are incinerated, they do not undergo complete destruction but instead give rise to PBDD/F (Weidlich, 2021), similar to the processes of PCDD/F formation during the incineration of chlorinated substances. Hence, we partially discuss the presence of BFRs in waste in the chapter dedicated to brominated dioxins (5.1.2), which also includes Figure 5.5 showing PBDE concentrations in some materials from scrapyards and electronic waste.

Among all residues, PBDEs concentrate most in bottom ash, ranging from 29 to 243 $ng.g^{-1}$, which is two orders of magnitude higher than in common rural or urban soil (Lin et al., 2014). There is no standard against which this concentration can be compared; in construction materials made from ash, as PBDE or PBDD/F content is not monitored, and there is no limit for PBDD/F in residues for waste incineration (Rollinson et al., 2022). According to Morin et al. (2017), the quantity of brominated flame retardants in bottom

Figure 5.8: *Median levels of the sum of PBDEs in blood serum in Thailand, Nonthaburi of e-waste workers and in various environmental compartments and foodstuffs of the Khok Sa-ad e-waste recycling area compared to background concentrations. Similar or higher levels of PBDEs as in Kalasin were also observed after incineration of plastic waste in Tropodo, Indonesia in both ash residues and free range eggs (Ismawati et al., 2021; Petrlik et al., 2020). (Source: Dvorska et al., 2023b)*

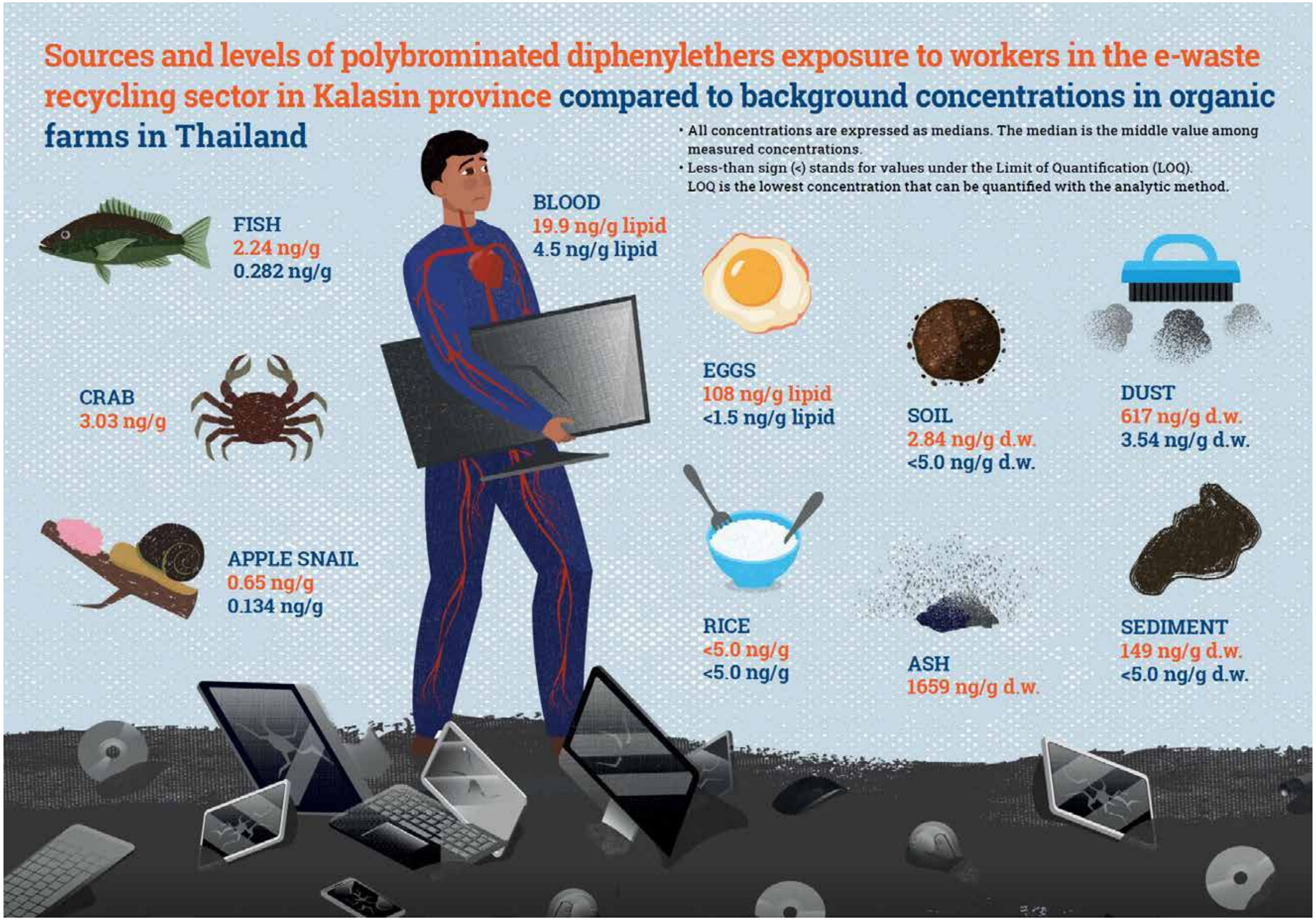


Photo 5.19: *PBDEs were found in a sample of SPRUK mixture from WtE Termizo Liberec (Czech Republic); Petrlik, 2006. The sample came from this heap near the Větrov landfill in Frýdlantsko. Photo: Marek Jehlička, Arnika.*

ash cannot be considered negligible and should be considered in landfilling bottom ash from waste incinerators or its use as fillers in engineering networks. Lin et al. (2014) recommend the precautionary principle be applied when using bottom ash as a building material or outright state that PBDE and PBDD/F concentrations in bottom ash are such that their use would lead to environmental contamination (Wang et al., 2010b).

High concentrations of PBDE (106.8 $ng.g^{-1}$ fat) were found in eggs from domestic poultry near a hazardous waste incinerator in Izmit, Turkey (Blake, 2005). Slightly lower concentrations (33.6 $ng.g^{-1}$ fat) were found in the vicinity of a hazardous waste incinerator in Aguadu, Philippines. Values exceeding 1,000 $ng.g^{-1}$ fat were measured in eggs from the vicinity of a municipal waste incinerator in Wuhan in the same sample where high PBDD/F concentrations were detected (Petrlik, 2016). Compared to other sites, this was the ninth highest value according to a study from 2021 (Petrlik et al., 2021).

In the Czech Republic, PBDEs were found, for example, in a mixture of fly ash and bottom ash, known as SPRUK, from WtE Termizo Liberec (Czech Republic);(Petrlik, 2006).

5.1.7.2 "Novel" Brominated Flame Retardants (nBFR)

New brominated flame retardants are a group of chemicals that, in many cases, have replaced already restricted BFRs. Various sources mention different chemicals in this group, but only some of them are measured in environmental matrices. In the Czech Republic, six nBFRs are mostly analyzed in this group:

- 1,2-bis(2,4,6-tribromophenoxy)ethane (BTBPE)
- Decabromodiphenyl ethane (DBDPE)
- Hexabromobenzene (HBBz)
- Octabrom-1,3,3-trimethylphenyl-1-indane (OBIND)
- 2,3,4,5,6-pentabromomethylbenzene (PBEB)
- Pentabromotoluene (PBT)

These substances are already widespread not only in the environment but also in food (Shi, Zhang et al., 2016). A more recent overview (Xiong et al., 2019) suggests: *"Toxicity data for nBFRs show that several nBFRs may have adverse effects such as hormonal disruption, endocrine disruption, genotoxicity, and behavioral disorders."*. It has been found that HBBz, PBEB, and PBT accumulate in aquatic organisms (Wu et al., 2011; Xiong et al., 2019). Decabromodiphenyl ethane (DBDPE) was introduced in the early 1990s as an alternative to DecaBDE in plastic and textile applications (Ricklund, Kierkegaard et al., 2010).

The new brominated flame retardant 1,2-bis(2,4,6-tribromophenoxy)ethane (BTBPE) was first produced in the 1970s and is used as a replacement for OctaBDE (Hoh et al., 2005). It has the ability to bioaccumulate and biomagnify in aquatic food chains (Law et al., 2006; Wu et al., 2011). Similar to DecaBDE, it has been found that the commercial BTBPE mixture contains brominated dioxins (PBDD/Fs) and/or promotes their formation during the processing of ABS plastic (Ren et al., 2017; Tlustos et al., 2010; Zhan et al., 2019).

HBBz is commonly used in the production of paper, wood, textiles, plastics, and electronic goods (Watanabe & Sakai, 2003). PBEB is a flame retardant that was mainly used in the 1970s and 1980s under the name FR-105 (de Wit et al., 2011; Straková et al., 2018). PBT is used in polystyrene casings for electronics, ABS plastics, and other plastic polymers, and is sold under the name FR-105 or Flammex (de Wit et al., 2011; Straková et al., 2018). OBIND is another replacement for PBDE, used in various plastics in electronic products (Straková et al., 2018).

Among nBFRs, the highest concentration found in eggs near an incinerator in Wuhan was BTBPE (51 $ng.g^{-1}$ fat); (Petrlik, 2016). Otherwise, these substances are rarely monitored in samples from waste incinerator surroundings (McGrath et al., 2017), if at all, even though they deserve attention.

5.1.8 Per- and polyfluoroalkyl substances (PFAS)

The term PFAS refers to per- and polyfluoroalkyl substances (PFAS) used both in industry and households. They have been manufactured since approximately the 1950s. This group, comprising around 10,000 compounds (ECHA, 2023), is primarily known for its use in Teflon or Gore-Tex,[39] as well as in paper food packaging, outdoor clothing, or carpets. Most PFAS representatives are either persistent themselves or act as precursors[40] to other persistent compounds. These substances are continuously released into the environment and subsequently bioaccumulate in living organisms (Duffek et al., 2020; Lanková et al., 2011; Lewis et al., 2022).

Humans primarily ingest PFAS through drinking water and food, but also through dust, personal care items, or consumer goods (Straková et al., 2022). PFAS bind to proteins, hence they are predominantly found in the livers, blood serum, plasma, or kidneys of living organisms, as well as in urine, placenta, or breast milk (Duffek et al., 2020; Llorca et al., 2010; Xu et al., 2022).

Some PFAS are considered suspected human carcinogens (Temkin et al., 2020) and are linked to kidney, ovarian, testicular, and prostate cancers. Some PFAS reduce women's fertility (Wang et al., 2023), increase the risk of high blood pressure during pregnancy, preeclampsia (placental disease), or lower birth weights of newborns (Borghese et al., 2020). PFAS can damage the immune system (Temkin et al., 2020). An overview of all possible effects of these substances on human health is shown in Figure 5.9.

Municipal waste can contain significant amounts of material contaminated with PFAS and/or other fluorinated compounds, which can lead to PFAS emissions and release during incineration. PFOS and PFOA were measured in air emissions from the WtE plant in Harlingen, Netherlands (Arkenbout & Petrlík, 2019), see Chapter 3.5.3. High and/or increased levels of PFAS were also detected in biological analyses of moss and needles near two European municipal waste incinerators (Arkenbout & Bouman, 2021). These results support suspicions of PFAS formation during waste incineration. Other studies have concluded that the flue gas could be a significant source of PFAS emissions from waste incinerators (Ahrens et al., 2011).

39 Gore-Tex and Teflon are also used for cleaning flue gases from incinerators. Paradoxically, they may be the source of new POPs pollution, this time PFAS.

40 Precursor – a compound from which another compound is formed by chemical transformation.

Figure 5.9: *Health risks of PFAS*

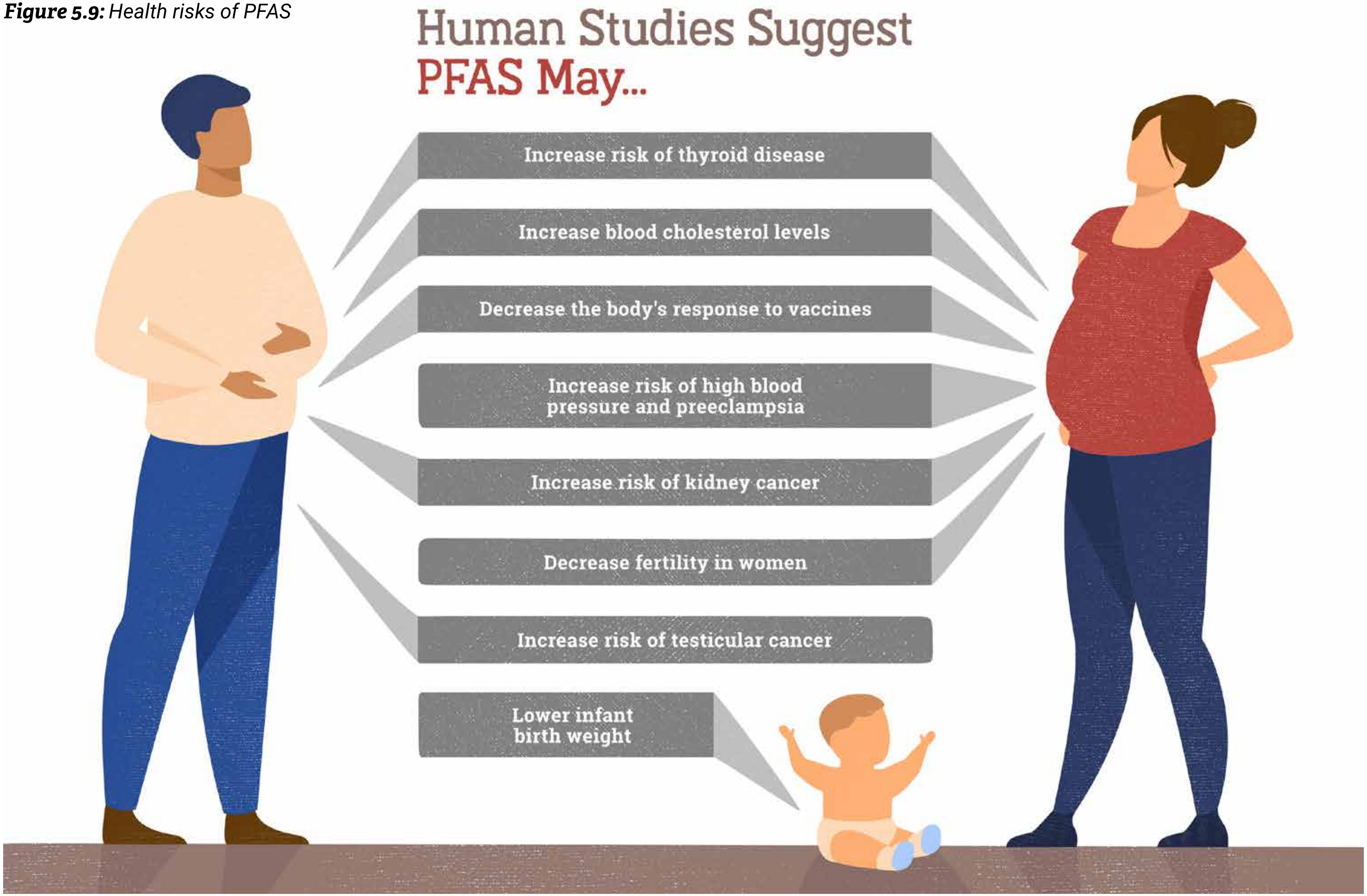


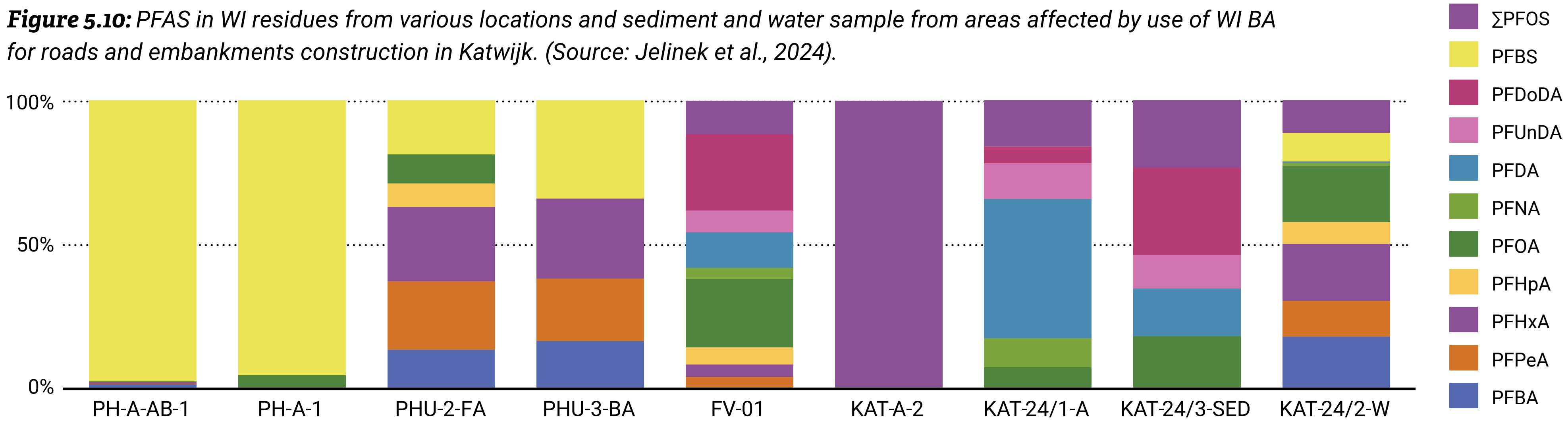
Photo 5.20 *Bricks made of WI ash are used for construction of houses in Aguado as well, despite their contamination with POPs. Photo: EcoWaste Coalition, 2020.*

A study of 31 Swedish waste incinerators (Strandberg et al., 2021), measuring PFAS in residues after waste incineration but not directly from chimneys, found PFAS presence in bottom ash from 9 sampled facilities, in fly ash from 15 facilities, and in condensate from 13 facilities. The same study concluded that PFAS were detected regardless of operational conditions. The highest levels of PFAS in condensate were found in incinerators operated at temperatures above 1,100°C. The authors noted that this occurred *"despite the general hypothesis that all organic substances decompose (burn) at temperatures above 1,000°C"* (Strandberg et al., 2021). The study also found high concentrations of PFAS in both incinerators burning solely municipal waste and in industrial (hazardous) waste incinerators, suggesting that both types of incinerators are significant sources of PFAS. Therefore, policymakers should not rely solely on "high temperature" in incineration facilities as a criterion that will lead to PFAS destruction.¨PFAS were detected in bottom and fly ashes from waste incinerators in various countries in a new study (Jelinek et al., 2024), ranging from <LOQ to 67.75 ng.g^{-1} dm, with the highest concentrations found in ash from a hazardous (mostly medical) waste incinerator in Aguado, Philippines. The detailed results are shown in Table 5.9, and the representation of individual PFAS is shown in the graph in Figure 5.10. In addition to the data presented here, the study also contains information on PFAS concentrations in eggs from free-range chickens around waste incineration facilities: *'In free-range chicken eggs from areas surrounding waste incineration facilities, PFAS values were 4 to 27 times higher than a reference sample from Jakarta (0.1 ng.g^{-1} ww), ranging from 0.38 to 2.69 ng.g^{-1} ww. The highest levels of 2.69 and 2.38 ng.g^{-1} ww were measured in samples from Phuket and Aguado, respectively"* (Jelinek et al., 2024). Contaminaiton of eggs in Aguado could occure also through bricks made of waste incineration ash (see Table 5.9) and sold to villagers (see Photo 5.20).

Table 5.9: *Summarized results of the analyses for PFASs of the samples from Aguado, Phuket, Bantar Gebang and Katwijk. Results are in ng.g^{-1} dry matter for ash and sediment samples and in ng.L^{-1} for water samples. Source (Jelinek et al., 2024).*

Locality (country)	**Aguado (Philippines)**		**Phuket (Thailand**		**Frydlant (Czechia)**	**Katwijk (Netherlands)**			
Sample ID	PH-A-AB-1	PH-A-1	PHU-2-FA	PHU-3-BA	FV-01	KAT-A-2	KAT-24/1-A	KAT-24/3-SED	KAT-24/2-W
Matrix	brick from ash	BA	ash	ash	BA/FA	BA (modified)		sediment	water
Units					ng.g^{-1} dm				ng.L^{-1}
Total	67.75	24.08	0.43	0.13	1.40	0.02	0.62	0.43	45.58

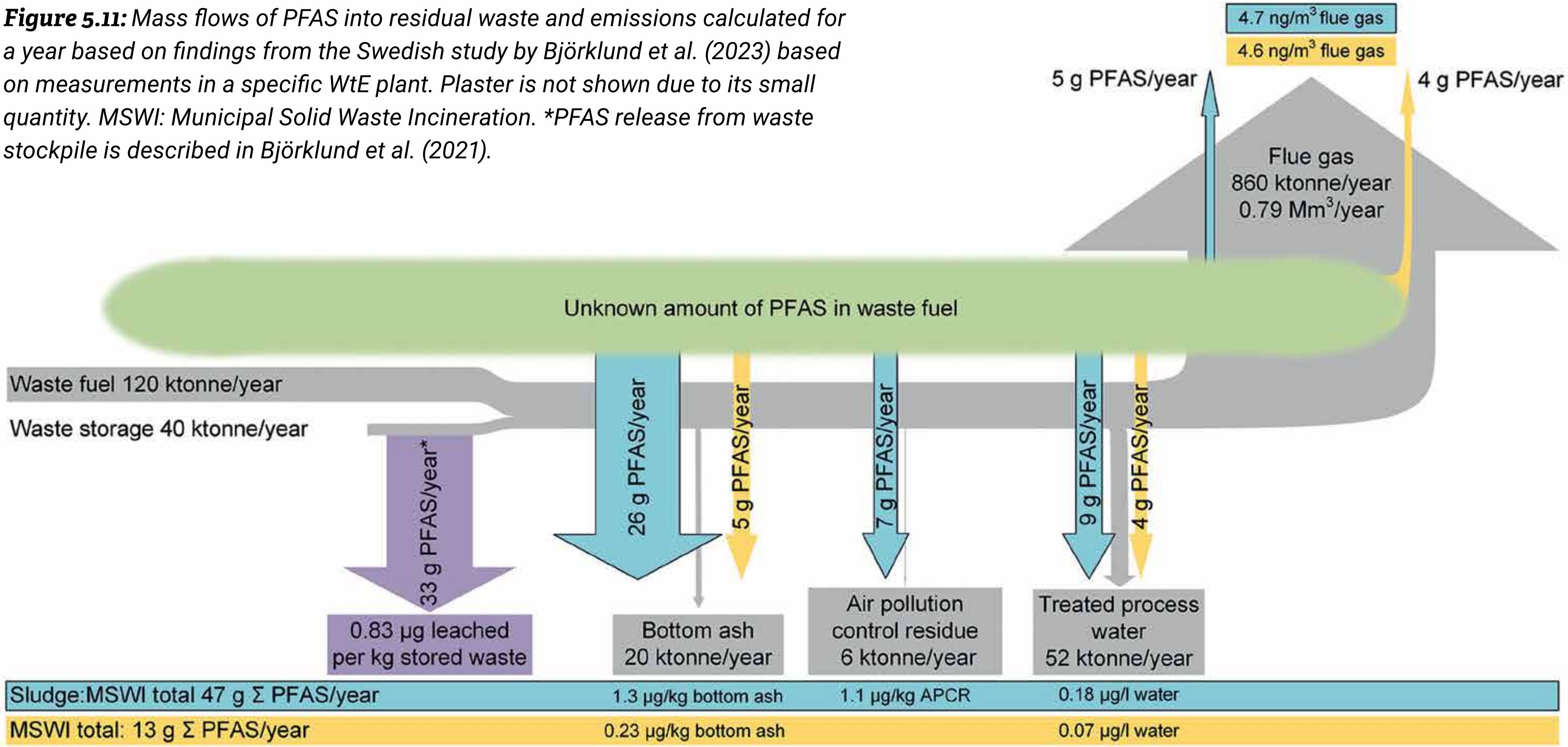
Figure 5.10: *PFAS in WI residues from various locations and sediment and water sample from areas affected by use of WI BA for roads and embankments construction in Katwijk. (Source: Jelinek et al., 2024).*



Similar to PBDD/F, PFAS are concentrated in bottom ash in incinerators. Liu et al. (2021) found that in two out of three incinerators, their concentration in bottom ash was 3 times higher than in fly ash. In addition to no European country systematically analyzing residues after waste incineration destined for use as construction materials for PFAS content (Blasenbauer et al., 2020), Liu et al. (2021) found that bottom ash represents an important vector for transferring PFAS to the environment, thus reliable techniques for PFAS decomposition in these materials are needed.

PFAS significantly contaminates drinking water. Its contamination mainly occurs around industrial plants, airports, and military bases where PFAS containing firefighting foam is used in training to suppress fires (Darlington et al., 2018; Milley et al., 2018). It is noteworthy that PFAS were found in sediments and fish from the Lužická Nisa River downstream of the WtE Termizo Liberec (Czech Republic) in a 2011 study (Lanková et al., 2011). This may not necessarily relate to the waste incinerated there, but to ash disposal, firefighting, or firefighter training in the incinerator's vicinity. Nevertheless, these substances need monitoring around waste incineration facilities, which have been shown to be significant sources of PFAS. This requirement is supported by new study results from Sweden, monitoring the fate of PFAS during incineration of both pure municipal waste and its mixtures with sewage sludge, concluding: *"...some PFAS are not fully degraded at high temperatures during the WtE conversion and can be emitted from the plant via bottom ash, gypsum, treated process water, or flue gas"* (Björklund et al., 2023). The study also found a significant increase in PFAS emissions and transfer during co-incineration of sewage sludge, a practice proposed by some WtE projects in the Czech Republic. The Swedish study well-mapped mass flows of PFAS during the incineration of municipal waste, displaying them in a diagram taken from this study (see Figure 5.11). The same team monitored the flows of PFASs in a full scale WtE plant in northern Sweden and concluded that: *"PFASs were found in all sample types except for boiler ash. The total levels of 18 individual PFASs (Σ18PFASs) in untreated flue gas ranged from 5.2 to 9.5 ng.m^{-3}, decreasing with 35% ± 10% after wet flue gas treatment. Σ18PFASs*

Figure 5.11: *Mass flows of PFAS into residual waste and emissions calculated for a year based on findings from the Swedish study by Björklund et al. (2023) based on measurements in a specific WtE plant. Plaster is not shown due to its small quantity. MSWI: Municipal Solid Waste Incineration. *PFAS release from waste stockpile is described in Björklund et al. (2021).*



*in the condensate ranged from 46 to 50 ng.L^{-1}, of which perfluorohexanoic acid (PFHxA) made up 90% on a ng.*L^{-1} *basis. PFHxA was also dominant in filter ash, where Σ18PFASs ranged from 0.28 to 0.79 ng.gl^{-1}"* (Björklund et al., 2024). This study shows that flue gas treatment can capture some PFASs and transfer them into WtE residues. Even this latest study confirmed that PFASs are not fully degraded at high temperatures during waste incineration in modern WtE plants.

PFAS in emissions from modern municipal waste incinerators were also found in Harlingen, Netherlands (see graph in Figure 3.7); (Arkenbout & Petrlík, 2019).

Pilot tests for incinerating PFAS are also being conducted in hazardous waste incinerators and cement plants. In such a pilot test at the Veolia Dry Creek high-temperature hazardous waste incinerator in South

Photo 5.21: *The Cement Australia Fisherman's Landing cement plant in Gladstone, Queensland, Australia, experimentally incinerating PFAS-containing firefighting foams, even though it did not reach the minimum target DRE of 99.9999 % for many PFAS compounds, suggesting their release into the atmosphere. Comparing these data with existing literature supports the hypothesis that high temperatures alone are not a guarantee of PFAS destruction in incineration facilities (Kuepouo et al., 2022); (Cement Australia, 2017). Photo: Jindřich Petrlík, Arnika.*

Australia (Veolia, 2019b), analyses revealed certain amounts of PFAS in emissions. However, the destruction and removal efficiency (DRE) could not be calculated because the PFAS concentration in bottom ash after processing was higher than the input PFAS concentration in waste. This indicates that incineration might be a source of PFAS rather than an effective means of their destruction.

Photo 5.22 *The East Liverpool hazardous waste incinerator incinerated PFAS-containing firefighting foams. Contamination of soil with these substances occurred in its vicinity. Photo: William D. Lewis, Mahoning Matters (Kruzman, 2022).*

In the vicinity of the East Liverpool hazardous waste incinerator in Ohio, USA, research was conducted on PFAS concentrations in soils, and measurable concentrations were found in all 35 sampled soil samples. PFOS was detected in the majority of soil samples (97 %) ranging from 50–8,300 $ng.kg^{-1}$. PFOA was measured in 94 % of soil samples ranging from 51 $ng.kg^{-1}$ to 1,300 $ng.kg^{-1}$ (Martin et al., 2023). This research shows that hazardous waste incinerators likely cannot safely break down PFAS, and their incineration leads to emissions of this group of substances into the surrounding environment. The East Liverpool incinerator obtained a contract to burn PFAS-containing firefighting foams in 2019 (Kruzman, 2022).

5.1.9 Other POPs

Residues from waste incineration contain not only UPOPs, which are subject to the Stockholm Convention, but also other POPs (Petrlik & Ryder, 2005), which are not covered by it. The Stockholm Convention was established with the aim of protecting health and the environment from (originally) 12 substances,[41] of which 4 were UPOPs. Since waste incinerators produce UPOPs in relatively large quantities, they have become major sources of these substances listed in Annex C of the Stockholm Convention.

5.1.9.1 Polychlorinated Naphthalenes (PCN)

Polychlorinated naphthalenes (PCN) were manufactured for similar purposes as PCBs. PCN creates efficient insulating coatings for electrical conductors. Other PCNs were used as wood preservatives, additives in rubber and plastics, in capacitor dielectrics, and in lubricants (Stockholm Convention, 2017). However, these chemicals are also formed unintentionally during high-temperature processes in the presence of chlorine, similar as PCDD/F and dl PCB.

PCN can induce toxic effects typical of dioxin-like compounds and are potentially teratogenic. Several short-term and medium-term tests show high acute toxicity, i.e., weight loss, liver damage, and delayed deaths at relatively low concentrations (>3 $mg.kg^{-1}$). Occupational studies have shown adverse effects on human health; some of them have also been observed in animal studies (dermal effects, liver disease). Certain evidence has been demonstrated for a correlation with an excess of specific cancers (POP RC, 2012b).

41 Among the top twelve – the „dirty dozen" – of the most dangerous POPs covered by the Stockholm Convention were eight substances used as pesticides, two industrial compounds and polychlorinated dibenzo-p-dioxins and dibenzofurans, which are formed involuntarily in chemical production and in the combustion of chlorinated substances. These substances were referred to as „dirty dozen".

Photo 5.23: *The Stockholm Convention regulating POPs was signed for the Czech Republic in May 2001 in Stockholm by the then Minister of the Environment, RNDr. Miloš Kužvart. Photo: Earth Negotiations Bulletin, 2001, https://enb.iisd.org/chemical/popsd.*

In simulated waste incineration conditions, Noma et al. (2004) measured 0.17 to 0.96 $ng.g^{-1}$ of polychlorinated naphthalenes (PCN) in fly ash and 0.95 to 1.7 $ng.g^{-1}$ in bottom ash. In the bottom ash of Japanese waste incinerators, concentrations from 0.74 to 610 $ng.g^{-1}$ were found (Kawano et al., 1998). These substances are not commonly monitored in solid residues from waste incineration.

5.1.9.2 Polychlorinated Dibenzothiophenes (PCDT)

Polychlorinated dibenzothiophenes (PCDT) are the sulfur analogs of polychlorinated dibenzofurans (PCDF). Similar to them, PCDT are an unintended by-products in chemical and combustion processes. The problem with measuring polychlorinated dibenzothiophenes is mainly because standards for these substances necessary for routine quality measurement of their concentrations are not available. Despite missing information, we consider PCDT as persistent substances with high potential for bioaccumulation. The toxicity of PCTD is lower than the most dangerous PCBs (Mantyla, 1992). Due to their high similarity, dioxin-like effects are expected, namely damage to the hormonal and immune system. However, they do not have as high a potential as their oxygenated counterparts (PCDF); (Kopponen et al., 1993).

Buser (1992) detected PCDT in fly ash samples from two municipal waste incinerators and in fly ash from operations where car wrecks were processed using an electric arc. PCDT was also found in gaseous emissions from waste incinerators. They are also formed in secondary metal production processes.

The results of analyses in various parts of the environment and matrices were summarized for the first time in a Finnish study (Sinkkonen, 1997). The second, Polish study also focused on polychlorinated thianthrenes (PCTA), which are the sulfur analogs of PCDD. This study includes, among other things, information about PCDT in the Elbe River sediments (Czerwinski, 2008). Trace amounts of PCDT have been determined, for example, in soil, sediments, airborne dust, some aquatic organisms, and conifer needles.

5.1.10 Limits for POPs in Waste

The determination of the "Low POPs Content Level (LPCL)" is related to the content of different POPs in waste. If waste contains any POPs above the LPCL then that waste is treated as "waste contaminated with POPs" and requires special treatment to ensure destruction of the POPs in the waste. Currently, in the European Union, this limit for dioxins is set at 5 µg TEQ.kg^{-1} (European Parliament and Council of the EU, 2022), and at 1 or 15 µg TEQ.kg^{-1} in the global recommendation approved by the Basel Convention in the General Technical Guidelines for POPs Waste (Basel Convention, 2023). For POPs present in waste from incinerators, these limits are summarized in Table 5.10.

Photo 5.24: *For the negotiations on tightening limits for POPs in waste that took place in 2022, Slovak Member of Parliament Martin Hojsík (Progressive Slovakia) from the Renew Europe group was the rapporteur in the European Parliament. Photo: Martin Hojsík's Archive.*

Table 5.10: *Limits for POPs in waste as established in the waste directive containing POPs, approved by the Basel Convention (Basel Convention, 2023), and in the Regulation of the European Parliament and Council (EU) on POPs (European Parliament and Council of the EU, 2019, 2022).*

	Global „provisional“ recommended limits for LPCLs	Limits set by EU regulation	Limits enforced by African states and the international IPEN network
PCDD/F	1 or 15 µg TEQ.kg^{-1}	-	1 µg TEQ.kg^{-1}
PCDD/F + dl PCB	-	5 µg TEQ.kg^{-1}	1 µg TEQ.kg^{-1}
HBCD	100 or 1,000 mg.kg^{-1}	500 mg.kg^{-1}	100 mg.kg^{-1}
Sum of PBDEs	-	500 mg.kg^{-1}	50 mg.kg^{-1}
Hexa-, hepta- and tetraBDE	50 or 1,000 mg.kg^{-1}	-	50 mg.kg^{-1}
PCB	50 mg.kg^{-1}	50 mg.kg^{-1}	50 mg.kg^{-1}
PCN	10 mg.kg^{-1}	10 mg.kg^{-1}	10 mg.kg^{-1}
PeCB	50 mg.kg^{-1}	50 mg.kg^{-1}	50 mg.kg^{-1}
HCB	50 mg.kg^{-1}	50 mg.kg^{-1}	50 mg.kg^{-1}
PFOS, its salts and PFOSF	50 mg.kg^{-1}	50 mg.kg^{-1}	-
PFOA, its salts and PFOA / related compounds	-	1 mg.kg^{-1}/ 40 mg.kg^{-1}	-
Sum of PFOS, PFOA, PFHxS Sum of PFOS, PFOA, PFHxS / their related compounds	-	-	0,025 mg.kg^{-1}/ 10 mg.kg^{-1}
PFHxS, its salts/compounds related to PFHxS	-	1 mg.kg^{-1}/ 40 mg.kg^{-1}	-

5.2 Other Organic Substances

Polycyclic aromatic hydrocarbons (PAHs) and volatile organic compounds (VOCs) are a broad group of chemical substances with a wide range of negative health effects. The types and quantities of individual VOCs waste incinerators emit into the air has not been extensively researched.

A new study from Vietnam found high concentrations of toxic VOCs, specifically benzene, toluene, ethylbenzene, and xylenes (BTEX) around four MSWIs: *“Concentrations of benzene, toluene, (m,p)-xylenes, o- xylenes and ethylbenzene ranged from 4.53 to 36.75 µg.m^{-3}, from 16.29 µg.m^{-3} to 125.36 µg.m^{-3}, from 2.82 µg.m^{-3}to 31.45 µg.m^{-3}, from 1.42 µg.m^{-3}to 25.61 µg.m^{-3}, from 1.32 µg.m^{-3}to 10.79 µg.m^{-3}, respectively. As a result of the risk assessment, it was determined that the incinerator’s exhaust gas caused secondary environmental damage, impacting the health of not only workers but also people living in nearby communities”* (Dung et al., 2023).[42]

Chlorinated benzenes (CBzs) are organic pollutants produced through various industrial and thermal processes, with limited and outdated studies on their full congener profiles.

Concentrations and congener profiles of seven CBzs were analyzed in bottom and fly ash (BA and FA) samples from a medical and a municipal waste incinerator in northern Vietnam, showing fly ash concentrations ranging from 6.98 to 34.4 ng.g^{-1} (median 19.1) in the medical incinerator and 59.1 to 391 ng.g^{-1} (median 197) in the municipal incinerator. Bottom

42 The waste incinerator in Nam Dinh uses a Losiho technology furnace, while in Vinh Phuc uses a SANKYO furnace made in Thailand, the fuel is natural gas. The composition of domestic waste in both these areas is mainly organic matter, glass bottles, paper (Dung et al., 2023).

ash concentrations were lower, with medians of 1.95 $ng.g^{-1}$ (range 1.53-5.98) in the medical incinerator and 17.4 $ng.g^{-1}$ (range 14.5-42.6) in the municipal incinerator, suggesting low-temperature catalytic formation of these pollutants in the post-combustion zone. Although estimated cancer risks from ash-bound CBzs for workers were below critical levels, the risks from other pollutants were not considered (Nguyen et al., 2021).[43]

Later study determined concentrations of 12 CBzs in fly ash (FA) and bottom ash (BA) from a MWI and an industrial waste incinerator (IWI) in northern Vietnam, finding again higher levels in BA (median 25.3 $ng.g^{-1}$) compared to FA (median 7.30 $ng.g^{-1}$). Emission factors for Σ12CBzs ranged from 21 to 600 µg/ton for FA and 190 to 4570 $ug.ton^{-1}$ for BA, resulting in annual emissions of about 6 $g.year^{-1}$ for the IWI and 3 $g.year^{-1}$ for the MWI. The study highlights the need for further investigations on the emission and environmental occurrence of all 12 CBzs rather than focusing only on regulated congeners (Nguyen et al., 2024).

As we mentioned in the air chapter (3.1), in 1995, Jay and Stieglitz published research attempting to determine organic substances contained in smoke emissions from municipal waste incineration. They identified approximately 250 individual compounds at concentrations exceeding 50 $ng.m^{-3}$. This represented about 42 % of all organic substances in emissions. The remaining 58 % consisted of unidentified aliphatic hydrocarbons (Jay & Stieglitz, 1995). Among the identified substances in emissions were several carcinogens or other substances harmful to health. It was a one-time study, and these compounds are not commonly measured in emissions from incinerators. Part of the identified substances are listed in Figure 5.12.

43 Only PeCB and HCB are regulated CBz congeners at international level, they are listed under the Stockholm Convention.

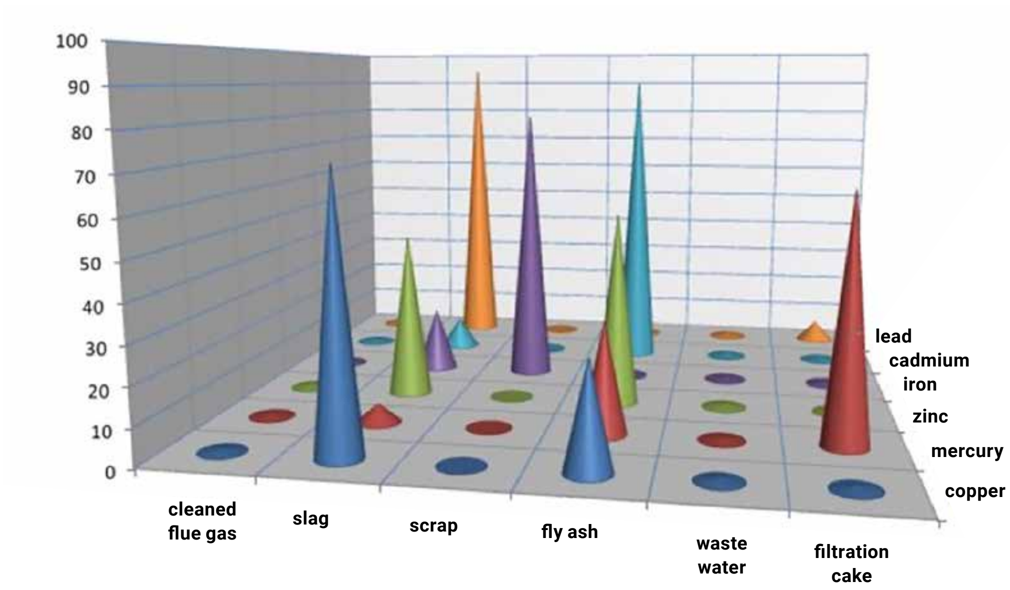
5.3 Heavy Metals

A significant amount of metals (Rollinson et al., 2022) that meet at least one of the REACH regulation criteria, end up in the bottom ash from waste incinerators. These include arsenic, barium, cadmium, cobalt, chromium, copper, lead, mercury, molybdenum, nickel, antimony, tin, vanadium, and zinc (Rollinson et al., 2022), as shown in Table 5.11. Solid municipal waste is highly heterogeneous, making its incineration a very complex process involving thousands of chemical reactions (Chagger et al., 2000). Theoretically, elements like Cd and Hg with lower boiling points than the temperature of grate should not appear in the bottom ash, while others like Pb and Zn with higher boiling points should always fall out through the grate. However, this does not happen; As, Br, Cd, and even Hg are commonly found in the bottom ash (Buchholz & Landsberger, 1995; Meima & Comans, 1999). Metals enter the incinerator in a less hazardous form than they exit. They leave it released from the materials they were bound to, reduced to elemental form or simpler compounds, making them more mobile and biologically available. This increases the likelihood of them reaching groundwater, surface water, or the food chain, where they can affect human health or other organisms. As these are elements that do not decompose over time, unless they remain in the same place where they were deposited, they lead to environmental contamination.

An idea of the distribution of selected metals in residues from municipal waste incineration in the Czech Republic and Austria can be obtained from the graphs in Figures 3.4 and 5.13, respectively. This resulted from detailed analyses conducted in 2004 at WtE SAKO Brno (Czech Republic);(Karásek, 2010) and in 1994 at the Vienna – Spittelau waste incinerator (Schachermayer et al., 1994) respectively. Table 5.11 below is derived from this monitoring and provides information on the amount of heavy metals ending up annually in residues from waste incineration in a plant with an annual installed capacity exceeding 200,000 tons

Figure 5.12: *List of a substantial portion of substances identified in smoke from municipal waste incineration. (Source: Jay & Stieglitz, 1995)*

Figure 5.13: *Transfer of heavy metals in air emissions, bottom ash, separated metals from slag, fly ash from electrostatic filter, waste water and filter cake in waste incinerator in Vienna – Spittelau in 1994. (Source: Karásek, 2010; Schachermayer et al., 1994).*



of waste but actually incinerating over 50,000 tons of waste per year (ČHMÚ, 2010).[44]

From the overview in Table 5.11 and graphs at Figures 3.4 and 5.13, it is evident that most heavy metals released during incineration end up in the solid waste products of waste incineration, primarily in the bottom ash.

44 The information about amount of incinerated waste in that MSWI applies to the year of analysis only.

Table 5.11: *Amount of heavy metals in combustion products in mg.t-1, quantity of individual elements in g.t-1 in various combustion products, and the total mass of all monitored elements (sum) in kg.t-1 in individual combustion products. (Source: Karásek, 2010)*

ement	Bottom ash	Fly ash	End-product	Flue gases	Total	Quantity [g.t^{-1}]
Antimony	9,763.51	3,189.42	1405.15	1.00	14,359.09	14.36
Arsenic	1,293.07	243.37	71.60	5.14	1613.13	1.61
Aluminum	7,318,132.80	672,034.20	130,294.80	379.35	8,120,841.15	8,120.84
Chromium	27,743.76	2,284.20	184.21	9.28	30,221.45	30.22
Cadmium	2,332.44	2,250.36	1,003.68	0.63	5,587.11	5.59
Cobalt	2,659.80	331.35	24.17	0.38	3,015.70	3.02
Manganese	147,843.96	13,084.80	468.79	92.02	161,489.57	161.49
Copper	849,499.20	7,467.36	20,936.52	52.15	877,955.23	877.96
Nickel	15,684.64	782.83	1,548.35	5.27	18,021.09	18.02
Lead	396,637.56	20,067.12	9,614.52	26.20	426,345.40	426.35
Mercury	116.21	15.20	609.18	143.17	883.76	0.88
Thallium	525.82	73.60	52.44	0.04	651.90	0.65
Vanadium	7,971.22	442.35	2,908.22	5.64	11,327.43	11.33
Zinc	1,318,115.04	148,086.66	49,859.64	226.41	1,516,287.75	1,516.29
Iron	5,279,498.40	210,907.80	46,267.20	491.43	5,537,164.83	5,537.16
Sum [kg.t^{-1}]	15.38	1.08	0.27	$1.44.10^{-3}$	16.73	

Mach (2017) collected samples from the vicinity of the waste processing facility at Hůrka, where waste biodegradation and stabilization occur, producing certified products for land reclamation of tailings, mine workings, and waste dumps. In 2014 and 2015, fly ash from municipal and hazardous waste incineration were also taken for processing. Some organic substances, organic compound groups, and 20 metals in sediments were determined in samples from the facility's surroundings. Cadmium and zinc concentrations were found to be several times higher at two out of three sampling points compared to average sediment values in watercourses in the Czech Republic from 1995 to 2004. The highest concentrations of PCDD/F, PAHs, and metallic elements were detected at the sampling point labelled "near the reservoir," located in immediate proximity (within meters) of the facility boundary.

Table 5.12: *Content of hazardous substances [in mg.kg^{-1}] in dry matter in the mixture of bottom ash and fly ash from Termizo a.s. in Czech Republic (Waste III, reference samples 6 and 8) compared to the threshold values specified in Appendix No. 1 to Regulation No. 294/2005 Sb. (Values exceeding the limits are marked in bold). (Source: Košařová, 2006)*

Element	Waste III	Appendix 6	Appendix 8	Limit values in Decree No. 294/2005 Sb.
As	16	**43**	**54**	10
Cd	**1.7**	**13**	**8**	1
Cr	120	113	90	200
Hg	**1.2**	**4**	**2.6**	0.8
Ni	**230**	**220**	**160**	80
Pb	**550**	**2,300**	**2,200**	100
V	**64**	**120**	**78**	180

The content of metals in a sample mixture of bottom ash and fly ash (Waste III) from WtE Termizo Liberec (Czech Republic), taken from the storage area belonging to the Strabag company, is summarized in Table 5.12 (Košařová, 2006). Alongside are reference samples 6 and 8 of WI ashes (from reports on hazardous waste properties assessments ordered by the Termizo Liberec). Simultaneously, the table includes the threshold values from the then-applicable Regulation 294/2005 Sb. on conditions for waste disposal in landfills and their utilization on the terrain surface. Upon comparison, it was concluded that "this mixture cannot be used for land reclamation of excavated surface mine workings, land modifications, or reclamations of areas affected by human activities without restrictions. This fact applies both to the utilization of waste on the terrain surface and its use as a building material" (Košařová, 2006).

5.3.1 Lead

Lead is a major global environmental health hazard that poses serious risks, particularly to young children. Approximately 80–90 % of daily exposure occurs through food consumption (Krejpcio et al., 2005; Liu et al., 2010b). Lead is cumulative and has a long half-life in bones, it remains in the human body for decades. Lead exposure during pregnancy is linked to miscarriage, while prolonged exposure reduces male fertility (Amadi et al., 2017; Vigeh et al., 2011). Elevated blood lead levels are associated with neurodevelopmental issues in children, including attention-deficit disorders and learning disabilities (Flora et al., 2006). Lead's impact on the nervous system manifests as irritability, attention and memory disturbances, headaches, muscle tremors, hallucinations, prolonged reaction times, decreased IQ, and nerve conduction velocity. Chronic lead exposure disrupts various body functions, causing next to neurological cardiovascular and hematologic issues (Debnath et al., 2019; Ministry of the Environment of the Czech Republic, 2021d; Pal et al., 2015; Rao et al., 2014). IARC classifies lead into group 2B, possibly carcinogenic, while its inorganic compounds fall into group 2A, probably carcinogenic. According to IARC, lead's organic compounds are not carcinogenic and are classified under group 3 (IARC, 2023).

Near a Korean municipal waste incineration plant, blood monitoring of 841 individuals revealed an average lead level of 43.1 μg.L^{-1} with a median of 41.9 μg.L^{-1} (Lee et al., 2012). The blood levels of lead and cadmium were slightly higher in the group of the subjects who had resided the longest near the municipal waste incinerators in Korea (Lee et al., 2012).

Environmental impacts include lead binding to airborne dust particles, settling on vegetation, and its presence in soil and water (Nieder et al., 2018). Lead is a highly toxic metal found in all components of the environment (Ministry of the Environment of the Czech Republic, 2021d). Lead from incinerator is released in air and can be found in both bottom ash and fly ash.

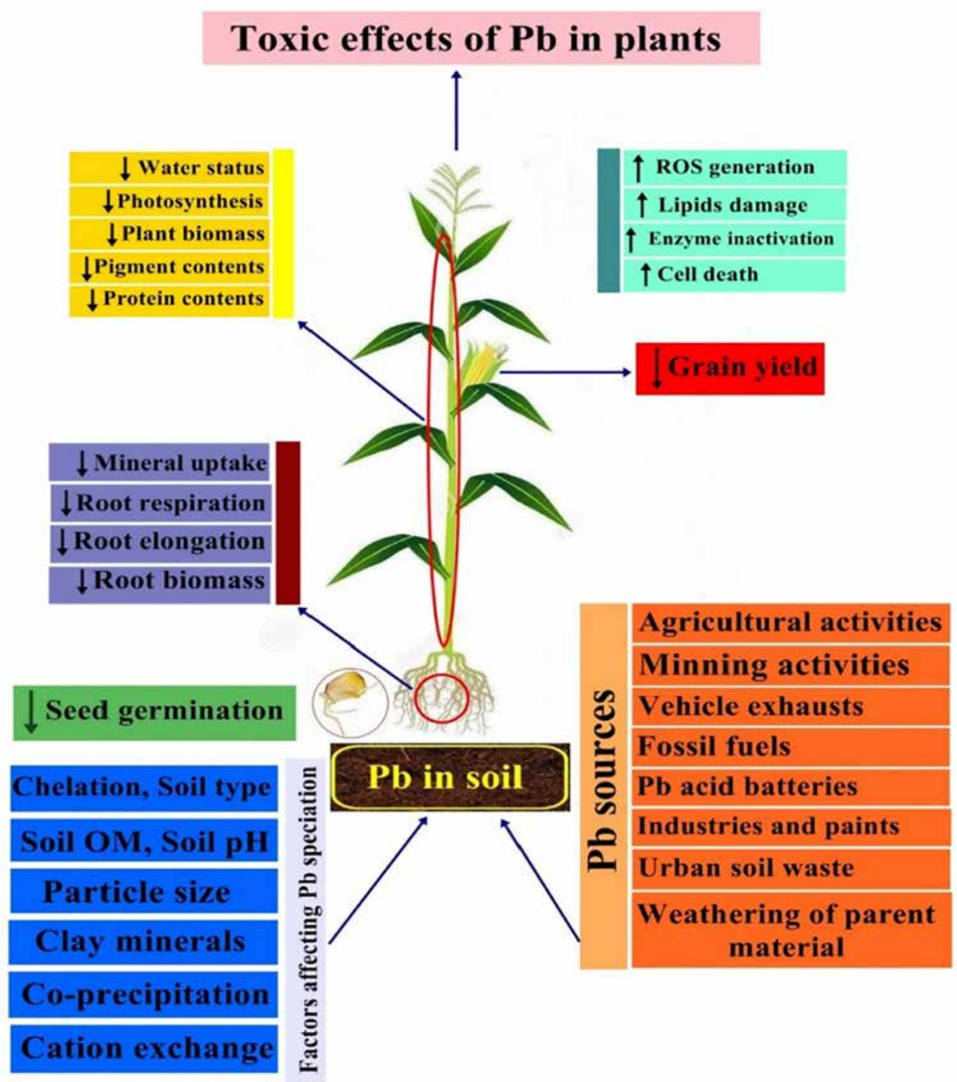
Lead is directly present in bottom ash at a concentration of 4,000 mg.kg^{-1} and 2.9 mg.kg^{-1} in cement (Rozumová et al., 2015). Fly ash shows concentrations between 1,036 and 5,090 mg.kg^{-1} (Ajorloo et al., 2022). Lead was detected at concentrations between 700 and 1,100 mg.kg^{-1} in bottom ash and fly ash samples from locations in southern Taiwan, with values exceeding 1,000 mg.kg^{-1} in dust from the vicinity of both bottom and fly ash repository (Bell et al., 2023a). In the mixture of bottom ash and fly ash from WtE Termizo Liberec (Czech Republic), lead concentrations exceeded 2,000 mg.kg^{-1} of dry matter (Košařová, 2006). According to Glauser et al., (2021), none of the Dutch bottom ash samples met Swiss regulations for landfill due to total concentrations of heavy metals Cr, Cu and Pb in some grain size fractions. Pb was found in in larger (>31.5 mm) and smaller (4-8 mm) fractions of bottom ash (Vateva and Laner, 2020), while Mantovani et al. (2021) detected higher concentrations of Cr and Pb in the largest (> 16 mm fraction). Concentration of Pb in leachate was found to be independent of pH (it was also found out for Cd, Cu and Mo), while As, Cd and Pb leached in mildly acidic conditions and were assigned as "long term leaching hazards" by Buchholz and Landsberger (1995). Pb next to Cr and Cu in leachate were above limit values of two current and one draft German standard for building aggregate (Vateva and Laner, 2020). Different quantities of Pb next to Co, Ni and Cd leached out of the bottom ash under different kinds of tests (Kalbe and Simon, 2020). According to Mehr et al. (2021), modern plants have extraction efficiency for Pb only 16 %.

In Britain, the spatial distribution of lead levels in soils showed a marked variation downwind from the Baldovie incinerator in comparison with the background level for the area but remained well within the typical range of lead in rural, unpolluted, British soils (Collett et al., 1998). In China, relatively high contents of cadmium, lead, antimony, and zinc in the soils at 250 m and 750–1250 m away from the MSW incinerators were found to be related to MSW incineration, while the elevated contents of the other four heavy metals (chromium, copper, mercury, nickel) were associated with other anthropogenic activities (Li et al., 2019). In a study conducted in the USA, the concentration of cadmium and lead in foliage was found to decrease with distance from the incinerator (Bache et al., 1992). Airborne dispersal of Pb is identified as particular critical risk factor with road and sub-base applications (Van Praagh et al., 2018).

5.3.2 Cadmium

Cadmium is a highly toxic element found naturally in soil, is prevalent in the environment due to human activities (Genchi et al., 2020b; Kubier et al., 2019; Musilova et al., 2017). Its primary route of human exposure is through the ingestion of contaminated foods (Hellstrom et al., 2007; Hosseini et al., 2013; Perez and Anderson, 2009) and water (Genchi et al., 2020b). The elimination of cadmium from an organism is very slow, leading to irreversible accumulation, primarily in the kidneys and liver when exposed. Prolonged exposure and accumulation leads to kidney disease, fragile bones, and lung damage. Chronic exposure is associated with hypertension, arthritis,

Figure 5.14: *Possible sources of Pb in soil, factors affecting Pb speciation in soil, and its toxic impacts on plant. (Source: Zulfiqar et al., 2019)*

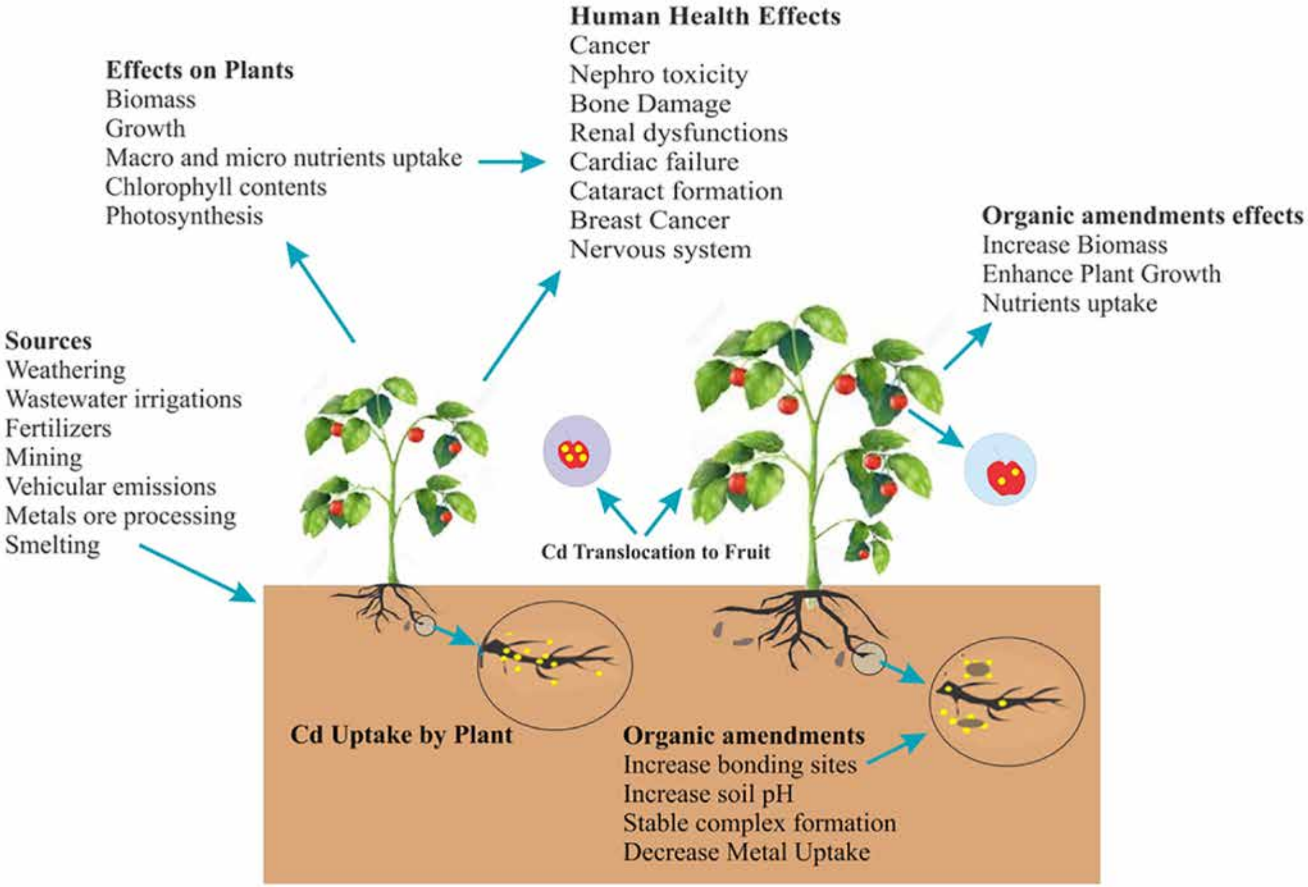


anemia, cardiovascular disease, diabetes, hypoglycaemia, headaches, osteoporosis, and an elevated risk of cancer (Nordberg et al., 2022). According to Ministry of the Environment in Czech Republic (2021b), "cadmium ions are also effective blockers of calcium channels, leading to the interruption of nerve impulse propagation. Cadmium is toxic to reproduction (jeopardizing sperm function and quality, damaging testicular germ cells), interferes with the metabolism of other metals, bone tissue, the immune, and cardiovascular systems. Inhalation exposure to cadmium can cause lung cancer in humans and animals and fetal damage." Furthermore, cadmium adversely affects the female reproductive system (Chen et al., 2015; Ju et al., 2012; Lin et al., 2015a). IARC classifies cadmium and its compounds as carcinogenic to humans (Group 1); (IARC, 2023). In the same study as in Chapter 5.3.1 (Lead), blood samples from residents around an incineration plant showed an average cadmium level of 1.7 $\mu g.L^{-1}$ (with a median of 1.6 $\mu g.L^{-1}$) (Lee et al., 2012). The blood levels of lead and cadmium were slightly higher in the group of the subjects who had resided the longest near the municipal waste incinerators in Korea (Lee et al., 2012).

Accumulation of cadmium from the environment by organisms is very high, hence the accumulation of cadmium in food chains. Mitigating sources of cadmium exposure is crucial for safeguarding human health and preventing associated detrimental effects. Cadmium is frequently detected in urine samples from communities affected by mining (Suta et al., 2020) or metallurgy, and it is also observed in sediments in those areas (Grechko et al., 2021b; Matoušková et al., 2023).

Cadmium from incinerator is released in air and can be found in both bottom ash (Buchholz and Landsberger, 1995; Meima et al., 1999; Klymko et al., 2017 and fly ash. Cadmium concentrations ranged from 30 to 350 ppm in fly ash (Ajorloo et al., 2022). Studies in southern Taiwan found cadmium concentrations between 13 and 92 $mg.kg^{-1}$ in bottom ash and fly ash samples (Bell et al., 2023a).

Figure 5.15: Diagrammatical presentation of Cd sources, uptake by plants, effects on plant growth and human health and how organic amendments reduce Cd uptake by plants and improve plant growth. (Source: Khan et al., 2017)



Meima et al. (1999) studied bottom ash and found that Cd leachability was not affected by pH but with Zn, Cd showed the highest leachability at low pH. As for Cd (and Pb), they leached under slightly acidic conditions and were classified as posing long-term leaching risks. Different quantities of Cd next to Co, Ni and Pb leached out of the bottom ash under different kinds of tests (Kalbe and Simon, 2020).

In the case of cadmium, the spatial distribution of the heavy metal showed neither a marked nor extensive contamination of the sampled area around the incinerator and remained within the typical range of cadmium levels in rural, unpolluted, British soils. Relatively high contents of cadmium, lead, antimony, and zinc in the soils at 250 m and 750–1250 m away from the MSW incinerators were related to MSW incineration, while the elevated contents of the other four HMs (chromium, copper, mercury, nickel) were associated with other anthropogenic activities. In a study conducted in the USA, the concentration of cadmium (and lead) in foliage was found to decrease with distance from the incinerator (Collett et al., 1998). A study in Taiwan found that the metal profiles detected in the air's aerosol were similar to those emitted from the MWI stack (Hu et al., 2003).

5.3.3 Arsenic

Arsenic, occurring naturally and via mining, metallurgy, and coal burning (Bencko, 1984; Bhattacharya et al., 2007; Rasheed et al., 2016), enters the body through inhalation, food, or water affecting gastrointestinal and nervous system (Rahman et al., 2011; Rodriguez et al., 2003). Chronic exposure leads to skin irritation and neurological issues (Chen et al., 2013; Tsai et al., 2003; Tseng et al., 2003). It deposits in the skin and its derivatives (nails, hair), can penetrate the placental barrier, and is primarily excreted through urine. Chronic exposure leads to allergic dermatitis and eczema, often affecting the nervous system (optic nerve degeneration, vestibular system damage), digestive tract, circulatory system, and blood formation. Epidemiological studies have observed increased mortality from cardiovascular diseases. Exposed individuals showed chromosomal aberrations in peripheral lymphocytes (EFSA CONTAM, 2009; Ministry of the Environment of the Czech Republic, 2021b). IARC considers arsenic and arsenic trioxide a human carcinogen, strongly linked to lung and bladder cancer; evidence for other cancers is partial (IARC, 2012). Non-carcinogenic risks include fetal development, children's neurodevelopment, nervous system impact, and heart/vessel diseases.

Arsenic can enter the food chain and is an inhibitor of biochemical reactions. Some fish and shellfish contain elevated levels of As, but in a less toxic organic form. Mass mortality of bee colonies, which are particularly sensitive to As compounds, may be an indicator of environmental contamination by these substances (Ministry of the Environment of the Czech Republic, 2021b).

Arsenic from incinerators is released into the air and can be found in both bottom ash and fly ash. Arsenic concentrations ranged from 25 to 262 ppm in fly ash (Ajorloo et al., 2022). In southern Taiwan, arsenic concentrations in bottom ash and fly ash samples ranged from detection limits to 51 $mg.kg^{-1}$ (Bell et al., 2023a). As was found in the widest range of compounds in bottom ash. An earlier study (Allegrini et al., 2015) used empirical data from leaching tests based on bottom ash obtained from a Danish bottom ash processing plant and with this data they modelled the toxicity impact for metals via three categories: carcinogenic human toxicity, non-carcinogenic human toxicity and freshwater ecotoxicity. While Cr dominated the human carcinogenic impact, As and Zn were more influential in the non-carcinogenic toxicity category.

Figure 5.16: *Arsenic uptake and bioaccumulation in plants. (Source: Bhattacharya et al., 2021)*

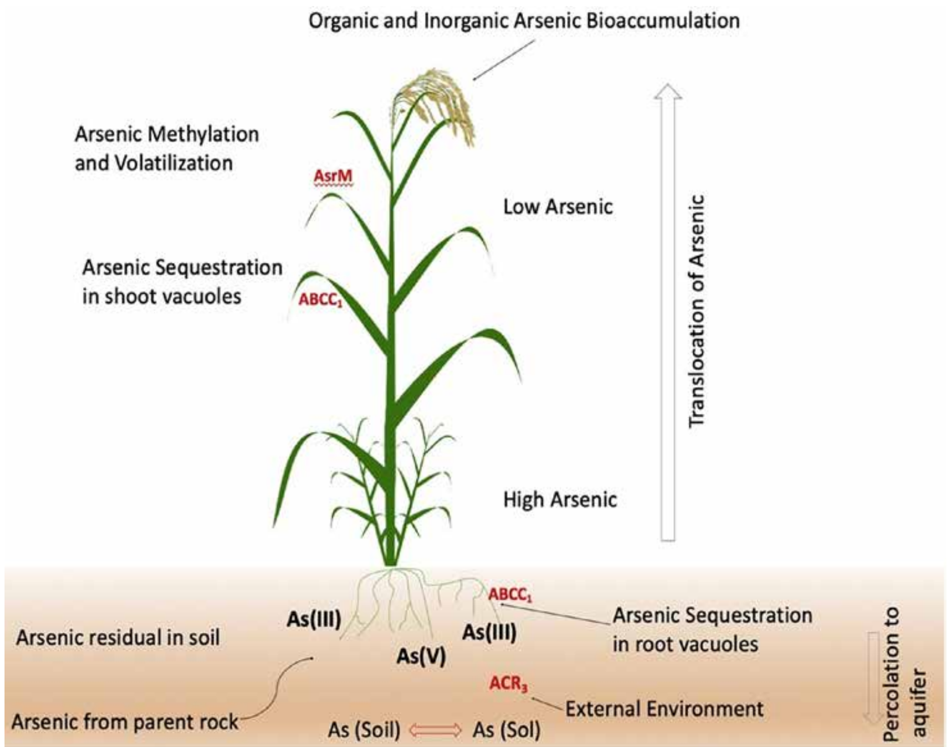
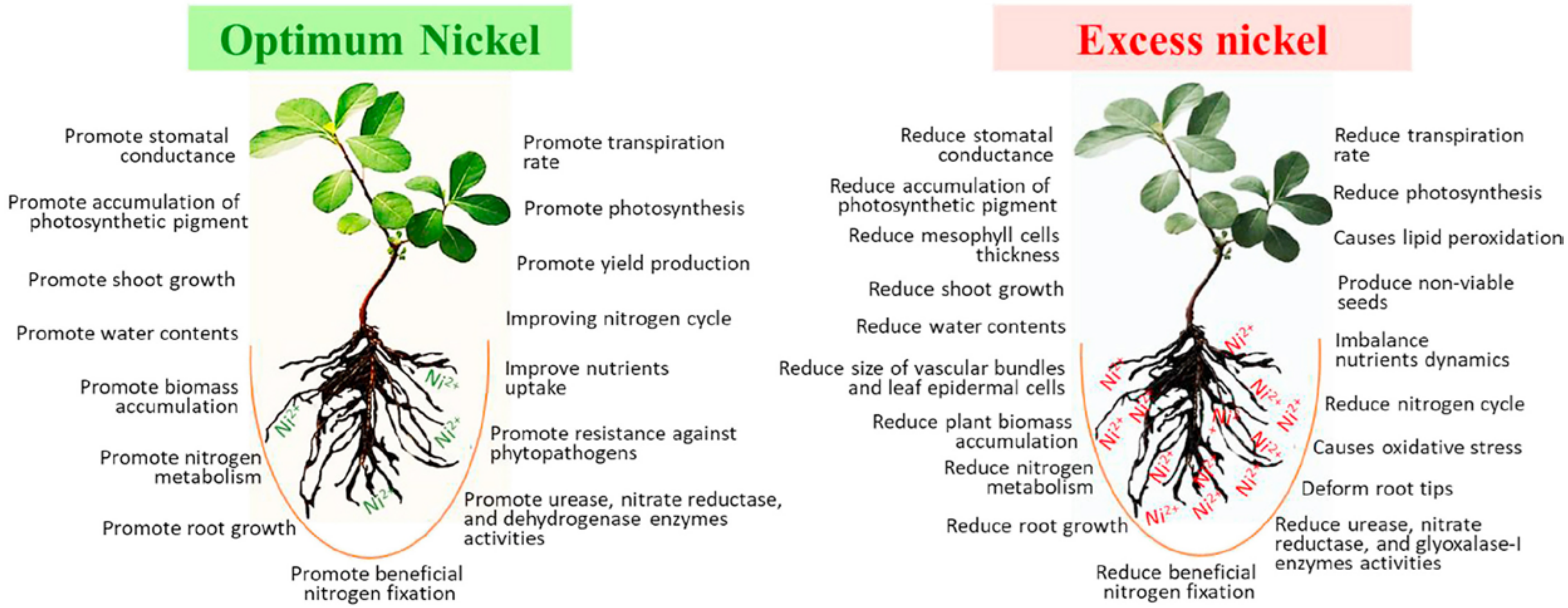


Figure 5.17: *Optimum and excess nickel effects on plants. (Source: Mustafa et al., 2023)*



5.3.4 Nickel

Nickel is a transition element prevalent in the environment from both natural sources and anthropogenic activities, poses risks to human health and the environment. Human exposure to nickel can result in various health issues, including allergies, cardiovascular and kidney diseases, lung fibrosis, and cancers of the lungs and nasal passages (Genchi et al., 2020a). Nickel compounds, classified as Group 1 human carcinogens by the International Agency for Cancer Research (IARC) in 1990 and reaffirmed in 2012 (IARC, 2023), exhibit genotoxic effects. Chronic exposure to nickel, even over weeks, leads to sufficient nickel uptake with persistent effects observed after exposure cessation (Klein and Costa, 2022). Similar to arsenic, nickel passes through the placental barrier and can directly affect prenatal development (Ministry of the Environment of the Czech Republic, 2021g). The toxicity of nickel is associated with mitochondrial dysfunctions and oxidative stress. Additionally, nickel-induced epigenetic alterations have been identified, contributing to genome perturbations (Klein and Costa, 2022). The Toxic Release Inventory (TRI) in the USA was used as an important source of information in a study focused on the association between six environmental chemicals, including nickel and lung cancer incidence in the United States (Luo et al., 2011).

Nickel poses dangers to aquatic organisms, leading to stricter limits in surface waters compared to drinking water (Fernandez-Luqueno et al., 2013; Ministry of the Environment of the Czech Republic, 2021g).

Nickel from incinerators is released in air and can be found in both bottom ash and fly ash. Nickel concentrations ranged from 12 to 203 ppm in fly ash (Ajorloo et al., 2022). The bottom ash also contains nickel. Kalbe and Simon (2020) discovered that it was present in larger quantities in the 0.25 to 45 mm fraction, rather than the finest fraction. In contrast, Vateva and Laner (2020) found the highest levels of Ni, along with cadmium and zinc, in the smallest fraction. Mantovani et al. (2021) found the largest amount of Ni in the 8 to 16 mm size fraction, while Caviglia et al. (2019) in 2-8 mm grain sizes. According to Alam et al., (2019), Ni showed high mobility during oxidising conditions (next to Cr, Cu and Sb).

Nickel in the air can enter soil or water through atmospheric deposition. Plants absorb nickel from the soil through their roots and can accumulate it. Lowering the pH increases nickel mobility and plant uptake (Ministry of the Environment of the Czech Republic, 2021g).

5.3.5 Chromium

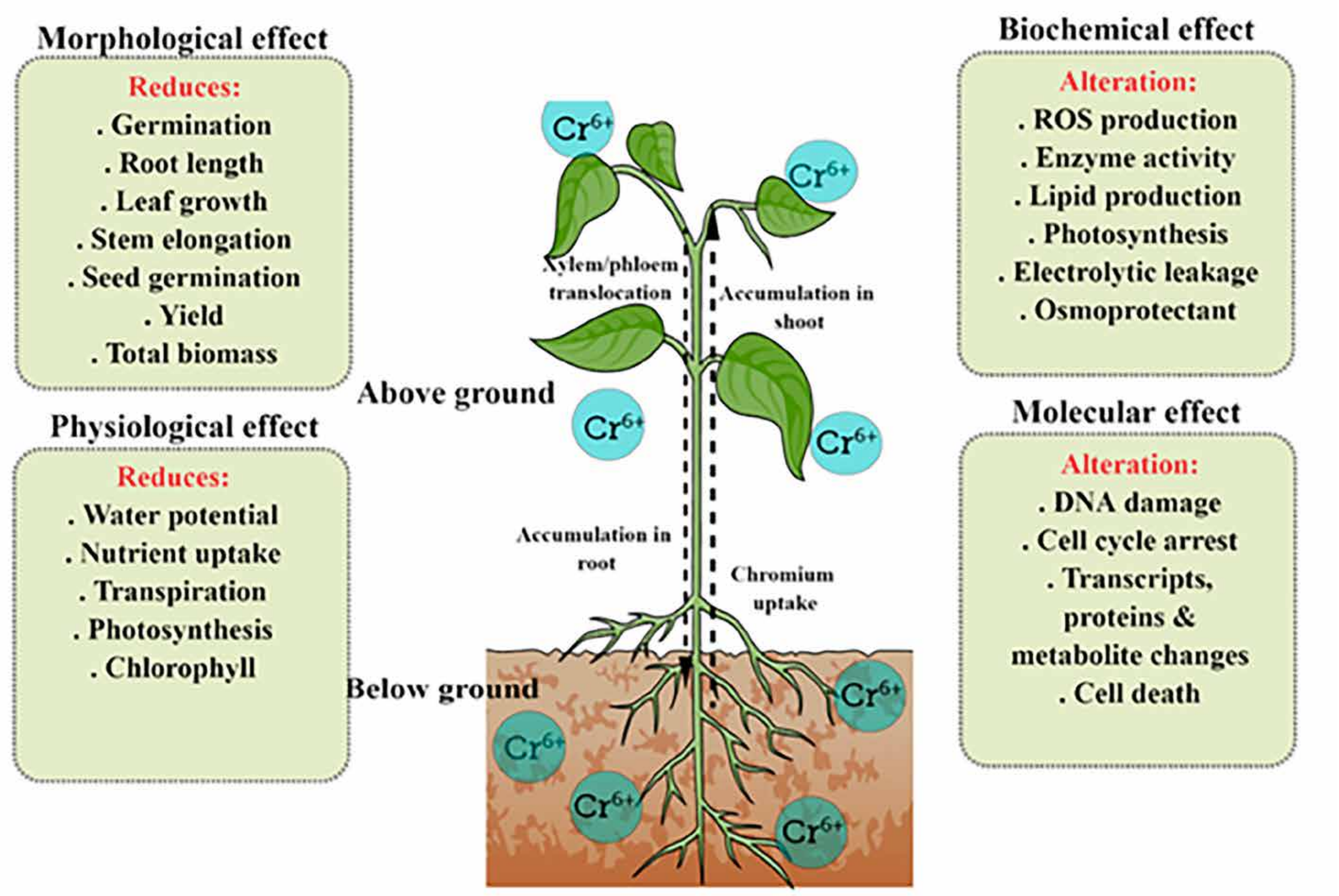
Chromium commonly occurs in two oxidation states, Cr (III) and Cr (VI), with different toxicities. While Cr (III) is an essential trace element, Cr (VI) compounds are toxic (oxidative effect), and their soluble compounds are mutagenic and classified as group 1 carcinogenic to men (IARC, 2023). Chromium (VI) causes skin issues, respiratory problems, including asthma-like symptoms, weakened immunity, and kidney/liver damage, inducing oxidative stress and DNA/protein damage (Guertin et al., 2004; Song et al., 2012). Inhalation of its compounds leads to nasal membrane ulcers, throat irritation, bronchitis, wheezing, and respiratory distress. Remarkably, chromium (III) is vital for human nutrition, found naturally in vegetables, fruits, meats, yeasts, and grains (Anderson, 1997; Pechova and Pavlata, 2007). TRI in USA was used as an important source of information in a study focused on the association between six environmental chemicals including chromium and lung cancer incidence in the United States (Luo et al., 2011).

In the presence of organic substances, Cr(VI) transforms rapidly to Cr(III). Cr(III) strongly binds to soil particles, limiting its solubility in water. Cr (VI) does not bind to soil particles, therefore it is highly mobile in soil and is highly toxic to aquatic organisms. If there is a lack of organic compounds, Cr(VI) can remain stable for a long time even under aerobic conditions. IN anaerobic conditions it is reduced to Cr(III) quickly (Ministry of the Environment of the Czech Republic, 2021a). Chromium does not bioaccumulate. Chromium exists naturally in minerals and is widely used in manufacturing, including metallurgy, textiles, papermaking, and various products like dyes and fertilizers. Its environmental presence stems from landfill leaching, ore extraction, and petroleum/coal combustion (Dellantonio et al., 2008; Jin et al., 2014). Emissions of chromium into the environment, where it can accumulate in soils or sediments, are highly undesirable. From such reservoirs, chromium may be released, even after many years, due to changes in external conditions, causing severe damage and health risks (Ministry of the Environment of the Czech Republic, 2021a).

Chromium from incinerator is released in air and can be found in both bottom ash and fly ash. Its concentrations in fly ash ranged from 55 to 612 ppm (Ajorloo et al., 2022). Vateva and Laner (2020) discovered elevated levels of Cr in larger (>31.5 mm) and smaller (4–8 mm) fractions of bottom ash, with greater concentrations after aging in the fraction > 4 mm. Kalbe and Simon (2020) demonstrated that the smallest fractions did not necessarily contain the most toxic elements, including Cr. Swiss landfill regulations were not met due to Cr concentrations in specific grain size fractions (Glauser et al., 2021). The carcinogenic effects of Cr on humans were mainly attributed to its presence in concrete specimens used as a road sub-base after carbonation (Allegrini et al., 2015). Compliance issues with Danish limit values for Cr were observed in these specimens (Allegrini et al., 2015).

Bottom ash, after subjecting to temperatures of up to 1,000°C, resulted in a significant increase in chromium (Cr) leachate concentrations by two orders of magnitude (Mantovani et al., 2021). Leaching tests conducted according to German standards showed that the limits for Cr were exceeded (Vateva and Laner, 2020). Sequential leach tests (Allam et al., 2019b) highlighted the high mobility of Cr under oxidizing conditions. In batch tests, the legal thresholds for leachate concentrations of Cr in The Netherlands were exceeded (Allam et al., 2019a).

Figure 5.18: *Effect of Cr toxicity (in the form of Cr^{6+} or CrO_4^{2-}) on various morphological, physiological, and biochemical traits in plants. (Source: Ali et al., 2023)*



5.3.6 Mercury

Mercury occurs naturally in various forms, spread through erosion (Ministry of the Environment of the Czech Republic, 2021h), weathering, and anthropogenic sources like combustion processes, coal burning and mining (Sundseth et al., 2017). Mercury is bioaccumulative, and as an element cannot break down. It can pass through the placenta to the fetus and to infants through breast milk. It's particularly dangerous for the youngest, affecting motor function development (walking, speech), causing mental retardation, seizures, cerebral palsy, blindness, and deafness. Inhaling mercury vapor poses significant risks to the nervous (NRC, 2000), immune, digestive, respiratory, and renal systems, with symptoms ranging from neurological disorders to potential fatality (Basu 2023; Tchounwou et al., 2003). In the same study as in Chapter 5.3.1 (Lead), residents around the incineration plant showed an average mercury level of 1.3 $\mu g.L^{-1}$ (with a median of 1.1 $\mu g.L^{-1}$) in blood samples (Lee et al., 2012).

In the aquatic environment, inorganic mercury transforms into highly toxic methylmercury (MeHg), accumulating in fish and shellfish and posing serious health risks upon consumption (Evers et al., 2013; Harris et al., 2003). MeHg adversely affects the nervous, cardiovascular, liver, kidney systems, and disrupts hormones, impacting developing fetuses and inhibiting plant growth (Kumari et al., 2020; Trasande et al., 2016). IARC classifies methylmercury compounds as possibly carcinogenic to humans (Group 2B); (IARC, 2023).

Mercury from incinerator is released in air and can be found in both bottom ash and fly ash. In fly ash, mercury concentrations ranged from 0.4 to 35 ppm (Ajorloo et al., 2022). Stabilized fly ash in one location in southern Taiwan showed mercury concentrations exceeding 7 $mg.kg^{-1}$ of dry matter (Bell et al., 2023a). Mercury should not be present in bottom ash because it has a lower boiling point and should fall out through the grate. However, it has been found along with As, Br, and Cd (Buchholz and Landsberger, 1995; Meima et al., 1999; Klymko et al., 2016).

Mercury leaked from the Megawaste incinerator into the municipal sewage system in Prostějov, Czech republic, in 2003 (MF Dnes & Jurčová, 2003), as consequences of incineration of hazardous waste in this incinerator. Unincinerated waste was the source of that leakage.

Bourtsalas & Themelis (2019) identified major sources of mercury emissions to the atmosphere in the US. The authors reported a significant decrease in mercury emissions from WtE in the last three decades: *"The emissions decreased from 81.8 t in 1989 (as reported by Earth Engineering Center) to about 0.4 t in 2014 (as obtained by authors' own survey of 77 WtE plants)."* However, the authors attribute the observed significant reduction in mercury emissions not to advances in waste incineration technology mainly, but to the reduction of mercury in waste. They suggested that *"The mercury content in MSW decreased from about 1.5 ppm in 2002 to about 0.33 ppm in 2015"* (Thanos Bourtsalas & Themelis, 2019).

In (Lü et al., 2019), the soil around the power plant showed a distinct spatial distribution of Hg, while other heavy metals were less noticeable and evenly spread. Hg levels peaked at 500m downwind from the plant, decreasing gradually with distance and falling below control levels. This Hg likely came from MSWI flue gas diffusion and sedimentation, accumulating in the soil. These findings highlight the need for focused monitoring of Hg pollution in soils near waste incinerators. A study conducted in the surroundings of the MSWI on Samui Island, Thailand observed *"low but elevated levels of Hg (76–275 $\mu g.kg^{-1}$)"* in surface soil and deeper layers (0–40 cm) in the predominant downwind direction of incinerator over a distance of between 0.5–5 km. Soil Hg concentrations measured from a reference/background track opposite of the prevailing wind direction were lower ranging between 7–46 $\mu g.kg^{-1}$. (Muenhor et al., 2009).[45]

45 The MSWI on Samui Island was shut down in the meantime.

Figure 5.19: *Sources and concentration of mercury in the environment. (Source: UNEP, 2024)*

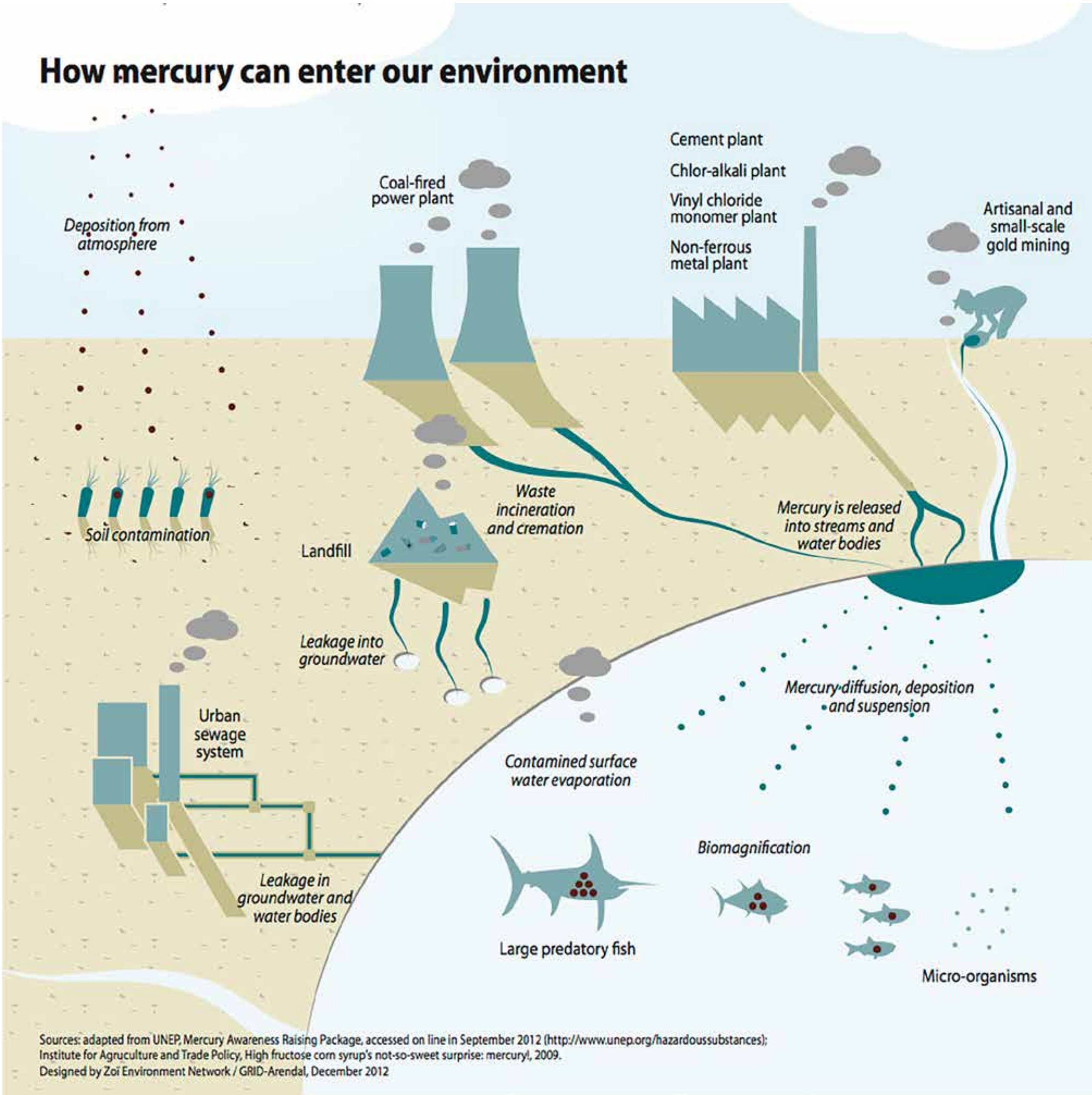


Photo 5.25 *High concentrations of mercury and lead were detected in stabilized fly ash from the Yan Chao site in southern Taiwan (Bell et al., 2023a). Photo: Tainan Community University.*

Deng et al. (2016) examined the blood mercury levels of 35 incinerator workers and 269 nearby residents exposed to the incinerator's emissions, along with 143 control subjects. They found elevated levels of mercury in both the incinerator workers and the exposed residents compared to the control group (see also Chapter 6).

5.3.7 Copper

Copper is among commonly used metals, also being a vital element for the human body, crucial for functions such as hormone secretion, nerve conduction, electron transfer, bone and connective tissue growth, and red blood cell synthesis. Despite its small quantity (50–120 mg) in the body, copper plays a critical role in various biochemical processes and its deficiency in adults can lead to blood and nervous system disorders (Ackah et al., 2014; Medeiros et al., 2012; Saracoglu et al., 2009). However, excessive copper intake can lead to health issues such as inflammation in the brain tissues, fatigue, hair loss, allergies, and even serious conditions like kidney dysfunction and cancer (Sobhanardakani et al., 2018).

It's released into the atmosphere during copper ore mining and processing and the combustion of fossil fuels and waste. Besides being ingested in food or water, copper can also enter the body through inhalation. Exposure to copper dust in the air may cause nasal and eye irritation, headaches, numbness, and diarrhea. Inhalation of copper dust can also cause flu-like illness symptoms, including metallic taste in the mouth, alternating fever and chills, chest tightness, and coughing (Ashish et al., 2013; Karalliedde and Brooke, 2012). Environmental impacts highlight that copper, while essential for animals and plants, can become toxic to aquatic organisms in higher concentrations (Hossain and Rakkibu, 1999). The adverse impact of Cu on freshwater ecotoxicity was highlighted in concrete specimens used as a road sub-base (Allegrini et al., 2015).

The presence of Cu and antimony (Sb) in waste incinerator was noted to create catalysis, accelerating unfavorable reactions that form chlorinated and brominated dioxins (Ebert and Bahadir, 2003; Weidlich, 2021).

Mercury from incinerators is released in air and can be found in both bottom ash and fly ash. Copper concentrations in fly ash ranged from 98 to 2,794 ppm (Ajorloo et al., 2022). None of the bottom ash samples met Swiss regulations for landfill due to total concentrations of Cu in certain grain size fractions (Glauser et al., 2021). Caviglia et al. (2019) found Cu concentrations exceeding Italian limit values for fractions below 10 mm.

For bottom ash, statistically significant correlations in various leaching methods were only observed for copper (Cu) when using deionized water as an eluent (Glauser et al., 2021). In Dutch column leaching tests, Cu was reported, with 62 % of the samples failing for Cu (Glauser et al., 2021). Allam et al. (2019a) demonstrated that leachate concentrations of Cu would have exceeded legal thresholds in The Netherlands for building aggregate use. Cu leachate concentrations were higher in the presence of dissolved organic matter, even under alkaline conditions (Glauser et al., 2021), but increased mobility of Zn and Cu below pH 8.5 was confirmed (Tiberg et al., 2021). After exposure to high temperatures, leachate concentrations of Cu in bottom ash were reduced (Caviglia et al., 2019). The presence of organic matter and ageing contributed to leaching issues, with short-term releases of high quantities of various substances, including Cu (Glauser et al., 2021; Kalbe and Simon, 2020).

5.3.8 Zinc

Similar to copper, zinc is essential for living organisms. Humans predominantly acquire it through food. However, excessive intake acutely causes gastrointestinal disorders and chronic damage to blood or the pancreas. Low zinc intake leads to growth and developmental disorders, with additional zinc intake being crucial for pregnant women. It's essential for healthy sexual development. Insufficient zinc in food causes unwanted weight loss, slow wound healing, impaired memory, sensory disorders (especially vision and smell), stunted growth, and mental lethargy. While not a significant risk to human health, chronic consumption of large amounts of zinc can increase the risk of heart disease and affect the immune system (Ministry of the Environment of the Czech Republic, 2021f; Nriagu, 2007). The first type of well-studied toxic reactions to zinc in human beings was "metal fume fever" induced by intense inhalations of industrial fumes containing zinc oxide. The most prominent respiratory effects of metal fume fever include fever, chills, gastroenteritis, substernal chest pain, and cough (Nriagu, 2007).

Figure 5.20: *Effect of deficiency and excess of copper on human and plants. (Source: Wołowicz and Hubicki, 2020)*

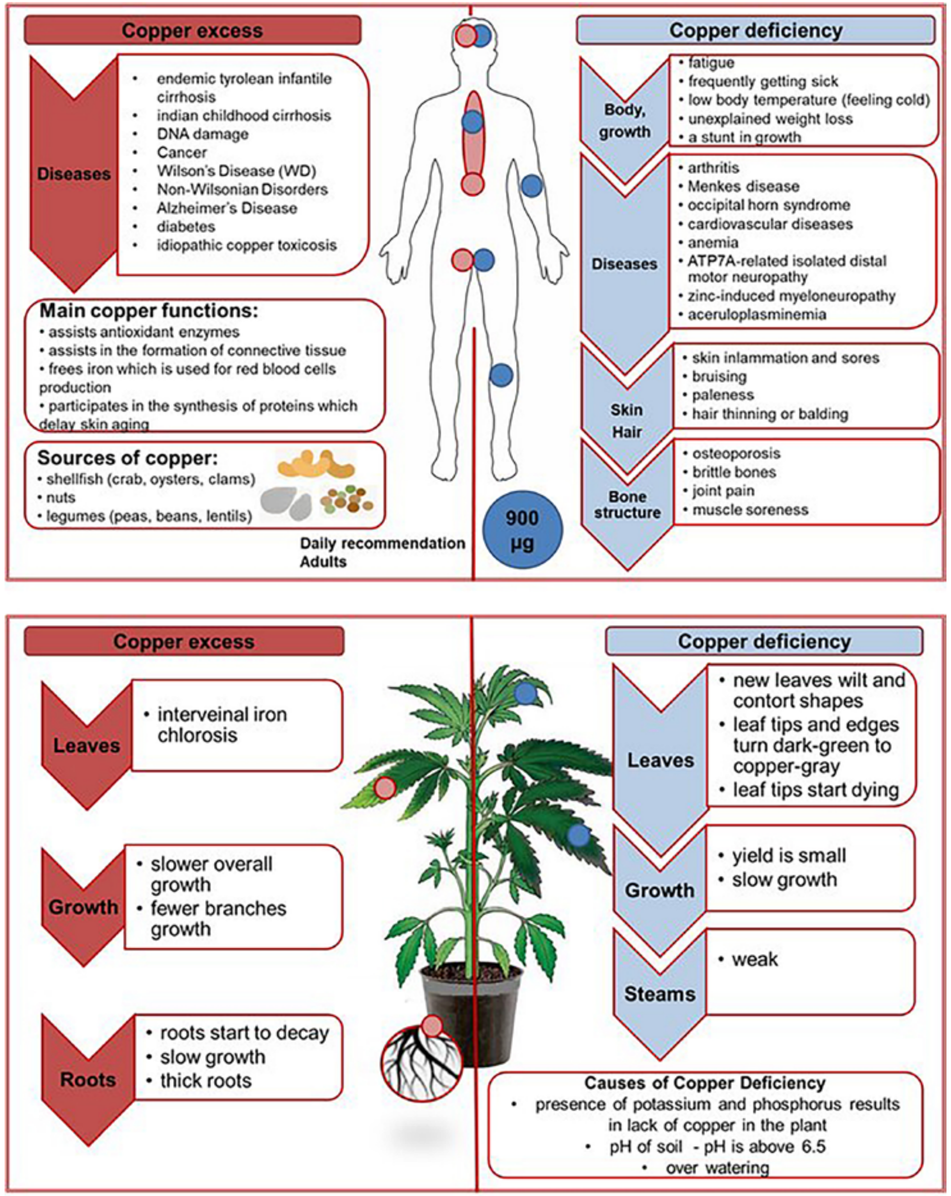


Photo 5.26: *High concentrations of zinc were measured in 2010, for example in the Elbe River below Lovosice (Havel et al., 2011). Photo: Milan Havel, Arnika.*

Zinc is considerably toxic to fish and other aquatic organisms, particularly sensitive are salmonid fishes. It dissolves minimally in water and typically binds to soil particles (Ministry of the Environment of the Czech Republic, 2021f; Rainbow and Luoma, 2011; Skidmore, 1964). As for humans, zinc is essential for plants but too much zinc affects plants' health (see Figure 5.21); (Kaur and Garg, 2021).

Zinc from incinerators is released in air and can be found in both bottom ash and fly ash. Zinc concentrations in fly ash ranged from 18 to 22,000 ppm (Ajorloo et al., 2022). In bottom ash and fly ash from locations in southern Taiwan, zinc was found at concentrations of 3,500 to almost 10,000 $mg.kg^{-1}$ (Bell et al., 2023a). Zinc can easily leak from the slags (Jin et al., 2014).[46]

Vateva and Laner (2020) measured Zn in the smallest fractions of bottom ash. Caviglia et al. (2019) found that the highest concentrations of Zn with Cu, Ni, Pb and Sr were in the medium grain size range (2 - 8 mm).

Secondary treatment (exposing bottom ash to up to 1,000°C) has been investigated by some authors, but its overall benefits and associated climate and cost impacts are questionable. Caviglia et al. (2019) found that after exposing bottom ash to temperatures up to 1,000°C, leachate Zn concentrations were reduced. Studies by Allam et al. (2019b) showed that leaching of certain elements often exceeded limits for non-isolated applications, and sequential leaching tests revealed high mobility of zinc (Zn), especially under low pH conditions. Tiberg et al. (2021) confirmed the increased mobility of Zn below pH 8.5. Glauser et al. (2021) observed significant differences in leachate concentrations when the batch test eluent was changed to a lower pH using CO_2-saturated water, with Zn mobility increasing 15-fold compared to the deionised water eluent. The authors noted a high buffering capacity in smaller fractions in the presence of CaO, resulting in transient stability. Meima et al. (1999) found that Zn leachability was highest at low pH. In a previous study, Allegrini et al. (2015) used empirical data from leaching tests on bottom ash and found Zn to be more influential in the non-carcinogenic toxicity impact.

46 Over the past few decades, zinc smelting activities in Guizhou, China have produced numerous slag dumps, which are often dispersed on roadsides and hill slopes throughout the region. During periods of acid rain, these exposed slags release heavy metals into surface water bodies. A column leaching study was designed to test the potential release of the heavy metals cadmium (Cd), chromium (Cr), copper (Cu), lead (Pb), and zinc (Zn) under simulated acid rain events. ….. Reaction rates (release amounts of heavy metals in certain period of leaching) of heavy metals in the leachates demonstrated the sequence of Zn>Cr>Cd, Cu>Pb. Leaching release of heavy metals was jointly affected by the pH of leaching solution and mineral composition of slags (including chemical forms of Cd, Cr, Cu, Pb, and Zn); (Jin et al., 2014).

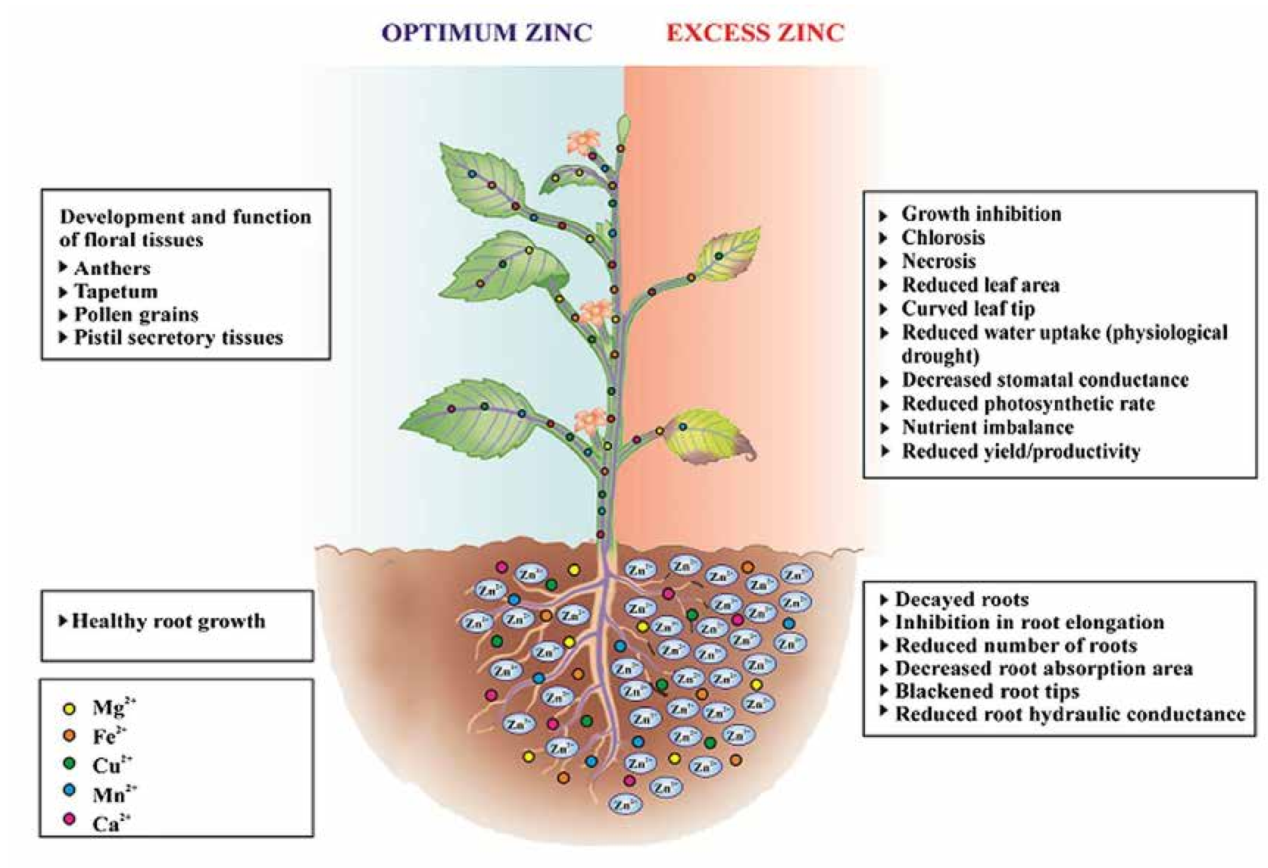


Figure 5.21: *Zinc toxicity for plants. (Source: Kaur and Garg, 2021).*

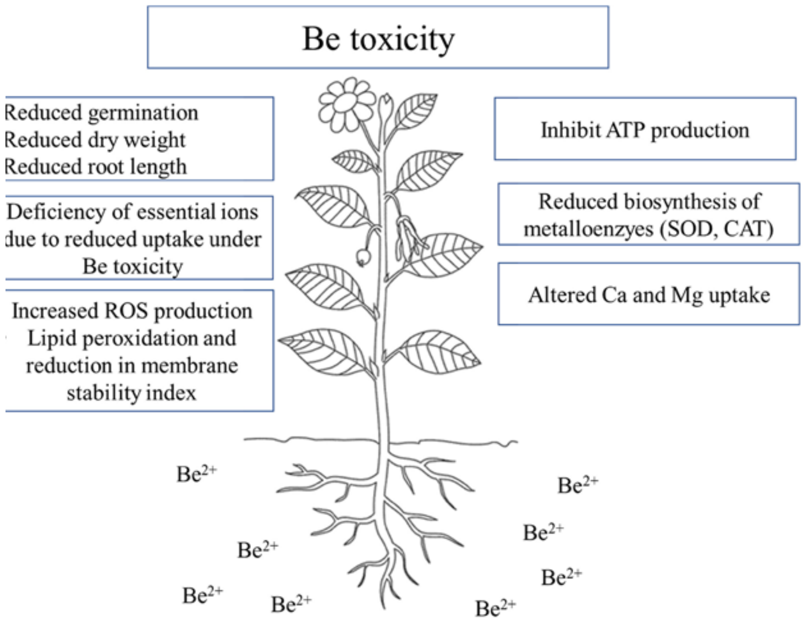
Relatively high contents of zinc, cadmium, lead and antimony in the soils at 250 m and 750–1,250 m away from the MSW incinerators were related to MSW incineration, while the elevated contents of the other four HMs (chromium, copper, mercury, nickel) were associated with other anthropogenic activities (Li et al., 2019).

5.3.9 Beryllium

Lung diseases associated with beryllium exposure have been recognized and studied since the early 1940s. Despite reduced workplace exposure, chronic beryllium disease continues to occur (NRC, 2008). Furthermore, beryllium has been classified as a human carcinogen category 1 (IARC, 2023).

The US EPA has set limits for beryllium in outdoor air at 0.01 µg/m³ as a 30-day average (Wambach & Laul, 2008). People are primarily exposed to beryllium through inhalation but also through food or drinking water (Bolan et al., 2023). Major sources of beryllium emissions include coal combustion, other fossil fuel processing, and waste incineration (Bolan et al., 2023; Taylor et al., 2003), specialized metal production, or ceramics manufacturing (Taylor et al., 2003).

Figure 5.22: *Beryllium toxicity. (Source: Bolan et al., 2023)*



It can concentrate in soils around incineration plants through transmission. One study from the late 1990s identified beryllium in soils around a waste incineration facility in Barcelona (Meneses et al., 1999).

5.3.10 Limits for heavy metals in waste from incinerators

Huber et al. (2019) study the use of bottom ash and related regulations in the EU, including Norway and Switzerland. The study assumes that the 463 incinerators in this region produce 17.6 million tons of bottom ash annually, with approximately half being used in road construction. Since there is no European-level regulation for the use of bottom ash, the conditions for its use vary at the national level. According to 2014/955/EU, residues from waste incineration are classified as waste. It is important to determine if they are hazardous waste under 1357/2014 and if they contain POPs under 850/2004.

Out of the 22 countries studied, 16 allow the use of the mineral fraction of bottom ash, while the remaining countries dispose of this waste in landfills. In Portugal, the use of bottom ash is assessed on a case-by-case basis. However, only 11 out of the 16 countries actually utilize bottom ash, with usage rates ranging from 20% to 100%. Most countries have legislative regulations for this purpose, while only four have non-binding guidelines (Austria, Germany, Sweden, and the UK). The countries that only landfill bottom ash are Estonia, Hungary, Ireland, Luxembourg, Slovakia, and Norway. These countries generate relatively small quantities of bottom ash, ranging from 28,000 to 250,000 t per year. Ireland and Luxembourg both allow exports. The only bottom ash treatment facility in Ireland exports to the Netherlands, while the only incinerator operator in Luxembourg exports to Germany. In the remaining countries, bottom ash is only landfilled. In Switzerland and Lithuania, using bottom ash outside of landfills is not common due to strict limits and lack of testing by companies for construction purposes. Finland has recently introduced limits, while Austria does not require the substitution of primary materials (Huber et al., 2019).

Huber et al. (2019) compared the limit values for countries that allow the use of bottom ash with those for inert waste as defined by the EU. In the case of inert waste, 18 indicators are monitored, mainly inorganic substances. Of these, 12 are commonly monitored in the mineral fraction of bottom ash. The limit for Pb is identical in 7 out of 17 states, while for Hg it is 5 out of 17, for As and Ni it is 4 out of 17, for Cr(total) it is 3 out of 17, for Cd it is 2 out of 17, and for Cu, Mo, Sb, Zn, and Cl-, it is 1 out of 17. Each state surveyed has its own limits for sulphides. The most common hazardous properties, according to Klymko et al. (2017), are HP 14 Reprotoxic and HP 10 Ecotoxic. Table 5.13 compares leaching limits (not total content) of selected countries with EU leaching limit values for disposing of in landfill for inert waste.

Table 5.13: *Comparison of leaching limit values for MIBA utilisation and EU leaching limit values for disposing of in landfill for inert waste. Values = 1: leaching limit value for MIBA utilisation matches exactly limit value for landfill for inert waste(cell colour yellow), values < 1: leaching limit value for MIBA utilisation is stricter than limit value for landfill for inert waste (cell colour yellowish to green), values > 1: leaching limit value for MIBA utilisation is less strict than limit value for landfill for inert waste (cell colour yellowish to red). Cells containing (-): no leaching limit value for MIBA utilisation is defined for the respective parameter. Factors determined for Portugal are based on an individual permit issued by Portuguese authorities. percolation test (perc.); limit value (LV). (Source: Blasenbauer et al., 2020)*

LV utilisation / LV inert waste landfill

	Austria	Belgium (Wallonia)	Denmark	Denmark	Finland	Finland	Finland	Finland	France	France	Germany	Italy	Lithuania	Poland	Portugal	Spain (Catalonia)	Sweden
	Base layer	Regular quality assurance test (base layer and possibly hydraulic bound material)	Category 1&2	Category 3	Roadway Covered	Roadqay Paved and Subgrade filling in industrial or storage building	Field Covered	Field Paved	Type 1	Type 2	Z2	Utilisation	Civil Engineering purposes	Sub-base of roads and highways	Aggregates for unbound and hydraulically bound materials for use in civil engineering work and road construction	Road subbase, Levelling of terrain and embanksment, Filling and restoration of degradable areas from extractive activities, others	Less than little risk (general use) - unbound material
As	1	2	0.16	1	2	4	1	3	1.2	1.2	-	1	-	2	10	2	0.18
Cd	-	25	0.13	2.7	1	1.5	1	1.5	1.3	1.25	13	1.3	0.75	2	50	25	0.5
Cr (total)	1	-	0.1	5	4	20	1	10	4	2	4	1	4	2	40	-	2.4
Cu	2	10	0.1	4.4	5	5	1	5	25	25	1.5	0.25	0.75	2	25	10	0.4
Hg	-	20	0.067	0.67	3	3	1	3	1	1	1	1	0.1	2	50	-	0.5
Mo	2	3	-	-	3	12	1	12	11	11	-	-	-	2	20	-	-
Ni	1	5	0.1	0.7	5	5	1	3	1.3	1.3	1	0.25	1	2	25	-	0.92
Pb	1	4	0.1	1	1	4	1	4	3.2	2	1	1	1	2	20	10	0.34
Sb	5	33	-	-	12	12	5	12	12	10	-	-	-	1	12	-	-
Zn	-	2.3	0.1	1.5	3.8	3.8	1	3	13	13	0.75	7.5	0.75	2	13	5	0.97
Chloride	4	6.3	0.55	11	4	14	1	3	13	6.3	3.1	2.5	13	2	63	-	0.16
Sulphate	5	10	0.89	14	5.9	18	1.2	10	10	5	6	2.5	20	2	20	-	0.046

Table 5.14: *Maximum limit values as total concentration (273/2021 Sb.), concentration in leachate (273/2021 Sb) in comparison with inert waste acceptable at EU landfills (2003/33/EC) in mg.kg-1.*

Parameter	Total concentration [$mg.kg^{-1}$ dm]	Leaching limit value [$mg.L^{-1}$] at 10 $l.kg^{-1}$	Inert waste acceptable at EU landfills (EU, 2003), batch test by 10 $l.kg^{-1}$ [$mg.kg^{-1}$ dry substance]
As	45	0.03	0.5
Cd	20	0.005	0.04
Cu	7,000	1	2
Hg	1	0.0008	0.01
Ni	500	0.03	0.4
Pb	1,000	0.05	0.5
Zn	10,000	0.6	4
Chloride	-	700	460
Fluoride	-	6	10
Sulphate	-	1,000	1,000
Ba	-	3	20
Cr (total)	-	0.2	0.05
Mn	-	0.3	-
Na	-	400	-
Mo	-	0.5	-
Sb	-	0.07	0.06
Se	-	0.1	0.1
V	-	0.3	-

Photo 5.27: *Storage of residues from the Covanta incinerator (so-called monofill) in Haverhill, Massachusetts, USA. (Source: Connett, 2013)*

5.3.10.1 Limits for heavy metals in the Czech Republic

In the Czech Republic, bottom ash must meet certain criteria regarding the content of selected metals and several groups of organic substances to be used for landfilling purposes according to decree 273/2021 Sb. The concentrations of these substances (excl. organics) are differentiated in the leachate ($mg.L^{-1}$) and in the dry matter ($mg.kg^{-1}$). An overview is presented in Table 5.14. Other requirements such as grain size, methods of use, and pH values and additional requirements as ecotoxicity testing (performed four times a year) are listed in the appendices of the same decree (Ministry of the Environment of the Czech Republic, 2021e).

5.4 Particulate Matter (PM)

The effect of these particles depends on their size, shape, and chemical composition. Their size is crucial for penetration and subsequent deposition in the respiratory system. Larger particles (PM_{10}) are mostly trapped in the upper respiratory tract, finer ones ($PM_{2.5}$) reach the bronchioles, and the finest can even enter the pulmonary alveoli. The effects are influenced by what the particles carry on their surface, generally having a wide range of effects on the cardiovascular and respiratory systems. They irritate the respiratory tract lining, alter the structure and function of cilia tissue, increase mucus production, and reduce the self-cleaning function of the respiratory system, facilitating infections. Repeated exposure can lead to chronic bronchial inflammation and chronic obstructive pulmonary disease, leading to overload of the right ventricle and circulatory failure, which is influenced by other factors (the body's immune system, allergic predisposition, smoking, exposure to occupational substances); (Ministry of the Environment of the Czech Republic, 2021b). These particles are associated with asthma, reduced lung function, and other respiratory problems, heart rhythm disorders, and increased mortality. In 2016, they were classified as known human carcinogens, Group 1 (IARC, 2023).

According to Howard's study (2009), modern incinerators in the European Union were a major source of emissions of ultrafine particles with a diameter equal to or less than 0.1 μm ($PM_{0.1}$), although one of the later studies questions the significant impact of nanoparticles from municipal waste incineration on human health and relies mainly on gaps in knowledge about their effects (Johnson, 2016). Some studies did not confirm waste incinerators as the main source of ultrafine particles in the air (Ragazzi et al., 2013). However, another study pointed out maintenance and other irregularities in the operation of large waste incinerators and small waste

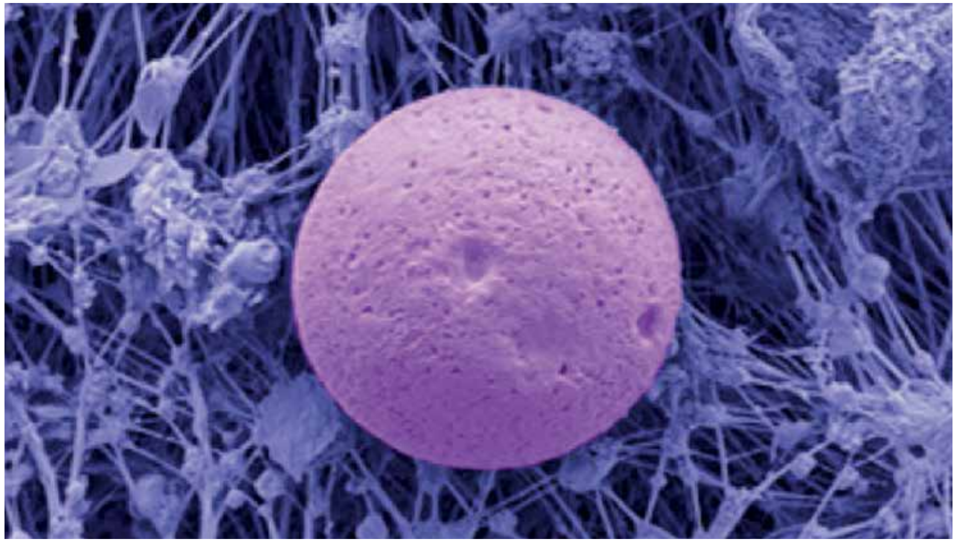


Photo 5.28: *Colored scan of dust particles from an electron microscope. Source: Czech Hydrometeorological Institute (ČHMÚ).*

incineration plants as potentially problematic sources of ultrafine particles (Walser et al., 2012). One study highlighted an interesting aspect in monitoring sources of ultrafine particles in the air: "The results of measuring flue gases from incinerators and atmospheric sampling at ground level near incinerators show that typical concentrations of ultrafine particles in flue gases are generally similar to concentrations in urban air, and subsequently, after the scattering process, the incinerator fumes are diluted. In the surrounding air, concentrations of ultrafine particles are typically indistinguishable from those that would occur in the absence of the incinerator" (Jones & Harrison, 2016). Thus, ultrafine particulate matter in the air can be added to the list of controversial topics regarding the influence of incinerators, but they should certainly be monitored in their emissions and should have a set limit for them similar to other sources.

6. Impacts of Incinerators on Human Health

In 2001, a scientific team from Greenpeace released a study titled "Incineration and Human Health" (Allsopp et al., 2001). This study reached the following conclusions: *"The research carried out on environmental contamination and human exposure to pollutants released by incinerators is limited and has focused mainly on dioxins and heavy metals. Research has demonstrated that both older and more modern incinerators can contribute to the contamination of local soil and vegetation with dioxins and heavy metals. Similarly, in several European countries, cow's milk from farms located in the vicinity of incinerators has been found to contain elevated levels of dioxins, in some cases above regulatory limits.*

Populations residing near to incinerators are potentially exposed to chemicals through inhalation of contaminated air or by consumption of contaminated agricultural produce (e.g. vegetables, eggs, and milk) from the local area and by dermal contact with contaminated soil. 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. Two studies in the Netherlands and Germany, however, did not find increased levels of dioxins in body tissues of residents living near incinerators. At an incinerator in Finland, mercury was increased in hair of residents living in the vicinity, most likely due to incinerator releases. Children living near a modern incinerator in Spain were found to have elevated levels of urinary thioethers, a biomarker of toxic exposure. Elevated levels or more frequent occurrence of certain PCBs occurred in the blood of children living near a hazardous waste incinerator in Germany.

Several studies have reported elevated levels of dioxins (total TEQ), and/or certain dioxin congeners, in the body tissues of individuals employed at both modern and older incinerators. This is thought to be a consequence of exposure to contaminated ashes in the workplace. Similarly, some studies have reported increased levels of chlorinated phenols, lead, mercury and arsenic in the body tissues of incinerator workers.

Experimental data confirm that incinerators release toxic substances and that humans are exposed as a consequence. Studies on workers at incinerator plants, and populations residing near to incinerators, have identified a wide range of associated health impacts (e.g. cancer, heart and respiratory diseases, elevated mutagens or thioethers in urine etc.). These studies give rise to great concerns about possible health impacts from incinerators even though the number of studies (particularly those that have been conducted to appropriately rigorous scientific standards) is highly limited. These should be seen, however, as strongly indicative that incinerators are potentially very damaging to human health" (Allsopp et al., 2001).

The cited study sparked a significant wave of counter-studies, some of which were promoted by industry associations of waste incineration operators such as CEWEP (Confederation of European Waste-to-Energy Plants), while others emerged as genuinely independent studies. In the following paragraphs, we will attempt to summarize the results of more recent studies.

There have been numerous epidemiological studies on the health effects of waste incineration, yet their results vary considerably. Generally, they agree that there is insufficient data to assess the issue and that further research should be conducted. In a review article on this topic (Negri et al., 2020), information on "third-generation" incinerators was gathered, revealing only short-term results, with the impact, particularly on chronic diseases, remaining uncertain. Another scholarly article on this subject (Tait et al., 2020) found several adverse effects of waste incineration on health, including significant associations with certain cancers, birth defects, infant deaths, or miscarriages, but noted fewer harmful effects in newer incinerators. However, this might be due to adverse effects not having manifested yet in these facilities.

Some studies confirmed increased risks of various cancers in the vicinity of waste incinerators (Elliott et al., 2000; Elliott et al., 1996; Franchini et al., 2004; Salerno et al., 2016; Salerno et al., 2015; Starek, 2005) or elevated incidences of non-Hodgkin's lymphoma near waste incinerators as the sole dominant source of PCDD/F (Bianchi & Minichilli, 2006; Floret et al., 2007; Floret et al., 2003; Floret et al., 2004). According to one study (Floret et al., 2004), the risk of soft tissue sarcoma did not significantly increase in a specific area near a waste incinerator, but another research team confirmed the opposite at the same location (Viel et al., 2008; Viel et al., 2011). Similarly, another Italian study (Zambon et al., 2007) concluded similar findings, examining various sources (Minichilli et al., 2016; Romanelli et al., 2019) of PCDD/F (waste incinerators and other industrial sources) and their impact on the health of people living nearby.

Fly ash generated from MSWIs often contains harmful PCDD/Fs, posing health risks to workers involved in its recycling and disposal. In a study focusing on fly ash from an MSWI in Southern Taiwan, several key findings emerged. Monte Carlo simulation revealed significant carcinogenic and non-carcinogenic risks for onsite workers, surpassing established threshold limits, indicating a pressing need for improved risk management strategies. Sensitivity analysis identified concentration and exposure duration as critical parameters in assessing both carcinogenic and non-carcinogenic risks, suggesting avenues for targeted intervention and mitigation measures. These findings underscore the importance of implementing effective health risk management strategies for onsite workers involved in waste incineration plants (Hsieh et al., 2018).

In a separate ecological study examining municipal cancer mortality between 1997-2006: The research uncovered excess cancer mortality in populations residing near industrial installations, particularly incinerators and facilities handling scrap metal and end-of-life vehicles. Elevated risks were observed for various cancers, including tumors of the pleura, stomach, liver, kidney, ovary, lung, leukemia, colon-rectum, and bladder, emphasizing the broad impact of industrial installations on public health. These findings lend support to the notion of a significant increase in cancer mortality risk in towns near incinerators and hazardous waste disposal facilities (Garcia-Perez et al., 2013).

Further studies focused on increased occurrences of respiratory diseases in women (Minichilli et al., 2016; Romanelli et al., 2019) or men (Golini et al., 2014). Maternal exposure to emissions from waste incinerators in Italy was associated with premature births (Candela et al., 2013), while in England and Scotland, Parkes et al. (2020) observed small but increased risks of congenital anomalies associated with the proximity to municipal waste incinerators. In Italy, an increase in PM_{10} from municipal waste incinerators was linked to increased miscarriage risks (Candela et al., 2015). A study in Japan concluded that the proximity of municipal waste incinerators to schools might be associated with hoarseness, headaches, stomach pain, and fatigue in school-age children (Miyake et al., 2005).

However, there have also been studies that did not confirm these problems (Federico et al., 2010; Fukuda et al., 2003; Hu & Shy, 2001; Ranzi et al., 2014; Thabuis et al., 2007). Therefore, it is necessary to continue research efforts and, most importantly, to approach each potential incineration facility in line with the precautionary principle. Negative impacts of waste incineration have certainly not been ruled out.

Study by de Titto and Savino (2019) promoted by CEWEP (CEWEP, 2020) has: *"… found no studies indicating that modern-technology waste incineration plants, which comply with the legislation on emissions, are a cancer risk factor or have adverse effects on reproduction or development"* (de Titto & Savino, 2019). The study also lists several factors in favor of that affirmation: *"(a) the emission levels of the plants currently built in the developed countries are several orders of magnitude lower than those of the plants in whose environments epidemiological studies have been carried out and which have found some kind of negative association in terms of health; (b) risk assessment studies indicate that most of the exposure is produced through the diet and not by a direct route; and (c) monitoring dioxin level studies in the population resident in the environment of incineration plants did not reveal increases of these levels when compared with a population living in reference areas."* (de Titto & Savino, 2019).

The study also emphasizes the need to implement "an emissions monitoring program to ensure the prevention of environmental damage" de Titto and Savino (2019). It should be added, however, that the study did not examine whether the assumption of rapid emission reductions is based on sufficient measurements. In fact, the establishment of an emission monitoring program is essential to objectively assess the real environmental impact of a given waste incinerator and to calculate the real emissions of substances such as dioxins and/or mercury (see Chapter 3.1.1.1). In a sense, this statement points to a common gap in the assessment of the health impacts of waste incinerators, as the food exposure pathway is often neglected. Contamination also occurs through improper management of incineration residues or APC residues.

However, it is necessary to add that the study did not examine whether the assumption of a rapid reduction in emissions is based on sufficient measurements. Moreover, the establishment of an emissions monitoring program is indeed a crucial prerequisite for an objective assessment of the real impact of each waste incinerator on the environment and for the calculation of actual emissions of substances such as dioxins or mercury (see Chapter 3.1.1.1). In a way, the affirmation "*(b) risk assessment studies indicate that most of the exposure is produced through the diet and not by a direct route*" (de Titto & Savino, 2019) points out a common gap in the assessment of the effects of waste incinerators on health because exposure through food pathways is often overlooked. Their contamination also occurs through improper handling of residues from incineration or APC residues (Air et al., 2003; Katima et al., 2018; Pless-Mulloli et al., 2000).

For most persistent organic pollutants (especially dioxins), the primary exposure pathway is food intake, particularly animal fats (Parzefall, 2002; Schecter et al., 2006). Neglecting this exposure pathway, especially for dioxins, does not make sense. Its inclusion in the assessment of waste incineration impacts on human health is often missing. Several studies recommend including exposure through locally grown foods (Ma et al., 2002; Nouwen et al., 2001).

A new study from the vicinity of an incinerator in Turin showed that local farmers had higher concentrations of PCDD/Fs and dl PCBs in their blood serum compared to the rest of the population living near the incinerator (Iamiceli et al., 2021). Similar conclusions were drawn from an older study in Flanders (Nouwen et al., 2001). The latest study that critically examined the health assessment of populations living near waste incinerators states, "... long-term consumption of food produced in an area affected by emissions from the incinerator may increase dioxin internal burdens in the population" (Campo et al., 2019). It confirms that the exposure pathway through domestically raised animals as food sources can lead to increased dioxin exposure in the vicinity of incinerators. It could be the farmers who consume their own cultivated products to a much greater extent, being more significantly affected by incinerator operations.

Increased levels of PCBs were observed in the soil and plants in the vicinity of both IWI and MSWI in Poland. The highest accumulation of PCBs was found in plants with large leaf area. Around the municipal waste incineration plant, these were Tanacetum vulgare leaves (12.45 $ng.g^{-1}$), and around the industrial waste incineration plant-grasses (4.3 $ng.g^{-1}$); (Gabryszewska & Gworek, 2020).

As part of the monitoring conducted by the IPEN network, 26 mixed samples from chickens raised near waste incinerators in 12 countries were analyzed.[47] The dioxin content in eggs ranged between 2.6 and 234.4 pg $TEQ.g^{-1}$ fat (PCDD/F and dl PCB), and in 24 out of 26 flocks living near incinerators, the regulatory limit in the EU for dioxins (PCDD/F) or dl PCBs and PCDD/F combined was exceeded. Therefore, almost all investigated areas near waste incinerators were unsuitable for free-range poultry (Petrlik et al., 2022).

47 It is obvious that this group of incinerators covered both modern waste incinerators equipped with filters (Petrlik et al., 2007; Petrlik, 2016) as well as very simple installations like e.g. tofu factories burning plastic waste (Petrlik et al., 2020)

Photo 6.1: *Farmers in the vicinity of a municipal waste incinerator in Turin had higher levels of dioxins and dl PCBs in their blood. The same incinerator reported excessive concentrations of mercury in air emissions due to an accident in August 2017. Source and photo: (Eco-dalle-Cittá, 2017)*

Heavy metals levels were studied in the blood of residents from the vicinity of waste incinerators in Korea. The blood levels of lead and cadmium were slightly higher in the group of the subjects who had resided the longest near the municipal waste incinerators in Korea (Lee et al., 2012).

Deng et al. (2016) studied 35 workers at incinerator sites and 269 people living nearby who were exposed to the incinerator's emissions. They also included 143 individuals as controls. They measured the amount of mercury in their blood. Even after considering factors like diet, they found that both the incinerator workers and the exposed residents had higher levels of mercury compared to the control group. This suggests that using local residents as controls might not be reliable, as they could share similar food sources, which can affect mercury levels.

The study, which focused on monitoring heavy metals in the vicinity of Portuguese solid waste incinerators, concluded: *"Compared with published reference values for similar conditions, blood levels of cadmium, lead, and mercury of the present investigation seem to be relatively higher, in median terms and for extreme values, mainly in the case of cadmium and mercury. In the case of lead, the differences are not so marked"* (Reis et al., 2007). On the other hand, a study conducted in the vicinity of a waste incineration plant in Spain, carried out three years after its commissioning, concluded that: "Populations near modern plants for solid waste incineration do not manifest increased levels of heavy metals" (Zubero et al., 2010).

China's increasing use of incineration for managing municipal solid waste (MSW) raises concerns regarding associated air pollution and health risks, which have been largely overlooked. Study by Boré et al. (2022) examined emissions from 510 incineration plants, focusing on PM, SO_2, NO_x, CO, HCl, and heavy metals. Hazard index (HI) and cancer risk (CR) assessments based on evaluation of the heavy metals levels reveal concerning levels, with the national average HI slightly exceeding[48] recommended thresholds and the CR surpassing safe levels.[49] Despite a decreasing trend in emissions of PM, SO_2, and CO, the current buffer protection measures may be inadequate. The multicriteria decision tool, RAFSI, suggests reconsideration of the minimum buffer distance, highlighting the need for more comprehensive measures to address the health impacts of waste incineration (Boré et al., 2022).

48 The hazard index (HI) assessment included Cd, As, Cr, Ni, Pb, and Hg.

49 The cancer risk (CR) included Cd, As, Cr, and Ni.

A new study from Northern Vietnam found that "*the incinerator's exhaust gas caused secondary environmental damage, impacting the health of not only workers but also people living in nearby communities*" (Dung et al., 2023). It justified this with high concentrations of BTEX in ambient air found during four sampling campaigns in April, June, September, and November 2021, with a total of 80 samples collected (see also chapter 5.2). On the other hand, study conducted in 2010 in the proximity of MSWI in Modena (Italy) did not find any differences in urinary BTEX differences between exposed and unexposed[50] subjects. PAHs were higher in exposed than in unexposed subjects for phenanthrene, anthracene, and pyrene (Ranzi et al., 2013).

Understanding the environmental and human impacts associated with PCDD/Fs and dl PCBs exposure from MSWIs is challenging because information on ambient and dietary exposure levels, spatial characteristics, and potential exposure routes is limited. Chen et al. (2006) investigated the relationship between food consumption and blood dioxin concentration in 1,709 residents near 19 incinerators in Taiwan, finding significantly higher blood PCDD/F levels in those consuming locally grown food compared to those who did not ($p<0.0001$). Similar results were found in Cordier et al. (2010). Cordier et al. (2010) also concluded that their study confirms *"previous observation of a link between the risk of urinary tract birth defects and exposure to MSWI emissions in early pregnancy and illustrates the effect of participation bias on risk estimates of environmental health impacts"*.

In the study by Zhang et al. (2023), 20 households from two villages located on the upwind and downwind sides of a MSWI were selected to characterize the concentration and spatial distribution of PCDD/F and dl PCB compounds

50 Between May and June 2010, 65 subjects living and working within 4 km of the incinerator (exposed) and 103 subjects living and working outside this area (unexposed) were enrolled in the study (Ranzi et al., 2013).

in ambient and food samples, such as dust, air, soil, chicken, egg, and rice samples. The source of exposure was identified using congener profiles and principal component analysis. Significant differences were observed ($p < 0.01$) in PCDD/F concentrations in chicken samples and dl PCB concentrations in rice and air samples between the upwind and downwind villages. The exposure assessment indicated that the primary risk source was dietary exposure, especially from eggs, which had a PCDD/F toxic equivalency (TEQ) range of 0.31–14.38 pg TEQ.kg^{-1} body weight (bw).day^{-1}, leading to adults in one household and children in two households exceeding the WHO-defined threshold of 4 pg TEQ.kg^{-1} bw.day^{-1}.[51] Chicken was the main contributor to the differences between upwind and downwind exposure (Zhang et al., 2023).

Principal component analysis (PCA) of PCDD/Fs in another study from China suggested that waste incineration was the primary source of PCDD/Fs in indoor air, whereas PCDD/Fs in indoor dust came from multiple sources. The results of the health risk assessment showed the carcinogenic risk due to indoor PCDD/F exposure was higher for adults than for nursery children and primary school children (Yu et al., 2023).[52]

Domingo et al. (2020) conducted a repeated food survey aimed at estimating the dietary daily exposure to PCDD/Fs by the population living in Tarragona County, where the HWI was being constructed, followed by a review study focused on adverse health effects for populations living near waste incinerators. They ask in the end of their study: *"Taking into account all the information presented above, and reflecting on 22 years of regular operations, a crucial question arises: Does this Hazardous Waste Incinerator (HWI) pose any health risks to the population living nearby? It prompts further questions: What would the concentrations of metals and PCDD/Fs be if the HWI did not exist? Would they be significantly lower than current levels? These questions gain significance considering the potential carcinogenicity of PCDD/Fs and trace elements such as As, Cd, and Cr.*

The regulation of PCDD/F emissions from waste incinerators under the EU Industrial Emissions Directive (2010/75/EU) implies an average emission limit value of 0.1 ng TEQ.Nm^{-3}.[53] However, is this limit truly safe for public health, and where is the evidence to support this claim? Drawing a parallel, if there is no risk-free level of exposure to tobacco smoke, why are current limit values considered risk-free for PCDD/Fs and carcinogenic metals?

It is argued that any facility emitting carcinogenic substances inherently poses health risks. For instance, Inoue-Choi et al. (2017) demonstrated that even minimal exposure to tobacco smoke carries increased mortality risks. Why then are the current limit values presumed to be risk-free?"

The establishment of a waste incineration plant, particularly a HWI, should prioritize preventing any environmental or health impacts. This necessitates rigorous public health surveillance and epidemiological studies, as suggested by de Titto and Savino (2019), Roberts and Chen (2006), and Signorelli et al. (2008).

However, existing studies on biomonitoring and health effects near incinerators exhibit methodological limitations (Campo et al., 2019). Recommendations for more robust epidemiological studies have been proposed

51 The authors of this study used a comparison with the TDI value established by WHO, which is more conservative (at a level of 4 pg TEQ.kg-1 bw.day-1), compared to the newly established value by EFSA (0.25 TEQ.kg-1 bw.day-1) (EFSA CONTAM 2018).

52 The carcinogenic risks of PCDD/Fs for age groups residing near the MSWI plant were all less than the risk threshold (Yu et al., 2023).

53 This limit value was lowered in more recent BAT conclusions for waste incineration European Commission. (2019). Commission Implementing Decision (EU) 2019/2010 establishing the best available techniques (BAT) conclusions, under Directive 2010/75/EU, for waste incineration. Available at: http://data.europa.eu/eli/dec_impl/2019/2010/oj

by Tait et al. (2020), emphasizing the importance of accurate exposure assessments (Hoek et al., 2018).

In summary, Domingo et al. (2020) suggested, while modeling, monitoring, and risk assessment studies are valuable, they may not sufficiently elucidate the risks posed by waste incinerators. More specific epidemiological studies are warranted to establish direct links between health effects and proximity to incinerators. Therefore, it is recommended that authorities conduct epidemiological studies without delay to assess the health risks posed by HWIs. Additionally, risk assessment studies should expand to include chemicals beyond those routinely analyzed, considering potential interactions among them, which current studies overlook.

Domingo et al. (2020) also inserted tables summarizing findings of various studies in their study focused on waste incineration health effects (see Tables 6.1 and 6.2).

Table 6.1: *A summary of international scientific studies with associations between health effects and proximity to MSWIs. (Source: Domingo et al., 2020)*

Location	Health Effect	Main Result	Reference
Great Britain	All cancers	Decline in the risk with the distance from MSWIs	(Elliott et al., 1996)
Great Britain	Liver cancer		(Elliott et al., 2000)
France	Soft-tissue sarcomas and non-Hodgkin's lymphomas	Highly significant clusters observed in the area around the MSWI	(Viel et al., 2000)
France	Non- Hodgkin's lymphoma	Increased non-Hodgkin's lymphoma incidence in the zones with higher concentrations of PCDD/Fs around MSWI	(Floret et al., 2003)
	Soft tissue sarcoma	Soft tissue sarcoma not significantly increased in zone with higher concentrations of PCDD/Fs around MSWI	(Floret et al., 2004)
Italy	Sarcoma	3.3 times higher risk among individuals with PCCD/F higher exposure level around MSWI	(Zambon et al., 2007)
France	Non- Hodgkin's lymphoma	Higher levels of PCDD/Fs in serum of people residing around MSWI Evidences between the incidence of non- Hodgkin's lymphoma and exposure to PCDD/Fs	(Viel et al., 2000)
Review	Lung cancer, larynx cancer and non- Hodgkin's lymphoma	Positive associations between residents near MSWIs and lung cancer, larynx cancer and non-Hodgkin's lymphoma	(Franchini et al., 2004)
Review	Cancer risk, respiratory symptoms, multiple pregnancy, congenital abnormalities, and disturbances in thyroid hormone levels	Increased of all reviewed health effects in individuals living in the vicinity of MSWIs	(Starek, 2005)
Italy	Lymphohematopoietic tumors and soft tissue sarcoma	Higher risk of non- Hodgkin's lymphoma in individuals (males) living near an MSWI	(Biggeri & Catelan, 2006)
Italy	Non- Hodgkin's lymphoma	Increase of mortality due non- Hodgkin's lymphoma in Italian municipalities with MSWI	(Bianchi & Minichilli, 2006)
Italy	Cancer risk	Very small incremental cancer risk around the MSWI	(Cangialosi et al., 2008)
Italy	Lung cancer	Excess risk of lung cancer for people living near the MSWI was below the WHO targe (1 × 10(-5))	(Scungio et al., 2016)

Location	Health Effect	Main Result	Reference
Italy	Neoplasia of nervous system, liver, and total of tumors	Significant increases of neoplesia of nervous system, liver, and total of tumors in persons residing near a MSWI	(Salerno et al., 2015)
Italy	All cancers	Increased risk for cancers	(Salerno et al., 2016)
Italy	Tumors of the lymphohematopoietic, cardiovascular diseases	Increased trends of mortality due to natural causes, the tumor of the lymphohematopoietic system, cardiovascular diseases	(Romanelli et al., 2019)
Italy	Lung cancer mortality	Cancer risk for females consistent with pollution measurements and other epidemiological findings. Lack of an excess risk in males related to strong confounding, due to occupational exposure and smoking habits.	(Parodi et al., 2004)
Italy	Morbidity levels for respiratory disorders	High PM_{10} levels due to the presence of two MSWIs was associated with increased morbidity levels for respiratory disorders in men.	(Golini et al., 2014)
Italy	Mortality and morbidity	Increased risk for cardiovascular diseases and also a trend for urinary diseases Mortality trend for general mortality in males, for cardiovascular diseases also in males, for respiratory diseases in females, at the highest exposure.	(Minichilli et al., 2016)
Review	Cancer, chronic diseases, soft tissue sarcomas	Direct evidences from third generation plants were scare The effect on chronic diseases, and particularly cancer, remains an open issue Potential excesses of soft tissue sarcomas corresponded to earlier incinerators	(Negri et al., 2020)
Review	Respirator effects, diverse cancers, reproductive effects	No significant effects on respiratory symptoms, pulmonary function, twinning, cleft lip and palate, lung cancer, laryngeal cancer, and esophageal cancer The reproductive outcomes were inconsistent	(Hu & Shy, 2001)
France	Cancer	Not able to establish whether had –or not- an excessive number of cancers around a MSWI due methodological difficulties	(Thabuis et al., 2007)
Japan	Mortality	Municipalities with MSWIs showed significantly higher mortality from female stroke than those without plants. The differences were not significant when using also socioeconomic indicators	(Fukuda et al., 2003)
Italy	Cancer risk	No detectable increases of cancer risk for people living near the MSWI	(Federico et al., 2010)
Italy	Morbidity and mortality	No increased risks of morbidity and mortality for the population living in area close to two MSWIs	(Ranzi et al., 2011)
USA	Non-Hodgkin lymphoma	The risks were reduced according to the lesser or greater distance to the MSWIs (until 5 km)	(Pronk et al., 2013)
Italy	Sex ratio, multiple births, preterm births, and small for gestational age births	Maternal exposure to incinerator emissions, even at very low levels, was associated with preterm delivery. However, it was not associated with sex ratio, multiple births, or frequency of small for gestational age births	(Candela et al., 2013)
England	Congenital anomalies in babies	Increased risks with the proximity to the nearest MWI were observed for all congenital anomalies combined, congenital heart defects, and genital anomalies, specifically hypospadias	(Parkes et al., 2020)
Review	Neonatal outcomes	Identified a number of higher quality studies reporting significant positive relationships with broad groups of congenital anomalies. Evidence-base is inconclusive and often limited by problems of exposure assessment, possible residual confounding, lack of statistical power with variability in study design and outcomes.	(Ashworth et al., 2014)

Table 6.2: *A summary of international scientific studies with associations between health effects and proximity to HWIs. (Source: Domingo et al., 2020).*

Location	Health Effect	Main Result	Reference
Italy	Soft tissue sarcomas	Significant increase in the risk of soft tissue sarcoma associated with residence within 2 km of the facility	(Comba et al., 2003)
Korea	Oxidative Stress	Increased levels of PCDD/F in blood of residents living around a HWI Increased oxidative stress of subjects living in the neighborhood of the HWI	(Leem et al., 2003)
Spain	Cancer mortality	Excess risks for all cancers combined as well as for lung cancer in towns in the vicinity of Spanish-based incinerators Marked increases in risk of tumors of the pleura and gallbladder (men) and stomach (women) near a MSWI located in Barcelona (Catalonia, Spain). Significant relative risks of non-Hodgkin lymphoma although no significant increases in the risk of dying of cancer, in the vicinity of that MSWI	(Garcia-Perez et al., 2013)
Spain	Cancer incidence	Higher cancer incidences in the vicinity of certain industrial plants but there was not a clear conclusion according to the statistical model used.	(Querejeta & Alonso, 2019)
Review	Carcinogenic and non-carcinogenic risks	Small carcinogenic and non-carcinogenic risk for people living near HWIs	(Travis & Hattemer-Frey, 1989)
Review	Health risks	It in order to minimize or prevent any potential adverse health effect from HW\is special care must be taken to ensure that facilities are well designed and well operated.	(Pleus & Kelly, 1996)
Finland	Health risk	Increase mercury in exposure (measured in hair) was minimal for residents living close the HWI and did not pose a health risk	(Kurttio et al., 1998)
England	Cancer incidence and mortality	The conclusion was clear: there was no evidence of elevated risk of cancer incidence -or mortality – in the vicinity of large industrial incinerators	(Reeve et al., 2013)
Review	Health effects	Any potential damage to the health of those living near HWIs is probably very small, if detectable. The authors remarked that any waste policy should be to minimize the negative effects of the generation and management of waste on human health and the environment.	(Block et al., 2014)

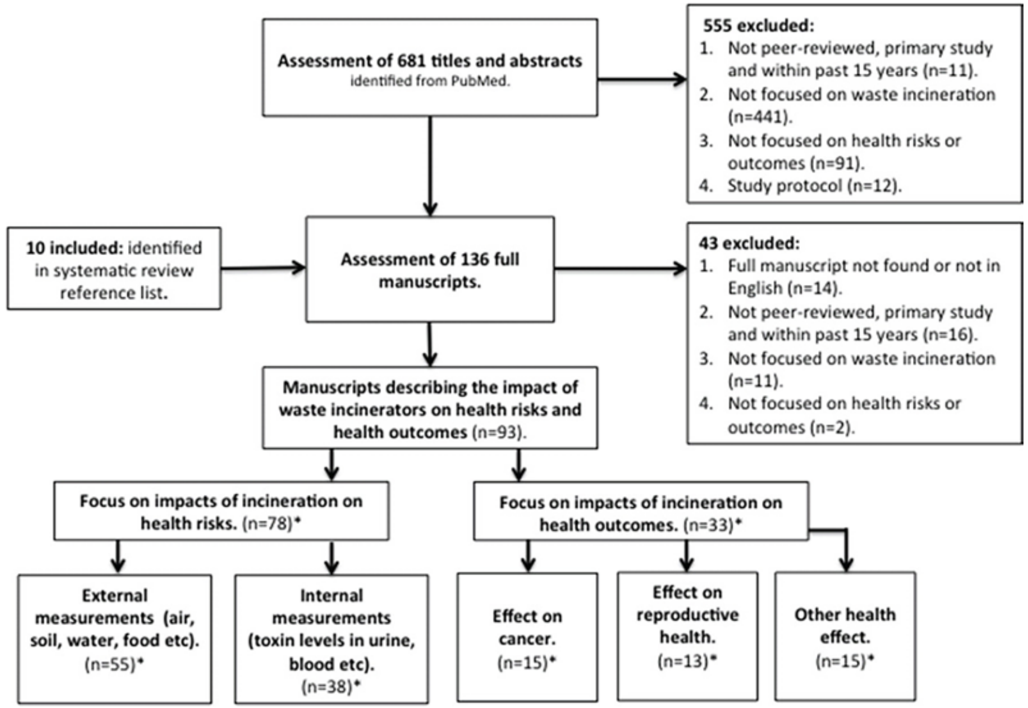
Studies from Japan (Fukuda et al., 2003) and Italy (Ranzi et al., 2011) found no increase in overall deaths linked to living near incinerators despite exposure to emissions. Galise et al. (2012) estimated a slight increase (0.12 %) in deaths attributable to fine particle exposure. Kim et al. found a small burden of disease near waste incinerators in Korea. Li et al. (2015) concluded that waste-to-energy incineration had the lowest non-cancer risks under normal operation but posed the highest cancer risk compared to other waste management methods. Li et al. (2015) also suggested *„that the option of compost with material recovery facility treatment may pose less negative health impacts than other options;“* (see also Chapters 8.1 and 9.1.3).

Tait et al. (2020) in a systematic review, examined 93 manuscripts meeting specific criteria, with methodological assessments based on the

National Health and Medical Research Council (NHMRC) criteria revealing predominantly low grades, with the highest being a grade C (satisfactory) (NHMRC, 2009). Study designs encompassed cohort and case-control investigations, with methodological quality varying from satisfactory to poor due to limitations such as absence of randomization and blinding. However, given the observational nature of the studies, these limitations were not unexpected (Tait et al., 2020). Methodology of the study based on the Preferred Reporting Items for Systematic Reviews and Meta-Analyses (PRISMA) guidelines (Moher et al., 2010) is described in diagram at Figure 6.1.

Of the reviewed papers, 61% demonstrated significant adverse outcomes related to waste incineration, encompassing exposure to elevated pollutant levels, increased risk of neoplasia, adverse reproductive outcomes, and other diseases like hypertension and reduced lung function. Notably, no studies investigated global health effects, but occupational exposure comparisons were made, suggesting incinerator workers could serve as sentinels for adverse effects. Overall, the literature demonstrates increased risk of adverse reproductive outcomes associated with exposure to waste incinerators, in particular preterm birth and congenital anomalies (Tait et al., 2020).

Figure 6.1: *PRISMA diagram for identification of peer-reviewed papers included in the Tait et al. (2020) review.*



The review underscores the significant risks associated with waste incineration, particularly older facilities linked with various health issues. While newer technologies show potential, long-term effects remain uncertain. Incineration's financial and ecological costs, alongside its potential health risks, necessitate careful consideration and close monitoring by policymakers. Community concerns must be addressed transparently, with early consultation essential for facility siting decisions. Study also warned that *„new incinerators should be located away from areas of food production"*, and *„food grown near and incinerator should be avoided"* (Tait et al., 2020).

7. The Accidents

Accidents in waste incinerators (fires or explosions) occur commonly and can be caused by various factors – inadequate safety standards, their non-compliance, equipment defects, human error, or simply a combination of circumstances. Even modern waste incinerators (considered state-of-the-art) are not immune to accidents, and certainly, those situated in the Czech Republic have all experienced at least one fire incident (Arnika, 2022a).

The peril of waste incinerator accidents mainly lies in the uncontrolled and unregulated burning of waste material and the subsequent uncontrolled release of highly toxic substances into the air. Incinerator accidents are relatively common occurrences that sooner or later are almost a certainty for every facility. Fires don't discriminate whether it's a hazardous waste incinerator, WtE facility, or waste pyrolysis unit – they can occur in any of them. Waste incinerator accidents also come with substantial costs – besides repairing the facility itself, in some cases, expensive environmental decontamination must be undertaken. Less frequent incidents in waste incinerators include toxic substance leaks into water sources (see Chapter 3.2.2) or the collapse of corroded structures (dpa, 2009), elaborated further in Chapter 9.2 (also see Photo 9.4).

7.1 Incidents, Fires and Explosions in Municipal Solid Waste Incinerators (MSWI)

For instance, WtE Malešice (Prague) faced frequent fires. In 2003, waste stored in the waste bunker caught fire, in 2020 waste near the shredder technology was ablaze, and in 2021, according to firefighters' statements, the reconstructed emission cleaning technology was on fire just before it was put into operation (iRozhlas, 2021). The initial estimates for the repairs were in the hundreds of millions of Czech crowns (CzK)[54] and were expected to take over half a year.

At the Termizo WtE in Liberec, Czech Republic, one of the bunkers caught fire in 2019. The fire was brought under control after approximately five hours, during which about one-third of the 1,500 tons of waste stored there was burnt (Ortová & Berka, 2019).

On the southern outskirts of London, in Beddington, a fire broke out in July 2019 at the most modern incinerator operated by the company Viridor (see Photo 7.1). The fire likely ignited due to a short circuit in a

54 1 EUR = approx. 24 CzK, so one hundred million of CzK = almost 4.2 million EUR

Photo 7.1: *Fire in the WtE (Waste-to-Energy) plant in Beddington in July 2019. Photo: London Fire Brigade (@LondonFire).*

Photo 7.2: *What remained of the original incinerator in Vienna – Spittelau after the fire in 1987. (Source: Wien Energie, 2024).*

lithium-ion battery (Doherty, 2019). This type of battery, commonly used in mobile phones, is a frequent cause of waste fires.

Many people are familiar with the Vienna waste incineration plant designed by the renowned architect Friedensreich Hundertwasser. However, few know that it stands on the site of an older large incineration plant, which was completely reduced to ashes on May 15, 1987, due to a fire caused by an explosion in the newly installed exhaust gas scrubber at the time (Wien Energie, 2024).

In Detroit, a waste incineration facility was shut down in 2019. While it burned thousands of tons of waste daily, it caused pollution and odors in its surroundings. Over its last 5 years of operation, the facility exceeded emission limits over 750 times. The plan included the remediation of the contaminated area, and ideally, the site would not be used for similar purposes again (Aguilar et al., 2019).

Even the largest Danish incinerator, Amager Bakke, experienced a fire in 2022, which broke out in the hydraulic pressing equipment (Freiesleben, 2022); refer to Chapter 10.2.4. Fires in Swiss incinerators may not be exceptions either (see Photo 7.4).

In Chapter 9.2 is described also serious incident in WtE plant in Prague – Malešice in October 2021. It is on Photo 11.8. A fire on February 2021

Photo 7.3: *Fire at the Košice municipal waste incinerator in 2004. Photo: Spoločnosť priatelov Zeme Archive, Košice.*

Photo 7.4: *A fire at an incinerator in the Aargau canton of Switzerland on May 28, 2015, affected part of the flue gas cleaning system. Black smoke was seen coming from the windows of one of the upper floors (badische-zeitung.de, 2015). Photo: Canton Police, Aargau, Switzerland.*

Photo 7.5: *Firefighters on October 4, 2013, stand in front of the smoking bunker of the Zella-Mehlis (Suedthueringen, Germany) waste incineration plant. This is a typical situation when a fire accident occurs in the bunker, and smoke from burning waste is released freely into the air. (Source: inSuedthueringen, 2013). Photo: frankphoto.de.*

Photo 7.6: *A typical image from the situation where firefighters are extinguishing a fire in the incinerator bunker. Here, they are extinguishing a fire in the bunker of the municipal waste incineration plant Gemeinschafts-Müll-Verbrennungsanlage Niederrhein (GMVA) in Oberhausen, Germany, which occurred in March 2018. (Source: Feuerwehr Oberhausen, 2018). Photo: Feuerwehr Oberhausen, Germany.*

at the Togari Clean Center in Toyota, Aichi Prefecture, damaged part of the incinerator, reducing its capacity by 30%. This led to a “garbage emergency” from April to July, with the city urging residents to cut waste and separate trash properly. The fire was caused by li-ion batteries mixed with combustible garbage. Similar incidents are common in Japan, with 12,765 fires linked to li-ion batteries in fiscal 2020, up from 9,732 the previous year. From fiscal 2018 to 2021, 5,529 fires caused 11.1 billion yen ($78 million) in damages (Matsumoto, 2023).

7.2 Incidents and Fires in Hazardous Waste Incinerators (HWI)

An accident in a hazardous waste incinerator in Leverkusen, Germany on 27 July 2021 is marked as probably one of the largest and most fatal industrial accidents in Germany. Seven people were killed and 31 injured in an explosion and subsequent fire at a hazardous waste treatment site operated by Currenta in Leverkusen (see Chapter 7.2.1.2).

In Malenovice near Zlín, at the hazardous waste incinerator operated by SUEZ at that time, one of the waste incineration furnaces caught fire in 2016 (tydenikpolicie.cz, 2016). However, the biggest fire occurred there on March 7, 1997, when the entire incinerator turned into ashes.

Several incidents also occurred at the Chropyně hazardous waste incinerator. In 2003, the solid waste shredder caught fire, resulting in damages reaching 1.5 million Czech crowns. In 2005, the storage area with barrels caught fire, involving solvents and chemicals intended for incineration, damaging the incineration technology itself and causing approximately 2 million Czech crowns in damages (Kapitánová, 2005). The facility experienced a second fire in the same year.

Photos 7.7 and 7.8: *Fire in are with barrels of toxic waste in hazardous waste incinerator in Chropyně, Czech Republic in November 2005. Fire-water runoffs can be toxic to water organisms, especially if fluorinated firefighting foams are used. Photos: HZS ZK (Fire-fighters CZ).*

Photo 7.9 *In 2005, the hazardous waste incinerator Ekotermex Vyškov caught fire twice, the photo shows the fire in May. Photo: FIRE BRIGADE.*

Photo 7.10 *A mismatch between incinerator technology and an untested dioxin filter can also lead to an incinerator accident, as happened in April 2013 at the hazardous waste incinerator in Lysá nad Labem (Arnika, 2013). Photo: Arnika archive.*

Photo 7.11 *Fire at the hazardous waste incinerator in Füzfó, Hungary, in July 2010. On June 15, 2010, a major accident occurred at a hazardous waste incineration plant near Lake Balaton (Tremmer, 2010). Ammonia, chlorine and nitrogen dioxide were released into the air. Photo: Gáspár Gábor.*

Photo 7.12: *One of the largest accidents occurred at a hazardous waste incinerator in Campana, Argentina, on November 18, 2004 (Red Proteger, 2004). The fire completely razed the incinerator, similar to what happened at the Emseko hazardous waste incinerator in Zlín in 1997. Photo: Red Proteger, Argentina.*

More examples of incidents in hazardous waste incinerators are in following case studies as well as in the case study focused on PCBs accidental leakage from the POPs waste treatment center Swan Hills in Alberta, Canada (see Chapter 5.1.3.1).

7.2.1 Case studies

The following case studies document accidents in hazardous waste incinerators in Germany, USA and Canada. They also represent various scenarios which were described in the studies by (Morrison et al., 2018) and/or by the Ministry of the Environment, Energy and the Sea (MoEES) in France (MoEES, 2016). Morrison et al. (2018) has documented another case in a cement plant incinerating waste caused by a conveyor belt, similar to what happened in Covanta's waste incinerator in Miami. MoEES (MoEES, 2016) suggests that also mixing of incompatible products can lead to accidents in waste incinerators such as the case of the hazardous waste incinerator in El Dorado, Arkansas.

7.2.1.1 Explosion in Waste Incinerator in El Dorado, Arkansas

An incident occurred at the Teris Inc. hazardous waste incineration plant in El Dorado, southern Arkansas in January 2005, resulting in the evacuation of hundreds of residents. The fire, which originated from a warehouse within the facility, caused concerns among nearby residents due to health issues such as headaches, nausea, and eye irritation. Despite no serious injuries reported, the evacuation disrupted the community (Associated Press, 2005).

The warehouse contained 4,000–5,000 drums of various wastes, including sodium chlorate solid waste erroneously profiled by the waste generator. This material, accompanied by organic contaminants, rail car components, and railroad ties, underwent spontaneous combustion, leading to a rapid spread of fire throughout the warehouse (Morrison et al., 2018).

Investigations revealed deficiencies in waste characterization, highlighting the need for thorough visual examination and proper waste analysis plans to accurately identify potential hazards. Following the incident, the facility implemented enhanced material analysis and handling protocols for wastes containing solid oxidizers to prevent similar incidents in the future (Morrison et al., 2018).

Photo 7.13: *Aerial image during fire event in January 2005 in El Dorado waste incinerator. (Source: Morrison et al., 2018). Photo: US EPA.*

A group of 26 residents in El Dorado, affected by fires at a hazardous waste processing plant in 2005, have filed a lawsuit against the company operating the site. Alongside this legal action, Teris LLC faced allegations of negligence from these residents, seeking compensation for expenses and losses incurred during evacuations and previous incidents. These incidents include the January 2005 explosion and fire, which forced the evacuation of 2,500 people, as well as a subsequent fire in July 2005. Additionally, Teris initiated a lawsuit against a company due to alleged mislabeling of waste that combusted after arrival at the plant (Associated Press, 2008).

Toxicity tests discovered water contamination in an outfall from the waste incinerator after the fire in El Dorado, in samples taken in 2006 and 2007. Fire-suppressant chemicals were suspected toxicants, and rain events during and after occurrence of the fire at the Teris facility provided a means for entry of fire-suppressant foam into the outfall from the waste incinerator. An analytical laboratory discovered that organophosphate-based surfactants as most likely source of the toxicity in water (FTN Associates Ltd., 2007).

7.2.1.2 Explosion in Leverkusen

Leverkusen, Germany: 9:40am, July 27th, 2021

In July 2021, an explosion occurred in a tank containing solvents at Chempark's chemical complex in Leverkusen, Germany. The complex is a major manufacturing hub for Bayer, Lanxess, and about 30 other companies. Seven people died and 31 were injured in an explosion and subsequent fire at a hazardous treatment site operated by Currenta in Leverkusen (Cartwright et al., 2021; Scott, 2021).

The blast occurred at 9:40 am, 27th July 2021 in the storage area of a hazardous waste incinerator, where production residues are collected for disposal in the waste incinerator (Cartwright et al., 2021). Three solvent storage tanks containing chlorinated solvents caught fire. Organic solvents are stored in containers at the center before being incinerated (Scott, 2021). The authorities closed roads and warned residents to stay indoors. They also warned of possible release of toxins into the atmosphere, telling residents not to eat fruit and vegetables from their gardens (Cartwright et al., 2021).

The explosion occurred in the complex's waste-management center, which features a sewage system, a landfill, and an incinerator. It was the worst chemical industry accident in Germany since 2016, when three

Photo 7.15

Photos 7.14 and 7.15: *Explosion in Leverkusen Chempark was worst chemical industry accident in Germany since 2016. Photos: Printscreens from videos on Youtube.com.*

maintenance workers died while working on pipelines at BASF's site in Ludwigshafen (Scott, 2021).

Following soil and plant analyses in the region, the State Environmental Agency of North Rhine-Westphalia initially found no relevant concentrations or exceedances of limits (bbr/me/sep/dpa, 2021).

However, on December 22, 2021, the Cologne district government announced that firefighting water containing clothianidin (a neonicotinoid pesticide) had been discharged into the Rhine during firefighting efforts. In the weeks following the incident, the Rhine waterworks in the Netherlands detected clothianidin in drinking water extracted from the Rhine for the first time. Additionally, Currenta released remaining liquids and firefighting water containing PFOS into the Rhine but failed to inform the responsible International Commission for the Protection of the Rhine about the discharge of toxic substances. Measurements by LANUV also revealed significantly elevated levels of PFOS in wastewater from Currenta's wastewater treatment plant. This underscores the gravity of the situation, prompting further investigation into Currenta's actions and potential charges for violating environmental laws and endangering public health (Wikipedia, 2022a).

The NRW Regional Association of the Federation for Environment and Nature Conservation in Germany (BUND) indeed filed a criminal complaint on January 17, 2022, with the Cologne Public Prosecutor's Office regarding water pollution and unauthorized waste disposal by Currenta and the Cologne district government. The complaint concerns illegal handling of toxic water for firefighting, which was mixed with wastewater from Chempark after an explosion at the hazardous waste incinerator in Leverkusen-Bürrig and discharged into the Rhine via the wastewater treatment plant. This information sparked an investigation by WDR Westpol magazine, revealing that Currenta, the supervisory authority, and the Minister for the Environment of North Rhine-Westphalia had withheld this information from the public. Nearly 10 million liters of water contaminated with toxic chemicals, including over 60 kilograms of the highly harmful insecticide clothianidin, were discharged into the Rhine. This action, along with the failure to disclose pertinent information, initiated an investigation, revealing serious environmental and public health concerns (BUND Landesverband Nordrhein-Westfalen, 2022).

7.3 Incidents, Fires and Explosions in Pyrolysis and Gasification Technologies

Incidents also extended to pyrolysis units. In Hamm, Germany, a pyrolysis unit was added to the local power plant in 2000. However, technical difficulties arose during practical operation, culminating in the collapse of the 60-meter-high chimney in 2009 due to material corrosion (see Photo 7.16). The power plant's roof was also damaged. According to the power plant operator, after this incident, the commercial operation of the pyrolysis unit was no longer feasible. In Fürth in the 1990s, a pyrolysis-combustion unit with a capacity of 100,000 tons.y^{-1} (low-value waste) was built. Despite optimization attempts, the long-standing problems led to the unit being decommissioned (dpa, 2009; Gleis, 2012).

Photo 7.16 *The Hamm pyrolysis stack corroded and collapsed in 2009. It was never rebuilt. Photo: Reiner Mors, tz.de, Source (dpa, 2009).*

Rollinson (2018) describes very well fire and explosion hazards in waste gasification processes: *"Fire and explosion hazard is created by the producer gas being ubiquitously within its explosive range combined with the high risk for contact with multiple ignition sources within the gasifier system. This is evidenced by historic antecedents which report 2865 gasifier fires over a six year period in Sweden. Explosive environments are also evidenced by historic antecedents. These are caused by both underpressure (oxygen ingress) and overpressure (flammable gas egress) in both the high temperature reactor and in ancillary components, again due to the multi-component and dynamic features of a gasifier system.*

Start-up and shut-down are identified as times when there will be a significantly heightened risk for fire, explosion and toxicity hazard. This is particularly concerning for modern "concept" systems which must necessarily operate on a test-basis, and which try and obviate less hazardous aspects such as noise and odour without a proper appraisal of the risk antecedents.

Raw waste which is pre-processed by sifting out some of the inorganic content, shredding, compacting, and drying has a propensity to self heat and auto-ignite. There have been several recent accidents due to the spontaneous combustion of stored RDF. If the waste industry is to avoid further process losses, it must learn from the lessons of gasification history and the lessons of risk assessment developed through major chemical process accidents of the past. At present however, risk is being aggravated by a reluctance to disclose or address these failures, preferences for novelty, a lack of stakeholder understanding, and a desire to operate beyond technological capabilities" concluded Rollinson (2018).

Fires can easily occur in pyrolysis plants. In February 2019, a tire pyrolysis unit caught fire in Nederweert, the Netherlands documented at Photo 2.4. It was also not the first time that the fire brigade has been sent to the company according the local press *(Scott, 2019).*

The Lučenec plastic waste catalytic pyrolysis plant experienced repeated fires in May 2016 (see Photo 9.5) and September 2017. Fire in May 2016 started on one of the machines and burnt also part of the stored plastic waste. The damage caused by the fire was estimated to be in five hundred thousands of euros (Hutková, 2016).

Hedlund (2023) analyses a recent case study of a plastic pyrolysis plant in Egebjerg, Nykøbing Sjælland, Denmark which suffered two explosions. The first *"on August 17, 2020, an internal explosion occurred in a P4O[55] reactor. The endcap was blown open with great force tearing all the heavy-duty bolts. There was significant material damage. A masonry wall was partly blown out, as was the roof above the reactor."* There were no casualties in this first explosion despite the pyrolysis plant being situated within 300-400 metres of a primary school and kindergarten. A subsequent investigation by an inspector found there was no written manual for the pyrolysis plant. The worker only received oral instructions from Spanish consultants and was unaware of any explosion risk. In addition, no risk assessment or ATEX[56] assessment has been carried out. According to Hedlund (2023) on *October 8, 2021, a P4O pyrolysis reactor experienced a repeat internal explosion. Again, the explosion tore open the hinged endcap and the blast wave and ensuring fire caused extensive structural damage, the roof was gone and walls blown out.* There we no casualties as all staff were in a morning meeting in the control room as the reactor exploded and "*Burning liquid plastic flowed out of the open reactor and ignited baled waste plastic stored near the reactor."*

In the case of this facility it was found that poor operating systems and lack of training for controlling flammable pyrolysis gases was the

55 An abbreviation of plastic for oil or plastic to oil.

56 ATEX assessments deal with assessments of explosive atmospheres and minimum safety requirements for workplaces and equipment used in explosive atmospheres.

Figure 7.1: *Pyrolysis plant located at the outskirts of the Egebjerg rural community. Source: (Hedlund, 2023).*

immediate cause of the explosions, Hedlund (2023) concludes that, *"The immediate cause of the two explosions is autoignition of flammable pyrolysis vapors. Because the pyrolysis process temperature is higher than the autoignition temperature of the pyrolysis gases, a source of ignition is always present inside the pyrolysis reactor. Inert gas purging is therefore essential to control explosion risk."*

The location of the plant so close to sensitive receptors like schoolchildren and the apparent lack of regulatory oversight of the company casts doubt not just upon the plant operators but the system of planning and regulation to manage such facilities – even in modern European countries. As Hedlund (2023) summarises, *"Pyrolysis is an inherently hazardous process but barriers to entry appear rather low and regulatory oversight limited."*

Photo 7.17: *Reactor endcamp torn open, deformed (2021). (Source: Hedlund, 2023).*

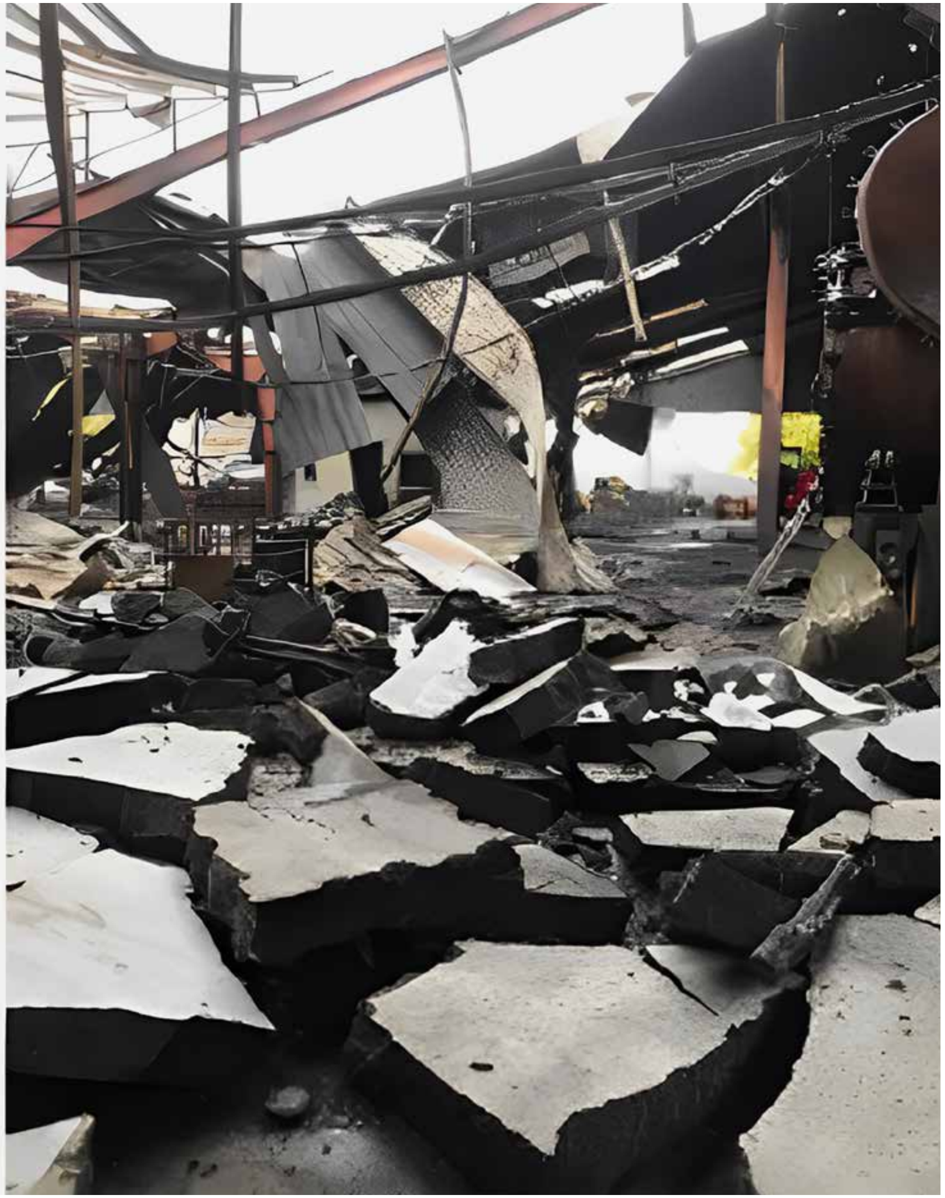


Photo 7.18: *Walls toppled due to explosion, roof gone (2021). (Source: Hedlund, 2023).*

7.4 Refused Derived Fuel (RDF) and Fires

Refused derived fuel (RDF) is another potential source of fires. Rollinson (2018) explained why: *"Due to the safety antecedents of self-heating during storage, the environment and duration of time stored must therefore be adequately appraised with a fire risk assessment to also cover stacking, avenues of moisture ingress, wrapping (if any) and monitoring of temperature and carbon monoxide, methane, and hydrogen levels"* (Rollinson, 2018).

(Morrison et al., 2018) described a fire which occurred at a cement production facility that used an engineered solid fuel composited from MSW and other industrial wastes to co-fire in the calciner. The engineered fuel was a blended mixture of solid wastes including paper, plastic, and sawdust as well as nonhazardous waste liquids, slurries, and sludge.

The engineered fuel, comprising oxidizing materials like polymerizing agents and vegetable-derived oils, introduced a significant risk of self-heating, necessitating vigilant inspection by material handlers for indications of elevated temperatures. Prior to loading onto trucks for transportation to the cement production facility, infrared pyrometers were employed to monitor surface temperatures. Upon arrival, the fuel navigated a complex conveyor system, including horizontal screw conveyors and a central drag conveyor, before reaching a long, enclosed belt conveyor. Despite the belt's initial flame-resistant nature, patches of non-flame resistant material compromised its integrity, leading to a fire near the head section of the conveyor upon its restart following a 2-day shutdown. This incident highlighted the critical need for enhanced safety measures and closer monitoring during conveyor system operation. Furthermore, subsequent occurrences, such as the ignition of a stockpile at the feed mixing facility, underscored the difficulty in detecting internal self-heating (Morrison et al., 2018).

7.4.1 Case studies

The following two subchapters discuss case studies on RDF incinerators in France and the United States.

7.4.1.1 Fos-sur-Mer, France

On November 2, 2013, at approximately 2:30 am, a fire erupted inside the 2,000 m² sorting building of a waste treatment facility. This facility had been operational since 2010 and was situated on an isolated 18-hectare parcel within an industrial/port zone. The fire rapidly spread due to wind, engulfing a compost storage and maturation zone covering 8,000 m² (holding 4,000 tonnes). Incandescent cinders were carried by fans used to regulate building pressure, causing the fire to extend to the air treatment and deodorization unit (which included biofilters spanning 3,000 m²). In under an hour, the fire reached another sorting zone comprising 5,000 m² of floor space, primarily

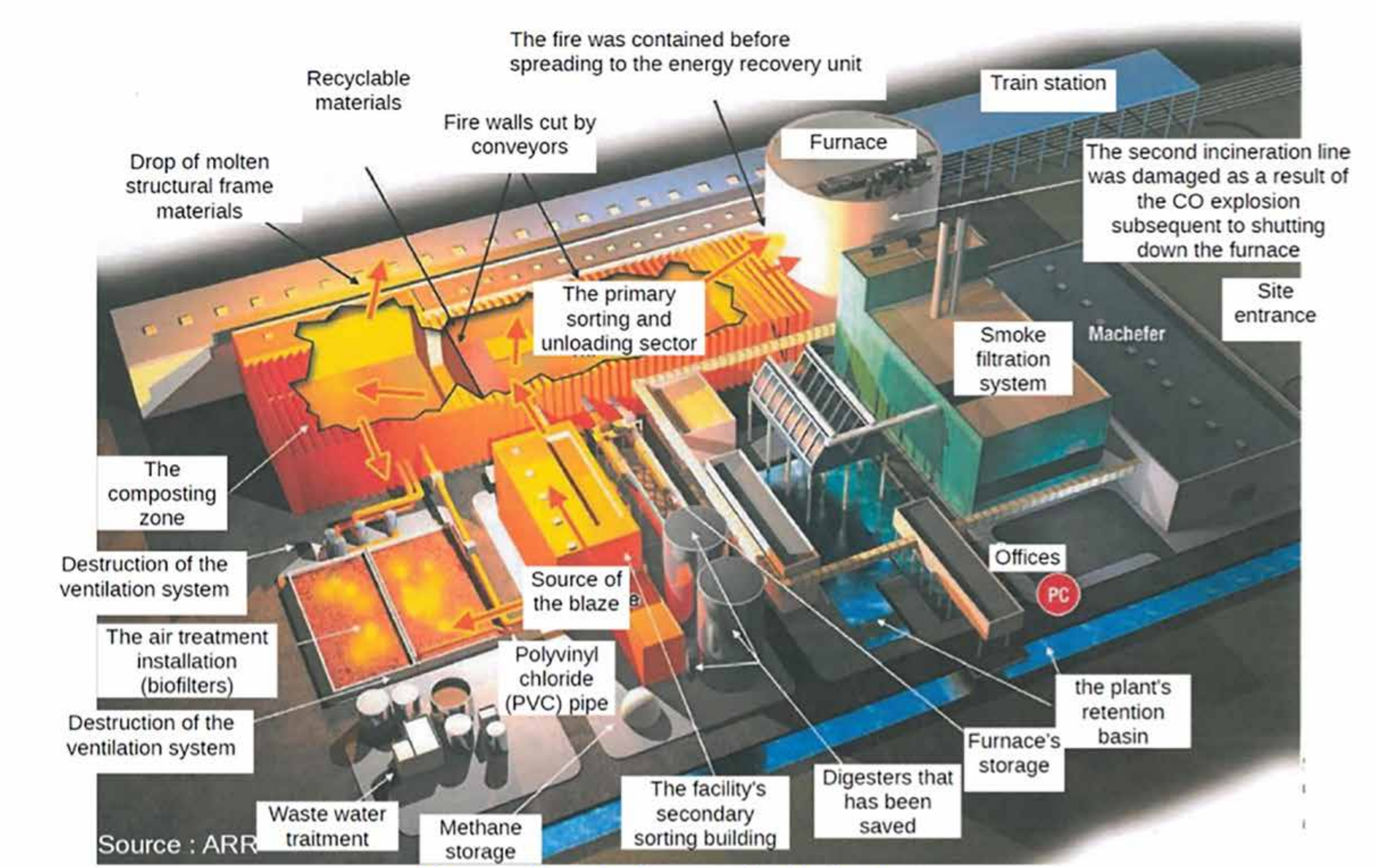


Figure 7.2: *Diagram of how the fire spread inside the waste treatment plant in Fos-sur-Mer, France. (Source: IMPEL and BARPI, 2015).*

containing plastics. The fire progressed via conveyor belts, penetrating firewalls and igniting the glued-laminated wood frame covering these walls. Burning timber fell, igniting two household waste pits (27,000 m^3 with a thickness of 20 meters) around 6:30 am (IMPEL and BARPI, 2015). For better understanding BARPI (2015) has provided a diagram (see Figure 7.2).

The primary and secondary sorting units, along with the biofilter and three buildings housing these facilities (totaling 18,000 m^2), were completely destroyed. However, two digesters and the incinerator were salvaged. Around 6 am, an incineration line (primary air inlet to a furnace) sustained damage from a CO explosion occurring three hours after the furnace shutdown. Property damage and production losses amounted to tens of millions of euros. Some of the waste typically treated on-site had to be redirected to other facilities. The site operated at 85% capacity for 18-24 months (IMPEL and BARPI, 2015; MoEES, 2016).

7.4.1.2 Fire at Covanta's Doral Incineration Plant in Miami, Florida

On February 12, 2023, a catastrophic fire erupted at the Covanta-operated waste-to-energy plant in Doral, Florida, engulfing a trash pit the size of a football field. The blaze left the facility's buildings in ruins, emitting plumes of toxic smoke that blanketed the surrounding Miami suburb of more than 75,000 residents. The disaster underscored residents' long-standing grievances about the environmental and health impacts of the plant's operations (Kapnick, 2023). The fire burned for almost three weeks and was finally placed under control on March 2, 2023 (Burkhardt et al., 2023).

This fire at the Doral incinerator was one of five in recent years. In July 2022, a fire broke out at the facility in a similar way to this recent one, with waste ("refuse-derived fuel") on a conveyor belt catching fire. In 2021, two small fires broke out, one in February in the trash pit, and another in March on a conveyor belt and in June of 2019, a fire broke out on one of the shredder lines at the facility (Burkhardt et al., 2023).

The fire in February 2023 occurred amidst contentious plans for a new trash incinerator plant adjacent to the existing facility, approved by the Miami-Dade Board of County Commissioners in the summer of 2022. The project faced backlash from residents and advocacy groups who criticized the lack of public input and transparency in the decision-making process (Kapnick, 2023).

Residents, represented by the Doral Community Coalition and led by former coalition president Ivette Gonzalez Petkovich, had been advocating against the new plant, citing concerns about the adverse effects of living near such a facility. The fire served as a potent reminder of the risks posed by the existing incineration hub, exacerbating fears and galvanizing opposition to the expansion plans (Kapnick, 2023).

In the aftermath of the blaze at Covanta's Doral waste incinerator, residents took legal action through a federal lawsuit, alleging exposure to hazardous substances that endangered health and property values. Despite Chief U.S. District Judge Cecilia Altonaga denying Covanta's motion for partial judgment and highlighting residents' concerns about health impacts and property damage, Miami-Dade County officials claimed no hospital admissions were linked to smoke exposure (DeLuca, 2024). They asserted that the air surrounding the plant contained low levels of particulate matter and lacked toxic chemicals, thus posing no harm to residents. However, plaintiffs cited an Earthjustice report (Burkhardt et al., 2023) revealing pollutants from the fire exceeded EPA-designated unhealthy levels, challenging the county's assertions (DeLuca, 2024).

"Covanta's industrial fire spewed dangerous contaminants into the air and polluted the surrounding area with smoke, ash, soot, creosote, and the numerous chemicals contained therein, including but not limited to dioxin," the lawsuit alleges (DeLuca, 2024).

Apart from legal actions, air and water pollution concerns emerged despite initial reassurances about air quality. Data revealed particulate matter concentrations surpassing moderate levels of concern, leading to unhealthy air quality for sensitive groups and hazardous conditions. Volatile organic compounds, chlorine, carbon monoxide, and hydrogen cyanide were detected at levels exceeding EPA action thresholds, posing risks to human health. Water testing post-fire by Department of Environmental Resources Management (Miami-Dade County) indicated exceedances of sanitary sewer standards for copper, lead, and zinc, as well as groundwater standards for sulfates and boron. Sulfates raised concerns for wastewater treatment processes and corrosion, as noted in the results (Burkhardt et al., 2023).

Despite meeting contamination standards, various pollutants such as total suspended solids, cyanide, PCDD/Fs, VOCs, petroleum hydrocarbons, metals, SVOCs, nitrate + nitrite, ammonia, and formaldehyde were present. This comprehensive assessment underscores the extensive pollution post-fire, necessitating thorough monitoring and remediation efforts to safeguard public health and environmental integrity in the affected area (Burkhardt et al., 2023).

The fire also prompted scrutiny of Covanta's safety protocols and operational history. Fire investigators identified the blaze's origin near the plant's conveyor belt, noting previous incidents of smaller fires in the years leading up to the catastrophic event. Despite Covanta's claims of safe waste management practices, the fire raised questions about the adequacy of pollution control methods and emergency response procedures at the facility.

Meanwhile, community leaders, including Doral Mayor Christi Fraga and former city mayor J.C. Bermudez, intensified calls for relocating trash incineration operations out of Doral. Miami-Dade County commissioners revisited the decision to build the new plant in Doral, ultimately agreeing

Photos 7.19 and 7.20: *Buildings at Covanta's Doral incinerator plant in Miami collapsed in the blaze in February 2023, creating a labyrinth of twisted metal during firefighting efforts. (Source: Kapnick, 2023). Photo: 7.ch - Screenshot from City of Doral fire footage; 7.ci - Screenshot from Miami-Dade Fire Rescue video.*

Photo 7.21: *Firefighters attempt to extinguish flames at Covanta's Doral trash incineration plant in February 2023. (Source: DeLuca, 2024). Photo: Screenshot from Miami-Dade Fire Rescue video.*

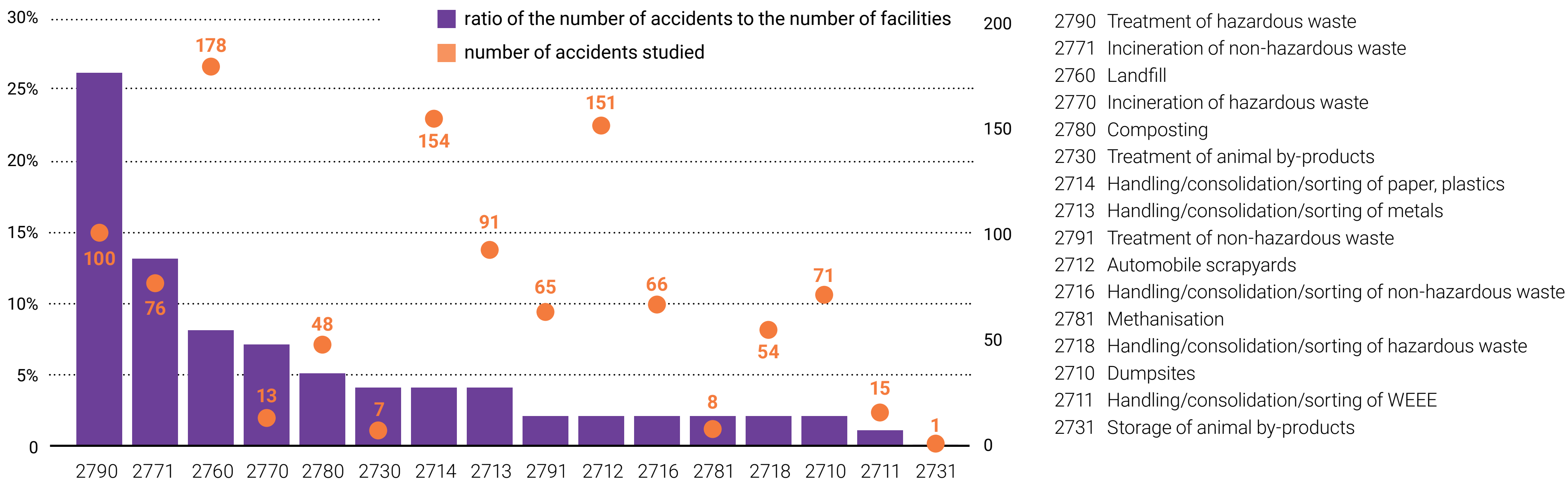
Photo 7.23: *Exterior of the Miami-Dade County Resources Recovery Facility after a fire in February of 2023. (Source: Burkhardt et al., 2023). Photo: Miami-Dade County Department of Environmental Resources Management*

Photo 7.22: *The Miami-Dade County Resources Recovery Facility during a fire in February of 2023. (Source: Burkhardt et al., 2023). Photo: Gina Romero / Florida Rising*

to reverse the designation, albeit leaving the city on a shortlist for potential future plant locations (DeLuca, 2024).

The Covanta fire highlighted broader environmental and regulatory challenges associated with waste management and incineration practices in densely populated urban areas. It underscored the importance of robust public engagement, transparent decision-making, and stringent safety measures in addressing residents' concerns and mitigating risks associated with industrial facilities in residential neighborhoods.

Figure 7.3: *Graph showing ratio of the number of accidents to the number of facilities in France between 2005 and 2014. (Source: MoEES, 2016).*



Note: In the above graph, the 27xx numbers associated with each activity correspond to the headings defined in the French nomenclature for classified facilities with an environmental protection designation.

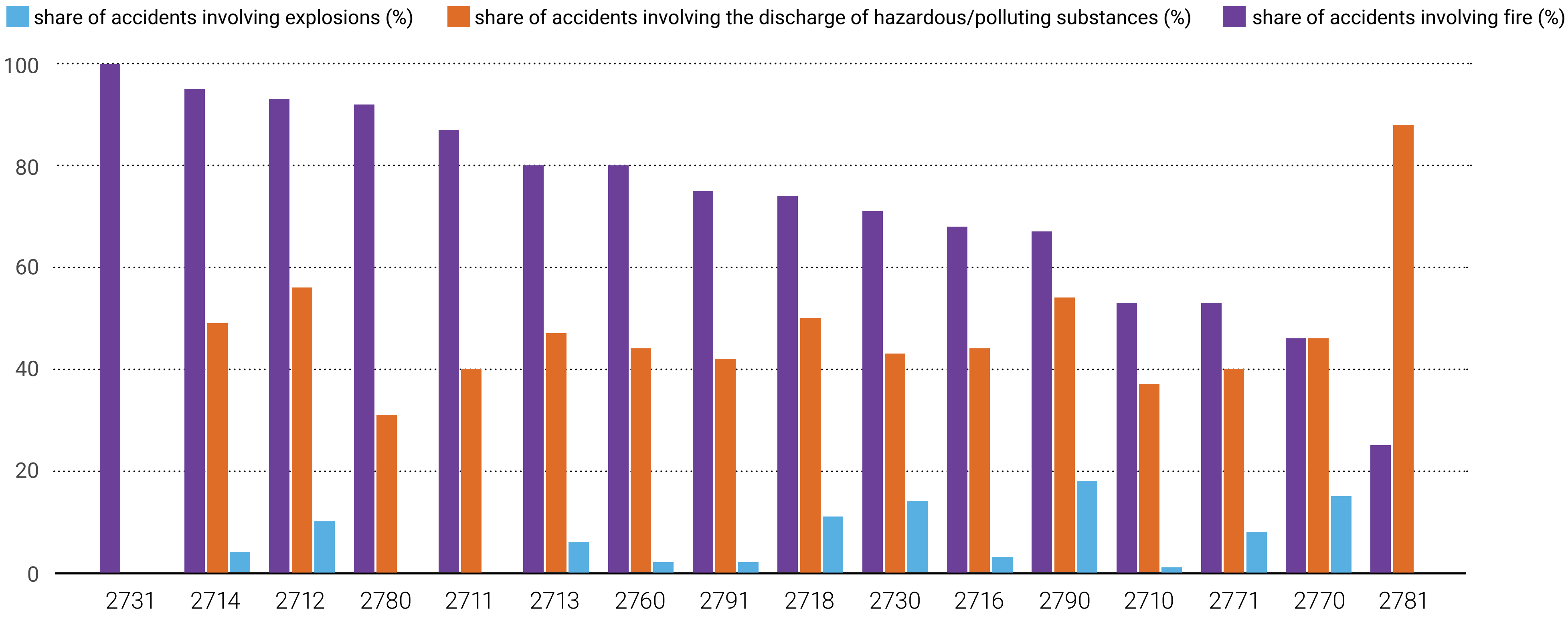
7.5 Analysis of Accidents in Waste Incineration Sector in France

Analysis, Research and Information on Accidents (ARIA), the French database catalogues incidents or accidents that were, or could have been, deleterious to human health, public safety or the environment. Ministry of the Environment, Energy and the Sea (MoEES) analyzed accidents occurring in waste incinerators and other waste management facilities in France, based on ARIA (MoEES, 2016). Its study, which is quite unique, evaluated accidents following a well described methodology: The reference used when describing the seriousness of consequences due to accidental events is the "European scale of industrial accidents" (BARPI, 2003).[57]

Overall, between 2005 and 2014, among all accidents recorded at classified facilities, irrespective of the activity, approximately 15% registered a

57 This scale is based on the four following indices, each of which is divided into 6 levels:
- Hazardous substances released
- Human and social consequences
- Environmental consequences
- Economic consequences

Figure 7.4: *This graph shows percentage of serious accidents in various waste management facilities in France between 2005 and 2014. The total obtained exceeds 100 % since several hazardous phenomena may be involved in a single accident. "Other phenomena" refers in particular to near accidents and workplace accidents. (Source: MoEES, 2016).*



Hazardous phenomenon	Percentage of accidents by phenomenon	
	Waste sector	All environmentally sensitive facilities
Fire	78%	62%
Discharge of hazardous/ polluting substances	47%	49%
Explosions	6%	8%
Other phenomena	12%	8%

2731 Storage of animal by-products waste
2714 Handling/consolidation/sorting of paper, plastics products
2712 Automobile scrapyards
2780 Composting
2711 Handling/consolidation/sorting of WEEE
2713 Handling/consolidation/sorting of metals
2760 Landfill
2791 Treatment of non-hazardous waste
2718 Handling/consolidation/sorting of hazardous
2730 Treatment of animal by-products
2716 Handling/consolidation/sorting of non-hazardous waste
2790 Treatment of hazardous waste
2710 Dump sites
2771 Incineration of non-hazardous waste
2770 Incineration of hazardous waste
2781 Methanisation

score of at least 2 on one or more of the 4 scale indices. Specifically for classified facilities designated by NAF code 38, Waste Collection, Treatment, and Disposal, only 11% of the accidents scored a "2" on one of the scale indices. The waste management sector ranks 12th in terms of "serious" accidents, while it holds the 3rd position in the total number of accidents. More serious and more often there were accidents in chemical industry sector for example (IMPEL and BARPI, 2015).

MoEES (2016) in its analysis compared the accidents in the incineration process (for either hazardous or non-hazardous waste) with other types of waste management operations. The result of such comparison is in graph at Figure 7.3. From the graph, it is apparent that contrary to the commonly held perception of the frequency of accidents at landfills and waste incinerators, France has experienced more serious accidents in a higher percentage of incinerators than at landfills, although there are more accidents investigated at landfills in terms of numbers. Another graph (see Figure 7.4) then explains that firefighters more frequently intervened in cases of hazardous substance leaks or explosions at incinerators than at landfills in France between 2005 and 2014 (MoEES, 2016).

MoEES (2016) analysis also got to the specific accidents case studies occurring in waste incinerators, such as explosions caused by inadequate procedures for controlling and maintaining combustion, release of toxic substances subsequent to the accidental mix of incompatible products during the transfer of reagents used to purify burned gases, and falling into waste pit.

"Accidents are typically associated with an error that can be traced back to the chemical product supplier (labelling error, inappropriate packaging) or the driver assigned the delivery (handling error). Consequently, the incineration plant operator's only course of action is to reinforce controls and supervision during the critical transfer step and encourage upstream partners to implement their own measures (procedures, training, etc.) to avoid encountering such problems," MoEES (2016) suggests.

7.5.1 Explosion caused by inadequate procedures for controlling and maintaining combustion

This scenario pertains to cases like explosions subsequent to a disaggregation of clumps of dust or substances (fouling), clogging in the waste loading hopper prompting the formation of CO combined with an undetected malfunction in the temperature probes, etc.

For example, on September 7, 2014, in Clermont-Ferrand, a significant incident occurred within a non-hazardous waste incinerator on a Sunday, characterized by a sudden pressure surge near the furnace combustion chamber. This surge activated various safety mechanisms, triggering automatic shutdown protocols and causing waste to scatter at the slag extractor outlet. Safety rupture discs beneath the furnace grating and the boiler expansion vessel hatch were also opened, resulting in untreated smoke being emitted through these openings for several minutes, causing concern among nearby residents.

Following the event, the incinerator operator collaborated with the builder to analyze its causes. It was determined that the pressure surge was caused by a substantial volume of materials falling onto the furnace grating and into the slag wells, creating a compressive effect similar to a piston within the gas-filled wells.

In response, the operator conducted comprehensive maintenance of the boiler expansion hatch and carried out a thorough inspection during a planned shutdown. Although no anomalies were immediately evident, the boiler was found to be clogged with soot and promptly cleaned. Additionally, adjustments were made to combustion regulation parameters to optimize operational efficiency. To enhance technician safety, several measures were implemented, including performing all works near the extractors with guillotines closed to prevent personal injury from debris spray, installing chains to secure skips and prevent tipping, and establishing a protected pedestrian crossing within the facility.

It is noteworthy that a similar incident involving an explosion and subsequent fire had occurred in the incineration furnace just months prior to this event (MoEES, 2016).

7.5.2 Incineration furnace explosion due to the presence of non-compliant waste

This scenario concerns explosions resulting from the presence of exogenous non-compliant waste in the furnace, which should not have been processed through this treatment stream, or from non-compliant waste deviating from the authorized specifications due to incorrect on-site preparation handling prior to incineration. For instance, on August 29, 2007, in Reims, around 8:30 pm, an explosion occurred inside furnace no. 1 of a municipal waste incineration plant. It appears that the explosion was triggered by non-compliant waste items such as gas bottles and munitions. The pressure loss led to a significant water leak in the boiler, prompting the emergency shutdown of line no. 1. As a result, a portion of the incinerator smoke was released into the atmosphere without proper treatment. The household waste was temporarily disposed of at a dumpsite until the furnace could be repaired.

According to the plant operator, the explosion in the furnace caused damage to the boiler tubes over a height of 30cm, as recorded during the initial survey inspection.

7.5.3 Release of toxic substances subsequent to the accidental mix of incompatible products

In many industrial activities, material transfers can pose significant risks, especially when potentially hazardous substances are involved. There are frequent instances of accidental mixing leading to toxic releases during the delivery of vital reagents by external shippers.

For example, on November 19, 2012, at Vaux-le-Penil, a driver was delivering a 25% hydrochloric acid (HCl) solution to a household waste incineration plant at around 8 am. The lorry carried three 1,000-liter bulk acid tanks and one 10% sodium hypochlorite (NaClO) bulk container in the same compartment. During the transfer process, the driver mistakenly connected the transfer hose intended for the acid tank to the sodium hypochlorite container, causing a chlorine (Cl2) release when approximately 200 liters were transferred.

Upon noticing the release at 8:15 am, the site's materials delivery agent halted the operation and sounded the alarm. Despite wearing individual protective gear, the driver felt ill from the release but managed to evacuate the transfer zone. Firefighters and municipal police were called to the scene, and a safety perimeter was established.

Due to the identical appearance of the HCl and NaClO bulk tanks and the inefficiency of the driver's mask, which had been used for several days, corrective measures were implemented by the supplier. A checklist procedure was introduced for all transfers on client premises, and an internal memo regarding the accident was circulated.

As a result of the incident, the on-site HCl was contaminated and had to be removed for destruction, with the tank thoroughly rinsed. The ion-exchanging resins used for demineralized water preparation on the site became unusable, necessitating the deployment of a mobile demineralization unit to cover the downtime during resin replacement. Additionally, the driver was placed under observation at the hospital and granted a one-week work leave.

7.5.4 Falling into the waste pit

The accident statistics for incineration installations encompass various workplace incidents, notably instances of individuals falling into waste pits.

For instance, on October 18, 2006, in Villejust, an employee tragically fell 4 meters to his death into a waste pit at a household waste incineration plant. The young man was discovered unconscious and succumbed to his injuries one hour after being removed from the pit by first responders. The exact cause of death remained undetermined, although the waste fermentation process was known to emit toxic gases, including CO and H2S. It was speculated that his fall may have been precipitated by fainting.

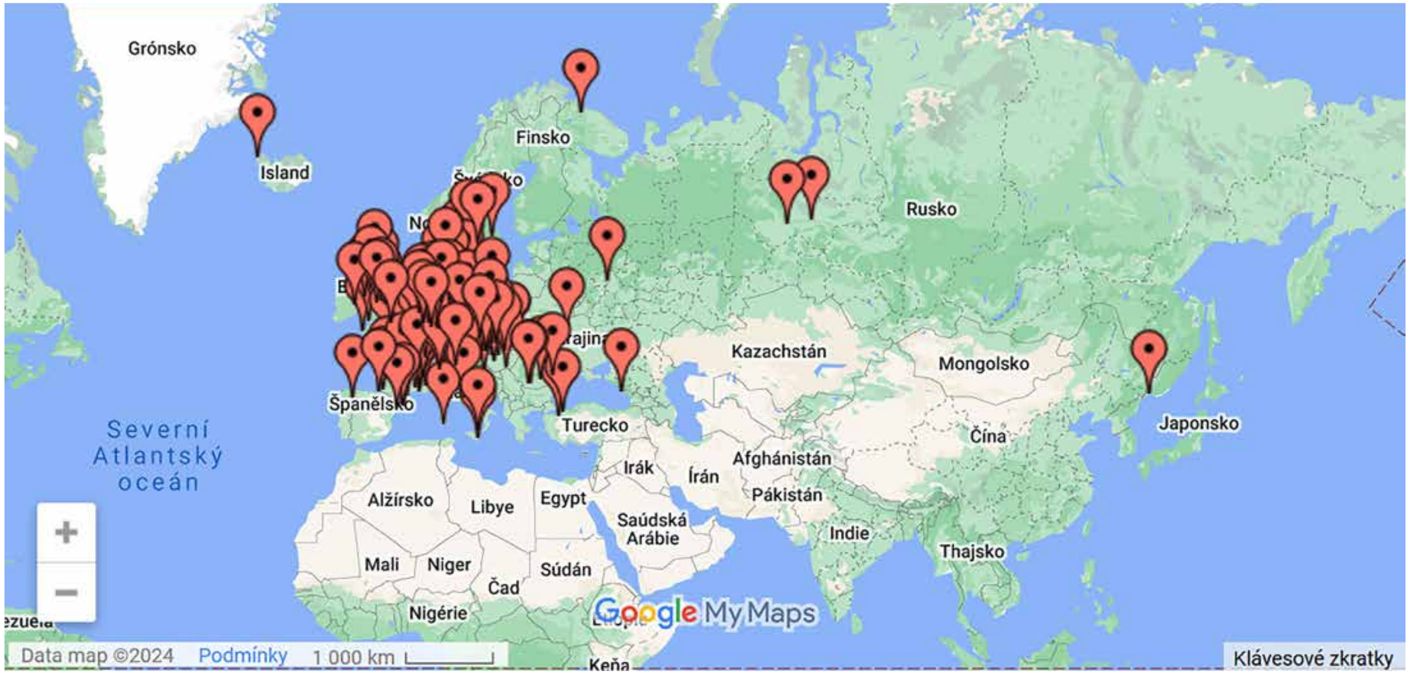
Such occurrences often stem from a combination of personal factors, such as lack of vigilance or fainting, alongside equipment-related issues, such as malfunctioning lorry tipping mechanisms or the absence of physical barriers to prevent falls (MoEES, 2016).

7.6 Summary of the Chapter

In this chapter, we have attempted to document the extent of fires, as well as other incidents leading to endangerment of human health or the environment, which occur during waste incineration, including storage prior to incineration. We have also utilized relatively rare publicly available analyses conducted in France and the USA. Undoubtedly interesting is the rarely available statistics comparing the frequency of interventions by the fire brigade in France, which dispels the notion that accidents and fires are not as common in waste incineration plants compared to other waste processing facilities. In some cases (such as Leverkusen or El Dorado), we have also managed to document the environmental impact of the fire extinguishing materials used, highlighting that their influence cannot be underestimated.

Accidents leading to serious fires are often overlooked or trivialized issues in Environmental Impact Assessments (EIAs) and during impact assessment processes for newly planned waste incineration plants or projects involving co-incineration in cement plants or power plants originally designed for the combustion of fossil fuels such as coal. For pyrolysis plants many proponents claim they are not incinerators and do not need EIA assessments. On both points this is incorrect as pyrolysis plants have been classified and regulated as incinerators for decades in the US and assessed accordingly. Both Rollinson (2018) and Hedlund (2023) note that pyrolysis plants are *inherently hazardous* due to the high flammability and explosive nature of pyrolysis gases requiring even higher levels of assessment than EIA such as ATEX and risk assessments. Therefore, we hope that this chapter also provides information for quality expert feedback in preparing responses to EIA consultation and submission processes.

It's worth noting that there have been many more incinerator fires than those listed above. While not exhaustive, an overview of the incidents of waste incinerators in Europe is available online (Arnika, 2022a; Arnika, 2024). A screenshot of the map of incinerator fires and accidents from the website gives some idea how many such accidents are happening in European countries (see Figures 7.5 and 7.6). Though not comprehensive, this chapter provided some insight into incinerator accidents in Europe, USA, and a few other countries. However no publicly accessible data is available about such accidents in China. In Japan, fires caused by Li-ion batteries have caused significant damage (see Chapter 7.1); however, information about accidents in waste incineration plants is less readily available compared to other countries.



Figures 7.5 and 7.6: *Screenshots of the map on Arnika's website with Google map showing waste incineration accident cases across Europe.*
This map is available also in English on Arnika's English website (https://arnika.org/en/our-topics/waste-plastics/incinerator-accidents-in-european-countries) (Source: Arnika 2022).

8. Alternatives to Incineration

8.1 Municipal Waste

Preventing waste generation is the preferred approach before resorting to landfilling or incineration. These can be achieved through avoiding consumption of products that lead to high waste generation (single use products, excessive packaging etc.), buying products that can be re-used and are durable and buying recyclable and recycled products. This approach can lead to savings in raw materials, energy, and reductions in harmful emissions throughout the production and consumption chain. Once waste is generated, the best practice involves separating and recycling usable materials, which further conserves resources and energy. Residual waste is often burned to reclaim calorific energy[58] in incinerators but with lower energy yields and resource loss. Embedded energy in the product from extraction of raw resources, production and transportation to market, is lost when the item is burned. Landfill locks up waste resources and generates methane gas, potent climate change emission. This can be mitigated by methane extraction and conversion to electricity

Photo 8.1: *Composting biowaste diverts one-quarter or more from municipal waste (Āriņa et al., 2023) and is also regarded as an important part of waste prevention. However, biowaste needs to be sorted directly at the source. Composting mixed biowaste from municipal waste is not suitable and, considering the presence of toxic substances, can even be dangerous. Arnika has made an interesting video about composting in the Broumov region (Arnika, 2017). Photo: Arnika.*

58 Calorific value of waste is the direct heating value in kilojoules (kJ) from burning the waste and is only a fraction of the embedded energy contained in the product that has become waste. Recycling retains much of the embedded energy of a product that has become waste as well as the material resources.

Figure 8.1: *Waste hierarchy according to Zero Waste Europe. (Source: Simon, 2019)*

via turbines. However, the main problem with landfill is groundwater contamination caused by toxic leachate liquid leaking from the landfill.

Comparatively, waste incineration might seem like an ideal solution compared to landfilling. However, incinerators themselves produce solid waste that must be deposited in landfills, part of it even in specially secured hazardous waste landfills. Although part of this waste (bottom ash) is currently not considered hazardous according to current criteria, this is likely to change in the future with broader monitoring of a wider range of harmful substances (see Chapters 3.3, 5.1.7, and 5.1.8). An ideal waste management approach for municipal waste can be considered "zero waste", which implements the higher principles in the waste management hierarchy rather than landfilling and waste-to-energy. Its aim is to minimize waste that needs to be landfilled or incinerated.

Generally, these systems consist of waste sorting, composting biodegradable components,[59] and recycling other waste (such as glass, plastics, metals, and other components). An important part of such systems is also reusing some products after repair, cleaning, or modification. It is clear that a completely waste-free system is currently not feasible, and the waste that cannot be further utilized will contain toxic substances or may not be recyclable. Addressing this issue can involve redesigning products to be free from added toxic substances, making them recyclable and avoiding the manufacture of non-recyclable products. By preventing waste generation and implementing zero waste systems, the risk of dioxin and other POPs (persistent organic pollutants) leakage into the environment decreases. Mapping and estimating this leakage reduction was attempted in a 2006 study in Central and Eastern European countries. It also described the case of Palárikovo in Slovakia, where the zero waste system helped drastically reduce the amount of landfilled waste from 1,300 tons in 1999 to 330 tons in 2005 (Havel et al., 2006).

Photo 8.2: *The foundation for setting up Zero Waste in Palárikovo was the analysis of the contents of bins. Photo: Spoločnosť priatelov Zeme, Košice.*

Zero waste is a more intricate system than waste incineration. Unlike incineration, it requires citizens' motivation and at the same time, acceptance of responsibility for what people purchase and how much waste they produce, as well as higher demands for waste sorting. However, this is not sufficient; a citizens' approach requires a combination of certain

59 Li et al. (2015) also suggested „that the option of compost with material recovery facility treatment may pose less negative health impacts than other waste management options. They also concluded that waste-to-energy incineration had the lowest non-cancer risks under normal operation but posed the highest cancer risk compared to other waste management methods (see Chapter 6).

top-down regulations (reducing the content of toxic substances in products, providing access to bins for sorted waste, etc.) and collaboration with companies, which are expected to take responsibility for their products throughout their life cycle under the Extended Producer Responsibility (EPR). While companies must currently ensure the take-back of their products (packaging, e-waste) under EPR, new legislation also emphasizes ecodesign (product lifespan, repairability, and recyclability).

The first region to embrace the zero waste system in 1996 was the Australian capital, Canberra (Arnika, 2020). Since then, many regions and cities have adopted this concept. Their list can be found on Zero Waste Europe website.[60]

8.1.1 Treviso, Italy

An example of good practice in the field of zero waste is the company Contarina (Simon, 2018), operating in the Italian province of Treviso with a population of 550,000 and an area of 1,300 km2. This company specializes in processing municipal waste for the entire region.

Waste in the region is sorted into five categories: wet waste, packaging (plastics, glass, metals), paper, plant-based biowaste, and mixed waste. There are collection centers for other types of waste, located in 49 locations. The total production of municipal waste in this area amounts to 413.34 kg per inhabitant (excluding waste from large shopping centers and businesses). In 2020, nearly 90 % of all waste in the region was sorted. As a result, the average production of mixed waste per inhabitant in the same year was only 42 kg, compared to 193 kg in Italy and 260 kg in the Czech Republic per inhabitant.

60 https://zerowasteeurope.eu/

Wet waste and plant-based biowaste are composted, and packaging materials and paper are further sorted and sent for processing. Even paper diapers are processed. From one ton of material, 150 kg of pulp, 75 kg of plastics, and 75 kg of highly absorbent materials are obtained. Since 2020, the outputs from this facility have been certified and can subsequently enter the market. Contarina is capable of processing street sweepings as well. These are cleaned, divided, and utilized according to different fractions.

High levels of waste sorting are achieved through awareness campaigns, collection systems, and citizen motivation. The waste collection company itself conducts awareness programs, and for this purpose, an educational academy has been established, providing training for both children and company employees.

Each household has its own waste bin (equipped with a chip) and also has its own account to track its waste production. The collection method depends on the type of residential area. A uniform waste fee is set for the entire region, consisting of a fixed amount (60 %) and a variable amount depending on the production of mixed waste. Households that compost at home receive a discount on the fee (30 %). Conversely, the collection of biowaste is subject to a fee. In 2020, the average fee per household reached 196 euros, approximately 80 euros per inhabitant.

Contarina is an example that should be followed not only in waste management but also in the operation of the waste collection and handling company itself. The company itself aims to fulfill the UN's Sustainable Development Goals by 2030.

8.1.2 Vrhnika, Slovenia

While in Slovenia in 1994 inadequate landfill capacities began to be addressed, leading in 2004 to a significant increase in landfill fees, the town of Vrhnika

Photo 8.3: *The Slovenian town of Vrhnika had approximately eighteen thousand inhabitants in 2013 and produced 80 kg of mixed waste per inhabitant. Photo: Sl-Ziga – Own work, Public Domain, https://commons.wikimedia.org/w/index.php?curid=6118754*

decided to implement separate waste collection. As a result, they managed to reduce landfill costs by more than half. The annual waste production per inhabitant compared to 2003 (201 kg) was reduced to 80 kg in 2013. In 2018, the municipality separated 83 % of the waste (McQuibban, 2021).

Initially, glass, paper and cardboard, plastics, metal packaging, residual waste, organic waste, hazardous waste, bulky waste, construction, and demolition waste were sorted. Recyclable municipal waste was collected at so-called eco-stations in the streets, where residents could bring their waste. In 2020, residents were encouraged to take waste directly to a collection center. In return, they received points after weighing the waste, reducing their monthly waste bill (pay-as-you-throw system). Thanks to this system, residents independently took approximately 30 tons of waste per year without the help of waste collection services (Van Vliet, 2014).

The intensity of residual waste collection, initially set at once a week, was reduced to once every two weeks in 2011, and since 2013, it has been collected only once a month. Bulky waste is collected in two ways – residents can deliver it directly to a collection center or request collection from home. All bulky waste is dismantled, and most materials are handed over for recycling. This is related to the reuse center (DEPO), which is used to upcycle waste into desired goods and utilize items that would otherwise end up in landfills. Items are repaired, improved, or disassembled into usable parts, from which something else is produced and then sold to the public at reasonable prices.

Education begins in schools, with lectures attended annually by 1,500 children and young people from across Slovenia. Regarding children, Vrhnika introduced nursery schools in cooperation with Ecologists Without Borders as a pilot project involving reusable diapers to prevent the disposal of disposable diapers in landfills (Van Vliet, 2014).

8.1.3 Kamikatsu, Japan

Kamikatsu, a town of 1,700 inhabitants in Japan, had its incinerator since 1991, which was closed in 2001 due to a lack of funds to acquire a "dioxin filter". In connection with the closure of the incinerator in 2001, the town decided to stop sending any waste to landfills or incinerators (Shenyoputro & Jones, 2023). "People saw that the incinerator was harming both the environment and their health. Therefore, a zero waste program was created," explained Akira Sakano, deputy chairman of the Zero Waste Academy, a non-profit organization that supports Kamikatsu in its environmental policy (Žák, 2017).

Their ambition was to produce no waste by 2020. The whole process was preceded by a five-year pilot project launched in 1998, primarily aiming to separate waste at the source. At the time of implementation, there were 22 waste categories, which increased to 45 in 2020. In the same year, Kamikatsu recycled 81 % of waste, which is a 65 % increase compared to 2000 (Shenyoputro & Jones, 2023), and an impressive number compared to Japan's average of 20 % (Wikipedia, 2022b). The evolution since 2000 is summarized in Figure 8.2.

This brought savings to the town because better waste sorting managed to reduce annual waste disposal costs by a third (Shenyoputro & Jones, 2023; Žák, 2017).

The system in Kamikatsu is sophisticated; it is not just about sorting and collecting waste. Citizens are motivated financially because the city introduced loyalty point cards for waste sorting, which can be exchanged for shopping vouchers. The city also purchased electric composters (Shenyoputro & Jones, 2023). Nothing is wasted. There are shops in the city where people can leave clothes, furniture, and things they no longer want, and anyone can take them for free. And when there's little interest? They take them to a factory where women turn them into something new – like a plush bear from an old kimono. "We are trying more and more to focus on changing our lifestyle," said Akira Sakano (Žák, 2017).

Kamikatsu residents have built a new town center entirely from recycled materials. An eight-meter-high window wall and other openings bring cool breezes in the summer, while carbon-neutral radiant heat warms the structure in winter (Wang, 2016). Similarly, the Zero-waste management center was built (see Photo 8.4); (Wikipedia, 2022b).

Photo 8.4: *Zero-waste management center in Kamikatsu built entirely from materials that were supposed to end up as waste. Photo: Own work (投稿者撮影), CC0, https://commons.wikimedia.org/w/index.php?curid=108202162.*

Three key principles in the town of Kamikatsu can be identified: recycling and resource utilization through source separation; cooperation between citizens and the government to reduce the tax burden; and encouraging

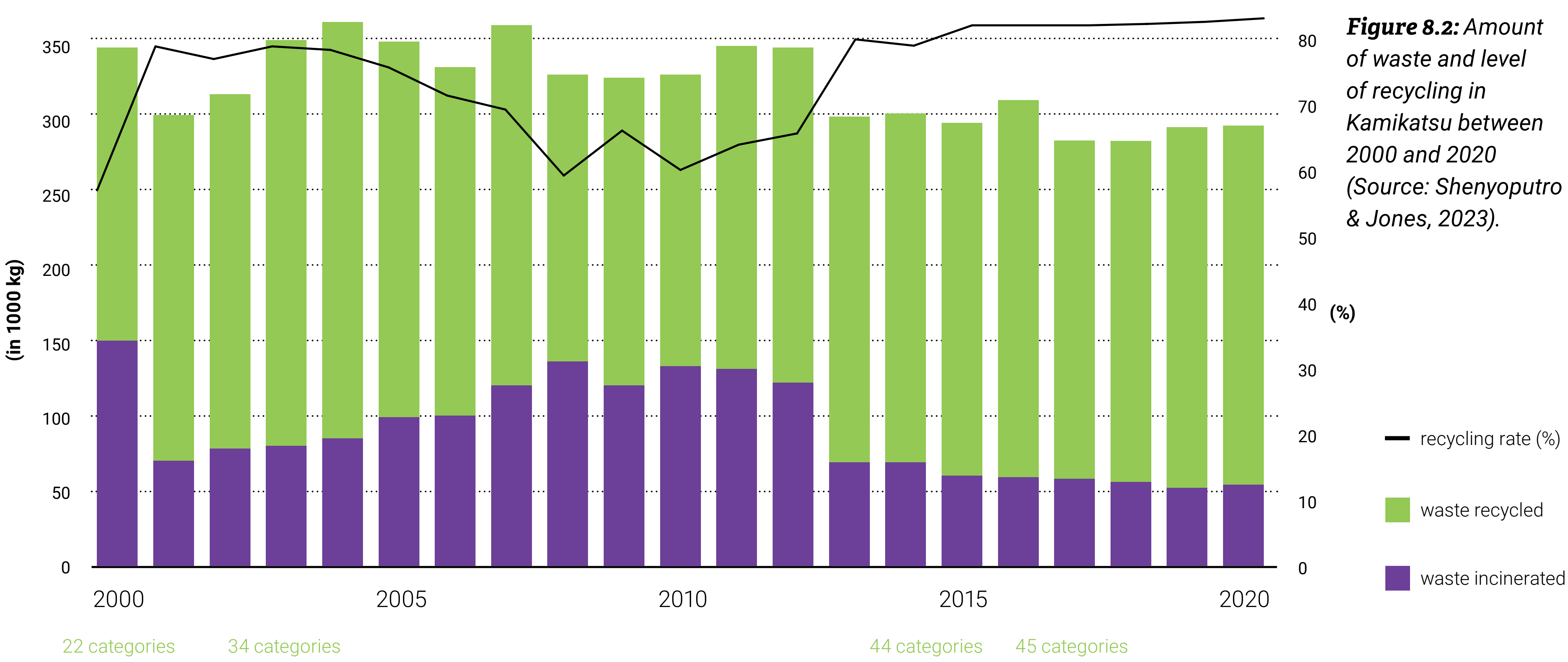


Figure 8.2: *Amount of waste and level of recycling in Kamikatsu between 2000 and 2020 (Source: Shenyoputro & Jones, 2023).*

manufacturers to reevaluate and redesign their products sustainably because the remaining 19 % of waste is considered non-recyclable (Shenyoputro & Jones, 2023).

8.2 Hazardous Waste

Hazardous waste encompasses a relatively wide range of waste that is deemed hazardous due to various properties, necessitating specific methods for handling them. Among the hazardous properties of waste can be toxicity, carcinogenicity, mutagenicity, infectivity, ecotoxicity, and others (European Union, 2024).

One group of hazardous waste, namely fly ash, has been addressed in chapters dedicated to residual waste from incineration (see Chapter 3.3) or in chapters focused on various toxic substances (see Chapter 5). Another significant group consists of medical waste, generated both in hospitals and other medical facilities, primarily hazardous due to their potential infectivity. Therefore, a separate Chapter, 8.3, will be devoted to them. A specific group includes waste containing mercury, which will be the focus of chapter 8.3.6. Another relatively specific group involves waste containing substances resistant to decomposition, which persist in the environment long-term, referred to as Persistent Organic Pollutants (POPs). A separate chapter (8.3.7) will also be dedicated to them. Technologies primarily aimed at breaking down POPs are applicable to a far broader range of hazardous waste, especially those containing halogenated compounds.

There are also wastes that contain hazardous substances and for which the only viable option is to place them in a secure landfill, such as waste containing asbestos, which does not break down by burning and poses a risk of dispersing its fibers. Similarly, it makes no sense to incinerate, for example, ash from domestic heating systems, which may contain hazardous substances in the form of heavy metals or PAHs (see Chapter 5.3 and 5.1.6). Burning it merely transforms it into another type of ash and partly into fly ash, but it certainly does not significantly reduce its volume. Therefore, it is better to dispose of it in a special landfill, regardless of the fact that most of the ash will be resolved with the end of coal burning in domestic heating systems since it will no longer be produced. However, it is pointless to have illusions that ash from domestic heating systems can be utilized, and in no case can its use as an additive to soil or for road and sidewalk gritting in winter be recommended (in Europe and cold climate countries), precisely due to the content of heavy metals and possibly organic substances.

8.3 Medical Waste

Waste from medical facilities comprises 75% to 90% of general waste, similar in composition to municipal waste. The remaining percentage forms hazardous waste Minimization techniques include sorting and recycling. Thorough recycling could reduce this amount to a mere 5 % to 3 %. The strict separation of infectious and hazardous material is a crucial step, compliance with which can lead to significant resource and cost savings in healthcare facilities. Recycling involves reusing material from waste. A significant amount of medical waste is similar to waste from offices or hotels – cardboard, paper, or food remnants.

Hospitals can introduce a simple program to divert these materials from infectious or toxic waste, which could contaminate them and thus render them unfit for further recycling. The options for recycling in healthcare facilities were extensively detailed by Głuszyński in his contribution "How to Create and Implement a Waste Minimization Program in a Hospital – a Practical Example from Poland" for a conference in 2012 (Kristian et al., 2012).

The UNEP handbook from 2006 designated the healthcare sector as one of the main sources of dioxins and mercury in the environment. These substances escape primarily due to improper handling and damage to mercury-contaminated equipment such as thermometers and blood pressure gauges (UNEP, 2006). However, the Minamata Convention on Mercury has reduced the global volume of mercury containing medical devices via mandatory phase-outs for most equipment. A major exception being dental amalgam which still does not have a firm phase out date though its use is declining in many countries.

The reduced residual waste from healthcare, after separating the non-infectious portion, can be processed using non-incineration technologies (Kristian et al., 2012; Petrova et al., 2008). Selecting these technologies requires knowledge of the waste's characteristics, quantity, and place of origin, and adherence to the technology description provided by the manufacturer. Non-incineration technologies can rid infectious waste of its hazardous properties, its "infectivity", allowing it to be handled as municipal waste. Implementing non-incineration technologies in healthcare operations significantly reduces the environmental impact and ensures the health safety of employees (Emmanuel, 2012; Emmanuel & Hrdinka, 2003). They can be categorized based on various criteria; however, concerning the decontamination process, they can be divided into four categories – low-temperature, chemical, radiation, and biological (Emmanuel, 2012; Emmanuel & Hrdinka, 2003). Their subsequent description is based on the compendium published by the World Health Organization (WHO); (Emmanuel, 2012).

8.3.1 Low-Temperature Processes

For waste disinfection, either hot steam or dry heat at temperatures of 93–177 °C is utilized. The action of hot steam forms the basic principle of autoclaves and retorts. These technologies commonly process tissue cultures and strains, sharp instruments, materials contaminated with blood and limited amounts of fluids, waste from infectious departments, surgical waste, laboratory waste (excluding chemicals), and so-called "soft" waste originating from healthcare.

A potential issue with these facilities might be the surrounding odor if adequate ventilation is not ensured. It's crucial to ensure thorough waste sorting to prevent the entry of hazardous materials into the chamber. The unequivocal advantage lies in the low acquisition cost and a wide range of products available in the market, allowing for the selection of the most suitable size. This category includes the use of autoclaves or retorts (Bondtech, ETC, Mark-Costello, Sierra Industries, SteriTech, and Tuttnauer), advanced autoclaves with functions like crushing, vacuuming, continuous filling, or stirring, etc. (San-I-Pak, Tempico Rotoclave, STI Chem-Clav, Antaeus SSM, Ecoletec, Hydroclave, Aegis Bio-systems, Log Med), microwave systems (SINTION, Medister), electrothermal deactivation, or hot air.

Photo 8.5: *Autoclave by the French company Ecodas installed in one of the French hospitals. Photo: Jindřich Petrlík, Arnika.*

8.3.2 Chemical Processes

The function of chemical processes is based on disinfection in the presence of chemical agents. Various chemicals are used – chlorine compounds, ozone, calcium oxide, sodium and potassium hydroxide, peroxyacetic acid, and others. Some chemicals do not change the physical appearance of waste, while others trigger chemical reactions altering the physical appearance and chemical properties. Technologies based on non-chlorine chemical agents are advantageous as they do not produce secondary chlorinated products. Chemical action-based technologies can process waste including cultures and strains, sharp objects, anatomical and pathological waste including blood and body fluids, surgical waste, waste from infectious departments, laboratory waste (excluding chemicals), and so-called "soft" waste. These technologies are mostly automated and user-friendly; no combustion by-products are produced, and liquid waste can be discharged into the regular waste system. However, the use of chemicals poses certain risks to workers, and the chemicals used may contaminate the air and waste water if present in the waste. Methods based on chemical agents without chlorine content include Steris EcoCycle10 (peroxyacetic acid), Lynntech (ozone), Delphi MEDETOX CerOx (metals as catalysts), or WR2 (alkaline substances).

8.3.3 Radiation Processes

Radiation processes utilize electromagnetic radiation for the decontamination of medical waste. It is a highly computer-controlled and automated method. For technologies based on electron beam radiation, it is necessary to use shredders or other mechanical devices to reduce the volume and homogenize the waste so that its origin is not recognizable. Specifically, UV radiation, cobalt 60, or electron beams are employed (BioSterile Technology).

8.3.4 Biological Processes

Biological processes decontaminate medical material using a blend of enzymes. This technology is suitable for large volumes of waste but is rarely used as of now (Kristian et al., 2012). A prototype by Bio Conversion Technologies was tested in Virginia.

8.3.5 Case Study: Comparison of Non-Incineration Technologies with Incineration

An example of very limited knowledge regarding alternative technologies for treating infectivity in medical waste using non-incineration technologies is the appendix (Dombek, 2018) to the EIA documentation for the expansion of the Ostrava hazardous waste incinerator in Czech Republic (Mynář et al., 2018). Because this is a case study of how such documentation deals with this issue, we will attempt to analyze how this appendix skews in favor of incinerating medical waste, often without adequate evidence.

The incomplete list of technologies and their characteristics suggests that the appendix's author likely isn't familiar with the technology compendium for medical waste processing by the WHO (Emmanuel, 2012) and might not have acquainted themselves with the autoclave technology used, among other places, in areas of Africa affected by Ebola epidemics (UNDP, 2015). This autoclave, combined with waste shredding, reduces waste volume by 85 %, making its efficiency comparable to incineration in this regard. It doesn't require waste transport over long distances since it can be installed directly in larger hospital facilities. Similarly, transport is eliminated for other facilities that can be resized according to healthcare facility needs (redakce Průmyslová ekologie, 2018).

As a claimed downside of non-incineration technologies, the authors assessing incinerators see that "the capacity of all the mentioned

Photo 8.6: *Jorge Emmanuel (left) and Johan Hoffman (right), co-designers of a new autoclave with one of the first units manufactured in 2014. Dr. Emmanuel was the main technical expert from the UN Development Program (UNDP)/Global Environment Facility (GEF)/World Health Organization (WHO)/Health Care Without Harm Global Healthcare Waste Project; Johan Hoffman (right) – executive director and chief engineer of Medi-Clave company, which co-designed and manufactured the autoclaves (HCWH, 2014). Photo: HCWH.*

non-incineration methods is significantly smaller compared to incineration technologies" (Dombek, 2018). However, this isn't their disadvantage; on the contrary, this enables easy installation of these technologies directly in healthcare facility premises, avoiding their transportation and the risks of accidents during transport. Waste transportation is a significant cost item in waste management budgets and a source of emissions of additional pollutants.

Other criticisms of using non-incineration technologies include that they do not reduce the volume and weight of decontaminated waste significantly and that the process does not guarantee the destruction of all pathogenic organisms.

Waste shredding is often already part of modern autoclaves (Emmanuel, 2012; UNDP, 2015). At the same time, the assertion that "non-incineration methods require additional subsequent operations, which will further increase the costs of the whole process" is flawed when it doesn't admit that waste from incinerators also requires and will require more "subsequent operations" in the future. Even for waste from smoke purification in hazardous waste incinerators, stabilization before landfill disposal is assumed, which obviously "increases the costs of the whole process," excluding the fact that some of them should undergo decontamination in a non-incineration technology capable of breaking down POPs (see Chapter 3.3.1). Given the problematic nature of waste from incinerators, their use for medical waste is at least questionable, and they also require further "subsequent operations."

Dombek (2018) further argues that "non-incineration methods may not significantly reduce the volume and weight of decontaminated waste", which he illustrates using the worst chosen example, namely microwave technology. Some autoclaves equipped with shredders reduce waste volume to as low as 15 % of the original volume and 50 % of the original weight. Thus, their effectiveness is comparable to incinerators in terms of waste volume reduction, without generating new toxic substances like dioxins in incinerators.

As an argument in favor of incinerating medical waste, Dombek (2018) states that "microwave technologies and autoclaves may not eliminate all

Photo 8.7: *A medical waste incinerator might appear inconspicuous at first glance with chimneys – this is an incinerator next to the hospital in Prague – Motol. Photo: Jindřich Petrlík, Arnika.*

Photo 8.8: *Medical wastes in incinerators can be identified by packaging in colored plastic bags. The photo shows medical waste prepared for incineration in the hazardous waste incinerator in Trmice, northern Czech Republic. Photo: Jindřich Petrlík, Arnika.*

Photo 8.9: *Infectious waste in special containers prepared for incineration in Trmice. Photo: Jindřich Petrlík, Arnika.*

Photo 8.10: *A fire occurred in the incinerator primarily for medical waste in Plzeň, Na Slovanech in June 2017, requiring a two-day intervention by firefighters. Photo: HZS Plzeňského kraje.*

pathogenic bacteria…," without demonstrating that this effectiveness has been extensively monitored in incinerators for medical waste. An older US EPA study stated that there's a lack of information for such evaluation, in other words, it hasn't been examined (US EPA, 1990). There's no scientific study focusing on the occurrence of pathogenic organisms in residual waste from medical waste incinerators. One study mentions that the incineration process of municipal waste may allow survival of bacteria like Salmonella (Klee & Peterson, 1971). The generally accepted hypothetical assumption that "everything is destroyed" in the high-temperature incineration process, without considering practical experience and empirical research, may not hold true. Modern non-incineration devices for medical waste decontamination have sensors for the elimination of biologically active microorganisms (PE, 2018). Nothing similar exists in incinerators.

8.3.6 Handling Mercury-containing Waste

Specific handling is required for waste containing mercury. Burning such waste poses significant risks to the environment because mercury vaporizes even at normal (room) temperatures and contaminates the air. Incineration does not remove "mercury compounds" from waste (Dombek, 2018), as claimed in one of the appendices of the EIA documentation for the expansion of the hazardous waste incinerator in Ostrava, Czech Republic (Mynář et al., 2019). Therefore, from waste containing mercury, it rapidly transfers into gaseous emissions in the incinerator, at best ending up in fly ash and other residues from gas cleaning (see Figure 3.4).

Incinerators are by no means recommended for waste containing mercury as mercury vapors can escape air pollution control systems (Basel Convention, 2012). The Minamata Convention identifies waste incinerators as one of the main sources of mercury emissions (UN Environment, 2016). Indirect thermal desorption is one way to extract mercury from mercury-containing waste. Concentrated mercury is best stabilized into mercuric sulfide and then deposited in below ground storage. Compared to other methods, this compound exhibits "very low leachability of mercury and high durability" (Rodríguez et al., 2012). Such mercury stabilization operations take place for example in Chvaletice, Czech Republic (Plachý, 2022).

8.3.7 Waste Containing Persistent Organic Compounds (POPs)

The Stockholm Convention in Article 6 defines the basic rules for handling waste containing POPs (Ministry of the Environment, 2006). These rules were further specified by experts from Greenpeace and the International

IPEN network (IPEN, 2010). Waste containing POPs must be disposed of or irreversibly transformed using environmentally non-damaging methods that largely meet the following criteria (IPEN, 2010):

- Almost 100% efficiency in eliminating POPs – concerning all input (during calculation) and output phases (gaseous, liquid, or solid).
- To ensure, if possible, 100% efficiency in eliminating POPs, all output components must be analyzable.
- If necessary, it must be possible to return waste back into the POPs disposal process.
- Prevention of uncontrolled releases of toxic substances during the process.

Technologies for decomposing POPs in waste have been evaluated for several decades. Some of them emerged due to the demand for technologies to dispose of chemical warfare agents or to clean up sites contaminated due to their use as military bases, where PCB was a common contaminant. Besides incinerators and cement kilns, a range of technologies capable of decomposing POPs with often higher efficiency has been developed, evaluated using the so-called Destruction Efficiency (DE) criterion. A summary of these technologies is provided, for instance, in the comprehensive IPEN network study from 2020 (Bell, 2020). Some of them are also described in the general guidelines for handling POPs waste updated within the Basel Convention on the transboundary movement of hazardous wastes (Basel Convention, 2023).

The efficiency of these technologies is also evaluated in other studies by the US EPA (2010) and UNEP (2004). At the same time, it's necessary to consider the potential for the creation and emission of unintended by-products of POPs, such as PCDD/F, dioxins, and other substances. This can also be addressed by a document prepared by the expert group for BAT/BEP of the Stockholm Convention (UNEP – EG BAT/BEP, 2006). Some of these technologies, intended for the decontamination of dioxin pollution in Spolana company in Czech republic, were presented at an international conference held in Prague in 2003 (IPEN et al., 2003). Eventually, a combination of indirect thermal desorption (ITD) and BCD was used for the decontamination (Kubal et al., 2004). Here is a list of at least some fundamental non-incineration technologies:

- Gas Phase Chemical Reduction (GPCR)
- Ball Milling – Mechanical-Chemical Destruction
- Supercritical Water Oxidation (SCWO)
- Alkaline Catalytic Decomposition (BCD)
- Catalytic Hydrogenation (CHD)
- Reduction by Alkali Metals (SR)
- Copper-assisted Catalytic Dechlorination (CDC)

Some of these have been described more closely in chapter 3.3.1, so we'll avoid detailed descriptions here. Some of them are also used for the decomposition of other halogenated substances, making them suitable for the disposal of a much wider range of hazardous wastes than just those containing POPs. In particular, GPCR and SCWO have been used for chemical weapon and nerve agent disposal by the US military.

Besides these technologies, gasification or plasma treatment of waste are considered alternatives to incineration of hazardous waste. However, they are merely other forms of waste incineration, so they cannot be classified as alternatives to incinerators, especially considering that they do not meet the criteria described above for their environmental impacts.

8.3.7.1 CreaSolv®

As mentioned in chapter 3.3.1, some technologies and waste require pre-treatment. Among such waste is polystyrene treated with BFR, specifically hexabromocyclododecane (HBCD). For this purpose, a method called

CreaSolv® was developed. Detailed information in Czech about this process is contained in an info sheet issued in 2012 (IPEN, 2010).

The material entering the reaction usually comprises at least 75 % plastic with BFR content (Malcolm et al., 2011). Brominated flame retardants are removed from the plastic (in this case, polystyrene) by dissolving the polymer and separating it from brominated and other additives, which are later concentrated. The amount of solvent used is very low in comparison with the processed plastic (< 1 %), as the vast majority returns back into the process, and only a small fraction, containing BFR, exits the equipment. In the presence of metals, an insoluble fraction rich in metals is formed alongside usable polymer recyclate (Schlummer et al., 2004). In Canada, it was possible to remove PBDD/F, present as co-contaminants, along with BFR (Schlummer et al., 2008). Concentrated BFR can be decomposed using another non-incineration technology or irreversibly transformed as agents during industrial processes.

This process can also handle, for instance, discarded mobile phones (after removing the battery), providing polymer particles suitable for compression molding and injection molding (Schlummer et al., 2004). In another case, expanded polystyrene waste was treated to produce polystyrene that can be re-expanded, possessing properties comparable to primary expanded polystyrene (Mäurer & Knauf, 2005).

A comparison of four different processes for handling electronic and electrical waste containing BFR found CreaSolv® to be the best in terms

Photo 8.11 *The PolyStyreneLoop operation in the Netherlands is essentially CreaSolv® at a fully commercial scale. Photo: Sustainable Plastics, https://www.sustainableplastics.com/news/new-life-breathed-polystyrene-loop-initiative*

of energy consumption and potential for photochemical oxidation. According to Freer's study (2005), CreaSolv® along with Centrevrap® has the least environmental impact. CreaSolv® is interesting due to the low solvent loss and its high renewability. CreaSolv® is in full commercial operation in the Netherlands. However, the EU member states do not sort enough old polystyrene containing HBCD used for building insulation, and the operation lacks enough material for processing.

9. Economics and Financial Aspects of Waste Incineration

From a financial standpoint, municipal waste incinerators pose a greater problem than hazardous waste incinerators.[61] Burning one ton of hazardous waste can generate much higher fees than for municipal waste. According to a study by the international GAIA network, in the USA, an investment of between 190 million to 1.2 billion USD is required to build a municipal waste incinerator with a capacity of one million tons per year. Operating costs for Waste-to-Energy facilities rank among the highest compared to other waste management methods like composting, anaerobic digestion, or landfilling (Moon, 2021).

The economics of waste incineration in WtE facilities are rather complex, much like most waste management methods. Let's go through it based on the following areas:

- Investment in construction
- Maintenance and repairs
- Operating costs and the price of waste incineration
- Related costs and fees
- Unaccounted costs caused by waste incineration.

61 Some economic aspects of hazardous waste incineration are discussed in Swan Hills case study – see Chapter 7.2.1.

Compared to landfilling, waste incineration is more expensive even in the Czech Republic. Consequently, landfill fees have been increased. This is arguably acceptable, but if fees for waste incineration are further raised, considering its negative impacts on the environment and public health (for which the state or municipalities bear the cost), only then will economic instruments help divert waste from environmentally unfriendly methods towards more sustainable ones: composting for organic waste, recycling for other waste, or even leading to waste reduction.

9.1 Investment in Construction

The initial investment in constructing WtE facilities varies depending on their size and significantly differs based on the country where they are built. The more sophisticated the country's environmental protection approach, the higher the investment costs. Other factors influencing the investment include labor costs and input materials, which are common considerations in construction. The construction of a large WtE plant (WtE) in Mallorca with a capacity of 730,000 tons.y^{-1} cost around 500 million EUR (approximately 11.5 billion CZK) (Environmental Justice Atlas, 2014; ESFC, 2023a).

Photo 9.1 *Municipal Solid Waste Incinerator in Likeng, China.*
In China, it is cheapest to build new WtE. Photo: Jitka Straková, Arnika.

For the investment in a WtE facility to be profitable, it must guarantee capacity utilization for 25 to 30 years (ESFC, 2023a). Therefore, future operators obligate municipalities through contracts to guarantee a continuous supply of sufficient waste for an certain period of time. This often leads to the blocking of other, more environmentally friendly waste management methods.

The cheapest place to build a new WtE facility is in China. While in the EU, UK, Canada, or the USA, the investment for one ton of incinerated waste capacity per year ranges between 600 and 1,000 EUR, in China, it's 250 EUR, and in developing countries, this investment rarely exceeds 400 to 500 EUR (ESFC, 2023b). A similar comparison was provided by an older study, stating that in China, one ton of annual WtE capacity could be installed for 250 USD, while in the USA, the average was 850 USD (Wu, 2018).

During their construction and maintenance, municipal waste incinerators often rely on loans and subsidies from the public budget. Out of four large-capacity municipal waste incinerators currently operational in the Czech Republic, only one (in Chotíkov), did not receive any subsidies, although it sought them.

Incinerators in Brno (WtE SAKO Brno) and in Malešice, part of Prague (WtE Malešice) were built before 1989. WtE Malešice (Prague) was completed in 1997 but operated as an unauthorized construction for one year without a valid building permit. Both of these incinerators underwent or are undergoing reconstruction or capacity increases. Both received substantial grants from EU funds (see the following Chapter). While the EU prohibited funding the construction of new incinerators, such restrictions did not apply to the modernization of those already built.

9.1.1 Case Study: WtE Termizo Liberec in Czech Republic

The construction process and financing of the Termizo WtE in Liberec are described in detail in a comprehensive study by the Hnutí DUHA (Kropáček, 2003). The investment was financed through a loan, for which the city of Liberec, owning 77 % of Termizo's shares, provided a guarantee. The original loan of 1.35 billion CZK was provided in 1996 by the bank (Investiční a poštovní banka) to Termizo, and the business plan did not account for additional interest costs. The city of Liberec also obtained a commitment from the State Environmental Fund of the Czech Republic to cover payments for interest commitments, which almost fell through due to deficiencies in issuing building permits.

The story of the WtE Termizo Liberec incinerator perfectly illustrates how a city can become subservient to a company incinerating its waste. By signing the so-called Guarantor's Declaration, the city of Liberec committed to block more environmentally friendly waste management methods.

Photo 9.2 *Construction of the municipal waste incineration plant in Liberec was largely funded by all citizens of the Czech Republic. Photo: Marek Jehlička (skyworker.cz).*

The declaration contained a clause stating that the guarantor (the city) commits to "...send all municipal combustible waste produced in the city or municipality for thermal utilization in the WtE facility." This commitment does not encourage, for example, composting organic waste, which typically constitutes about a quarter of all household waste.

The city of Liberec was unable to repay the loan due to faulty economic assumptions regarding the profitability of heat production through waste incineration. Eventually, the loan ended up among the problematic projects transferred to the Czech Consolidation Agency (ČKA) after the bank collapse. In April 2002, the government approved the sale of Czech Consolidation Agency's receivables, amounting to 1.92 billion CZK, to the main guarantor, namely the city of Liberec, for a significantly reduced price of 715 million CZK. Subsequently, in June 2002, the Office for the Protection of [Economic] Competition approved the transaction regarding public support. Thus, WtE Termizo Liberec was partly paid for by taxpayers from across the Czech Republic – the expenses per capita exceeded 115 CZK.

Based on the technical advisors' policy, the Termizo Liberec WtE facility was built without a dioxin filter, which was necessary for the incinerator to comply with the European limit for dioxin emissions at 0.1 ng $TEQ.m^{-3}$. Its construction in 2002 represented an additional investment of 250 million CZK. The city did not have these funds, so it sold the majority of shares to particular a member of the PPF Group. This story shows that the Termizo WtE in Liberec could not exist without repeated injections of funds from the state.

9.1.2 Case of the Incinerator in Plzeň – Na Slovanech, Czech Republic

Similar to the city of Liberec contributing to the construction of a municipal waste incinerator, Plzeň supported the construction of a hazardous waste incinerator at Na Slovanech. The controversial hazardous waste incinerator in Plzeň, Na Slovanech, was built by the Navrátil company in the early 1990s using a loan from the Plzeň City Council. The city ended up paying for it by eventually repaying the loan, including interest, which amounted to nearly a hundred million Czech crowns, using the proceeds from the sale of municipal property. Therefore, the T. O. P. Eko Plzeň company, which operated the incinerator until 2009, was more than 90 % owned by the city. Since January 2009, the city became the sole owner of the company and consequently the operator of the hazardous waste incinerator SITA CZ, now under the Recovera group of Veolia. The incinerator's

Photo 9.3: *Hazardous waste incinerator in Plzeň, Na Slovanech (pictured in 2011), narrowly missed the obligation to undergo an environmental impact assessment; the city of Plzeň paid the debt for its construction. Photo: Jindřich Petrlík, Arnika.*

capacity is 2,500 tons of waste per year. The incinerator's story is a perfect example of how a city can economically bear the consequences of investing in waste incinerator construction. Throughout its history, the incinerator has also faced several environmental impact problems. Right at the beginning of its existence, it evaded an assessment of these impacts under the Environmental Impact Assessment Act because the law was not in effect when the construction proceedings began.

A study prepared for a bank Komerční banka in 1991 regarding the loan for the hazardous waste incinerator in Plzeň recommended, among other things: a) approving an increase of 30 million CZK annually in operating costs for hospital bed facilities in Plzeň; b) preventing the establishment of another competitive medical waste incinerator until the loan is repaid; c) contractually ensuring the supply of medical waste in a constant amount per day (7 tons) (DZP, 2002).

9.1.3 Incinerators versus Composting Facilities

Composting of biowaste has several times lower investment costs per 1 ton of installed capacity compared to incineration plants (Ščasný, 2002). However, they compete with each other. For example, WtE Malešice needs to incinerate a portion of wetter waste because its technology is not designed for high calorific value of plastics (Info.cz, 2023). According to a recent study in Lithuania, for instance, 30 to 40 % of municipal waste consists of biodegradable materials (Āriņa et al., 2023), which are compostable or processable at biogas plants. The authors suggest supporting home composting. Some cities in the Czech Republic, such as Jihlava (Tulis, 2011), Příbor (ČTK, 2018c), Jablonec nad Nisou (ČTK, 2018b), and Prague 10 (ČTK, 2015), have offered free or municipally supported composters.

9.2 Maintenance and Repairs

Waste incinerators, whether municipal or hazardous, require not only regular maintenance during their operation but also the replacement of various worn-out or corroded parts. The costs of such repairs can often reach amounts close to the costs of building a new incinerator, as documented in the cases of the Brno and Prague (Czech Republic) municipal waste incinerators. However, let's first take a broader look at these costs.

WtE in Prague faced initial problems with excessively high chlorine concentrations (chlorinated substances) in the incoming waste, leading to corrosion in the flue gas cleaning part of the technology. The extent of corrosion is well depicted by a detail of the iron lining from one of the decommissioned waste incinerator chimneys in the Czech Republic (see Photo 9.4).

The corroded chimney was paid for by a power plant co-incinerating waste using pyrolysis technology in Hamm, North Rhine-Westphalia. The sixty-meter-high chimney corroded due to overly acidic emissions and collapsed in 2009 (dpa, 2009). Four months later, the RWE company announced the closure of this facility because repairing the chimney wasn't viable due to the risk of repeated corrosion (wa.de, 2010).

The damage caused by the fire in the catalytic pyrolysis technology in Lučenec (Slovakia) in 2016 was estimated to be in the hundreds of thousands of euros (see Photo 9.5). The Lučenec plastic waste catalytic pyrolysis plant experienced repeated fires in May 2016 and September 2017 (Hutková, 2016).

Unlike the WtE Termizo Liberec, the WtE in Prague already had a dioxin filter installed but had to modernize it in 2007, costing them 260 million CZK (Mach, 2007). This amount is similar to what the WtE Termizo Liberec had to invest in equipping itself with a dioxin filter. The tightening of environmental protection legislation is reflected in additional costs for the modernization and maintenance of incinerators.

During its last modernization, WtE in Prague replaced all four boilers and increased its capacity to almost 400,000 tons of waste per year.

Photo 9.4: *Corroded iron lining of a decommissioned waste incinerator chimney. Photo: Arnika's archive.*

Photo 9.5: *The catalytic pyrolysis plant for plastic waste in Lučenec, Slovakia, suffered repeated fires in May 2016 and September 2017. (Source: Hutkova, 2016)*

The total investment in the incinerator's modernization was planned to be around 2.8 billion Czech crowns (iDnes, 2023). Pražské služby a. s. (owner) issued bonds to finance part of this action. According to a report in E15, Prague's leadership approved the city's repurchase of bonds from Pražské služby a. s. up to one billion Czech crowns in August 2018 (Euro.cz editorial office, 2019). As reported by iDnes: "The problem arose during the renovation in 2021 when a fire damaged the third production line. The fire damaged almost half of the incinerator, and the damaged technology needed repair. The damage amounted to hundreds of millions of crowns," (iDnes, 2023); (see also Photo 11.8).

A similar investment is looming in Brno: "The operator of WtE SAKO Brno (Czech Republic) plans billion-dollar investments in the coming years. A new waste incineration boiler, including related investments, will cost 2.3 billion crowns," (Tramba, 2022). According to an article in the Economic Journal, they intend to utilize the EU Modernization Fund for its financing (Tramba, 2022). Like the Prague incinerator, it aims to use this opportunity to increase its incinerator capacity.

Photo 9.6: *WtE SAKO Brno (Czech Republic) makes ample use of European grants. Photo: BíláVrána – Own work, CC BY-SA 4.0, https://commons.wikimedia.org/w/index.php?curid=62476281.*

Non-governmental organizations criticized the allocation of funds from the EU Modernization Fund, primarily to heating companies. "The most money will go to existing 'big players'—energy and heating companies. They will receive 26 % of the fund for the transformation of heating," wrote legal expert Laura Otýpková in 2020 (Otýpková, 2020).

However, it's not the first time the WtE SAKO Brno intends to use European grants. "An important milestone and the beginning of Stage I for evaluating the operation of the WtE SAKO Brno before and after modernization is the year 2002 when documentation was prepared and submitted for a financial grant request from the EU ISPA fund (a financial instrument for funding infrastructure projects in the areas of the environment and transportation) for the Waste Management Brno project. The following year, a financial grant of 1.5 billion crowns was approved from the ISPA fund. The total investment value was 2.2 billion crowns, with the remaining part financed by the Statutory City of Brno, the State Environmental Fund of the Czech Republic, and WtE SAKO Brno" (Bajzová, 2017).

The investments in the modernization of incinerators in Prague and Brno are similar to those found in Switzerland and used as an example for the planned construction of a WtE in Písek, Czech republic: "...the facility in Switzerland, KVA Horgen, referenced in the feasibility study for the waste incinerator in Písek, had to make investments in the incineration facility every 2.5 years from 1950 to 2017 due to tightening environmental standards and emission limits, in the order of millions of CHF per case," (Kajtman, 2023). Nevertheless, both the Brno and Prague incinerators were significantly helped by public budgets.

A case study discusses the complexity of repairs for the largest Amager Bakke incinerator project in Denmark (Chapter 10.2.4).

9.3 Operating Costs and Waste Incineration Fees

According to available data, the cost of incinerating municipal waste in the Czech Republic in 2020 ranged from 850 to 1,500 CZK (approx. 32 to 60 EUR) per ton of waste (Blahut, 2020).

Among the operating costs of waste incinerators are payments for the disposal of residual waste, such as ash, slag, or fly ash (see Chapter 3.3). Incinerators seek and find ways to classify their waste as a product, avoiding payment for its disposal. It is extensively used as a material for technical securing of landfills, bypassing disposal fees for municipalities where the landfills are located, resulting in significant profits. For instance, the ash from the WtE Prague is used at the landfill in Benátky nad Jizerou (Bočan, 2014).

WtE Termizo Liberec mixes fly ash from flue gas cleaning with bottom ash and sells this mixture as a product called SPRUK. Even if they earn just one crown per ton of this product, it's profitable as it saves a significant amount of money they'd otherwise pay for disposing of hazardous waste, which the fly ash constitutes. The primary driving force for incinerators to strive for the treatment and reclassification of their waste into

Photo 9.7: *The AVE CZ landfill in Benátky nad Jizerou stores bottom ash from WtE Malešice (Prague) in its rear section and uses it, sorted by fractions, for surface treatments. Approximately 70,000 tons of such waste ended up in the landfill in 2018. Photo: Marek Jehlička (skyworker.cz).*

products (building material) is the desire to save their own money, not environmental protection or material conservation.

Investment costs for WtE facilities are cheap in China. Plants that incinerate domestic waste are heavily reliant on government subsidies. Since 2020, the industry has struggled to get the government to pay up. According to an August 2022 study, 11 incineration plants across Zhejiang, Jiangsu, Anhui, Shandong and Jiangxi were found to be owed 478 million yuan (US$ 65.61 million) in national and provincial electricity-generation subsidies and waste-disposal fees (Jiacheng, 2023).

9.4 Associated Costs and Fees

Waste incinerators require a centralized waste management system and are demanding in terms of transportation. Usually, these are high-capacity facilities requiring waste transportation from significant distances. This was well described by Petr Borecký, the mayor of Úvaly, in a 2016 interview regarding the ČEZ-planned high-capacity waste incinerator in the Mělník, Horní Počaply power plant area: "A relatively expensive system is being built, where you have to establish transfer stations, and the cost to build one is approximately 35 million CZK. That's a total of 600 million CZK that municipalities will have to spend on this," (iRozhlas, 2016). The then governor of the Central Bohemian Region, Miloš Petera, confirmed the region's need for this facility, adding that "...the revenue points of the municipalities would be under the European Union subsidy, so there would be an 80% subsidy for the 600 million," (iRozhlas, 2016). Thus, EU subsidies can be obtained for logistics costs associated with the WtE construction.

Municipal waste incinerators also receive support as renewable energy sources for a portion of heat generated from incinerated waste, based on the provisions of the Supported Energy Sources Act (EnergetikaInfo.cz editorial office, 2022).

Photo 9.8: *Waste incinerators are sources of contamination for domestic poultry farming, which can have health impacts on the population, resulting in additional costs borne from public budgets. The illustrative photo is from the egg sampling around the Košice municipal waste incinerator in 2005 (Hegyi et al., 2005). Photo: Archive SPZ Košice.*

9.5 Unaccounted Costs Resulting from Waste Incineration

In other chapters of this study, the impacts of incinerators on the environment and human health are discussed. The consequences for these impacts are not accounted for, even though the costs, predominantly borne by taxpayers and the state, are incurred in healthcare and destroyed resources. The investigation of contamination around closed incinerators in Lausanne, Switzerland, or Maincy, France (see Chapter 3.5), for instance, was funded by state or municipal (i.e., public) institutions. The same applies to the contamination case of poultry farms in Newcastle, where in 2000, 44 sites of dioxin and heavy metal pollution were identified due to the use of incinerated municipal waste ash (see also Chapter 5.1.1.3.5).

The introduction of the obligation to purchase emission allowances for WtE operations is an attempt to at least partially consider the impacts of waste incineration on the environment, specifically CO_2 emissions. In the "Fit for 55" package, the EU Council and European Parliament agreed on a compromise requiring European WtE facilities to participate in the emission allowance payment system by 2030 at the latest (Garrett, 2023). A study prepared for ZWE rather thoroughly analyzed what this would entail (Warringa, 2021). The inclusion of incinerators in the emissions trading system (EU ETS) will inevitably increase waste fees (or price of thermal and electrical energy) for households and create greater pressure for sorting, but primarily for reducing the production of mixed municipal waste ending up in incinerators.

A study comparing the most economically viable waste management solution in Bombay concluded that incineration remains the most expensive option in terms of investment and operating costs throughout the incinerator's lifespan (Sharma & Chandel, 2021).

Professor Lars Stoumann Jensen from Copenhagen also mentioned an interesting aspect, pointing out that in Danish incinerators, nearly 10,000 tons of phosphorus from compostable organic waste ends up in landfills instead of fields (see Chapter 9.1.3), which roughly matches the amount Denmark imports annually in the form of phosphates (Borking, 2011), for which they have to pay.

9.6 Summary of the Chapter

In summary, waste incinerators, whether WtE or hazardous waste incinerators, have received numerous financial reliefs funded by public sources, i.e., taxpayer money. Many of these operations couldn't have been established or would have become unprofitable without them. Waste incineration also incurs numerous hidden costs, which the state or its citizens must bear (e.g., health protection, destroyed resources, etc.).

Certainly, paying for better environmental protection is necessary, and it cannot be acquired for free. However, when we look at the impact of waste incineration facilities on the environment, is this indeed the higher-quality environmental protection in waste management?

An otherwise quite favorable analysis of waste incineration summarized their economic cost well: "Installing a waste incinerator is an expensive process, mainly due to expensive infrastructure and equipment required for the incinerator's construction. Besides high initial costs, waste incineration requires the employment of trained and devoted staff to operate it. Regular maintenance of the facility, the intensity of which increases with the facility's aging, adds significant operating costs," (conserve-energy-future.com, 2023).

In centrally controlled China, incinerators cannot achieve economic profitability without state support, as noted in an analysis published in 2019: "(1) Waste-to-Energy incinerators cannot make profits solely from waste disposal fees without state subsidies. In the calculated case, the subsidy per ton should be 100–300 yuan, considering profitability requirements. (2) The current waste disposal fee subsidy for waste-to-energy is 292.5 yuan per ton, and the minimum subsidy is 197 yuan per ton. To achieve the repayment period, it is necessary to slightly increase the waste disposal fee subsidy," (Ye et al., 2019).

10. Overcapacity of Waste Incineration

10.1 Global capacity of waste incineration

The total capacity of MSWI/WtE globally was estimated at over 228.24 million $t.y^{-1}$ in 2013 (Coenrady, 2013). Estimates in 2017 were around 250 to 258.4[62] million $t.y^{-1}$ (Lu et al., 2017; Makarichi et al., 2018). The rapid increase of estimated capacity at almost 390 million $t.y^{-1}$ in 2020 (Coenrady, 2020) can be attributed to development in China, from more than 25 million $t.y^{-1}$ in the 2013 database (Coenrady, 2013) to 162.5 million $t.y^{-1}$ in the 2020 database (Coenrady, 2020). According to the Ministry of Housing and Urban-Rural Development (MoHURD) figures, 210 million tonnes of domestic waste were incinerated in China's cities and county towns in 2021,[63] with 73 % of domestic waste from cities burned. This rapid increase in incineration, both in percentage and absolute terms, has helped reverse the sprawl of landfill sites encircling Chinese cities. However, it has also worsened economic, health and environmental risks (Jiacheng, 2023). By 2023 in China, there were 927 WtE plants operating (Jiacheng, 2023).

More information about waste incineration in China is in subchapter 10.3.

The capacity of WtE plants in the USA and Japan has changed little in recent years. There were 31 facilities and 60.1 million $t.y^{-1}$ of waste burned in the USA in 2013 and 31.6 facilities and 60.3 million $t.y^{-1}$ of waste burned in Japan, according to Coenrady's database (Coenrady, 2013; Coenrady, 2020).

In the UK, the capacity of WtE plants has almost doubled from 8.6 million $t.y^{-1}$ in 2013 to 15.7 million $t.y^{-1}$ in 2020 according Coenrady's database. Such a rapid increase of capacity in the UK can also be confirmed by data from CEWEP which shows an increase from 6.1 million $t.y^{-1}$ in 2013 (CEWEP, 2015) to 13.96 million $t.y^{-1}$ in 2020 (CEWEP, 2023).

In the European Union (EU);(including Norway, Switzerland and UK), 101 million tons of municipal waste was treated in 504 Waste-to-Energy plants in 2020 (CEWEP, 2023). The largest capacity for WtE was found in Germany, France and the UK with 27, 14.26 and 13.96 million tons of municipal solid waste in 100, 117 and 54 WtE plants respectively (CEWEP,

62 Calculation based on information that WtE plants burn daily approximately 700.000 metric tons of waste. (Makarichi et al., 2018)

63 There is discrepancy between data from Coenrady (2020) database which counts 162.5 million t.y-1 and data from Chinese MoHURD which estimates 210 million $t.y^{-1}$ of municipal waste being incinerated in Chinese WtE plants. Coenrady's database probably does not necessarily cover all WtE plants as some projects were finished within two to three years of his counting and with the most recent start-ups being recorded in 2019 in Coenrady's database.

2023). Bulgaria, Greece, Latvia and Romania have no WtE plants as of 2020, according to data from CEWEP (CEWEP, 2023).

Plants that incinerate domestic waste are heavily reliant on government subsidies in China. But since 2020, the industry has struggled to get the government to pay up. According to an August 2022 study, 11 incineration plants across Zhejiang, Jiangsu, Anhui, Shandong and Jiangxi were found to be owed 478 million yuan (US$65.61 million) in national and provincial electricity-generation subsidies and waste-disposal fees (Jiacheng, 2023).

10.2 Case Studies from Europe

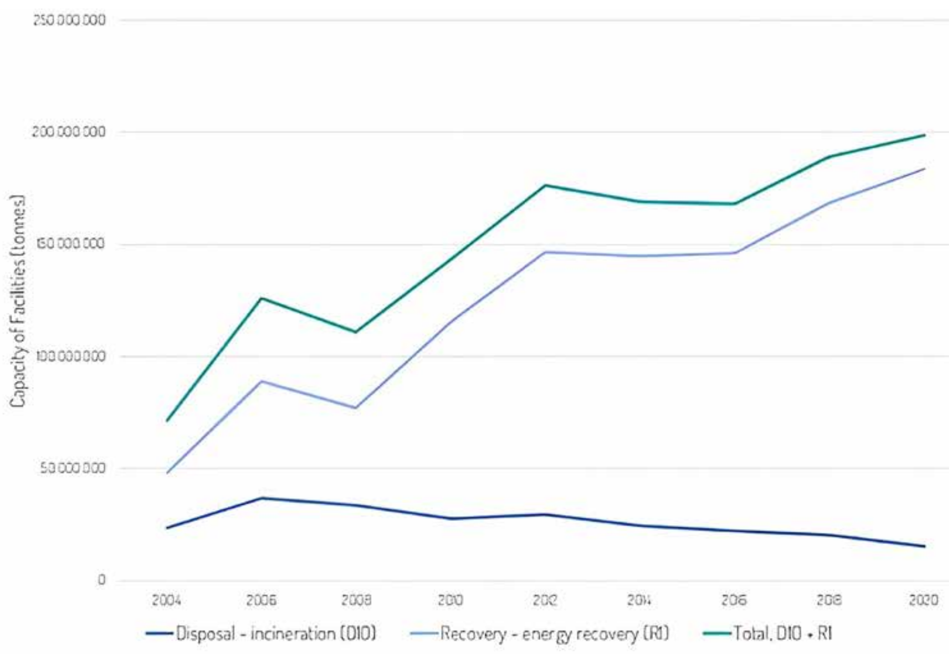
In the following two case studies, we will attempt to show how the exaggerated capacity of the WtE incinerator plants have influenced waste management, and foreign policy. Some aspects have already been mentioned, for example, in the chapter dedicated to residues from waste incineration (Chapter 3.3). The Netherlands, which has exaggerated incineration capacities, faces problems concerning where to dispose of the bottom ash (see Chapter 3.3.3.1).

10.2.1 European Union

A 2023 report by Zero Waste Europe suggests that the EU has an excess of incineration capacity and recommends considering a moratorium on the construction of new incinerators. The report also notes that between 2004 and 2020, there was an annual increase in waste incineration capacity of approximately 8 million tonnes, and that by 2023, total capacity could reach up to 220 million tonnes.

Additionally, the report calls on Member States with excess capacity to implement a moratorium and potentially reduce capacity. Janek Vahk, Zero Pollution Policy Manager at Zero Waste Europe, commented on the situation at the EU level. He stated that waste management and environmental sustainability are becoming increasingly important. The conclusions of the study for Zero Waste Europe are clear: it is time for a moratorium on incineration. The European Union must rethink the role of incineration in the waste hierarchy, especially with overcapacity looming and recycling targets on the horizon. The report even suggests decommissioning 5 % of incinerators annually. Implementing a moratorium would promote sustainable waste management (Hogg, 2023).

Figure 10.1: *Evolution in Capacity of D10, R1 and D10+R1 Facilities, 2004-2020 (tonnes) according to Eurostat. (Source: ZWE, 2023).*



It is also important from the point of view that the EU has set specific targets for waste recycling, and building more incinerators may lead to waste

Figure 10.2 *Recycling rates in Europe by waste stream. (Source: EEA, 2023)*

going there instead of being recycled. The growth rate of municipal waste recycling is slow (see Figure 10.2).

Furthermore, the European Commission has made it clear in a number of policies, targets and regulations that govern the EU Taxonomy and guidance for major European Financial Institutions, that waste to energy incineration is not supported in terms of climate impacts and that it is an industry that represents a threat to the Circular Economy, undermining better waste management including recycling and resource conservation (European Commission, 2020; Makavou, 2021).

10.2.2 Case study: Czech Republic

There are currently four WtE plants in operation in the Czech Republic that use municipal waste (MSW) for energy recovery. These four plants have a total capacity of 962,000 tons of waste per year. Their total capacity is higher than the actual amount of waste incinerated per year due to planned shutdowns or accidents, but the difference is mainly due to the increase in capacity of these plants in recent years.

In 2021, the Czech Republic had 23 incinerators that cannot be classified as WtE and that incinerate industrial or medical waste. By way of comparison, in 1992 there were more than 230 waste incineration plants in the Czech Republic, most of which had a small annual capacity. (ČEÚ, 1992), see Figure 10.3. In the same figure you can see how the number of incinerators and their capacity in the Czech Republic developed from 1994 to 2006 (Brožová et al., 2008). A large number of them failed to meet stricter requirements - emission limits or technological requirements - and had to be closed (ČHMÚ, 2003).

According to the Czech Statistical Office (ČSÚ), in 2020, 5,271,690 tons of municipal waste were generated in the Czech Republic. From the status of municipal waste management in 2020 (Table 10.1) it can be observed that the Czech republic still landfills an excessive amount of municipal waste and, conversely, do not utilize waste to energy incineration (acc. to EU). It is also interesting to note that between 2020 and 2021 there has been a 1.2% decrease in municipal waste generation (Ministry of the Environment of the Czech Republic, 2021c) and a 6.5% decrease in hazardous waste generation (Harák, 2023).

The EU recommends to its member states, including the Czech Republic, that a MSW recycling rate of 55 % by 2025, 60 % by 2030 and 65 % by 2035, while setting a maximum of 10 % MSW to landfill, be applied.

Figure 10.3: *Development of the number of incinerators and their capacity in the Czech Republic in 1994-2006. (Source: Brožová et al., 2008)*

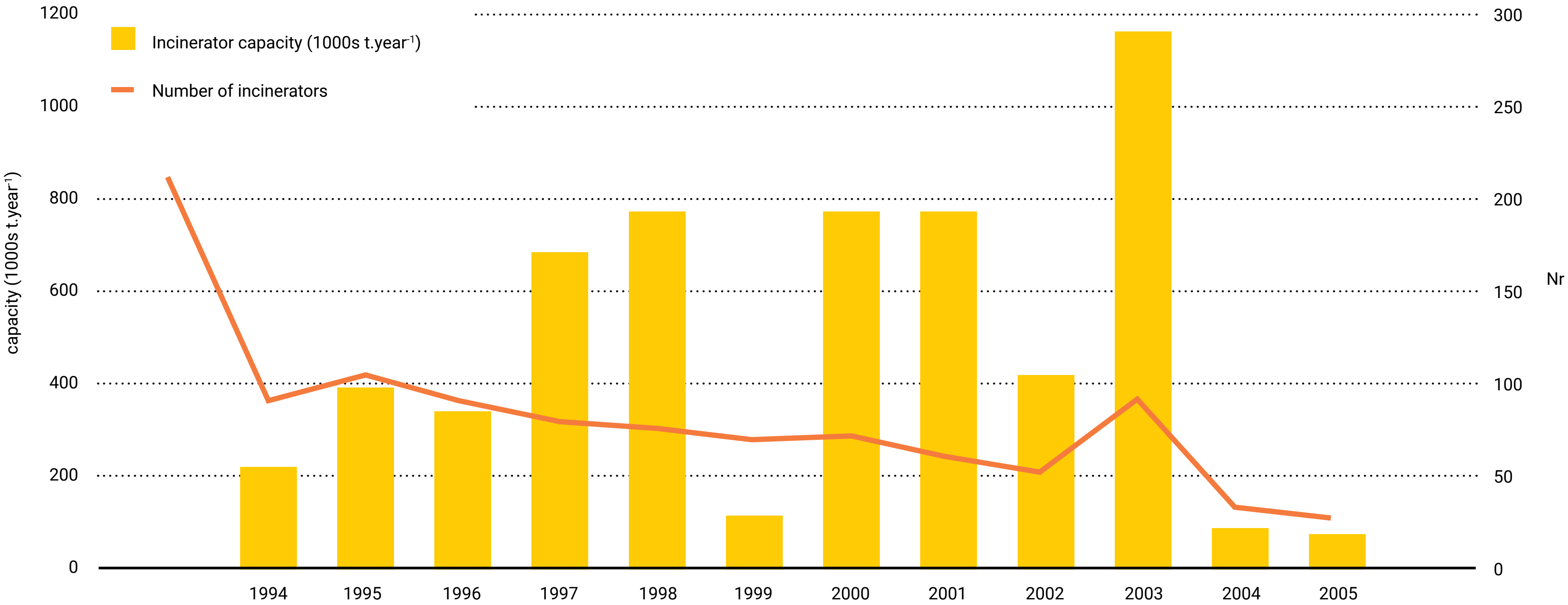


Table 10.1: *Waste management in the Czech Republic compared to the EU for municipal waste in 2021. Source: ČSÚ (2022) and EUROSTAT (2023)*

	Energy recovery [%]	Landfill-ing [%]	Material recovery and com-posting and backfilling [%]	Others [%]
EU (2021)	26.8*	23.4	30.3 + 19.5**	0
CZ (2021)	15.8	46.9	24.6 + 12.3 + 0.2	0.1

**The distinction between incineration and energy utilization is not made.*
***The distinction between landfilling is not made.*

If we consider the generation of 6,000,000 $t.y^{-1}$ of municipal waste by 2035 (even with a slight increase despite the year-on-year reduction in production mentioned above), we will be able to recover approximately 30 % of MSW (2,000,000 t) for energy in 2035.

The current capacity of operating waste-to-energy facilities is 962,000 $t.y^{-1}$. However, during the period 2011 to 2023, several other plants have gone through the Environmental Impact Assessment (EIA) process and received approval from the Ministry of the Environment, and it can be assumed that they will be built if they receive funding. Their capacity is 948,000 $t.y^{-1}$ of MSW. Approximately 496,000 $t.y^{-1}$ of waste with an unknown proportion of MSW, ends up as solid recovered fuel (SRF) in cement plants. Therefore, the total annual capacity of the facilities that will presumably be built, is 1,910,000 $t.y^{-1}$, and with the inclusion of SRF used in cement plants, this will be 2,406,000 tonnes.

Currently, other plants (with a total capacity of about 300,000 $t.y^{-1}$ of MSW) are seeking approval in the EIA process also. If they were to receive permission to build and operate then the total capacity of facilities in the Czech Republic would increase to more than 2,700,000 $t.y^{-1}$ (2,200,000 $t.y^{-1}$ excluding co-incineration in cement plants).

Photo 10.1: *The company Ecowaste near the Prachovice cement plant prepares waste for incineration in the cement plant: it's enough to shred it and it goes into the cement plant. This has been reported by Eko-kom for many years as plastic recycling. Photo: Pardubický region.*

The existing and approved facilities already substantially exceed (in terms of today's waste production) the maximum amount of mixed municipal waste that the Czech Republic will be able to energetically utilize in and after 2035. Their capacity will also exceed the maximum energy recovery scenario in Waste Management Plan for the Czech Republic with an outlook to 2035 (1,869,600 $t.y^{-1}$); (Ministry of the Environment of the Czech Republic, 2022b). Further increases in waste incineration capacities in the Czech Republic will undermine the ability to meet recycling targets after 2035. Incineration and recycling facilities will increasingly compete for the same materials (because the current composition of wastes that

Photo 10.2: *Even in the preparation of plastics for the Prachovice cement plant, there was a fire, specifically on April 26, 2016. Photo: HZS (Fire Rescue Service).*

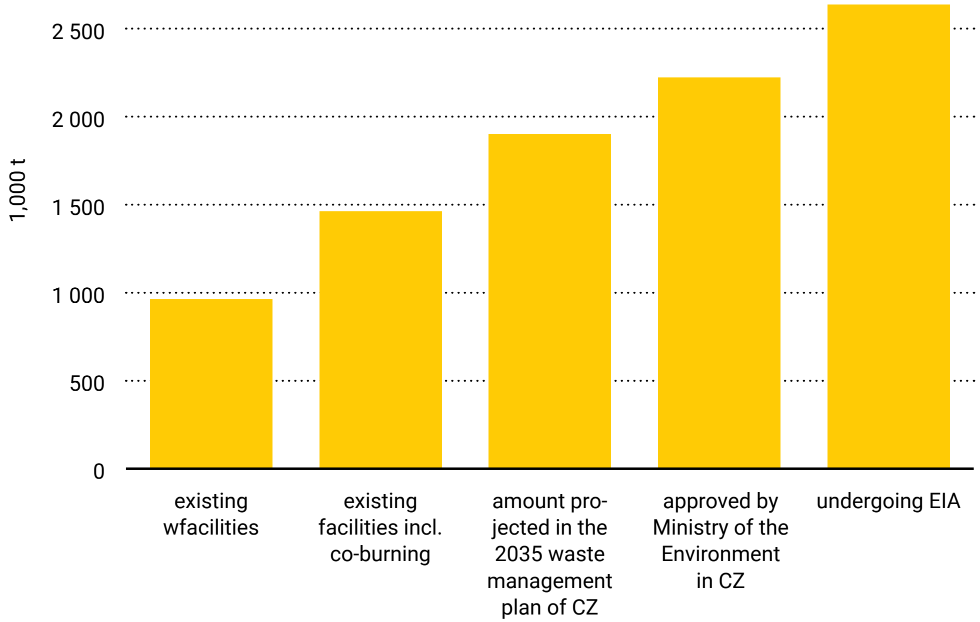
Photo 10.3: *Senator RNDr. Jitka Seitlová (KDU-ČSL) criticized the further increase in waste incineration capacities (WtE) in the Czech Republic before the approval of the new waste act, which significantly favors incinerators. Photo: Martin Holzknecht, Arnika.*

are directed to WtE facilities, contain materials that can be composted, reused or recycled). The excessively high capacity of waste incinerators was mentioned on November 12, 2020, during the discussion of the new Waste Act in the Senate of the Czech Parliament by Senator RNDr. Jitka Seitlová (Göblová, 2021).

According to Mgr. David Surý (Chief Director of the Environmental Protection Section of the Ministry of the Environment in Czech Republic); (Göblová, 2023): *"We see projects that are still being planned, they are neither financed nor planned, they have only undergone EIA and the associations are telling us we have to put the brakes on, don't build. We say we are not going to build, the capacity is clearly there and no investor in a court of law will invest in a project for which they don't have the commodity they want to use for energy and they know that the state will not support the import of waste from abroad."* Incinerators are not subsidized by the state, but they have received funding from the State Environmental Fund of the Czech Republic (Hospodářská komora České republiky, 2023).

From the provided data and information, it unequivocally follows that the capacity of existing and (within the EIA process) approved facilities for energy recovery from waste is more than sufficient and that there is no need to grant any further approvals. On the contrary, it would be desirable to keep

Figure 10.4: *Existing and planned capacities for waste incineration in the Czech Republic*



the capacity for energy recovery from waste lower to create mild pressure on waste recycling and avoid the so-called "lock-in" effect, as described in the next chapter 10.2.3. The data also indicates that the problem does not lie in the low incineration rate or energy recovery from waste, but primarily in their excessive production and inadequate recycling, which in the Czech Republic is currently mainly addressed by excessive waste landfilling.

10.2.3 More on overcapacities in Europe

The surplus capacities for waste incineration in Nordic countries (Denmark, Norway, Estonia) result in their dependence on importing waste from neighboring states – in Denmark's case, it's approximately 1,000,000 tons of waste annually (Schart, 2020), see Figure 10.6. However, Danish legislators decided in 2020, as part of carbon emission reduction efforts, to reduce the capacity of their incinerators by 30 % within ten years, leading to the closure of 7 of 21 waste incineration plants in Denmark (Gardiner, 2021). Norway's Klemetsrud incinerator, on the other hand, imports waste from Manchester or Leeds in the United Kingdom. As waste processing incurs fees, it's economically viable to export waste from Norway to Sweden for incineration, as incineration there is cheaper (Bevanger, 2015).

Countries with substantial incineration capacity tend to recycle less. Data from Denmark in 2005 clearly showed that regions with higher

Photo 10.4: *The municipal waste incinerator on the island of Bornholm will be among those Denmark plans to shut down (Christensen et al., 2021). Photo: BOFA.*

incineration rates have lower recycling levels, while regions with lower incineration rates have higher recycling rates. Denmark's recycling rate lags behind other European regions. According to Eurostat data, Denmark consistently generates one of the highest amounts of waste per capita in the European Union, and over 80 % of what is incinerated in Denmark is recyclable (GAIA, 2013). A similar situation exists in Sweden. More than 70 % of the waste sent to incinerators there is recyclable, as highlighted by an analysis of waste designated for incineration (Politico, 2022).

The lack of waste (or surplus incineration capacity) also means that incinerators burn waste that could otherwise be recycled under different circumstances. If incinerators were indeed utilizing only non-recyclable materials, as claimed on their websites, for example, by ČEZ (ČEZ, 2022), there wouldn't be competition for waste materials, the same state funds, or contractual commitments for waste deliveries from municipalities. This is more apparent in Waste-to-Energy Facilities and less direct in incinerators that solely dispose of waste (GAIA, 2013). Incinerators require materials with high calorific value (paper, cardboard, plastic), often the same materials that are easily recyclable. Moreover, incineration only utilizes approximately 20 % of the energy stored in these waste materials, while recycling saves 3 to 5 times more energy compared to primary resource utilization or virgin production (GAIA, 2013). Specifically, in the case of office paper production, it's 2.5 times more energy – see Table 10.3.

In Germany and the United Kingdom, the paper industry and incinerators have been in competition for several years for paper material (lestercycle.com, 2007). Up to 70 % of fibers used in the paper industry come from household and business recycling. The paper industry is aware of this and advocates prioritizing the recycling of high-quality paper over incineration (CPI, 2012). In the Netherlands, recycling companies addressed an open letter to several ministers in 2009 out of concerns about competition from waste incineration (GAIA, 2013).

Table 10.2: *Energy consumption in office paper production from primary resources and recycled paper. (Source: Havel, 2022).*

	Production from primary raw materials		Production from recycled paper	
	Total Energy ($MJ.t^{-1}$)	Fossil Energy ($MJ.t^{-1}$)	Total Energy ($MJ.t^{-1}$)	Fossil Energy ($MJ.t^{-1}$)
Wood / Old Paper	803.6	730.5	807.5	774.1
Pulp/DIP*	28365.8	5507.9	5352.3	4221.1
Transport of Pulp	463.5	419.3		
Paper Production	8975.8	7956.7	8975.8	7956.7
Total	38608.7	14614.3	15135.5	12952.0

*DIP - Ink Removal from Old Paper

Although incinerators rank lower in the waste management hierarchy, they are often favored at the local level over recycling, perpetually impacting waste prevention efforts and attempts to increase the recycling percentage negatively (GAIA, 2013). For local authorities, this means redirecting waste management funds into incineration facility construction, leaving little for prevention, recycling, or composting support (GAIA, 2013; ZWE, 2019). Also excess incineration capacity leads to reduced waste disposal fees, prompting municipalities to choose incineration over recycling.

The Japanese city of Minamata is also struggling to move away from incineration. Although it recycles twice as much as the Japanese average, 51.6 % of waste that is compostable or recyclable still ends up in incinerators (GAIA, 2013).

In Madeira and the Azores, where several waste incinerators operate, recycling facilities have been dismantled or were never built because they would disrupt the waste flow crucial for sustaining the economic operation of incinerators. Constructing and maintaining waste incinerators consume a considerable portion of available waste management funds, hindering investments in waste management alternatives (ZWE, 2019).

However, coal incineration also generates energy with significant environmental impacts (CO_2 emissions, fine particulate matter, sulfur, and nitrogen oxides), affecting people's health, particularly those living near coal-fired power plants (European Union, 2024). The European Union, of which the Czech Republic is a part, has committed through the Green Deal to achieve carbon neutrality by 2050, including halting coal mining and incineration. In the Czech Republic, this target is set for 2033. Besides, waste incineration is one of the most expensive and least efficient forms of energy production. Burning one ton of waste produces approximately 550 kWh of electricity (US EPA & OLEM, 2016). The same amount of energy is generated by burning 280 kg of coal (or burning 1 ton of coal yields about 1971

Table 10.3: *Energy Saved by Recycling vs. Energy Obtained by Incineration for Different Materials. (Source: Jofra, 2013).*

Material	Energy saved by recycling ($MJ.kg^{-1}$)	Energy recovered by incineration (without energy recovery)	Energy recovered by incineration (with energy recovery)
Glass	2.85	*	*
Office paper	10.54	2.55	7.17
Newsprint paper	17.81	2.98	8.38
Steel cans	21.61	*	*
PET	34.36	3.98	11.17
Copper wire	87.59	*	*
Aluminum cans	161.58	*	*

**For materials and uses marked with an asterisk, the energy balance is negative because additional energy input is needed to raise the material's temperature.*

kWh of energy) (US EIA, 2022). Moreover, substantial energy savings can be achieved through material recycling – see Table 10.3. Despite this, ČEZ claims on their website that the energy utilization of waste saves non-renewable energy sources like coal or oil (ČEZ, 2022), even though waste is not a renewable resource. This topic is further developed in Chapter 4.1.

10.2.4 Copenhagen, Denmark

In Denmark's capital, Copenhagen, a new waste incineration plant was built and launched into operation in 2017 by five municipalities (Dragør, Frederiksberg, Hvidovre, Copenhagen, and Taarnby) that owned the 40-year-old Amager waste incineration plant. Compared to the old incinerator, it was expected to generate 20 times more heat and electricity per ton of waste incinerated. Its case was well described by Madsen (2019), and the following analysis significantly draws from it, alongside articles from the Danish press. In 2012, a loan guarantee of EUR 534 million for the incineration plant project was rejected by the Copenhagen city council because building such a project might signal to residents that incinerating otherwise recyclable materials is acceptable. Instead, the council wanted a plant with smaller capacity, focused more on recycling and reuse.

The City of Copenhagen requested a new tender for a smaller furnace. The board of the Amager Bakke company rejected the proposal, citing it as economically unviable. Despite this, during the summer of 2012, after a series of secret negotiations, the City of Copenhagen approved the plans with minor changes. This happened despite the fact that the new project would direct the city's waste treatment for incineration for 30–40 years, undermining its climate plan (Bredsdorff & Wittrup, 2012). Moreover, it was agreed that the facility couldn't import waste.

The total capacity of the new incineration plant, which started operating in 2017, is 560,000 tons of waste annually (120,000 tons more per year than the old incinerator), on two lines, each with a capacity of 30–35 tons of waste per hour (Madsen, 2019). However, during the summer months, the incineration plant cannot operate at full capacity because excessive production would mean other power plants could not distribute their heat and electricity (and would be forced to shut down). Therefore, only one line operates at Amager Bakke during the summer.

In an attempt to be a flagship for sustainable development, the Amager Bakke incineration plant changed its name to the Amager Resource Center (ARC) and committed an additional EUR 8 million to research alternative technologies. For this purpose, a new waste sorting plant was built alongside the incineration plant, providing space for household waste storage and recycling (Madsen, 2019). Over time, it became apparent that operating at full capacity required the use of imported waste, which was originally prohibited. However, in 2016, the five municipalities that own Amager Bakke changed the original agreement to allow waste imports (Wittrup, 2016a). The initial estimated amount of waste was too low because a decreasing volume of waste would have led to the plant's bankruptcy after a few years. The municipalities were forced to adjust the assumed 480,000 to 350,000 tons of waste annually, even though the maximum capacity of the incineration plant is 560,000 tons. Now, the facility not only allows the incineration of imported waste but also permits the incineration of biomass, again in contradiction to the original agreement (Madsen, 2019).

In 2018, Amager Bakke incinerated over 451,000 tons of waste, approximately 30,000 tons of which were imported from Great Britain and Ireland (Gurzu, 2019; Wittrup, 2016b).

In 2019, the incineration plant planned to import 50,000 to 70,000 tons of waste from Great Britain, Germany, Italy, Ireland, and the Netherlands, aiming for further increases up to 90,000 tons of imported waste annually (Gurzu, 2019). Plant proponents continue to attempt to justify the environmental

Photo 10.5: *To fill the capacity of the largest Danish municipal waste incinerator Amager Bakke, known for having a ski slope on its roof, up to 90,000 tons of waste must be imported annually. Photo: https://www.youtube.com/watch?v=BIa8bGMAUvI.*

benefits of pursuing waste imports yet the full life cycle analysis of these waste resources which include paper, cardboard and plastic (15-40%) demonstrates otherwise - recycling these same waste imports outcompetes incineration. After ten years of project development, several bailouts, and interventions by the finance minister, the incineration plant still faces financial and technical problems. For instance, in 2016, Babcock & Wilcox Vølund, suppliers to the incineration plant, discovered an error in the furnaces. The subsequent resolution caused both the company and ARC a loss of millions of euros (Martini & Sandøe, 2017). In 2017, the facility was shut down for fourteen days when a design flaw in the heat exchanger meant it couldn't handle temperature changes. In May 2022, one furnace line at Amager Bakke was out of operation for slightly over two weeks after a fire broke out in the hydraulic waste pushers leading waste into the

Photo 10.6: *Professor Brian Vad Mathiesen from Aalborg University stated in 2012 that focusing on heat pumps, geothermal, and solar heat would be significantly more advantageous than waste incineration (Bredsdorff & Wittrup, 2012). Photo: https://thinkeuropa.dk/en/advisory-board/brian-vad-mathiesen.*

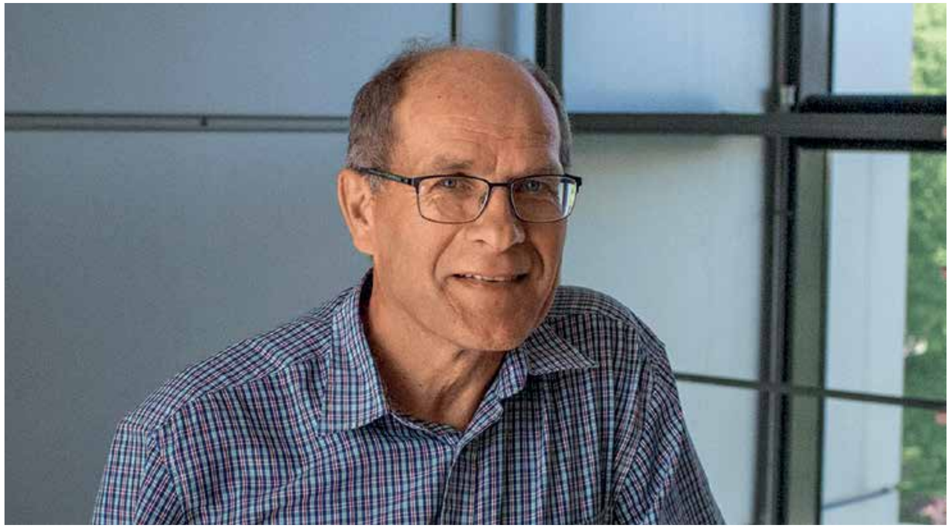


Photo 10.7: *Lars Stoumann Jensen from the University of Copenhagen claims that "nearly 10,000 tons of phosphorus are lost" in Danish incineration plants, corresponding to the annual import of phosphates to Denmark. Photo: University of Copenhagen (https://plen.ku.dk).*

furnace. The company estimated that extinguishing the fire would cost 8 to 10 million Danish crowns (Freiesleben, 2022). Independent experts such as Professor Brian Vad Mathiesen from Aalborg University warned in 2012 that there are significantly better ways to generate heat and energy than incinerating resources. He noted that focusing on heat pumps, geothermal, and solar heat would be substantially more advantageous (Bredsdorff & Wittrup, 2012). As the Amager Bakke incineration plant is financed through a 30-year loan, Danish taxpayers will bear the continuing costs of this facility while the technology providers, Babcock & Wilcox, benefit from it.

"Nearly 10,000 tons of phosphorus are lost in Danish incineration plants every year," estimated Professor Lars Stoumann Jensen from the Faculty of Science, University of Copenhagen (KU-LIFE) in 2011 (Borking, 2011). This roughly corresponds to the amount Denmark imports annually as phosphate. He pointed out that large amounts of compostable waste disappear in incineration plants.

Other parts of Denmark are heading in a different direction than that represented by Amager Bakke. On the island of Bornholm, they aim to achieve zero waste production by 2032 (BOFA, 2019; Gurzu, 2019).

10.2.5 Tallinn, Estonia

According to data from Eurostat for the year 2015 (see Figure 10.5), Estonia utilized 58 % of municipal waste for energy and only 9 % was landfilled (EUROSTAT, 2015). From older data, it's evident that this was achieved

Figure 10.5: *European statistics showing how individual countries managed municipal waste in 2015. (Source: EUROSTAT, 2015).*

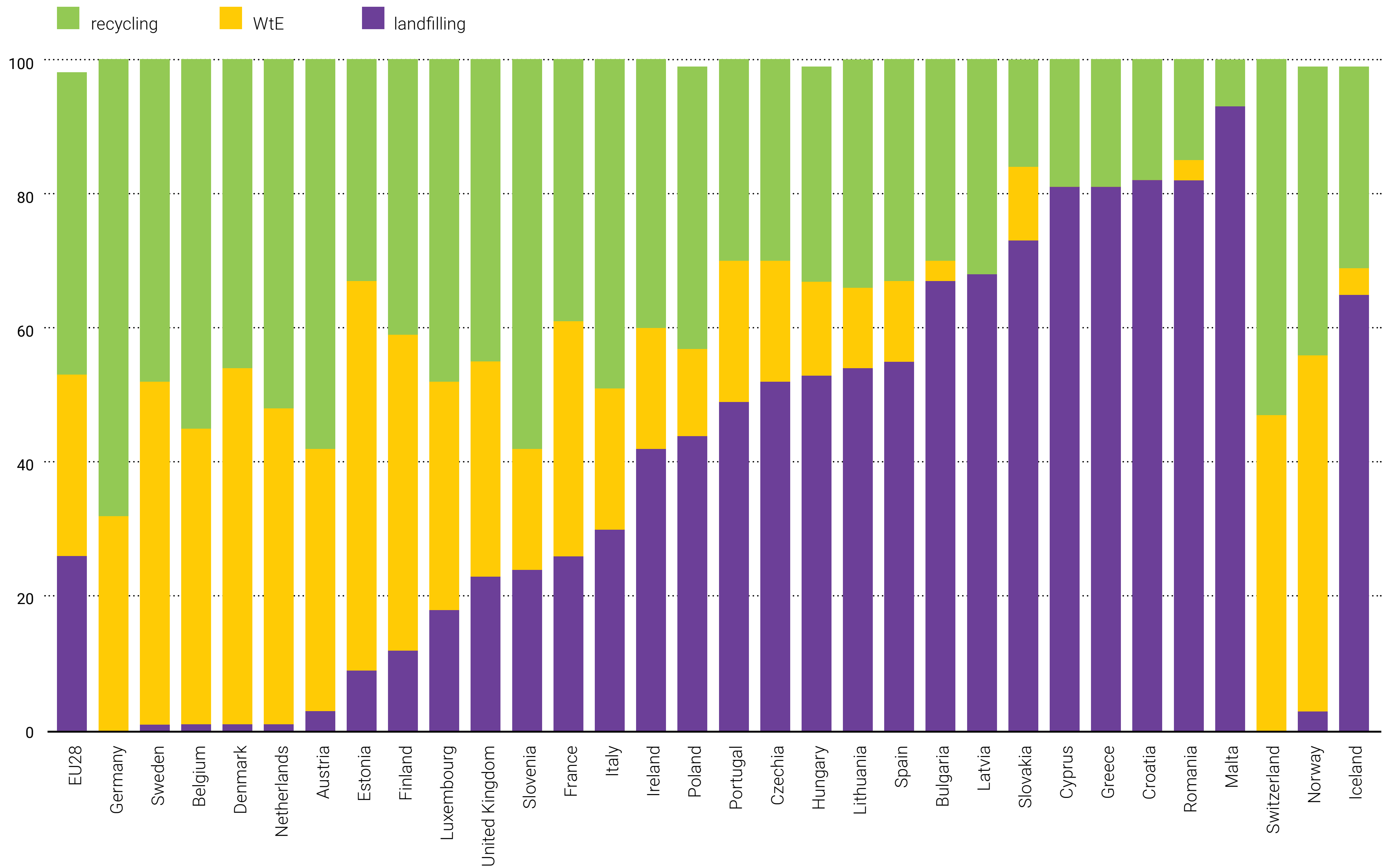


Photo 10.8: *The Tallinn WtE in Iru is close to the sea, where waste is imported. Photo: Bjoertvedt – Own work, CC BY-SA 3.0, https://commons.wikimedia.org/w/index.php?curid=9110788.*

Photo 10.9: *Ash and slag from the Iru incineration plant are also used in road construction, as in the Netherlands. Photo: Jäätmed Artiklite arhiiv.*

through the construction of the waste-to-energy plant (WtE) in Tallinn, where most of the previously landfilled waste is directed. According to a report from 2010, three-quarters of municipal waste in Estonia was previously landfilled (EUROSTAT, 2010; Watkins et al., 2012).

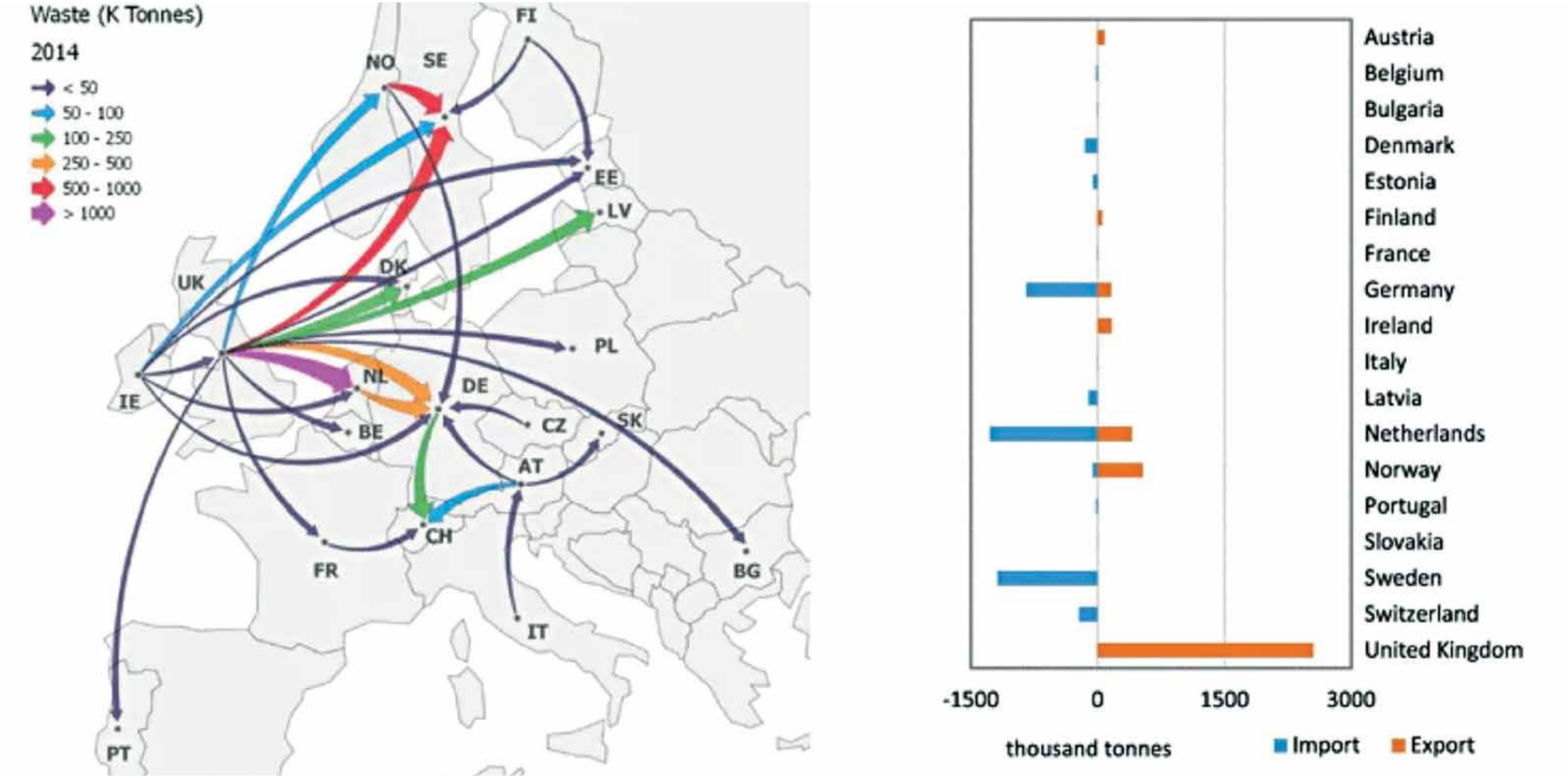
ČEZ (Czech Republic) or heating companies in general seem eager to follow Estonia›s path. Things went smoothly there; hardly anyone protested against the local large waste incineration plant built in the area of the power plant in Iru near Tallinn. It was launched in 2013 and has a capacity of 220,000 tons of waste.y^{-1} (Petrlík, 2018).

Looking at how Estonia deals with municipal waste raises the question of how it will meet the EU target for municipal waste recycling. The political framework for a circular economy in the EU set recycling at 65 % of municipal waste. Today, Estonia recycles only 33 % of it. Even if it adds nine percent from landfills, it will still fall short by 23 %. Will the waste-to-energy plant pick up the slack? If more than a third of the waste diminishes at the Iru plant, it will. People in Tallinn are reliant on energy supplies from this facility.

For instance, an article from September 2013 states that the Iru plant is already unable to manage with Estonian waste alone. Hence, it needs to add 10 % from imports from Ireland or Finland to reach ninety percent of its required waste inputs (see Figure 10.6). These are transported by sea on ships (Kallas, 2013).

“Unfortunately, the waste market in Estonia is still evolving, and it’s not possible to get waste from all regions of the country into our plant. Since we cannot afford a production outage, we have to supplement our capacity

Figure 10.6: *This graph shows that in Estonia, there's a predominance of importing municipal waste over exporting it. However, in other countries with even larger incineration capacities (Netherlands, Germany, Switzerland, or Sweden), this disproportion is much more pronounced. (Source: Scarlat et al., 2018).*



occasionally with waste from Finland and Ireland," said Eliis Vennik, a spokesperson for the operator of the Tallinn WtE, to the online magazine Ärilehele (Economic List);(Kallas, 2013). The Iru incineration plant in Tallinn is operated by Eesti Energia, the Estonian counterpart of the ČEZ Group.

Additional available information about the Iru incineration plant is also interesting. Like other incineration plants in Europe, even the one in Tallinn doesn't want to pay for bottom and fly ash disposal in landfills. Hence, they try using bottom ash as a base in road construction and apply a process called carbonization to fly ash. Allegedly, this will enable using the fly ash in products (likely cement or as an additive in other construction materials) or storing it in a regular landfill (Ruutelmann, 2017). It's not clear how and whether dioxins are fixed in the fly ash.

Photo 10.10: *Reppie WtE Plant. (Source: Fidelis, 2019)*

10.2.6 Ethiopian Reppie Waste to Energy Plant, a Flagship of Next Development in Africa?

In Ethiopia, the absence of an effective municipal solid waste management system leads to widespread littering, open dumping, and burning, resulting in persistent odor, pollution, flooding, and disease outbreaks. Addis Ababa's primary dump site, Koshe, also known as Repi or Reppie, tragically experienced a landslide in 2017, claiming over 114 (GAIA, 2018).

Following this disaster, the Ethiopian government fast-tracked the Reppie Waste-to-Energy (WtE) Project, led by a consortium including Cambridge Industries Ltd (CIL), China National Electric Engineering Co (CNEEC), and Ramboll of Denmark. This $118 million initiative aims to convert 350,000 tons of solid waste annually into 50MW of electricity, fulfilling 30 percent of household energy requirements. Planned to commence operations in 2018, the project's scope expands to include the establishment of WtE plants in Uganda, Kenya, Cameroon, Senegal, and Djibouti (AFDB, 2021; GAIA, 2018; SCS & Rebecca, 2022).

Despite media and some international institutions, including the UN Environment Program, promoting waste incineration in Africa, there's a lack of acknowledgment of its adverse effects on human health and the environment (GAIA, 2018).

The Reppie WtE Plant, the first in its kind in Africa, was built on the Koshe landfill site in the capital, Addis Ababa. It was launched in 2013 as a municipal solid waste treatment plant. The plant was inaugurated in September 2018 in the presence of high-level government officials and representatives of international partners and consequently started operation (Scott, 2018). However, due to the low quality of the waste entering the facility and a shortage of skilled manpower, the plant has not managed to continue its operation as expected. In April 2019, the operator announced plans for a restart (Fidelis, 2019), but it wasn't until four years after its opening that an article on the relaunch emerged (fanabc.com, 2022).

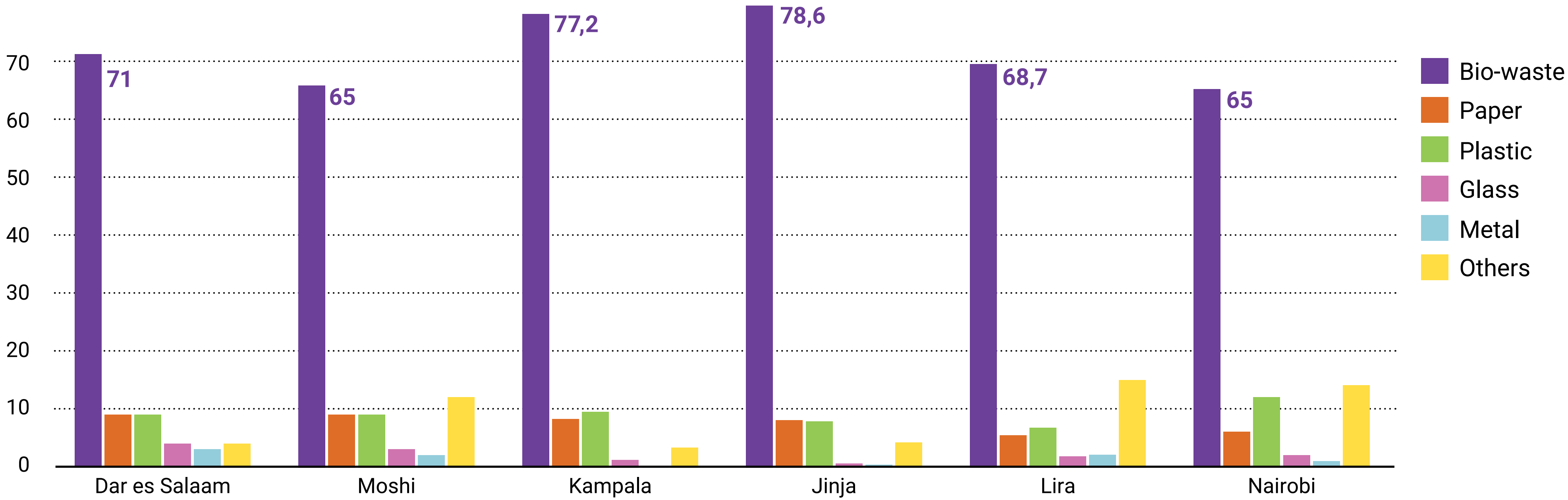
As of 2022, the steam turbine and generator of the plant are undergoing repairs under a two-month maintenance scheme scheduled to conclude at the end of December 2022, according to Project Manager Biruk Eba. The plant is currently incinerating 600 tons of waste daily, albeit with disruptions in power generation. If the project resumes, it can only generate 25 MW of power, casting doubt on the initial 50MW power estimation in terms of time, money, and resource allocation (fanabc.com, 2022).

Alemu (2019) belongs to rare experts who are critical of the Reppie WtE project, and he wrote that *"Despite the accolades, it was a misguided investment from the outset, beginning with the initial decision-making process."* Alemu (2019) continues, *"considering the composition of domestic waste in most African urban cities—comprising 60 % to 75 % dirt and biodegradable waste—incineration for energy generation is fundamentally misplaced. Additionally, mining combustible waste from the existing dumping site at Repi would significantly increase the embedded energy required for every megawatt generated."*

For the same investment amount, the Addis Ababa city administration could have implemented an efficient Integrated Solid Waste Management System (ISWMS), creating thousands of jobs, Alemu (2019) suggests.

Despite the failures of the Reppie WtE plant, other African capital cities such as Kampala and Nairobi are considering investment in WtE plants. Swedish experts from Hifab and IVL have been contracted by NLS Waste Services Ltd Kampala, Uganda, to support and project manage the construction of the first large waste-to-energy plant in Uganda (SCS East, 2022). Kenya has also announced its intention to build a WtE plant in

Figure 10.7: *Municipal waste composition in African cities. Sources: (Adebayo Bello & bin Ismail, 2016; GAIA, 2018).*



2019 , envisaged to cost US $197 million (Najimesi, 2019). The project has commenced the process of securing an energy generation license from the Kenya Energy Regulatory Commission (KERC) in 2021 (AFDB, 2021). Building of WtE plants in Kenya is supported and funded by the African Development Bank Group.

Since the major composition of wastes generated in most African cities is biodegradable, organic materials as shown in graph at Figure 10.7 (Adebayo Bello & bin Ismail, 2016), with low calorific value and high moisture, cities should prioritise investment in composting systems which come with a much smaller budget instead of investing in wasteful WtE.

Further, the UNEP's latest recommendations for low to middle income countries where organic waste volumes are high, include that national authorities should: *"take care to ensure strategies and technologies are fit-for-purpose and tailored to the needs of the country's economy, geography and culture; avoid technologies that lock in linear resource use*" (UNEP & ISWA, 2024).It is clear that waste to energy incineration is an expensive and financially risky, linear technology unsuitable for Africa and many other low to middle income countries who are trying to implement more sustainable and cost effective Zero Waste and Circular Economy models in line with such national and international policy recommendations.

10.3 Challenges in China's Waste-to-Energy Sector

In recent years, China has experienced an enormous rise in waste incineration and the development of Waste-to-Energy (WtE) capacity as part of its broader strategy to address mounting municipal solid waste issues (Shapiro-Bengtsen, 2020). According to the data of the China Statistical Yearbook (National Bureau of Statistics of China, 2017; National Bureau of Statistics of China, 2021), in the five years from 2016 to 2020, the amount

Photo 10.11: *A municipal waste incinerator in Nanjing, Jiangsu province. (Source: Jiacheng, 2023). Photo: Alamy.*

of municipal solid waste incineration (MSWI) in China increased by about 98 %. A development pattern has gradually formed in which incineration mainly treats new waste (Pei et al., 2023).

10.3.1 Waste Sorting Initiatives and Unintended Consequences

The introduction of mandatory waste sorting, notably in major cities like Shanghai (Yixiu, 2019), was intended to improve resource recovery and reduce reliance on incineration. However, these efforts resulted in unexpected challenges. Widespread waste sorting policies led to reduced incineration fuel availability. In May 2023, the Shanghai Laogang Waste Disposal Company recorded 88 days of stoppage across its 12 incinerators, meaning 24 % of capacity was wasted that month. The system had reached overcapacity. This shortage is attributed to the success of waste sorting policies, causing a mismatch between the reduced waste available for incineration and the expanded incineration capacity (Jiacheng, 2023).

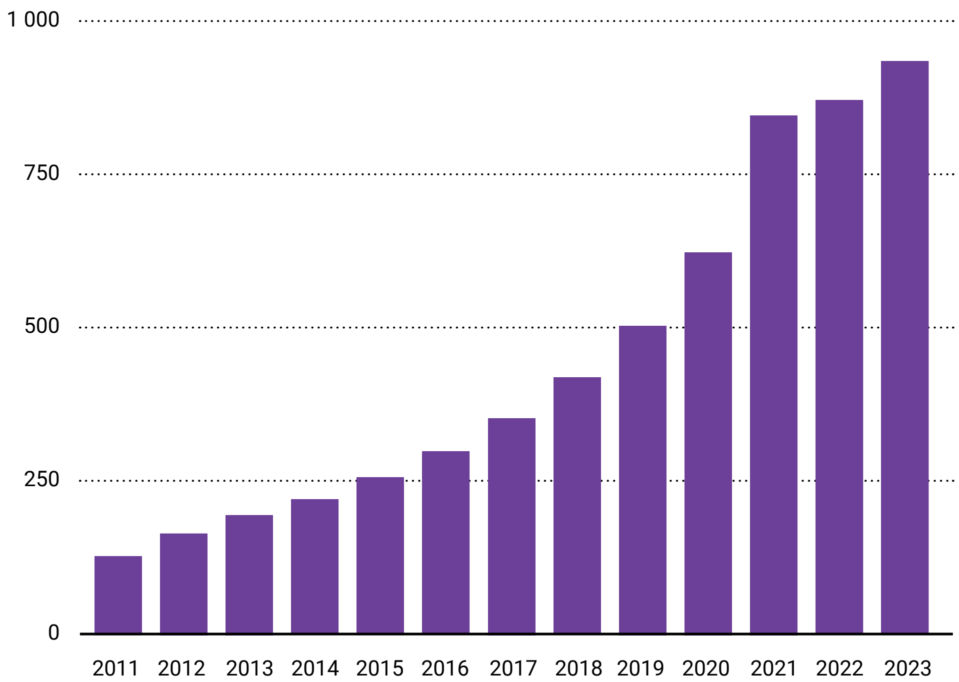
As China expands waste sorting policies to more regions by the end of 2025, balancing waste reduction, sorting, and incineration becomes crucial, especially in addressing waste incineration overcapacity challenges. The composite challenge lies in adhering to the more important principles of reduction, recycling, and safety outlined in China's Solid Wastes Law (Jiacheng, 2023). It also highlights just how significantly waste incineration undermines the recycling sector and better waste management outcomes.

10.3.2 Overcapacity Issues

China's focus on achieving incineration targets set by the Ministry of Housing and Urban-Rural Development (MoHURD) fueled a rush in incineration plant construction. The number of incinerators surged from 130 in 2011 to 927 by 2022, surpassing the 1 million-ton daily incineration capacity target three years ahead of schedule. Overcapacity issues were compounded by factors such as overestimated waste collection capabilities and conflicting policies, leading to a misalignment between waste sorting success and incineration demands (Jiacheng, 2023).

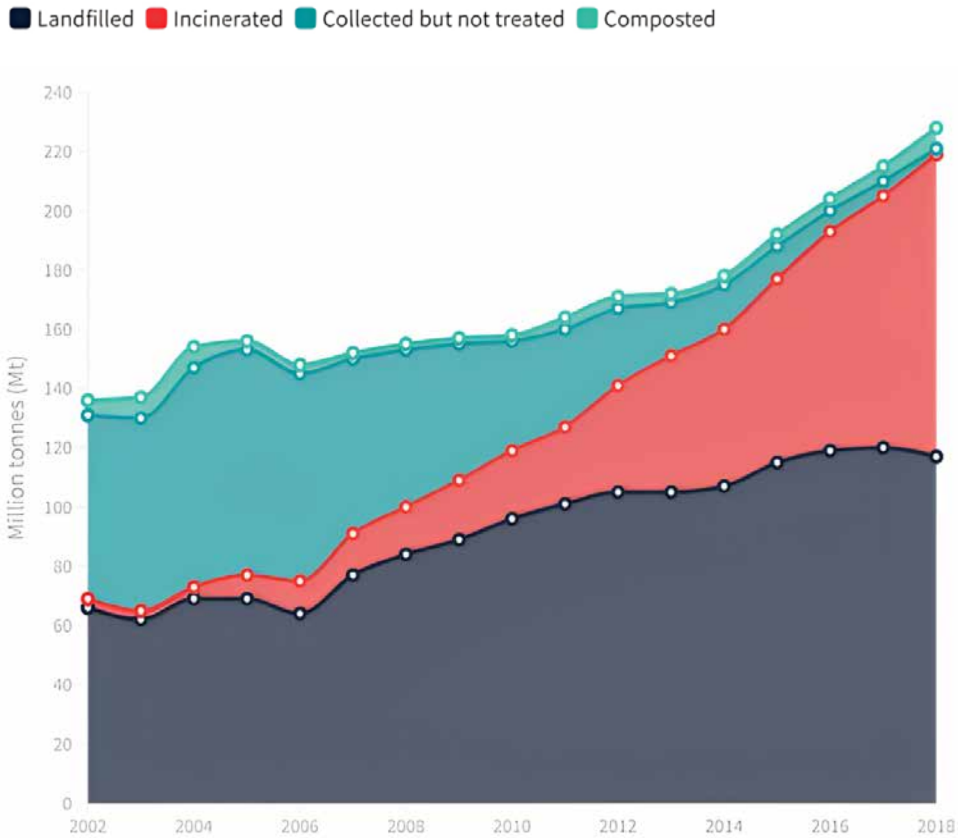
Recent studies highlight the risk of overinvestment in municipal solid waste incineration (MSWI) and landfill capacity due to discrepancies between planned and projected MSW quantities. Even without sorting food waste, overcapacity is anticipated in Anhui and Tianjin by 2030, potentially discouraging more effective sorting and recycling efforts (Shapiro-Bengtsen et al., 2020).

Figure 10.8: *Graph shows development of number of WtE instalations in China. (Source: Jiacheng, 2023)*



By comparison, in Shanghai, great quantities of recyclables and organic waste have been separated in the city in a short time (Jiacheng, 2023; Shapiro-Bengtsen, 2020; Yixiu, 2019). Key to the success of the scheme has been extensive enforcement. This level of enforcement may not be possible across China. However, based on assumptions that are conservative compared to the Shanghai case, Shapiro-Bengtsen et al. (2020) projects 39 % of food waste to be sorted by 2030 and 57 % by 2050, amounting to 61 million tons in 2030 and 109 million tons in 2050. This

Figure 10.9: *Graph shows fast growing amount of municipal waste which is incinerated in China. (Source: Shapiro-Bengtsen, 2020)*



waste should be managed properly, for example in anaerobic digesters, to produce biogas and fertiliser. The gas produced can be combusted or upgraded to natural gas quality for use in the energy system. China plans to have waste-sorting systems in place in 46 cities by the end of 2020, and in most major cities by 2025. What happens in Shanghai will guide that process (Yixiu, 2019).

10.3.3 Health, Environmental and Economic Concerns

While incineration has aided in reducing landfill sites, it has raised concerns about economic, health, and environmental risks. Incineration plants, reliant on government subsidies, face challenges in receiving timely payments (Jiacheng, 2023). Studies indicate health risks associated with waste-to-energy facilities, emphasizing the need for safe buffer distances of at least 1,500 meters, which is five times the current required distance of 300 m (Boré et al., 2022) and showcasing substantial disagreements between government support and public concerns (Yuan et al., 2019).

Photo 10.12: *Waste sorting station in Shanghai in 2019. Photo: Wu Yixiu via China Dialogue under Creative Commons License; (Source: Yixiu, 2019).*

Photos 10.13 and 10.14: *Waste to energy plant in Wuhan (10.13) and its close vicinity (10.14). Nearest housing was destroyed in order to build a "green belt" around the waste incinerator. Photo: Jindřich Petrlík, Arnika, June, 2016.*

Plants that incinerate domestic waste are heavily reliant on government subsidies in China. But since 2020, the industry has struggled to get the government to pay up. According to an August 2022 study, 11 incineration plants across Zhejiang, Jiangsu, Anhui, Shandong and Jiangxi were found to be owed 478 million yuan (US$65.61 million) in national and provincial electricity-generation subsidies and waste-disposal fees (Jiacheng, 2023).

10.3.4 Public protests

In 2013, the city of Guangzhou was a focal point for protests against the construction of a new waste incinerator, with thousands of residents taking to the streets to voice their opposition (RFA, 2013). The uproar centered around Shiling township in Huadu district, where demonstrators marched to the township government offices, brandishing banners and chanting slogans in defiance of the proposed incinerator plant, which would be located too close to their homes. Eyewitnesses estimated that at least 10,000 people joined the protest, highlighting the depth of community concern over the environmental and health implications of the project.

This strong resistance echoed a broader trend in China's environmental activism, with instances like the Jiangmen protests prompting officials to reconsider controversial industrial ventures (RFA, 2013). Despite assurances from local authorities that environmental experts would address community concerns, residents remained skeptical, fearing the potential

Photo 10.15: *Residents of Shiling march in protest of a planned waste incinerator plant, July 15, 2013. (Source: RFA, 2013)*

ramifications of having waste facilities in such proximity to densely populated areas. This sentiment resonated with Liulitun's anti-incineration movement, which emerged earlier in Yongfeng, Beijing, in 2006 (Da, 2017), illustrating a longstanding grassroots resistance to environmentally questionable projects across the nation.

10.3.5 Waste to Energy Plants and Dioxins and Mercury in China

Large municipal waste incineration capacities also result in significant dioxin emissions. A study by (Guo et al., 2023) concluded that most Chinese incinerators met concentration and temperature standards, controlling total emissions to acceptable levels as off 2018. The study revealed substantial benefits from curbing MSW-related dioxin pollution, with waste sorting programs contributing significantly.

A more detailed picture of their results is given in Figure 10.10, copied from their study. It shows that the overburden of dioxin emissions is in the same regions which were found to have overcapacity of municipal waste incinerators (Shapiro-Bengtsen et al., 2020).

An earlier study assessed health risks in relation to PCDD/Fs in areas surrounding two MSWI in China and concluded that the atmospheric pollution by PCDD/F surrounding one MSWI was relatively serious; the environmental impact of the other MSWI was not significant (Jin et al., 2012).

Higher PCDD/F levels were found in household dust in the town of Taopu compared to those in the town of Changzheng. Principal component analysis (PCA) of PCDD/Fs suggested that waste incineration was the primary source of PCDD/Fs in indoor air, whereas PCDD/Fs in indoor dust came from multiple sources. The results of the health risk assessment showed the carcinogenic risk due to indoor PCDD/F exposure was higher for adults than for nursery children and primary school children (Yu et al., 2023).

In another study, the association between PCDD/Fs in paired hair and serum samples from workers was examined in a municipal solid waste incinerator (MSWI) plant in South China. Fly ash and flue gas from the MSWI plant were also analyzed to determine the source apportionment of PCDD/Fs in the hair. The median level of PCDD/Fs in serum and hair were 28.0 pg TEQ.g^{-1} (lipid weight) and 0.30 pg TEQ.g^{-1} (dry weight), respectively. Flue gas was identified as the primary source of PCDD/Fs in human hair. Blood and flue gas were accountable for, on average, 37 % and 61 % of the PCDD/Fs in hair, respectively.

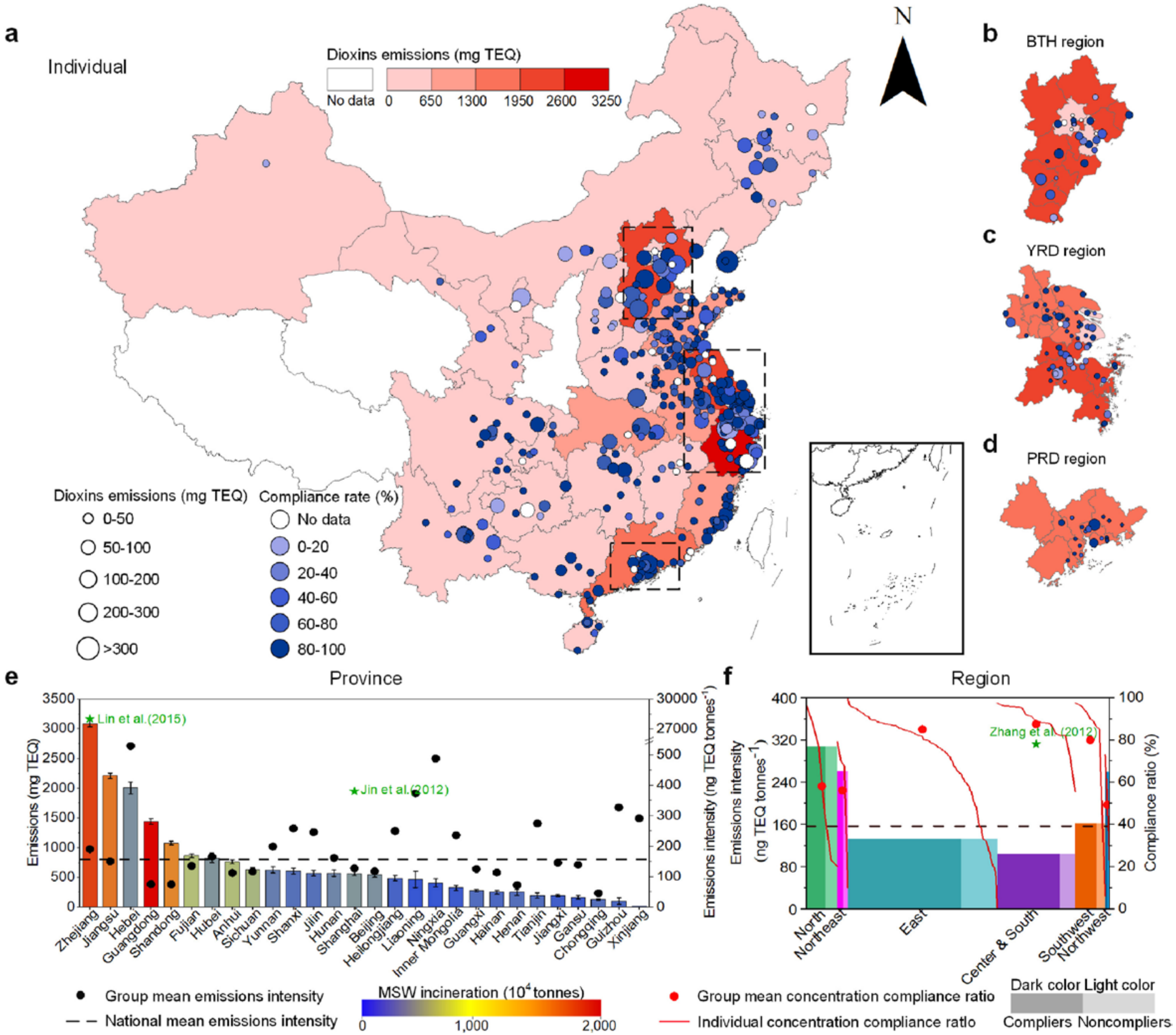


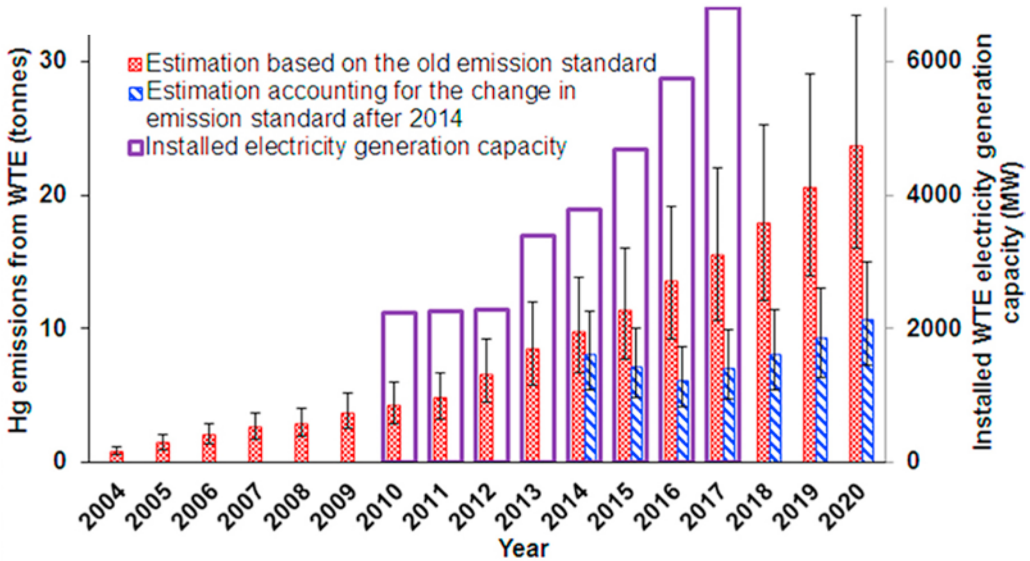
Figure 10.10: *Dioxins emission from Chinese municipal solid waste (MSW) incineration power plants. Notes: a-d, Individual emissions (mg toxic equivalent quantity (TEQ)) of MSW incineration power plants operating in 2018 in (a) mainland China (b) the Beijing-Tianjin-Hebei (BTH), (c) Yangtze River Delta (YRD) and (d) Pearl River Delta (PRD) regions. The dots indicate individual plants, with the size indicating emissions and the colour indicating the compliance rate (defined as the proportion of observations complying with the current standards); the coloured background denotes the provincial emissions. (e), Provincial emissions (left y axis), with the colours of the bars denoting the MSW incineration amount (104 t) and the error bars indicating 2 standard deviations of the emissions estimates. The black dashed line and points indicate the national and provincial means, respectively, of the emissions intensity (ng TEQ tonnes 1; right y axis). (f), Regional emissions, where the colour of the bars indicates the region, the height indicates the emissions intensity (left y-axis), the width is proportional to the MSW incineration amount, the area is proportional to the emissions level, and the shaded area is proportional to the emissions of standards compliers (dark) and noncompliers (light). The black dashed line indicates the national mean emissions intensity; the red curves and points indicate the individual and regional concentration compliance ratios (the exceedance of the smokestack concentration standard; right y axis), respectively. The green asterisks denote the previous sparse observations of the emissions intensities in the associated provinces (Jin et al., 2012; Lin et al., 2015b) (e) and regions (Zhang et al., 2012) (f). (Source: Guo et al., 2023)*

Photo 10.16: *Landfill with fly ash in big bags in Wuhan, China.*
Fly ash can be a source of dioxin contamination in the environment.
Photo: Jindřich Petrlík, Arnika, June, 2016.

Hu et al. (2018) highlights the increasing significance of WtE incineration as a source of mercury emissions in China. It provides an overview of mercury control practices at WtE facilities, estimates current mercury emissions and predicts future trends, emphasizing the need for measures to reduce these emissions. In 2016, WtE incineration in China was estimated to emit around 6.1 t of mercury, with projections indicating a rise to 10.6 t by 2020 due to rapid industry growth (see Figure 10.11). The adoption of stricter emission standards and the implementation of the Minamata Convention on Mercury are expected to help mitigate this increase in the long term. However, uncertainty remains in mercury emission inventories due to limited data availability, underscoring the necessity for improved monitoring and quantification of mercury content in MSW and the efficiency of air pollution control devices (APCDs) at WtE facilities.

To address these challenges, China should focus on enhancing recycling and waste-sorting programs to prevent mercury-containing waste from entering incinerators. Additionally, efforts should be directed towards improving data collection and analysis to better understand mercury emission factors and removal efficiencies at WtE facilities (Hu et al., 2018).

Figure 10.11: *Graph summarizes projection of mercury emissions from WtE plants in China by Hu et al. (2018).*



11. Waste incineration and civil society – case studies

Civil society can also be involved in waste incineration projects. The strategies used vary depending on the country and the specific situation and are described in more detail in the following case studies. These include studying the environmental impacts of incinerators and disseminating information to others who are interested in the issue or seeking guidance, creating petitions and collecting signatures, which can increase pressure on local politicians and the company responsible for the incinerator. Community Information Systems (CIS) are an important part of achieving environmental justice for frontline communities especially those who face the disproportionate impact of hazardous waste and waste incineration pollution (Lloyd-Smith, 2009). These strategies can be used even before the incinerator project enters the environmental impact assessment process, where they can apply lessons learned and propose alternatives for waste management. The public can participate in public hearings related to the EIA or organize their own discussions and engage with experts on waste issues, incinerator-environment relations, or environmental lawyers. However, in all the cases presented below, these are long-term issues with their own local specificities. We have already mentioned the activities of CSOs in the chapters 8.1.2, 8.1.3 or 10.3.4. Protests are common also in countries, which are not described in the following chapters.

Photo 11.1 *Protests have taken place to oppose the expansion of Edmonton incinerator in England. (Sources: Tranah, 2021; Allin, 2021)*

Photo 11.2 *Protests have taken place to oppose the expansion of Edmonton incinerator in England. (Sources: Tranah, 2021; Allin, 2021)*

Photo 11.3 *Photo from the report of the anti-incineration protests in Gipuzkoa from Basque Zero Waste Europe member Zero Zabor. (Source: ZWE, 2017)*

11.1 Spain

In Spain, CSOs undertook a variety of specific actions to combat the environmental and health risks posed by co-incineration. These actions included legal challenges, education and awareness-raising activities and building networks and coordination also on international collaboration.

They specialized in developing legal procedures to challenge environmental authorizations for co-incineration, often enlisting the support of professional lawyers and toxic experts. This approach proved successful on several occasions, demonstrating the movement's ability to engage effectively with the legal system to protect community health and the environment.

The movement also focused on educating the public about the dangers of waste incineration in cement plants. They organized awareness campaigns with the help of health, toxic, and waste management experts to increase social pressure and support for their cause. These activities aimed to raise understanding among the wider population about the risks of co-incineration and the benefits of alternative waste management strategies.

They prioritized local campaigns to prevent co-incineration in their communities but also devoted significant efforts to create coordination structures at regional, national, and international levels. This strategic networking helped to counteract criticisms of NIMBYism (Not In My Backyard), showing that their concerns were not just local but part of a broader environmental and health issue. The Spanish network against waste incineration in cement plants was formally established in Madrid in 2009, leading to annual gatherings and the development of a cohesive movement that has strengthened over time.

The movement's reach extended beyond Spain, with international gatherings against co-incineration taking place in Italy and Spain, involving representatives from various countries. These events facilitated the exchange of information and strategies, bolstering the movement's global stance against waste incineration and aligning it with broader environmental justice and Zero Waste goals.

11.1.1 Coimbra

The case study of the protest against co-incineration of dangerous industrial waste in Souselas, Coimbra, Portugal, was described by Matias (2014).

The Souselas community's fight began in 1998 against the proposed co-incineration project due to concerns over environmental and health impacts. This resistance movement highlights the deep engagement of local populations with environmental issues and their ability to influence national debates and agendas. The protest not only questioned the relationship between political decision-making, scientific knowledge, and citizens' participation but also demonstrated the transformative power of collective action in challenging and redefining political strategies and outcomes. The movement's success in mobilizing a wide range of actors, from local communities to national scientific communities and opposition parties, underscores the potential of grassroots activism to effect change and promote more inclusive and deliberative forms of public participation in environmental decision-making processes.

11.2 Ireland

11.2.1 Galway

In Galway, Ireland (Davies, 2005), the Galway Safe Waste Alliance (GSWA) spearheaded a dynamic campaign against the proposed municipal solid waste incinerator, drawing on a blend of local actions and global networks to amplify their message. GSWA's strategy included lobbying local councils, orchestrating public marches, gathering thousands of petition signatures, and facilitating community submissions to local authorities. These efforts were aimed at fostering widespread public opposition and influencing local government decisions. The alliance's approach was multifaceted, combining traditional forms of protest with innovative tactics like leveraging international advocacy networks and utilizing information and communication technologies (ICTs) for broader engagement and knowledge sharing.

GSWA's campaign was not only about local resistance but also about connecting with and learning from global movements against incineration. By joining forces with transnational advocacy networks such as the Global Anti-Incineration Alliance (GAIA) and the Zero Waste Alliance, GSWA tapped into a wealth of expertise and support, showcasing the power of global solidarity in local environmental struggles. This international collaboration enriched the campaign with diverse strategies and information, enhancing the local movement's credibility and impact. The utilization of ICT further facilitated the sharing of resources and strategies across borders, enabling GSWA to present a well-informed and globally connected front against the incineration project in Galway. This case exemplifies how localized environmental activism can effectively leverage global networks to challenge and influence waste management policies and practices.

11.2.2 Carranstown

The Carranstown anti-incineration campaign in Ireland, detailed in the study by (Davies, 2008), presents an insightful narrative of civil society activism against waste management policies favoring incineration. This campaign emerged in response to the local authorities' plans, influenced by the 1996 Waste Management Bill, to incorporate municipal solid waste incineration into their waste management strategies. Despite facing significant challenges such as a lack of resources, opposition from government and industry, and the marginalization of their voices in policy discussions, the Carranstown campaigners mobilized community support, leveraging petitions, public meetings, and legal challenges to contest the planning permissions for the incinerator.

Their efforts highlight the complexities of engaging with and influencing waste management policies in a context where economic development priorities often overshadow community and environmental concerns. Despite the obstacles, the campaign fostered community solidarity, raised awareness about alternative waste management strategies, and exemplified the critical role of grassroots activism in advocating for sustainable environmental practices. The case study underscores the potential impact of civil society in shaping waste governance, even as it reflects on the constraints that limit such influence, including financial limitations, strategic challenges, and the broader political and economic forces at play.

11.2.3 Jeremy Irons involvement

Oscar winner Jeremy Irons got actively involved in anti-incineration campaigns. Oscar-winning actor Jeremy Irons, who owns the nearby 15th century Kilcoe Castle, was among those who spoke out against the development at a four-week public hearing in Cork last September of 2003 (Riegel, 2003). He also stared documentarry movie about waste incineration. In response to the question, *"What do you want people to do once they've seen the film?"*, Jeremy Irons says: *"I would like them to research whether there is a waste-to-energy plant [incinerator] planned for their area, and, if there is, to oppose it. If there is not, then to discover how their local council deals with their waste…I would like them to use their ingenuity to discover how they can reduce waste both at home and in their workplace…And I would like them to tell their friends to see Trashed."* (Shlomo, 2012).

Photo 11.4: *Oscar winner Jeremy Irons got actively involved in anti-incineration campaigns. Photo was taken during a sell-out screening of the documentary, Trashed, at Vue cinema in Stroud, England, another site where local community opposed waste incinerator (Stroud News & Journal, 2015).*

11.3 China

In China, Lang & Xu (2013) provided an in-depth look at how grassroots activism against waste incineration projects has evolved within the context of China's rapid urbanization and environmental challenges. Highlighting case studies from Beijing, Guangzhou, and Wujiang, the study reveals the successful tactics employed by protestors, including leveraging social media, engaging in public demonstrations, and utilizing legal channels to challenge government decisions. These campaigns not only resulted in the postponement or cancellation of incinerator projects but also prompted a broader discourse on ecological modernization and the importance of public participation in environmental governance. It showcases the power of community mobilization and strategic activism in influencing policy decisions and advancing environmental protection efforts in an authoritarian political context. The study details three significant anti-incinerator campaigns in China, each highlighting different aspects of public resistance and its impacts:

Beijing: The campaign against a proposed incinerator in the densely populated suburb of Liulitun showcased the power of public opposition through organized protests and the use of social media to rally support. This movement ultimately led to the government reconsidering the placement of waste-to-energy facilities in close proximity to residential areas.

Photo 11.5 *Activists protest waste-to-energy incinerators in the southern Chinese city of Guangzhou. Photo: e360.yale.edu, 2017*

Guangzhou: In Panyu District, activists employed a multifaceted approach that included public demonstrations, extensive media campaigns, and legal actions to challenge the environmental assessment processes. Their efforts resulted in a significant delay of the incinerator project and forced a more transparent and participatory reconsideration of waste management policies.

Wujiang: The campaign in Wujiang was notable for its emphasis on legal challenges against the local government's decision-making process regarding the incinerator project. Activists successfully highlighted issues of procedural transparency and environmental risk assessment, contributing to a broader debate on governance and public engagement in environmental decisions.

These cases collectively underline the effectiveness of grassroots activism in influencing environmental policy and project implementation, demonstrating varied strategies from legal action to media engagement that can be adapted by other communities facing similar challenges.

11.4 Portugal

The conflict over co-incineration of hazardous industrial waste in cement kilns in Portugal, as detailed in Jerónimo & Garcia (2011), illustrates a pivotal instance of environmental activism and its broader societal impacts. The government's decision to pursue co-incineration sparked widespread protests from residents, environmental organizations, and scientific communities, leading to a deepened dialogue on environmental health, democratic participation, and the legitimacy of scientific expertise. Activists and local communities, initially motivated by the immediate threat to their health and environment, utilized various strategies including legal actions, public demonstrations, and the mobilization of counter-expertise to challenge and ultimately transform governmental waste management policies. This collective opposition was instrumental in steering the controversy towards more sustainable waste treatment methods, emphasizing the role of civic engagement and public scrutiny in shaping environmental governance.

The narrative of this conflict reveals the complexities of negotiating environmental risks, the power dynamics between government, industry, and civil society, and the transformative potential of grassroots activism. Despite the government's attempt to legitimize co-incineration through expert consultations, persistent public resistance highlighted the limitations of technocratic approaches to environmental decision-making and underscored the importance of inclusive, transparent, and participatory processes. The shift towards a multifunctional waste treatment method, emerging from this protracted conflict, exemplifies how sustained civic action can lead to more environmentally sound and socially acceptable solutions. This case study not only reflects the challenges of managing hazardous waste in a way that respects public health and environmental integrity but also demonstrates the critical role of democratic engagement in navigating the intersections of science, policy, and community values.

11.5 South Africa: Durban

The case study of activism against the Mondi incinerator in Durban, South Africa, provides a compelling narrative of environmental justice efforts by (Leonard & Pelling, 2010). In 2002, the local community opposed Mondi Paper's proposal to construct a multi-fuel boiler (MFB), which activists argued was essentially a polluting incinerator. Despite initial legal victories, the government eventually granted Mondi permission. The campaign's challenges included inconsistent mobilization, weak collaboration among local leaders, and reliance on legal strategies without sufficient community engagement. This story highlights the complexities of environmental activism, including the difficulties in sustaining mobilization and the strategic choices between legal battles and community protests. The Mondi case serves as an instructive example for activists globally, emphasizing the importance of consistent community mobilization, education, and the need to engage in both legal and direct-action strategies.

11.6 Czech Republic

11.6.1 Civil Society Engagement in the Case of the Prague – Malešice Municipal Waste Incinerator

The case of the Prague – Malešice MSWI in Czechoslovakia, and in the Czech Republic later on, illustrates the significant role of civil society in advocating for environmental protection and public health. This case is described and analyzed by Konopásek et al. (2004). Originating from decisions made in the communist era, the incinerator's construction faced opposition due to its inadequate environmental standards. However, after the fall of the regime in 1989, civil society groups, particularly the ecological section of the Civic Forum, reignited the debate, calling for improved waste management strategies and better air pollution control technologies for the incinerator (Konopásek et al., 2004).

Photos 11.6 – 11.7: *The WtE plant in Liberec was another controversial project in the Czech Republic (see case study in subchapter 9.1.1). Children of the Earth (Děti Země), together with local civic activists, organized a public protest against its start of operation in September 2000 when the WtE plant did not meet the dioxin emission limit. Over 500 citizens took part in that protest. Photos: Jindrich Petrlik and Simona Jašová, September 2000.*

Despite legal challenges and protests, construction proceeded, leading to administrative errors being identified in the construction permits issued before 1989. This rendered parts of the incinerator illegal until 1997 when a retroactive construction permit was granted, allowing the incinerator to operate.

Amidst ongoing construction and legal battles, the NGO Children of Earth (COE) emerged in this case in mid-1990s, focusing on the issue of dioxin emissions. Although initially aiming to halt the incinerator's operation entirely, COE shifted its focus to advocating for improved air pollution control technology to reduce dioxin emissions. Their campaign led to legislative changes, including the establishment of dioxin emission limits in Czech legislation, even before EU requirements (Konopásek et al., 2004).

The COE campaign brought public attention to the dangers of dioxins and pressured the incinerator's management to address the issue seriously. Eventually, the incinerator adopted new technology, becoming the only household waste incinerator in the country to meet the legally required dioxin emission limit of 0.1 $ng.m^{-3}$. However, doubts persisted regarding the effectiveness of the new technology and concerns about potential manipulation of emission results (Konopásek et al., 2004).

As the COE campaign concluded, a new organization, Arnika, continued the anti-dioxin efforts, achieving further successes such as the creation of a Pollutant Release and Transfer Register (PRTR) and ratification of the Stockholm Convention on POPs by the Czech Republic (Petrlik et al., 2023). Despite these victories, the incinerator's improved technology and public relations efforts diminished the public's interest and activism in the issue.

Photo 11.8: *WtE plant in Prague – Malešice remained problematic: In October 2021 there was a fire (iDnes, 2023) in technological part (see also Chapter 9.2). Photo: twitter/HasiciPraha.*

The establishment of the Civic Commission for the Control of the Incinerator demonstrated civil society's ongoing efforts to hold the incinerator accountable. However, without the active involvement of groups like COE, local citizens felt sidelined and disempowered, leading to a decline in civic engagement and a sense of futility in public involvement.

In conclusion, the Prague - Malešice incinerator case highlights the pivotal role of civil society in advocating for environmental protection and public health. While successful in achieving legislative changes and improvements in emission standards, the decline in activism after initial victories underscores the challenges of sustaining public engagement in long-term environmental campaigns (Konopásek et al., 2004).

11.6.2 From Opposition to Waste Incineration to Promotion of 3R

A CSO Arnika, established in 2001, continued the anti-dioxin efforts of the Children of the Earth (COE), achieving further successes such as the creation of a Pollutant Release and Transfer Register (PRTR) and ratification of the Stockholm Convention on POPs by the Czech Republic (DiGangi, 2011; Petrlik et al., 2023). These were the fundamental goals of the "Toxics Free Future" campaign led by Arnika from 2001 to 2008. It also involved assisting local civic initiatives in resisting projects for new incinerators, including pyrolysis, waste gasification, and plasma incineration technologies, which are now encompassed under the term "chemical recycling." Among such projects was the attempt by BDW Line company to operate a hazardous waste incinerator in Lysá nad Labem, which would have involved burning waste containing PCBs (Marcanikova et al., 2005; Skalsky et al., 2006), among other substances. The project was opposed

Photo 11.9

Photos 11.9 – 11.13: *Probably the largest public hearing in the EIA (Environmental Impact Assessment) process for any project took place in the case of AVE CZ's efforts to reconstruct and put the hazardous waste incinerator in Pardubice back into operation. The meeting, attended by 7,000 people, had to take place at the hockey stadium (Zlinský, 2022). Photos: Michal Klíma (MAFRA) – photo 11.9, Jiří Sejkora (Deník) – photos 11.10 – 11.12, and Jindrich Petrlik (Arnika) – photo 11.13.*

Photo 11.14: *The event of the Arnika association in 2002 pointed out that the hazardous waste incinerator in Lysá nad Labem is a source of emissions of toxic substances, including dioxins. Photo: Jindřich Petrlík, Arnika.*

by the local civic group Lysin (Arnika, 2023). The incinerator is effectively non-operational today, ending in April 2013 with an attempt to operate an untested dioxin filter (see Photo 7.10); (KÚSK, 2009).

For example, in 2004, Arnika helped the local civic association A21 in the Polička engineering area to halt a project to build plasma technology, which was intended to dispose of waste containing PCBs. The untested Russian technology was planned for an otherwise very clean environment, an area of drinking water resource accumulation (Arnika, 2004a). The construction plan for this technology was eventually halted.

Since 2011, Arnika has initiated the "Don't Burn, Recycle" project, within which it continues to support local civic initiatives opposing waste incineration projects. Cases of planned WtE plants, waste incinerators, or "chemical recycling" plants, which Arnika has addressed, can be found on a summary page of their cases (Arnika, 2023). Some projects have been halted during the environmental impact assessment phase based on well-founded objections submitted by both Arnika and local civic groups, municipal representatives, and citizens.

From protests against waste incinerators, local civic initiatives have formed the Coalition Pro3R, which since 2012 has been creating a counterbalance to political and lobbying efforts to support waste incineration and landfilling, including valuable secondary raw materials or compostable components. The coalition's name itself encapsulates its goal: "3R" in the coalition's name is an abbreviation for the English words "reduce, reuse, recycle." The coalition possesses knowledge and arguments from the field of sustainable waste management and has examples of good practice from the Czech Republic. The coalition submits comments and proposals on conceptual and legislative materials at the national level and occasionally at the European level (Koalice Pro3R, 2024).

11.7 India: Zero Waste Kovalam Project

The Zero Waste Kovalam project, spearheaded by Thanal in Kerala, India, emerged as a response to the looming threat of a municipal waste incinerator in Kovalam. Aligned with the principles outlined in international agreements such as the Stockholm Convention and the Strategic Approach to International Chemicals Management (SAICM), the project sought to embody zero waste resource management, waste prevention, substitution, and toxics reduction (DiGangi, 2011). Through a meticulous strategy, the project evaluated the potential of biogas plants for resource

Photo 11.15: *Kovalam is a pleasant touristic location on the sea shore in Kerala, India. Photo: Jindřich Petrlík, Arnika, August 2008.*

recovery while concurrently empowering women's groups with knowledge on alternative materials for plastics. This multifaceted approach was further fortified with the establishment of a "Zero Waste Center" in 2003, fostering endeavors such as poison-free farming, water conservation, and community capacity-building (DiGangi, 2011; Jayaraman, 2005; Vignesh et al., 2021).

Furthermore, the project's impact transcended just waste management, as evidenced by its contribution to localizing chemical-free food production and economic upliftment, as highlighted in the study by Shinogi et al. (2018). This holistic approach not only fostered a new, more sustainable and less toxic farming culture but also bolstered the economy of a small village, showcasing the potential for replicating successful community waste management models nationwide to mitigate the growing waste menace in both rural and urban India (Shinogi et al., 2018).

Photos 11.16 and 11.17: *The „Zero Waste Kovalam" project helped many local women to participate in the production of new products by recycling what would otherwise become waste. Photo: Thanal (11.16) and Jindrich Petrlik (11.17).*

Photo 11.18: *Here is an example of a piece of jewelry made by a cooperative of women in Zero Waste Centre Kovalam from recycled waste. Photo: Jindřich Petrlík, Arnika.*

Photo 11.19: *The team of the CSO Thanal based in the state of Kerala, India, is responsible for bringing the protest against the construction of a new waste incinerator into the "Zero Waste Kovalam" project, which taught the local community to prevent waste generation and recycle it. Photo: Thanal.*

Key milestones in the project's timeline further illustrate its evolution and impact:

- In 1998, the Kerala Tourism Department's announcement of plans to set up a waste incinerator in Kovalam sparked initial opposition.
- Throughout the late 1990s and early 2000s, Thanal and its partners, including Equations and Greenpeace, launched campaigns against the incinerator, eventually leading to the shelving of the proposal.
- In 2000, the concept of Zero Waste Kovalam was conceived at a meeting in Bangkok, marking the project's formal inception.
- Subsequent events, such as the launch of studies into garbage generation patterns and skill-sharing workshops, laid the groundwork for community engagement and capacity-building.
- The inauguration of various initiatives, including a Zero Waste Center and organic bazaars, underscored the project's commitment to sustainable practices and community empowerment.
- Notable achievements, such as the establishment of India's first zero waste ward in Muttakadu and the adoption of Zero Waste Tourism goals by the Philippines government, highlighted the project's broader impact beyond Kovalam (Jayaraman, 2005).

The Zero Waste Kovalam project not only influenced local agencies in Kovalam to adopt proper discard management practices but also spurred hotels to establish their own biogas plants for kitchen purposes. These biogas plants, integral to the "zero waste" approach, are functioning effectively, such as the one at Samudra Hotel, meeting the gas needs for the hotel's water boiler. Additionally, the project's impact extends to the Institute of Hotel Management and Catering Technology (IHMCT), where waste management has significantly improved, leading to benefits like reduced waste disposal issues and employment opportunities. Moreover, the initiative has enabled the reuse of various waste materials, leading to the creation of new products and economic opportunities, particularly for women (Dileep, 2007).

In essence, the Zero Waste Kovalam project stands as a testament to the power of community-driven initiatives in addressing complex environmental challenges and fostering sustainable development which started from opposition to a new waste incinerator proposal, and developed into citizens' driven community waste management.

11.8 Malaysia: Gabungan Anti-Insinerator Kebangsaan (GAIK)

Gabungan Anti-Insinerator Kebangsaan (GAIK), established in 2014, emerged from the collaboration of various organizations in Malaysia to resist the development of waste-to-energy (WtE) incinerators across the nation. Comprised initially of four organizations—Selamatkan Bukit Payong, Gabungan Anti Insinerator Cameron Highlands, Jawatankuasa Anti Insinerator Tanah Merah, and Jawatankuasa Bertindak Kuala Lumpur Tak Nak Insinerator (KTI)—GAIK aimed to persuade the Malaysian Government to halt WtE facility constructions and adopt a more sustainable waste management strategy: Zero Waste. Presently, GAIK comprises ten individuals and five non-government organizations (NGOs), steadfast in their opposition to incinerators (Abril, 2023).

Among their achievements, GAIK successfully lobbied against a WtE incinerator project in one state and influenced authorities in Kepong, Kuala Lumpur, to abandon a similar proposal. Despite these successes, challenges persist, including limited resources, difficulty tracking WtE proposals, and resistance from some residents fearful of reprisals (Abril, 2023). Tang et al. (2021) stated that *„the incineration plants have failed due to the opposition of the public"* in Malaysia.

GAIK collaborates with organizations such as GAIA, attending regional meetings to foster alliances and exchange knowledge. They recognize the interconnectedness of environmental issues and advocate for collective action to address them (Abril, 2023).

Photo 11.20: *Paul Connett who promotes zero waste philosophy in many communities across the world gave a presentation to GAIK in April 2019. (Source: Abril, 2023).*

Photo 11.21: *Residents of Rawang in Gombak district, who are opposing a plan by the Selangor state government to build an incinerator in Batu Arang, show a map of the area and the proposed plant site (Hassan, 2024). Photo: Hazlin Hassan, The Straits Times.*

There is also strong opposition against building of new municipal waste incinerator in Batu Arang (Selangor state), some 50km north-west of Kuala Lumpur, will have the capacity to burn 2,400 tonnes of waste daily (Hassan, 2024).

Residents of Batu Arang said the incinerator may jeopardise their health and safety through the emission of poisonous fumes such as cancer-causing dioxins close to homes, schools, mosques and temples located within some 700m to 2km of the proposed waste plant. Ms Esther Woo, spokeswoman for Jaringan Rawang Tolak Incinerator (JRTI), a coalition of resident associations that oppose the move, said she is worried about the health of her children who attend a school nearby (Hassan, 2024).

Malaysia's recycling rate was 35.38 per cent as at 2023, and the government aims to hit 40 per cent by 2025, and it expects similar development as in Europe to convert most of municipal waste from landfilling by 2050 (Hassan, 2024).

11.9 Australia

Australia's resistance to waste incineration goes back to at least 2003 in Tasmania when local campaigners, The National Toxics Network and Greenpeace successfully defeated the Test incinerator. This was followed by successful campaigns in Western Australia (WA) which held off a number of proposals for more than a decade, up until 2015 and 2020 when two major waste incinerators were finally approved. As of April 2024, neither of these incinerator projects are operational yet, with numerous technical and legal delays forcing massive cost blowouts.

The industry has turned its attention heavily to Australia's east coast where numerous projects are proposed. None have yet been built or are operational.

Western Sydney community campaigners launched a successful campaign bringing the issue of incineration directly to the NSW and federal Parliament with a 10 000+ strong petition to oppose what would have been the largest WtE plant in Australia. Zero Waste Australia (a campaign of the NGO – Toxics Free Australia) provided a critical Community Information System (CIS) utilising independent research and reports to challenge narratives about the safety and real-life operations of incinerators. A strong focus on collaborations with international advocacy organisations like the International Pollutants Elimination Network (IPEN), the Global Alliance for Incinerator Alternatives (GAIA), Zero Waste Europe (ZWE) and US groups like Energy Justice Network (EJN) and independent

***Photo* 11.22** *Massive protests in Sydney. Photo: Jane Bremmer, TFA.*

experts like Professor Paul Connett, delivered successful outcomes in terms of accessing latest studies, strategies and information to support the Australian campaign. Developing factsheets and other public interest materials, convening public meetings, webinars, events and widespread community resistance actions, at every opportunity from the most local level to the national, contributed to the success of this campaign.

The Australian Capital Territory (ACT) now has a policy that prohibits building of WtE plants and promotes Zero Waste Policy. WtE is also prohibited in the Sydney CBD and the Victorian government have set a 1 million tonne cap on waste incineration.

“Where the Australian Government sees efforts towards stronger regulations to address the risks of chemicals and pollution, communities and civil society denounce the capture of the State for the benefit of mining, oil, gas, agrochemical and other corporate interests”, said Marcos Orellana, UN Special Rapporteur on toxics and human rights, following an official visit to Australia (UNHRC, 2023). “Draconian restrictions on the right to peaceful protest in several states aggravate the distance between State and society,” said Orellana. Proposed petrochemical, offshore oil and gas, hydraulic fracking, and waste incineration projects pose serious health, water, agricultural and climate concerns (UNHRC, 2023).

The Australian experience shows that the determination of those most affected communities when supported by international campaign groups and independent experts, with access to reliable information, can even defeat incinerator projects in a country where environmental justice is not a priority. Promoting knowledge directly into the hands of the people, is a powerful antidote to the environmental injustice that waste incinerators bring.

12. Final summary

In this study, we have highlighted the key impacts of waste incinerators on the environment, human health and the economy. As can be seen, waste incineration contributes to the disruption of the Planetary Ecosystem, particularly through global chemical pollution (Chapter 4.2), greenhouse gas emissions (Chapter 4.1), biodiversity loss (Chapter 4.3), and biogeochemical flows (Chapter 4.4).One of the biggest problems associated with waste incineration is dioxins, which have serious negative effects on human health (Chapter 6), including cancer, damage to the immune system, reproductive problems and developmental defects (Chapter 5.1.1).

Despite strict emissions limits, waste incinerators are responsible for almost one fifth of all dioxins released into the air in the European Union (Chapter 5.1.1.1). It is evident that pyrolysis and plasma gasification of waste, as well as technologies now summarized under the name "chemical recycling" of plastic waste, do not represent functional substitutes for waste incineration and are similarly problematic in terms of environmental impacts or have different negative effects than "classical" waste incinerators (Chapters 3 and 6). The most suitable alternatives in the field of waste management therefore appear to be greater investment in waste prevention, sorting and recycling, which primarily includes bio-waste composting (Chapters 8 and 9.1.3). For municipal waste, the most appropriate solution is to set up systems called zero waste (see Chapter 8.1), even though some residual waste still remains. However, there is no need to build expensive, largescale waste incinerators for what is the smallest fraction of the waste stream – residual waste. Waste prevention (reduction), reuse, recycling, and composting (for biowaste) have proven to be more environmentally friendly and cost-effective approaches to waste management in all aspects. Developed countries with further growth of waste incineration capacity are in danger of having to import waste, because cities will become dependent on WtE as heat sources (Chapter 10). The rapid increase in waste incineration capacity in China has brought a whole host of problems, including impacts on health and the environment, and the need for continuous economic support from the state (Chapter 10.3).

Medical waste does not have to be incinerated to decontaminate infectious waste, there are a number of proven non-incineration technologies. Even in the healthcare sector, it makes sense to sort waste, not all of it is infectious (Chapter 8.3). POPs in hazardous waste can be destroyed and decontaminated far more effectively by so-called non-incineration technologies (Chapter 8.3.7), including fly ash from incinerators containing high concentrations of dioxins (Chapter 3.3.1). It is absolutely necessary to avoid incineration of waste containing mercury, which easily escapes even at normal (room) temperatures. Among other things, it is completely contrary to the Minamata Convention on mercury, which the Czech Republic and many other countries have ratified (Chapter 8.3.6).

Despite the claims of waste incineration proponents and governments that the EU Best Practice Standards for waste incineration operations are robust and protect human health and the environment, the fact is that the most dangerous substances (such as dioxins or mercury) that are produced during combustion are monitored in emissions only twice a year, and many of them are not monitored at all (Chapters 3.1 and 5.1.1.1). Due to emission limits, incinerators must clean their flue gases. However, this creates another flow of toxic waste in the form of ash and air pollution control (APC) residues, which should require strict handling and treatment regulations as a hazardous waste. (Chapters 3.3 and 5.1.1.3). The failure to adequately account for and regulate fly ash, and therefore, the dioxins and other POPs it contains, significantly contributes to exceeding the planetary limits of chemical pollution (Chapter 4.2). The amount of unregulated dioxins in fly ash is out of control and corresponds to the maximum tolerable intake of these substances for the population of up to 133 planets of the Earth.

In addition to dioxins, other toxic substances such as brominated dioxins, PFASs, polychlorinated biphenyls and other organic substances are also released during waste incineration (Chapters 5.1 and 5.2). Brominated dioxins have similar toxicity to dioxins and similar effects on human health, yet they are not yet measured in flue gas from incinerators (this obligation is new), not to mention their concentration in solid waste incineration residues (Chapter 5.1.2). Waste incinerators also release significant amounts of mercury and other toxic metals into the environment with negative effects on health (Chapter 5.3). These metals are released into the air to a lesser extent but end up mainly in solid residues such as fly ash and APC residues and bottom ash. Unburnt plastic particles, known as microplastics, also remain in the bottom ash (Chapter 4.2). There are also many other potentially hazardous substances that are unknown or have no limits set for waste incinerators effluents (Chapter 5). This is problematic when considering the further use of residues from waste incinerators, especially in construction and agriculture and adds to the ever increasing burden of pollution, which is already exceeding planetary limits.

Despite the whole range of toxic substances that waste incinerators leave in emissions into the air and water, but above all in waste, slag, bottom ash and fly ash, the assessment of their impact on the health of residents living in the vicinity remains a controversial topic (Chapter 6). Although there have been a number of studies demonstrating their negative impact on human health, there are also a number of studies that have not proven this impact. Chapter 6 provides a rough cross-section of the issue of assessing the impact of incinerators on human health. However, it also concerns the assessment of local food contamination (Chapters 3.4. with case studies in 3.5., 5.1.1.3.3 and 5.1.5.1).

Waste incinerators do not only process materials that cannot be recycled, but they also compete for the same funds and materials as recycling facilities. At the same time, waste incineration means the loss of valuable raw materials, which must be extracted, produced and transported again. They thereby discourage the conservation of resources and their maintenance in a circular economy. Incinerators waste energy that was invested in the production of products that ended up in waste and in their collection. For these reasons, waste incineration was removed from the EU Taxonomy and from the list of financing sustainable activities.

The construction of WtE and waste incinerators is heavily dependent on the financial support of the public sector (Chapters 9 and 10.2.4), which often paid extra for their construction or is paying extra. WtE receives support from EU funds in a hidden way. In addition to the initial investment costs, incinerators (WtE) swallow a lot of repair and maintenance funds, separate to the expenses related to the effects of incinerators on human health and the environment (Chapters 9.3, 9.4 and 9.5). Other financial costs are related to accidents, mostly fires, which occur quite often in waste incinerators, and which often destroy a large part of the equipment and threaten the health of residents living in the vicinity (Chapter 7). In environments surrounding waste incinerators, soil contamination with toxic substances (primarily dioxins) and the related contamination of domestic produce, poultry and/or livestock were also observed. Their research alone represented additional costs (Chapters 3.4 and 9.5).

Waste incinerators represent an outdated, unsustainable, and expensive way of managing waste that has negative effects on the environment, human health, and even the entire planetary ecosystem.

Assessing the impacts of certain human industrial activities on human health is never simple, considering the multitude of factors influencing our health status. This is especially true for the impacts of waste incineration, and therefore, studies attribute a share of increased illness among people living nearby, even to modern waste incinerators equipped with better filtration systems, while other studies have found no such influence (Chapter 6). A significant portion of recent studies emphasizes the assessment of all exposure pathways, particularly exposure to toxic substances accumulating in local food sources (from backyard animal husbandry or locally grown crops). This exposure pathway is often overlooked in official evaluations of the effects of new waste incinerators on human health (Chapter 6).

Traditionally, it's assumed that landfills pose a greater risk of fire accidents than incinerators, but recent statistics from France challenge this notion. An analysis comparing accidents during waste incineration with other waste management methods reveals that France has experienced more severe accidents at a higher rate in incinerators than at landfills, despite more incidents being investigated at landfills. Firefighters in France intervened more frequently in incidents involving hazardous substance leaks or explosions at incinerators between 2005 and 2014. Additionally, we include descriptions of several case studies, highlighting also intriguing water contamination following fire extinguishments (Chapter 7).

Civil society engages in waste incineration projects through diverse methods like environmental studies, petitions, and public discussions to propose alternatives and pressure stakeholders (Chapter 11). These proactive efforts aim to influence decisions before and during the environmental assessment phase. Despite varying by region, such activism is a widespread response to the threat of waste incineration and other waste management impacts (Chapters 11.1–11.9).

Modern incinerators are trying to be included in the circular economy system and are therefore looking for ways to use the bottom ash, which remains up to one third of its original weight from the incinerated waste (Chapter 3.3.3). In this regard, too, for example, the oversized Dutch incinerators have already hit an imaginary ceiling, and the Nobel Prize winner Ernst Worrell therefore described the Dutch roads built from incinerator bottom ash as "linear landfills" (Chapter 3.3.3.1).

Incinerating waste, while producing the energy that powers our modern, energy-intensive lives, also actively contributes to the cycle of climate change. Emissions of carbon dioxide, created by the combustion process, are one of the driving forces behind the greenhouse effect, which has serious consequences in the form of global warming and climate change. By 2050, the conversion of plastic waste to energy (including incineration in WtE) will lead to greater emissions of carbon dioxide than the burning of fossil fuels. Energy utilization of waste therefore does not help solve global climate change but contributes to it and thus represents a dead end in replacing coal (Chapter 4.1).

While waste seems to magically disappear, the reality is that by burning waste we destroy valuable raw materials that we can no longer reuse, recycle or compost, while an unusable third of the original weight of waste remains enriched with toxic substances. By operating incinerators, we support linear waste management, which requires a constant supply of waste and conversely, generates significant volumes of hazardous waste as a result.

The choice of waste management technologies our governments make inherently comes with significant local and global impacts. The destruction of finite resources by entrenching waste incineration leads to a linear instead of circular economy. Waste incineration also pollutes the environment, endangers human health, and leaves a legacy of hazardous waste. The disproportionate impacts of waste incineration pollution on indigenous communities cannot be ignored and represents a violation of their human rights. Indeed, as the UN Special Rapporteur on toxics and hazardous waste recently reported after speaking with impacted communities in Australia, "Waste incineration is the end of the line for fossil fuels" (Chapter 11.9). However, as the examples in this book also show, there are safer, more effective solutions found in non-combustion technologies, Zero Waste city models and the Circular Economy.

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