Australian Government
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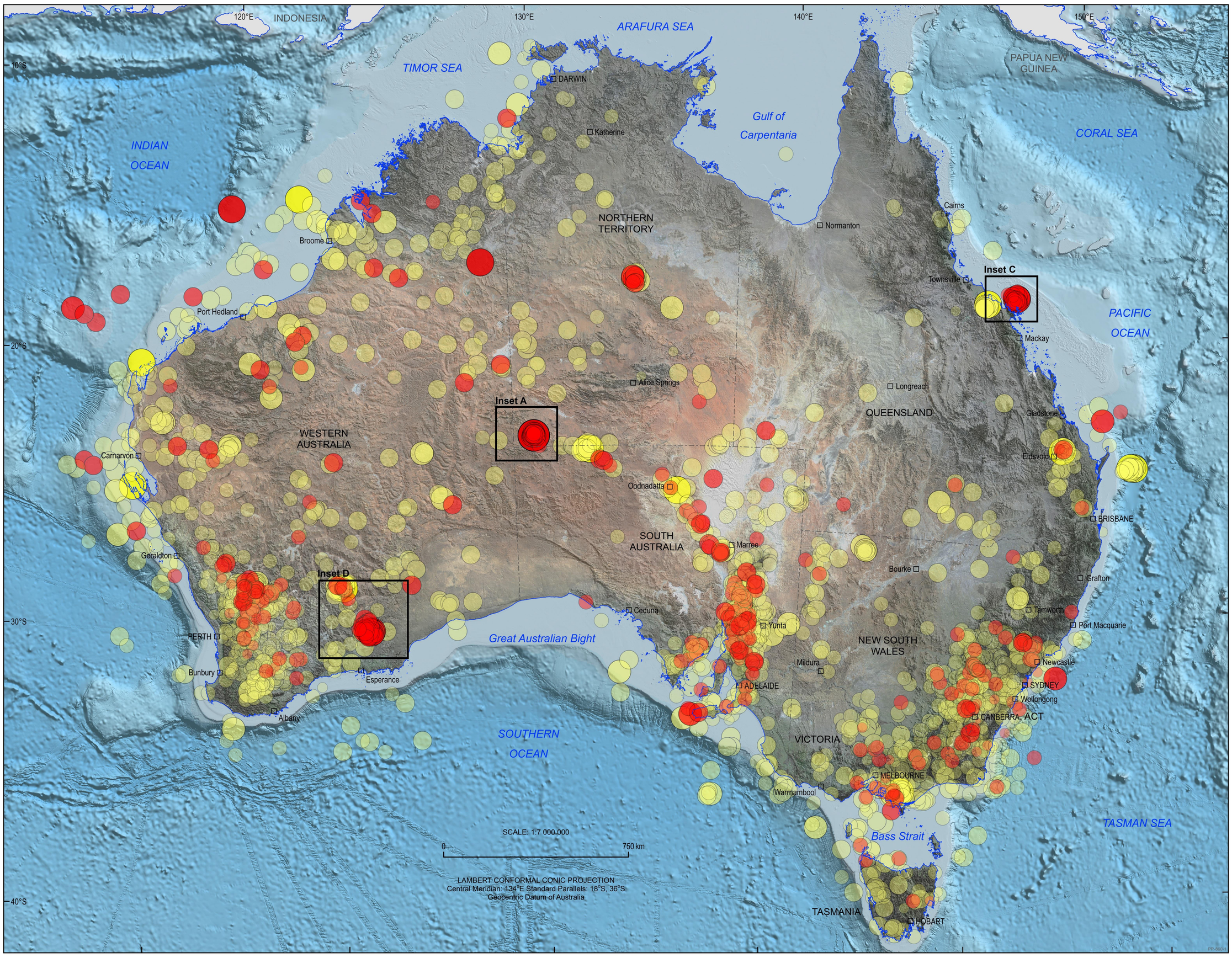
Australian Earthquakes 2016

A. Thom, Geoscience Australia.

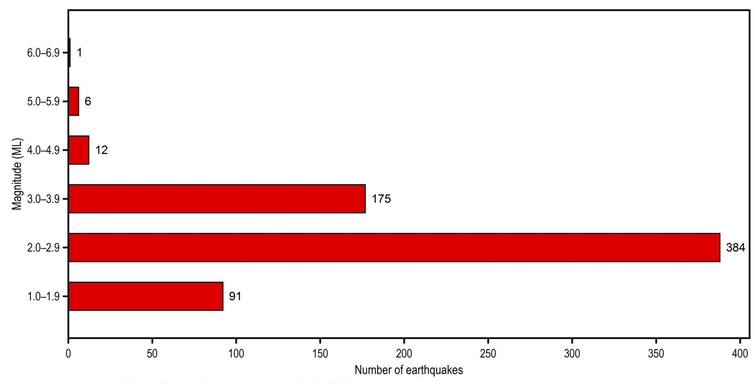


Earthquake data for 2016
Magnitude (ML)

- 0.0–0.9
- 1.0–1.9
- 2.0–2.9
- 3.0–3.9
- 4.0–4.9
- 5.0–5.9
- >6.0

Earthquake data for the previous 10 year period Magnitude (ML)

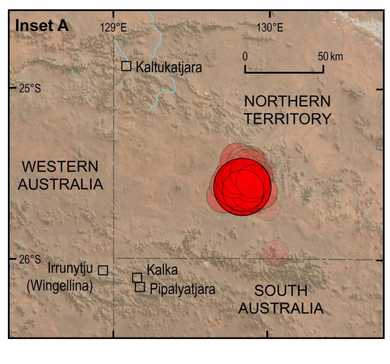
- 0.0–0.9
- 1.0–1.9
- 2.0–2.9
- 3.0–3.9
- 4.0–4.9
- 5.0–5.9
- >6.0

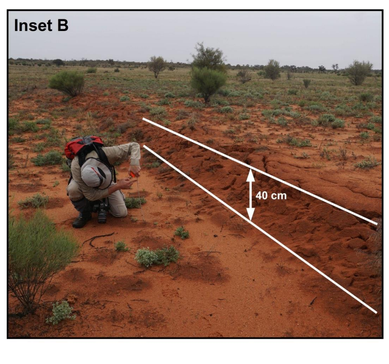


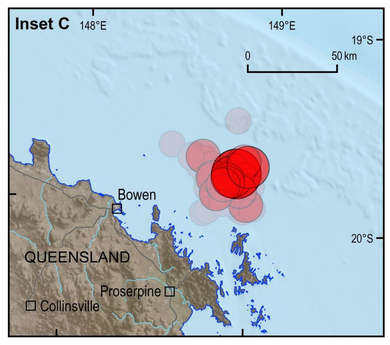
Number of Australian earthquakes by magnitude 2016

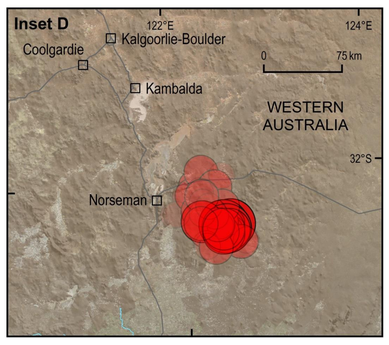
Magnitude (ML)	Date	Time (UTC)	Latitude	Longitude	Depth (km)	Location
6.1	20 May	18:14:02	-25.579	129.832	0	Petermann Ranges, NORTHERN TERRITORY
5.8	18 Aug	04:30:07	-19.768	148.863	0	Offshore northeast of Bowen, QUEENSLAND
5.6	8 July	09:40:50	-32.458	122.511	0	East of Norseman, WESTERN AUSTRALIA
5.3	10 May	09:44:35	-16.397	118.690	10	Offshore northwest Australia
5.1	6 Nov	09:54:31	-19.106	127.987	10	Southeast of Hall's Creek, WESTERN AUSTRALIA
5.1	28 May	16:38:44	-32.460	122.438	0	Southeast of Norseman, WESTERN AUSTRALIA
5.0	28 May	15:30:26	-32.497	122.466	0	Southeast of Norseman, WESTERN AUSTRALIA
4.5	19 Jun	01:54:09	-35.916	136.450	10	West of Kangaroo Island, SOUTH AUSTRALIA
4.4	13 Aug	15:31:13	-23.821	152.867	10	Offshore northeast of Bundaberg, QUEENSLAND
4.4	8 Jun	02:01:09	-32.506	122.493	0	Southeast of Norseman, WESTERN AUSTRALIA

Australia's top 10 earthquakes 2016









Earthquake data for 2016
Magnitude (ML)

2.0–2.9 3.0–3.9 4.0–4.9 5.0–5.9 >6.0

Major road *Major river* *Town*

Inset A: Petermann Ranges earthquake sequence 2016, Northern Territory. The largest earthquake in this sequence was a magnitude 6.1 event on 21 May. **Inset B:** Fault scarp created by the M6.1 Petermann Ranges earthquake 2016. **Inset C:** Offshore Bowen earthquake sequence 2016. The largest earthquake in this sequence was a magnitude 5.8 event on 18 August. **Inset D:** Norseman earthquake sequence 2016. There were three events with magnitudes larger than or equal to M5. A M5.0 and a M5.1 event occurred on 28 May less than two hours apart, followed by the main M5.6 earthquake of the sequence on 8 July 2016.

This poster shows earthquakes occurring in Australia in 2016 with a background of Australian earthquakes over the past 10 years. Also included are images produced as part of the analysis of the Petermann Ranges earthquakes in Northern Territory, the Norseman earthquake sequence in Western Australia, the offshore Bowen earthquakes in Queensland, as well as the yearly summary of earthquake occurrences in Australia in 2016.

For additional information the Australian Seismological Report 2016 provides a summary of earthquake activity for Australia for 2016 including dedicated state and territory earthquake information. It also provides summaries of earthquakes of magnitude 5+ in the Australian region and of magnitude 6+ earthquakes worldwide for 2016.

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Copies of this map may be downloaded from the Geoscience Australia internet site at:
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Map, graph and table data sourced from Geoscience Australia's earthquake database
http://www.ga.gov.au/earthquakes/searchQuake.do

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THE NEW SCHOOL

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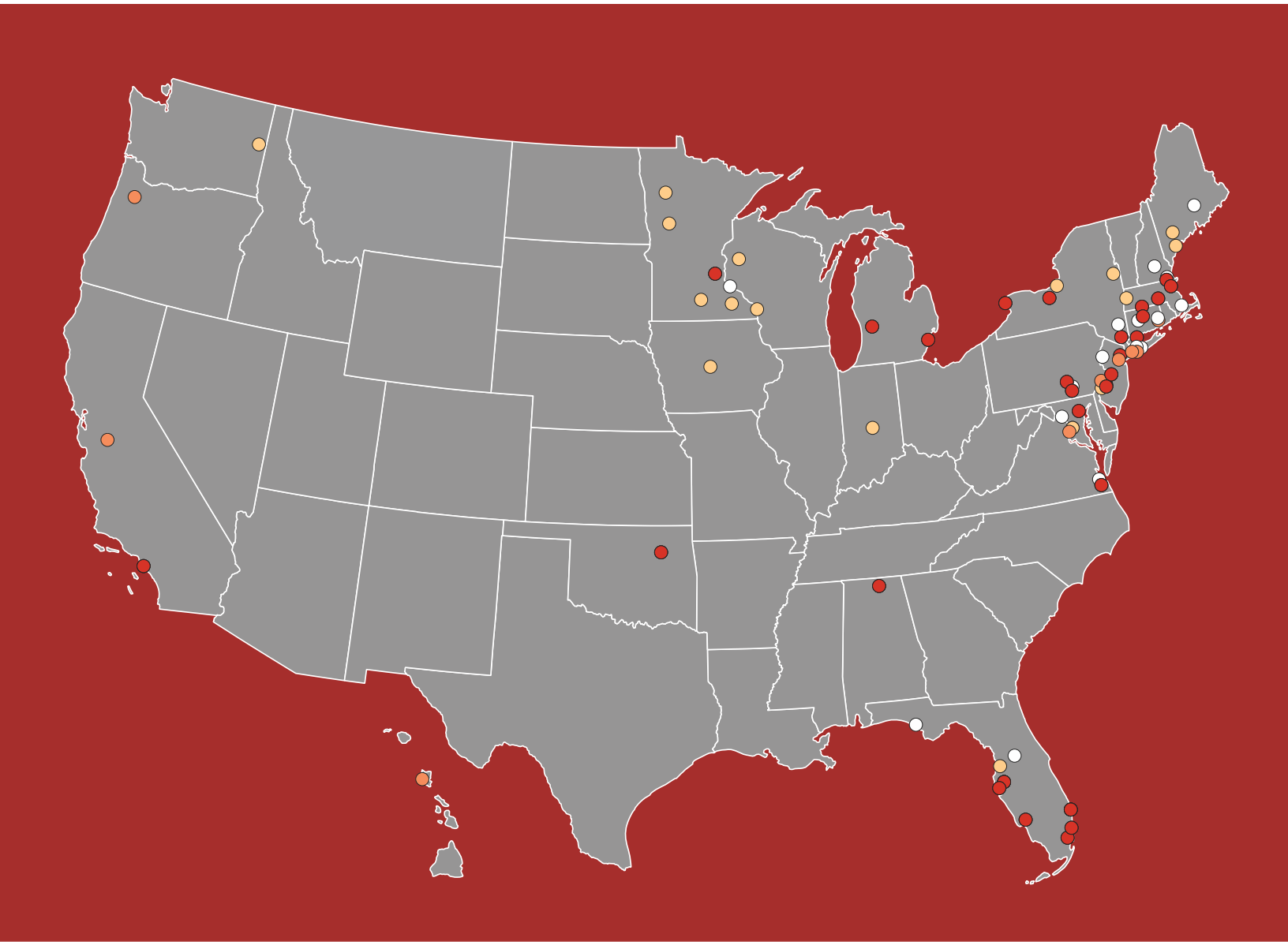

U.S. Municipal Solid Waste Incinerators: An Industry in Decline

May 2019

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About the Tishman Environment and Design Center

The Tishman Environment and Design Center at The New School fosters the integration of bold design, policy, and social justice approaches to environmental issues to advance just and sustainable outcomes in collaboration with communities.
Tishmancenter.org

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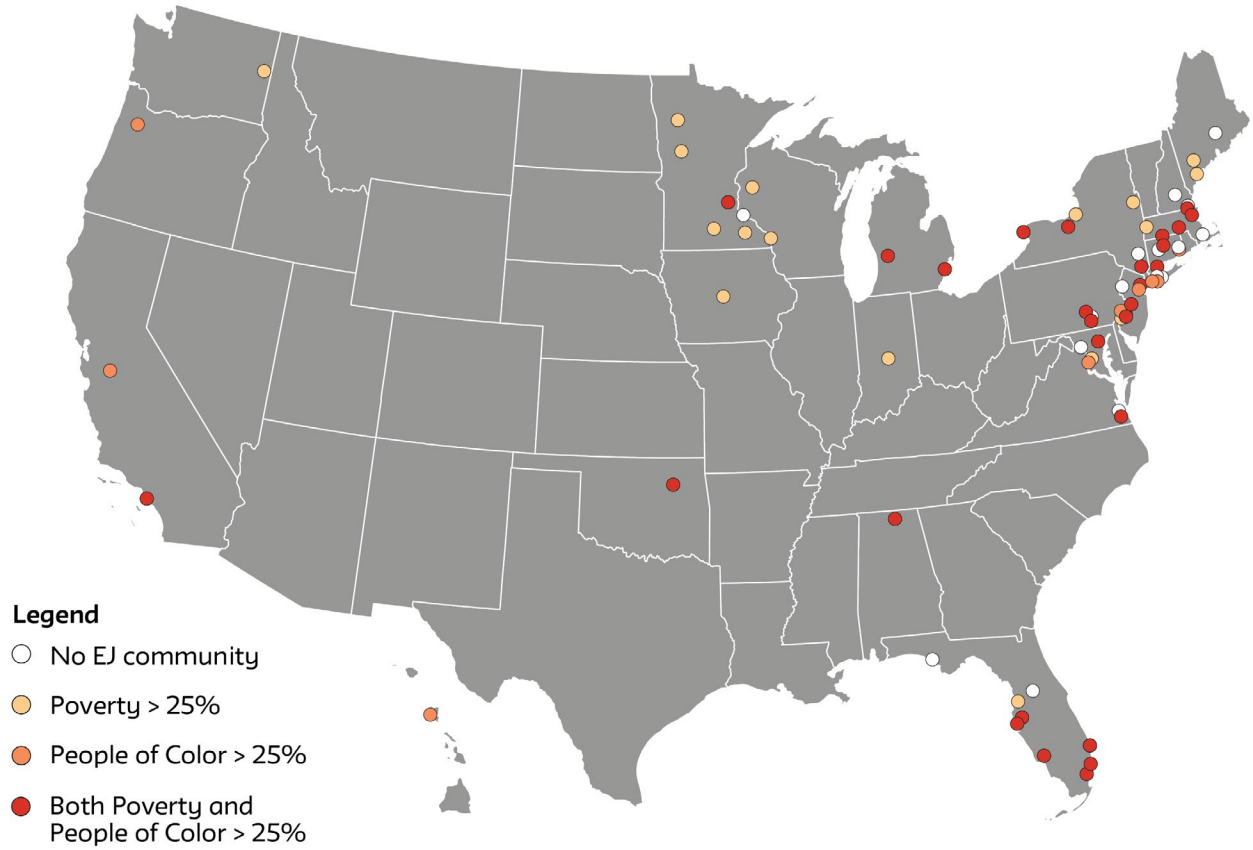
Executive Summary

Municipal Solid Waste (MSW) incinerators have a long history in the United States as a waste disposal system and an equally long history of resistance among communities where they are sited. The current state of MSW incineration seems to be in decline due to a volatile revenue model, aging and costly operation and maintenance costs, and increasing attention to issues of zero waste, environmental justice and climate change. Seventy-three MSW incinerators remain in operation in the U.S., not including those currently designated for closure.[1] The industry saw at least 31 MSW incinerators close since 2000 due to issues such as insufficient revenue or the inability to afford required upgrades.[2]

This report examines three major economic vulnerabilities in the MSW incinerator industry. First, construction and maintenance costs are significant and relatively more capital intensive compared to other forms of waste disposal. Second, the current pool of MSW incinerators have reached or are close to reaching their life-expectancy and now require another round of capital investment, often at the expense and risk of local taxpayers. Third, the industry's revenue streams are volatile, dependent on competitive tipping fees and access to the renewable energy markets. Additionally, the report reveals the relationship between MSW incinerators and environmental justice communities as well as the air pollution and potential health risks related to the incineration industry.

One of the distinct characteristics of garbage incinerators in the United States is that they are often sited in communities of color and low-income communities, also referred to as environmental justice (EJ) communities. **58 incinerators, or 79 percent of all MSW incinerators in the U.S. are located in environmental justice communities.**[3] The incineration industry represents an affront to environmental justice as they contribute to the cumulative and disproportionate pollution placed on communities of color and low-income communities.

MSW Incinerators & Environmental Justice Communities

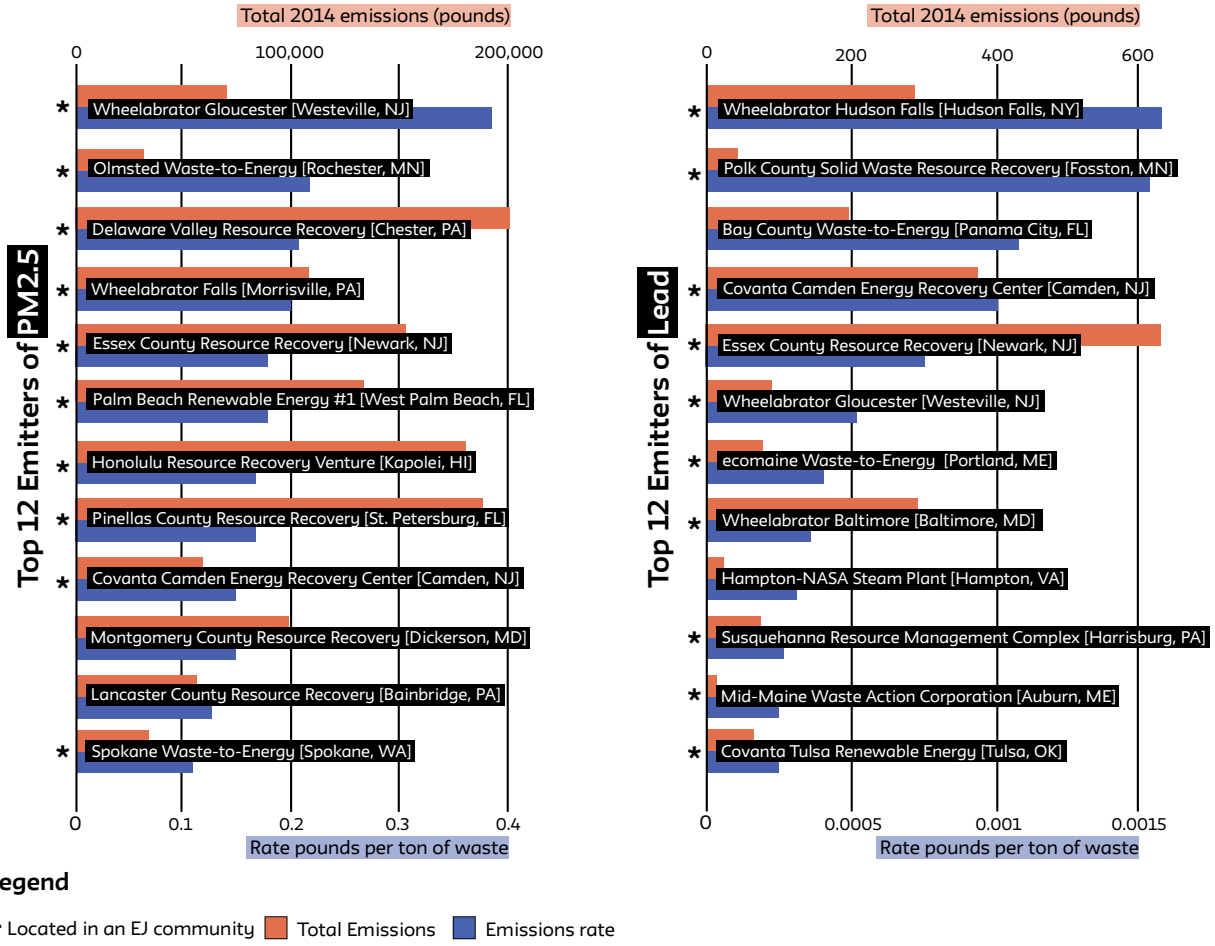


Municipal solid waste incinerators rely primarily on revenue streams from tipping fees and secondarily on energy sales (i.e. steam and electricity). As an example, Covanta Corporation, which controls a large share of MSW incinerators in the country, gets approximately 71 percent of its revenues from tipping fees and 18 percent from electricity sales.[4] These two revenue streams are volatile and can undermine the financial stability of the industry. There is close competition for tipping fees between landfills and incinerators, which means that in places where landfill tipping fees decline or where volumes of waste decrease, an incinerator's primary revenue source can be jeopardized.

Many municipalities are also removing long term "put or pay" clauses from contracts so that they are not required to deliver a set amount of waste to incinerators over time with a threat of financial penalties. Similarly, renewable energy subsidies can change over time, depending on the regulatory and political environment in each state. This leads to an underlying business model at risk, "*As our historic energy contracts have expired and our service fee contracts have transitioned to tip fee contracts, our exposure to market energy prices has increased.*" (Covanta Annual Report, 2018)[5] Another factor that contributes to this industry's potential decline is the average age of incinerators in the U.S., which is 31 years.[6] The life expectancy of an incinerator is 30 years[7] and upgrading decades-old facilities requires another large capital investment, often paid for or subsidized by local taxpayers. Municipalities that finance these upgrades or that are required to deliver large volumes of waste often end up burdening taxpayers, sometimes with ruinous outcomes. Cities like Baltimore, Maryland;[8] Harrisburg, Pennsylvania;[9] and Detroit, Michigan,[10] all faced debt payments to and lawsuits from the incinerator industry that threatened their cities' fiscal stability.

The increasing fixed costs of maintaining and operating incinerators together with competition for tipping fees can mean that the industry relies on energy sales to stay profitable. But burning trash is one of the most expensive forms of energy generation in the U.S., costing $8.33/MWh compared to $4.25/MWh for pulverized coal and $2.04/MWh for nuclear, the second and third most expensive forms of energy generation.[11] Despite these costs and the fact that MSW incinerators produced a negligible 0.4 percent of total U.S. electricity generation (2015), two-thirds of all the incinerators in the U.S. today have access to renewable energy subsidies.[12] These energy subsidies are coming under increased scrutiny as environmental advocates question the classification of waste burning, particularly non-biogenic waste, as renewable energy. The introduction of new carbon pricing policies in states like New York may mean that incinerators, which emit significant amounts of CO2, will face new financial challenges.

One of the primary reasons that communities oppose new and existing incinerators is their contribution to air pollution and related health risks. MSW incinerators are relatively large emitters of air pollutants with some studies showing that they emit several pollutants at a rate exceeding that of fossil fuel power plants.[13] Incinerators also have associated diesel sanitation trucks that deliver waste and emit air pollution in host communities. Stack emissions from incinerators include a variety of pollutants harmful to health such as particulate matter, dioxins, lead, and mercury. Globally, waste disposal, primarily from incineration, contributes to ~8 percent of the total anthropogenic mercury emissions.[14] The Dirty Dozen lists illustrate the incinerators, among the 73 in the country, that emit the largest amounts of air pollutants for PM2.5, NOx, Lead, and Mercury. Approximately 1.6 million people live within a three-mile radius of these facilities (See Appendix E).[15] There are 4.4 million people that live within a three mile radius of all 73 incinerators in the U.S. **Ten of the twelve incinerators that emit the greatest total amount of lead emissions (annually), are in environmental justice communities.** Three of the incinerators that emit the largest total amounts of lead (annually) of all the incinerators in the U.S. are located in Baltimore, Maryland, and in Camden and Newark, New Jersey.



The incinerator industry is in trouble. These aging facilities are too expensive to maintain, too risky to finance, and too costly to upgrade. Incinerators in the U.S. operate under volatile economic and regulatory conditions that threaten their major sources of revenue, tipping fees and energy sales. The current state of the U.S. incineration industry and its economic and environmental impacts serves as a warning to regions around the world considering incineration as an approach to solid waste. These facilities can create financial burdens while generating health-harming air pollution for local communities. Finally, these plants represent an environmental injustice because they burden communities of color and low-income communities where they are located. Incinerators are coming under increasing pressure in the United States and around the world to be replaced with more just and sustainable alternatives to waste management.

Chapter 1: HISTORY OF THE INCINERATION INDUSTRY

Municipal solid waste (MSW) incinerators have a long and troubled history as a waste management strategy dating back more than a century in the United States. These facilities have taken many forms over the years and have faced an equally long history of resistance among communities where they are sited. While the trajectory of the industry has waxed and waned in the last 50 years, the current state of MSW incineration seems to be in decline.

There are currently 73 MSW incinerators, not including those currently designated for closure.[16] Collectively, these 73 incinerators burn about 13 percent of all MSW produced in the United States and have an annual revenue estimated to be $3.2 billion.[17] Despite these profits, the industry saw at least 31 MSW incinerators close since 2000.[18] Closures are largely due to insufficient revenue and inability to afford required upgrades.

Most incinerators were built in the 1980s and have exceeded their life expectancy of 30 years.[19] The age of these facilities is a prime contributor to the industry's overall decline and a factor in the various equipment issues and shutdowns that have taken place over the last decade. The industry has also sought to generate additional revenue streams through federal and state classification as a "renewable energy source," hence the shift in branding incinerators from "refuse facilities" to "waste to energy" (WTE) plants.

Table 1: Age of MSW Incinerators

Year of Construction	Number of Facilities
1970-1979	3
1980-1989	45
1990-1999	24
2000s	1

While garbage incineration as "waste-to-energy" has been sold to governments and the public as a technologically-advanced approach to handling solid waste, with the bonus of producing energy, relatively little energy is actually derived from these plants.[20] Combined, these facilities produced approximately 0.4 percent of total electricity generation in the U.S. in 2015.[21] In fact, MSW incinerators are expensive to operate and produce criteria air pollutants like particulate matter as well as relatively more greenhouse gas emissions than coal-fired power plants.[22] Approximately 25 percent of the trash incinerated at MSW plants also remains as toxic ash requiring landfill disposal.[23] Emissions from incinerators include hazardous air pollutants like mercury, lead, and dioxins.[24] The air pollution and associated public health impacts will be further explored in Chapter 3 of the report and implications around energy production will be discussed in Chapter 2.

The history of garbage incineration in the U.S. dates back more than a century. The first garbage incinerator was introduced in the U.S. in 1885 to dispose of waste from an army post on Governor's Island in New York.[25] That same year, the first municipal solid waste incinerator was built in Allegheny, Pennsylvania.[26] From 1885 to 1908, an estimated 180 waste incinerators were constructed across the United States.[27] These early incinerators were mass burn plants using specialty furnaces developed by European manufacturers.[28] In densely populated areas like New York City, incinerators were popular due to the lack of cheap land nearby to develop and expand large landfills. But the cost of building and operating an incinerator was also expensive relative to landfills.[29] It has been estimated that by the late 1930s, the United States had more than 700 garbage incinerators.[30] In the 1960s, New York City had 22 municipal incinerators and thousands of incinerators in apartment buildings, burning nearly one-third of all of the city's trash.[31] While use of incinerators continued to grow in the first half of the 20th century, landfilling remained a relatively cheaper and more commonly used option throughout the country.[32]

Consumption, Waste Management and the Growth of the Incineration Industry: 1970s – 2000s

> *"The U.S. produces more than 30 percent of the planet's total waste, though it is home to only 4 percent of the world's population."*[33]

During the second half of the 20th century, numerous factors impacted how municipal solid waste was produced, managed and disposed.[34] One of the most significant factors driving this was Americans' growing appetite for consumption fueled in part by increased marketing to stimulate consumer habits after World War II.[35] This increased consumption also produced immense amounts of waste. There is a correlation between increased wealth and waste generation. Richer countries are far likelier to produce more waste per capita than poorer countries.[36]

Production of garbage rose steadily since the 1960s. The growth in consumption and production of plastics was particularly harmful to public health. Figure 1 shows total MSW generation and per-capita generation over the past 60 years. In 1960, Americans produced 2.68 lbs/person/day of waste, a total of 88 million tons. [37] By 2015, that increased to 4.48 lbs/person/day and a total of 262.4 million tons of

waste.[38] The amount of plastics in the waste stream in 1960 was negligible.[39] But by 2015, plastics made up about 13.1 percent,[40] or 34.5 million tons of the waste stream.[41] As Figure 1 illustrates, total MSW generation grew 199 percent from 1960 to 2015.

Prior to the introduction of plastics, American waste was primarily composed of organic or biogenic materials. The introduction of plastics in the consumer marketplace in the 20th century, while heralded as an important innovation also introduced new public health and environmental concerns. The properties, which popularized plastics, its versatile and durable qualities, also made disposal difficult.[42] Most plastic products produced since the 1950s have not been recycled but have been landfilled, incinerated, or remain as pollution in oceans and waterways.[43] In fact, the U.S. only recycles 9.1 percent of plastic waste, less than the 15.5 percent that is incinerated and some studies estimate plastics recycling as low as 2%[44] after accounting for plastics exportation that is counted as recycled.[45]

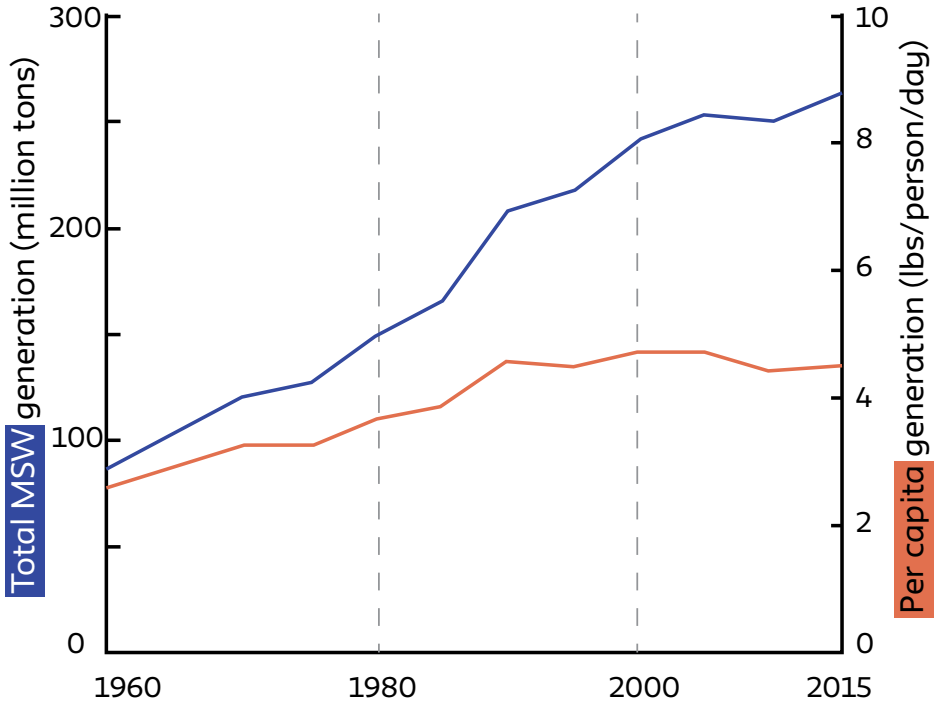
Studies have shown that recycling plastic saves more energy than combustion.[46] Unfortunately, the recent boom in hydraulic fracturing has aided the growth of the plastics industry as a surge of natural gas supplies makes plastic production cheaper.[47] Figure 3 shows the type of waste generated in the U.S. in 2015 by material. Much of this waste, about 90 percent, could be reused, recycled, or composted instead of landfilled or burned.[48] As shown in Figure 2, the U.S. landfills 52 percent of the MSW generated; incinerates ("Combustion with Energy Recovery" in the Figure) 13 percent of MSW; recycles 26 percent; and composts 9 percent.

The growth in household waste and the increasing composition of non-biogenic waste directly impacts incinerator emissions. As MSW incinerators burn more materials containing toxic chemicals, the subsequent emissions will also include more hazardous air pollution. In vulnerable communities, where the U.S. incineration industry is mostly located, burning waste products with toxic compounds impacts the health and well-being of people in these overburdened areas. Ironically, these low-wealth areas that host incinerators tend to contribute the least to the problem because these households consume less on average than wealthier households.[49]

Federal Oversight of the Incineration Industry

Federal oversight and regulation of the incineration industry has evolved over time through diverse air, energy, and solid waste related policies. Figure 5 details this history of federal laws, legal decisions and regulations pertaining to the incineration industry.

Figure 1: MSW generation rates: 1960 - 2015



Source: U.S. EPA, Advancing Sustainable Materials

Figure 2: U.S. MSW Waste Disposal Methods (2015)

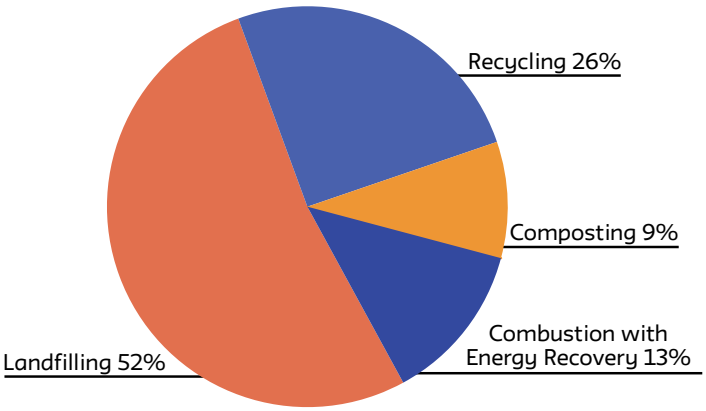
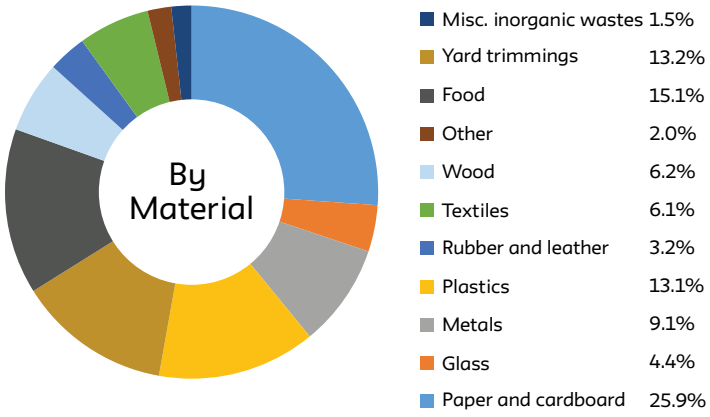


Figure 3: U.S. MSW Waste By Material (2015)



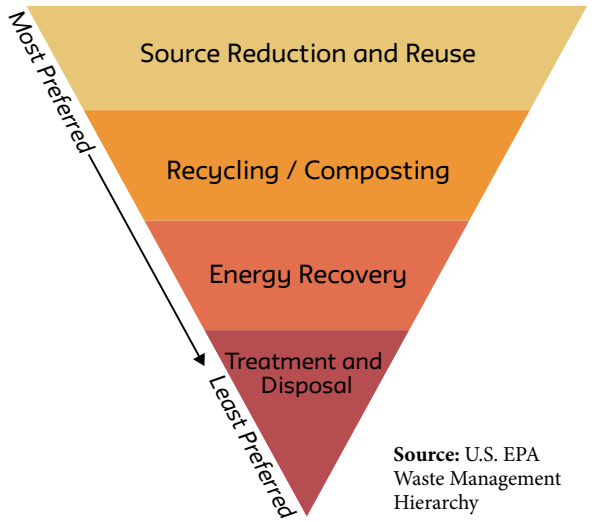
Source: U.S. EPA National Overview Facts and Figures on Materials, Waste and Recycling 2015.

In 1970, the **United States Environmental Protection Agency (U.S. EPA)** was established as a new federal agency and the federal **Clean Air Act (CAA)** was enacted. Under the CAA, the EPA banned uncontrolled burning of MSW and placed restrictions on particulate matter.[50] The law led to the closure of many of the older incinerators because they lacked required emissions controls, which were expensive to retrofit. The share of municipal solid waste being processed by incinerators declined from 31 percent in 1960 (these were primarily incinerators without energy recovery) to 9 percent in 1980.[51] Between World War II and 1979, the number of incinerators plummeted from 300 to 67.[52]

In the early 1970s, as the U.S. EPA expanded research and guidance on waste management in the United States, it became clear that the Solid Waste Disposal Act of 1965 was not sufficient to protect human and environmental health.[53] In 1976, the federal government enacted the **Resource Conservation and Recovery Act (RCRA)** that is still the defining law regulating solid waste today.[54] RCRA gave the EPA authority to regulate and create policies for managing solid and hazardous waste. Landfills became more tightly regulated.[55] Many open dumps shut down across the country.[56] Between 1980 and 1986, the number of landfills went from 20,000 to 6,000.[57] These regulations made landfill maintenance more expensive and over time, helped consolidate waste management into a smaller handful of larger, well-financed private sector companies that could keep pace with costs.[58] Since the enactment of RCRA, state environmental agencies and county authorities were charged with implementing solid waste management laws and issuing solid waste permits.[59]

The U.S. EPA also created a Waste Management Hierarchy (Figure 4), which prioritized source reduction and reuse first, recycling and composting, and then incineration (energy recovery) and landfilling last. Many recyclable, compostable and largely biogenic materials are being burned in MSW incinerators instead of composted, recycled, reused or reduced as recommended by the U.S. EPA's Waste Management Hierarchy. One of the central critiques of relying on large incineration facilities is that they require high volumes and constant flows of waste to remain profitable. This need for running the facilities at their maximum capacity undermines more sustainable and preferable methods of preventing or diverting waste from burning or landfilling.

Figure 4: U.S. EPA Waste Management Hierarchy



Source: U.S. EPA Waste Management Hierarchy

Figure 5: Time line of Incinerator Industry Laws & Regulations

Other regulations and policies enacted in this time period impacted MSW incineration. The **Public Utility Regulatory Policies Act** (1978) allowed investor-owned utilities to purchase electricity from independent producers, including MSW incinerators, through power purchase agreements, up to a limit of 80 MW of electricity.[60] This gave incinerators another source of revenue. In 1988, the federal government stopped the dumping of industrial, medical, and sewage waste into the ocean through the **Ocean Dumping Ban Act.** This narrowed the list of MSW disposal methods. In 1990, as part of new amendments to the Clean Air Act, officials implemented the **Maximum Achievable Control Technology standards (MACTs)** that limited pollution from MSW combustion plants.[61] These standards forced plants to achieve a similar level of emission control "already attained by an average of the best performing, top 12 percent, sources in each pollutant category."[62] While MACTs helped reduce criteria and hazardous air pollutants emitted from MSW combustion, there are still significant emissions that pose a risk to human and environmental health.[63]

In the 1980's the MSW incinerator industry saw a resurgence in new facilities. The closure of thousands of landfills was due to the introduction of RCRA rules, the energy crisis in the 1970's, and the industry's efforts to rebrand itself as an energy source. As shown in Figure 6, the proportion of MSW being combusted with energy recovery systems grew during the 1980s and 1990s along with the shift to branding incinerators as 'waste-to-energy' plants.

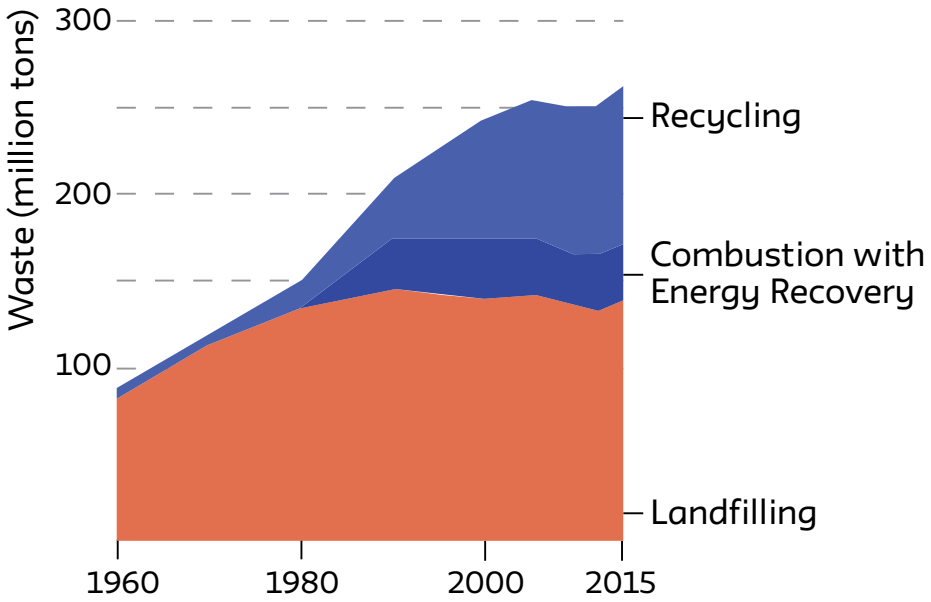
In the late 1990s, cities largely stopped building MSW incinerators. Communities targeted for hosting incinerators fiercely opposed the siting and municipalities were increasingly wary of the large capital costs to build and maintain these facilities. Dioxins and mercury research in the early 1990s helped to inform the opposition to the incineration industry as public concern grew over the link between cancer and dioxins.[64] In the 1990s, incinerators were found to contribute to the growth of mercury pollution in the atmosphere, while at the same time, the U.S. began to ban products with mercury due to health risks.[65]

Privatization and Deregulation of Waste

During the 1980s and 1990s, multinational corporations were able to consolidate their control of the municipal solid waste system as a valuable commodity.[66] As new regulations required more capital and technological capacity to manage larger quantities of waste, the industry began to regionalize in order to achieve economies of scale. Private corporations began to enter this market to create regional systems for waste transfer, processing and disposal.[67] By 2000, four waste management corporations across the entire waste disposal sector (including Waste Management and Allied Waste) controlled 85 percent of the total waste industry revenues.[68]

Three key court decisions also significantly impacted the business model for MSW incineration. The deci-

Figure 6: Municipal Solid Waste Management: 1960-2015



Source: U.S. EPA National Overview: Facts and Figures on Materials, Wastes and Recycling

sions in C&A Carbone Inc. v. Town of Clarkstown, and Oregon Waste Systems, Inc. v. Department of Environmental Quality of the State of Oregon, defined waste and disposal capacity as commodities and limited governments' ability to control the movement of waste within their jurisdictions.[69] In 1994, in the C&A Carbone, Inc. v. Town of Clarkstown decision, the court found that "flow control ordinances" violated the Interstate Commerce Clause.[70] The town of Clarkstown signed a contract with a waste processing plant promising at least 120,000 tons of waste per year. In order to meet their contract, the town passed a flow control ordinance mandating all city waste be processed at this designated plant. This provided a guaranteed revenue stream to the waste processing company.[71] Such contracts, called "flow control ordinances" were commonplace, and many incinerators entered into these contracts with municipal governments. In the Carbone case, the Supreme Court found these mandates or "flow control" ordinances unconstitutional and defined waste as a commodity that should not be restricted for the benefit of some competitors.[72] After this decision, two city-owned incinerators in Ohio, unsure of their ability to meet financial obligations absent the flow control ordinances, were closed.[73]

The second decision, Oregon Waste Systems, Inc v. Department of Environmental Quality of the State of Oregon, found that surcharges on out-of-state trash being disposed of at in-state facilities, violated the Inter-state Commerce Clause.[74] The State of Oregon argued that the surcharge was fairly used so as to make out-of-state waste producers pay the same amount for waste disposal as in-state producers.[75] But by striking down these surcharges, incinerators benefited because they could receive out-of-state trash without additional fees that would make their facilities less competitive in the waste disposal marketplace and they could better ensure enough waste flow to their facilities to be profitable.

In 2007, the Supreme Court returned to the question of flow control ordinances in United Haulers Association v. Oneida-Herkimer Solid Waste Management Authority. Waste haulers and a trade association sued the Oneida-Herkimer Solid Waste Management Authority over a flow control ordinance requiring them to deliver trash to the city-owned facility.[76] The flow control ordinance directed waste haulers from Oneida and Herkimer counties to dispose exclusively at facilities under the agency's control. In a 6-3 decision, the Supreme Court ruled in favor of the Oneida flow control ordinance. The Carbone decision previously ruled that flow control ordinances were unconstitutional; however, this Oneida decision found such ordinances constitutional as long as the waste disposal facility was owned by a public agency.[77] In Carbone, the case centered on flow control that benefited privately-owned disposal facilities. The Oneida case made a distinction in the use of flow control based on the rationale that public agencies have different objectives from privately controlled facilities, one serves a public purpose and the other threatens competition among private entities.[78]

Incineration and Environmental Justice Communities

The association of communities of color and low-income communities with waste dumps has a long history of resistance in the environmental justice movement.[79] Since publication of the seminal study, "Toxic Waste and Race in the United States," in 1987, studies have continued to show that race is the most significant predictor of living near a toxic facility along with income.[80] In 1984, the Cerrell Report, commissioned by the California Waste Management Board, stated that "*All socioeconomic groupings tend to resent the nearby siting of major facilities, but middle and upper socioeconomic strata possess better resources to effectuate their opposition.*"[81] The results of this report confirmed the suspicions of environmental justice communities that charged the waste industry of targeting low-income and communities of color for facility siting. "The Cerrell Report fit us to a T', says Mary Lou Mares, one of the leaders of El Pueblo."[82] One of the distinct characteristics of garbage incinerators in the United States is that they are often sited in communities of color and low-income communities, also known as environmental justice (EJ) communities. The stigma and pollution burdens from the association of waste with EJ communities has become a central point of organizing opposition to incinerators.[83]

The siting of incinerators and other polluting facilities in environmental justice communities is not a coincidence but rather it is a product of historic residential, racial segregation and expulsive zoning laws[84] that allowed whiter, wealthier communities to exclude industrial uses and people of color from their boundaries.[85] While suburbs zoned primarily for single family, residential developments, cities

retained and hardened industrial zoning - effectively depressing land values in areas where people of color and low-income people were pushed to reside.[86] Over time, the effect of structural and institutional racism in the U.S. that relegated people of color to marginal lands, close to industry and pollution, continues to be seen today in the patterns of disproportionate siting of incinerators.

There are many reasons why the co-location of communities of color and low-income communities and incinerators is worrisome. These communities face underlying social vulnerabilities due to their socio-demographic status and they are often, already overburdened with disproportionate amounts of pollution from a multitude of sources. Incinerators pose potential health risks for any host community, but these risks are particularly pernicious when one considers the fact that a majority of plants are located in environmental justice communities that are contributing the least to the waste problem and yet are asked to bear the brunt of the larger society's consumptive, throw away lifestyles.[87] Furthermore, the racialized nature of land use patterns means that incinerators are exacerbating environmental racism. This makes incinerators particularly problematic in the U.S. context. In addition to incinerators' implication in perpetuating environmental racism there are a variety of reasons why incineration is considered a "false solution" on the part of environmental justice and environmental advocates across the country. These groups cite the following concerns with incinerators:

- Health impacts from air pollution associated with stack emissions and diesel trucks transporting waste. Exacerbation of underlying health problems such as childhood asthma & cardiac disease.
- Public debt related to financing the construction & maintenance of the incinerator can drain local taxpayers.
- The creation of waste processing hot spots. One facility is located in the area, it can create a precedent for concentrating other waste-related facilities nearby due to depressed land values.
- The stigma of being a dumping ground for waste from wealthier, often whiter communities.
- Decrease in recycling, composting, and waste reduction due to perverse incentives to burn more waste.
- Decrease in property values and commercial businesses because of stigma and nuisance issues.
- Exacerbation of cumulative impacts from multiple sources of pollution.

One of the critical reasons why incinerators are particularly problematic in environmental justice communities is because of their contribution to the cumulative impacts of pollution in these areas. The effect of multiple pollutants from many sources and their interaction with underlying socio-demographic vulnerabilities in overburdened communities' results in what is often termed "cumulative impacts." "Cumulative impacts" is a framework for thinking about and assessing the vulnerability of communities considering both environmental and socio-demographic factors. The California Environmental Protection Agency (CALEPA) defines the term as:

> *Cumulative impacts means exposures, public health or environmental effects from the combined emissions and discharges, in a geographic area, including environmental pollution from all sources, whether single or multi-media, routinely, accidentally, or otherwise released. Impacts will take into account sensitive populations and socio-economic factors, where applicable, and to the extent data are available.*[88]

Though the federal government does not have an official designation for "environmental justice" communities, a number of states and municipalities have working definitions based on race and income thresholds. These thresholds range from relative measures compared to state averages or absolute percentages of racial and income categories within census tracts or block groups. Based on a review of these existing definitions and national averages,[89] the threshold chosen for this national study falls within the range of percentage thresholds used by other states or policies (i.e. Massachusetts, New York).[90] In order to examine the co-location of MSW incinerators and environmental justice communities, the percent of people who identify as "minority" (according the U.S. census definitions[91]) and the percent of people that are below the federal poverty level in the census tracts within a three-mile radius of the plants was compiled from the U.S. EPA's Enforcement and Compliance History Online (ECHO) database.

The definition selected is based on census tracts where: (a) the percentage of people living below the federal poverty rate is above 25 percent OR (b) the percentage of people identify as "minority" is above 25 percent. Some communities met both income and race thresholds. Most existing environmental justice definitions use either the race or income thresholds, but few require both conditions to determine if an area can be deemed an EJ community.[92] Figure 7 depicts the 73 MSW incinerators currently in operation in the U.S. and identifies the facilities located in environmental justice communities according to this definition. The figure shows:

- **58 incinerators, or 79 percent, are located in environmental justice communities.**[93]
- **31** incinerators, or **40 percent**, are in communities where both the thresholds for poverty AND the percentage of people of color is above 25 percent.
- **48** incinerators are in communities where more than **25 percent** of the population is below the federal poverty level (national poverty rate of 12 percent)[94]
- **44** incinerators are in communities where the population is at least **25 percent people of color.**

Source: Kim Hunter. Will Copeland speaking at Breathe Free Detroit Press Conference, May 18 2018.

Detroit Incinerator Closes Down

Renamed the Detroit Renewable Power (DRP) facility in the 1990s, this incinerator reflects many of the industry trends across the market with respect to its declining performance, fiscal troubles and its failed efforts to rebrand itself as an energy facility. The scale of the fiscal burden that the facility imposed on local tax payers was immense – beginning with a $478 million construction bond in the 1980s and then an additional $179 million bond in the 1990s. Ultimately Detroit paid out over $1 billion to operate a facility that polluted the community. The facility was the source of sustained and intense community-led opposition from the time it was proposed until the present day. Groups such as Breathe Free Detroit and Zero Waste Detroit rallied residents to oppose the public financing and public health burdens that the facility imposed on surrounding EJ communities. These groups cited the persistent odor and air pollution violations that emanated from the plant as the drivers for the push to permanently close the facility.

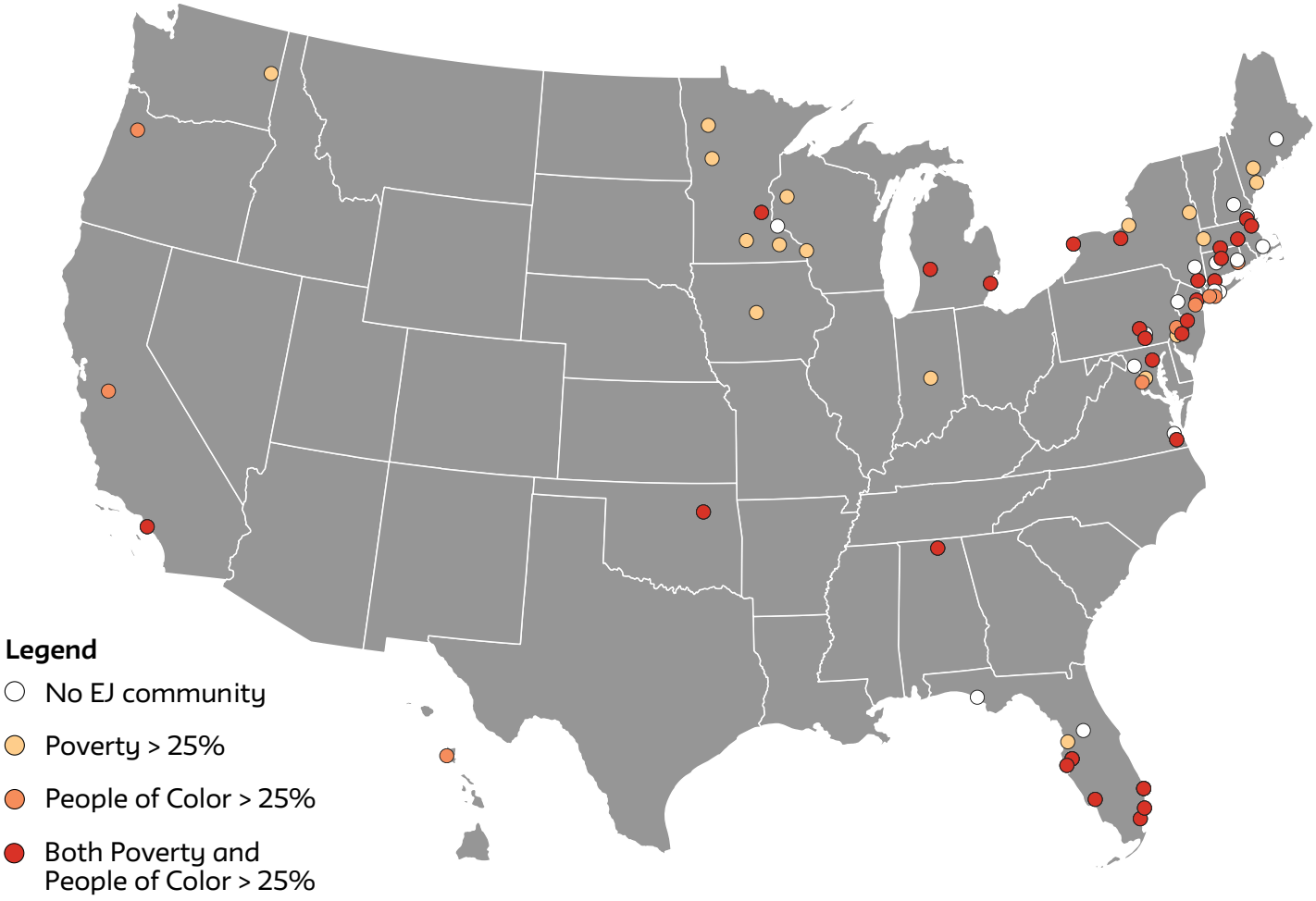
In January 2019, the Great Lakes Environmental Law Center (on behalf of Ecology Center and Environment Michigan) issued a 60-day Notice of Intent to Sue the Detroit Renewable Power Incinerator for violating the Clean Air Act over 600 times in the past 5 years. The threat of this citizen suit, which would likely have required DRP to invest tens of millions of dollars to come into compliance, was a critical factor in the incinerator's closure, which was announced just days before the groups would have actually filed the lawsuit in Federal court. Local organizers celebrated the closure of the Detroit incinerator as a community victory that illustrates the power of long- term, grassroots environmental justice organizing. "We celebrate the closure of one of the world's largest incinerators, a facility that has been a bad neighbor for over 30 years, unable to comply with Clean Air laws and odor restrictions." (Breathe Free Detroit!)

It is important to note that several of the largest and relatively most polluting incinerators (*incinerators reporting high total annual emissions for NOx, PM, Lead, or Mercury relative to all 73 MSW incinerators, please see Appendix E for more detail*) in the U.S. are in census tracts, within 3 miles, that are predominantly low-income or people of color communities. These communities include:

- Honolulu Resource Recovery Venture (Kapolei, Hawaii) has census tracts within a 3-mile radius with a population that is 81 percent minority and 13 percent below the federal poverty rate
- Essex County Resource Recovery (Newark, New Jersey) has census tracts within a 3-mile radius with a population that is 71 percent minority and 37 percent below the poverty rate
- Palm Beach Renewable Energy Facility #1 (West Palm Beach, Florida) has census tracts within a 3-mile radius with a population that is 56 percent minority and 34 percent below the federal poverty rate
- Wheelabrator Baltimore (Baltimore, Maryland) has census tracts within a 3-mile radius with a population that is 66 percent minority and 50 percent below the federal poverty rate

For more detailed information on where incinerators are located in relation to environmental justice communities, refer to Appendix A. Most of the existing incinerators in the U.S. are located in environmental justice communities that are disproportionately impacted by other polluting facilities. Many environmental justice organizations are actively involved in the advancement of alternatives to incineration which can provide economic and environmental benefits to their communities. In the next section, the economic vulnerabilities of the industry will be explored in detail.

Figure 7: MSW Incinerators and Environmental Justice Communities



Source: Global Alliance for Incinerator Alternatives (GAIA)

Environmental Justice Communities Advance Zero Waste & Just Transition Solutions

Environmental justice communities that host incinerators are not only opposing existing facilities, they are leading the way on alternative solutions to waste disposal. EJ organizers are proposing practical pathways toward phasing out incinerators and establishing zero waste systems. The zero waste goals proposed by EJ organizations include advocating for policies such as pay-as-you-throw, financial incentives for waste reduction, recycling, and composting, mandates for worker safety, and ensuring democratic participation of residents.

In Baltimore, resident activists are developing a Zero Waste Implementation Plan that phases out the Wheelabrator Incinerator and replaces it with alternative waste diversion industries like composting. The Plan defines the problem of incineration through a health, equity and racial justice lens and also details policy goals. EJ activists are building their network through the Fair Development Roundtable where they are advancing zero waste goals and community land trusts. The organizers will also support demonstration projects that highlight the deep commitment of residents to environmental sustainability by increasing composting and recycling as well as green space stewardship.[95]

EJ organizations are also deploying "Just Transition" principles in their efforts to move away from incineration towards zero waste goals. Just Transition refers to a set of principles, processes and practices of shifting economic and political power from an extractive economy toward, "a low-carbon and climate-resilient economy that maximizes the benefits of climate action while minimizing hardships for workers and their communities."[96] At the core of this approach is the fair treatment of workers in the transition, so that those that have been most negatively impacted by polluting practices in the past, directly benefit from future economic opportunities. In Detroit, local groups, including Breathe Free Detroit, sought protections for workers and residents as part of their campaign to shut down the Detroit incinerator.[97] They engaged with the city to hold it accountable for worker protection and raised funds for former employees.

Gentrification is another potential threat to local residents once an incinerator closes. EJ groups are raising awareness of the potential adverse impacts of the decommissioning process for shuttered incinerators and advancing Community Benefits Agreements (CBAs) to ensure that future development does not displace local residents. In Commerce, California, East Yard Communities for Environmental Justice is closely monitoring the decommissioning process after successfully advocating for the closure of the Commerce Refuse to Energy Facility.[98]

Chapter 2:
ECONOMIC INDICATORS OF DECLINE IN THE INCINERATOR INDUSTRY

The municipal waste incineration industry has profited by branding itself as a sustainable waste management and renewable energy industry. However, the industry relies on a risky business model that is costly to run and maintain as it ages, produces air pollution and toxic ash, and is dependent on public taxpayer dollars, which is ultimately not sustainable. The incineration industry in the United States is estimated to earn about $3 billion annually in gross revenue[99] and is expected to reach $4 billion in 2019.[100] Despite these profits, the industry faces serious economic challenges.

Two companies, Covanta and Wheelabrator, dominate the industry with 54 of the 73 "waste to energy" facilities under their control.[101] Incinerators are expensive to operate and maintain and "*the industry's performance is highly dependent on […] local and state government investment.*"[102] The incinerator industry relies on competitive tipping fee revenues and energy sales for a large proportion of its revenues. In order to secure funds from the sale of energy, the industry lobbies policymakers to secure access to tax credits, subsidies, power purchasing agreements, net metering, renewable energy credits and loan assistance through classification as a "renewable energy" source.[103] Even with these government supports; the industry still struggles to meet annual revenue demands. The vast majority of closures which took place over the past decade were due to economic losses.[104]

Figure 8 illustrates the financial structure of a typical MSW incinerator, showing capital investment sources, fixed and variable costs, and sources of revenue. Industry vulnerabilities are present in each of the quadrants depicted in the schematic. Incineration companies typically secure financing for the large capital costs of construction by securing publicly issued bonds or private loans. Wall Street firms have capitalized on this industry in which they profit from fees involved in structuring bonds that provide capital to build MSW incinerators. Between 1982 and 1989, Wall Street "floated $13.5 billion in bonds to build garbage incinerators and investment bankers earned nearly $200 million in fees."[105]

To get this financing, incinerator firms typically have to show evidence of economic viability by securing large, long-term sanitation contracts from county and municipal governments or other large institutions that can guarantee constant volumes of waste. Facilities built since the 1980s are relatively larger in size in order to guarantee enough volume of waste to be profitable. Incinerator revenues are derived largely from tipping fees; thus, these sanitation contracts are critical to their profitability.[106]

Despite rebranding themselves as energy companies, incinerators are primarily waste disposal companies. In addition to tipping fees, incinerators also sell steam and electricity as well as metal recovered from ash. The sale of energy from these plants has become another important stream of revenue as facilities capture more generous subsidies from the sale of electricity under the category of renewable energy. Energy sales account for approximately 20-30 percent of revenues and help cushion against decreases in tipping fees. As the 73 remaining incinerators age, the maintenance and upgrading costs also tend to increase and jeopardize a facility's profitability.

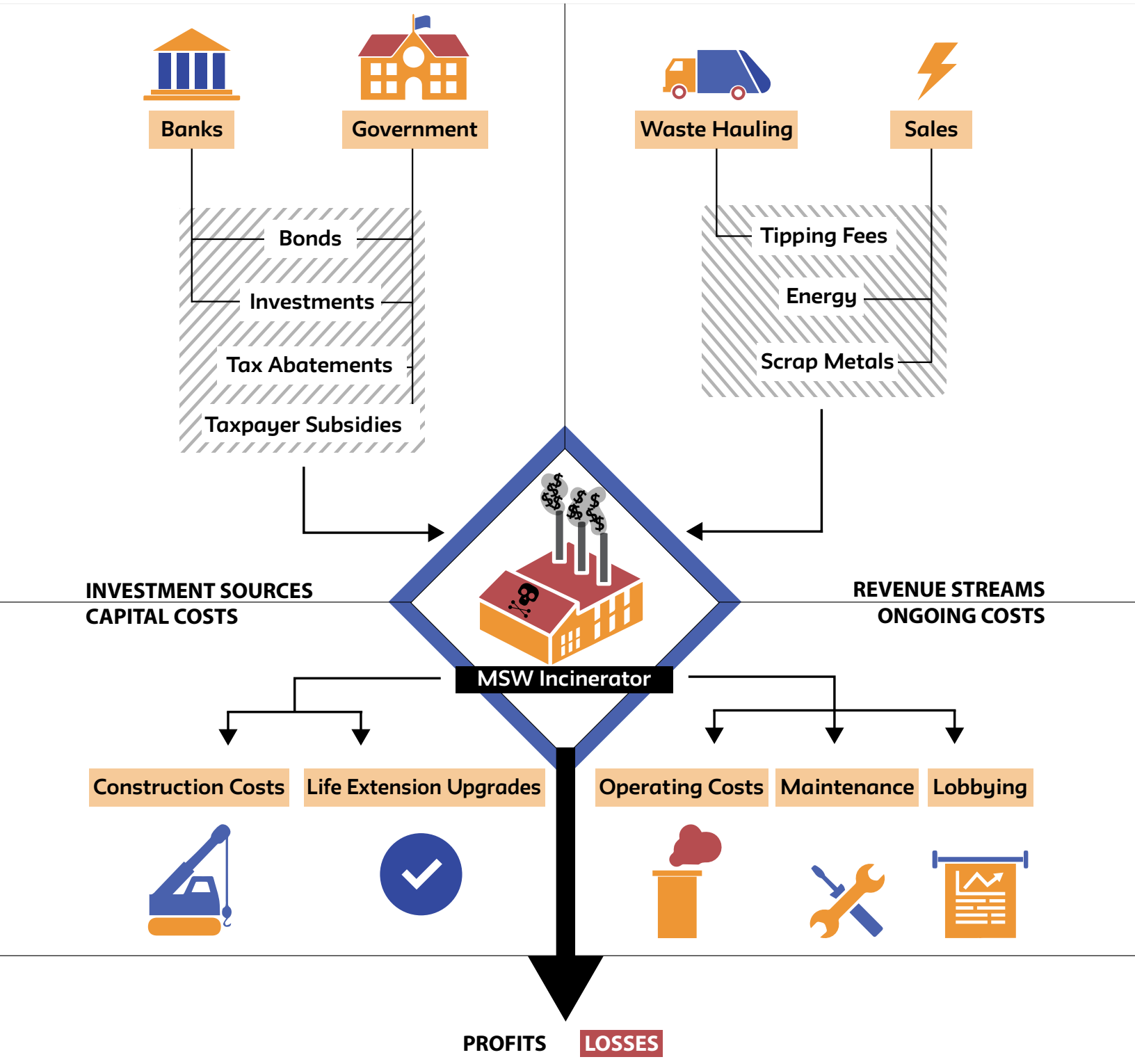
This report examines three major economic vulnerabilities in the MSW incinerator industry. First, construction and maintenance costs are significant and relatively more capital intensive compared to other forms of waste disposal. Second, the current pool of MSW incinerators have reached or are close to reaching their life-expectancy and now require another round of capital investment if they are going to continue operations, often at the expense and risk of local taxpayers. Third, the industry's revenue streams are volatile, dependent on competitive tipping fees and access to the renewable energy market.

Construction and Maintenance Costs

Incinerators are risky investments for cities[107], highly capital-intensive, and the most expensive form of garbage disposal. In order to raise the capital needed to build a new facility, companies often require assistance from government through various subsidies (companies typically qualify for some of these subsidies by being designated as 'electricity-generating' facilities) including access to low or no-cost municipal bonds.[108] Incinerator firms must first prove profitability to potential investors and local governments through executed service agreements with local governments, private waste haulers, and electricity purchasers.

According to the U.S. Environmental Protection Agency, construction of an average-sized incinerator can cost approximately $100 million.[109] However, construction costs often run well beyond $100 million. An MSW incinerator proposed for the Finger Lakes region of New York was estimated to cost $365 million to build and would have burned 2,640 tons of trash per day.[110] This facility proposal was halted in March 2019 because of community opposition and local lawmakers' concerns about the environmental and economic risks of the plant. High costs and community opposition have prevented hundreds of facilities from being constructed since the 1980s.[111] Only one facility in the U.S. has been built this century, the Palm Beach Renewable Energy Facility #2 in Florida.

Figure 8: Schematic of an MSW incinerator's financial structure



This facility was built in 2015 and is owned by the Solid Waste Authority (SWA) of Palm Beach County and operated by Covanta.[112] It cost $672 million to build and burns 3,000 tons of trash per day.[113]

Historically, municipalities issued bonds and used the proceeds to finance construction costs of a new facility. Although this is normal practice for states, counties, and cities looking to borrow money for major public projects like roads, schools, and hospitals, incinerator projects have proven to be risky public ventures. Christopher Taylor, formerly head of the Municipal Securities Rulemaking Board, told Reuters, in 2010, when reporting on the Harrisburg, PA incinerator, that, "*anybody who studied incinerator bonds for the last 30 years would find most of them had great difficulties, if not defaults.*"[114] The proceeds from bond sales are provided to the constructing company as a tax-exempt loan, anticipating that the bond debt will be repaid over time with revenues generated from tipping fees and electricity sales or from taxes.[115] One of the reasons Palm Beach County, Florida decided to support financing the construction of such an expensive facility, was in order to extend the life of their landfill by sending ash to the landfill, instead of solid waste.[116]

The large municipal bonds associated with incinerators are paid by local taxpayers and put municipalities at financial risk during construction and operation of an MSW incinerator. While incinerators may earn money for the owner/operator, costs are often borne by the public in the form of public financing and fees.[117] If the plant is unable to raise enough revenue through tipping fees or electricity sales to service the debt, taxpayers may be on the hook for the debt. In some cases, property taxes may be used to service the debt from construction.[118] At the Wheelabrator Westchester incinerator in Westchester County, New York, the county levied a property tax for solid waste services that provided $44 million in revenue to the incinerator company, or roughly 60 percent of the public solid waste budget, in 2009.[119]

Larger plants provide economies of scale that may make profitability more secure. It has been estimated that a larger facility may cost an average of $10/ton less to operate.[120] For host communities, larger plants mean increased air pollution from stack emissions and diesel sanitation trucks that service incinerators. The size of a WTE facility is dependent on the availability of MSW to burn and the ability to sell the net electrical generation.[121] At the Palm Beach Solid Waste Facility's Renewable Energy Facility #2 incinerator in Florida, the county planned to import waste from out-of-county waste haulers and therefore constructed a facility with excess capacity, meaning it was built to handle more waste than Palm Beach County alone produced. SWA and its operating partner planned to issue lower tip fees to out-of-county waste haulers than Palm Beach residents would pay, as an incentive to send their waste to the new facility, essentially putting residents in the position of subsidizing waste disposal for other municipalities.[122] This is a common practice, where facilities originally constructed via local bonds by county solid waste authorities with the purpose of handling the waste from municipalities in that county are constructed much larger than the volume of waste generated by the county. Communities that host these facilities are asked to not only bear the brunt of the pollution from the regional waste-shed, but also the debt and sometimes even disproportionate fees for waste outside their area.

Many incinerators negotiate contracts, or service agreements, with multiple municipalities in the region and/or private waste haulers to secure enough waste on a daily basis to feed the incinerator and raise enough revenue to stay in business. Historically, service agreements were executed for 20-30 year terms.[123] Municipalities may take these risks with the promise of reduced tipping fees for the host community and may be able to receive a "host fee," that returns some revenue to the municipality. For instance, Covanta pays Hempstead Town in Long Island, New York, an annual $7.7 million host fee for allowing them to operate the facility.[124] Harrisburg, Pennsylvania, also collects approximately $250,000 a year as a host fee from its incinerator, which according to state law is supposed to be used for environmental improvements.[125]

One of the worst examples of the financial burden that incinerators can have on municipal finances is the Detroit incinerator. In March 2019, operators of Detroit's infamous incinerator abruptly announced its closure. Detroit's incinerator struggled through decades of financial woes. In 1986, a total of $438 million was issued in bonds to build the facility, which opened in 1989 under city control.[126] At the time of closure, Detroit Renewable Energy CEO Todd Grzech reported, "…*when we looked at it, there was just not enough money in the world to be a good neigh-*

bor, create value for our customers and go forward as a business entity. It just doesn't all match up."[127] After more than 30 years, the Detroit incinerator ended up costing local taxpayers close to one billion dollars to construct, operate and maintain over time due to the significant debt financing that was paid on the original bonds.

In addition to the high capital costs for construction, MSW incinerators are very expensive to operate and maintain and may leave operators/owners with tight margins and operating deficits. The U.S. Energy Information Administration reports that the fixed operating and maintenance (O&M) costs for running an MSW incinerator makes it the most expensive way to generate electricity.[128] In order to estimate the Annual Operating and Maintenance Costs for an average MSW incinerator, three methods were used: (1) the World Bank estimates of operation and maintenance costs for a median size incinerator based on average tonnage and tip fees, (2) the U.S. Energy Information Administration (EIA) estimates of waste burning based on costs per kilowatt-year, and (3) an example case of the York County Resource Recovery Facility in Pennsylvania using publicly available financial records. Table 2 summarizes these methods and the resulting estimates of operation and maintenance costs (see Appendix B for complete calculations).

According to the three different methods, average operation and maintenance (O&M) costs for incinerators fall within a range of $17-$24 million annually. These fixed costs are relatively high in relation to the profit margins that incinerators like the York facility may expect on average. In order to compare the profit margins and fixed costs that most incinerators face, Table 3 summarizes the annual revenue and expenses for the York County Resource Recovery Facility in Pennsylvania. This is a 30-year-old, mid-sized facility publicly-owned and privately-operated by Covanta. Pennsylvania treats trash burning as 'renewable energy' through its net-metering policy and Renewable Portfolio Standard. The facility has the capacity to incinerate 1,344 tons of waste per day and its tipping fee is $62 per ton,[129] which falls in the average range for MSW incinerators. Its gross annual electricity generating capacity is 42 MW. This facility was selected because its annual waste capacity is close to the median value of all MSW incinerators, and since it is publicly owned, its financial reports are publicly available.

The profit margins of this plant are notably thin at approximately $1.2 million annually. Without electricity sales totaling over $9 million, the facility would not raise enough revenue from tipping fees to meet annual operating and maintenance costs. This case study illustrates the incinerator industry's increasing reliance on electricity sales to cushion their tipping fee revenues and offset the potentially increasing O&M costs as the plant ages. If tipping fees fall by as little as 15-20 percent, or the O&M costs increase by the same amount, the facility would no longer be profitable. Some municipalities are forced to cover operating deficits for failing incinerators. In 2016, Covanta's Pittsfield Resource Recovery Facility threatened to close its Pittsfield, Massachusetts facility because of high operating costs and declining profitability. Pittsfield lawmakers passed incentives totaling $562,000, coming from an economic development fund, for the company to stay open for at least another four years.[130]

Life-Extension of Incinerators

Most MSW incinerators currently in operation today were built in the 1980s. The average age of these facilities is 31 years[131] yet the average life expectancy of an incinerator is 30 years.[132] Upgrading decades-old facilities requires another large capital investment, often paid for or subsidized by local taxpayers. The age of these facilities can be a major contributor to equipment breakdowns, shut downs, fires and permitting violations under the Clean Air Act. Upgrad-

Table 2: Cost Calculations for Average Annual Operation & Maintenance Costs for MSW Incinerators

SOURCE	ESTIMATE OF O &M (ANNUAL $)
World Bank estimates for median size incinerator based on tonnage & fees[278]	(1,050 tons/day x 365 days x $44-$55/ton) = **$17 million - $21 million**
U.S. EIA estimates of waste burning costs per kilowatt-year[279]	$392,820 X 61 MW = **$24 million**
York County Resource Recovery Facility	Publicly available financial records[280] **$20,440,360**

Table 3: York County (PA) Incinerator Revenues & Expenses (2017)

REVENUES (ESTIMATE)		EXPENSES (ESTIMATE)	
Tipping Fees	$24,320,550	Operation & Maintenance	$20,440,360
Electricity Sales	$9,350,730	Processing Fee	$716,640
		Misc. Operating Costs	$11,330,020
TOTAL	**$33,671,280**	TOTAL	**$32,487,020**

ing air emissions control technology is particularly expensive and requires large capital investments, typically generated from additional municipal bonds. Municipalities that finance upgrades with bonds use the proceeds from the bonds to loan to the operating company. For example, in Niagara Falls, New York, a Covanta-owned facility received $165 million from the municipality for upgrades in 2012, which served as a tax-exempt loan for the company.[133] In 2015, Niagara Falls Covanta received two new fixed rate tax-exempt corporate bonds totaling $130 million.[134] At the Essex County facility in Newark, New Jersey, the Essex County Improvement Authority issued $90 million in bonds in 2015, to mature in 2045, to finance the upgrade of the facility's emissions control technology to a baghouse.[135] Covanta's Delaware Valley facility in Pennsylvania accessed $40 million in public bonds and partially used it to refinance the debt from upgrading projects at its facility.[136] In Red Wing, Minnesota, a $12.54 million upgrade for the incinerator will be funded by Xcel Energy and the City of Red Wing, with 62 percent of the total cost covered by the City.[137]

The most infamous example of financially ruinous investments in incinerator upgrades can be found in Harrisburg, Pennsylvania. Between 1969 and 2003, the City of Harrisburg issued 11 sets of bonds to build, expand or repair the incinerator facility. In 2003, due to excessive dioxin emissions, the U.S. EPA threatened to shut down the plant.[138] By this time, the facility already held more than $100 million in debt. Instead of shutting down the facility, then Mayor, Stephen Reed, and his administration chose to retrofit it using $130 million in city-backed debt. This debt became a financial nightmare for the city leading to a major budget deficit that caused government layoffs, a 17 percent increase in property taxes and an attempt at Chapter 9 bankruptcy.[139]

A court decision blocked the bankruptcy.[140] However, the Governor intervened and declared a fiscal state of emergency. In 2018, the state filed a lawsuit against responsible parties, including law firms and private investors, who made millions of dollars in fees from structuring this financial debacle. At the time the suit was filed, Governor Tom Wolf released a statement:

> *"It is time to hold those responsible for the failed incinerator debt scheme accountable and recoup the taxpayer dollars wasted by their negligence and deception. This project, started in 2003, represents the worst of how lobbyists and special interests bill taxpayers for their own gain."*[141]

Fire and Accidents

As incinerator facilities age, the incidence of equipment failure or poor operating practices can lead to fires, failures or other accidents at the facility. Flammable, reactive or toxic materials may enter the incinerator via the tipping floor where trucks dump materials before entering the furnaces. These materials may ignite on the tipping floor or in the pit where sparks from materials such as a decaying battery, or spontaneous combustion of organic material.[142] During incineration, chemicals that are incompatible might react and generate heat or produce flam-

The Harrisburg Incinerator on South 19th St.
Source: PennLive, Paul Chaplin, The Patriot News/file.

mable, toxic, or inert gases or mixtures that produce toxic substances, fires, or explosions. These incidences may indicate poor management and declining operations within a facility.

Even if facilities are upgraded, the risks of fires, accidents, equipment failure, and breakdowns can persist. The Montgomery County Resource Recovery Facility in Dickerson, Maryland, is 22 years old and among the newest MSW incinerators in the country. In recent years, however, it has experienced increasing equipment issues and at least six waste pile fires between 2015 and 2017.[143] The waste-to-energy facility in the city of Hartford, Connecticut was the primary waste facility for the state but was fully offline after both turbines broke on November 5th, 2018. An estimated 20,000 tons of waste had to be stored indoors and pre-processed waste was also held in outdoor containers, in violation of state permits.[144] The facility's aging equipment is prone to unplanned outages and Connecticut's quasi-public agency, the Materials Innovation and Recycling Authority (MIRA), previously warned state officials that it would be unable to bear the costs of needed upgrades.[145] According to MIRA officials, member municipalities could see tip fees increase from approximately $72 per ton to $83 per ton by March 1, 2019, to help offset the millions of dollars in extra costs generated by the equipment failure.[146]

The federal government does not collect or maintain a central repository of reports on fire incidences or other accidents in the incineration industry. In order to compile information on incinerator fires and accidents, a search of local newspaper articles reporting these incidences in nearby facilities was tabulated. Four notable incinerator fire accidents were identified since 2008; (1) Montgomery County Resource Recovery Facility in Maryland; (2) Covanta Fairfax County incinerator in Virginia, (3) Spokane City incinerator in Washington, and the (4) Bay County incinerator in Florida.

In December 2016, there was a trash fire inside the Montgomery County Resource Recovery Facility (24 years old, burns 1,800 tons MSW/day) in Maryland which lasted almost two weeks. A "tower of trash eight stories high and 200 feet wide" caught fire in the 30-foot-deep storage pit. The county warned residents living within a mile of the plant to stay indoors or leave the area if they had asthma, lung or heart issues.[147] The Covanta Fairfax County incinerator in Virginia (29-years old facility, burns 3,000 tons MSW/day) experienced a fire that lasted multiple days in February 2017, causing regional concern about air quality. Fire investigators determined that the fire originated on the tipping floor of the building and extended to the holding pit which was filled to capacity at three stories high.[148]

Source: Photo taken by Ari Herzog at Haverhill Resource Recovery Facility in Haverhill, Massachusetts, September 17, 2008.

Vulnerability in Revenue Stream

The incineration industry in the U.S. operates in a volatile economic and regulatory environment. The industry's profit margins are tight, and they rely on steady streams of waste with accompanying tipping fees and generous energy subsidies to ensure their profitability. According to Covanta's 2018 Annual Report,

> *"We also expect that an increasing portion of system capacity will be contracted on a shorter term basis, and so we will have more frequent exposure to waste market risk...As our historic energy contracts have expired and our service fee contracts have transitioned to tip fee contracts, our exposure to market energy prices has increased."*[149]

This volatility coupled with debt burdens and fixed or increasing maintenance and operating costs makes this industry particularly vulnerable to decline as incinerators reach the limits of their life expectancy. Municipal solid waste incinerators rely primarily on tipping fees and secondarily on electricity sales for revenues. As an example, Covanta (which owns 22 facilities and operates 39 facilities in the U.S.), on average, derives its revenues: 71 percent from tipping fees, 18 percent from electricity sales, 5 percent from metal recycling and 6 percent from "other" (i.e. revenues derived from construction revenues, resale of purchased energy, fees from operating transfer facilities, etc.).[150] This distribution of revenues seems to be common among the industry and electricity sales have become an important component in shoring up the profitability of the industry as waste volumes and tipping fees fluctuate. But the market for WTE electricity as a "renewable" energy has also fluctuated as regulatory environments shift. If renewable energy subsidies decline or become unavailable, incinerators may quickly go out of business. Additionally, if new climate mitigation policies that regulate, or price carbon are applied to the incineration industry, it threatens the economic viability of these plants.

Tipping Fees

Tipping fees are the most significant revenue for MSW incinerators and represent one of the most vulnerable parts of their revenue stream. "Tipping fees" or gate fees, are charged by a waste disposal site, such as an incinerator or landfill, to a municipality or private waste hauler for each tonnage of waste deposited at the site. Incinerators are dependent on a steady waste volume and seek to burn waste at their maximum capacity to remain profitable. The more trash they burn, the more revenue they can generate. These tipping fees vary greatly from facility to facility depending on a variety of factors. One important factor is the going price in regional markets where tipping fees at landfills, which are direct competitors for incinerators, can set the lower boundary for fees. If a city or hauler has the option to dump its waste in an incinerator or in a landfill, they will often turn to the lowest cost option in their locality (factoring in transportation costs).

Thus, landfill tip fees are important markers that can outcompete incinerators for trash volumes. Tip fees also vary across the country based on the amount of available, cheap land for landfills. According to *Solid Waste Environmental Excellence Protocol (SWEEP)* 2016 tip fee survey, the average landfill tipping fee was $49, and the following regional trends persisted: "*Regional trends remained the same, with the highest costs in the Northeast and the lowest in the West. Approximate average tip fees at the end of 2016 were $78 in the Northeast, $57 in Pacific states, $48 in the Midwest, $41 in the Southeast and $35 in the West.*"[151] In places where tipping fees at landfills decline or where volumes of waste decrease, incinerator tipping fee revenues can be jeopardized. For example, in New Jersey, Covanta recently closed their Warren County Resource Recovery Facility because of the decline in tip fees as reported in their 2018 Annual Report.[152]

Tipping fees can also vary across different sanitation contracts within the same facility. For instance, trash hauled from Olmsted County to the Rochester, Minnesota facility is set at $83 per ton.[153] Yet waste haulers from Dodge County to the same facility pay about $108/ton or 30 percent more. Dodge County is further away at 23 miles from the facility, while Olmsted County is roughly 7 miles away. In order to ensure incinerators raise enough revenue through tipping fees, municipalities often agree to "put or pay" clauses with incinerators. These clauses stipulate that communities must supply a certain amount of waste or pay a penalty. This guarantees a set revenue stream regardless of the quantity or quality of waste delivered, and it creates a significant financial obligation for the city. These clauses are also criticized by environmental advocates who point to the perverse incentives embedded in these agreements to undermine diversion of waste to more sustainable disposal options like composting or recycling. One

example of the financial costs and perverse incentives that these clauses create for waste diversion can be found in Honolulu, Hawaii. The City of Honolulu has a 20-year "put-or-pay" contract with the Covanta incinerator (HECO) to deliver 800,000 tons of waste annually to the facility or face steep financial penalties.

> *"From 2013 to 2016, the city had to pay Covanta over $6.2 million, according to an audit (PDF) of the city's recycling program released in October. Honolulu could save $7 million in disposal costs and generate $29.5 million in revenue by diverting its plastic and paper recycling from the H-POWER facility. The city also has a profit-sharing arrangement with Covanta for energy sold to HECO, which some see as a perverse incentive to produce more waste rather than less."*[154]

In April 2019, Wheelabrator filed suit against Baltimore County for breaching their sanitation contract by not sending enough waste to their facility and claiming defendants caused over $32 million in damages.[155] "Put or Pay" clauses lock a municipality into generating waste at levels that do not allow for meaningful increases in diversion or waste reduction, following the U.S. EPA's waste hierarchy. A 2011 study found 65 percent of incinerated waste could have been recycled or composted.[156] Burning trash directly conflicts with recycling and composting goals and is a hindrance to local and state Zero Waste targets.[157]

Some cities have caught on to the financial and environmental burden of these "put or pay" clauses and begun re-negotiating contracts. For example, the City of Bridgeport, Connecticut, previously had a "put or pay" contract with the Wheelabrator incinerator but in 2018, when a new contract was signed with the company the city removed this clause. The Housatonic Resources Recovery Authority Executive Director in Bridgeport emphasized that the contract "creates no risk of financial exposure to the town," explaining that eliminating the practice of put-or-pay as one of the major advantages of this new contract.[158]

In order to better understand the vulnerability of incinerators to price fluctuations in tipping fees, the fees for 54 of the 73 MSW incinerators were compiled (Tip fees for 19 facilities were not publicly disclosed or available. For a complete list of tip fees and source information please see Appendix C).[159] Using these fees, the average incinerator tipping fee nationally was estimated to be about $65.35/ton. The national average for landfill tipping fees is approximately $51.82/ton.[160] However, the national average for landfill tipping fees for states *with* incinerators was estimated to be higher at $63.26, as shown in Table 5. Tipping fees for incinerators range from $15/ton of waste for Detroit's former incinerator to as high as $130.55/ton for Covanta's Essex County incinerator in Newark, New Jersey (this tip fee is for some haulers bringing waste from outside of Essex County). The market for waste disposal is regional and many waste haulers export waste to other states, particularly in the Northeast where there is less available landfill space. This dataset represents an estimate of the tipping fee market at a state scale, but regional tipping fees may diverge from this.

Table 4 compares average landfill tipping fees to estimated average incinerator fees by state. In about half the states, the difference between the average landfill tipping fee and the average incinerator-tipping fee is relatively small, which means incinerators in these markets are likely competing head to head with landfills for waste. If incinerator tipping fees increase or landfill fees drop, incinerator revenues could be jeopardized.

> *"The biggest impediment for us is cheap landfilling, particularly in the middle part of the country," Covanta's Van Brunt says. Tipping fees can be as low as $20 per metric ton in land-rich states like Oklahoma. More densely populated coastal regions tend to have more waste-to-energy facilities because of their landfills' relatively high tipping fees—more than $70 in parts of New Jersey, for instance."*[161]

Hawaii, Massachusetts, Maryland and New Hampshire have much higher landfill fees than incinerator fees. This may be due to a lack of landfills or available landfill space within a state, or regionally. The costs of exporting waste might also be much higher, adding to the relative cost of landfilling. Hawaii, for example, will pay much more for out of state export of waste to landfills than a state in the middle of the U.S. In Minnesota, New Jersey, Washington, and Wisconsin, incinerator fees appear much higher than landfill fees. Northeast states have some of the most expensive landfill and incinerator tip fees. This is likely because of the high volumes of waste and shortage of available land compared to other parts of

Table 4: Average Landfill Tip Fees Compared to Average Incinerator Tip Fees by State

States	# Incinerators	Incinerator Tip Fee Data Points	Average Incinerator Tip Fee (i)	Average Landfill Tip Fee (ii)	Difference Between Average Landfill & Incinerator Fees
Alabama	1	1	$40.00	$33.49	($6.51)
California	2	2	$59.50	$58.42	($1.08)
Connecticut	5	3	$65.67	NA	NA
Florida	11	9	$55.36	$54.67	($0.69)
Hawaii	1	1	$45.00	$96.33	$51.33
Iowa	1	1	$55.00	$48.28	($6.72)
Indiana	1	0	NA	$45.02	NA
Massachusetts	7	4	$68.48	$95.00	$26.52
Maryland	2	2	$55.00	$68.28	$13.28
Maine	3	3	$78.83	$78.20	($0.63)
Michigan	2	2	$35.00	$37.81	($2.81)
Minnesota	7	5	$83.20	$61.67	($21.53)
New Hampshire	1	1	$64.00	$80.00	$16.00
New Jersey	4	4	$81.96	$97.43	($15.47)
New York	10	5	$76.82	$66.17	($10.65)
Oklahoma	1	0	NA	$34.81	NA
Oregon	1	0	NA	$69.58	NA
Pennsylvania	6	5	$66.35	$69.59	$3.24
Virginia	4	3	$59.14	$53.48	($5.66)
Washington	1	1	$107.53	$83.44	($24.09)
Wisconsin	2	2	$64.00	$49.09	($14.91)
TOTAL/AVERAGE	73	54	$65.63	$63.26	($2.09)

* Numbers in red parenthesis indicate amount that average incinerator tip fees exceed landfill tip fees in respective states.

(i) Staley, Kantner, and Choi, Analysis of MSW Landfill Tipping Fees, 1-5.

(ii) Average landfill tip fees serve as a proxy for regional waste management prices. States can export waste to landfills out of state in the region which may have different tipping fees from in-state facilities.

the country. These higher tip fees may also be a result of lucrative, long term sanitation contracts with large metropolitan cities in the region that can export their waste easily to nearby receiving incinerators. While tipping fees are subject to regional market changes and the terms of specific sanitation contracts, the relatively small differences in price between landfill and incinerator tipping fees means that there is strong competition in the market for waste and incinerators are at a significant risk if these prices or waste volumes drop.

Electricity Sales

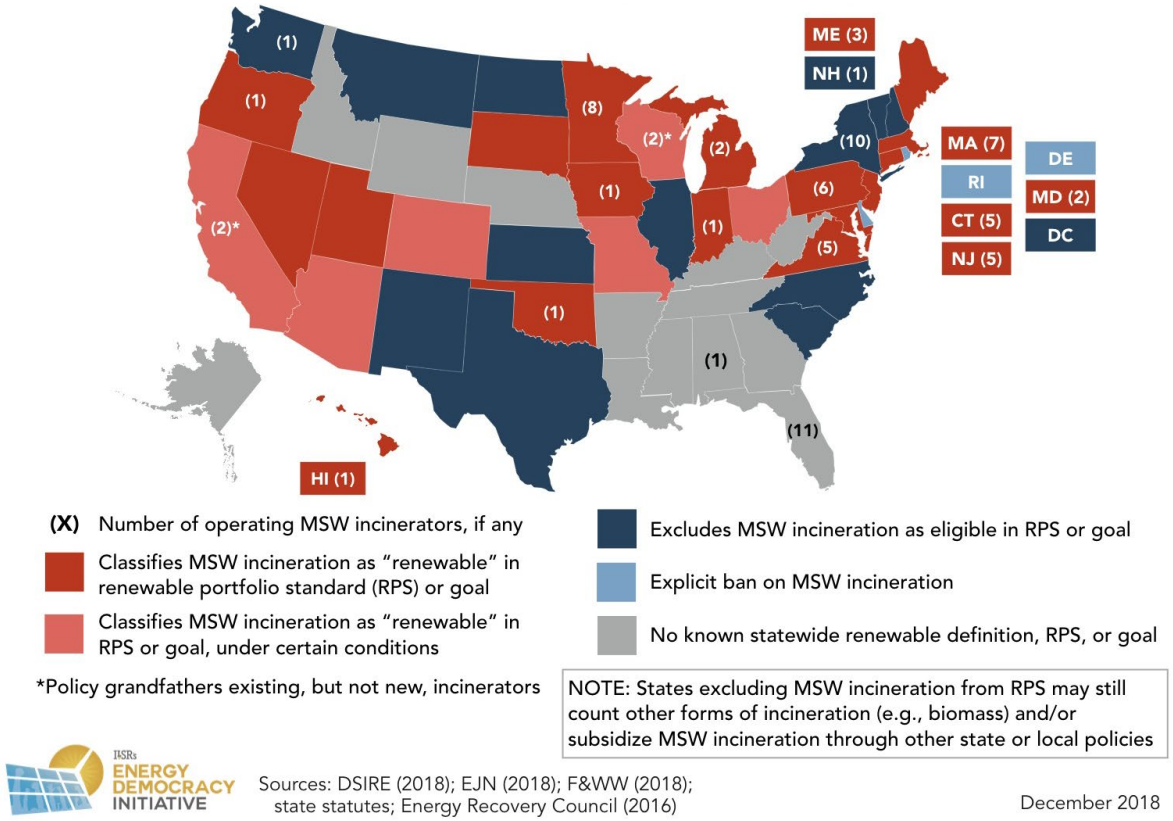
In addition to tipping fees, incinerators depend on sales from electricity generation to boost their revenues. MSW incinerators produced a negligible 0.4 percent of total U.S. electricity generation in 2015.[162] Electricity sales serve to augment the gap between operating costs and tipping fee revenues.[163] However, burning trash is one of the most expensive forms of energy generation in the U.S., with higher capital and fixed costs compared to other energy sources, including wind, solar, natural gas, coal and even nuclear power.[164] For example, waste incineration costs $8.33/MWh compared to $4.25/MWh for pulverized coal and $2.04/MWh for nuclear, the second and third most expensive forms of energy generation.[165] The incineration industry has taken advantage of lucrative renewable energy subsidies because the U.S. EPA and several states have allowed waste incineration to be defined as a "renewable energy" source.

State Renewable Portfolio Standards (RPS) are one example of the way in which states have allowed waste incineration to benefit from the increased interest in investing in renewable energy.[166] Thirty-seven states and the District of Columbia have an RPS.[167] RPS programs set renewable electricity generation targets and define allowable technologies, such as solar and wind that qualify as renewable. Qualifying producers are authorized to sell electricity generated beyond their required obligation and may trade or sell renewable energy credits (RECs), typically receiving one REC per MWh of power produced each year.[168] Twenty-three states include municipal solid waste incineration as a "renewable" form of energy.[169] How much capital is allocated to renewable energy sources depends on what "tier" within the RPS it is placed. Tier I generates more revenue than Tier II, and although most states place incinerators in the Tier II category, the designation grants incinerators valuable access to the renewable energy markets.[170] Only Maryland classifies incineration as a Tier I source of renewable energy on par with solar and wind and this designation was likely a factor in catalyzing a proposal to build a new MSW incinerator in Baltimore that was defeated by local residents.

Figure 9 shows which states have an RPS, if it includes MSW incineration, and the number of MSW incinerators in each state. According to this report, 52 incinerators are located in states that include MSW incineration as an allowable technology; however, at least three of these facilities have closed since the report was published in 2018 (in Minnesota, Michigan, and New Jersey) making the current total 49.[171] Two thirds of all the incinerators in the U.S. today have access to renewable energy subsidies that contributes to the profitability of these plants.

These same subsidies are under increased pressure from advocates to be eliminated or significantly curtailed. In Gonzalez, California, residents opposed a potential waste-to-energy facility that sought access to the state's renewable energy credits. California includes one of two existing MSW incinerators in the state RPS as an allowable technology. When the company behind the proposed facility failed to persuade state officials to include them in the RPS, the

Figure 9: Number of MSW Incinerators included in RPS by State



Source: Institute for Local Self-Reliance, "Waste Incineration: A Dirty Secret in how States Define Renewable Energy."

company withdrew their proposal.[172] This example illustrates the power of advocates to threaten the industry's renewable energy subsidies.

Burning trash is not a renewable or "clean" source of energy. Incineration releases greenhouse gases into the atmosphere, contributing to climate change. MSW incinerators may be at-risk from climate mitigation policies that put a price on carbon pollution such as a carbon tax. Covanta recently reported that if New York State passes a proposed carbon tax bill, they may need to close four incinerators on Long Island because of increased costs.[173] Referring to the potential impacts of a new carbon tax on their business, an industry representative highlighted the likelihood of plant closures with the lack of exemptions for incinerators in the bill:

> *"It's a pretty brutal policy" for waste-to-energy plants, said Scott Henderson, senior director of government relations for Covanta, which estimates the four waste-to-energy plants it operates on Long Island would incur between $31.1 million and $42.7 million a year in new costs as a result of the policy. The combined $332 million in costs over 10 years Covanta expects to incur from the carbon pricing plan "will likely result in waste-to-energy facilities closing."* [174]

These significant costs to meet carbon emissions reductions targets reflect how much carbon pollution is emitted from burning waste. The industry has long argued that their emissions should be considered carbon neutral because they burn waste that is biogenic, hence the carbon they emit would have cycled into the atmosphere in the form of decomposition over time. But MSW incineration delivers a burst of carbon in a short time span (as opposed to natural decomposition over years) and they also burn increasingly large proportions of non-biogenic waste made from fossil fuels like plastics, which contributes to GHG emissions and co-pollutants.[175] The United Nations Environment Programme (UNEP) challenges the carbon neutrality logic of waste incineration:

> *Climate change is time-critical – it is widely accepted that immediate reductions in global GHG emissions are essential to reduce the impact of climate change. The atmosphere does not differentiate between a molecule of biogenic CO2 and a molecule of fossil-derived CO2, therefore it appears logical that immediate efforts should be made to minimize emissions of all CO2 regardless of source.*[176]

Incineration of non-biogenic waste like plastics produces toxic compounds detrimental to human health. Burning organic waste also produces more carbon dioxide than coal-fired power plants.[177] In either case, biogenic or non-biogenic, waste is not a renewable source of energy and thus advocates have rightly criticized industry efforts to exploit these subsidies to the detriment of actual renewable sources. The incineration industry faces the possibility of continuing to lose access to valuable renewable energy subsidies which puts their whole revenue model at risk.

Net metering is another way the industry has used its identification as a renewable energy source to buttress its financial sustainability. Net metering is designed to promote the expansion of renewable energy by allowing renewable energy generators to sell their excess energy to a utility.[178] As of 2015, 44 states have net metering policies. According to the DSIRE database, 14 states and three cities include municipal solid waste incineration in their net metering regulatory policies.[179] As of 2015, 44 states had net metering policies. According to the DSIRE database, 14 states and three cities include municipal solid waste incineration in their net metering regulatory policies. In 2018, the outgoing Republican Governor in New Jersey, Chris Christie, signed a bill, AB 2204, that extended net metering to MSW incinerators and allowed them to sell power directly to up to 10 end-use customers located within 10 miles of the facility.[180] These net metering subsidies can give incinerators unfair access to renewable energy subsidies and deflect important resources from truly renewable energy technologies like solar and wind.

Community Victory: Commerce Incinerator Closure

In June 2018, a waste incinerator, in Commerce, California, named the Commerce Refuse-to-Energy Facility was permanently shut down. The Covanta run facility began operations in 1987 burning over 120,000 tons annually of municipal solid waste. When the facility was originally proposed, it was promoted by the City of Commerce and County Sanitation Districts of Los Angeles as a state-of-the-art alternative to landfilling in Los Angeles County. This incinerator reflects the weak financing model for an industry that has become increasingly dependent on renewable energy subsidies to stay afloat. The facility spokesman stated,
"It really was all because of the expiration of a 30-year power purchase agreement we had with the local utility, Southern California Edison, that expired on December 31, 2016, he said, explaining this cut previous rates of 11 cents per kWh by nearly two-thirds. 'That was insurmountable.' CREA raised tip fees to $84, as far as the local market would allow when factoring in cheaper rates at nearby landfills, but that wasn't enough. Energy comprised two-thirds of the plant's revenue model" (Charles Boehmke, LASDC).[181]

The industry attempted, over the last decade, to lobby California state lawmakers to consider incineration on par with renewable energy sources like solar in order to capture valuable renewable energy subsidies. These efforts were effectively thwarted by community and environmental justice advocates' opposition. East Yard Communities for Environmental Justice is a community based environmental justice organization that works together with community members in East Los Angeles, Lynwood and Long Beach. This group fought alongside the community in opposing the incinerator and advancing calls for the closure of the plant siting both the financial and health impacts on nearby residents. In 2017, East Yard Communities for Environmental Justice together with Valley Improvement Projects quickly organized to prevent incineration from qualifying for renewable energy subsidies. In June 2018, the owners closed this plant because of rising costs without any new forms of revenues.

The final closure of the Commerce incinerator shows that the sustained efforts by EJ advocates can effectively curtail the incineration industry's fiscal viability by removing renewable energy subsidies from the equation. The advocates are continuing their efforts in shutting down another local incinerator in California, as there are two remaining facilities, both operated by Covanta. East Yard Communities for Environmental Justice has been actively opposing another local incinerator, the Southeast Resource Recovery Facility, a 30-year-old incinerator in Long Beach, CA. Community organizers have been putting pressure on the incinerator and potential revenue streams, which included defeating a bill qualifying incineration as renewable energy, monitoring air emissions records, raising awareness of the potential health impacts of incinerators on low-income communities and communities of color, and opposing financial incentives by the City for Covanta. Despite the Long Beach City Council's recent decision to provide financial support for costly upgrades of the aging facility, the voices against these public investments and the increasing call for zero waste are gaining strength. The financial vulnerability and the declining nature of the incineration industry was clearly demonstrated in the case of the Commerce incinerator.

Power Purchase Agreements

Power Purchase Agreements (PPAs) are another way incinerators' boost their revenue through electricity sales. PPAs are contracts between an electricity provider and a power purchaser, typically a utility or trader, in which the purchaser commits to acquiring a certain amount of energy. This long-term contractual commitment to buy energy has been the driving factor behind the development of new projects.[182] Examples of cities that have entered PPAs with MSW incinerators include the District of Columbia; Palo Alto, California; Georgetown, Texas; and Pendleton, Oregon.[183]

Sometimes electricity prices drop or PPA agreements expire and are not renewed. This puts the facility at financial risk. Spokane, Washington's city-owned incinerator previously sold its electricity to Puget Sound Energy for about $12 million per year in revenue.[184] However, the agreement expired in 2011, and a state law the following year removed MSW generation from the qualified list of renewables. Now the Spokane incinerator sells its electricity to Avista, for 3.8 to 5.2 cents per kilowatt-hour, lower than the 9 cents per kilowatt-hour agreement with PSE.[185] Under this new agreement, the Spokane incinerator will earn roughly 58 percent less in electricity sales.

The Miami-Dade County Resource Recovery Facility sold electricity through a Power Purchase Agreement with Florida Power & Light until 2013 when the agreement expired. Electricity sales revenues dropped from slightly over $30 million in 2013 to $14 million in FY2014. After the PPA expired, the rate dropped from $85 per megawatt hour to the market rate of about $28 per megawatt hour.[186] The Commerce, California, incinerator shuttered in 2018 as a direct result of the expiration of its power purchase agreement, a year after legislation aimed at providing incinerators with renewable energy subsidies failed to pass. These examples reflect the vulnerabilities inherent in facilities that rely on these contracts and the power of advocates to challenge the incineration industry's claims to renewable subsidies.

Closures and a Future in Decline

The incinerator industry is in trouble. Aging facilities are often too expensive to maintain, too risky to finance and too costly to upgrade. These plants operate under volatile economic and regulatory conditions that threaten their major sources of revenue, tipping fees and energy sales. Since 2000, at least **31 MSW incinerators closed,** largely due to economic factors. Table 5 lists all 31 facilities and the primary reasons for closure. For eighteen of the facilities listed in Table 6, related news articles sited economic conditions for closing, particularly a decrease in revenue from either loss of tipping fees or electricity sales. Some facilities also cited an insufficient waste stream. According to news reports, six of the facilities closed because they were unable to afford the necessary upgrades in air pollution control equipment (Davis Energy Recovery Facility, Harrisonburg WTE Facility, Southernmost WTE Facility, Miami Incinerator and Nottingham Incinerator). In North Charleston County, South Carolina and Ossipee, New Hampshire, both municipalities shut down their incinerators as part of their strategy to increase recycling and improve environmental management systems. In Detroit and Dearborn Heights, Michigan, facility operators included community opposition as part of the reason they shut down.

Advocates and local environmental justice communities are increasing the pressure on states and cities to reject new incinerators, as well as tighten the requirements and reduce access to subsidies for existing facilities. The combined pressures from increasing costs, risky revenue streams and environmental justice advocacy and zero waste policies creates a picture of an industry in decline. In the following chapter, a review of the health implications and risks associated with this declining industry is explored in depth.

Table 5: Incinerator Closures Since 2000

Facility Name	Location	Year of Closure	Reason(s) for Closure
Detroit Renewable Power	Detroit, MI	2019	Economic conditions[281]
Great River Energy - Elk River Station	Maple, Grove, MN	2019	Economic conditions[282]
Covanta Warren County Resource Company Facility	Oxford, NJ	2018	Economic conditions[283]
Commerce Refuse-to-Energy	Commerce, CA	2018	Economic conditions[284]
Davis Energy Recovery	Layton, OH	2017	Upgrades[285]
Little Miami Waste Incinerator	Hamilton County, OH	2016	Upgrades
Harford Waste-to-Energy	Joppa, MD	2016	Economic conditions/loss of contract[286]
Wheelabrator North Broward	Pompeo Beach, FL	2015	Economic conditions[287]
Wallingford Resource Recovery	Wallingford, CT	2015	Economic conditions/Emissions violations[288]
Harrisonburg Resource Recovery	Harrisonburg, VA	2014	Economic conditions/upgrades[289]
Jackson County Resource Recovery	Jackson, MI	2013	Economic conditions/loss of contract[290]
Wheelabrator Claremont	Claremont, NH	2013	Economic conditions[291]
Coos County Beaver Hill Municipal Waste Incinerator	Beaver Hill, OR	2012	Economic conditions[292] /Safety hazard[293]
Maine Energy Recovery Company	Biddeford, ME	2012	Lack of owner interest[294] /odor complaints[295]
New Hanover County - WASTEC	Wilmington, NC	2011	Economic conditions[296],[297]
Montenay Waste-to-Energy Recycling	North Charleston, SC	2010	Emissions violations[298] / Recycling[299]
Ossipee Solid Waste Incinerator	Ossipee, NH	2009	Recycling mandate[300]
Candia Incinerator/Recycling Center	Candia, NH	2008	Loss of contract[301]
Savannah Resource Recovery	Savannah, GA	2008	Economic conditions[302]
Fergus Falls Resource Recovery	Fergus Falls, MN	2006	Economic conditions[303]
Park County-Livingston Incinerator	Livingston, MT	2005	Emissions violations[304]
Juneau Incinerator	Juneau, AK	2004	Economic conditions[305]
Harrisburg Resource Recovery*	Harrisburg, PA	2003	Economic conditions/Emissions violations[306]
Central Wayne Energy Recovery L.P.	Dearborn Heights, MI	2003	Economic conditions/Emissions violations [307],[308]
Southernmost Waste to Energy	Key West, FL	2002	Air Pollution Control Upgrade cost[309]
Osceola Incinerator	Osceola, AR	2002	Federal Fraud Conviction[310]
Pascagoula Energy Recovery	Moss Point, MS	2002	Economic conditions[311]
Sutton Incinerator	Sutton, NH	2001	Unknown[312]
Miami Incinerator	Miami, OK	2000	Emissions violations/Upgrades[313]
Nottingham Incinerator	Nottingham, NH	2000	Upgrades[314]
Sitka Waste-to-Energy	Sitka, AK	2000	Unknown[315]
Hebron-Bridgewater Refuse District	Bristol, NH	Unknown	Unknown[316]

*Harrisburg, PA facility reopened in 2006 after major upgrades.

Chapter 3:
PUBLIC HEALTH AND COMMUNITY IMPACTS

Waste incinerators produce a variety of pollutants from the combustion of municipal solid waste, to the transport of the waste via diesel sanitation trucks to the ash that is a byproduct of the combustion process. The heterogenous nature of MSW means that waste incinerators are burning a variety of consumer waste laden with heavy metals and other toxic compounds that results in the release of harmful air pollutants when combusted. Populations in close proximity or downwind to the facility may be exposed directly through inhalation of air pollutants or indirectly through consumption of contaminated food or water.

Despite air pollution control technologies and regulatory permit limits, incinerators still emit relatively large quantities of hazardous and criteria air pollutants. As noted in Chapter 1, these air pollutants contribute to and exacerbate cumulative impacts that exist in many environmental justice communities where the population is already overburdened and vulnerable. Furthermore, aging incinerators can experience accidents, malfunctions of their equipment, and declining maintenance, resulting in exceedances of their permitted pollution limits. This is particularly worrisome since studies show that environmental justice communities, where many incinerators are located, have underlying stressors that make them more susceptible to the detrimental health impacts of incinerator pollution.

Incineration Regulations and Public Health

MSW incinerators are relatively large emitters of air pollutants with some studies showing that they emit several pollutants at a rate exceeding that of fossil fuel power plants. Stack emissions include a variety of pollutants such as particulate matter (PM2.5, PM10, Ultrafine particles), nitrogen oxides (NOx), sulfur oxides (SOx), dioxins, nanoparticles, lead and mercury. Ash byproducts also contain dioxins and heavy metals like lead and mercury. Various factors impact the severity and spread of pollutants from a given MSW incinerator. These factors include the size and age of the incinerator, composition of the waste, emissions control technology, stack height and local weather conditions. For metals and other pollutants that are persistent in the environment, the potential effects may extend well beyond the area close to the incinerator and these toxins can build up in the human body over time.

> *"The unintended and uncontrolled release of toxic substances into the environment from waste incineration can occur because of malfunctioning equipment, large changes in the waste feed-stream, poor management of the incineration process, or inadequate maintenance or housekeeping. Off-normal operations (e.g., upsets and accidents) at various points in the incineration process might result in explosions; fires; the release of smoke, ash, or noxious odors into the atmosphere; and the spilling or leakage of contaminated or toxic substances."*[193]

The U.S. EPA regulates air pollutants with the expressed purpose to "protect public health and welfare." They do this primarily under the federal Clean Air Act (CAA) regulations with accompanying state laws. MSW incinerators are primarily regulated under Title V (CAA) permits typically issued by state environmental regulatory agencies. These permits establish atmospheric concentrations of six criteria pollutants that include carbon monoxide, lead, nitrogen oxides, ozone, particulate matter, and sulfur oxides. The CAA uses "MACT" or Maximum Achievable Control Technology standards to establish emissions requirements. The law also limits emissions of 187 hazardous air pollutants (HAPs).[194] Stationary sources like incinerators, which emit or have the potential to emit, ten or more tons per year of any one HAP or 25 tons per year or more of any combination of HAPs are regulated as a "major source" of air pollution and have to implement "maximum achievable control technology" ("MACT").[195]

The CAA does not require the U.S. EPA to eliminate health risks, but rather serves the purpose of reducing risk "sufficiently" to protect public health with an "adequate margin of safety."[196] This is an important consideration for environmental justice communities where a pattern of cumulative and disproportionate pollution exists and where the effects of multiple pollutants, from multiple sources and their synergistic and additive impacts are not well known or regulated.[197] Studies have demonstrated patterns of disproportionate, cumulative impacts in communities of color and low-income communities across the country.[198] These communities are known to experience adverse health outcomes related to socio-demographic characteristics, also known as social determinants of health. Some of the health burdens that have been documented in environmental justice communities include elevated blood lead levels, asthma, preterm births, and increased cardiovascular disease related morbidity and mortality rates.[199] These underlying health disparities combined with the cumulative impacts of multiple sources of pollution create a riskscape where incinerator emissions exacerbate environmental injustice.

Environmental justice communities' critique federal and state regulatory approaches that rely on permitting that only considers chemical by chemical and facility by facility assessments of environmental hazards. Regulations like the CAA and Title V permits for incinerators do not take into consideration

the multiple environmental and social stressors that contribute to the overall impact each facility has on health risks in the exposed population.[200] Another critique of the regulatory process for incinerators is related to emissions data and monitoring. Most of the criteria air pollutants and HAPs are self-reported to the U.S. EPA by facilities on an annual basis. Emissions estimates are typically derived from calculations based on operating conditions and confirmed via stack testing that occurs infrequently (1-5 years) and under "normal" operating conditions.[201] In limited cases, incinerators install Continuous Emissions Monitoring Systems (CEMS) for specific pollutants, such carbon monoxide, NOx, SOx, and opacity but CEMS are not in wide use by MSW facilities for pollutants such as dioxins, mercury or PM.[202]

The emissions reporting from incinerators may be underrepresenting the extent of emissions like dioxins or mercury because the release of these compounds is linked to the composition of the waste being burned at any one time and the assumption of optimal operating conditions which often are interrupted due to malfunctions in the equipment. Emissions measurements are also taken during "optimal operating" times and not during, for instance, start-up and shutdowns or operating upsets, when emissions are often at their highest.[203] Permit exceedances reported by incinerators are not always fined by state regulatory agencies due to relief granted to plants during periods of shut down, start up and malfunctions (SSM).[204] Some researchers and advocates believe emissions data pertaining to incinerators is underestimated or poorly characterized.[205]

Another critical consideration in assessing the health impacts of incinerators is the impact of poor operations and weak oversight and enforcement. In Chapter 2, anecdotal evidence suggests that incinerators in the U.S. have a pattern of accidents which can put local communities at risk. As these facilities age, the lack of proper enforcement coupled with increasing incidences can increase the emissions and related health risks from incinerators.

Environmental Justice and Incinerator Health Risks

Even if one assumes that the existing regulatory structures are sufficient to be protective of human health, environmental justice communities often do not receive the same levels of protection in terms of the enforcement and application of penalties for the violation of environmental laws.[206] Studies show that enforcement officials are slower to respond to incidences of violations and the fines have historically been set lower for facilities located in low-income and communities of color compared to those in whiter or wealthier communities. One study showed that penalties for pollution violations were 46 percent higher in white communities than communities of color.[207] This evidence of underestimating the potential health harm from the emissions of incinerators, the lack of attention to cumulative impacts assessment, the underlying social and health vulnerabilities of exposed populations, and the lax enforcement of existing laws, leads communities to justifiably worry that their health and well-being are not sufficiently protected when it comes to incinerators.

Existing Health Studies

The direct health impacts resulting from exposure to pollutants emanating from incinerators is not well understood or extensively studied in the epidemiological literature in the U.S. In the book, *Waste Incineration and Public Health* (2000), the authors note the reasons for this dearth of studies related to health and incinerators: relatively small study populations; emissions from other pollution sources; variations in human activity; and weaknesses in methodology and data sources.[208] Studies have shown that pollutants emitted from MSW combustion are known to be persistent, bio-accumulative and toxic and once dispersed into the environment these compounds can enter soil, water, and food systems.

> *"Incineration of chlorinated substances in waste, such as polyvinyl chloride (PVC) plastic, leads to the formation of new chlorinated chemicals, such as highly toxic dioxins, which are released in stack gases, ashes and other residues. In short, incinerators do not solve the problems of toxic materials present in wastes. In fact they simply convert these toxic materials to other forms, some of which may be more toxic than the original materials."*[209]

Source: United Workers.

Baltimore Incinerator Proposal Defeated

In 2009, Energy Answers International applied to construct the largest municipal solid waste incinerator in the United States in Curtis Bay, Maryland— a mile or less from Benjamin Franklin High School and Curtis Bay Elementary School. The Curtis Bay community suffered historically from disinvestment and the health impacts of polluting industries in their neighborhoods. These same neighborhoods have been ranked among the most polluted zip codes in the state and the country. In addition to existing polluting industries, the planned incinerator would have been permitted to emit 1,000 pounds of lead and 240 pounds of mercury annually. The company planned to spend nearly $1 billion on the plant which would burn 4,000 tons of waste per day, including plastic, rubber, auto parts and demolition debris.

Benjamin Franklin High School students began organizing when they were made aware of the plans for an incinerator in their community. Destiny Watford and her fellow students co-founded a group called "Free Your Voice" which planned to not only stop the largest incinerator in the U.S. from being constructed but advocated for long term neighborhood-driven development in Curtis Bay. The students went door-to-door informing other residents about the dangers of the incinerator project, held a march and led an act of civil disobedience, sending a message to the Maryland Department of Environment. When they learned that their own high school planned to buy energy from the incinerator, they gave a presentation at their school in opposition, effectively persuading the Baltimore City Public School system to end their proposed contract with the incinerator.[210] In time, 22 customers that planned to buy energy from the incinerator were persuaded to cancel their contracts, eliminating the financial viability of the project.[211]

Interestingly, Maryland is one of the few states in the U.S. that considers incineration a Tier 1 renewable energy source (on par with traditional renewables like wind and solar) in their Renewable Portfolio Standard. These energy subsidies, along with the potential to secure long-term public sanitation contracts with large institutions, allowed for the financing of this proposed facility. In 2016, the Maryland Department of Environment responded to the public pressure and determined that the Energy Answers International permit had expired, making it illegal for the company to construct the incinerator.[212] The defeat of this incinerator proposal in Baltimore reflects the importance of local, grassroots efforts to prevent the adoption of long term public contracts that finance these facilities and lock them into a polluting infrastructure.

After pollutants from an incineration facility disperse into the air, some people close to the facility may be exposed directly through inhalation or indirectly through consumption of food or water contaminated by deposition of the pollutants from air to soil, vegetation, and water.[213] In the European Union, MSW is the second most important emission source type for dioxins (iron ore sintering ranked highest).[214] Globally, waste disposal, primarily from incineration, contributes to ~8 percent of the total anthropogenic mercury emissions.[215] In a 2010 study of China's mercury source categories, emissions from incineration of municipal solid waste (MSW) was shown to experience the fastest growth due to the rapid expansion of the MSW incineration industry in China. According to this study "MSW incineration should be considered a high priority source in China's mercury control strategy."[216]

While the literature on the direct health impacts of waste incineration is limited in the U.S., there are a handful of studies from Asia and Europe in particular, where MSW incinerators are prevalent, that provide some insights into health-related impacts that can be applied in the U.S. context.[217] There are also case studies that point to specific health impacts such as a study that showed that dioxin emissions increase the risk of non-Hodgkin lymphoma among the population living in the vicinity of a municipal solid waste incinerator in France.[218] Another study in France considered all births (n = 21,517) of women residing within a 4-km radius of an incinerator at the time of delivery (2003-2010) and found that pre-term delivery increased with increased exposure.[219] A study in Italy analyzed the occurrence of miscarriages in women aged 15-49 years residing near seven incinerators of the Emilia-Romagna Region (Northern Italy, 2002-2006) and found that an increase of PM10, due to incinerator emissions was associated with an increased risk of miscarriage.[220] A 2005 study in Japan found that proximity of schools to municipal waste incineration plants may be associated with an increased prevalence of wheeze, headache, stomach ache, and fatigue in Japanese school children.[221] These health studies help shed light on the potential health risks posed by MSW incinerators in the U.S.

In order to characterize the nature of the potential health risk that aging incinerators in the U.S. might pose, several factors are summarized in this Chapter, including: (1) the health risks associated with specific air pollutants from incinerators, (2) a ranking of incinerators based on a snapshot of their emissions profiles for the most health harmful air pollutants and their presence in EJ communities, (3) a review of the coincidence of incinerator facilities in nonattainment areas, and (4) an estimation of emissions from waste hauling associated with incinerators.[222]

Incinerators as Major Sources of Air Pollutants

In 2017, the Environmental Integrity Project compiled a report, *The Truth is in the Trash*, comparing MSW incinerator emissions to coal-fired power plants and found that incinerators: produced, NOx, lead, and mercury at a higher rate than coal and Greenhouse Gases at an average rate that is 68 percent higher, per unit of energy delivered to the grid, than coal plants.[223] An example of the relative scale of pollution emitted by incinerators can be seen in the Montgomery County Resource Recovery Facility in Maryland. The plant releases approximately 740 tons of air pollutants annually and sends 180,000 tons of toxic ash to Virginia landfills.[224] The Environmental Integrity Project found that:

> *"On average between 2007 and 2009, the amount of mercury produced per hour of energy at MCRRF was 2-4 times and at WBI [Wheelabrator Baltimore Incinerator] 2.5-5.6 times that of the coal power plants. Between 2007 and 2009, MCRRF produced on average 3-8 times more lead per hour of energy than the coal power plants, while WBI produced on average between 6.5 and 18 times as much lead per hour. As with mercury, these emissions rates make WTE incinerators among the largest sources of lead in the state."*[225]

Table 6: Major Pollutants and their Sources

Pollutant	Examples of Sources
Dioxins	Plastics or fuels such as wood, coal and oil
Heavy metals	Batteries, pigments, leather, solder, cans, and consumer products and packaging
Chlorine	Polyvinyl chloride plastics and some bleached paper
Polystyrenes	Food service products such as rigid trays and containers and disposable eating utensils
Sulfur Oxides	Tires and gypsum wallboard
Nitrogen Oxides	Food and yard waste
Lead	Lead-acid car batteries, electronic items, leaded glass and plastics, batteries, fluorescent tubes, thermometers, and thermostats
PFOS, PFOA	Carpets, clothing, fabrics for furniture, paper packaging for food and other materials that are resistant to water, grease or stains

Some of the most health harmful pollutants emitted by incinerators include heavy metals like lead and mercury, as well as other hazardous air pollutants, particulate matter, nanoparticles, dioxins and furans.[226] Table 6 describes some of the primary sources of air pollutants emitted by incinerators. Because MSW incinerators burn a heterogenous mix of household and other waste, the resultant emissions from these facilities also varies significantly.

The combustion of household waste, plastics, fuel oil, electronic components or batteries for example, can emit dioxin. Dioxin emissions from incinerators have generated significant public health concerns because exposure, even in small amounts, can result in neurologic, immunologic, and reproductive impacts. According to the U.S. EPA, dioxins are "*are highly toxic and can cause cancer, reproductive and developmental problems, damage to the immune system and can interfere with hormones.*"[227] Dioxins are also extremely persistent compounds that take a long time to break down and can bioaccumulate. Studies show that "*epidemiologic data suggest that there is little or no margin of exposure for humans, [considered safe] with respect to these developmental effects.*"[228] Nanoparticles are another understudied but potentially harmful source of emissions from incineration of MSW. A 2014 study suggests that the fate of these particles, when incinerated is unclear, "*Due to the large variety of nanoproducts, the toxicity potential of nanomaterials and the wide range of potentially affected waste streams, the consequences for future waste management are currently unpredictable… The few available studies which address the incineration of nanoproducts have indicated that ENM [Engineered nanomaterials] removal efficiencies may vary significantly and depend on properties such as particle type and size.*"[229] Nanoparticles, ultrafine and PM2.5 particles can pose serious health risks to humans from the inhalation of these tiny particles.

> *"Epidemiological studies demonstrated associations between deaths and particulate air pollution even at extraordinarily low mass concentrations (Pope et al. 1992; Schwartz 1994) ….We pointed out that the majority of deaths associated with air pollution in the epidemiological studies were from cardiac rather than respiratory disease and attempted to explain the apparent fact that toxicologically tiny doses of particulate matter (PM), mainly carbon, to the lungs could cause death from failure of another organ."*[230]

A recent study concluded, "…. *anthropogenic $PM_{2.5}$ was responsible for 107,000 premature deaths in [U.S.] 2011, at a cost to society of $886 billion.*"[231]

There are a variety of health risks and uncertainties associated with the release of toxic air pollutants from incineration. The lack of conclusive scientific certainty relating to the causes and the consequences of the harm caused by certain substances or activities, however, should not be viewed as a reason to postpone preventative measures, as affirmed by many international conventions.[232] The precautionary principle was defined at the Wingspread Conference in 1998 as, "*When an activity raises threats of harm to human health or the environment, precautionary measures should be taken even if some cause and effect relationships are not fully established scientifically*".[233] This principle aims at ensuring a higher level of environmental protection through preventative decision-taking in the case of risk.[234] The precautionary principle tries to prevent harm before it occurs and is a foundational tenant of the Environmental Justice Movement. While the direct health implications of incineration are not well studied, incinerator emissions contribute to the overall cumu-

lative impacts that may harm EJ communities. Thus, the precautionary principle would lead communities to prefer less harmful alternatives to waste embodied in the approach of zero waste and waste reduction and diversion over incineration.

Danger on the Horizon: 2017 China Waste Ban

In 2017, China announced a ban on 24 types of solid waste, including certain plastics, unsorted scrap papers, and discarded textile materials. This ban sent shock waves through the waste management systems in the U.S., which are heavily reliant on the export of recyclables. Since the China Ban, municipalities are scrambling to find disposal options for their low quality, hard-to-recycle waste materials. In the short-run, many cities are sending recyclable materials to incinerators or landfills or letting them pile up.[235] If addressed properly, China's ban can activate additional investment in domestic recycling capacity, secondary material markets, and programs for reducing consumption.[236] Some of this plastic may end up in MSW incinerators. According to a Guardian article from February 2019, the Covanta incinerator in Chester, PA received a significant amount of Philadelphia's sorted recyclables in response to the ban from China. "*About 200 tons of recycling material is sent to the huge Covanta incinerator in Chester City, Pennsylvania, just outside Philadelphia, every day since China's import ban came into practice last year, the company says*."[237] In April 2019, Philadelphia announced that they would stop sending their recyclable material to the incinerator.[238] Increased plastic combustion is particularly worrisome because burning plastics releases toxic air pollution such as dioxins which increase the risk to host communities like Chester, Pennsylvania.[239]

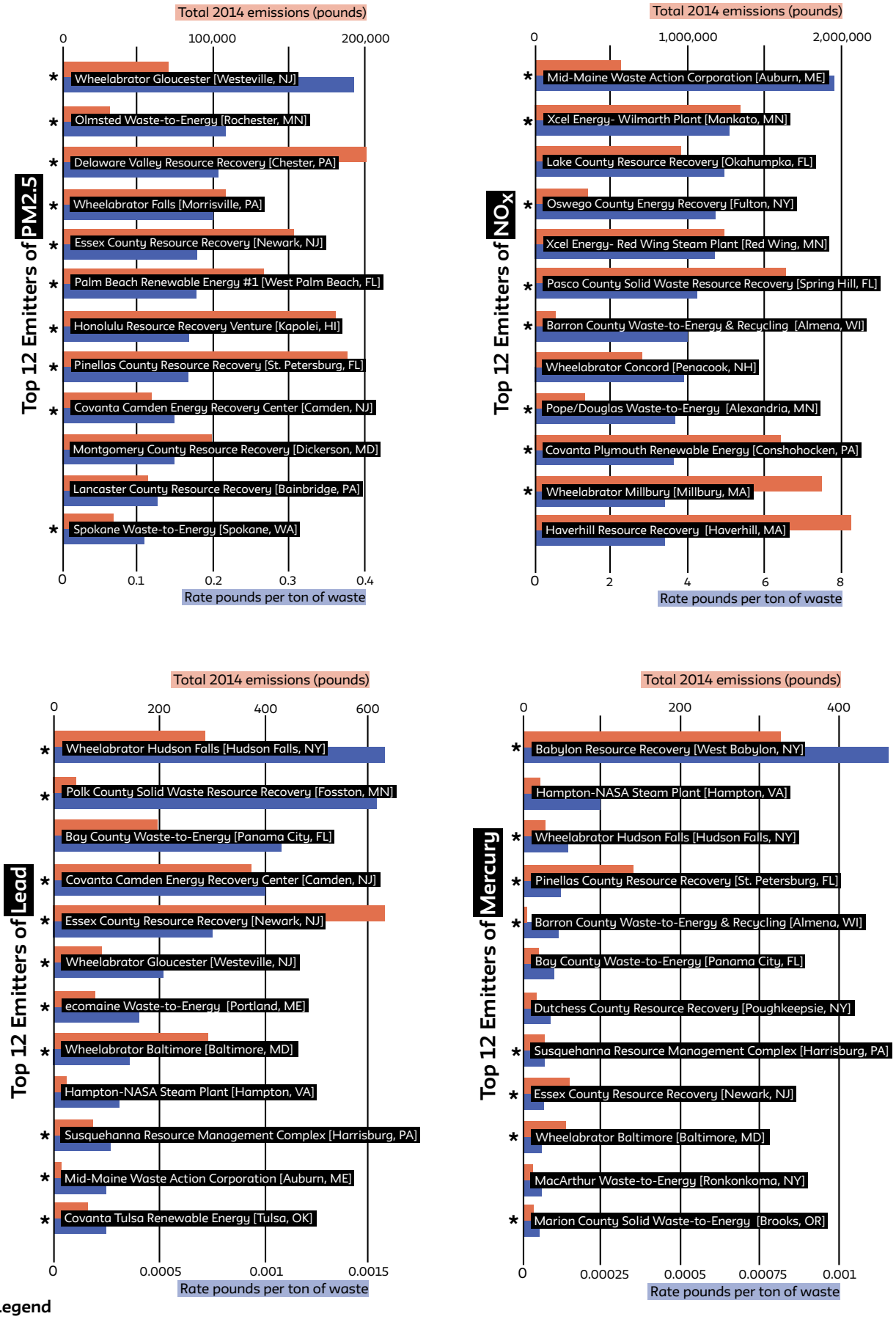
Incinerator Emissions Data: The Dirty Dozen

In order to assess the relative impact and health risks associated with MSW incinerators, a snapshot of air pollutant emissions data was compiled for all incinerators in 2014 (latest available data). Air pollution emissions data was obtained from the U.S. EPA's Enforcement and Compliance History (ECHO)[240] online database. ECHO provides facility-level compliance data for environmental regulations and Air Pollution Reports from the National Emissions Inventory,[241] Greenhouse Gas Reporting Program,[242] Toxic Release Inventory,[243] and Clean Air Markets Division.[244] Stack test data and emissions calculations are reported by the facility to state or tribal officials, who then report emissions to the EPA through the Emissions Inventory System (EIS). The EIS collect and publish this data every three years in the National Emissions Inventory online system that feeds into the ECHO website.

Emissions data for all 73 incinerators was collected for the following pollutants: NOx, SOx, mercury, lead, particulate matter (PM10 and PM2.5), and carbon monoxide. These pollutants represent some of the most health harming air pollutants emitted by incinerators, for which a complete dataset is available.[245] (See Appendix E for additional tables showing data for all seven pollutants). Facilities were ranked according to the top twelve highest emitters (among the 73 facilities nationwide) for each pollutant according to total annual emissions (lbs) and the rate of emissions (lbs/ton) per ton of waste incinerated. These top emitting facilities were then identified according to whether they are located in environmental justice communities (see Chapter 1 for definition of EJ communities). Figure 10 summarizes the results of this ranking exercise for particulate matter (PM 2.5), NOx, lead and mercury.

The "Dirty Dozen" Incinerators charts in Figure 10, illustrate the most polluting MSW incinerators according to PM2.5, NOx, Lead, and Mercury emissions. Approximately **1.6 million people** live within a three-mile radius of the "Dirty Dozen" incinerators for these four pollutants.[246] There are **4.4 million people** that live within a 3 mile radius of all 73 incinerators in the U.S. The relative emissions produced by an incinerator are in part dependent on the amounts of waste burned so that one would expect the largest incinerators to be most likely to emit the largest amount of pollutants. Since daily capacity to burn waste varies significantly among the 73 incinerators, it was important to examine both the total air pollutants (lbs) emitted annually as well as the rate of emissions (lbs/ton) per ton of waste combusted. The emissions rate was calculated by dividing the annual emissions (lbs) by the annual tons of waste burned at the facility. ***The Dirty Dozen charts reveal that most of the highest emitting facilities in each pollutant category (NOx, SO2, mercury, lead, PM 2.5, PM 10, CO), are in environmental justice communities.***

Figure 10: Dirty Dozen Incinerators



The following represents the number of "Dirty Dozen" incinerators that are located in EJ communities by pollutant category:

- PM10: 10
- PM2.5: 10
- Lead: 10
- NOx: 8
- SO2: 9
- CO: 8
- Mercury: 8

Ten of the twelve incinerators that emit the greatest amount of lead emissions, are in environmental justice communities. Exposure to lead can affect virtually every organ and can cause severe neurological damage in humans, especially in children and fetuses.[247] The Covanta owned, Essex County Resource Recovery incinerator in Newark, New Jersey emits the largest total amount of lead of any MSW incinerator in the country with over 600 pounds of lead reported in 2014, far above the next highest emitter, Covanta Camden (also in New Jersey) at 380 pounds. The Newark plant is emitting total annual lead levels higher than the largest incinerator facility in the U.S. These lead emissions are particularly troubling when considered in the context of the overall lead risk already present in the population. Children in Newark for example, represent 13 percent of the children in the state with elevated blood lead levels (Newark has 3.8 percent of the state's children).[248] The City of Newark is also experiencing widespread lead contamination in the City's drinking water supplies and more than thirty public schools tested above the federal action levels for lead in their drinking water.[249] The incinerator's lead emissions combine with multiple sources of lead in the home and school environments and may compound the potential health risks of already overburdened EJ communities in Newark. The Wheelabrator Hudson Falls incinerator in Washington County, New York is the highest per ton emitter of lead in the country and is also in an EJ community.[250]

Incinerators are also significant emitters of mercury. Mercury can cause neurologic, renal, developmental and reproductive damage.[251] Eight of the twelve incinerators with the highest emissions of mercury pollution in the U.S. are located in environmental justice communities. The Babylon Resource Recovery Facility in New York is located in an EJ community and it stands out as both the largest total emitter of mercury, releasing over 319 pounds of mercury annually as well as the highest per ton emitter in the country. The Pinellas County Resource Recovery Facility in St. Petersburg, Florida, emits 134.89 pounds of mercury annually and is also in an EJ community.

The incinerator that emits the most PM2.5 pollution in the country is the Delaware Valley Resource Recovery Facility in Pennsylvania, owned and operated by Covanta. In 2014, the facility emitted over 200,000 pounds of PM 2.5. This incinerator is in a non-attainment area for both PM2.5 (2012) and 8-hour Ozone (2015).[252] The PM emissions from the incinerator contributes to the overall air quality in the region and related health risks. PM2.5 is associated with decreased life expectancy and can cause or worsen several heart and lung problems.[253] Recent studies have shown that PM2.5 can have significant health and morbidity impacts on the US population.

> *"This translates to PM 2.5 causing an extra 20,000 deaths a year," said a co-author, Joel D. Schwartz, a professor of epidemiology at Harvard. "Separately, a 10 parts per billion decrease in ozone would save 10,000 lives per year. The effect was greater for low-income people, African-Americans, women and those over 70, and the risk remained significant even at levels below what the Environmental Protection Agency considers safe."*[254]

In 2012, Delaware County, PA had the highest pediatric inpatient hospitalization rate for asthma, after Philadelphia, in the state.[255] Even within the County, in 2013, Latino and Black children were more likely to have asthma than White children (2.5 and five times respectively).[256]

NOx (Nitrogen Oxides) is also a significant health impacting pollutant that is a major contributor to ozone, acid rain, and particulate matter.[257] NOx contributes to respiratory disease, cardiovascular disease and asthma.[258] The incinerators with the highest total annual emissions of NOx, are the I-95 Energy/Resource Recovery facility in Lorton, Virginia and the Pinellas County Resource Recovery Facility in St. Petersburg, Florida, both of which are located in EJ communities. Looking at the rate of NOx emissions per ton of waste burned, Mid-Maine Waste Action Corporation in Auburn, Maine and Xcel Energy-Wilmarth Plant in Mankato, Minnesota rank the highest, both are located in EJ communities.

Clean Air Act Violations

MSW incinerators are required, under the Clean Air Act, to have Title V operating permits that identify the amount of allowable emissions per year at a facility. If a facility exceeds the allowable emissions limits and operating parameters (i.e. temperatures, record keeping, monitoring, etc.) specified in the permit, these exceedances or violations of the permit are required to be reported to state regulatory authorities. The U.S. EPA collects and publicly reports enforcement and compliance information through a system called ECHO (Enforcement and Compliance History Online).[259] In order to assess the relative frequency and types of compliance issues occurring at incinerators across the country, a review of Clean Air Act violations data was compiled and assessed from the ECHO website. The ECHO website has known data gaps due to its reliance on a diverse range of inputs from various states. Each state tracks permit violations, enforcement actions and compliance differently, and each reports their information differently to the U.S. EPA. Thus, there are known gaps in the completeness and accuracy of this federal database.

The violations and compliance issues reported in ECHO are likely conservative estimates based on known case studies where state level data on permit violations and exceedances are much higher than what is reported in ECHO. For example, in January of 2019 the nonprofit groups Environment Michigan and the Ecology Center filed a Notice of Intent to Sue the Detroit incinerator alleging 600 violations of federal hourly limits on carbon monoxide and nitrogen oxide emissions over the past five years. According to the Detroit Free Press, the incinerator, "*exceeded pollution emissions standards more than 750 times over the last five years, Michigan Department of Environmental Quality records show.*"[260] In 2007, the Eastern Environmental Law Clinic filed a notice of intent to sue Covanta Energy, the owners of the Newark, NJ incinerator for noncompliance with the Clean Air Act, alleging hundreds of violations of federal clean air standards for sulfur dioxide, opacity, carbon monoxide and particulate matter.[261] These violations were likely not reported to the ECHO system, either because the state did not consider them violations or the state did not adequately report these exceedances into the federal database. Also, important to note is evidence that states have varying approaches to compliance and enforcement, with some states adopting more aggressive inspection and enforcement oversight than others.[262]

ECHO data for the 73 incinerators reveals that an estimated 21 incinerators received 126 "Federally Reportable Violations" under the Clean Air Act between 2016 – 2019.[263] Data were pulled from the Three-Year Compliance History table from each facility's page on ECHO as well as facility fines (fines levied by state agencies). Twenty-one incinerators received 49 fines totaling $535,737. Table 7 summarizes the incinerators with the greatest number of violations logged in ECHO between 2016 and 2019.

Incinerators may receive violations for exceeding emissions limits under their Title V permits for one or more pollutants, or for "facility or administrative issues." These administrative issues may refer to poor record keeping or monitoring practices, failure to submit or file reports with the state, or to maintain operational parameters required in the permit such as specific temperature controls, feed rates or oxygen levels.[264] Pollutants that appear the most often as violations include carbon monoxide, sulfur dioxide, and particulate matter. These violations may be the result of incomplete combustion, equipment malfunction or other compromised conditions within the facility. Interestingly, many of the same pollutants that are typically monitored via Continuous Emissions Monitoring Systems (CEMS) like carbon monoxide, also appear frequently in the list of compliance issues (stack-gas concentrations of O_2, CO, NO_x, SO_x, and opacity are often monitored via CEMs).

This points to another potential limitation in the oversight of incinerators - without CEMS for pollutants of greatest health concern like dioxins, mercury, and lead - facilities may be underreporting the instances of exceedances occurring at incinerators. CEMS for these pollutants is not currently required for most existing MSW incinerators in the U.S. "*Reliable continuous emission monitors (CEMs) for dioxins and furans or for metals would be desirable, because automatic devices electronically linked to such devices could directly control those emissions of greatest potential health consequence.*"[265] The Baltimore City Council recently passed a bill to require incinerator facilities to install CEMS for many of these pollutants as well as institute more stringent emissions limits.[266] This bill may result in the closure of the Baltimore incinerator due to the costs to retrofit the plant,

Table 7: MSW Incinerator Violators and Fines Levied (2016-2019)

Top MSW Violators and Fines Levied (2016-2019)					
	MSW Incinerator	State	# of Violations	# of fines (amount of fine $)	Example of recent Violations
1	Covanta Plymouth Renewable Energy	PA	33	8 ($73,045)	Administrative
2	Detroit Renewable Power*	MI	27	1 ($149,000)	Sustained High Priority Violations for every quarter between April 2016 and March 2019 when it closed. Sulfur Dioxide, Carbon Monoxide.
3	Delaware Valley Resource Recovery	PA	11	4 ($34,217)	Administrative
4	Lancaster County Resource Recovery	PA	8	1 ($42,196)	Administrative
5	York County Resource Recovery	PA	8	1 ($9,148)	Administrative
6	Covanta Camden Energy Recovery Center	NJ	5	4 ($7,050)	Particulate Matter, Sulfur Dioxide, Carbon Monoxide
7	Perham Resource Recovery	MN	5	1 ($11,370)	Cadmium, Particulate Matter, Administrative
8	Essex County Resource Recovery	NJ	3	6 ($90,960)	Particulate Matter, Sulfur Dioxide, Carbon Monoxide
9	Covanta Tulsa Renewable Energy	OK	3	0	Unresolved continuous Carbon Monoxide since 2014
10	Wheelabrator Portsmouth	VA	2	1 ($7,669)	Chlorinated Dioxin and Furans
11	Xcel Energy French Island Generating Station	WI	2	0	Total Hazardous Air Pollutants
12	Wheelabrator Bridgeport	CT	1	0	Unresolved continues Mercury emissions
*Closed in March 2019					

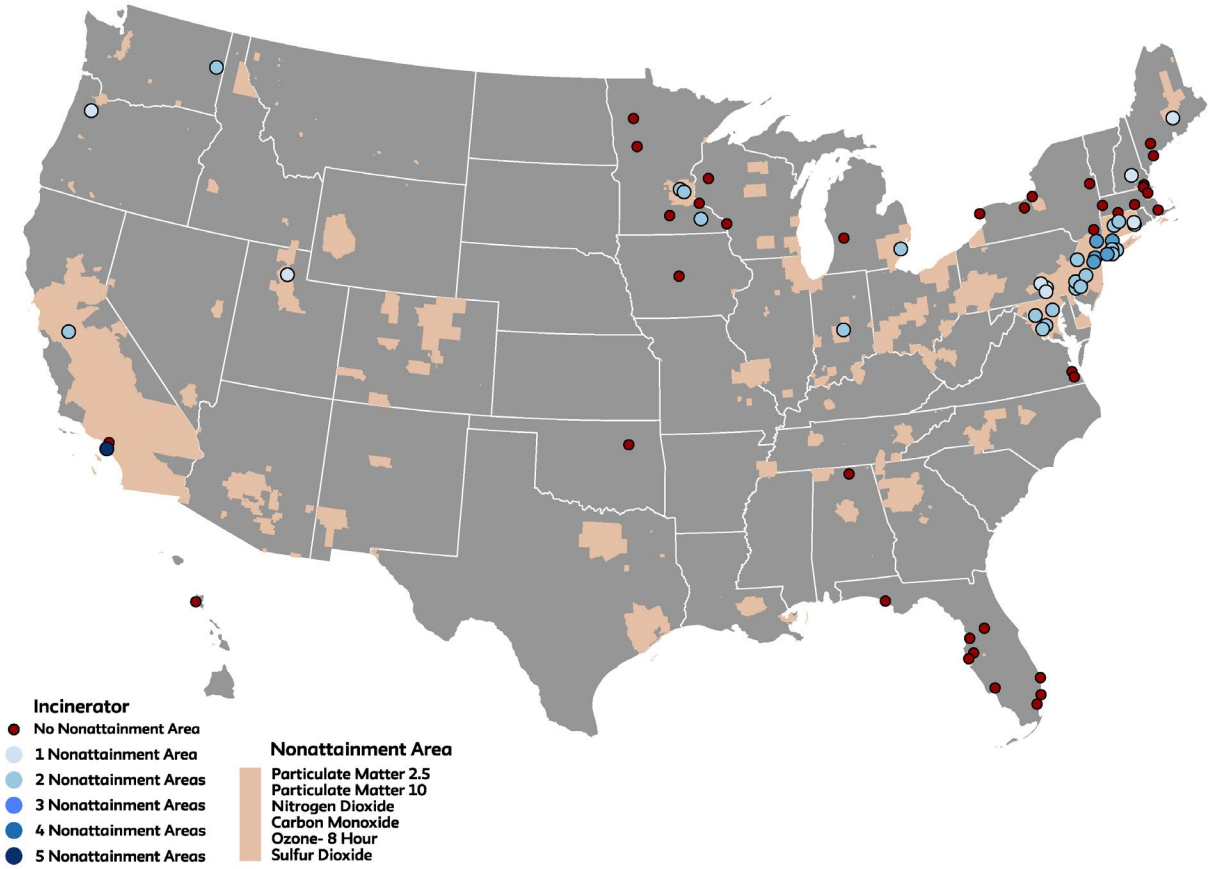
"*The incinerators' owners say it would be impossible to retrofit their plants to meet the standards set out in the legislation and so would have to close if the strict standards go into effect.*"[267] The added risk from poorly functioning and non-compliant facilities exacerbates existing health risks.

Incinerators and Areas Out of Attainment with the National Ambient Air Quality Standards (NAAQS)

The National Ambient Air Quality Standards (NAAQS) sets national limits for the six criteria pollutants based on atmospheric (ambient) concentrations. Areas of the country are assessed for these six pollutants: ground level ozone, particulate matter (PM), lead, sulfur dioxide (SOx) and nitrogen dioxide (NOx) and carbon monoxide. If an area is determined to be "not in attainment" for any of the criteria pollutants, states are expected to develop a State Implementation Plan (SIP) for achieving attainment through state-selected and enforced controls on emissions.

In order to assess the underlying air quality conditions in the places where incinerators are located, the Nonattainment Areas for Criteria Pollutants Green Book (2019) was used to generate a map showing the location of MSW incinerators within nonattainment areas (all nonattainment areas for all six criteria pollutants combined). There are 39 incinerators that fall within a nonattainment area for one or more criteria pollutants. Twenty-two incinerators fall within two nonattainment areas and five incinerators fall within three nonattainment areas. The Southeast Resource Recovery Facility in Long Beach, California is the only facility that falls within five nonattainment areas. Figure 11 depicts the incinerators located in non-attainment areas in the shaded areas on the map.

Figure 11: MSW Incinerators in Non-Attainment Areas



The presence of incinerators in areas that are in non-attainment for criteria air pollutants indicates places where the industry is contributing to already poor air quality. A recent study has shown that there are significant gaps in air pollution monitors used to designate nonattainment areas and therefore the scope of the problem may be underestimated. Using satellite data, this study found that 47.6 million Americans (up from 23.3 million) live in counties that do not meet that standard for PM2.5.[268] Many of these communities are burdened with pollution from multiple sources impacting public health and well-being, including MSW incinerators.

Diesel Emissions from Waste Hauling to Incinerators

In addition to stationary source air pollution, waste incineration impacts environmental and human health via mobile source emissions derived from the largely, heavy-duty diesel (HDD) sanitation trucks that collect and haul almost all MSW in the country and concentrate near MSW facilities.

> *"Garbage trucks are one of the least efficient vehicles on the road. Powered by diesel fuel, they average just 3 miles per gallon, burn about $42,000 of fuel per year, and emit about 20 times the carbon emissions of the average US home. As they rumble down city streets waking residents at dawn, they make more than 1,000 stops a day and log an average of 130 miles a day."*[269]

Sanitation trucks release significant health harming diesel particulates including black carbon and soot as well as nitrogen oxides, particulate matter, carbon monoxide, and volatile organic compounds.[270] One of the most direct and localized sources of air pollution associated with proximity to MSW incinerators are diesel emissions from sanitation trucks. Since MSW incinerators operate 24 hours a day, seven days a week, the impact of these diesel trucks on local communities can be significant. Many of these communities have multiple waste facilities, such as transfer stations, and may see thousands of diesel trucks per day from a variety of sources. Waste delivered to incinerators may originate from more affluent neighborhoods or even different states and spend time queuing at the incinerator or traveling into the

facility via residential streets. Living near a waste site may mean chronic exposure to diesel fumes which have been classified as a carcinogen by the National Cancer Institute[271] and may contain up to 40 types of hazardous air pollutants.[272]

Diesel trucks have the worst fuel economy of highway vehicles[273] and emit approximately 20 percent of global anthropogenic emissions of nitrogen oxides (NO_x), which are key PM2.5 and ozone precursors.[274] Rear-loader refuse trucks are most common for collecting residential trash and have an average fuel economy of between 1 and 3 miles per gallon.[275] Table 8 summarizes the pounds of pollutants (VOCs, carbon monoxide, nitrogen oxides, and particulate matter 2.5 and 10) emitted per day by these trucks.[276] These calculations on based on estimates of the average refuse truck which travels an estimated 130 miles per day and 25,000 miles per year.[277] Sanitation trucks consume 43-130 gallons of diesel fuel daily, based on an average fuel economy.

The average incinerator handling 1,300 tons/day requires a sanitation truck fleet of approximately 186 diesel trucks per day. According to the estimates of emissions in Table 9, a fleet of this size would emit (annually) approximately:

- 8,760 lbs of volatile organic compounds
- 33, 215 lbs of carbon monoxide
- 142,715 lbs of nitrogen oxides
- 3,285 lbs of PM 2.5

The Miami-Dade County Resource Recovery Facility is the largest MSW incinerator in the country (4,200 tons/day) and its truck fleet would require double or triple the amount of trucks required of the average incinerator. Based on the total amount of tons hauled in a year and the tonnage an average sanitation truck can haul, Miami-Dade County Resource Recovery Facility's truck fleet was estimated to be between 672 and 840 diesel trucks daily. The total emissions from a fleet of 672 trucks (each 35 cubic yards in size hauling 7 tons of waste) would emit (annually):

- 31,755 lbs of volatile organic compounds
- 120,085 lbs of carbon monoxide
- 515,015 lbs of nitrogen oxides
- 12,410 lbs of PM 2.5

The resultant emissions contribute to the health burden and risk in host communities, particularly for communities that face the cumulative exposure to multiple mobile and stationary sources of pollution. These emissions are not factored into the regulatory permits or emissions thresholds for incinerators. Thus, the full extent of their impact on local health is underestimated by regulatory agencies.

Table 8: Pollutants Released by Heavy Duty Diesel Sanitation Trucks

Heavy Duty VII Diesel-Burning Refuse Trucks (130 miles/day)			
Pollutants	**One Truck** (lbs/day)	**Fleet of 119 Trucks** (lbs/day)	**Fleet of 181-265 trucks** (lbs/day)
Volatile Organic Compounds	0.13	15.43	23.48 - 34.37
Carbon Monoxide	0.49	58.31	88.69 - 129.85
Nitrogen Oxide	2.1	249.90	380.1 - 556.5
Particulate Matter 2.5	0.05	5.95	9.05 - 13.25
Particulate Matter 10	0.05	5.95	9.05 - 13.25

Conclusion

MSW incinerators in the U.S. are aging facilities that face an increasingly uncertain economic future. This industry benefits from a lax regulatory system and government support in a variety of forms from direct public expenditures to renewable energy subsidies. Incinerators represent an affront to environmental justice communities by contributing to disproportionate, cumulative impacts in communities of color and low-income communities. These communities are host to a majority of the incinerators in the country which emit large amounts of health harming air pollution. Two multinational corporations, Covanta and Wheelabrator, dominate the incinerator industry, relying on large public sanitation contracts and energy subsidies to remain profitable. However, incinerators face increasing scrutiny and community opposition as cities and states advance zero waste alternatives to incineration. More than thirty plants have closed in the last twenty years largely due to economic conditions like the loss of waste volume. The incineration industry must also deal with tight competition for tipping fees, and tight profit margins that are vulnerable to abrupt changes in waste or electricity markets. Additionally, these facilities are experiencing rising operation and maintenance costs as they reach the end of their 30-year life expectancy.

Incinerators emit significant amounts of air pollutants that can contribute to overall environmental and public health risks. Despite the existence of environmental regulations, state and federal regulatory agencies tasked with protecting human health are not doing enough to monitor and regulate this industry. Some of the largest emitters of air pollutants among the MSW incinerators in the U.S. are located in EJ communities. Finally, the relationship between incinerators and environmental justice communities reveals the disproportionate impact that this industry has on the most overburdened areas of the country who contribute the least, proportionately, to the waste problem. In the last year alone, two more incinerators were shuttered, in Detroit and Commerce. These facility closures reflect the power of environmental justice communities to advance the case against incineration and the impending decline of MSW incinerators in the U.S.

Endnotes

1 At this report was written, Detroit Renewable Power announced imminent closure of one of the dirtiest MSW incinerators in the country built in an environmental justice community in 1989. This report will refer to 73 MSW incinerators, acknowledging that there are now 72 incinerators left. Detroit Renewable Power cited lack of sufficient funds as their reason for closure.

2 For more information on recent closures, please refer to the end of Chapter 2.

3 Environmental justice communities are commonly identified as those where residents are predominantly minorities or low-income; where residents have been excluded from the environmental policy setting or decision-making process; where they are subject to a disproportionate impact from one or more environmental hazards; and where residents experience disparate implementation of environmental regulations, requirements, practices and activities in their communities. (California Energy Commission, "Environmental Justice," Accessed April 9, 2019).

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243 Toxic Release Inventory: U.S. EPA's Toxic Release Inventory requires U.S. facilities in different industry sectors to report annually how much of each chemical is released to the environment and/or managed through recycling, energy recovery and treatment. TRI is meant to inform the public about information

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Agencies use the FRV Determination Date and the Resolved Date to display an FRV date range in ECHO. These include Alabama, Florida, Massachusetts, and North Carolina.

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APPENDIX A: List of 73 MSW Incinerators in the U.S.

***Red highlight indicates incinerators located in an Environmental Justice community.**

Name	City, State	Operator	Initial Operation Year
Alexandria/Arlington Resource Recovery	Alexandria, VA	Covanta	1988
Arnold O. Chantland Resource Recovery Plant	Ames, IA	City of Ames	1975
Babylon Resource Recovery	West Babylon, NY	Covanta	1989
Barron County Waste-to-Energy & Recycling	Almena, WI	ZAC Inc	1986
Bay County Waste-to-Energy Facility	Panama City, FL	Engen	1987
Bristol Resource Recovery Facility	Bristol, CT	Covanta	1988
Connecticut Solid Waste System Resource Recovery	Hartford, CT	NAES Corporation	1987
Covanta Camden Energy Recovery Center	Camden, NJ	Covanta	1991
Covanta Hempstead	Westbury, NY	Covanta	1989
Covanta Plymouth Renewable Energy	Conshohocken, PA	Covanta	1982
Covanta Tulsa Renewable Energy	Tulsa, OK	Covanta	1986
Delaware Valley Resource Recovery	Chester, PA	Covanta	1992
Detroit Renewable Power	Detroit, MI	Detroit Renewable Energy	1989
Dutchess County Resource Recovery	Poughkeepsie, NY	Wheelabrator	1987
Ecomaine Waste-to-Energy	Portland, ME	ecomaine	1988
Essex County Resource Recovery	Newark, NJ	Covanta	1990
Hampton-NASA Steam Plant	Hampton, VA	City of Hampton	1980
Haverhill Resource Recovery	Haverhill, MA	Covanta	1989
Hennepin Energy Resource Center	Minneapolis, MN	Covanta	1989
Hillsborough County Resource Recovery	Tampa, FL	Covanta	1987
Honolulu Resource Recovery Venture	Kapolei, HI	Covanta	1990
Huntington Resource Recovery	East Northport, NY	Covanta	1991
Huntsville Waste-Energy	Huntsville, AL	Covanta	1990
I-95 Energy/Resource Recovery	Lorton, VA	Covanta	1990
Indianapolis Resource Recovery	Indianapolis, IN	Covanta	1988
Kent County Waste-to-Energy	Grand Rapids, MI	Covanta	1990
Lake County Resource Recovery	Okahumpka, FL	Covanta	1991
Lancaster County Resource Recovery	Bainbridge, PA	Covanta	1991
Lee County Resource Recovery	Fort Myers, FL	Covanta	1994
MacArthur Waste-to-Energy	Ronkonkoma, NY	Covanta	1990
Marion County Solid Waste-to-Energy	Brooks, OR	Covanta	1987
McKay Bay Refuse-to-Energy	Tampa, FL	Wheelabrator	1985
Miami-Dade County Resource Recovery	Doral, FL	Covanta	1982
Mid-Maine Waste Action Corporation	Auburn, ME	Mid-Maine Waste Action Corp	1992
Montgomery County Resource Recovery	Dickerson, MD	Covanta	1995
Niagara Falls Resource Recovery	Niagara Falls, NY	Covanta	1980
Olmsted Waste-to-Energy	Rochester, MN	Olmsted County	1987
Onondaga Resource Recovery	Jamesville, NY	Covanta	1995
Oswego County Energy Recovery	Fulton, NY	Oswego County	1986
Palm Beach Renewable Energy #1	West Palm Beach, FL	Covanta	1989
Palm Beach Renewable Energy #2	West Palm Beach, FL	Covanta	2015
Pasco County Solid Waste Resource Recovery	Spring Hill, FL	Covanta	1991

APPENDIX A: Continued

Name	City, State	Operator	Initial Operation Year
Penobscot Energy Recovery Company	Orrington, ME	ESOCO	1988
Perham Resource Recovery	Perham, MN	Prarie Lakes Municipal Solid Waste Authority	1986
Pinellas County Resource Recovery	St. Petersburg, FL	Covanta	1983
Pioneer Valley Resource Recovery	Agawam, MA	Covanta	1988
Pittsfield Resource Recovery	Pittsfield, MA	Covanta	1981
Polk County Solid Waste Resource Recovery	Fosston, MN	Polk County	1988
Pope/Douglas Waste-to-Energy	Alexandria, MN	Pope/Douglas Solid Waste Joint Powers Board	1987
SEMASS Resource Recovery	West Wareham, MA	Covanta	1988
Southeast Resource Recovery	Long Beach, CA	Covanta	1988
Southeastern Connecticut Resource Recovery	Preston, CT	Covanta	1991
Spokane Waste-to-Energy	Spokane, WA	City of Spokane	1991
Stanislaus County Resource Recovery	Crows Landing, CA	Covanta	1989
Susquehanna Resource Management Complex	Harrisburg, PA	Covanta	1972
Union County Resource Recovery	Rahway, NJ	Covanta	1994
Wheelabrator Baltimore	Baltimore, MD	Wheelabrator	1985
Wheelabrator Bridgeport	Bridgeport, CT	Wheelabrator	1988
Wheelabrator Concord	Penacook, NH	Wheelabrator	1989
Wheelabrator Falls	Morrisville, PA	Wheelabrator	1994
Wheelabrator Gloucester Company	Westeville, NJ	Wheelabrator	1990
Wheelabrator Hudson Falls	Hudson Falls, NY	Wheelabrator	1991
Wheelabrator Lisbon	Lisbon, CT	Wheelabrator	1995
Wheelabrator Millbury	Millbury, MA	Wheelabrator	1987
Wheelabrator North Andover	North Andover, MA	Wheelabrator	1985
Wheelabrator Portsmouth	Portsmouth, VA	Wheelabrator	1988
Wheelabrator Saugus	Saug, MA	Wheelabrator	1975
Wheelabrator South Broward Inc.	Fort Lauderdale, FL	Wheelabrator	1991
Wheelabrator Westchester	Peekskill, NY	Wheelabrator	1984
Xcel Energy French Island Generating Station	La Crosse, WI	Xcel Energy	1988
Xcel Energy- Red Wing Steam Plant	Red Wing, MN	Xcel Energy	1987
Xcel Energy- Wilmarth Plant	Mankato, MN	Xcel Energy	1987
York County Resource Recovery Center	York, PA	Covanta	1989

APPENDIX B: Cost Calculations for Average Annual Operation & Maintenance Costs for MSW Incinerators

SOURCE	CALCULATION	ESTIMATE OF O &M (ANNUAL $)
World Bank estimates for median size incinerator based on tonnage & fees	• Median size MSW incinerator = 1,050 tons of waste/day • World Bank average annual operating costs for an incinerator = $44 to $55 per ton of waste • 1,050 ton-per-day facility costs ~$17 million to $21 million annually to operate • Calculation: 1,050 tons per day of waste X $44 or $55/ton X 365 days	(1,050 tons/day x 365 days x $44-$55/ton) = **$17 million - $21 million**
U.S. EIA estimates of waste burning costs per kilowatt-year	• Waste burning costs (2013 estimate) $392.82 per kilowatt-year in fixed operating & maintenance cost. • Median gross capacity of electricity production of MSW incinerators = 61 MW • $392.82 MW-year X 61 MW ~ roughly $24 million in operation costs per year	$392,820 X 61 MW = **$24 million**
York County Resource Recovery Facility	• 1,344 tons/year capacity • $62/ton = Tipping fee • 42 MW/year = electricity sales • O & M reported = $20,440,360	Publicly available financial records **$20,440,360**

APPENDIX C: Incinerator Tip Fee Sources

State	Incinerators	Tip Fee	Source of Tip Fees
AL	Huntsville Waste-Energy	$40.00	Ulloa et al, [report], 2019
CA	Stanislaus County Resource Recovery	$39.00	Government Technology, [article], 2015
	Southeast Resource Recovery	$80.00	City of Long Beach, CA, [article], 2018
CT	Wheelabrator Lisbon	$65.00 - $75.00	Town of Lisbon, CT, [report], 2011
	Wheelabrator Bridgeport	$60.00	City of Bridgeport, CT, [report], 2018
	CT Solid Waste System Resource Recovery	$72.00	Hartford Courant, [article], 2018
FL	Wheelabrator South Broward Inc.	$64.21	Golden Beach, FL, [document], 2019
	Pinellas County Resource Recovery	$37.50	Pinellas County, FL, [website], 2019
	Pasco County Solid Waste Resource Recovery	$59.30	Lee County, FL, [report], 2018
	Palm Beach Renewable Energy #1	$42.00	Lee County, FL, [report], 2018
	Palm Beach Renewable Energy #2	$42.00	Lee County, FL, [report], 2018
	McKay Bay Refuse-to-Energy	$71.00	City of Tampa, FL, [document], 2019
	Lee County Resource Recovery	$50.20 - $67.45	Lee County, FL, [report], 2018
	Miami-Dade County Resource Recovery	$62.67	Miami-Dade County, FL, [website], 2019
	Hillsborough County Resource Recovery	$69.40	Lee County, FL, [document], 2018
HI	Honolulu Resource Recovery Venture	$45.00	City and County of Honolulu, [report], 2016
IA	Arnold O. Chantland Resource Recovery Plant	$55.00	City of Ames, IA, [report], 2016
IN	N/A	N/A	N/A
MA	Wheelabrator North Andover	$69.54	Town of Waterton, MA, [document], 2014
	Wheelabrator Millbury	$67.99	Town of Northborough, MA, [report], 2017
	SEMASS Resource Recovery	$78.37	The Patriot Ledger, [article], 2018
	Haverhill Resource Recovery	$58.00	Town of Bedford, MA, [website], 2018
MD	Wheelabrator Baltimore	$50.00	Inst. for Local Self-Reliance, [report], 2017
	Montgomery County Resource Recovery	$60.00	Montgomery County, MA, [document], 2018
ME	Penobscot Energy Recovery Company	$81.50	CommonWealth, [document], 2018
	Mid-Maine Waste Action Corporation	$82.00	Sun Journal, [article], 2018
	ecomaine Waste-to-Energy	$73.00	Sun Journal, [article], 2018
MI	Kent County Waste-to-Energy	$55.00	Michigan Live, [report], 2017
	Detroit Renewable Power	$15.00- $25.00	Great Lakes Enviro. Law Ctr, [report], 2018
MN	Perham Resource Recovery	$80.00	Minn. Pollution Control Age., [report], 2012
	Pope/Douglas Waste-to-Energy	$98.00	Echo Press, [article], 2018

APPENDIX C: Continued

State	Incinerators	Tip Fee	Source of Tip Fees
	Olmsted Waste-to-Energy	$83.00-$108.31	Governmental Advisory Assoc. [report], 2012
	Hennepin Energy Resource Center	$85.00	Hennepin County, MN, [report], 2019
NH	Wheelabrator Concord	$64.00	Concord Monitor, [article], 2013
NJ	Wheelabrator Gloucester	$83.50	Town of Rockport, MA, [report], 2019
	Union County Resource Recovery	$107.00	Union Co. Utilities Authority, [website], 2018
	Essex County Resource Recovery	$130.55	Atlantic Co. Utilities Authority, [website], 2018
	Covanta Camden Energy Recovery Center	$68.68	Town of Berlin, New Jersey, [document], 2018
NY	Wheelabrator Westchester	$75.95	USA Today, [article], 2014
	Wheelabrator Hudson Falls	$62.00	Hamilton County, NY, [report], 2012
	Oswego County Energy Recovery	$75.00	Oswego County, [document], 2018
	Onondaga Resource Recovery	$95.00	Syracuse, [article], 2018
	Dutchess County Resource Recovery	$76.15	Dutchess County, NY, [report], 2017
OK	N/A	N/A	N/A
OR	N/A	N/A	N/A
PA	Delaware Valley Resource Recovery	$63.00	City of Philadelphia, [report], 2018
	Susquehanna Resource Management Complex	$85.00	Press & Journal, [article] 2016
	York County Resource Recovery Center	$62.00	YC Solid Waste Authority, [website] 2019
	Lancaster County Resource Recovery	$62.00	SWANA, [report], 2012
	Covanta Plymouth Renewable Energy	$59.76	The Inquirer, [article], 2019
VA	Wheelabrator Portsmouth	$62.00	The Virginia- Pilot, [article] 2018
	I-95 Energy/Resource Recovery	$66.00	Fairfax County, [report], 2018
	Alexandria/Arlington Resource Recovery	$49.42	City of Alexandria, VA, [report], 2012
WA	Spokane Waste-to-Energy	$107.53	City of Spokane, [website], 2019
WI	Xcel Energy French Island Generating Station	$62.00	La Crosse Solid Waste Dpt, [website], 2019

APPENDIX D: Pollutants and Related Health Impacts

Pollutant	Short Term Health Impacts	Long Term Health Impacts and High Exposure
Nitrogen Oxides (NOx)	Aggravates asthma, leading to respiratory symptoms, hospital admissions.[317] Causes coughing and choking, nausea, headache, abdominal pain, and difficulty breathing.[318]	Asthma and respiratory infections.[319] Very high exposure may cause death, genetic mutations, decreased female fertility, spasms, swelling of the throat, rapid pulse, and dilated heart.[320]
Sulfur Dioxide (SO2)	Inflames and irritates the respiratory system and causes breathing difficulties especially during heavy physical activity.[321]	Reduces lung function and causes incidences of respiratory symptoms and diseases.[322] High concentrations can affect lung function, worsen asthma attacks, and worsen existing heart disease.[323]
Dioxins	The most harmful man-made toxins known to humans.[324] Causes poor liver and immune functioning, and neurological impairment.[325]	Causes cancer, reproductive and developmental problems, damage to the immune system, and interference with hormonal systems.[326]
Mercury	Neurological and behavioral disorders.[327] Symptoms include tremors, insomnia, memory loss, neuromuscular effects, headaches and cognitive and motor dysfunction.[328]	Overexposure may cause permanent neurological damage. [329] Toxic effects on the kidneys, nervous, digestive and immune systems, and on lungs, skin and eyes.[330]
Lead	Relatively low levels can disrupt normal development of the central nervous system, especially during fetal life and early childhood.[331] May cause miscarriage, stillbirths, and infertility.[332]	Can affect virtually every organ system.[333] Prolonged exposure may increase risk of high blood pressure, heart disease, and kidney disease. [334]
Particulate Matter >10 µm (includes PM10 and 2.5)	Deposits into the trachea and deeply into the lungs, irritates and corrodes the alveolar wall, and impairs lung functioning.[335] Causes aggravation of asthma, respiratory symptoms and an increase in hospital admissions.[336]	Overall mortality and mortality of lung cancer increases by 4%, 6% and 8%, respectively, for every 10 µg/m^3 PM2.5 increase.[337] Cardiovascular disease Respiratory disease

APPENDIX E: Dirty Dozen List Tables (2014)

Environmental justice communities are marked with a red square at the start of the row

Emissions data in the table below is sourced from the U.S. EPA ECHO Database

NITROGEN OXIDE MSW Incinerator	City	State	Tonnage per day	Tonnage per year	Nox emissions (2014, pounds)	NOx Rate (pounds per ton of waste)
Mid-Maine Waste Action Corporation	Auburn	ME	200	73000	563,885	7.72
Xcel Energy- Wilmarth Plant	Mankato	MN	720	262800	1,331,571	5.07
Lake County Resource Recovery	Okahumpka	FL	528	192720	950,783	4.93
Oswego County Energy Recovery	Fulton	NY	200	73000	341,157	4.67
Xcel Energy- Red Wing Steam Plant	Red Wing	MN	720	262800	1,226,000	4.67
Pasco County Solid Waste Resource Recovery	Spring Hill	FL	1,050	383250	1,615,941	4.21
Barron County Waste-to-Energy & Recycling	Almena	WI	90	32850	130,658	3.98
Wheelabrator Concord	Penacook	NH	500	182500	702,486	3.85
Pope/Douglas Waste-to-Energy	Alexandria	MN	240	87600	319,023	3.64
Covanta Plymouth Renewable Energy	Conshohocken	PA	1,216	443840	1,586,220	3.58
Wheelabrator Millbury	Millbury	MA	1,500	547500	1,871,826	3.42
Haverhill Resource Recovery	Haverhill	MA	1,650	602250	2,045,774	3.4

SULFUR DIOXIDE MSW Incinerator	City	State	Tonnage per day	Tonnage per year	SO_2 emissions (2014, pounds)	SO_2 Rate (pounds per ton of waste)
Hampton-NASA Steam Plant	Hampton	VA	240	87600	161,040.00	1.84
Barron County Waste-to-Energy & Recycling	Almena	WI	90	32850	42,250.90	1.29
Wheelabrator Millbury	Millbury	MA	1,500	547500	603,770.00	1.1
Wheelabrator Baltimore	Baltimore	MD	2,250	821250	621,703.00	0.76
Palm Beach Renewable Energy #1	West Palm Beach	FL	2,000	730000	491,910.62	0.67
Wheelabrator Concord	Penacook	NH	500	182500	113,259.48	0.62
SEMASS Resource Recovery	West Wareham	MA	3,000	1095000	647,847.60	0.59
Niagara Falls Resource Recovery	Niagara Falls	NY	2,250	821250	450,413.00	0.55
Wheelabrator Portsmouth	Portsmouth	VA	2,000	730000	398,981.58	0.55
Mid-Maine Waste Action Corporation	Auburn	ME	200	73000	35,986.98	0.49
Xcel Energy French Island Generating Station	La Crosse	WI	400	146000	65,811.60	0.45
Pope/Douglas Waste-to-Energy	Alexandria	MN	240	87600	39,136.10	0.45

APPENDIX E: Continued

LEAD MSW Incinerator	City	State	Tonnage per day	Tonnage per year	Lead emissions (2014, pounds)	Lead Rate (pounds per ton of waste)
Wheelabrator Hudson Falls	Hudson Falls	NY	500	182500	289.83	0.0016
Polk County Solid Waste Resource Recovery	Fosston	MN	80	29200	45.37	0.0016
Bay County Waste-to-Energy	Panama City	FL	500	182500	197.95	0.0011
Covanta Camden Energy Recovery Center	Camden	NJ	1,050	383250	380.00	0.0010
Essex County Resource Recovery	Newark	NJ	2,277	831105	631.80	0.0008
Wheelabrator Gloucester	Westeville	NJ	500	182500	95.20	0.0005
ecomaine Waste-to-Energy	Portland	ME	550	200750	80.20	0.0004
Wheelabrator Baltimore	Baltimore	MD	2,250	821250	293.93	0.0004
Hampton-NASA Steam Plant	Hampton	VA	240	87600	26.53	0.0003
Susquehanna Resource Management Complex	Harrisburg	PA	800	292000	77.20	0.0003
Mid-Maine Waste Action Corporation	Auburn	ME	200	73000	17.90	0.0002
Covanta Tulsa Renewable Energy	Tulsa	OK	750	273750	66.00	0.0002

MERCURY MSW Incinerator	City	State	Tonnage per day	Tonnage per year	Mercury emissions (2014, pounds)	Mercury Rate (pounds per ton of waste)
Babylon Resource Recovery	West Babylon	NY	750	273750	319.79	0.001168
Hampton-NASA Steam Plant	Hampton	VA	240	87600	21.29	0.000243
Wheelabrator Hudson Falls	Hudson Falls	NY	500	182500	26.00	0.000142
Pinellas County Resource Recovery	St. Petersburg	FL	3,150	1149750	134.89	0.000117
Barron County Waste-to-Energy & Recycling	Almena	WI	90	32850	3.83	0.0001165
Bay County Waste-to-Energy	Panama City	FL	500	182500	18.16	0.0000995
Dutchess County Resource Recovery	Poughkeepsie	NY	450	164250	15.96	0.0000869
Susquehanna Resource Management Complex	Harrisburg	PA	800	292000	25.40	0.00006714
Essex County Resource Recovery	Newark	NJ	2,277	831105	55.80	0.0000641
Wheelabrator Baltimore	Baltimore	MD	2,250	821250	52.68	0.000064
MacArthur Waste-to-Energy	Ronkonkoma	NY	486	177390	11.36	0.0000597
Marion County Solid Waste-to-Energy	Brooks	OR	550	200750	12.00	0.0000562

APPENDIX E: Continued

PM2.5 MSW Incinerator	City	State	Tonnage per day	Tonnage per year	PM2.5 emissions (2014, pounds)	PM2.5 Rate (pounds per ton of waste)
Wheelabrator Gloucester	Westeville	NJ	500	182500	70,463.00	0.39
Olmsted Waste-to-Energy	Rochester	MN	400	146000	31,577.00	0.22
Delaware Valley Resource Recovery	Chester	PA	2,688	981120	201,191.11	0.21
Wheelabrator Falls	Morrisville	PA	1,500	547500	108,230.44	0.2
Essex County Resource Recovery	Newark	NJ	2,277	831105	153,748.40	0.18
Palm Beach Renewable Energy #1	West Palm Beach	FL	2,000	730000	133,364.59	0.18
Honolulu Resource Recovery Venture	Kapolei	HI	3,000	1095000	182,757.22	0.17
Pinellas County Resource Recovery	St. Petersburg	FL	3,150	1149750	191,063.17	0.17
Covanta Camden Energy Recovery Center	Camden	NJ	1,050	383250	59,094.80	0.15
Montgomery County Resource Recovery	Dickerson	MD	1,800	657000	98,760.26	0.15
Lancaster County Resource Recovery	Bainbridge	PA	1,200	438000	57,033.04	0.13
Spokane Waste-to-Energy	Spokane	WA	800	292000	33,400.00	0.11

PM10 MSW Incinerator	City	State	Tonnage per day	Tonnage per year	PM10 emissions (2014, pounds)	PM10 Rate (pounds per ton of waste)
Wheelabrator Gloucester	Westeville	NJ	500	182500	70,472.00	0.39
Palm Beach Renewable Energy #1	West Palm Beach	FL	2,000	730000	233,481.65	0.32
Olmsted Waste-to-Energy	Rochester	MN	400	146000	34,562.30	0.24
Pinellas County Resource Recovery	St. Petersburg	FL	3,150	1149750	248,555.57	0.22
Wheelabrator Falls	Morrisville	PA	1,500	547500	117,515.00	0.21
Honolulu Resource Recovery Venture	Kapolei	HI	3,000	1095000	207,877.43	0.19
Essex County Resource Recovery	Newark	NJ	2,277	831105	153,750.40	0.18
Susquehanna Resource Management Complex	Harrisburg	PA	800	292000	51,696.80	0.18
Montgomery County Resource Recovery	Dickerson	MD	1,800	657000	102,090.80	0.16
Covanta Camden Energy Recovery Center	Camden	NJ	1,050	383250	59,094.80	0.15
Spokane Waste-to-Energy	Spokane	WA	800	292000	41,600.00	0.14
Bay County Waste-to-Energy	Panama City	FL	500	182500	25,131.29	0.14

APPENDIX E: Continued

Carbon Monoxide MSW Incinerator	City	State	Tonnage per day	Tonnage per year	CO emissions (2014, pounds)	CO Rate (pounds per ton of waste)
Palm Beach Renewable Energy #1	West Palm Beach	FL	2,000	730000	1,278,240.83	1.75
Bay County Waste-to-Energy	Panama City	FL	500	182500	298,058.13	1.63
Wheelabrator Hudson Falls	Hudson Falls	NY	500	182500	201,226.82	1.1
Miami-Dade County Resource Recovery	Doral	FL	4,200	1533000	1,532,163.55	1
Dutchess County Resource Recovery	Poughkeepsie	NY	450	164250	160,557.00	0.98
Xcel Energy- Wilmarth Plant	Mankato	MN	720	262800	234,146.38	0.89
Mid-Maine Waste Action Corporation	Auburn	ME	200	73000	58,108.53	0.8
SEMASS Resource Recovery	West Wareham	MA	3,000	1095000	777,220.60	0.71
Southeastern Connecticut Resource Recovery	Preston	CT	669	244185	166,789.51	0.68
Connecticut Solid Waste System Resource Recovery	Hartford	CT	2,850	1040250	692,894.45	0.67
Hampton-NASA Steam Plant	Hampton	VA	240	87600	54,664.65	0.62
Wheelabrator Portsmouth	Portsmouth	VA	2,000	730000	448,816.25	0.61

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“Deliver or pay”, or how waste incineration causes recycling to slow down

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Written by
SARA MUZNIK

Is incineration compatible with recycling?

A common argument in the past has been that we could recycle as much of the waste as possible and burn what remains. In reality, however, incineration can discourage recycling. Here’s why.

Recycling VS incineration

In Europe, burning waste in the so-called “waste-to-energy” plants is an increasingly common practice. About a quarter of all municipal solid waste is burned in 450 incinerators in mostly central, northern and western European Member States. This practice is often presented as a sustainable option to manage municipal waste. But trash is not a renewable resource. Producing resources that end up as waste requires great amounts of energy, which can be saved by recycling the materials instead of extracting virgin ones. Recycling is also more profitable and creates more

jobs than incineration. From an environmental as well as social point of view, there is no doubt that recycling is the best method for dealing with waste.

How incineration is slowing recycling down

Incinerators are expensive to build, so in order to make profit and repay the investment costs, they need a guaranteed stream of waste. Therefore, “waste-to-energy” plants require municipalities to sign long-term contracts compelling them to deliver a minimum quantity of waste for 20 to 30 years, or pay fees to compensate the incinerator company for lost profits. With such contracts in place, municipalities commit to generating a certain amount of waste, instead of decreasing that amount while increasing their recycling rate.

A remarkable case from Italy illustrates this problem in practice. In 2002 a medium sized incinerator in Pietrasanta was built by the order of the regional government of Tuscany without the approval of the surrounding municipalities. The incinerator was managed by the private firm Veolia for 7,5 years, until it was shut down in July 2010. The main reason for closing down the plant was the violation of environmental standards, mostly due to inappropriate wastewater treatment.

However, during the operation, there was a conflict between the 6 municipalities and the plant management due to the “deliver or pay” contracts imposed by the regional government.

Municipalities had implemented a door-to-door separate collection scheme in line with European Waste Framework Directive in order to meet national recycling targets of 65%. Separate collection in the municipalities had therefore reduced the amount of dry waste below the minimum of 10 000 tons per year – the minimum quantity stated in the contracts. In response, the plant management demanded a total of 13 million Euros to be paid by the municipalities, which started a legal war between the years of 2010 to 2015. The municipalities, trapped in a lose-lose situation, ended up paying 5 million Euros of fines due to their unfulfilled contractual obligations, just because they had successfully implemented separate collection of waste.

Such cases are increasingly common around Europe, and especially in the United Kingdom.

In Nottinghamshire, the County Council refused separate food waste collection in order to avoid the fines for not meeting the waste delivery targets for incineration. Another City Council in Stoke-on-Trent has been fined for delivering lower than minimum waste tonnage levels at their local incinerator. In Derby the recycling rate have fallen from 42% to 31% over a course of a year, due to specific provisions on the composition of the waste in the contract with the waste burning plant, which encouraged the incineration of recyclable and compostable material.

The situation in the EU, and what the European Parliament can do about it

Because of misconceptions and sometimes poorly transparent decision making process, incineration is unfortunately still a common practice around Europe. Every year less than 40% of waste is recycled or reused, whereas up to 90% recycling should be attainable. Instead of selling the recyclables for reuse, which would be both economically and environmentally efficient, the “deliver or pay” contracts require municipalities to burn valuable resources. This approach is counter intuitive to the already modest European waste reduction targets for 2020. Not to mention that recycling saves massive amounts of CO_2 emission and can play an important role in meeting the objectives on climate change as set in the Paris Agreement. We currently burn 81 million tons of waste in EU every year. There is a potential to reduce the amount of waste to 25 million tons per year, if we implement proposed zero waste and circular economy plans, as many cities have already done. By 2030, EU’s incineration capacity could be reduced by 75% when all European cities would repair, reuse or recycle at least 85% of their materials, like Treviso does today. If we want to increase the recycling rate we need to stop financing incineration now.

The European Parliament is currently discussing the European Directive on Renewable Energy. RED II will be implemented in the following decade, influencing the choices of local policy makers and financial investors. It’s important that financial support for renewables is in line with the recommendations of Commission’s

communication on waste to energy to phase out support schemes for waste incineration. The European Parliament's Committee on Industry, Research and Energy (ITRE) will vote on the issue of renewable energy support schemes for energy from mixed wastes on November 28th.

Claims that recycling and incineration are compatible practices are misleading, as incineration stifles recycling. "Deliver or pay" contracts cause a lock-in effect and hamper efforts on reduction and separate collection, hence are in conflict with the European environmental objectives.*"Incinerators are expensive to build, so in order to make profit and repay the investment costs, they need a guaranteed stream of waste"*

ASX: DVP

ASX Announcement
31/10/2022

SEPTEMBER 2022 QUARTERLY REPORT

Develop outlines clear strategy to capitalise on surging demand for energy transition metals and underground mining skills

Highlights of the Quarter

- **Develop launched its five-year business plan which outlines a clear pathway for value creation**
  - **The strategy is based on production of energy transition metals and provision of underground mining services – both set for exponential demand growth over the coming years.**
  - **Exposure to key energy transition metals via ownership of two assets (currently Woodlawn and Sulphur Springs) in tier-one locations of Australia.**
  - **Aim to produce over 50,000 tonnes per annum of copper equivalent metal.**
  - **Mining Services: Poised to capitalise on the huge increase in demand and pricing for underground skills via team of world-class underground mining specialists**
  - **Leading global fund BlackRock comes on as a 5.2% substantial shareholder**
- **Delivered a robust Mineral Resource Estimate for the Woodlawn Underground Mine of 7.3Mt at 12.0% ZnEq[1]; Two thirds of the Resource is in the higher-confidence category of Measured and Indicated**
- **In addition to the Woodlawn Resource, the assessment identified 5.1Mt of mineralisation next to the historical underground workings, which had mined grades of ~15.9% ZnEq[1]**
- **Excavation of underground drilling platforms at Woodlawn is well advanced and will enable an extensive mine life extension and exploration drilling program to commence in the December quarter**
- **Delivered a substantial increase in Mineral Resources for Sulphur Springs Deposit to 13.8Mt @ 8.3% ZnEq[1]; Contained metal increased to 786Kt Zn, 153Kt Cu & 10.4Moz Ag**
  - **Substantial increase in geological confidence with ~90% of Resource in the Indicated category**
  - **Grade increases significantly: zinc up +50%, silver up +15%**
  - **The Resource is conservative because it is calculated on a net smelter return basis, or payable metal, making it the project's most robust Resource to date**
- **Updated Sulphur Springs mine plan and project cost studies are underway; All project approvals have been obtained**
- **Exploration drilling completed at Sulphur Springs, Kangaroo Caves and Evelyn in WA**
- **Develop's Mining Services division completed 1047m of underground development at Bellevue Gold, in line with the mining schedule and generating A$9.1M in contract revenue**

Develop (ASX: DVP) is pleased to report on a pivotal quarter for the Company, during which it outlined a clear strategy to create substantial value by capitalising on the forecast growth in demand for energy transition metals and underground mining skills.

Develop Managing Director Bill Beament said: “We have laid out a robust plan for creating significant value in two core areas where demand will exceed supply by a substantial margin.

“Woodlawn and Sulphur Springs zinc-copper projects provide us with two exceptional, high-growth assets in tier-one locations. These assets will produce the metals which will enjoy exceptionally favourable supply-demand fundamentals as the decarbonisation and energy transition themes accelerate.

“We will also benefit from the significant growth already underway in demand for underground mining services. The current supply deficit in this area will only increase and we have established an outstanding team who can provide these services.”

Mr Beament said Develop was entering what would be an extremely active period on all three fronts.

“At Woodlawn, the development of underground drilling platforms is progressing well,” he said.

“We published an updated JORC Resource based on our extensive review of the drilling data we inherited as part of the acquisition. The Resource is not only large and high-grade, but it is extremely robust due to the application of number of technical parameters during the estimation process.

“At Sulphur Springs, we also released an outstanding updated JORC Resource result which demonstrates that the project is well on track to becoming a significant producer of zinc and copper.

“This will be followed by a reserve update, optimised mine development plan, revised project costings and exploring funding options.

“Our underground services division continues to perform extremely well as part of the agreement we have at the Bellevue Gold Mine.

“The world-class team of underground specialists we have put together for the contract, combined with Bellevue’s highly experienced team, are adhering well to the contract schedule.”

Occupational Health, Safety and Environmental

Group loss time injury frequency rate “LTIFR” is 0.0 (injuries per million work hours), WA Underground metalliferous industry average is 2.4

Group total recordable incident frequency rate “TRIFR” is 0.0, WA Underground metalliferous average is 9.2.

There have been no major or reportable environmental or heritage incidents for the past quarter.

Woodlawn Zinc Copper Mine

Develop’s Woodlawn Mine is located in the world class Lachland Fold belt in NSW, 250km south-west of Sydney and 40km south of Goulburn. Historically, the Woodlawn Mine operated from 1978 to 1998 and processed 13.8Mt of ore from the Woodlawn open pit, underground and minor satellite deposits grading 15.9% ZnEq[1] (9.1% Zn, 1.6% Cu, 3.6% Pb, 0.5g/t Au and 74g/t Ag).

The Company recommenced exploration mining activities at the Woodlawn site during the September Quarter, with the initial focus on the assessment of the mines extensive infrastructure as well as commencing the underground development for the excavation of drill platforms.

The new drill platforms, coupled with existing cuddies will enable an extensive underground exploration drilling campaign to commence in the December quarter. The drilling strategy is aimed at converting Inferred Resources to Indicated, extending the mineralised lenses at depth/along strike and drill-testing recently-identified EM conductors.

The September quarter saw the continued arrival of several key management staff as well as the mobilisation of the final mining equipment required for this development work. Recruitment for additional mining, geology and maintenance staff also took place with high levels of interest in the mine from experienced personnel.

Re-establishment of surface pumping, explosives storage, electrical infrastructure and site security was also completed, with refurbishment of the coreyard infrastructure and contractor laydown underway.

Woodlawn Mineral Resource Update

The Company also released an updated Mineral Resource Estimate (MRE) for the Woodlawn (Underground) deposit (see ASX release 2 August, 2022).

The updated Woodlawn Underground MRE of 7.3Mt @ 12.0% ZnEq[1] (5.7% Zn, 1.8% Cu, 2.0% Pb, 44.9/t Ag & 0.6g/t Au) represent the most robust and resilient resource for the deposit to date and includes applying a NSR, geometallurgical domaining and Minable Stope optimisation (MSO) to fully elucidate the potential for economic extraction.

Approximately 65% of the MRE has been classified as Measured and Indicated, with the remaining resources in the Inferred category (Figure 1).

A further 5.1Mt of remnant mineralisation has been identified proximal to historic workings (excluded from updated MRE). The remnant mineralisation has potential to significant increase the projects economic outcomes, evaluation of this material is currently underway.

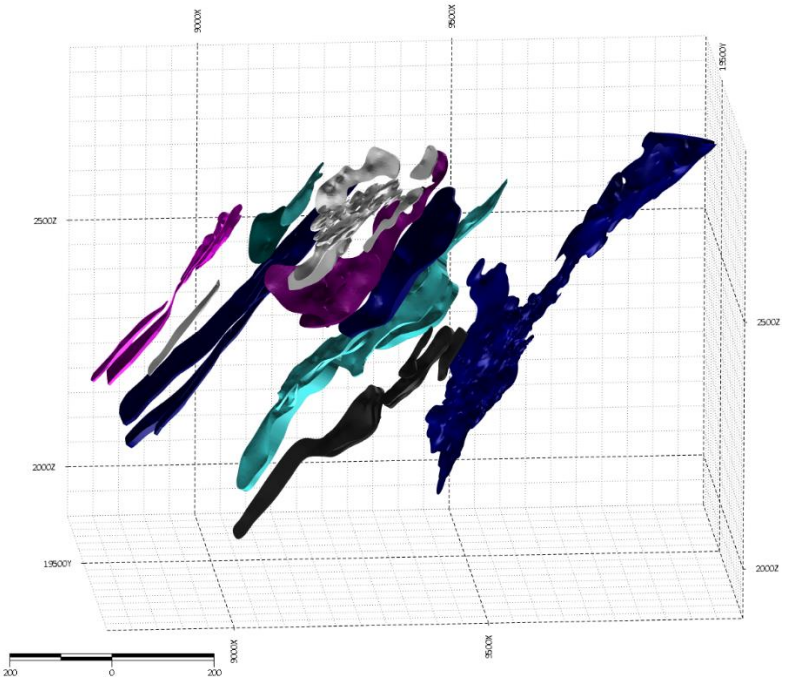

Figure 1. Woodlawn MRE oblique long section.

Sulphur Springs Zinc Copper Project

Develop's Sulphur Springs Project is located 112km south-east of Port Hedland in Western Australia and hosts a total Mineral Resource comprising 17.4 million tonnes grading 8.3% ZnEq[2] (5.8% zinc, 1.0% copper and 21g/t silver).

The Definitive Feasibility Study (see ASX release 10 October 2018) delivered a Pre-Tax NPV[8%] of A$472 million based on a copper price of US$6300/tonne and zinc price of US$2650/tonne. Current prices are significantly higher for both metals.

Sulphur Springs Environment Approvals

The Sulphur Springs Mining Proposal and Mine Closure Plan was approved by the Department of Mining, Industry Regulation and Safety on 15 August 2022 along with the granting of the Sulphur Springs works approval by Department of Water and Environmental Regulation on the 8 September 2022. Receipt of these two approvals, in combination with the approvals currently held, allows full regulatory implementation of the project.

Sulphur Springs Mineral Resource Update

During the quarter, the Company also released an updated Mineral Resource Estimate (MRE) for the Sulphur Springs deposit (see ASX release 6 September, 2022).

The updated Sulphur Springs MRE of 13.8Mt @ 8.3% ZnEq[2] (5.7% Zn, 1.1% Cu, 0.3% Pb, 22.5g/t Ag & 0.2g/t Au) is reported on the basis of a $80/t Net Smelter Return (NSR) and represents the most robust resource for the deposit to date, including geometallurgical domaining and recoveries to fully elucidate the potential for economic extraction.

The updated MRE has resulted in Indicated resources increasing from to ~68% to ~90% of the total (Figure 2). Step-out and exploration drilling has also highlighted ongoing potential for expansion to the known mineralisation.

The update MRE has also led to a 260Kt increase in contained zinc metal, a 2.4Moz increase in silver metal, with a decrease of 56Kt in contained copper metal.

The updated MRE further paves the way for an increased Reserve, optimised mine development plan, revised project costings and exploring numerous funding options which are all currently underway.

Resource Category	Metallurgical Domain	Tonnes (kt)	NSR ($A/t)[1]	Zn %	Pb %	Cu %	Ag g/t	Au g/t	Fe %
Indicated	Oxide	209	$381	0.3	0.1	4.2	18.9	0.1	29.8
	Transitional	6,655	$313	5.7	0.3	1.4	21.8	0.1	23.9
	Fresh	5,495	$289	5.8	0.3	0.9	22.0	0.1	21.0
	Sub-total	**12,360**	**$303**	**5.6**	**0.3**	**1.2**	**21.9**	**0.1**	**22.7**
Inferred	Fresh	1,401	$249	6.4	0.5	0.2	38.4	0.2	20.8
	Sub-total	**1,401**	**$249**	**6.4**	**0.5**	**0.2**	**38.4**	**0.2**	**20.8**
GRAND TOTAL		**13,760**	**$298**	**5.7**	**0.3**	**1.1**	**23.5**	**0.2**	**22.5**

Table 1: Sulphur Springs MRE by Resource category and metallurgical domain.
NSR reported at A$80/t cut-off. Tonnages are dry metric tonnes. Minor discrepancies may occur due to rounding.

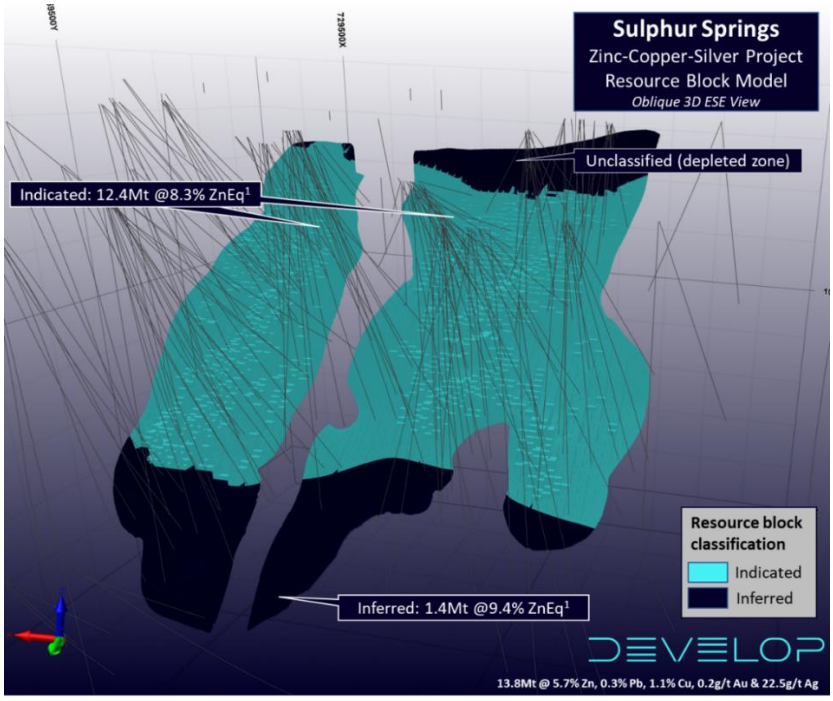

Figure 2. Sulphur Springs Resource block model classification.

Exploration

During the quarter the company commenced an ~5,500m Reverse-Circulation exploration drilling programme at the Sulphur Springs and Kangaroo Caves deposits. The programme was designed to test extensions to open mineralisation identified at the Trouser Leg and Eastern Lens targets during the updated Sulphur Springs MRE, and the down-plunge extension to mineralisation at Kangaroo Caves. Assay results from this drilling programme are expected in the March Quarter 2023.

Develop Underground Services Division

Develop continues to build strongly upon its first Underground Mining Services agreement with Bellevue Gold. The ~$A400 million agreement covers a period of almost four years for the construction, development and production activities at the underground mine.

During the quarter, the division completed 1047m of underground development at Bellevue Gold, in line with the mining schedule and generating A$9.1M in revenue. The site continues to build up its stores and maintenance inventory to achieve the significant project ramp up over the coming 12 months.

The achievement of the mining physicals under the agreement since May-22 at Bellevue is a credit to the quality of the operational team as the site has predominately only had the use of a second-hand mining fleet. Post the end of the quarter the site has been delivered a number of new mobile mining equipment pieces to continue delivering to the contract schedule. This equipment will allow for further improvements in mining physicals and contract revenues.

Also, post the end of the quarter the operation ramped up the workforce by 34%. Develop is not experiencing any issues attracting and retaining its work force. The next significant contract ramp up occurs in early 2023.

Whim Creek Join Venture (20% free carried)

Develop has a 20% free carried interest in the Whim Creek Base Metal project. During the quarter project partner Anax Metal Limited (ASX: ANX) released an update Mineral Resource Estimate for Salt Creek and drilling results for Evelyn (refer ASX September 12 and September 6, 2022). Highlights include:

- Updated Salt Creek MRE comprising.
    - 80% increase in indicated resources
    - 99% increase in contained copper
    - 22% increase in contained zinc
- Massive sulphide mineralisation intersected down plunge at the Evelyn Deposit

The company will continue to update shareholders as further results are released.

Evelyn Project [E47/1209]

A small ~450m exploration drilling programme was also completed at the 100% owned Evelyn Prospect in August 2022. The programme was designed to test several FLEM targets north of Anax Metals Evelyn Resource. Assay results from this drilling programme are expected in the December Quarter 2022.

Corporate

At the EGM held on 5 September 2022, Shareholders voted in favour of the acquisition of the Premium Group. Completion is expected in November.

The Company sold it’s redundant exploration camp at Spinifex Ridge for A$2.5 million and discharged its rehabilitation liabilities.

Securities Information

The Company's issued capital at the date of this announcement is:

Security Class	Issued Capital
DVP Fully Paid Ordinary Shares	161,906,472
Unlisted Performance Rights	1,000,333
Unlisted Options (various expiry dates and exercise prices)	48,966,688

Financial Information

The Company's cash position on 30 September 2022 was A$36 million.

Appendix 5B – Statement of Consolidated Cash Flows is provided in a separate report.

Information as disclosed in the Cash Flow Report:

- Exploration and Evaluation during the quarter was $172,000.
- There were no mining production and development activities during the quarter.
- Payments to related parties of the Company and their associates during the quarter was $146,000. The Company advises that this relates to executive directors' salaries, non-executive directors' fees, and superannuation**.**

This announcement is authorised for release by Bill Beament, Managing Director.

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About Develop

Develop (ASX: DVP) has a twin-pronged strategy for creating value. The first of these centres on the exploration and production of future-facing metals. As part of this, the Company owns the Sulphur Springs copper-zinc-silver project in WA's Pilbara region. This project is currently the focus of ongoing exploration to grow the inventory and various development studies. Develop also owns the Woodlawn zinc-copper project in NSW. Woodlawn, which is on care and maintenance, comprises an underground mine and a new processing plant. The second plank of Develop's strategy centres on the provision of underground mining services. As part of this, Develop has an agreement with Bellevue Gold (ASX: BGL) to provide underground mining services at its Bellevue Gold Project in

Interest in Mining Tenements

AREA OF INTEREST	TENEMENTS	GROUP INTEREST	EXPIRY
Sulphur Springs	M45/494	100%	21/10/2032
	M45/587	100%	6/09/2032
	M45/653	100%	28/09/2037
	M45/1001	100%	21/01/2029
	E45/4811	100%	27/03/2023
	E45/4993	100%	10/04/2023
	E 45/6033	100%	Application
	E 45/6034	100%	Application
	L45/166	100%	30/04/2030
	L45/170	100%	19/09/2030
	L45/173	100%	24/08/2033
	L45/179	100%	31/03/2032
	L45/188	100%	19/11/2030
	L45/189	100%	19/11/2030
	L45/287	100%	27/09/2033
	M45/1254	100%	10/10/2038
Evelyn	E47/1209*	100%	26/09/2021
Whim Creek Anax JV	M47/236	20%	26/07/2032
	E47/3495	20%	31/07/2022
	M47/237	20%	26/07/2032
	M47/238	20%	26/072032
	M47/443	20%	1/06/2040
	L47/36	20%	18/01/2023
	M47/323	20%	3/062035
	M47/324	20%	3/06/2035
	M47/1455	20%	3/04/2033
Woodlawn	S(C&PL)20	100%	16/11/2029
	EL7257	100%	14/11/2026
	EL8325	100%	2/12/2023
	EL8353	100%	17/03/2024
	EL8623	100%	17/07/2023
	EL8712	100%	5/03/2024
	EL8796	100%	25/09/2024
	EL8797	100%	25/09/2024
	El8945	100%	19/02/2023
	EL8318	20%	3/11/2023
	EL5878	20%	24/07/2023
Alchemy JV	EL7941	20%	23/05/2022
	EL8267	20%	12/05/2023
	EL8356	20%	12/05/2023
	EL8192	20%	30/10/2021
	EL8631	20%	26/07/2025
	EL8711	20%	5/03/2023
	EL7954	20%	19/06/2022
	EL8400	20%	20/10/2024
SKY Metal JV	EL8573	20%	23/05/2023
	EL8400	20%	20/10/2024
	EL8573	20%	23/05/2023

*The company has made an application for a 12-month extension of term on E47/1209 to DMIRS.

Mineral Resources Statements

SULPHUR SPRINGS PROJECT (DVP 100%)

SULPHUR SPRINGS

Classification	Tonnes (kt)	Zn %	Pb %	Cu %	Ag g/t	Au g/t
Indicated	12,359	5.6	0.3	1.2	21.9	0.1
Inferred	1,401	6.4	0.5	0.2	38.4	0.2
TOTAL	**13,760**	**5.7**	**0.3**	**1.1**	**23.5**	**0.2**

KANGAROO CAVES

Classification	Tonnes (kt)	Zn %	Pb %	Cu %	Ag g/t	Au g/t
Indicated	2,300	5.7	0.3	0.9	13.6	0.0
Inferred	1,300	6.5	0.4	0.5	18.0	0.0
Total	**3,600**	**6.0**	**0.3**	**0.8**	**15.0**	**0.0**

WOODLAWN (DVP 100%)

WOODLAWN

Classification	Tonnes (kt)	Zn %	Pb %	Cu %	Ag g/t	Au g/t
Measured	104	4.3	1.9	2.1	100	1.4
Indicated	4,776	5	1.8	1.8	42.2	0.7
Inferred	2,461	6.9	2.5	1.8	47.8	0.3
Total	**7,341**	**5.7**	**2**	**1.8**	**44.9**	**0.6**

WHIM CREEL JV PROJECT (DVP 20% : ANX 80%)

MONS CUPRI

Classification	Tonnes (kt)	Zn %	Pb %	Cu %	Ag g/t	Au g/t
Measured	1,070	1.6	0.7	1.5	38.0	0.3
Indicated	3,500	0.8	0.3	0.8	17.0	0.1
Inferred	500	1.5	0.6	0.5	14.0	0.0
Total	5,100	1.0	0.4	0.9	21.0	0.1

SALT CREEK

Classification	Tonnes (kt)	Zn %	Pb %	Cu %	Ag g/t	Au g/t
Indicated	1,017	3.3	0.9	1.2	20.0	0.2
Inferred	839	5.3	1.5	0.7	43.0	0.2
Total	**1,856**	**4.2**	**1.2**	**1.0**	**30.0**	**0.2**

WHIM CREEK

Classification	Tonnes (kt)	Zn %	Pb %	Cu %	Ag g/t	Au g/t
Indicated	1,760	0.6	0.2	1.1	6	0
Inferred	660	0.2	0.1	0.6	2	0
Total Cu Resources	2,420	0.5	0.1	0.9	5.0	0.0
Indicated	120	3.2	0.4	0.1	12.0	0.1
Inferred	45	2.5	0.4	0.1	9.0	0.0
Total Zn Resources	**165**	**3.0**	**0.4**	**0.1**	**11.0**	**0.1**

EVELYN

Classification	Tonnes (kt)	Zn %	Pb %	Cu %	Ag g/t	Au g/t
Indicated	443	3.9	0.3	2.4	40	0.9
Inferred	106	1.8	0.1	1.3	15	0.2
Total	**549**	**3.5**	**0.3**	**2.2**	**35**	**0.8**

Tonnages are dry metric tonnes. Minor discrepancies may occur due to rounding.

Note:

- *The Sulphur Springs Mineral Resource Estimate has been extracted from the Company's ASX announcements "Sulphur Springs Updated Mineral Resource Estimate" issued 6 September 2022.*
- *The kangaroo Caves Mineral Resource Estimate has been extracted from the Company's ASX announcements Kangaroo Caves Resource Upgrade" dated 22 September 2015*
- *The Woodlawn Mineral Resource Estimate has been extracted from the Company's ASX announcements "Woodlawn Updated Mineral Resource Estimate" issued 2 August 2022.*
- *The Mineral Resources estimates for The Whim Creek Project are based on information supplied by Joint-Venture Partner Anax Metals (ASX: ANX), full details of these estimates, including the applicable JORC statements, on their websites: https://anaxmetals.com.au*

*1.The zinc equivalent grades for Woodlawn (Zn Eq) are based on zinc, copper, lead and silver prices of US$3011/t Zinc, US$7690/t Copper, US$1900/t Pb, US$19.05/oz Silver and US$1654/oz Gold with metallurgical metal recoveries of 88% Zn, 70% Pb, 70% Cu, 33% Au and 82% Ag based on historical recoveries at Woodlawn and supported by metallurgical test work undertaken. The zinc equivalent calculation is as follows: Zn Eq = Zn grade% * Zn recovery + (Cu grade % *Cu recovery % * (Cu price $/t/ Zn price $/t)) + (Ag grade g/t /31.103 * Ag recovery % * (Ag price $/oz/ Zn price $/t)). It is the opinion of Develop Global and the Competent Person that all elements and products included in the metal equivalent formula have a reasonable potential to be recovered and sold.*

*2.The zinc equivalent grades for Sulphur Springs (Zn Eq) are based on zinc, copper and silver prices of US$3011/t Zinc, US$7690/t Copper and US$19.05/oz Silver with metallurgical metal recoveries of 93.6% Zn, 86.8% Cu and 46% Ag and are supported by metallurgical test work undertaken. The zinc equivalent calculation is as follows: Zn Eq = Zn grade% * Zn recovery + (Cu grade % *Cu recovery % * (Cu price $/t/ Zn price $/t)) + (Ag grade g/t /31.103 * Ag recovery % * (Ag price $/oz/ Zn price $/t)). It is the opinion of Develop Global and the Competent Person that all elements and products included in the metal equivalent formula have a reasonable potential to be recovered and sold.*

Competent Person Statement

The information contained relating to the Woodlawn Underground Mineral Resources is based on information compiled or reviewed by Ms Jillian Irvin of Entech Pty Ltd who is a Member of the Australian Institute of Geoscientists. Ms Irvin consents to the inclusion. Ms Irvin has sufficient experience relevant to the style of mineralisation, type of deposit under consideration and to the activity being undertaking to qualify as Competent Persons as defined in the 2012 – Refer Edition of the "Australasian Code for Reporting of Mineral Resources". Ms Irvin consents to the inclusion in this announcement of the matters based on this information in the form and context in which it appears. The Company confirms that it is not aware of any further new information or data that materially affects the information included in the original market announcement entitled 'Woodlawn Updated Mineral Resource Estimate' issued 2 August 2022 and, in the case of estimates of Mineral Resources, that all material assumptions and technical parameters underpinning the estimates in the relevant market announcement continue to apply and have not materially changed. To the extent disclosed above, the Company confirms that the form and context in which the Competent Person's findings are presented have not been materially modified from the original market announcement.

The information contained relating to the Sulphur Springs Mineral Resources is based on information compiled or reviewed by Ms Jillian Irvin of Entech Pty Ltd who is a Member of the Australian Institute of Geoscientists. Ms Irvin consents to the inclusion. Ms Irvin has sufficient experience relevant to the style of mineralisation, type of deposit under consideration and to the activity being undertaking to qualify as Competent Persons as defined in the 2012 – Refer Edition of the "Australasian Code for Reporting of Mineral Resources". Ms Irvin consents to the inclusion in this announcement of the matters based on this information in the form and context in which it appears. The Company confirms that it is not aware of any further new information or data that materially affects the information included in the original market announcement entitled 'Sulphur Springs Updated Mineral Resource Estimate' issued 6 September 2022 and, in the case of estimates of Mineral Resources, that all material assumptions and technical parameters underpinning the estimates in the relevant market announcement continue to apply and have not materially changed. To the extent disclosed above, the Company confirms that the form and context in which the Competent Person's findings are presented have not been materially modified from the original market announcement.

The information in this announcement that relates to Exploration Results is based on information by Mr Luke Gibson who is an employee of the Company. Mr Gibson is a member of the Australian Institute of Geoscientists and has sufficient experience with the style of mineralisation, type of deposit under consideration and to the activity being undertaking to qualify as Competent Persons as defined in the 2012 – Refer Edition of the "Australasian Code for Reporting of Mineral Resources". Mr Gibson consents to the inclusion in this announcement of the matters based on this information in the form and context in which it appears.

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The Mulloon Rehydration Initiative: The project's establishment and monitoring framework

By Luke Peel, Peter Hazell, Tony Bernardi, Stephen Dovers, David Freudenberger, Carolyn Hall, Donna Hazell, Walter Jehne, Leah Moore and Gary Nairn

The Mulloon Rehydration Initiative is a case study highlighting the challenges of integrating research into a catchment scale land-repair project involving multiple landowners and partners. Starting with an innovative project in 2006 to install 'leaky weirs' on a single property, the project has now expanded to include stream rehabilitation works on 16 properties and aims to cover an area of 23,000 ha of the Mulloon Creek, NSW and its main tributaries. Here, we describe the establishment phase of the project and the design of its monitoring framework.

Key words: *long-term research and monitoring, landscape rehydration, stream rehabilitation.*

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Introduction

The impacts of agriculture have been identified as having diminished the environmental and production qualities of Australian landscapes (Woods 1983; Jackson *et al*. 2016). Impacts include soil erosion, soil structural and nutrient decline, vegetation simplification and degradation, stream incision and gullying, loss of soil water-holding capacities and loss of native species and their habitats. In response, Australian governments, NGOs, landholders and community groups have invested significant resources in land rehabilitation and ecosystem restoration (Salt 2016). It is widely recognised that such projects lend themselves to long-term monitoring and research to test and inform efforts to address degradation in Australian Agricultural landscapes, but there are significant barriers to such long-term environmental research and monitoring (LTERM), for complex reasons. As a result, the ecological and social outcomes of thousands of these projects are often poorly documented; there are only a few well-designed and funded projects that explicitly integrate multi-dimensional monitoring (e.g. HLW 2020; ARI 2021). Monitoring integrated with scientifically peer-reviewed research and publication if it occurs at all is commonly an add-on rather than an integral project component.

Here, we document the process of establishing the Mulloon Rehydration Initiative (MRI) and describe the design of its monitoring framework. This is a significant, long-term (2006 - ongoing) catchment-scale stream and floodplain rehabilitation Initiative located in the Mulloon Creek catchment in the Southern Tablelands, NSW, Australia (Fig. 2). It is conducted under the auspices of the Mulloon Institute (Box 1). The Initiative's particular focus is to install in-stream structures (logs and rocks) and restore native riparian vegetation (Fig. 1). Our research aim is to monitor how these activities may improve hydrological, ecological, agricultural and social values derived from this catchment. These activities are based on the Natural Sequence Farming approach which aims to slow the drainage of rainfall, countering landscape drying that commenced with the clearing of catchments for agriculture (Williams 2010).

The context of this Initiative is complex, featuring as it does: contested management interventions; long-term landscape functions; diverse stakeholders; regulatory constraints; multiple environmental, social and production goals; strong community engagement and an interdependence of hydrological,

Ecological Management & Restoration *Linking science and practice*

Figure 1. The Mulloon Rehydration Initiative is restoring hydrological function along all degraded reaches of the Mulloon Creek integrated with a research and monitoring program. Seen here in the lower front is a typical 'leaky weir' (long arrow) and fenced riparian revegetation (short arrow).

geomorphological, production and ecological variables. The project is described and assessed against principles for LTERM derived from the literature, which largely draws on biodiversity-focused studies with a smaller range of parameters. These principles are extended to generate insights to inform further research activities and management interventions in production landscapes.

This paper proceeds in four parts. The first describes the origins of the Initiative, and the second, most extensive part describes the establishment of a long-term environmental research and monitoring program guided by features of successful LTERM programs distilled from the literature. The third part discusses lessons learned in this process, against the same features. The final part concludes the paper.

The Mulloon Rehydration Initiative

Commencing the project and starting conditions

The Mulloon Rehydration Initiative (MRI) had its genesis in 2005 during the Millennium drought. It was initiated by a project to rehydrate a 2.5-km section of creek at Mulloon Creek Natural Farms, within the 23,000 ha Mulloon Creek catchment, a tributary of the Shoalhaven River on the NSW Southern Tablelands (Fig. 2). The landowner, the late Tony Coote invited independent landscape thinker, Peter Andrews, to his farm after Andrews had recently featured on ABC TV's *Australian Story* program for his unconventional, and some would argue, controversial approach to landscape restoration based on the use of in-stream 'leaky weirs' to slow water flow and rehydrate landscapes (ABC 2018).

Coote and Andrews agreed to work together to regenerate the section of Mulloon Creek that flows through the farm, based on principles Andrews had been developing and promoting for many decades, called Natural Sequence Farming (Williams 2010). Such principles sought to reinstate natural landscape functions based on high levels of water infiltration and retention and that Andrews argued existed before European occupation of the Australian continent (Ripl 2003; Andrews 2006; Kravcik *et al*. 2007). Prior to clearing for agriculture commencing in the mid-1800s, much of the creek and adjacent floodplain was likely a discontinuous watercourse containing chains of ponds and swampy meadow complexes that allowed stream flow to be retained in the landscape for longer periods resulting in higher ecological productivity (Fig. 3a) (Johnson & Brierley 2006). By the 21st century, landscape clearing was causing

Box 1. The Mulloon Institute (TMI)

In 2011, Tony Coote and his wife, Toni, established *The Mulloon Institute* based at their Mulloon Creek Natural Farms with the vision of the farm becoming a living laboratory for long-term regenerative agricultural research and education. The Institute is a non-profit organisation with charitable status. A Board of Directors and an independent Scientific Advisory Council were appointed to oversee the operation and the scientific endeavours of the Institute.

The Institute's mission is "To actively demonstrate, validate and share landscape rehydration, restoration and regenerative practices in order to create sustainable, profitable and resilient agricultural and environmental systems now and into the future." The Institute's strategic goal is to rehydrate and restore 2.5 million hectares of land and positively impact the livelihoods of 5,000 farming families (https://themullooninstitute.org/mission-vision).

Following Tony's passing in August 2018, he bequeathed the farms to The Mulloon Institute to ensure the scientific research work would continue on the farms in perpetuity, a wonderful legacy to the nation.

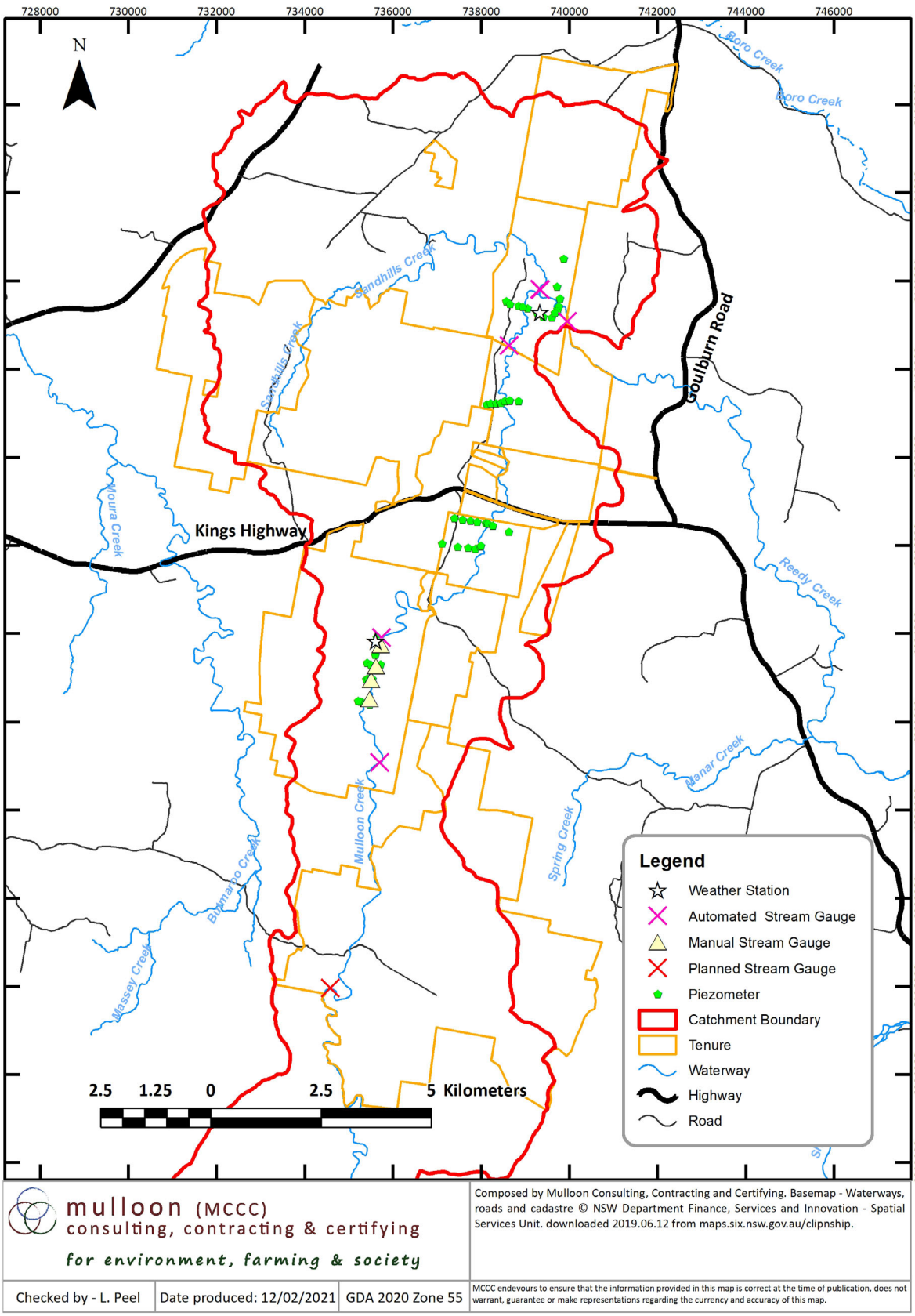

Figure 2. Mulloon Rehydration Initiative (MRI) is focused along the Mulloon Creek east of the town of Bungendore, southeast New South Wales, Australia. Core monitoring installations are shown and described in the legend.

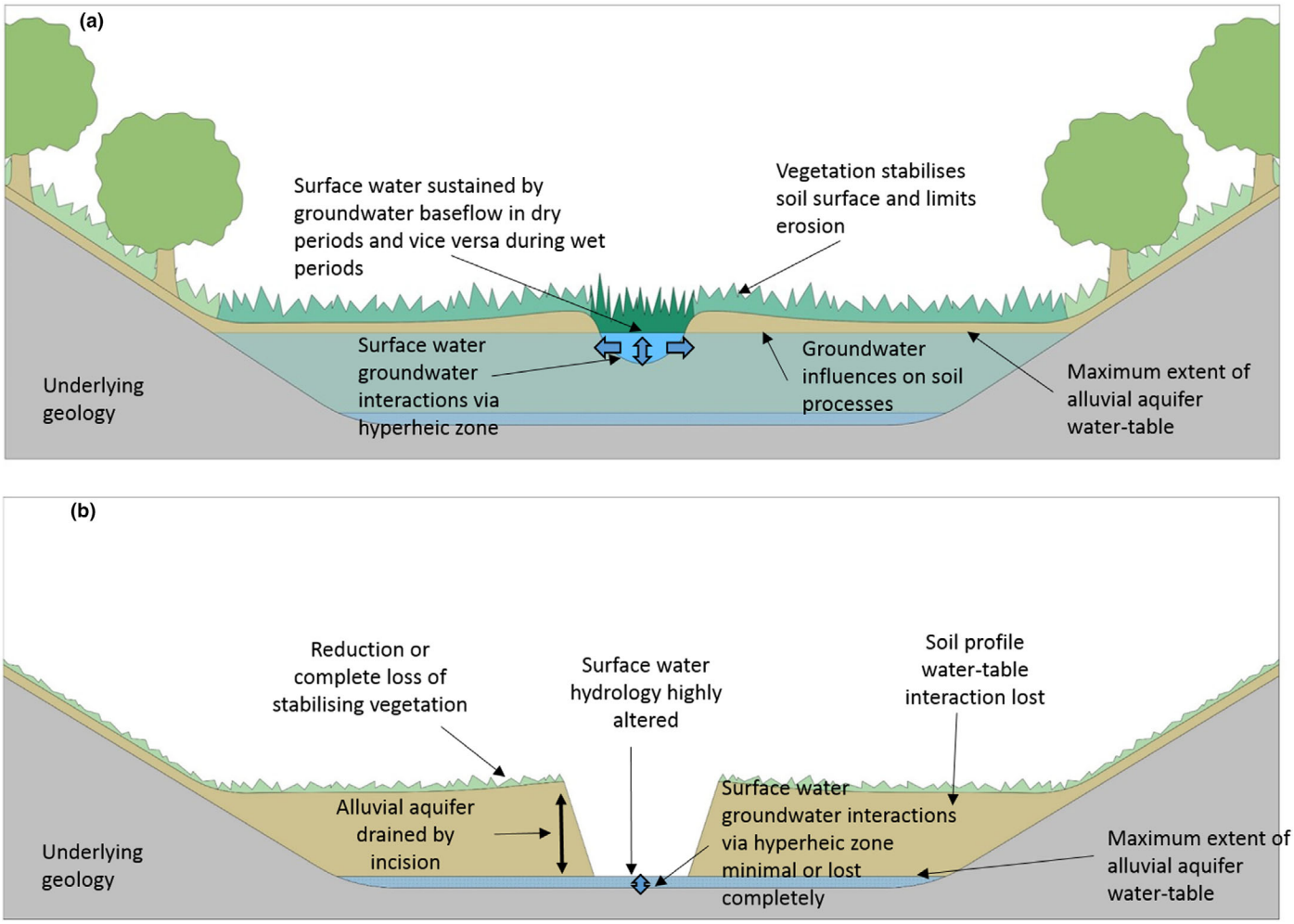

Figure 3. Schematic representation of an (a) intact versus (b) incised stream.

increased runoff and the creek had become a deeply incised single continuous channel throughout the entirety of the floodplain pocket, draining its catchment more rapidly (Fig. 3b). Andrews set out to functionally reconnect the stream with its adjacent floodplain (Fig. 3a).

In July 2005, the regional NRM body, Southern Rivers Catchment Management Authority (SRCMA) and the executive of the Upper Shoalhaven Landcare Council were approached by Coote to gain institutional and community support, and all the neighbours that bordered or bounded Mulloon Creek for a distance of 10-km downstream of Coote's property were approached. Aware of the public interest generated by Andrews's appearance on *Australian Story*, the Southern Rivers CMA agreed to support the project provided it had the backing of the neighbours and the upper Shoalhaven catchment landcare community. The CMA's role included engagement with other state and local government agencies, and supervision of project design, approvals and on-ground implementation. After support was gained from neighbours and the Upper Shoalhaven Landcare Council, albeit with concerns over the potential impact on stream flow, the CMA agreed to be the primary proponent of a National Landcare Program grant application to implement and monitor a Natural Sequence Farming demonstration within the Mulloon Creek floodplain on Coote's property.

Stage 1: Pilot works and initial responses

This first grant was awarded in January 2006 and work commenced in April 2006. Over a period of 8 weeks, undertaken in two phases in autumn and spring, a total of 42 log, rock, earth and vegetation structures were built along a 2.5-km stretch of creek and into several tributary gullies within the 100 ha floodplain pocket. The creek was also fenced from livestock. The structures were designed to slow water flow, raise the creek level, re-invigorate biological activity and ultimately re-establish the functional connection between the creek and adjacent floodplain. Figure 4 shows before and after views of one 2006 trial site.

Instream bed control structures, known as 'leaky weirs', were designed and installed by Andrews under the supervision of the CMA. Some conflict was encountered during this process. For example, Andrews expected to be able to spill pulses of water onto the floodplain during elevated flows to assist in achieving recharge of the floodplain aquifer. However, this was not possible due to regulatory constraints under the NSW *Water Management Act* 2000. The CMA, as the supervising authority, was also uncomfortable with the fluid approach to planning and implementation undertaken by Andrews. Coote was adamant that Andrews should be able to implement the project the way he saw fit.

Figure 4. (a) A typical degraded pilot site showing the incised stream bank and lack of ponding (March 2006). (b) The same reach ("Peter's Pond") by February 2018 showing extensive stream bank revegetation and extensive ponding following the establishment of a stone-based 'leaky weir'. (Photos by Peter Hazell).

The project was successfully implemented albeit with teething difficulties. From the perspective of Andrews and Coote, the key issue was that a full demonstration of the Natural Sequence Farming technique could not be implemented due to the prohibitive regulatory environment. From the perspective of the CMA, the soft engineering approach taken by Andrews posed an elevated risk of structure failure in the short term (SRCMA 2011). Both perspectives had some justification. Importantly, once the environment directly adjacent to the creek and gully structures stabilised with armouring vegetation, the stream rehabilitation process appeared to progress rapidly (Fig. 4b). However, repair of instream structures was required due to damage by high flows before armouring vegetation was sufficiently established. This was noted in a 2011 Southern Rivers CMA report on the trial of the Natural Sequence Farming approach to catchment repair (SRCMA 2011).

Some monitoring was established during this pilot project, including: stream and ground water hydrology; water quality; instream and riparian ecology; flora and fauna observations; a stream and floodplain geomorphology survey and a baseline photo log (SRCMA 2008). In the early stages, support from research organisations was difficult to secure, making it hard to establish comprehensive baseline monitoring for most of the variables. The CMA subsequently took on the monitoring role between 2007 and 2008. From 2008, researchers and students from the Australian National University (ANU) began to engage in monitoring ongoing changes post-treatment.

Stage 2: Expanding the scope

In 2014, under the auspices of The Mulloon Institute (Box 1), the small-scale demonstration project dramatically expanded into the Mulloon

Rehydration Initiative (MRI). This Initiative aims to be a catchment-wide landscape regeneration project experiment involving over 20 landholders, covering an area of 23,000 ha and stretching along over 50 km of the Mulloon Creek and its main tributaries (Sandhills and Shiel Creeks) (Fig. 2). The earlier concerns about the effect that multiple creek interventions would have on water availability downstream remained an issue amongst some neighbours. Despite this, all the neighbours bordering or spanning Mulloon Creek accepted the invitation to actively participate in the broader rehydration Initiative. A commitment that the Initiative would be scientifically monitored provided re-assurance. The science program is described below, and to November 2021 the MRI has implemented 16 individual projects during the past 6 years totalling over $4.7 Million of external funding (Table 1).

This funding has supported extensive on-ground activities as summarised in Table 2. It has been the experience of The Mulloon Institute that funding for outreach and education efforts, and to a lesser extent demonstration on-ground works, is more easily gained than for research and monitoring costs.

For the MRI, every in-stream work must be carefully planned and monitored, requiring a "Controlled Activity Approval" under the NSW *Water Management Act* 2000. Consent requires a detailed schematic, map and description of each leaky weir (Box 2, Figs. 5 and 6). Hardwood logs are sourced from the Forestry Corporation of NSW and rocks from a nearby licensed quarry. A vegetation management plan accompanies construction of all leaky weirs. The aim of the weirs is to correct the physical structure and function of the system. Rapid revegetation is critical in stabilising and transforming the system into one that is aggrading and not eroding. All sites are fenced to exclude livestock and revegetation uses a diversity of native trees, shrubs and rushes (see https://themullooninstitute.org/blog/2020/7/8/mulloon-rehydration-initiative-update?rq=species, with link to species list). Long-term responsibility for weed management is the responsibility of the landholder. The Mulloon Institute recommends that it is undertaken in accordance with the NSW Biosecurity Act 2015. Skilled earth moving equipment contractors are used for the construction of the leaky weirs, and the Green Army and volunteers have been involved in revegetation.

Establishment of the Larger MRI as a Long-Term Research Program

Unlike the pilot Natural Sequence Farming demonstration where monitoring was more ad-hoc, The Mulloon Institute aimed to establish the MRI as a long-term research project in landscape rehydration and regeneration with the key overarching question: *What is the effect of stream interventions on the ecology and farm productivity of the landscapes within the Mulloon catchment*? The MRI is not principally a research experiment, but rather an active, multi-participant restoration project supported by a research program to evidence impact and inform further initiatives.

A Science Advisory Committee was established by The Mulloon Institute to guide development of a monitoring and research plan linked to the on-ground activities. Members of the Committee are from various institutions, including the ANU, Universities of Canberra and Melbourne, NSW Department of Primary Industries, NSW Biodiversity Conservation Trust, South East Local Land Services and the Murray Darling Basin Authority. The Committee reports to The Mulloon Institute's Board, and works closely with the Institute's Project Coordinator, Research Coordinator and CEO. The Institute's Board chair is a member of the Science Advisory Committee.

The Institutes Science Advisory Committee adopted a framework based on Long-Term Environmental (Ecological) Research and Monitoring (LTERM). Such research has been recognised globally as essential to understanding changes in the state of the environment and the associated impacts of humans (both positive and negative). Researchers have acknowledged the importance of LTERM through the establishment of the International Long-Term Ecological Research Network in 1993 (Mirtl *et al*. 2018). There is a growing literature establishing LTERM as a research methodology that includes identifying critical success factors and providing guidance on design (e.g. Lindenmayer & Likens 2009, Lindenmayer *et al*. 2012, 2014; Youngentob *et al*. 2013; Burns *et al*. 2018). This literature provided the basis for establishing the research and monitoring associated with the MRI.

Lindenmayer *et al*. (2012, 2014) identified features that have been drawn from a wide body of literature and practice that are important for effective LTERM (Box 3); below, we describe how we are addressing these features (Note: the source uses the term LTER; we use the term LTERM to explicitly include monitoring).

LTERM Feature 1 – Conceptual system model

We applied a 'State and Transition' conceptual modelling approach developed by Westoby *et al*. (1989) for rangeland dynamics in Australia and used elsewhere (Bestelmeyer *et al*. 2017). The MRI broadened this modelling methodology following the approach used in the *South East Catchment Action Plan* (NSW Government 2014) that includes the Mulloon catchment. State and Transition models are used to describe and communicate the dynamics of a region's ecological systems but less often socio-economic conditions. These conceptual models describe the different states that can exist in a

Table 1. A list of the funded projects contributing to the Mulloon Rehydration Initiative (MRI) as of November 2021

Project title	Funder (grant)	Purpose	Duration (years)
Australian Government Programmes			
Rehydrate Australia to improve the environment, farm productivity and community education and engagement	Landcare-Smart Farms-Capacity Building Program; Australian Government-($3.86 M)	1) Comprehensive planning and scientific evaluation of hydrology, flora and fauna, and production improvements, including financial, social and community outcomes, linked to landholder support through on-farm training, workshops and educational materials.	2020–2025
Mulloon habitat restoration for threatened species (4 projects)	Australian Government - Green Army Programme ($38K)	Labour for on-ground works, particularly planting	2017–2018
Mulloon's habitat restoration for threatened species.	National Landcare Program - 20 Million Trees ($69K)	Establish 7,200 trees and 4,800 understorey plants on 30 hectares to improve the resilience and ecological connectivity of threatened species habitat along the creek corridor	2017
Outreach, engagement and uptake of landscape rehydration projects by agricultural communities	National Landcare Programme - Sustainable Agriculture Small Grants ($46K)	To engage with agricultural communities across the country to measure the uptake of the Mulloon Community Landscape Rehydration Project (now labelled the MRI).	2017
NSW Government Programs			
Landscape rehydration capacity building: developing curriculum	NSW Environmental Trust ($250K)	Development, in collaboration with two case study catchments, advanced level training course and materials.	2020–2025
Restoration of Mulloon Catchment to protect its ecosystems (Phase 1)	NSW Environmental Trust ($100 K)	Improve the condition of aquatic and terrestrial areas, including gullies and tributaries, creating wildlife corridors, vegetation linkage and habitat and food sources for flora and fauna.	2015–2018
Above (Phase 2)	NSW Environmental Trust ($100k)	To improve the creek condition and riparian functionality and habitat connectivity along Mulloon Creek linking Tallaganda National Park and Reedy Creek State protected lands.	2019–2022
Mulloon community landscape rehydration project	South East Local Land Services Rural Landscapes Program ($50k)	Improve water quality in the Sydney water catchment, via water quality monitoring and stream gauges.	2015
Benchmarking biodiversity within the Mulloon Community Landscape Rehydration Project (MRI) area	South East Local Land Services ($40K)	Baseline monitoring to assess biodiversity impacts	2015–2018
Mulloon Community Landscape Rehydration Project (the MRI): bringing the community along	South East Local Land Services ($87K)	Support groups within the local Landcare community for natural resource management and sustainable agricultural outcomes.	2016–2018
Philanthropic funding			
Mulloon watershed community project	Vincent Fairfax Family Foundation ($150 K)	Establishment funding for the MRI	2014–2015
Mulloon watershed community project	Veolia Mulwaree Trust ($34K)	Establishment funding for the MRI	2016
Project Governance	The Mulloon Institute	Maintenance of Board, Scientific Advisory Committee, CEO, core staff, etc	Long–term endowment
Generated income			
Income from various sources, as a registered not-for-profit	Various, with part income sequestered to TMI	Earnings from farming to TMI; income Mulloon Consulting; public donations; fees from training programs.	Ongoing

landscape. Through participatory processes, undesired and desired states can be identified, and socio-ecological processes that drive the transitions between states can be elucidated.

The value of these models is in fostering a general understanding, or theory of change, of how a system functions and responds to management actions and climatic events (Bestelmeyer *et al*. 2017). Causes of and constraints to change are often incompletely understood, but they can be tested by monitoring the effects of management and restoration actions. In this way, these models have helped us specify system uncertainties requiring research, have assisted our development of the monitoring program, are useful for specifying management objectives for sites and serve as our guide to maintain and restore ecosystem services.

Table 2. On-ground works of the MRI project as of November 2021. 'Leaky weirs' refers to in-stream structures designed to slow but not dam creek flow, with riparian restoration including fencing and revegetation

Activity	Number	Total area or length	Date completed	Funding agency
Leaky weirs installed on *Mulloon Creek Natural Farm*	14	2.5 km	Sept 2006	National Landcare Programme
Leaky weirs installed at *Mulloon Farm* (North) and revegetation	7	1.5 km	March 2018	NSW Environment Trust
Leaky weirs installed at *Mulloon Farm* (South referred to as *Mulloon*)	4	1 km	November 2018	NSW Environment Trust
Leaky weirs installed at *Westview* (also fencing, revegetation)	3	1 km	December 2018	NSW Environment Trust, LLS* Rural Landscapes Fund
Leaky weirs installed at *Palerang* (also fencing and revegetation)	15	3.5 km	December 2019	NSW Environment Trust, LLS* Rural Landscapes Fund
Leaky weirs installed at *Duralla* (also fencing and revegetation)	7	2 km	July 2020	National Landcare Programme 2
Planting of 12,000 native plants	12,000	20 ha	March 2018	20 million Trees
Planting of 2,000 native plants	2,000	5 ha	March 2019	NSW Save Our Species – Save the Scarlet Robin - Program

*Local Land Services, an NSW Government regional organisation, replaced the CMAs.

Figure 7, Box 4 and the detail in Appendix 1 synthesise our knowledge of the historical, undesired and desired conditions of the system. These show our current understanding of how the Mulloon catchment and its human communities have changed to the present time and how we predict it will respond in the future given current interventions. This conceptual synthesis is open to new information. We expect the catchment to go through multiple 'transition states' since restoration processes often take decades to yield our desired state, but we have left these possible transition states out of Figure 7 for brevity. Our State and Transition models are used for bridging the science–management divide and assist in communicating the objectives, timescales and drivers of change across the Mulloon catchment.

LTERM Feature 2 – Research questions and hypotheses

Our broadest question is: *What are effective management practices that restore catchment health across a range of ecosystem services and human values*? Our initial research is on the effectiveness of installing leaky weirs and restoring riparian vegetation along an entire catchment. Our broad hypotheses ($H_{1\text{-}4}$) include, based on our conceptual models above, that instream structures and riparian revegetation will:

- H_1 Significantly improve native floral and faunal diversity and abundance.
- H_2 Significantly improve measures of hydrological function at multiple scales.
- H_3 Be a catalyst for significant improvements in measures of farm productivity and profitability.
- H_4 Be a catalyst for improving community engagement and cohesion within the catchment linked to greater support from a broad range of stakeholders outside the catchment.

As individual research partners and projects develop, these questions and hypotheses will be refined, but we expect that most research projects can fit under one of more of these guiding questions and hypotheses linked to our conceptual model above.

H_1 and H_2 invite more typical scientific approaches, whereas H_3 expands the domain of data gathering and communication into the financial and production arenas. H_4 aims to fulfil the Institute's goal of informing future initiatives elsewhere, and focuses attention on social, regulatory and project management information and lessons, considerably expanding the envelope of the 'experiment'.

LTERM Feature 3 – Experimental design

Ideally, one would apply a rigorous and well-replicated Before-After-Control-Intervention (BACI) design (Underwood 1991) informed by decades of detailed 'before' data on all four dimensions shown in Figure 7 above – as well as collecting data from a comparable control catchment. However, such baseline data are generally not available due to the paucity of fine-scale monitoring in the Australian landscape; and comparable catchments are difficult to locate in the region. Rigorously controlled replication is also a challenge due to the complex environment of 20+ private landholders, all holding different management objectives and practices and across a catchment.

Nonetheless, as shown in Figure 5 and Table 2 above, in-stream structures (leaky weirs) and associated riparian revegetation ('interventions') are well replicated in space and time. Detailed pre-intervention data, such

Box 2. Plan and description for a typical MRI site (see Figure 5)

Site MN7 is a log sill with a sandbar rehydration channel. The site is bedrock controlled. Two rows of logs have been placed on top of each other in a v-notch configuration. They are pinned and keyed into the banks. Knitted brush and Poa tussock matting to 400 mm were placed underneath the logs for scour protection. Clay bank material and gravel was pushed up behind the logs. *Typha* and *Phragmites* were transplanted into the sediments behind the logs. A sandbar rehydration channel was cut into the sandbar adjacent to the left extent of the log sill at the spill height of the log sill. The left edge of the log sill and the rehydration channel entry was rock armoured to prevent scouring. Rock rip rap was placed around the edges of the sill to one metre up the bank and up to five metres downstream of the sill on the bank. Bedrock control will prevent scouring of the bed. The site around the structure and the rehydration channel was extensively planted with native reeds, sedges, shrubs and trees. Dimensions of this site include: 43 m top bank to top bank, 18.5 m width of control structure and 6 m^2 control structure cross-sectional area. This structure will impound approximately 0.17 ML of stream flow. An existing stream crossing 50 m upstream of MN7 will be inundated by up to 300 mm. The crossing was augmented with 100 mm of stream gravel to ensure continued utility.

as stream bank profiles (Fig. 6), are being collected. At this scale, assessing riparian changes through time directly associated with each weir is achievable. However, long-term monitoring of experimental controls (no interventions) is problematic. As more and more weirs are established, dependent on funding, then upstream or downstream local 'controls' may disappear because the aim of MRI is full riparian restoration along the entire Creek.

The lack of within-catchment controls would be offset by the establishment of controls at the catchment scale. At this point, we do not have the resources to establish a monitoring program in a nearby catchment with similar hydrology, condition and land use history. However, potential space and time comparisons could be feasible in the future should resources become available and a comparable catchment can be found without in-stream structures having been installed. While lack of such a control catchment will limit the project's potential to attribute catchment scale changes to improved management practices, we are nonetheless monitoring at the catchment scale to test our conceptual models to identify trajectories of change in the indicators of catchment health or functionality and to improve our understanding of catchment processes.

Ideally, we would also have sufficient resources to find and monitor a 'reference' catchment. That is, a catchment still in our 'Desired State' as described in Figure 7, Appendix 1 and Box 4. We are not aware of any existing catchment scale research that comprehensively documents the behaviour or performance of a comparable catchment in very good condition (least disturbed).

LTERM Feature 4 – Measurement of key entities

As shown in Figure 2 and Table 3, we have made substantial progress in implementing a broad monitoring program across our four dimensions of interest that is coordinated by the Institute's Research Coordinator (L. Peel). We expect other response variables to be measured as more research partners become involved with specific interests.

INSERT Table 3

LTERM Feature 5 – Data management and use

The Institute partnered with HydroTerra Pty Ltd to develop rigorous site descriptions, data capture and storage standards. Protocols have been established for field equipment installation, maintenance and repair. Meta-data requirements (rules by which data are described) have been developed in detail. Roles and responsibilities are documented. Procedures have been developed to facilitate accurate and transparent data sharing with collaborating researchers. System documents include: site description and management plans, DataStream™ manual, monitoring plans and system design and specifications.

LTERM Feature 6 – Scientific productivity (publishing)

LTERM projects often encounter a problem in regard to publishing papers prior to long-term results being available. This makes communication of research planning and baseline data, and thus the sharing of practical lessons, difficult. While an open-access policy on the part of the Institute has seen project reports, data, etc. made public (https://themullooninstitute.org/projects), this paper is the first peer-reviewed full publication at this relatively early stage in the MRI. Explicit, accessible communication of intervention plans assists the understanding of stakeholders regarding the nature of planned interventions (e.g. Fig. 8). This paper is analogous to Shorthouse *et al.* (2012) describing the logic of and early lessons from another key Australian LTERM initiative. We agree with Lindenmayer and Likens (2009) that our long-term monitoring program can be used as a

Figure 5. A typical Mulloon Rehydration Initiative site (WVM1): (a) pre-works; summer 2018; (b) site works summer of 2018; (c) initial recovery – autumn 2020. The hydrological and ecological outcomes of these constructions are the focus of on-going monitoring. Preliminary results to be published in the near future (Photos by Peter Hazell).

framework around which shorter term projects can be conducted. For specific, peer-reviewed outputs (e.g. post-intervention hydrological impacts), often up to a decade is needed for reliable data across stream morphology evolution, vegetation establishment phases and seasonal climatic variations.

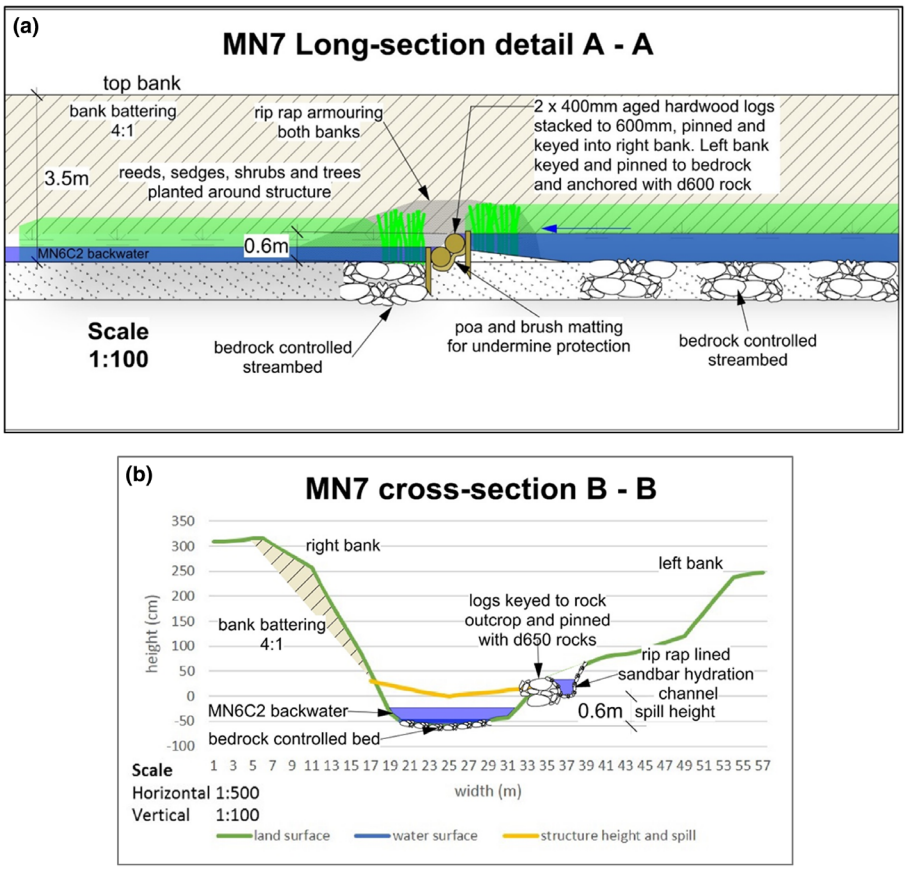

Figure 6. Detail of typical 'leaky weir' installation and revegetation. (a) Site MN7 long section AA, (b) Site MN7 cross-section B-B.

Box 3. Features of Effective Long-Term Ecological Research and Monitoring (LTERM) (adapted from Lindenmayer et al 2012, 2014)

1. A conceptual model of the system, developed and continually reviewed to guide question framing and experimental design.
2. Posing good questions/hypotheses: measurable, scientifically based and tractable, capable of testing management options, and can evolve with improved knowledge.
3. Sound experimental study design, including use of statistical expertise; early use of reference and control sites, long-term security of research sites.
4. Measurement of appropriate entities: guided by (1-3) above, to ensure that important factors and relationships are monitored.
5. Frequent use of data: high-quality record keeping and ongoing interrogation of data to assure quality and generate new questions and management possibilities.
6. Scientific productivity: publication of methods and results in academic and more widely accessible formats.
7. Project management: well-developed partnerships between scientific disciplines, policy makers and resource managers; early resolution of intellectual property issues; strong and enduring leadership; and succession planning; ongoing funding.
8. Research logistics: access to field equipment, qualified field staff and data storage and management capacities, established field and laboratory protocols.

LTERM Feature 7 – Project management

The Mulloon Institute recognises that effective leadership is pivotal to all features of successful monitoring programs as described by Lindenmayer *et al*. (2012, 2014). Through the governance structures of the Institute, the MRI provides comprehensive management of on-ground activities as well as guiding strategic programs, including the research program. The Institute employs a Research Coordinator, Project Coordinator and

Box 4. Catchment scale drivers of change between States as shown in Figure 6

Transition 1 – Drivers of change from the *Pre-Colonial State* to the *Undesired State*

- Displacement or extermination of Indigenous communities and extensive loss of their culture and knowledge
- Colonisation by large- and small-scale pastoralists and farmers
- Regional extermination of the dingo
- Introduction of grazing pressure by domestic livestock
- Introduction of foxes, cats and rabbits then the recent increase in deer and pig populations
- Deliberate and unintentional introduction of non-native pasture and weed species
- Extensive clearing of trees across the mid and lower catchment
- Cropping of alluvial valley and lower slopes
- Logging of the upper catchment
- Cycles of droughts and floods

Transition 2 – Drivers of change to a *Desired State*

- Construction of numerous leaky weirs and other structures that enable greater retention of water in the catchment
- Improved grazing management to promote diverse perennial pastures
- Riparian revegetation and fencing to control access to the creek
- Re-introductions of regionally extinct or rare species of flora and fauna
- Control of feral pest animals and invasive weeds
- Control of kangaroos to sustainable levels
- High levels of cooperation between landholders
- High levels of shared indigenous and non-knowledge
- Adaptive management based on sustained monitoring and research
- Sustained investment in landscape repair

Transition 3 – Drivers of change back to the *Undesired State*

- Failure to maintain the function of leaky weirs post-flood events
- Sustained over grazing by livestock
- Failure to maintain fencing to control access to the riparian zone
- Break down in cooperation between landholders
- Loss of landscape management knowledge due to landholder turnover
- Failure to adapt to changing conditions informed by research and monitoring
- Deliberate and unintentional introduction of non-native pasture and weed species
- Re-emergence of populations of foxes, cats, rabbits, deer and pigs

Hydrologist, as well as field officers that have time allocated to monitoring activities, along with communications and administrative staff who contribute to the initiative. The Science Advisory Committee meets quarterly. This Committee has been instrumental in setting appropriate questions, developing a workable conceptual model, resolving what to measure and guiding the research program described here. The Committee also established research

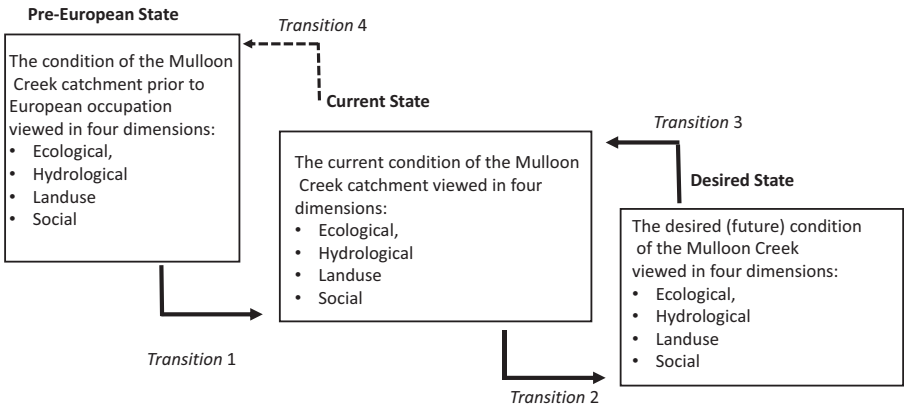

Figure 7. A broad outline of our State and Transition conceptual model for the Mulloon Creek Catchment. Appendix 1 and Box 4 provide more detail, such as the Transitions which drive State change. Transition 4 is shown as a dotted line as we consider a return to a pre-colonial State to be unlikely due to system wide shifts, including climate change and introduction of pest species unlikely to be exterminated at this scale.

collaborations ranging from small projects to formal partnerships with institutions, including the ANU and the universities of Canberra, Melbourne, UTS, RMIT, UNSW, Wollongong, CSU and WSU, along with the NSW Department of Primary Industries and commercial entities, including Cibo Labs and HydroTerra.

The Scientific Advisory Committee follows an adaptive monitoring process (Lindenmayer and Likens 2009). A key approach of the MRI research program is that question-setting, study design, data collection, data analysis and data interpretation are iterative processes. The Committee is committed to evolving the research program and associated monitoring in response to new information or new questions. New monitoring protocols and new technologies will no doubt need to be carefully adopted without losing consistency in baseline measures of catchment responses, extending the already detailed monitoring plan.

LTERM Feature 8 – Research logistics

As noted above under Feature 5 (Box 2), the MRI has established data management protocols, monitoring equipment specifications and data collecting procedures in partnership with HydroTerra Pty Ltd. These are to be adopted by all research partners including student researchers, and communicated to stakeholders and the wider public, with the stated principle of, where possible, maximising stakeholder access to data and research findings.

Discussion: Reflections on Applying LTERM in Production and Social Contexts

In the light of The Mulloon Institute and MRI experience over the past 15 years, we now revisit the LTERM features presented in Box 2 and described above. We have found these literature- and experience-derived features to be useful in shaping our approach to establishing an LTERM project at scale. However, as the literature recognises, contexts vary. The following summarises our lessons against these features, extending these based on experience in a cross-tenure, production landscape context with diverse stakeholders and motivations, regulatory constraints, community and landholder engagement and multiple ecological, hydrological, economic and social variables. The following proceeds through the 'Features of Effective LTERM' in Box 2:

1 *A conceptual model of the system*. We found a shared conceptual model of the system to be indispensable, and recommend the use of one that is based on established models from the formal literature (in this case, a State-Transition model). However, in complex situations, it is to be expected that (i) the model must be tested and adapted, and (ii) while relative simplicity is needed in a model, important ancillary matters will exist that cannot be captured by our simple conceptual model (Fig. 7 above). For example, our initial understanding of surface-ground water connectivity is likely to be overly simplistic. Preliminary data suggest that this connectivity is highly variable at finer scales than previously assumed. Capturing all variables would make for an impossibly complicated model unsuitable as an organising and communication device. Balance between conceptual soundness, communicability and detail linked to other sources is required.

2 *Posing good questions/hypotheses*. Without hypotheses, experimentation is impossible. A balance is needed between a smaller number of core questions that are rigorous, testable and able to be comprehended by diverse stakeholders, and what can become a 'shopping list' of endless research questions, and thus monitoring and data needs. In settings such as the MRI, the span including the hydrological, ecological,

Table 3. Catchment scale monitoring: key response variables and replication

Response variables	Method	Spatial Replication	Frequency	Commenced
Ecological				
Patchiness, indices of infiltration, soil surface stability and nutrient cycling	Landscape Function Analysis (Tongway and Hindley, 2004)	30+ transects stratified by land type	2-3 years	2015
Riparian vegetation condition	Rapid Appraisal of Riparian Condition (Jansen *et al*. 2005)	50+ transect sites	2 years	2017
Bird diversity	Point counts	17 transects	2 years	2015
Frog diversity	Frog calls in Spring	30+ transect sites, plus farm dams and wetlands	2 years	2017
Fish	Electro-backpack, with additional bait-less traps	6 in Mulloon, and 3 comparative sites in nearby similar montane creeks	3-5 years	2016
Aquatic invertebrates	Australian Rivers Assessment System (AUSRIVAS 2013)	6	2 years	2015-16
Hydrological				
Stream surface water height	Automated pressure sensors	6	Continuous	2007
Stream surface water	Automated sensors; Temp, pH, EC, ORP, turbidity and DO	6	Continuous	2017
Ground water level	Piezometers	80	Continuous	2007
Ground water quality	Temp, EC, pH	80	Continuous	2018
Climate	Automated Weather stations	2	Continuous	2006
Stream profile	Survey station	All leaky weir sites	Pre-construction and every 3 years	2015
Stream bank cross-sectional contours	Ground survey	All leaky weir sites	3 years	At time of construction of in-stream works
Soil moisture sensors	Automated Sentek with 6 sensors for each at 10, 30, 50, 70, 90, 120 & 150 cm depth	31	Continuous	2021
Productivity and economics				
Pasture biomass	Remote sensing (Cibo Labs Pty)	Whole of catchment	Continuous	2020
Farm profitability	Project staff and landholders	Participating landholders (voluntary)	TBC	TBC
Social dimensions				
Landholder participation	Raw count	Property owner and manager	continuous	2014
Landholder attitudes	Direct liaison, periodic survey	Property owner and manager	continuous	2014

production and social can produce unmanageable scope: care and clarity in framing questions is required, and a preparedness to focus on key, achievable research questions (see further below).

3 *Sound experimental study design*. Again, multi-tenure, -stakeholder, -value and -motivation contexts limit the possibility of straightforward experiments aimed at a few cause-effect links. For some participants, interventions that retain in-stream water and revegetated banks are sufficient, assuming that water quality and biodiversity will be co-benefits. Other participants, and regulatory requirements, demand water and soil chemistry monitoring and species surveys. Also, the lack of experimental controls at both site and catchment scales and comprehensive before-intervention data make experimental purity difficult, if not impossible. We acknowledge that the Mulloon Creek catchment is unique – lessons learned here may not be directly applied to other catchments. However, tools, such as the Hydrogeological Landscapes Framework (Moore *et al*. 2018), provide a means of exploring similarities and differences with other catchments. There is always a tension between investing in improving degraded catchments versus investing in research to understand how a least disturbed catchment functions. To date, The Mulloon Institute and its diversity of funders are focussing on restoration complemented by research and monitoring.

4 *Measurement of appropriate entities*. Features (2) and (3) (Box 2) notwithstanding, there is scope for later regret over not having captured a particular variable or process. A useful warning is "In 2050, which aspects of ecological change will we regret not having measured?" (Lindenmayer *et al*. 2015: 213.) As per above, for

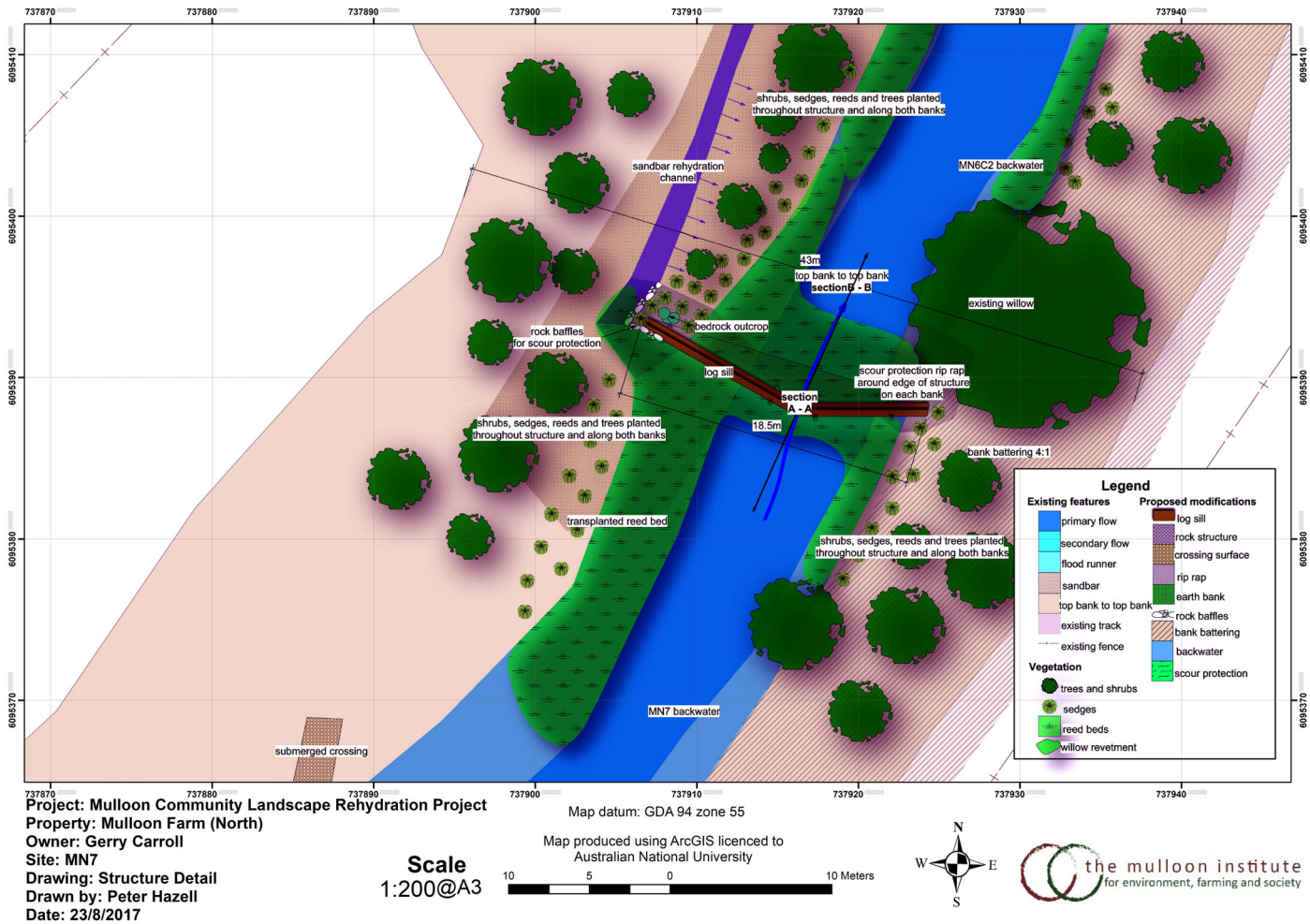

Figure 8. Aerial plan of a typical structure (leaky weir) and revegetation with locally native species (Site MN7).

multi-variable 'experiments' such as the MRI, a full research plan and monitoring system invites an unachievable array of entities, even for what is a relatively well-resourced initiative led by an NGO and local community. Even with the Institute's dedicated staff and resources, fulfilling its monitoring program will be difficult and expensive. For a typical community-led initiative, comprehensive monitoring would be impossible. A major need in landscape rehabilitation, especially stream reconstruction, is for the development of agreed minimum but rigorous standards for design and monitoring, an aim the Institute is pursuing with others.

5 *Frequent use of data*. Sustained data curation, analysis and communication are vital not only from a research perspective but also for maintaining interest, trust and motivation in participants, and to support reporting and compliance. Recourse to additional data also reframes questions: we have increased attention to ground water hydrology and ground-surface water connectivity in light of data indicating greater complexity than generally assumed. For other, purely practical restoration projects not driven by a mandate for or commitment to research, high level data capacities will be deemed unnecessary and too resource intensive. Community-led projects would likely require support from government agency staff or research organisations to undertake data-intensive monitoring and reporting, or combine resources to establish shared capacities. The scale of past and especially future monitoring and data demands of the MRI represent major undertakings and require advance skills and field and data management infrastructure. Again, the development of minimum but sufficient standards for application across similar projects is desirable.

6 *Scientific productivity*. Peer-reviewed papers in the open literature is the ideal, but as noted, long-term projects may take perhaps a decade to generate sufficient time

series. We suggest that other forms of communication should be explicitly aimed for as well as academic publication. The Institute's approach has been to pre-empt formal publication with public release of monitoring reports, data summaries, communication of activities and preliminary findings to an extensive membership list (e.g. a quarterly newsletter), conference presentations and numerous field days. While short of formal research publishing, these measures invite scrutiny and develop trust via openness. One issue encountered is where student or early-career research projects are a core part of the research effort. Release of student data or theses must be managed to protect the intellectual property rights and future publishing opportunities.

7 *Project management*. The establishment of partnerships has been both essential and time consuming, and we would add a further feature of effective LTERM (see 9 below). For leadership and succession planning, the formal governance structure of The Mulloon Institute makes this more achievable than for any one-off project, and is needed when activities and especially monitoring span decades. This suggests use of a properly constituted organisation with reasonable expected longevity to manage administrative, financial and data matters. 'Leadership' in Box 3 (point 7) is not the correct term for a co-owned, multi-party initiative, as opposed to a traditional research project led by a senior scientist. 'Champions' with shared visions, ongoing engagement and supporting transparent and participatory process are more suitable. On funding, the on-ground works component of the MRI have been considerably less than the LTERM-related costs (staff, field equipment, instrumentation, etc.), not even counting significant in-kind academic contributions. Any restoration project wishing to include an LTERM dimension should be aware of such costs, and that research and especially monitoring funding may be difficult to obtain.

8 *Research logistics*. The demands of LTERM have been challenging even for a relatively well-resourced project. (See Youngetob *et al* 2018 further on the costs and resourcing of LTERM). Underestimation of required staff, field equipment, information technology, etc., will undermine the research and monitoring capacity. It is likely that external support will be required for any restoration project especially at larger scales and over many years. As per the investments in Table 2, central to LTERM is security of site access, which for simple restoration projects needs only to be for the duration of management actions, not long-term monitoring. For the MRI, sites under the ownership of the Institute are secure; however, trust and engagement are required to assure access to other properties (thus far, without issue). As the MRI spreads to public land tenure in future implementation stages, additional issues of managing ongoing access and collaboration arise. We recommend MoUs or similar, which while non-binding and general, articulate mutual goals and expectations.

9 *Stakeholder engagement and management* (New feature). With a cross-tenure context, complex regulatory environment and multiple stakeholders, the Institute's experience instructs that effort and skills are required to engage with and manage communication and activities across participating partners. To date, these include: researchers from multiple disciplines, landholders (who vary from major commercial operators to non-resident amenity), agencies and philanthropic and public supporters with varied motivations. Partners include numerous public agencies at local, regional, state (multiple within one state) and federal levels. The demands of such diverse and cross-sector engagement well exceeds that typical in a purely research-focussed LTERM. The MRI is not just about participation, but more about comprehensive and equal partnerships closer to the ideals of co-production and co-governance (e.g. Wyborn 2015). Hence, a diversity of skills is required including in relevant social sciences.

Conclusion

The genesis of Mulloon Rehydration Initiative began 15 years ago with paddock walks and conversations between Tony Coote and Peter Andrews, resulting in a demonstration project on Tony's property. The visually dramatic changes along that section of Mulloon Creek (Fig. 4) have since inspired neighbours and many visitors, including senior public figures (https://themullooninstitute.org/testimonials). Funding for expanded activities is now flowing for rehydration projects along the full length of the Mulloon Creek, integrated with a multi-dimensional research and monitoring program.

We look forward to further community- or NGO-led major rehydration initiatives being informed by the MRI experience. We aim to develop efficient and sufficient project design and long-term monitoring approaches that test our understanding of desired catchment outcomes and informs similar activities in other catchments. However, if our Initiative and other catchment scale projects are to be properly monitored and contribute to a larger body of data and understanding, critical support will

be required by government agencies and research organisations.

The journey, to date, has been complex. While the principal focus of the MRI is, on the face of it, simple: rehabilitation of stream and floodplain morphology, hydrology, soil condition and vegetation, it is the scale that makes it such a complex challenge. It is large scale, more than 20 km of stream and over 20 landholders. It involves contested management interventions with regulatory hurdles to negotiate; it requires active engagement from multiple landholders with multiple environmental, social and production goals' and, there is a clear demand for all these on-ground activities to be integrated into a multi-dimensional and long-term research and monitoring program. Thus, the MRI provides a case where effort is being made to test the 'science, art and craft' of long-term practice and monitoring and to inform other evidence-based, multi-stakeholder management interventions at larger scales. We hope the next 15 years demonstrates this to be case.

Acknowledgements

The Mulloon Institute acknowledges the Indigenous custodians of the lands and waters where its work is undertaken. We acknowledge the landholders, officials, donors and other researchers who have contributed to the Initiative over a decade and a half, and especially the inspiration of Peter Andrews and the late Tony Coote AM and Mrs Toni Coote. Financial and other contributions are identified in the text and tables of this paper.

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Supporting Information

Additional supporting information can be found in the following online files.

Appendix S1. Description of three catchment condition States (see Figure 6) across ecological, hydrological, landuse and social dimensions.

Gardening for health: a regular dose of gardening

Author: Richard Thompson[A]

ABSTRACT

There is increasing evidence that exposure to plants and green space, and particularly to gardening, is beneficial to mental and physical health, and so could reduce the pressure on NHS services. Health professionals should therefore encourage their patients to make use of green space and to work in gardens, and should pressure local authorities to increase open spaces and the number of trees, thus also helping to counteract air pollution and climate change.

There is anxiety that the NHS cannot cope now and in the future with the health needs of an increasing and ageing population. It is also realised that pharmaceutical drugs, transformative though they have been, are increasingly expensive and are not always as effective as they appear in the results of early, enthusiastically reported, clinical trials. Drugs are also prescribed at the cost of side effects, which are a leading cause of admissions to hospital, particularly for the elderly, who are poorly represented in trials.

Health depends on a range of social, economic and environmental factors, as is emphasised by the shaming disparity between the length of life in different areas of the country.[1] In addition to improving the information given to patients and health professionals on the true efficacy of drugs and on their risks, and thus empowering choices for patients, there are opportunities to treat some physical and mental conditions with alternative or complementary therapies, and to encourage changes in lifestyle. Such treatments could reduce the workload and financial pressure on the NHS, particularly in primary care, but they clearly must only be recommended by health professionals if there is good evidence that they are effective; many are without merit. Ten million of the UK population are defined as disabled, among them 6.9 million of working age. Can we do better for these people without using drugs?

Green care

One group of holistic therapies that aim to treat the whole person and has been well researched through surveys and randomised trials is so-called green care, or therapy by exposure to plants and gardening.[2,3] Several trials have revealed the beneficial effects on mood and mental health of simply observing nature, or even images of natural scenes. In a Japanese study, viewing plants altered EEG recordings and reduced stress, fear, anger and sadness, as well as reducing blood pressure, pulse rate and muscle tension.[4] Another Japanese study simply found that it more beneficial physiologically to view a green hedge rather than a concrete fence.[5] In a pioneering randomised study by the environmental psychologist Roger Ulrich,[6] views of plants and trees from post-operative wards improved the mood of patients, and reduced analgesic use, surgical complications and length of stay. Similar beneficial results have been found for patients undergoing dental treatment,[7] and viewing natural scenes together with natural sounds improved the experience of bronchoscopy.[8] Another carefully controlled study showed that viewing sculpture gardens without any greenery through the windows of an oncology ward caused a negative reaction in many patients.[9] Even randomly exposing post operative patients to pictures of countryside on the walls of their rooms can reduce pain and anxiety, while, perhaps not surprising to everyone, abstract images increased anxiety.[10,11] It would be interesting to study the mental effects of visiting art galleries! Paintings on the walls of a Swedish psychiatric hospital were often vandalised, but only if they were of abstract images, not if they were landscapes.[12] The charity MIND compared short walks through a garden with walks in a shopping complex, and showed that the former improved mental health, whereas the latter made it worse.[13] In a prison in Michigan, residents who had a view of the countryside from their cells used the prison medical services less than those with an internal courtyard view.[14]

In another randomised experiment, when post-operative patients were exposed to eight different species of indoor plants, both pain and length of stay were once again reduced and patients' satisfaction with their hospital rooms was improved.[15] Exposing pictures of flowers in the dictator game, which is an economic game that questions whether individuals are solely driven by self interest, can change the decisions made by the players.[16] In another study, putting plants in a computer room improved productivity and lowered blood pressure.[17] Indoor gardening has been used to treat patients with mental health problems.[18] It is not only the appearance of plants that is beneficial: their leaves remove toxins, dust and microorganisms from the air and they also produce the so-called negative ions from their leaves. The overall evidence that charged ions affect mood is, however, unconvincing,[19] despite advertisements strongly recommending their benefits.

Many studies in the UK and other countries concur that higher proportions of green space, especially biodiverse habitats,[20] are associated with less depression, anxiety and stress, even after controlling for potential confounding factors such as deprivation.[20–22] In Japan, green space has been linked with increased longevity.[23] Exposure to green space seems to reduce

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health inequalities related to deprivation,[24] but associations are not proof of an hypothesis and a few studies have disagreed.[25,26] In reality, studies such as these suffer from embedded residual confounding correlations between green space and higher income, better housing, and healthier lifestyles (such as less smoking), which can be difficult to disentangle. Interestingly, the benefit of green space may not be simply related to physical activity,[27,28] but might rely more on improved social interaction.[29]

Gardens

Therapeutic gardens have been used in hospitals for thousands of years, and were strongly supported by Florence Nightingale; they improve the surroundings for patients, visitors and staff. Ulrich[11] has emphasised their beneficial effects on stress, especially if the spaces support biodiversity, with increased satisfaction reported by those who use them.

A small central garden between buildings at St Thomas' Hospital was created for the millennium; another at St George's Hospital was successfully commissioned by Harold Lambert FRCP. Gardens that are attached to hospices, such as Maggie's cancer centres,[30] and care homes are now widespread and provide that important view from the rooms and an area to visit. Recently, impressive gardens for wheelchair users and those confined to beds have been designed around spinal injury units; these are known as Horatio's gardens and have been set up in memory of Horatio Chapple, who died in an accident in the Arctic.[31] They can include facilities for therapy and training in gardening skills.

Gardens around prisons have a long history of improving the lives of the prisoners and offering training towards employment in the horticulture industry. At the urban prison in Wandsworth, a collaboration with The Conservation Foundation has seen green areas introduced into the prison and an exercise yard dug up to make way for a vegetable garden where produce can be grown.[32] In the First World War, British prisoners in the civilian internment camp at Ruhleben in Germany were sent seeds and plants by the Royal Horticultural Society in London to help them to develop a successful garden.[33]

A recent survey by Mintel for the charity Thrive,[34] which enables social and therapeutic horticulture, showed that among people with disabilities, a quarter listed gardening as a hobby. Two-thirds of the respondents owned a garden and 87% had access to a garden that they thought was beneficial to their health. Surveys in the general population have given similar results,[35] with a large majority believing that gardens were beneficial to health. Numbers of visitors to garden centres and private gardens, such as those in the National Garden Scheme or run by the National Trust, are increasing. Gardening has been associated with a lower prevalence of dementia and with positive health effects in several countries,[36,37] and economic benefits have been shown, for instance, for mental health services.[38]

In northern Europe, Green Care Farms have proved popular and have grown in number so that there are now hundreds of such facilities in Norway and in the Netherlands. Patients who have impaired mental health, learning disabilities or drug dependency, as well as older people, are referred for a period of work in functioning farms, often involving animals.[39] In England, the University of Essex has set up the National Care Farms network. By 2012, the network included 180 farms, which were visited by 3000 patients a week;[40] their positive benefits have been independently reviewed.[41] The charity Thrive has identified 800 therapeutic horticulture projects across England and Wales.

The effects of gardening on body and mind

Why does gardening seem to be so beneficial to health? It combines physical activity with social interaction and exposure to nature and sunlight. Sunlight lowers blood pressure as well as increasing vitamin D levels in the summer,[42] and the fruit and vegetables that are produced have a positive impact on the diet. Working in the garden restores dexterity and strength, and the aerobic exercise that is involved can easily use the same number of calories as might be expended in a gym. Digging, raking and mowing are particularly calorie intense;[43] there is a gym outside many a window. The social interaction provided by communal and therapeutic garden projects for those with learning disabilities and poor mental health can counteract social isolation. Furthermore, it has also been reported that the social benefits of such projects can delay the symptoms of dementia[44] (an effect that might be partly due to the beneficial effects of exercise). Patients who are recovering from myocardial infarction or stroke find that exercise in a garden, using constraint therapy of a paretic limb, for example,[45] is more effective, enjoyable and sustainable than therapy in formal exercise settings. For some patients, gardening can even lead to employment. There are also successful schemes that involve volunteers to help older people who cannot manage their gardens, with both the volunteer and the owner benefitting from the social interaction and from the produce and a shared interest.

Intelligent Health points out that the pandemic of physical inactivity is the fourth leading cause of premature death, and contributes to preventable physical and mental disorders.[46] The Department of Health calculates that an increase of only 10% in average exercise by adults would postpone 6000 deaths and save £500 million annually.[47] Regular moderate intensity exercise may reduce the risk of dementia,[48] mental health problems, cardiovascular disease, diabetes, and cancer of the breast and colon, and in an Australian study, gardening was found to be more effective than walking, education or maintaining alcohol intake at moderate levels in protecting against dementia.[44] It enhances self esteem and alters the EEG.[49] Similarly, moderate exercise in leisure time is associated with increased longevity, regardless of weight,[50–52] particularly if combined with exposure to natural scenes,[53] although some studies have suggested that exercise declines with reduced cognition; a reverse causation bias.

Thankfully, high intensity exercise is not needed to obtain these benefits,[51,52] which is perhaps as well given that the uptake of cycling- and gym-based exercise is poor in the older population, and that these activities can be expensive. Gardening or simply walking through green spaces could therefore be important in preventing and treating ill health. The Five Year Forward Plan for the NHS[54] emphasises the potential importance of prevention in reducing the mounting pressure on the NHS and on social services. There are 152,000 strokes annually and a total of 1.2 million stroke survivors in the UK.[55] Also in the UK, a quarter of a million patients are admitted to psychiatric hospitals each year and dementia is predicted to affect a million people by 2025.

Few complementary therapies have been convincingly shown to be effective, but gardening and nature, which are alternative therapies, offer a proven, cheap and nearly universally available means to improve the nation's health. Although there is evidence that knitting can also help![56]

The green environment

The 2016 RCP report on pollution[57] underlined the deleterious effects of air pollution on respiratory and cardiovascular health globally, with an estimated 8000 premature deaths a year in the UK alone being linked to this issue. The House of Lords has reported specifically on the poor air quality in London.[58] Poor air quality can be associated with higher mortality in acute medical wards.[59] Not only larger forests[60] but also urban forests[47] can offset this, as can plants in buildings, gardens, parks, and roadways. Trees, for instance, remove large quantities of toxins and particulates through their leaves,[61] transmitting toxins to the soil where microorganisms metabolise them, or trapping them in hairs on leaves that later fall.[61] Roadside trees reduce the indoor concentration of particulates.[62] Although evergreen trees have smaller leaf areas than their deciduous cousins, they are more effective in the winter months. Trees themselves do emit varying amounts of volatile compounds,[63] but overall they reduce the levels of pollutants close to roads.[64–66] For instance, a single maple tree can remove 48 lb (22 kg) of particulates and 100 lb (45 kg) of carbon each year, as well as toxic metals, nitrogen oxides and sulphur dioxide. The link between residence close to roads and dementia and other problems[67] could be due to exposure to the many pollutants emitted by vehicles, such as nitrogen oxides, carbon dioxide, ozone, metals, organic compounds and differently sized particulates.

Trees, hedges, and other plants counter climate change by trapping carbon and emitting oxygen; and worldwide, forests may offset a quarter of man-made carbon dioxide. They also improve the environment by reducing noise, heat, glare, wind, water run-off, erosion and dust. Cooling from shading and the evaporation of water from leaves can reduce the need for air conditioning in buildings, and cooling also reduces the formation of some pollutants, such as ozone. Even lawns and turf are helpful,[68,69] also trapping pollutants and passing them on to soil microorganisms, in addition to providing recreational space for exercise. Plants may also help to solve the problem of polluted soils in industrial areas. Architects are reluctant to preserve old trees or add them to their developments, and so trees must be protected or included in planning consent conditions, and later properly maintained.

What can health professionals do?

Health professionals should encourage their patients not to see danger in exercise in the garden, green spaces, parks and the countryside. Instead, they should emphasise the potential benefits to patients' health,[70] such as improvements in strength, balance and dexterity. When appropriate, patients can be referred to local community and therapeutic gardening projects, where occupational therapists trained in horticulture help them to manage and treat their medical issues.[71] This is part of what has become known as social prescribing[72] or community referral, which has the potential to improve the physical and mental health of the population by preventing illness or by ameliorating the effects of established disability. Gardens can also help to improve parity between the treatments for mental and physical disabilities.

The particular benefit of gardening to veterans of the armed services has been fully reviewed.[73] Both mental conditions, such a post traumatic stress disorder, and the effects of physical injuries can be improved,[74] and there are opportunities to train for a new career in the expanding horticulture industry. Health professionals should encourage the development of gardens in hospitals, hospices, schools[75] and prisons. They should try to influence the design of new health service buildings by insisting that there are views of outside nature from every patient and staff room, and by placing internal plants in atria, communal areas, surgeries, clinics and staff rooms, even if they are misguidedly banned from wards. Even window boxes and balconies can be used. Health professionals should also encourage the teaching of the skills and benefits of gardening in schools.[75]

In addition, health professionals should encourage local authorities to plant more trees; the Greater London Authority alone plans to plant two million more trees by 2025.[76] Green spaces, parks, gardens and allotments will improve the environment,[77–79] particularly where gardens are in short supply, as in deprived urban areas. Despite the apparent density of buildings in our towns, they do contain gardens and green areas, with an average of one fifth of the land in UK towns being given over to green space. Even in the most crowded cities, such as New York and Singapore, roof gardens, green walls and hanging containers are popular. A well-kept local environment improves local pride and can reduce crime and social isolation. Urban planners must be convinced of the importance of including green space,[79,80] as they are in Holland.[81]

I endorse Buck's proposal[2] that gardens and gardening should be incorporated in NHS England's programmes for improving public health, and hope that health professionals will be in the vanguard of the campaign. They should also support the long-standing charity Fields in Trust (previously the National Playing Fields Association), which campaigns to preserve and increase public green spaces. ■

Conflicts of interest

The author is a trustee of the National Garden Scheme, and past trustee, now patron, of the charity Thrive. He is a member of the Royal Horticulture Society's Health and Horticulture Forum, and he gardens in London.

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INCINERATION AND HUMAN HEALTH

State of Knowledge of the Impacts of Waste Incinerators on Human Health.

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INCINERATION AND HUMAN HEALTH

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EXECUTIVE SUMMARY

Management of municipal and industrial waste is a growing problem throughout the world. In the European Union, while waste output is continually increasing, new regulations are imposing more stringent restrictions on the amount of waste permitted to go to landfill. At the same time, many incinerators have been closed over the past few years because of stricter regulations on their atmospheric emissions. In Europe, all incinerators will soon have to comply with new standards set out in a recent EC draft directive.

Fortunately, there are alternative solutions to turn around the waste crisis on a long-term basis. Primarily, this means the implementation of waste prevention strategies, and in conjunction with this, waste re-use and recycling. Despite this option, there is an emerging trend for constructing, and planning to construct, new incinerators in an attempt to provide a "quick fix" solution to the waste crisis. Incinerators are deemed as favourable in this respect because they are perceived as reducing waste to one tenth of its original volume, and therefore reduce the volume of waste going to landfill sites.

Incinerators, however, are controversial in terms of their potential impacts on the environment and human health, as well as in terms of the economic considerations which do not favour this technology. They are known to emit numerous toxic chemicals into the atmosphere and produce ashes and other residues. One country, the Philippines, has taken serious note of the many concerns about incineration at a governmental level. Following strong public opposition to incinerators, the Philippine Clean Air Act of 1999, banned the incineration of municipal, medical and hazardous waste. Waste reduction, re-use and recycling are being promoted while non-burn technologies are recommended for waste that needs some form of treatment. Meanwhile, some governments in Europe are advocating the construction of even more incinerators.

This report was undertaken to draw together scientific findings on incinerator or releases and their impacts on human health. A broad range of health effects have been associated with living near to incinerators as well as with working at these installations. Such effects include cancer (among both children and adults) adverse impacts on the respiratory system, heart disease, immune system effects, increased allergies and congenital abnormalities. Some studies, particularly those on cancer, relate to old rather than modern incinerators. However, modern incinerators operating in the last few years have also been associated with adverse health effects.

Despite reductions of some chemicals in stack emissions, modern incinerators nevertheless still emit numerous toxic substances to the atmosphere as well as in other residues such as fly ash and bottom ash. Moreover, reductions of dioxins and other chemicals in stack gases commonly leads to increased releases of these same chemicals in the other incinerator residues. In most cases, health effects which have been associated with incinerators cannot be tied down to a particular pollutant, Together with the limited data available, it is, therefore, impossible to predict health effects of incinerators including new or updated installations. With such factors in mind, this report demonstrates that there is an urgent need for the complete phase out of incineration and the implementation of sound waste management policies based on waste prevention, re-use and recycling.

INCINERATORS – WASTE GENERATORS

It is a common misconception that things simply disappear when they are burned. In reality, matter cannot be destroyed – it merely changes its form. This can be exemplified by looking at the fate of some substances in wastes which are burned in municipal solid waste (MSW) incinerators. These incinerators are typically fed mixed waste streams that contain hazardous substances, such as heavy metals and chlorinated organic chemicals. Following incineration, heavy metals present in the original solid waste are emitted from the incinerator stack in stack gases and in association with tiny particles, and are also present throughout the remaining ashes and other residues. Incineration of chlorinated substances in waste, such as polyvinyl chloride (PVC) plastic, leads to the formation of new chlorinated chemicals, such as highly toxic dioxins, which are released in stack gases, ashes and other residues. In short, incinerators do not solve the problems of toxic materials present in wastes. In fact they simply convert these toxic materials to other forms, some of which may be more toxic than the original materials. These newly created chemicals can then re-enter the environment as contaminants in stack gases, residual ashes and other residues.

All types of incinerators release pollutants to the atmosphere in stack gases, ashes and other residues. A multitudinous array of chemicals is released, including innumerable chemicals that currently remain unidentified. The chemicals present in stack gases are often also present in ashes and other residues. Such chemicals include dioxins, polychlorinated biphenyls (PCBs), polychlorinated napthalenes, chlorinated benzenes, polyaromatic hydrocarbons (PAHs), numerous volatile organic compounds (VOCs), and heavy metals including lead, cadmium and mercury. Many of these chemicals are known to be persistent (very resistant to degradation in the environment), bioaccumulative (build up in the tissues of living organisms) and toxic. These three properties make them arguably the most problematic chemicals to which natural systems can be exposed. Some of the emitted

chemicals are carcinogenic (cancer-causing) and some are endocrine disruptors. Others such as sulphur dioxide (SO^2) and nitrogen dioxide (NO^2) as well as fine particulate matter, have been associated with adverse impacts on respiratory health.

It is a popular misconception that the weight and volume of the original raw waste are reduced during incineration. It is often quoted that the volume of waste is reduced by about 90% during incineration. Even if only the residual ashes are considered, however, the actual figure is closer to 45%. The weight of waste is supposedly reduced to about one-third during incineration. However, this once again refers only to ashes and ignores other incinerator emissions in the form of gases, which result in an increased output in weight. In sum, if the mass of all the outputs from an incinerator, including the gaseous outputs, are added together, then the output will exceed the input.

ENVIRONMENTAL AND HUMAN EXPOSURE TO INCINERATOR RELEASES

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 in 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.

HEALTH IMPACTS

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 (see tables below). 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.

HEALTH IMPACT	COMMENTS
Biomarkers of Exposure	
Elevated mutagens in urine	Incinerator ashes and stack emissions are mutagenic (have the ability to damage DNA). Workers are therefore exposed to mutagenic compounds. Elevated mutagens in urine indicate exposure to mutagenic compounds. (Study dates 1990 & 1992).
Elevated levels of hydroxypyrene in urine	Hydroxypyrene is an indicator of internal exposure to PAHs. The result suggests elevated exposure to PAHs. (Study date 1992).
Increased quantity of thioethers in urine	Thioethers in urine are an indicator of exposure to electrophilic compounds such as PAHs. The results suggest exposure to electrophilic compounds. (Study date 1981).

HEALTH IMPACT	COMMENTS
Cancer	
3.5-fold increased probability of mortality from lung cancer	Workers who were employed at a Swedish MSW incinerator in Sweden at sometime between 1920 and 1985. (Study date 1989).
1.5-fold increased likelihood of mortality from oesophageal cancer	Workers who were employed at a Swedish MSW incinerator in Sweden at sometime between 1920 and 1985. In conjunction with evidence from other research, the result implies an increased health threat to workers. (Study date 1989).
2.79-fold increase in mortality from gastric cancer	Workers employed at an MSW incinerator in Italy at sometime between 1962 and 1992. Some of the increase may have been due to other confounding factors.
Other Impacts	
Increased mortality from ischemic heart disease	Workers who were employed at a Swedish MSW incinerator in Sweden at sometime between 1920 and 1985. The result was statistically significant in workers with greater than 40 years employment. (Study date 1989).
Excess hyperlipidemia. A significant association between blood dioxin levels and natural killer cell activity (immune system effect). Altered sex ratio among offspring. Decreased liver function. Increased allergy.	Workers employed at an incinerator in Japan, that operated between 1988 and 1997. Excess of hyperlipidemia was significant. Change in immune system cells. Altered sex ratio was not statistically significant. Correlation between allergy and dioxin exposure must be donfirmed. (Study date 2000).
Other Impacts	
Excess of proteinuria (urine abnormality) and hypertension. Possible increased incidence of small airway obstruction (unconfirmed diagnosis). Abnormal blood chemistry.	Workers at a MSW incinerator in the US. An excess of workers with significant proteinuria. (Study date 1992).
Chloracne (a skin condition due to dioxin-exposure)	Chloracne found in one worker from an old incinerator in Japan, who had high blood levels of dioxin. (Study date 1999).

Summary of Studies on Health of Populations Living in the Vicinity of Incinerators

HEALTH IMPACT	COMMENTS
Biomarkers of Exposure	
Elevated levels of thioethers in children's urine	Urinary thioether levels were t higher among children living near a recently built incinerator in Spain. (Study date 1999)
No abnormal chromosomal damage	No excess chromosomal damage among children living near two Belgian incinerators. (Study date 1998)
Cancer	
44% increase in soft tissue sarcoma and 27% non-Hodgkin's lymphoma.	Significant clusters of these cancers in residents living close to an incinerator in France. Possibly due to exposure to dioxin from the incinerator, but more research is needed to confirm if this is the case. (Study date 2000).
6.7-fold increase in likelihood of mortality from lung cancer	Significantly increased occurrence in residents living close to a MSW incinerator in an urban area of Italy. (Study date 1996).

HEALTH IMPACT	COMMENTS
Cancer	
Increased incidence of cancer of the larynx	Found around one UK hazardous incinerator of waste solvents (1990), but not nine others. In Italy, excess mortality from this cancer was found in residents living near to an incinerator, a waste disposal site and an oil refinery.
37% excess mortality due to liver cancer	A study on 14 million people living within 7.5 km of 72 MSW incinerators in the UK. Further research to eliminate possible confounders found the increased probability of liver cancer to lie between 20 and 30%. Social deprivation could not be totally ruled out as a confounder. (Study dates 1996 and 2000).
2-fold increased probability of cancer mortality in children	A study conducted on 70 MSW incinerators in the UK (1974-87) and 307 hospital waste incinerators (1953-1980). These results are consistent with another study in which an increased probability of childhood cancer was observed for hospital incinerators and large-scale, high-temperature combustion industries (Study dates 1998 and 2000).
Respiratory Impacts	
Increased purchase of medicine for respiratory problems.	A study at a village in France that had a MSW incinerator. Results suggest increased use of medicine for respiratory illness but a cause-effect relationship cannot be concluded (Study date 1984).

HEALTH IMPACT	COMMENTS
Respiratory Impacts	
Increased respiratory symptoms, including 9-times increase in reporting of wheezing or cough.	A study in the US on residents living near to a hazardous waste incinerator. The results are of limited utility because of methodological concerns about the study. (Study date 1993).
Adverse impacts on lung function of children.	A study on children living near to a wire reclamation incinerator in Taiwan. Results indicate that higher air pollution, but not the incinerator itself, is linked to altered lung function in children. (Study date 1992).
Increased respiratory symptoms including lung disease, wheezing, persistent cough and bronchitis.	A study on 58 individuals living near to cement kilns burning hazardous waste in the US. Significant increase in respiratory symptoms. (Study date 1998).
No adverse effect on the prevalence or severity of asthma in children.	A study on children living near to sewage sludge incinerators in Australia. (Study date 1994).
No increase in respiratory effects or decrease in lung function	A study on 3 communities (6963 individuals) living near to a municipal, hazardous and hospital waste incinerator in the US. The lack of association between exposure to particulate air pollution and respiratory health in this study should be interpreted cautiously due to limitations in data on individual exposures.

HEALTH IMPACT	COMMENTS
Sex Ratio	
Increase in female births	A study on populations living near to 2 incinerators in Scotland, UK. The effect was found in the area potentially most exposed to incinerator releases. Other studies have found an increase in female births among fathers who were accidentally exposed to high levels of dioxins. (Study dates 1995 and 1999).
Congenital Abnormalities	
Increased incidence in orofacial clefts Other midline defects including spina bifida and hypospadias (genital defect)	The significant increase in orofacial clefts was observed for births in an area located near to an incinerator site where open burning of chemicals took place 1960-69. A link between the conditions and living near the incinerator is likely but not confirmed.
1.26-fold increased probability of congenital malformations among new born infants	A study conducted on a population living near to 2 MSW incinerators in Wilrijk, Belgium. (Study date 1998).
Increased congenital eye malformations (anecdotal report)	Reported at an area near two chemical waste incinerators in Scotland, UK. Further research in the UK found no link, although the study was hampered by lack of data on the condition. (Study date 1989).
Multiple Pregnancy	
Possible increase in rate of twinning/multiple pregnancy.	An increase in twinning was significant in 1980 in a population living near to an incinerator in Scotland, UK. A 2.6-fold probability of multiple pregnancy found near incinerator in Belgium (Study date 2000). No impact on multiple pregnancy found on a survey of an incinerator in Sweden. Data from different studies is conflicting and inconclusive.
Other Impacts	
Lower thyroid hormone levels in children	Children living near a German incinerator had significantly lower blood levels of certain thyroid hormones. (Study date 1998)
Increased allergies, increased incidence of common cold, increased complaints about health in general, increased use of medication in school children	A study conducted on school children living near to two MSW incinerators in Wilrijk, Belgium. (Study date 1998).

INCINERATOR RELEASES AND REGULATION

• Stack Gases

As previously mentioned, numerous chemicals are emitted to the atmosphere from incinerators thrtough the stack gases. Important points regarding some of these chemical emissions are given below.

• Dioxins

Extensive research has demonstrated that dioxins can cause a diverse array of toxic effects. They have become widespread contaminants throughout the globe and are present in the body tissues of human beings across the whole globe. Research suggests that, in industrialised countries, dioxins have now reached levels in tissues of the women which may cause subtle, adverse effects upon the immune system, and nervous system of their babies.

Incineration, particularly MSW incineration, was identified as a major source of dioxins during the 1980s and early 1990s. It has been estimated as accounting for between 40 and 80% of atmospheric dioxin emissions in various industrialised countries. The true figure may be even greater because there are several methodological flaws in nearly all of the dioxin inventories that estimate atmospheric emissions from incineration.

Considerable improvement in air pollution control technologies that have been installed in new or updated incinerators during the 1990s is thought to have led to substantial reductions in the quantity of dioxins released to the atmosphere from incinerator stacks. However, recent estimates suggest that MSW incinerators are still a main source of dioxins in the environment. In the UK, it was estimated that MSW incinerators were responsible for 30-56% of dioxin emissions while in Denmark a recent mass balance study identified MSW incineration as the

dominant source of dioxins to atmosphere and a highly significant contributor (via ash residues) to landfill. Moreover, reduction of dioxins emitted in stack gases has most likely resulted in a corresponding increase in dioxins emitted as contaminants of ash residues.

While measurements taken from some new or modernised incinerators have shown that they comply with limits set by the new EC directive, others have not. Those not fulfilling the EC regulatory limit include incinerators that have recently been tested in Spain, Poland, Sweden, and Belgium. In Belgium, testing was carried out on an incinerator using the routine technique of taking "point measurements" which involves monitoring dioxin levels over a period of several hours. However, when testing was carried out by "continual monitoring", over a 2 week period, the results were substantially different. The point measurement technique underestimated dioxin emissions by a factor of 30 to 50. It is therefore of great concern that very few incinerators are tested using continual monitoring or tested under their normal operating conditions. Moreover, the new EC regulations do not stipulate that measurements should be taken using this technique, so current routine monitoring of incinerator stack gases, using point measurements, could be grossly inaccurate and underestimate dioxin emissions to air.

• Other Organic Compounds

For regulatory purposes, the EC has proposed a limit for total organic carbon emissions to atmosphere to regulate all the organic chemicals emitted. This regulation, however, fails to take into account the toxicity/health impacts of known organic chemicals that are emitted from incinerator stacks. Similarly it totally ignores unknown chemicals of unknown toxicity and the potential health effects they could cause.

• Heavy Metals

Heavy metals, including lead and cadmium, are emitted in stack gases from incinerators. Many heavy metals are persistent and exert a wide range of adverse impacts on health.

With the exception of mercury, the levels of heavy metals released in stack gases from incinerators have decreased considerably over the past decade due to improvement in air pollution abatement technologies. Nevertheless, the quantities in which they are still emitted from modern incinerators potentially add to current background levels in the environment and in humans. As is the case with dioxin emissions to the atmosphere, the reduction of levels of heavy metals emitted in stack gases causes a corresponding increase in levels in the ashes, which will, ultimately, result in contamination of the environment when these are disposed of.

• Particulate Matter

Incinerators of all types emit particulate matter into the atmosphere. The majority of this particulate matter is ultrafine in size. Current air pollution control devices on incinerators only prevent 5 to 30% of the "respirable" (<2.5 μm) sized particles from entering the atmosphere, and can do very little to prevent ultrafine (<0.1 μm) particulates from escaping. It is these respirable particles, and especially the ultrafine particles, which can reach the deepest regions of the lungs, and which are thought to be responsible for causing adverse impacts on human health. Incinerators therefore contribute to the type of particulate air pollution that is the most dangerous for human health. In addition, recent evidence suggests that particles containing heavy metals, such as those emitted from incinerators are especially of concern with regard to health. Incinerators are, therefore, likely to produce particulate air pollution which is even more toxic than, for example, that emitted from a coal-fired power station.

The new EC Draft Directive does not set any limits for the release of fine particulate matter. Given the scale of the health impacts resulting from such particulate air pollution, this can be considered as an outstanding neglect of factors relevant to human health, and which requires rigid control and regulation.

• Ashes

Fly ashes from air filtration equipment on incinerators and the bottom ashes that remain after incineration contain numerous hazardous chemicals, such as dioxins and heavy metals. Despite the potential toxicity of ashes, there are no EC limits for levels of persistent organic chemicals and heavy metals in ashes.

Because of their contamination, disposal of incinerator ashes presents significant environmental problems. The majority of ash is landfilled. This can result in contamination of sub-soils and groundwater. In some cases, the contamination of groundwater by compounds that have leached from the waste, in particular, heavy metals like lead and cadmium from fly ash has been documented. In an attempt to reduce leaching, fly ash is sometimes stabilised in cement before disposal. Although this method reduces the immediate leaching of heavy metals and other toxic chemicals, weathering and erosion over time will ultimately cause their release back to the environment

There has been a recent tendency in some European countries to use bottom ashes and/or fly ashes for construction purposes, a practice that reduces the financial costs of "secure" ash disposal. Ash has been used in road and path construction. Again, however, the future releases of persistent toxic substances due to erosion over time could result in the release of substances back to the

environment and, therefore, potentially to human exposure. This has recently been exemplified in Newcastle, UK where fly ash and bottom ash from a presently operating, modern incinerator, were used for path making and also spread over allotments as fertiliser between 1994 and 1999. Recent analysis of ash from the allotments found that it is contaminated with extremely high levels of heavy metals and dioxins. Clearly, the use of ashes from incinerators represents a potential threat to human health, but this practice is not being discouraged either by the EC or at a national level by the regulatory regimes proposes or currently in place.

• The Way Forward

A limited amount of epidemiological research has been directed at investigating the health impacts of incinerators. Despite this, scientific studies reveal that MSW and other incinerators have been associated with detrimental impacts on health.

The new EC draft directive on incinerators is not formulated to take human health impacts into account in relation to the regulation and control of these facilities. Rather, the regulatory limits that are set for the permissible release of substances are based on what is considered to be technically achievable. In any case, the draft EC directive on incinerators, not yet in force, can be regarded as already outdated. Many European countries have already committed themselves at the OSPAR Convention to phase out all releases of hazardous substances to the environment by 2020. In this context no emissions of hazardous chemicals would be allowable in stack gases or ashes. This is likely to prove impossible for incineration technology to ever achieve.

In addition, at the Fifth Intergovernmental Negotiating Committee Meeting (INC5) on the Elimination of Persistent Organic Pollutants (POPs), held in December 2000, a world-wide agreement was reached to reduce total dioxin releases, with the ultimate aim of their elimination. Incineration is listed as one of the main industrial source categories for dioxins, and requires the use of BAT (Best available Techniques) for new installations and substantially modified existing facilities. It was also agreed to promote the development and, where deemed appropriate, require the use of substitute or modified materials, products and processes to prevent the formation and release of dioxins. In this context, incineration is acknowledged as a significant source of dioxins that, in the longer term, should be replaced by alternatives.

To comply with the provisions of the OSPAR agreement and of the emerging POPs Convention implies a radical rethink of industrial and manufacturing processes. Instead of waste-generating "dirty" technologies, which rely upon incineration and other environmentally dubious waste disposal techniques, OSPAR implies the need to develop and use "clean-production" technologies which eliminate toxic waste. The adoption of "zero-waste" as a central tenet of environmental regulation also implies that the Precautionary Principle of environmental protection will occupy an equally key position in the development of policy and regulatory frameworks. The precautionary principle requires that the burden of proof should not be laid upon the protectors of the environment to demonstrate conclusive harm, but rather on the prospective polluter to demonstrate no likelihood of harm. On this premise of precautionary regulation it can be argued that there is already sufficient evidence of environmental contamination and adverse human health impacts to call for a complete phase out of incineration.

In the case of waste management, adoption of a zero releases strategy and the reduction of health impacts from waste management means a move towards an environmental management paradigm based upon the three axioms of reduce, re-use and recycle in relation to the generation of both municipal and industrial wastes.

GREENPEACE DEMANDS

A drive towards waste prevention, re-use and recycling, and therefore also towards lessening the adverse health impacts from waste management, should include the following measures:

- The phase out of all forms of industrial incineration by 2020, including MSW incineration. This is in line with the OSPAR Convention for the phase out of emissions of all hazardous substances by 2020.

- Financial and legal mechanisms to increase re-use of packaging (e.g. bottles, containers) and products (e.g. computer housings, electronic components).

- Financial mechanisms (such as the landfill tax) used directly to set up the necessary infrastructure for effective recycling.

- Stimulating markets for recycled materials by legal requirements for packaging and products, where appropriate, to contain specified amounts of recycled materials.

- Materials that cannot be safely recycled or composted at the end of their useful life (for example PVC plastic) must be phased out and replaced with more sustainable materials.

- In the short term, materials and products that add to the generation of hazardous substances in incinerators must be prevented from entering the the waste stream at the cost of the producer. Such products would include electronic equipment, metals and products containing metals such as batteries and florescent lighting and PVC plastics (vinyl flooring, PVC electrical cabling, PVC packaging, PVC-*u* window frames etc) and other products containing hazardous substances.

and more generally:

- Further the development of clean production technologies which are more efficient in terms of material and energy usage, produce cleaner products with less waste and which, ultimately can be designed to operate in a "closed loop" configuration in order to fulfil the needs of society in a more equitable and sustainable manner;

- Fully implement the Precautionary Principle, such that, in the future, problems are avoided before they occur. The continuation and further development of scientific research has a fundamental role to play in identification of potential problems and solutions. Nonetheless, we must be ready to take effective precautionary action to prevent environmental contamination and degradation in the face of the considerable and often irreducible uncertainties associated with determination of health and other environmental impacts from incineration.

1. INTRODUCTION TO HEALTH EFFECTS OF INCINERATORS

The impact of waste incinerators on health and their releases of hazardous combustion products, such as dioxins and PAHs are of great public concern (Ardevol *et al.* 1999). Research has identified numerous toxic compounds, which are emitted in stack gases and in ashes, as well as many unidentified substances of unknown toxicity (see section 5). Individuals who are exposed to the hazardous substances resulting from incineration, and whose health can, therefore, be potentially affected by such exposure, include workers connected with incinerator facilities and populations living within their local vicinity. Studies on exposure and health impacts of incinerators have focused entirely on these two groups of individuals.

Importantly, a recent publication by the National Research Council (NRC 2000), an arm of the National Academy of Sciences that was established to advise the U.S. government, concluded that it was not only the health of workers and local populations that could be affected by incinerators. The NRC reported that populations living more distantly from incinerators are also likely to be exposed to some incinerator pollutants. For example,

"*Persistent air pollutants, such as dioxins, furans and mercury, can be dispersed over large regions – well beyond the local areas and even the countries from which the sources first emanate.… Food contaminated near an incineration facility might be consumed by people close to the facility or far away from it. Thus, local deposition on food might result in some exposure of populations at great distances, due to transport of food to markers. However, distant populations are likely to be more exposed through long-range transport of pollutants and low-level, widespread deposition on food crops at locations remote from a source incineration facility.*"

and,

"*The potential effects of metals and other pollutants that are very persistent in the environment may extend well beyond the area close to the incinerator. Persistent pollutants can be carried long distances from their emission sources, go through various chemical and physical transformations, and pass numerous times through soil, water, or food. Dioxins, furans, and mercury are examples of persistent pollutants for which incinerators have contributed a substantial portion of the total national emissions. Whereas one incinerator might contribute only a small fraction of the total environmental concentrations of these chemicals, the sum of the emissions of all the incineration facilities in a region can be considerable. The primary pathway of exposure to dioxins is consumption of contaminated food, which can expose*

a very broad population. In such a case, the incremental burden from all incinerators deserves serious consideration beyond a local level."

In the present report, published research on human exposure to pollutants from incinerators and health impact studies of workers and local populations is discussed. A broad range of health impacts has been documented in these two groups, including adverse effects on children living in local populations near to incinerators. However, whether the observed associations with pollutants from incinerators are causal is often difficult to confirm.

Types of Research Study

The impacts of incinerators on human health have been assessed primarily using three types of study. These are human exposure studies, epidemiological studies and finally, risk assessment studies. Exposure studies and epidemiological studies provide the most compelling evidence about the health impacts of incineration since they involve generating scientific data directly from the individuals under investigation. On the other hand, risk assessments are theoretical estimations of what health effects may occur based on mathematical calculation.

• *Exposure Studies*

Exposure to compounds emitted from incinerators may occur, for example, by inhalation of contaminated air, or by consumption of local agricultural produce or soil that has been contaminated by deposition of airborne pollutants. In addition, workers at incinerator plants may also be exposed to contaminated ashes.

To assess possible health impacts resulting from exposure to incinerator releases, reliable methods of assessing exposure are required. One method to assess potential exposure is to monitor levels of contaminants in air from incinerators, and in soils, vegetation and agricultural produce (e.g. see section 4). However, such investigations do not permit the "internal exposure" in humans to be assessed directly (Ardevol *et al.* 1999). Evaluation of internal exposure requires the quantification of compounds in the human body. In recent years, with technological advances, it has become possible to monitor the level of certain toxic compounds from incinerators in the body tissues of humans. This involves the determination of contaminant concentrations in biological samples, for instance, in blood, urine, hair or breast milk.

Exposure studies analyse biological samples for:
1) chemical pollutants that are released from an incinerator, or, 2) metabolites (breakdown products) of these chemicals, or, 3) biomarkers of exposure, (which show biological effects of a toxic exposure). The results of the analyses are compared with a control group of individuals considered to be unexposed.

A number of studies have been conducted to assess exposure of incinerator workers (see section 2.1) and populations living near to incinerators (see section 3.1) using the analyses described above.

• Epidemiological Studies

Epidemiological studies attempt to establish the incidence or prevalence of health effects that may be related to the intake of pollutants released from an incinerator. Information pertinent to the potentially contaminated people is used, for example, birth and death certificates, disease registries, physicians' reports, self-reported symptoms and illnesses. This is compared with similar information from potentially uncontaminated or less contaminated people. Some of the major challenges to establishing a cause-and-effect relationship through epidemiological studies are (NRC 2000):

- Identifying suitably exposed populations that are large enough to establish a useful degree of statistical significance.
- Identifying the many factors that modify the effect (e.g., age, sex, etc.) and/or potentially confounding factors (e.g., smoking, diet, etc.)
- Identifying biases (including reporting biases) in data collection.
- Measuring the frequency of occurrence and concentrations of specific pollutants within the affected population and a potentially unaffected control group.
- Measuring effects that are small, occur infrequently, take many years to appear, and/or occur not in the exposed individuals but in their offspring during infancy, child hood or adulthood.

Epidemiological studies have investigated a variety of health outcomes from exposure to incinerator releases in both workers and in populations living close to incinerators. In particular, cancer and respiratory effects have been analysed. Such human epidemiological studies are however limited in number given the widespread concern about potential health effects of incinerators. The rarity of these studies is possibly due to their expense and difficulty of performance. These studies are generally more valid than other health studies insofar as exposure to all pollutants emitted from incinerators are explicitly or implicitly accounted for, thus mirroring the "real" situation (Rowat 1999) although their power is determined by their design.

• Risk Assessment

Risk assessment attempts to estimate exposure to a particular chemical from the releases in question and ultimately calculates the probability of health effects from the estimated exposure. Risk assessment is a step by step process which involves the use of mathematical equations to estimate pollutant releases, their transport and their transformation in the environment, together with human exposure and finally the likelihood of suffering health effects from this exposure. The use of risk assessment is largely for regulatory decision making.

The process of risk assessment itself is, however, fraught with uncertainties, is necessarily over-simplistic of environmental processes and warrants being viewed with deep scepticism on whether it can actually be protective of human health (e.g. Johnston *et al.* 1996). A fundamental problem of risk assessment is that the estimation of the health consequences of pollution is still a poorly understood science. Even for dioxin (TCDD), one of the most intensely studied chemicals, many unknowns remain and since risk assessment relies on toxicological data to estimate health effects, it is can only be as good as the data on which it is based. Indeed for many chemicals there is a substantial lack of toxicological information. This could obviously lead to imprecision in results generated by risk assessments. Moreover, in the case of the developing foetus and infant, there is a huge uncertainty in the toxicological significance of long-term low-dose exposure to pollutants. It is clear though that the developing stages of life are the most vulnerable to toxic insult. Risk assessments however are generally based on estimation of risk in adults and ignore the potential impacts on the foetus and developing young.

In estimating the probability of health impacts, many uncertainties appear at every stage of the risk assessment process. For instance, there is uncertainty in estimating the quantities of releases, in estimating the transport and transformation of pollutants in the environment and, from this calculation, estimating human exposure. It is indeed extremely difficult, if not impossible, to determine the actual doses involved in environmental exposures. In order to overcome uncertainties in estimations, risk assessors use "conservative" estimates and so assume that overestimating risks overcomes these problems and is therefore protective of public health. However, the notion of "conservative" is ill-defined and in practice raises significant questions concerning exactly how conservative a risk assessment should be and whether all uncertain parameters should be conservatively treated or just a selected few. In this way it becomes apparent that risk assessment not only contains many uncertainties, but it is also a subjective rather than a scientifically objective process. This again calls into question whether regulations derived from risk assessments can be truly protective of human health.

With regard to incineration and risk assessment, the National Research Council (NRC 2000), noted that the procedures used to perform risk assessment "*vary widely, from a snap judgment to the use of complex analytic models,*" and described other related difficulties:

"The committee's evaluation of waste incineration and public health was substantially impaired by the lack of available compilations of ambient concentrations of pollutants resulting from incinerator emissions. In addition, large variabilities and uncertainties associated with risk-assessment predictions often limit the ability to define risks posed by incinerators.... Emission data needed to fully characterize environmental concentrations for health-effects assessments are not readily available for most incineration facilities. Such information is lacking especially for dioxins and furans, heavy metals (such as lead, mercury, and cadmium), and particulate matter.... Generally, data are not collected during startup, shutdowns, and upset conditions – when the greatest emissions are expected to occur. Furthermore, such data are typically based on a few stack samples for each pollutant. Thus, the adequacy of such emissions data to characterize fully the contribution of incineration to ambient pollutant concentrations for health-effects assessments is uncertain."

A further point on the estimation of releases with regard to incinerators is that data are usually based on test burns that are carried out under optimal conditions. It is likely that such data underestimate releases under operational conditions (see further section 5.1.1), (Webster and Connett 1990).

Risk assessments on incinerators generally focus on only one or a few substances that are known to be emitted, in particular dioxins and selected heavy metals. However, in reality, releases from incinerators consist of complex mixtures of hundreds of chemicals including many unknown compounds with unknown toxicity. Risk assessment omits to take into account health impacts of many of the known chemicals and all the unknown chemicals (Johnston *et al.* 1996). In addition, in looking at just single chemicals, it does not address the issue of the combined toxicity of the chemical mixtures in stack emissions (Johnston *et al.* 1998). For instance, the combination of two or more chemicals together may cause an additive or even greater than additive (synergistic) effect or a less than additive (antagonistic) effect.

A further problem in risk assessment is that it is very difficult to determine which are the most appropriate and sensitive endpoints for detecting the toxicity of chemicals. An adverse effect on the immune system or respiratory system may, for instance, be more sensitive and be instigated at lower chemical concentrations than another sort of health impact. For health risk assessment on incinerators, toxicological endpoints can include both cancer and non-cancer effects on health. Whatever endpoint is chosen, it is thereby accepted as a key metric capable of being used to protect human health in an holistic manner. It is questionable, however, whether the correct endpoints are ever used in risk assessment. In addition, as discussed above, the developing young are likely to be more sensitive to some adverse chemical effects than adults.

In summary, there is a dauntingly wide spectrum of inadequacies and uncertainties inherent in the process of risk assessment, from the estimation of type and quantity of pollution, to estimates of exposure and health effects. Each of these problems alone can fatally compromise risk assessment procedures. It is particularly important that these limitations are recognised when risk assessment is applied in the formulation, implementation or enforcement of regulations. Risk assessment should be viewed with deep scepticism unless all the areas of uncertainty are explicitly defined (Johnston *et al.* 1998). Risk assessments that have been performed for incinerators are briefly discussed in section 3.3.

2. OCCUPATIONAL HEALTH IMPACTS

2.1 Exposure

Municipal incinerator workers have considerable exposure to incinerator ash and this raises the possibility that they might absorb significant quantities of dioxins, and other toxic substances present in ash. The greatest potential for exposure to the toxic components of ash occurs during ash cleaning operations (Schecter *et al*. 1991). Respiratory personal protective equipment is designed to give workers protection from pollutants although it is important to note that dioxins and many other contaminants are also absorbable through the skin.

According to the NRC (2000):

"*Incinerator operators and maintenance workers, and those involved in the collection, transport, and disposal of fly ash and emission control equipment residues, have the potential to be most exposed to toxic substances associated with incineration*."

Noting that "*incinerator workers have been exposed to high concentrations of dioxins and toxic metals, particularly lead, cadmium, and mercury,*" the NRC assigned its highest level of concern to incinerator workers, irrespective of the implementation of maximum achievable control technologies (NRC 2000).

Studies on exposure of incineration workers are limited in number and in their focus. The majority have investigated exposure to dioxins and a few heavy metals whilst two studies investigated other organic compounds. Nearly all the studies investigated MSW incinerators but not other types of incinerator.

2.1.1 Dioxins

Research has indicated that incinerator workers can be exposed to elevated levels of dioxins in workplace air. Studies published on incinerator workers themselves during the 1990s also implied that workers have suffered from exposure to dioxin levels in the workplace that were above background levels. Specifically, some studies reported elevated levels of dioxins (total TEQ) in workers' blood. Other studies did not find an increase in total dioxins but did find an increase in certain dioxin congeners. Overall, these studies demonstrated that workers at incinerators could be subjected to increased exposure to dioxins.

A study in the US at a refuse-derived fuel, coal co-fired incinerator showed that incinerator workers are exposed to higher than background levels of dioxins in air in the workplace (Pilspanen *et al.* 1992). Levels of dioxins in workplace areas were higher than in other ambient air concentrations from the region. The source of these pollutants was thought to be associated with particulate matter emitted through combustion backpressure or leakage from the boiler, and subsequently carried by air currents through the plant. Commenting on this study, Marty (1993) notes that it demonstrated that worker exposure to dioxins is considerable in the incinerator occupation compared with exposure of the general population. The National Institute for Occupational Safety and Health (NIOSH) investigated three MSW incinerators in New York in 1992. They determined that airborne concentrations of dioxins during a cleaning procedure (of the lower chamber) were high enough to exceed the protection capabilities of the protective respiration equipment worn by the workers during this operation (NIOSH 1995). The study concluded that cleaning operations at the incinerators poses a health hazard.

Kitamura *et al*. (2000) demonstrated that the average dioxin concentrations in the blood of workers was 3.7 times higher than the levels measured in nearby residents at an incinerator in Japan. The incinerator operated between 1988 until 1997 when it was closed due to high stack emissions of dioxins and contamination of local soil. The study measured dioxin concentrations in the blood of 94 workers. The level of dioxins in blood samples ranged from 13.3 to 805.8 parts per trillion (ppt) TEQ, with a mean value of 93.5 ppt TEQ (lipid basis). Workers who had cleaned the inside of incinerators had the highest dioxin levels. In addition, contact with fly ash was identified as an important factor for high dioxin levels in blood. Workers who had no direct contact with fly ash had lower blood levels (average 34.2 ppt TEQ). In comparison, the average concentration of dioxins determined in blood from residents living within 2 km of the incinerator was 25.3 ppt TEQ. Co-planar PCBs were also monitored in the study. Some workers showed high PCB levels (range 3.1 to 54.2 ppt TEQ) and the study noted that environmental measurement of PCBs might be necessary. Health impacts observed in workers from this study are discussed below in Section 2.2.2.

A study at a municipal waste incinerator in Germany was undertaken in which blood samples were taken from 56 male workers, and the level of dioxins was measured in a pooled (combined) blood sample (Schecter *et al.* 1991). The sample was compared to a pooled blood sample taken from a control group of males who had no known exposure to toxic materials. Results showed that the incinerator workers had a 30% higher level of dioxins (total TEQ) in blood compared to the control group. Dibenzofurans were particularly elevated (103 ppt in workers versus 47 ppt in controls). In addition, the congener profile (pattern of levels of the different dioxin congeners) in the workers blood was similar to the congener profile of incinerator ash. This implied, together with the elevated levels of dioxins in workers blood, that workers had been exposed to dioxins

in the workplace. These findings subsequently led to the implementation of more stringent worker protection methods at the plant (Schecter *et al.* 1994).

A study in Japan on two workers who had been employed for over eight years, at what is now an obsolete MSW incinerator, found blood levels of dioxins were still elevated several years after their employment ceased (Schecter *et al.* 1999). The concentrations of dioxins in the workers blood was extremely high (360 ppt TEQ and 278 ppt TEQ, lipid basis). By comparison, the average Japanese blood level is 24 ppt TEQ, making levels in the two men to be 15 and 11.5 times higher than the general population. Dibenzofurans accounted for most of the TEQ. The incinerator had burned household waste including PVC and other plastics. It is of note that elevation of dibenzofurans rather than of dibenzodioxins has been reported more frequently for such combustion processes. One of the workers wives also had elevated dioxins in her blood (98 ppt TEQ), but the wife of the other worker did not have elevated levels (18 ppt TEQ). It was proposed that the high level in one of the women might have resulted from her washing contaminated clothing brought home by her husband.

In contrast to the above studies, research at some incinerators has not always found elevated levels of total TEQ dioxins in workers blood, but has nevertheless found elevations in certain dioxin congeners, in particular, hexa- and hepta-dibenzofurans. A study of 10 workers from a MSW incinerator in Germany (Papke *et al*. 1993) found elevated levels of these congeners together with total dioxins in some of the workers. Another study on four workers from a MSW incinerator in the Netherlands reported elevated levels of these congeners as well as of hepta- and octa-dibenzodioxins as compared to the blood fat of five local residents (van den Hazel and Frankort 1996). The values reported show that the average concentration of hepta-dibenzodioxins was about three times higher in the workers; octa-dibenzodioxins, 1.7 times higher; hexa-dibenzofuran, almost two times higher; and hepta-dibenzofuran, 1.9 times higher.

A study on 31 workers from 3 chemical waste incinerators in Germany, by contrast, did not find elevated levels of dioxins in workers blood (Papke *et al.* 1994). Levels of dioxins in blood were within the normal range. However, two workers were identified who showed elevated levels of hepta- and hexa-dibenzofurans as described in the studies above.

2.1.2 Other Organic Compounds

A study was undertaken on workers at an incinerator in Germany regarded as having modern health and safety standards (Wribitzky *et al.* 1995). This investigated 45 workers who were in contact with the incinerator and others who were not, namely 54 periphery workers and 23 management workers. The biological exposure limits set in Germany were not exceeded for benzene, toluene or xylene, although levels of these substances were elevated above general population levels for some employees. Significantly higher toluene concentrations, however, were detected in blood of incinerator workers relative to other employees. In addition, higher levels of chlorophenols were found in incinerator workers compared to other employees, suggesting occupational exposure among these workers. The degree of elevation of both toluene and chlorophenols were nevertheless small and were not considered by the authors as relevant to occupational health.

NRC (2000) reviewed a study by Angerer (1992) which investigated levels of various organic chemicals in the blood and urine of 53 workers from a MSW incinerator in Germany compared to 431 control subjects. The study reported elevated levels of hexachlorobenzene in plasma and of chlorophenols in urine. Workers had significantly higher levels of 2,4-and 2,5-dichlorophenol, and 2,4,5-trichlorphenol, and of plasma hexachlorobenzene (HCB). These chemicals were analysed because they are dioxin precursors compounds. Of other chemicals detected in urine, 4-monochlorophenol and tetrachlorophenol, were higher in the control group and there were no significant differences between workers and the control group for levels of plasma PCBs, blood benzene and urinary 2,4,6-trichlorophenol and pentachlorophenol. The NRC review commented that due to the lack of consistent findings between the worker and control groups, no conclusion could be drawn from this study on the exposure to combustion precursors of dioxins and hence no inferences could be drawn concerning exposure to dioxins. However, it is possible that workers suffered increased exposure to PAHs based on evidence of a biomarker of exposure (see section 2.1.4).

2.1.3 Heavy Metals

Studies have been carried out in order to investigate whether workers may be exposed to elevated levels of some heavy metals in air in the workplace. Three studies on incinerator workers have suggested that workers had experienced increased exposure to certain heavy metals, but one study, by contrast, found little evidence of increased exposure.

NRC (2000) describe a study performed by the National Institute for Occupational Safety in 1992 at three MSW incinerators in New York that investigated levels of heavy metals in the workplace (NIOSH 1995). The airborne concentrations of aluminium, arsenic, cadmium, lead and nickel during some periods of the clean-out of the electrostatic precipitator were high enough to exceed the

protection capabilities of the air purifying respirators worn by the workers during these operations. This led to the conclusion that working during these clean-out operations at incinerators poses a health hazard.

In 1989, a study on incinerators in New York found that, in some cases, workers were exposed to high levels of lead in air. Following this report, a study was conducted on the levels of lead in workers blood at three New York incinerators (Malkin *et al.* 1992). Results showed that the average lead level was statistically significantly increased in workers compared to a control group of workers (mean 11.0 mg/dl versus 7.4 mg/dl), although it did not exceed the maximum US limit that is deemed acceptable in the workplace of 40 mg/dl. The study suggested that the presence of lead in incinerator ash is capable of increasing lead concentrations in workers blood. In contrast, a study on a German incinerator with very modern health and safety standards did not find that lead was elevated in workers blood (Wrbitzky *et al.* 1995).

A study by Bresnitz *et al.* (1992) was conducted on exposure to heavy metals and health impacts amongst 86 MSW incinerator workers in the US. This revealed that these substances were generally not elevated in the worker's blood and urine. Only 8 of the 471 tests for heavy metal exposure showed levels of heavy metals that were elevated above the expected levels for an unexposed population. These included zinc, mercury and lead. However, the elevated levels were not related to workers exposure categories and were not deemed to be clinically significant.

A study on a hazardous waste incinerator in Finland assessed levels of mercury in 11 workers in 1984 and again in 1994 (Kurttio *et al.* 1998). Results showed that during the ten-year period, mercury levels, measured in hair, increased in the workers from 0.62 to 0.98 mg/kg. However, mercury levels were not high in workers in comparison to unexposed population levels worldwide (0.5 – 4.0 mg/kg in hair). A study of workers at an incinerator in Germany that has notably high safety standards also did not find that background levels of mercury in workers were exceeded significantly. Nonetheless, it was found that concentrations of arsenic were elevated above background in several individuals from both the incinerator itself and in periphery and management workers. (Wrbitzky *et al.* 1995). Concentrations were highest in the incinerator workers. The study concluded that the elevation probably occurred as a result of exposure at work. Arsenic is highly carcinogenic and the study commented that the source must be identified so that exposure to workers could be reduced.

2.1.4 Biomarkers

Two studies used biomarkers to investigate exposure of workers to hazardous chemicals. One study indicated that workers might be exposed to increased levels of PAHs and another showed that they might be exposed to elevated levels of electrophilic compounds such as PAHs.

NRC (2000) reviewed a study by Angerer (1992) which investigated levels of various organic chemicals in the blood and urine of 53 workers from a MSW incinerator in Germany compared to 431 control subjects (see also above, section 2.1.2). The study reported elevated levels of hydroxypyrene in the urine of workers. Hydroxypyrene is known to be a good indicator of internal exposure to PAHs. The results showing higher hydroxypyrene in workers urine suggest, therefore, that workers had suffered higher exposure to PAHs.

A study at a chemical waste incinerator was conducted which examined thioether concentrations in the urine of workers (Van Doorn *et al.* 1981). Thioethers excreted in the urine are the end products of detoxification of electrophilic compounds in the body such as PAHs and benzene. They can therefore be used as a biomarker to indicate the extent to which an individual has been exposed to electrophilic compounds. The study tested the urine of 3 workers from the incinerator both before work and after work. Results were compared to urine samples taken at the same times from non-exposed men at the plant. The study revealed that thioethers in the urine of the incineration workers was consistently higher following work compared to the start of the working day. This pattern of thioether excretion was not observed in the non-exposed control subjects. Moreover, the level of thioethers in urine in the incineration workers after work was consistently higher than thioethers in control subjects. From these results the study concluded that incineration workers were likely to inhale or absorb electrophilic compounds while at work, which were subsequently metabolised and excreted as thioethers in urine.

2.1.5 Mutagenic Compounds

Mutagenic compounds, also termed mutagens, are compounds that have the ability to damage DNA in cells of the body. Studies have shown that mutagenic compounds are present in incinerator emissions to air and in incinerator ashes. According to Ma *et al*. (1992), mutagenic compounds that are present in incinerator releases of gases, particulates and in ashes inevitably result in the exposure of incinerator workers to these compounds. One study on incinerator workers, discussed in this section, indicated that internal exposure to mutagenic compounds in the workplace might occur.

Laboratory studies have shown that incinerator emissions to air (e.g. Fomin and Hafner 1998, DeMarini *et al.* 1996), and fly ashes and bottom ashes (e.g. Shane *et al.* 1993), are mutagenic. Incinerator emissions to air consist of a complex mixture of organic chemicals and other elements. Within the organic fraction, it has been proposed that mutagenicity may be due to just one or a few chemical classes present in the chemical mixture. A recent study identified PAHs and nitroarenes (nitro-aromatic hydrocarbons) as being important contributors to the mutagenicity of incinerator emissions (DeMarini *et al.* 1996).

To investigate exposure to mutagens in workers, Scarlett *et al*. (1990) conducted a study to determine whether mutagenic compounds were present in the urine of incinerator workers. The study found that municipal waste incineration workers had a significantly increased frequency of mutagenic compounds in their urine compared to a control group of water treatment workers. Further research on workers from the same incineration plants was conducted to determine whether the quantity of mutagens in urine was consistently elevated in the workers or varied with time (Ma *et al.* 1992). The first in a series of tests again showed significantly increased mutagens in the urine of incinerator workers compared to water treatment workers. However, further tests showed lower frequencies of mutagens in urine of the incinerator workers. At the same time, the quantity of mutagens in the urine of water workers remained consistent. The study noted that likely explanations for lower amounts of mutagens in the urine of incineration workers on second and third testing was that exposure to mutagens in the incinerator workplace is highly variable. Alternatively, or in addition to this, the workers who were being investigated may have taken extra measures to reduce their exposure by wearing protective clothing and masks after they suspected that they were being exposed to toxicants. Although mutations play a role in the process of carcinogenesis, the study noted that the presence of mutagens in the urine of incinerator workers did not establish per se, that mutations were taking place. Hence, it was not possible to relate these findings to an evaluation of the likelihood of cancer or other adverse health outcome through this particular study although the findings are suggestive.

2.2 Health Impacts

Studies on mortality and morbidity (illness) among incinerator workers are very limited innumber. Research has reported a broad range of health impacts associated with working atincinerators including death from heart disease and certain cancers and hyperlipidemia, allergy and hypertension. One study reported chloracne in a highly exposed worker, a condition specifically associated with dioxin exposure.

2.2.1 Mortality

Gustavsson (1989) investigated mortality among 176 incinerator workers who were employed for one year or more between 1920 and 1985 at a municipal waste incinerator in Sweden. The study noted that the working environment at this incinerator was more contaminated than would be expected at modern incineration plants. Results revealed an excess of ischemic heart disease. It was calculated that the excess was caused by occupational factors. The excess was found to be highest and (statistically significant) in workers with more than 40 years exposure.

An excess of deaths from lung cancer was also found in the study. Compared with national rates of lung cancer in Sweden, the workers had a 3.5-fold higher probability of dying from lung cancer. Compared with local lung cancer rates they had a 2-fold higher likelihood of dying from the disease. The small sample size in the study precluded fully conclusive statements with respect to the statistical significance of the rate of lung cancer among workers (Marty 1993). Gustavsson *et al.* (1993) point out, however, that analysis of the exposure time and latency period suggested that it was improbable that the high rates of lung cancer among workers were due to smoking. Additionally, according to calculations from previous studies, only very excessive smoking habits (100% smokers) could produce the size of cancer excess seen in this study. It was noted that exposure of workers to polycyclic organic compounds, especially PAHs, may have been be an important factor in the lung cancer excess.

The study also found a 1.5-fold increased probability of death from oesophageal cancer. Taken in the context of the study on its own, the evidence for an occupational origin of oesophageal cancer was weak. However, other studies on workers in Sweden exposed to combustion products, for example, gas workers, chimney sweeps and bus garage workers have also reported an excess of oesophageal cancer. Considering the results of excess oesophageal cancer among incinerator workers in the context of these other studies supports the existence of an increased health threat due to occupational exposure. It appears that this increased threat cannot be attributed to cigarette smoking and alcohol consumption which are both known contributory factors (Gustavsson *et al.* 1993).

In contrast to the above study by Gustavsson (1989), a study on 532 workers employed at two municipal waste incinerators in Rome, Italy, between 1962 and 1992 did not find an excess of lung cancer (Rapiti *et al.* 1997). Mortality from lung cancer was reduced in comparison to the general population and cancer mortality from all cancers combined was similar to that of the general population. However, there was a 2.79-fold increased likelihood of dying from

gastric cancer among the workers. This excess probability was evident for workers who had more than 10 years latency since first employment. Increased gastric cancer has also been observed among sewage workers and, to some extent, incinerator workers have similar occupational exposures such as to inhalation of volatile pathogens, bacterial toxins and organic dust. There are also other contributory factors for gastric cancer including alcohol consumption, deficient intake of fruit and vegetables and lower socio-economic status. Such factors could have been present in the incinerator workers in this study and may explain the excess of gastric cancers to some extent. The study concluded that incinerator workers deserve increased surveillance by means of epidemiological studies and the role of dust and bacterial toxins in waste management requires further investigation.

2.2.2 Morbidity

Kitamura *et al.* (2000) investigated morbidity in 94 municipal incinerator workers who had worked at an incinerator in Japan. The incinerator had operated between 1988 and 1997 when it was closed due to high stack emissions of dioxins which resulted in contamination of local soil. The study found elevated dioxin levels in the blood of workers (range 13.3 to 805.8 ppt TEQ, mean 93.5 ppt TEQ (lipid basis) (see also section 2.1.1). At blood dioxin levels above 100 ppt TEQ, a statistically significant relationship with hyperlipidemia was found. In addition, there was a marginal correlation between dioxin levels and allergy. However, the study noted that these health conditions were self-reported by the workers and that confirmation of the diagnosis may be necessary because there was no association between dioxin levels and plasma lipid levels.

Investigations of blood biochemistry did not find any significant associations with dioxin levels in the blood of workers although some decreased liver function was noted. Tests for the immune system revealed significant associations between dioxin blood levels and natural killer (NK) cell activity and PHA stimulation. Dioxin has previously been associated with effects on the immune system and the authors noted that consequently a follow-up study was necessary.

The study also investigated the sex ratio of children born to workers. Theoretically the number of female and male children born should be equal, but in reality there is a very slight male excess (see discussion section 3.2.3). In this study, when the workers were divided into high (greater than 49 ppt TEQ dioxins in blood) and low (less than 49 ppt TEQ) exposure groups, there were 16 boys and 17 girls born to the low exposure workers. This compared to 2 boys and 5 girls among the high exposure group. The slight excess in the number of females born in the high exposure group was not, however, considered to be statistically significant.

A study by Bresnitz *et al.* (1992) was conducted on morbidity amongst 86 MSW incinerator workers in the US. The study investigated several different health parameters. It divided the workers into two hypothetical groups, those considered to experience high exposure in the workplace and those with low exposure. Results revealed an excess of workers (31%) who had urinary abnormalities, namely significant proteinuria (protein in the urine). For this effect, however, no difference was apparent between the high and the low exposure groups. The prevalence of hypertension was also above normal among the workers and it was suggested that the hypertension might explain the increased prevalence of proteinuria. Tests on lung function in the workers showed that this parameter was affected by smoking status. Tests also suggested an increased potential for small airway obstruction of the lungs although the diagnosis was not confirmed. The high exposure group had a 19% increased likelihood of possible small airway obstruction of the lungs compared to the low exposure group. Among workers who had never smoked in the groups, there was an 85% increased potential for small airway obstruction in the high exposure group. The study concluded that additional studies were needed to assess the potential health effects of MSW incinerator by-products. The authors also suggested that increased efforts in reducing personal risk factors and potential occupational exposures were needed in order to reduce morbidity among incinerator workers.

Schecter *et al.* (1999) found particularly high dioxin levels in the blood of two men several years after they had worked at an old Japanese waste incinerator (see also section 2.1.1). One worker, who had a dioxin level of 360 ppt TEQ, had chloracne, a skin condition caused by exposure to dioxin. The other worker, who had a dioxin level of 278 ppt TEQ, did not have chloracne. At the time of the investigation he was recovering from two episodes of gastrointestinal cancer of unknown aetiology (medical cause).

3. HEALTH IMPACTS ON POPULATIONS LIVING NEAR TO INCINERATORS

Hens *et al.* (2000) note that incinerator release of pollutants to air and to water is effectively a dilution and dispersal of pollutants over space and time. This causes a slow, but gradual accumulation of pollutants in the food chain and the human body, such that health effects may often only become visible and measurable after a long latency period

3.1 Exposure Studies

A limited number of studies have been conducted to determine whether individuals residing near to incinerators have been exposed to pollutants. Studies are restricted to investigations of exposure to dioxins and heavy metals. Results of these studies are mixed. Some reported elevated exposure among nearby residents while others found no evidence of increased exposure.

3.1.1 Dioxins and PCBs

Three studies have reported increases in dioxin levels in residents living near to incinerators, while two studies have found no evidence of increased exposure. One study also reported that certain PCB congeners were possibly increased in the blood of child residents.

Gonzalez *et al.* (2000) investigated exposure in residents living in the vicinity of a newly built incinerator in Mataró Spain, both before and two years after the plant started running. The study determined the level of dioxins in 1995 and again in 1997 in pooled blood samples from 104 individuals living between 0.5 and 1.5 km from the incinerator and from 97 individuals in subjects living further away at between 3.5 to 4.0 km. In 1995, prior to start up of the incinerator, dioxin levels in blood of those living near and those living farther away were 13.5 ppt TEQ and 13.4 ppt TEQ, respectively. In 1997 after 2 years of operation of the incinerator, dioxin levels had increased in both groups of people by about 25% and PCBs increased by about 12 %. When further repeated analyses were conducted, the increase in dioxins among residents was of the order of 10 to 15%, rather than 25%. The increase in dioxins was not different in the residents living near to the incinerator and those living further away and the authors commented that the increase in dioxin blood levels was unlikely to be attributable to the incinerator. The incinerator's dioxin stack emissions were reported as 0.98-2.5ng TEQ/m^3.

A study in Japan was undertaken in an area close to a MSW incinerator for which high local dioxins in soil (see section 4.2.1), and an unusually high rate of cancer (2-fold excess) among residents, had been reported (Miyata *et al.* 1998). The study tested blood samples from 13 women and 5 men living within 2 km of the incinerator. Levels of dioxins were raised considerably in the residents compared to background levels found in the general population. For instance, women had an average blood level of 149 pg TEQ/g lipid and men 81 pg TEQ/g lipid, whereas the background level for the general population is in the range of 15 to 29 pg TEQ/g lipid. The authors commented that increased exposure in the residents was considered to be due to direct inhalation of dioxins from the stack gas of the incinerator and by intake of local vegetables contaminated by stack gas.

Following reports of high levels of dioxins in cow's milk at farms near to Coalite chemicals, in Derbyshire, UK where an incinerator was operating prior to 1991 (see also section 4.2.2), a study was undertaken on levels of dioxins in the blood of 10 residents from the farms (Startin *et al.* 1994). Results of the study revealed elevated levels of dioxins in blood of all the residents. Their blood levels were compared to available data on background dioxin levels for the German population since no relevant UK data were available. Three of the residents had blood levels (49, 85 and 95 pg TEQ/g lipid) which were just above or at the upper end of background levels, and the other 7 residents had levels (137-291 pg TEQ/g lipid) that were unmistakably higher than background levels.

Holdke *et al.* (1998) analysed levels of PCBs in the blood of 348 children between the ages of seven and ten years who lived near a hazardous waste incinerator in Germany. The results were compared to a control group of children who lived in a region with similar industrial pollution and with a second control group of children who lived in an area with lower industrial pollution. Among those who lived in the vicinity of the hazardous waste incinerator, PCB 170 and PCB 180 were present at statistically significant higher concentrations, while PCB 183 and PCB 187 were detected with greater frequency than among the control children from the area with lower industrial pollution. According to the study, while the results can only be viewed as a regional comparison of the three groups and the effects small, the statistically significant results do indicate a regionally plausible pattern.

Two other studies in Europe have found no increase in levels of dioxins of individuals residing close to incinerators. Deml *et al.* (1996) sampled blood from 39 persons and breast milk from 7 persons in 1993 who lived near to a MSW incinerator in Germany. The study reported that there was no indication of increased blood levels of dioxins in the residents. Levels of dioxins in the blood (mean 17.0 ppt TEQ lipid, range 5.2 to 34.5 ppt TEQ lipid) and breast milk (mean 12.4 ppt TEQ lipid, range 6 to 19 ppt TEQ lipid) of residents were not significantly different from background levels in the German population (range 10 to 48 ppt TEQ lipid in blood and mean 30 ppt TEQ lipid in breast milk).

Similarly, a study on exposure of a limited number of residents (five) living near to an incinerator in Duiven, The

Netherlands, did not find increased dioxin levels in their blood (van den Hazel and Frankort 1996). This study was undertaken specifically to test whether residents had elevated levels of dioxin congeners in their bodies because of their potential exposure to fly ash blown away from the storage site near the waste incinerator. Levels of dioxins in blood of the residents (mean 31.4 ppt TEQ lipid) were similar to levels in the control group of 5 individuals from the Dutch general population (mean 33.8 ppt TEQ lipid). In addition, the study did not find increases in the levels of any particular dioxin congeners amongst the residents.

3.1.2 Heavy Metals

Only one study was located in the scientific literature on exposure to heavy metals in individuals residing near to incinerators. Kurttio *et al.* (1998) investigated changes in the level of mercury between 1984 and 1994 in the hair of 113 individuals who lived near to a hazardous waste incinerator in Finland. Mercury concentrations were found to increase in workers at the plant (see section 2.1.3) and in residents such that levels increased with decreasing distance from the incinerator.

For example, levels increased by 0.16 mg/kg in residents living 1.5-2 km (high exposure group) from the plant, 0.13 mg/kg at 2.5-3.7 km (medium exposure group) and 0.03 mg/kg at about 5km (low exposure group). The results indicated that the incinerator was the likely source of exposure among residents. Exposure was most likely mainly due to inhalation and possibly via ingestion of local well water and vegetables. The authors concluded that the increase in mercury concentrations in the residents over time was small and, on the basis of current knowledge, did not pose a health threat.

3.1.3 Biomarkers

The theory behind using biomarkers in epidemiology studies relies on early biological effects of a toxic exposure, (i.e. the biomarker), being more prevalent and easier to detect in a potentially exposed population than clinical disease.

A study conducted at a recently built incinerator in Spain, compared children living in the vicinity to children living outside the zone of influence of the incinerator using urinary thioethers as a biomarker (Ardevol *et al.* 1999). The use of urinary thioethers as a biomarker relies on the fact that when electrophilic compounds such as PAHs are detoxified in the body, the metabolic end products can be detected as thioethers in the urine. Electrophilic compounds are generally potent mutagenic and carcinogenic compounds.

The study assessed the possible contribution of incinerator releases to urinary thioethers in children aged 7-10 during 1997. Research on children, rather than on adults, eliminates other potential effects on health that may interfere with study results, including smoking or occupational or other lifestyle toxicants. The study reported that there was a greater quantity of urinary thioethers in the urine of children living near to the incinerator than the control group of children living further away, although the result was not statistically significant.

The study also found that parental smoking of tobacco predicted a statistically significantly greater quantity of urinary thioethers in children of both groups. In addition, among children who were exposed to smoking by both parents in the home, there was a significantly greater quantity of thioethers in urine from the incinerator group compared to the control group. It is possible that this effect was caused by a greater extent of exposure to tobacco smoke in these children. Alternatively, it may have been caused by the addition of exposure to the tobacco smoke and exposure to incinerator releases. In the case of releases from incinerators, the higher amounts of thioethers in urine of the children may have been caused by exposure to PAHs and possibly dioxins.

3.2 Health Effects - Epidemiological Studies

The majority of epidemiological studies on the health of populations residing near to incinerators have focused either on incidence of cancer or of respiratory symptoms. Additionally, some research has investigated other potential effects including congenital abnormalities and changes in the sex ratio. Considering the widespread use of incinerators on a global scale, the number of studies that have investigated health effects in residents near to these facilities is sparse.

3.2.1 Cancer

Some of the substances emitted from incinerator stacks, including cadmium, PAHs and dioxin (TCDD), have been classified as human carcinogens or likely/possible human carcinogens by the International Agency for Research on Cancer (McGregor *et al.* 1998, see Elliot *et al.* 1996). A number of studies have been undertaken on cancer incidence on populations living near to incinerators or other industrial sites. The majority of these studies have found an association between elevated rates of cancers and living close to incinerators or other industrial sites, including childhood cancer. Most research in this field necessitates consideration of exposure to material released from incinerators over a number of years because the time taken for cancer to develop (the latency period) is long for many cancers.

Soft Tissue Sarcoma and Non-Hodgkin's Lymphoma

A study in the area of Doubs, eastern France, was

conducted to investigate clustering of two types of cancer, soft tissue sarcoma and non-Hodgkin's lymphoma, near to a MSW (Viel *et al.* 2000). The study was undertaken following a report of high dioxin emissions from the incinerator. The study found highly significant clusters of both cancers in areas close to the incinerator but not in other surrounding regions.

In a press release in 1998, the French Ministry of the Environment revealed that 71 MSW incinerators had dioxin emissions to atmosphere above 10 ng I-TEQ/m^3. One of the incinerators, Bescançon, had a reported dioxin air emission of 16.3 ng I-TEQ/m^3. A general cancer registry covered the area local to this incinerator and this provided an opportunity for researchers to study the incidence of cancer in the region. Soft tissue sarcoma and non-Hodgkin's lymphoma were selected for investigation because previous work has suggested that dioxins increase the probability of contracting these cancers. The incinerator had operated from 1971 onwards.

The study divided the region of Doubs into 26 areas (statistical units) for the purpose of the analysis. During the period 1980 to 1995, 110 cases of soft tissue sarcoma and 803 cases of non-Hodgkin's lymphoma were recorded. Analysis showed that statistically significant clusters of both cancers were present in 2 of the 26 areas, Bescançon and Audeux that were closest to the incinerator. There was a 44% increased incidence of soft tissue sarcoma and a 27% increased incidence in non-Hodgkin's lymphoma. No other clusters were found in the remaining 24 areas. Possible confounding factors of socio-economic status and urbanisation were discussed as unlikely to bias results. Furthermore, to make sure that distance to health centres did not confound the results (that is, due to closer residence and the consequent wider access to specialised care leading to more frequent cancer diagnoses), the study also considered the frequency of Hodgkin's disease as a control cancer. Hodgkin's disease is a cancer that is not associated with dioxin exposure. The study found no clusters of Hodgkin's lymphoma within the entire study area. The authors concluded from this that the clusters of soft tissue sarcoma and non-Hodgkin's lymphoma near the incinerator were not attributable to the presence of University Hospital in the Bescançon -Audeux area resulting in more reliable diagnosis of the diseases.

In conclusion, the authors stated that the consistency of the findings for clustering of soft tissue sarcoma and non-Hodgkin's lymphoma around the incinerator was remarkable. However, they cautioned that before clusters of both these cancers are attributed to dioxin release from the incinerator, the findings should be confirmed by further investigation. If dioxin is involved, the route of exposure among residents remains to be determined.

Lung Cancer

A study in Trieste, an industrialised city in northeast Italy, was undertaken to investigate the impact of air pollution from a number of sources (ship yard, iron foundry, incinerator and city centre) on lung cancer (Biggeri *et al.* 1996). It found that the prevalence of all types of lung cancer was increased both by residence close to the incinerator and to the city centre.

The method used in this study involved the identification of individual subjects who had died from lung cancer in the region, and subsequently, the identification of matched control individuals who had died at a similar time but not from cancer or lung disease. A total of 755 male individuals who had died from lung cancer between 1979 and 1981 or between 1985 and 1986 were identified. The two enrolment periods were selected to cover an extended period of time bearing in mind the costs of the study. Analysis of results accounted for confounding factors including smoking habits, age, likelihood of exposure to carcinogens in the workplace, and approximate levels of air particulate.

Results of the study showed there was a statistically significant increased probability of dying from all types of lung cancer among people living by the incinerator. The likelihood of dying from lung cancer for individuals living within a short distance of the incinerator was 6.7 times greater than for those living in other areas. Independently from this, living close to the city centre was also associated with a higher probability of death from lung cancer (2.2-fold higher at the city centre). This study confirmed the findings of a previous study in Trieste, which had also identified an increased likelihood of lung cancer in the neighbourhood of the incinerator (Babone *et al.* 1994). Some possible confounding of results through factors such as length of time at the death address, (i.e. change of residence), could not be excluded. The study concluded that the results provided further evidence that air pollution is a moderate contributory factor for lung cancer and that this was consistent with the hypothesis of an independent impact on health of residing close to the incinerator and to the city centre.

Cancer of the larynx

An investigation of cancer rates around an incinerator of waste solvents and oils at Charnock Richard, Lancashire, UK was undertaken during the late 1980's by the local council. Statistical analysis of the results identified a significant excess of cancer of the larynx close to the incinerator, which decreased as distance from the incinerator increased (Diggle *et al.* 1990). Following this report, another study was undertaken to investigate incidence of laryngeal cancer at this incinerator and at nine other similar incinerators in the UK which began operation before 1979 (Elliot *et al.*

1992). This study found no excess of larynx or lung cancers up to 10 km from the incinerator sites when lag periods (between incinerator start up and cancer incidence) of 5 and 10 years were used. Consequently, the study concluded that the apparent cluster of cases of cancer of the larynx at Charnock Richard, Lancashire, was unlikely to be due to the incinerator. However, there were several recognised limitations in the data used in this study. For instance, the lag time of 5 and 10 years for the development of laryngeal cancer is short considering the epidemiology of solid tumours. A study on mustard gas workers, for example, showed that cancer of the larynx was evident only after a follow-up of at least 10 years since first employment and another study showed excess mortality from larynx cancer in workers exposed to dioxin was only apparent after 20 years.

A more recent study on the incidence of various cancers in a population living within the vicinity of an incinerator, a waste disposal site, and an oil refinery plant in Rome which had operated since the early 1960s, did find an increased probability of mortality from cancer of the larynx (Michelozzi *et al.* 1998). The investigation was undertaken following concerns about pollution from the industries affecting the resident population. No excesses were found for liver, lung and lymphohaematopoietic cancers. However, an increased likelihood of cancer of the larynx was evident at 0-3 and 3-8 km distance from the industries, although the result was not statistically significant. Nevertheless, the authors hypothesised a possible link between emissions from the industries and larynx cancer since there was a statistically significant decline in this cancer among men with increasing distance from the industries. The study noted that this was interesting because results from other studies on cancer incidence in the vicinity of these industries were conflicting. It concluded that the results for laryngeal cancer were based on a limited number of cases, and further studies would be necessary to determine whether the presence of refineries or incinerators actually does represent a factor contributing to an increased probability of contracting this disease in resident populations.

Liver Cancer and Other Cancers

A study was undertaken on cancer incidence in individuals living near to incinerators in Great Britain as a consequence of concerns about possible health effects of residing close to these facilities (Elliot *et al.* 1996). The research showed a statistically significant excess of liver cancer among residents.

The study investigated cancer incidence among over 14 million people living within 7.5 km of 72 MSW incinerators. Data on cancer incidence among residents from 1974-1987 was compiled using the national cancer registration scheme. Cancer incidence rates for populations living close to incinerators were compared with national cancer rates to determine whether there was excess of observed numbers of cancer cases versus expected numbers. Results showed that there were statistically significant excesses of all cancers combined and of stomach, colorectal, liver and lung cancer for populations living within 7.5 km of incinerators. The incidence of cancer decreased with increasing distance from the incinerators. The greatest increased probability of occurrence was found for liver cancer, for which there was 37% excess from 0 to 1 km distance from incinerators compared to national rates. However, further analysis of the data indicated that the excesses of all cancers combined and of stomach and lung cancer were likely to be due to confounding, in this case social deprivation. Social deprivation tends to be high in polluted areas and it is strongly predictive of disease occurrence. With regard to liver cancer it was noted that social deprivation could account for at least part of the increased likelihood of contracting this disease as observed in the study. It was also noted that there was some misdiagnosis of primary liver cancer due to secondary liver tumours (i.e. tumours arising subsequently to, and as a result of, other types of primary tumours). The study concluded that further research would be needed to confirm whether or not there was an excess of primary liver cancer in the vicinity of incinerators. Further work on the diagnosis of liver cancer in this study was subsequently carried out (Elliot *et al.* 2000). It also indicated an increase in the rate of liver cancer in residents living near to incinerators.

The initial study (Elliot *et al.* 1996) utilised information recorded on death certificates. For further analysis of the data, the second study, (Elliot *et al.* 2000), included a review of histology slides and reports and medical records in order to clarify whether liver cancers were primary cancers or secondary cancers. Out of the 235 original cases of liver cancer recorded on death certificates a review of 119 cases (51%) was performed. Primary liver cancer was confirmed in 55% of these cases and secondary cancers in 18%. If these figures are used to re-calculate the incidence of liver cancer in the first study, the excess of 37% liver cancers (23 cases) reported in the first study is reduced to 12.6 cases, and 18.8 cases when only definite secondary cancers are excluded. This translates to 0.53 and 0.78 excess cases per 1,000,000 cases per year (or an increased probability of contracting liver cancer of 20 and 30% within 1 km distance of MSW incinerators). The study concluded that the true excess would be expected to lie somewhere between these two figures. The study could not rule out the possibility of confounding on the results from social deprivation. Elliot *et al.* (2000) commented that if the findings of excess liver cancers in this and the

previous study were caused by residence near to MSW incinerators, then the results related to historical exposure patterns around these installations.

Childhood Cancer

An analysis by Knox (2000) has recently been published that used data on MSW incinerators from the initial study by Elliot *et al.* (1996), (see above), to determine whether the likelihood of contracting childhood cancer is increased near to incinerators. The study considered childhood cancer around 70 MSW incinerators between 1974-1987, and 307 hospital waste incinerators between 1953-1980. Latent intervals for childhood cancers are short and this mitigates problems of the often-longer latency periods for cancer in adults encountered in the "all ages" study by Elliot *et al.* (1996). The analysis used a newly developed, sensitive method that could consider the distance of the birth address of each child from an incinerator and also the death address, if this was different. In this regard, the "migration method" used for this analysis could compare distances from incinerators to the birth addresses and to death addresses of children with cancer who had moved house. The study identified an increased incidence of child cancer in children who were born near to incinerators.

The developmental stages of life are generally the most vulnerable to toxic insult. Thus, exposure of the developing foetus in the womb and during the early stages of life to toxic substances may lead to a greater potentila for adverse effects on health, such as cancer, than exposure in later years. In the study by Knox (2000), if exposures to toxic substances from living near to an incinerator during the early stages of life predicted an increase in cancer incidence, then there would be closer associations with the birth address of children rather than with the death address.

Results of the analysis indeed showed a highly statistically significant excess of migration away from birthplaces close to incinerators within 5 km of the sites. Thus, exposure to incinerators at the birth address, and hence during the early developmental stages of life, was associated with a higher probability of contracting cancer than exposure at the death address or exposure in later years. For children who were born within a distance of 5 km of MSW incinerators, there was a 2-fold increase in the probability of dying from cancer.

These results are also in agreement with previous research which showed an increased likelihood of childhood cancer in children who were born within a short distance of hospital waste incinerators, large-scale high temperature combustion sources or installations emitting VOCs (Knox and Gilman 1998). The excess numbers of leukaemias and solid tumours of all types were similar to those found in the study on childhood cancer in the vicinity of MSW incinerators by Knox (2000). This phenomenon was also observed in previous studies on proximity of childhood cancer to industrial sites and exposures to pre-natal medical radiation. Such a result would be expected for agents/chemicals that have systemic access (i.e. access via the circulation) to the DNA/RNA in all types of foetal cells (Knox 2000).

The study conducted by Knox and Gilman (1998) on the level of childhood cancers near to many different industries concluded that increased cancer rates were apparent in children born near to hospital incinerators, other combustion sources and industries emitting VOCs. From these results it was concluded that multiple toxic sources are responsible for many birth –time or pre-birth (foetal) initiations of cancer. This effect on the developing young is likely to be mediated through various VOCs and products of combustion. With regard to waste incineration itself, concordance of results on childhood cancer from MSW incinerators (Knox 2000) and hospital waste incinerators (Knox and Gilman 1998) suggests a common direct effect of being born near to incinerators and childhood cancer. Knox (2000) noted, however, that it is difficult to say whether the apparent cancer-related threats to health near incinerators might also stem from other hazards in the nearby environments. In this regard, the most "toxic" incinerators in the study were close to industrial sources of kinds implicated in previous studies. For this reason, the present overall conclusion of the study was that the increased probability of childhood cancer stems from residence near to large-scale combustion processes as a whole, of which incinerators are one component (Knox 2000).

3.2.2 Respiratory Effects

Incinerators, in particular cement kilns, emit considerable quantities of SO_2 and NO_2. Long term exposure to these substances is known to have negative impacts on respiratory health (see e.g. Ayres 1998). Similarly, incinerators emit fine particulate matter and many studies have reported that long term exposure to particulate matter is associated with adverse effects on respiratory symptoms (see further appendix A). Despite the potential negative impacts on respiratory health from substances known to be emitted from incinerators, there have been only a limited number of epidemiological studies on respiratory effects in individuals who live near to incinerators. Of the research that has been undertaken, some studies have suggested negative impacts on respiratory health whilst others have found no effect.

An early study by Zmirou (1984) suggested there was an increased use of medication for respiratory illnesses among residents living near to a MSW incinerator in a French village. The study found that medicines for respiratory

problems such as bronchodilators, expectorants and cough medicines were purchased significantly more often at distances closer to the incinerator. The investigators pointed out that it is not possible to conclude a cause-effect relationship from this study, but stated that the observation made by the study is consistent with the hypothesis that pollution generated by the incinerator may exacerbate respiratory problems (see: Marty 1993).

At a hazardous waste incinerator in western North Carolina, US, an investigation of the health of nearby residents was conducted following reports of illness and neurological symptoms in workers employed at the plant (ATSDR 1993). After adjustment for confounding factors such as age, sex and cigarette smoking, the study found significant increases in the prevalence of self-reported respiratory symptoms. For instance, residents living near to the incinerator were almost nine times more likely to report recurrent wheezing or cough compared to residents living further from the site, and they were almost two times more likely to report other respiratory symptoms. In addition, chest pain, poor co-ordination, dizziness and irritative symptoms were also significantly elevated. However, no difference between the two groups was reported for the prevalence of physician-diagnosed diseases and hospital admissions for these diseases. Although this study found an increase in respiratory symptoms among residents living near to an incinerator there are several concerns surrounding the study, which limit the interpretation of the findings. For instance, according to a review of this study by NRC (2000), there are concerns regarding the retrospective nature of the study (incinerator operated from 1977 to 1988 and the study was not conducted until 1991), and regarding adverse publicity before the incinerator shutdown. The NRC commented that the study was of limited utility in evaluating the effect of incinerator exposures.

A study in Taiwan investigated the respiratory health of children living near a wire-reclamation incinerator and reported associated adverse effects on lung function (Wang *et al.* 1992). The study tested 86 primary school children and compared the results to a control group of 92 children from a "non-polluted" city. Air pollution in the incinerator district as measured by SO_2 and NO_2, was notably greater than in the comparison city. Questionnaires administered to the children revealed no differences in respiratory symptoms. However, abnormal forced expiratory volume in 1 second (FEV1), a measure of lung function, was significantly greater in the incinerator group (17.5%) than the control group (3.2%). Further testing of lung function of 26 children from each group revealed a positive methacholine-challenge test in 9 of the incinerator group but only 1 of the control group. From these results, the authors concluded that the high level of air pollution to which the children living near to the incinerator were exposed was associated with a detrimental effect on lung function. A review of this study by NRC (2000) noted that the study appears to demonstrate that higher concentrations of air pollutants alter lung function in children, but does not directly allow any inference about the contribution of incinerators as opposed to other pollutant sources to the observed health effects.

In the US, it has become common practice for hazardous wastes to be used as part of the fuel to achieve high temperatures during cement kiln operation. A study conducted on cement kilns operating in Midlothian, Texas, documented statistically significant increased self-reported respiratory symptoms among a sample of nearby residents as compared to residents living further away (Legator *et al.* 1998). Risk assessments based on measurements of incinerator emissions recorded in 1997/8 in the area had reported that there was no threat posed to human health from the cement kilns (see Legator *et al.* 1998). In addition, a study by the Department of Health in the region in 1992 concluded that there

"*did not appear to be any consistent patterns of illness or symptoms that might be an indication of …a common-source health problem among the study respondents*".

However, subsequent analysis of both the above pieces of research concluded that there were deficiencies, flaws and inadequacies in the methodology of the studies. Since then, Legator *et al.* (1998) have conducted research aimed at identifying whether exposure of nearby residents to pollutants from the cement kilns resulted in adverse health effects.

A randomised sample of 58 individuals living near to the incinerators was selected and requested to undergo an interview based on an extensive health questionnaire. The results were compared to a control group of 54 individuals who lived further away from the incinerators. The study showed that no areas of health appeared to be significantly adversely affected with the exception of the respiratory category. The population living close to the incinerators reported significantly higher frequencies of respiratory symptoms ($p = 0.002$) than the control group. All of the respiratory symptoms on the questionnaire, including lung disease, wheezing, emphysema, persistent cough and bronchitis were elevated, the only exception being pneumonia. The study was not subject to some of the limitations that can hinder studies of this nature such as biases on questionnaire reporting. In addition, the control population was older than the incinerator-exposed population and since older individuals are more sensitive to the effects of chemicals, it is likely that impacts on the incinerator-exposed population were underestimated. The

study concluded that the results add to the growing body of information that persons living next to incinerator-generated air pollution experience increased prevalence of respiratory symptoms.

Gray *et al.* (1994) conducted a study on the incidence of asthma among two groups of children living near sewage sludge incinerators in Sydney, Australia. Respiratory illness was monitored by way of questionnaire, and various physiological tests, including tests on lung function. The study found no adverse effect on either the prevalence or the severity of asthma in the children compared to a control comparison group of children living in a different region of Sydney. In addition, no differences were apparent in lung function. Measurements of SO_x, NO_x, hydrogen sulphide, ozone (O_3) and particulate matter did not detect any statistically significant differences between the incinerator and comparison areas. The study concluded that releases from high-temperature sewage-sludge incinerators appeared to have no adverse effect on the prevalence or severity of childhood asthma.

A study in the US reported no significant difference in levels of particulate air pollution or respiratory health in communities residing near to three waste incinerators from 1992 to 1994 (Shy *et al.* 1995). The study was conducted on three communities living near to a municipal, hazardous and medical waste incinerator in North Carolina US and three comparison communities that were more than 3 km upwind of incinerators. The study simultaneously monitored air quality in the communities and respiratory health effects in individuals.

In all, 6963 individuals participated in a telephone survey of respiratory health over the course of 3 years and 100-144 individuals per community per year participated in having tests on lung function. Shy *et al.* (1995) reported results for the first year of the three-year study. The study found no significant difference in the concentration of particulate matter (PM10) in the incinerator communities relative to the comparison communities. Incinerators were calculated to contribute to less than 3% of particulate matter measured in the communities, the remainder coming from other sources around the region. However, consistently higher levels of particulate zinc, lead, and chloride were found in incinerator communities when winds were coming from the direction of the medical and MSW incinerators. The study noted that if a chemical component of incinerator releases can cause respiratory effects in an exposed community, standard measures of air pollution may fail to detect the relevant differences in human exposure.

With regard to respiratory health in the study, no significant differences were found in respiratory symptoms recorded by telephone survey between the incinerator and comparison communities. In addition, results of lung function from this study for 1992/3, plus a subsequent more in depth analysis of lung function (Lee and Shy 1999) did not find any relationship between particulate levels (PM10) in the communities and lung function. Furthermore, there were no apparent differences in lung function between the incinerator and comparison communities. This is in contrast to previous studies which have reported increased respiratory effects associated with increases in PM10 (see appendix A). The different results of this study may be because the particulate levels were relatively low by comparison with previous studies and therefore effects on lung function might not be possible to detect even if they existed (Shy *et al*. 1995).

Although the above study (Shy *et al.* 1995) did not find an association between living near to the three particular incinerators in the study and an increase in acute or chronic respiratory symptoms, it was emphasised that the study did have several limitations. For instance, there was more cigarette smoking and greater use of kerosene heaters in the homes of the comparison communities and this would tend to mask any moderate-sized respiratory effects in the incinerator communities. Also, a major problem was the possible significant misclassification of exposure to pollutants from the incinerators because different sections of the community were likely to experience different air pollution levels due to the prevailing wind direction. It was reported that this would tend to lead towards the result of no effect upon respiratory health. Further analysis of lung function results by Lee and Shy (1999) also noted that the study was limited by a lack of information on individual exposures to releases. The authors commented that the lack of association between PM10 and respiratory health in the study must be interpreted cautiously because exposure estimation based on monitoring of ambient air likely resulted in misclassification of true exposure levels.

3.2.3 Sex Ratio

In humans, the ratio of males to females born should theoretically be 1:1. But in reality there is a slight excess of males at birth. This may be attributable to a number of different factors including age of parents and time of insemination within the cycle (Moller 1996).

The sex ratio has been found to vary somewhat between different countries. Abnormal sex ratios have been associated with some occupational environments, for instance, an excess of male births was reported for tax experts and chartered accountants and an excess of female births was reported for librarians and quantity surveyors (see Williams *et al.* 1992). The mechanisms which cause

sex ratio to vary have not been clarified but a hormonal influence has been implicated in having an effect.

Recent research has indicated a decreased proportion of male births in the general population of Denmark, Netherlands, USA, Canada, and in sawmill workers who were exposed to trichlorophenate that was contaminated with dioxins. It has been hypothesised that these changes in sex ratio may be caused by exposure to chemical pollution. There is some evidence for this from studies on populations who have been exposed to dioxins. For example, a study on the population at Seveso, Italy, who were exposed to high levels of dioxin (TCDD) following an explosion at a chemical herbicide plant in 1976 has investigated sex ratio (Mocarelli et al. 2000). Individuals were included in the study who were exposed at the time of the accident, whether as a child or as an adult, and have subsequently had children of their own. Exposure was assessed by analysing blood samples taken around the time of the accident and kept frozen in storage. The results showed that increased levels of dioxin in the blood of the fathers increased the probability of siring female children. The concentration in the blood of the fathers at the time of the accident was about 20 times the estimated average concentration of TCDD currently found in humans in industrialised countries, although the levels in blood had fallen in some cases by the time children were conceived. The study showed that male exposure to TCDD before and during puberty is linked to this sex-ratio effect of siring more females. This indicates that the time before and during puberty may be a very sensitive time for dioxin toxicity in men. Men in adulthood at the time of exposure were also affected. Overall, the data showed that exposure of men to TCDD is linked to a lowered male/female sex ratio in their offspring, which may persist for years after exposure.

One study has investigated sex ratios in populations living near to incinerators. This study was carried out on residents living in the vicinity of two incinerators in Scotland, UK (Williams *et al.* 1992). For the purpose of the study, the area was hypothetically divided up into 16 different districts (by postcodes) including 6 districts further away from the incinerators, which were used as a comparison area in the analysis of results. No difference was found in the sex ratios of births among residents between the potentially exposed area (comprising 3 districts) and the comparison area. However, when districts were considered individually, the area identified as being most vulnerable to air pollution from incinerators had a statistically significant excess of female births. Another district considered to be potentially vulnerable also had an excess of female births and another had an excess of male births although these were not statistically significant. The study proposed that incinerator releases such as polychlorinated hydrocarbons, dioxins and pesticides, if present, might have affected the sex ratios. However, the authors noted that it is not possible to attribute causality of increased female births to materials released by incinerators this study alone and proposed that more research was needed.

3.2.4 Congenital Abnormalities

Research on populations living near to incinerators has reported an increased incidence of congenital abnormalities. One study in Amsterdam found increased numbers of orofacial clefts and other midline defects in a community living near to an incinerator site used for the open burning of chemicals. Another study near to an incinerator in Belgium found an increased prevalence of congenital abnormalities. Other research on congenital eye malformations has not detected an increased prevalence near to incinerators.

Between 1961 and 1969, a poorly regulated incinerator carried out open-burning of waste chemicals in Zeeburg, Amsterdam. A recent analysis of data from the region has revealed a dramatic increase in the incidence of orofacial clefts in babies born after incineration began (ten Tusscher *et al.* 2000). In comparison, no increase in orofacial clefts during the same time period was observed in births from another area that was unaffected by the waste burning. For instance, the average incidence of orofacial clefts in Zeeburg between 1961 and 1969 was 2.5 per 1000 births compared to 1.2 per 1000 births in the comparison area. In particular, for the years 1963 and 1964, the incidence of congenital abnormalities at Zeeburg was respectively 5.1 and 7.1 per 1000 births. This result for 1963/4 was statistically significantly different to the comparison area.

The residential location of many of the women who gave birth to babies who had orofacial clefts was found to be situated in an area along a corridor of wind-flow from the incinerator. It is known that chemical exposure can be a cause of orofacial clefts and it was noted that dioxin (TCDD) is known to cause cleft palate in mice. The authors concluded that although a cause-effect relationship cannot be proven in this case, it seems very likely that there is a link between the open incineration of chemicals and the increased incidence of orofacial clefts in Zeeburg , Amsterdam for the years 1960 to 1969. Furthermore, as well as orofacial clefts, the majority of babies born in Zeeburg with some other midline defects were born in the area corresponding to wind-flow from the incinerator. These conditions included central nervous system defects (mainly spina bifida) and genital defects (mainly hypospadias).

A cluster of congenital abnormalities was discovered among inhabitants of the Neerland neighbourhood in the Wilrijk region, Belgium. This stirred up unrest in the local community. The area is situated between two municipal

waste incinerators, one at a distance of 1200 meters and the other at a distance of 800 meters. Research had previously shown that the area around Wilrijk was among the regions in Flanders that received the highest dioxin deposition. This was due to the incineration of municipal wastes between 1980 and 1996. Following concern by residents about the cluster of congenital abnormalities, two health studies were ordered by the government and took place between 1997-1998. The first (Verschaeve and Schoeters 1998) investigated genetic damage in chromosomes in certain types of blood cells (peripheral lymphocytes), and the second investigated children's health (Aelvo *et al*. 1998). Van Larebeke (2000) recently reviewed these studies.

The first study, on chromosomal damage, compared 24 children from the area with a control group of 20 children from another neighbourhood in the Antwerp region. The study did not detect any differences in chromosomal aberrations between the two groups. However, van Larebeke (2000) commented that if genetic effects were present, they would be expected to be of low intensity. The study did not have the statistical sensitivity to be able to detect such effects at low intensity. Therefore it is possible that such effects could be present but would not have been found by the study. If present, such genetic effects could have a significant impact on health.

The second study assessed health problems of children from Neerland. The study found an increased incidence of congenital malformations in babies from Neerland compared with incidence within Belgium as a whole, although the result was not statistically significant. The probability of giving birth to a baby with congenital malformation was 1.26 times greater for Neerland women than for Flemish women in general. There was also an increased incidence in congenitally malformed babies born in Neerland compared to babies who were born in the same clinics but whose families resided elsewhere. The increased probability of having a congenitally malformed baby appeared to be confined to children born to parents who had not resided for a long time in Neerland.

In addition to congenital abnormalities, the second study also investigated performance and health of children from Neerland at school compared to other children from a nearby area and to Flemish children in general. No differences were apparent in failure at school. However, increased allergies and repeated episodes of the common cold were significantly increased in Neerland children in the third class of maternal school, and increased complaints on health in general were also more frequent. At the age of 9, in third class of primary school, there was a significant increase in allergies and "use of medication". Use of medication is considered to be an indirect measure of ill health caused by pollution.

Van Larebeke (2000) concluded in a review of the studies that a more detailed in-depth analysis of the health status of children from the Neerland neighbourhood might reveal other health effects possibly related to pollution. For instance, data on individual exposure determined as blood levels of dioxins were lacking in the study as were data on early (pre-symptomatic) biological effects. The present results were considered to be sufficiently indicative to warrant further study to include investigation of these points. Both incinerators were shut down in November 1997 due to their exceeding the dioxin air emission standards and consequent considerations of public health (Nouwen *et al.* 1999).

In a review of incineration and health, Gatrell and Lovett (1989) discuss findings on congenital eye malformations in children born in the vicinity of incinerators. At two chemical waste incinerators in Scotland, UK, (owned by ReChem), there were reports in national newspapers of congenital eye malformations in children born near to the incinerators. However, government studies found no evidence of increased congenital eye malformations in children born in the vicinity of these plants, or another Rechem chemical waste incinerator in Wales. The government studies have, however, been questioned on accuracy because the database of eye malformations that was used has a voluntary rather than obligatory notification system and therefore some genuine cases may be missed out. Further research on this subject by Gatrell and Lovett (1989) investigated whether there was any evidence of clustering of eye malformations around incinerator sites in areas across England and Wales, but found no evidence of a link. Again, this study was limited by the database of registered congenital eye malformations.

3.2.5 Multiple Pregnancy

There are inconsistent results reported in the scientific literature about a possible increase in multiple pregnancies near to incinerators. An initial study (Lloyd *et al.* 1988) investigated twinning rates around two chemical waste incinerators in Scotland, UK, between 1976 and 1983, following anecdotal reports of increased twinning in cattle in the area. For the years 1980 to 1983, the study found the highest values of twinning were apparent in the areas considered to be most vulnerable to incinerator releases. Values for 1980 were statistically significant. Comparatively high values were also observed in areas designated as less vulnerable in the period for 1976-9. Analysis of results indicated that the late 1970s and early 1980s was the period when spatial clustering of excess twinning became evident.

During the late 1970s to the early 80's, the rate of twinning in cattle in the area was also seen to increase dramatically. Lloyd *et al.* (1988) proposed that the increased rate of twinning in cattle and in humans in the area was consistent with the hypothesis that environmental air pollution may have affected obstetric parameters of the local populations of people and animals. However, not all confounding factors could be ruled out and based on the results, the authors commented that it would be premature to attribute a causal link between the pollution from incinerators and twinning.

In addition to increased twinning in cattle, farmers in the area had reported other effects in cattle including increased abnormalities and stillbirths and unexpected deaths. A subsequent study, (Lenihan Inquiry Report) however did not find a link between the incinerator releases and problems in the cattle (cited in Petts 1992, Gatrell and Lovett 1989).

Van Larebeke (2000) noted that data from a Belgian study on incineration and health effects (discussed above, section 4.2.5), showed that there was a statistically significant increased (2.6-fold) probability of having multiple pregnancies in a population living in the neighbourhood of two MSW incinerators. In another study however, data of twin deliveries in Sweden between 1973 and 1990 did not show evidence of clusters of twin births in the vicinity of incinerators (Rydhstroem 1998). The study used a method that could compare the number of twin deliveries both before and after the commissioning of an incinerator.

3.2.6 Hormonal Effects

Thyroid hormones in the blood of children living in industrial/agricultural municipalities close to the Beibesheim incinerator in Germany were compared to those of children living in an industrial/agricultural area without an incinerator and in a second comparison area (Osius & Karmaus, 1998). The incinerator was licenced to burn highly PCB contaminated materials (Osius *et al.* 1999). The initial 1998 study determined thyroid hormones- free thyroxine and free triiodothyronine-in blood samples from 671 children, aged 7-10 years. Blood serum levels of free thyroxine (FT4) and, to a lesser extent, free triiodothyronine (FT4) were statistically significantly lower in children living in the area in which the incinerator was operating. In this group, it was also found that there was a higher prevalence of FT3 values below clinical references. Mean levels of thyrotropin stimulating hormone (TSH), however, were only marginally different. The authors concluded that their results, considered with those of Holdke *et al.* (1998), (see: Section 3.1.1), suggested that children exposed to toxic waste incineration in the studied area had lower free blood t hyroid hormone levels.

In the later 1999 study, the authors attempted to correlate blood contaminant levels with the highly complex thyroid hormone system, which regulates the development of brain function and cell growth. It was found that increased concentrations of the mon-ortho congener PCB 118 in the blood were statistically significantly associated with increased levels of TSH. Elevated levels of PCB congeners 138, 153, 180, 183 and 187 were associated with reduced blood FT3 levels. No associations were found for PCB congeners and FT4 although elevated blood cadmium concentrations were associated with raised TSH levels and diminished FT4 levels. The authors concluded that the study supported the hypothesis that cadmium and PCBs could have a detrimental effect upon levels of thyroid hormones. Given the importance of the thyroid hormone system in the growth and development of children the authors suggested that future studies should be made of impacts of these contaminants upon thyroid hormones in different age groups and to consider neurological development as a component of these studies.

3.3 Risk Assessment

The present regulatory system aims to set quantities or rates at which chemicals can be legally released into the environment. In Europe, limits are generally based on the process of hazard assessment but in recent years the process of risk assessment has been increasingly implemented.

A risk assessment on health effects attempts to estimate exposure to a particular chemical from the pollutant releases in question and finally calculates the probability of health effects from the estimated exposure. Many risk assessments have been reported on health effects expected to arise from exposure to incinerator releases, particularly cancer risk. Nearly all such risk assessments from the 1980s through to the 1990s have concluded that contmainants from incinerators do not pose a significant health risk to populations living within their vicinity. This is in direct contrast to human epidemiological studies, some of which have found evidence of health impacts.

For example, a review of risk assessment data for hazardous waste incinerators in the 1980s by Oppelt (1990) points to a conclusion that stack emissions from burning hazardous waste pose little risk to human health. Data on which many of these assessment were based has however been criticised by the US EPA Science Advisory Board as being insufficient. Dempsey and Oppelt (1993) discuss risk assessments conducted for cement kilns burning hazardous waste. These concluded that no adverse health effects would be expected due to emissions. A US health risk assessment on MSW incinerator air emissions estimated that carcinogenic and non-carcinogenic risks from exposure via inhalation were within acceptable limits (Roffman and

Roffman 1991). Similarly, a study in Germany estimated that cancer risk caused by inhalation of selected heavy metals and dioxins emitted from modern municipal waste incinerators would not endanger health. (Eikman 1994). A study on a MSW incinerator in Mokdong, Seoul, Korea, also reported that cancer risk from inhalation was less than the acceptable risk value of one cancer case in a million (Lee *et al.* 1997).

One of the criticisms of many health risk assessments on incinerators is that they only account for exposure via inhalation and do not consider other routes of potential exposure such as ingestion of soil and vegetation and absorption through the skin (Webster and Connett 1990). This criticism applies to many of the above mentioned studies. Given that ingestion of food is the dominant route of exposure for incinerators sited in or near to agricultural areas this brings all the above risk assessments into substantial question (see e.g. Meneses *et al.* 1999, Webster and Connett 1990).

A recent risk assessment of a MSW incinerator in Montcada, Catalonia, Spain, did take into account all of the known potential exposure routes for dioxins. The risk assessment estimated exposure of local residents to incinerator air emissions of dioxins via inhalation of air and particulate matter, and via soil and vegetable intake from the area and via absorption of soil through the skin (Meneses *et al.* 1999). It compared the intake of dioxins through these routes with intake through a normal diet. Results showed that incinerator air emissions accounted for less than 6% of the total dioxin intake for the population, while diet accounted for greater than 94%. The study concluded that according to the WHO standard for tolerable daily intake (TDI) of dioxins, (i.e. the daily intake of dioxins per person proposed as safe based on current knowledge), intake of dioxins from the incinerator would not imply health risks for the general population of the area. However, the study failed to mention that the incinerator emissions would nevertheless add to levels of dioxins already stored in the tissues of the nearby population as well as those already present in the food supply..

Interestingly, a recent risk assessment was published which did indicate an increased health risk from exposure to dioxins for some child members of a population living near to incinerators (Nouwen *et al.* 1999). The risk assessment was conducted for the population living within the vicinity of the two incinerators in Neerland, Wilrijk, Belgium. Epidemiological studies in this region indicated that an increased probability of congenital malformations at birth and some impacts on respiratory health of children as discussed in section 4.2.5 above (van Larebeke 2000). The risk assessment considered 3 possible exposure scenarios. These were, firstly, a worst case scenario in which individuals lived exclusively from foods (meat, milk and vegetables) produced in the vicinity of the incinerator. Secondly, a scenario which assumed people consumed a mixture of commercial produce and produce from the area (25% crops and 50% meat) was considered. Thirdly, a scenario was proposed in which individuals consumed only commercial produce containing background concentrations of dioxins. This latter scenario was assumed to be the exposure situation for the majority of the residents at Neerland. Exposures were estimated based on dietary exposure, inhalation and dermal exposure. The present tolerable daily intake set by WHO for dioxins is 1-4 pg TEQ/kg bw/day. The risk assessment estimated that children in the first, high exposure scenario, would have exceeded this limit in 1980 by a factor in excess of 4 (16.62 pg TEQ/kg bw/day). Children with an estimated medium exposure in the second scenario, also exceeded the WHO TDI in 1997 by a 2-fold margin (8.17 pg TEQ/kg bw /day). The study considered that this would be the situation for a relatively few families.

4. ENVIRONMENTAL CONTAMINATION

4.1 Deliberate and Fugitive Releases from Incinerators.

Incinerator wastes in the form of stack gas, fly ash, bottom ash/slag, scrubber water, scrubber water filter cake, etc. are deliberately dispersed or otherwise released to the environment, carrying with them the diversity of pollutants formed or redistributed during the incineration process. Some of the incinerator wastes as well as the wastes actually burned are also released unintentionally as fugitive emissions.

One important difference between the two types of releases, deliberate and fugitive emissions, is the extent to which they are subject to regulatory oversight and control. Thus far, stack gas would appear to be the most highly regulated of the deliberate releases from incinerators. Characterisation and oversight of the other deliberately released incinerator wastes is sparse at best.

Fugitive emissions are vapours or particles that escape during waste tipping, waste feeding, incineration, and ash handling. For example, fugitive dusts can be released from bottom-ash pits and fly-ash hoppers as well as during the process of transferring these ashes to transport vehicles and during the transfer from transport vehicles to the final repositories, such as a landfill. These dusts, particularly fly ash dusts from particulate air pollution control devices, are enriched in toxic metals and contain condensed organic matter (NRC 2000).

At hazardous waste incineration facilities, fugitive emissions are, in the case of liquid wastes, released as vapours from liquid waste tank vents, pump seals, and valves. For solid wastes, fugitive emissions escape as dust from solid-materials handling and during the handling and transport of fly ash captured by air pollution control devices. Also, the high-temperature seals on rotary-kiln incinerators are a potential source of vapour and dust emissions generated by such incineration facilities (NRC 2000).

Fugitive emissions can be minimised by designing buildings to be under negative pressure so that air is drawn from the areas where both the incinerator ashes and the wastes to be burned are handled and stored. The National Resource Council noted,

"*Although some facilities have partially closed ash-removal systems, few have completely enclosed ash-handling systems throughout the plant*"

Fugitive emissions that are released near ground level may well have a greater impact on the nearby environment than those released into the air from the incinerator stack. The pattern of dispersal of both fugitive emissions and stack releases depends on a potentially infinite number of variables, for example, type of terrain, presence of nearby structures or trees, wind direction and velocity, weather conditions and relative humidity and the interactions between them.

4.2 Studies on Environmental Contamination

Pollutants that are emitted into the atmosphere from an incinerator stack, as well as fugitive emissions, may be deposited on the ground near to the incinerator and so contaminate the local environment. Some pollutants, including PM10 particulate matter and volatile and semivolatile organic compounds, such as dioxins and PCBs, may also be transported great distances on air currents. For example, Lorber *et al.* (1998) estimated that only around 2% of the dioxin emissions to air are deposited in soil near to an incinerator while the remainder is much more widely dispersed.

Most research on environmental contamination in the vicinity of incinerators has focused on dioxins and heavy metals, ignoring most other pollutants. Studies show that soil and vegetation close to incinerators may become contaminated with incinerator releases of dioxins and heavy metals to levels above normal background concentrations. As a consequence, there is a possibility of agricultural produce, such as crops, becoming contaminated. Livestock may also take in pollutants, largely through ingestion of contaminated vegetation and soil. In some instances this has led to cow's milk being banned from sale due to unacceptably high levels of dioxins, and recommendations to avoid the consumption of eggs and poultry.

This section discusses studies on levels of dioxins and heavy metals found in soil and vegetation in the vicinity of incinerators, both in the past, and more recently. Levels found in cow's milk are also discussed. Considering the potential for contamination of agricultural produce close to incinerators of all types, the research in this area is very limited.

4.2.1 Soil and Vegetation

Research has shown that soil and vegetation can be used as suitable media for monitoring contamination from atmospheric deposition of dioxins and heavy metals (see Schuhmacher *et al.* 1999a, Schuhmacher *et al.* (1997a), Gutenman *et al.* (1992).

Levels of dioxins in soils have been widely used to describe long-term exposure to these chemicals. On the other hand, vegetation is a more representative index of short-term exposure to dioxins (Schuhmacher *et al.* 1999b).

With regard to vegetation, dioxins and heavy metals may simply be deposited on the surface of leaves or be present in soil particles on the plants. In addition, metals may enter the leaves through the small pores on the leaf surfaces (stomata) and be taken up by the roots in woody plants (see Bache *et al.* 1992). Dioxins are, however, apparently not taken up by the root system of plants (Hulster and Marschner 1992).

Dioxins

There are many sources of dioxins in urban areas besides incinerators and consequently, in urban/industrial areas, it is difficult to clarify whether dioxin contamination is coming from an incinerator as opposed to other sources. Nevertheless, studies have shown that high levels of dioxins are present in soils near to some incinerators. In many instances, they have also shown that the level of dioxin found in soil and vegetation is dependent upon the distance from the incinerator, a phenomenon which implicates incinerators as a primary source of the contamination.

For instance, a study which took soil samples from the surroundings of a clinical waste incinerator in Spain, found the highest levels were located at distances nearer to the incinerator compared to further away (Jimenez *et al.* 1996). Levels in soils close to the incinerator were 2.1 to 7.5 times higher than usual background levels of dioxins in soils. In another study, extremely high dioxin levels (e.g. 252 and 211 ppt TEQ) were found in soils on the leeward side of a Japanese MSW incinerator (Ohta *et al.* 1997). These levels are exceedingly high compared to background levels found in soils of industrialised countries (e.g. 3.6 ppt TEQ for rural soils and 11.9 ppt TEQ for urban soils of North America, and similar levels for Europe), (USEPA 2000). This incinerator has generated considerable controversy due to the high number of cancer deaths recorded nearby. The study showed that the high soil levels correlated with the area of high cancer incidence.

In 1993, exceptionally high levels of dioxins and PCBs were reported in soils near to the Shanks hazardous waste incinerator (formerly the Rechem incinerator) in Wales, UK (see ENDS 2000b). The highest level of dioxins was 1740 ng I-TEQ/kg (Foxall and Lovett 1994). The study indicated that releases other than those from the incinerator stack could have been substantially responsible for the high levels, including fugitive emissions during ash disposal operations from waste storage areas and the transformer handling facility (Foxall and Lovett 1994). The plant has since been upgraded and recent data suggest that levels of dioxins in soils have now fallen to around two-thirds of the 1993 values. The decline in PCBs is however less marked. Moreover, average air emissions of PCBs (2 ng/m^3) from the plant are well above levels in urban areas of Britain which rarely exceed 1ng/m^3 and are usually well below 0.5 ng/m^3. Even so, despite being previously challenged by the Environment Agency, the company now has authorisation to import another 200 transformer carcasses, a known source of PCBs, from Italy (ENDS 2000b).

A 1998 study on an old MSW incinerator in Montcada, Barcelona, reported dioxin levels in soils ranging from 0.06 to 127.0 ng I-TEQ/kg (ppt) with a mean concentration of 9.95 ng I-TEQ/kg (ppt), (Granero *et al.* 1999). The study found that levels had increased at all sites that were monitored between 1996 and 1997 and again between 1997 and 1998. However, the increases were not statistically significant. The authors noted that although the conditions of incineration remained constant during 1996 to 1998, it is possible that the potential accumulation of dioxins in soils that may be expected from incineration could be counteracted by a decrease in emissions from other sources in the area.

Some studies have shown that incineration is not always associated with high levels of dioxins in local soils. For instance, a study in Spain showed that levels in the vicinity of an old incinerator in Catalonia in 1997 (range 0.11 to 3.88 mean 1.17 ng I-TEQ/kg (ppt) were not unduly high and indeed were consistent with levels found at MSW incinerator sites in other studies in the US and The Netherlands (Schuhmacher *et al.* 1999a). The levels found in 1997 had, however, increased slightly (8.3%), but not significantly, from levels previously determined in 1996.

Studies on vegetation near a MSW incinerator in Catalonia, Spain in 1996/7 showed that the incinerator contributed to dioxins in vegetation because higher levels were apparent closest to the incinerator with lower levels recorded further away (Domingo *et al.* 1998). However, between 1996 and 1997, unlike levels in soils at the incinerator which remained relatively constant, (see above: Schuhmacher *et al.* 1999a), in some areas there was a reduction in the levels of dioxins in vegetation. Since vegetation largely reflects short-term changes in dioxin exposure, whereas soil reflects longer-term exposure, the study noted that the reduction in dioxin levels in vegetation might be due to better pollution control at the incinerator. It could also, however, be reflective of a reduction of dioxin emissions from other sources in the area in general. Similarly, another study in Spain at an old MSW incinerator in Montcarda, Barcelona, found a decrease in levels of dioxin between 1997 and 1998. The study proposed that the reduction in dioxin levels was probably due to general abatement actions to reduce dioxin air emissions (Schuhmacher *et al.* 1999b).

Heavy Metals

Heavy metals released into the environment from incinerators can contaminate soils and build up

(bioaccumulate) in plants and animals. In this way they eventually make their way to humans via the food chain or through contamination of drinking water. In addition, for people living in the vicinity of incinerators, and especially for children, exposure to heavy metals may also occur due to the consumption of dirt or dust originating from contaminated soil. Other routes of heavy metal intake include inhalation and absorption via the skin (Schuhmacher *et al.* 1997b).

Data on levels of heavy metals in soils near to incinerators are very limited. A study on soils near to an industrial incinerator in Italy found lead pollution in soils was increased by some 600% (Zanini and Bonifacio 1991). A more recent investigation into levels of cadmium and lead in soils surrounding the Baldovie MSW incinerator in Scotland determined that the incinerator was responsible for the long-term distribution of the metals in soils within a 5 km vicinity of the incinerator (Collett *et al.* 1998). It was found that long-term concentrations of cadmium and lead in emissions to air from the incinerator were related to levels found in local soils. Although the levels of cadmium and lead were linked to incinerator air emissions, the study also reported that levels of cadmium and lead were within their normal background ranges in soils. A study on a sewage sludge incinerator near to Birmingham, UK, found evidence of lead and cadmium contamination in surface dusts close to the incinerator (Feng and Barratt 1999).

A recent study in Spain on the old MSW incinerator in Montcada, Barcelona did not find levels of heavy metals that were considered to be high (Schuhmacher *et al.* 1997b). For cadmium and lead, levels were similar to concentrations reported for uncontaminated soils.

There are only a few studies in the scientific literature on levels of heavy metals in vegetation in the vicinity of incinerators. Bache *et al.* (1991) reported contamination of vegetation surrounding a MSW incinerator (without air emission controls) in the US by several heavy metals. Another study on a MSW incinerator in the US, this time fitted with pollution control equipment, found that levels of cadmium and lead in tree foliage were linked to distance from the incinerator (Bache *et al.* 1992). The study concluded that even a MSW incinerator with pollution control equipment (electrostatic precipitator) could result in the significant deposition of metals such as cadmium and lead in surrounding areas. A study on a MSW incinerator in New Jersey reported that levels of mercury in vegetation (moss placed at specific sites around the incinerator) was related to distance from the incinerator (Carpi and Weinstein 1994). The highest concentrations of mercury were located closest to the incinerator.

One study has been documented in the scientific literature, which did not find a link between incineration and heavy metals in surrounding vegetation. The study reported that contamination of vegetation with cadmium and lead was not apparently related to distance from a MSW incinerator (Gutenman *et al.* 1992).

4.2.2 Cow's Milk

Cattle that graze in areas subject to air deposition of pollutants, such as dioxins, can ingest the pollutants that have been deposited on vegetation and soils. Dioxins can subsequently be passed to their milk, and hence, ultimately to humans. This is because elimination via milk is a major route of excretion of dioxins in cow's (Baldassarri *et al.* 1994). Research conducted in several countries during the 1990's has demonstrated elevated levels of dioxins in cow's milk from farms located near to incinerators.

A decade ago, a study in the Netherlands reported high concentrations of dioxins in cow's milk (up to 13.5 pg I-TEQ/g fat [ppt]). This led to a decision by the Dutch government to instigate an upper limit for dioxin levels in milk and milk products of 6 pg I-TEQ/g (fat), (Liem *et al.* 1990). Some other European countries including Germany, Holland and Austria (Ramos *et al.* 1997) later also adopted this limit-value. A study in Austria reported high dioxin levels in cow's milk obtained from farms sited close to incinerators (up to 8.6 pg I-TEQ/g fat), (MAFF 1997a). In the UK, exceptionally high levels of dioxins (up to 1.9 pg TEQ/g whole milk, equivalent to 48 pg TEQ/g fat) were reported in milk from farms located near to a chemical waste incinerator at the Coalite chemicals plant in Derbyshire (MAFF 1992, EA 1997, Sandells *et al.* 1997). The incinerator was subsequently closed down in November 1991.

More recent studies have also found evidence of increased levels of dioxins in cow's milk from farms situated near to incinerators. For instance, a UK study on milk from farms located within the vicinity of potential dioxin sources in the UK, found that farms near to two out of eight MSW incinerator sites investigated had milk with levels of dioxins which exceeded the Dutch 6 pg I-TEQ/g fat limit in 1993-5 (MAFF 1997b). In 1995, milk from a farm sited near a MSW incinerator in Bristol had a level of 6.1 pg TEQ/g fat whilst milk from farms near to an incinerator in West Yorkshire had levels ranging from 3.1 to 11 pg I-TEQ/g fat. A retest in of farms in the latter area in 1996 showed that high levels remained in milk (1.9-8.6 pg I-TEQ/g fat). The incinerator was closed later in the year for failing to meet newly imposed pollution control standards.

A study in Switzerland of cow's milk from farms located in both rural and more industrial areas reported that local influence of incinerators on dioxin levels in milk was clearly

detectable (Schmid and Schlatter 1992). Similarly, a more recent study in Spain found that milk from rural areas had levels of dioxins (1.3-2.47 pg I-TEQ/g fat) that were lower than levels in milk from a farm located in the vicinity of potential dioxin sources. Of the dioxin sources, it was determined that a waste incinerator had the greatest effect such that the highest levels (3.32 pg I-TEQ/g fat) in milk were obtained for a farm situated close by (Ramos *et al.* 1997).

5. INCINERATOR RELEASES

All waste incinerators are also waste generators - incineration of waste results in output of waste products. This is because physical matter cannot actually be destroyed, but can only be transformed into new forms. Thus when things are burned, they do not disappear as is the common perception, but merely change their form. Waste products resulting from incineration take the form of stack gas emissions to the atmosphere, bottom ashes (slag) and fly ashes (caught in filters in the incinerator stack) which ultimately are disposed of to landfill sites. Where water is used for cleaning processes in an incinerator, there are also releases of waste products to water.

It is a popular misconception that the weight and volume of the original raw waste are reduced during incineration. Although it is often stated that the solid residues (ashes) remaining equate to about one third of the initial weight of the raw waste (Pluss and Ferrell 1991), and volume reduction of about 90% is achieved (Williams 1990) neither of these statistics stand up to scrutiny. If all the waste outputs from an incinerator are summed, then the output will exceed the original waste input. The gases present in the flue stack result from the combination of carbon-based materials with oxygen and are usually ignored in calculating the mass of residues, but the combination with oxygen to form CO2 increases actual weight. Residues from wet gas cleaning systems can generate appreciable volumes of contaminated water and solids. In the case of the statistic concerning volume reduction, this is usually generated by reference to the volume of uncompacted wastes. Landfilled MSW, however, is generally compacted to increase stability and prevent water infiltration as well as reduce the volume of the wastes. Comparing unburned waste and incinerator ash, the actual volume reduction achievable is closer to 45% (DoE 1995).

Numerous chemicals are released into the wastes generated by incineration, including hazardous chemicals. For instance, MSW incinerators are typically fed a mixed waste stream and the combustion of such waste leads to hazardous substances originally present within the waste being mobilised into releases from the incineration plant. While some chemicals remain in their original form, others are changed into new chemical species. For example, heavy metals are not destroyed by incineration but are simply concentrated in the remaining wastes. They can remain in their original form during incineration or may react to form new compounds such as metal oxides, chlorides or fluorides (Dempsey and Oppelt 1993).

The exact nature of the substances released during incineration depends on the composition of the waste that is incinerated. For instance, incineration of chlorinated organic compounds will cause the formation of hydrogen chloride (HCl) and this in turn can contribute to the formation of dioxins. Technical standards that are applied both to the incineration process and to pollution control equipment will also influence the final products of incineration (EEA 2000). However, whatever control technology is applied, all types of incineration result in releases of toxic substances in ashes and in the form of gases/particulate matter to air. These substances include heavy metals, numerous organic compounds, such as dioxins, and gases, such as nitrogen oxides, sulphur oxides, hydrogen chloride, hydrogen fluoride, together with carbon dioxide. According to the NRC (2000):

" … *the products of primary concern, owing to their potential effects on human health and the environment, are compounds that contain sulfur, nitrogen, halogens (such as chlorine), and toxic metals. Specific compounds of concern include CO, NO_x, SO_x, HCl, cadmium, lead, mercury, chromium, arsenic, beryllium, dioxins and furans, PCBs, and polycyclic aromatic hydrocarbons.* …"

In many countries over the past few years, new regulatory air emission standards have forced the closure or updating of many old incinerators or the building of new ones. Upgraded plants (together with new ones) may be fitted with modern, improved air pollution control technology. For example, out of the 780 incinerators in operation in the UK in the early 1990s (30 for municipal waste, 700 for clinical waste, 40 attached to chemical companies, 6 for sewage sludge and 4 for hazardous waste), only 110 remained after the tightening of standards (Murray 1999). Presently there are 12 MSW operating incinerators in the UK. The closure or updating of old incinerators is considered to have led to a substantial reduction of emissions of toxic substances to air.

One study in the Netherlands has also estimated that dioxin emissions to air have been significantly reduced (Born 1996). Murray (1999) states that the most sophisticated German technology developed during the early 1990's has cut atmospheric emissions broadly by a factor of ten. Although this is a significant improvement, the problem of toxic waste products from incineration has not disappeared. In fact, the problem has shifted so that more of the dioxins and other toxic substances generated now appear in the ashes, thereby creating new disposal and pollution problems. The European Environment Agency (EEA 2000) has warned that even if total air emissions from incineration are reduced in the future as standards improve "this might be offset with increased incineration capacity". In this regard, it is of great concern that an increase in the use of incineration is being proposed in some European countries. In the UK for instance, following the closure of numerous old incinerators, up to 177 possible new ones have been proposed by the government (ENDS 1999).

On the regulatory front, among the various incinerator outputs, stack gas has received the greatest share of attention and is the most highly regulated since the gas and its toxic components are dispersed directly into the open air. However, the other incinerator wastes also contain toxic pollutants and, consequently, pose threats to public health that may be less obvious and/or immediate but are no less real.

The Commission of the European Communities (CEC) has been drafting a proposal for a new waste incineration directive since 1998 (EC 1998, EC 1999). The proposed new directive will establish controls on the incineration of most wastes that are not covered by the previous 1994 directive. The new directive will set limits for releases of some hazardous substances in stack gases and water. The directive is expected to be adopted by the end of 2000 or early 2001. All new MSW incinerators built after the directive comes into force must satisfy the limits in the directive within 2 years, whilst existing incinerators have a period of 5 years to satisfy the criteria. In addition to the EU regulations, various national guidelines for incinerators are also presently in place and these will have to comply with the directive within two years of it coming into force (EC 1999).

5.1 Releases to Air

This section presents data on substances known to emitted in stack gases from incinerators. Most research on air emissions has focused on dioxins and upon the behaviour of a few toxic heavy metals. Data from research upon other emitted chemicals are sparse. In addition, a very large number of the chemicals emitted from incinerators remain unidentified.

Emissions from incinerator stacks to air are discussed below under the following categories: organic compounds; heavy metals; gases and particulates. The EC have proposed limits for air emissions from incinerators in their new directive for only a few of the compounds falling under these categories. These proposed limits are given in the table below.

Table 5.1 EC Air Emission Limit Values

Substance	**Proposed EC Limit (mg/Nm^3)**
Dioxins	0.1 ng TEQ/Nm^3
Mercury	0.05 b
Cadmium + Thallium	Total 0.05b
Sb, As, Pb, Cr, Co, Cu, Mn, Ni, V	Total 0.5b
Carbon Monoxide	50c
SO2	50c
Nox	200c
HCl	10c
HF	10c
Particles	10

[a] Average values measured over a sample period of a minimum of 6 hours and a maximum of 8 hours.
[b] All average values over the sample period of a minimum of 30 minutes and a maximum of 8 hours.
[c] Daily average value

5.1.1 Organic Compounds

Dioxins

Polychlorinated dibenzo-*p*-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) are a group of chemicals often referred to simply as dioxins. There are more than 200 individual congeners (members) of the PCDD/Fs group. The most widely known and most toxic congener is 2,3,7,8-TCDD. It has been described as the most toxic chemical known to mankind and is a recognised as a human carcinogen. Dioxins are persistent in the environment, toxic and bioaccumulative (build up in the tissues of living organisms). A more detailed description of the toxic impacts of dioxins on health is given in appendix A.

The toxicity of individual dioxins and furans varies by several orders of magnitude. Because analytical data may report 17 different congeners as well as totals for homologue groups (i.e. all congeners containing the same number of chlorine atoms), it is often necessary to summarise data so that individual samples can be directly compared. This is generally done by expressing the amount of dioxins present as toxic equivalents (TEQs) relative to 2,3,7,8-TCDD. The most common TEQ system used is the international toxic equivalents system (I-TEQ). The TEQ system works by assigning TCDD, the most toxic congener, a toxic equivalence factor (TEF) value of 1. The toxicity of all other congeners is expressed relative to this, such that they are assigned a TEF value between 0 and 1. The I-TEQ of a sample containing a mixture of dioxins is obtained by multiplying the concentration of each congener by its TEF and summing the results.

One important consideration in relation to dioxin air emissions from incinerators is that regulations consider only the chlorinated varieties. It has been known for some time that incinerators generate and emit brominated and mixed chloro-bromo substituted dioxins in appreciable quantities (see: Schwind *et al.* 1988). These are regarded as of an equal toxicological significance relative to the chlorinated dioxins, producing a similar array of biological impacts at similar molar concentrations (Weber and Greim 1997). Despite these compounds being highly persistent when associated with fly ash particles, little attention has been directed at evaluation of their significance to human health and there are currently no obligations on the part of incinerator operators to monitor and control these chemicals.

Formation of Dioxins in Incinerators

Dioxins are produced as unintentional by-products of many manufacturing and combustion processes, especially processes that use, produce or dispose of chlorine or chlorine derived chemicals. All types of incinerators produce them. Research has shown that while dioxins can be destroyed in the combustion zone of incinerators, they can be regenerated in the post-combustion zone by processes that are dependent on the temperature profile (Blumenstock *et al.* 2000, Huang and Buekens 1995, Fangmark *et al.* 1994). The predominant formation pathway of dioxins has been reported to be de novo synthesis (Johnke and Stelzner 1992), and they are also formed from precursors that are either constituents of the waste or are also formed by chemical recombination of materials in the waste. The chlorobenzenes and chlorophenols are two such groups (Huang and Buekens 1995). PVC, a common constituent in municipal waste, has also been identified as a dioxin precursor (USEPA 1997).

Prior to incineration, raw waste is itself known to contain dioxins. However, it has been demonstrated that the process of waste incineration creates dioxins. For instance, past and present, calculations (mass balances) both show that the total amount of dioxins coming out of an incinerator in the various waste products is greater than the amount going into the incinerator as raw waste (Williams 1990; Hansen 2000). This appears to still be the case for modern and updated incinerators operating in the late 1990s, although very little in the way of scientific data is available from the scientific literature with the exception of a recent Danish study (Hansen 2000).

In another example from Spain, a mass balance estimate based on measurements from eight operating municipal waste incinerators showed that more dioxins are emitted from the incinerators than were present in the raw waste (Fabrellas *et al.* 1999). Estimates showed the level of dioxins (PCDD/Fs) input in raw waste to the incinerator amounted to 79.8 g I-TEQ/year. This compared to the total estimated output of flue gases (1-1.2 g I-TEQ/year), fly ashes (46.6-111.6 g I-TEQ/year), and bottom ashes (2-19 g I-TEQ/year). An alternative dioxin mass balance conducted on another Spanish municipal waste incinerator was ambiguous. One test showed a greater dioxin output than input whereas another test showed a greater dioxin input than output (Abad *et al.* 2000). This is not particularly surprising because emissions of dioxins and other substances from individual incinerators are highly variable depending on waste input and combustion conditions. In addition, the precision of such estimates is often not high, encompassing a wide range of values.

Dioxin Inventories and Incineration

During the 1980s and early to mid-1990s MSW incineration, in particular, was identified as a major source of dioxins emitted to atmosphere. For example, the Dutch government organisation RIVM estimated that incineration was responsible for about 79% of all dioxins emitted to air in the Netherlands for the year 1991. In the UK, MSW incinerators were estimated as responsible for about 53-82% of all dioxins emitted to air in 1995. In the US such facilities accounted for about 37% of total annual air emissions (see Pastorelli *et al.* 1999). A summary of data from 15 countries, described as a "global" inventory, showed that incineration accounted for about 50% of dioxin emissions to air in 1995 (Fiedler 1999). MSW incineration has been identified as being responsible for the greatest proportion of dioxin air emissions compared to other types of incineration (eg. Alcock *et al.* 1998), although from "global" inventory data for 15 countries, Fiedler (1999) noted that all sectors of incineration in 1995 were major emitters in many countries. This included MSW incinerators, hazardous waste incinerators, sewage sludge incinerators, waste wood incineration and crematoria. Table 1.2 shows the estimated dioxin air emissions for different types of incinerators for 1997 in the UK.

Even recently, incinerators have been estimated to account for a high proportion of atmospheric dioxins. For example, (Hansen 2000), has conducted a flow analysis for dioxins in Denmark for 1998-1999. Notwithstanding improvements in technology, municipal solid waste incineration was identified as the single largest source of dioxin releases to atmosphere, estimated at between 11-42g I-TEQ per year. It is estimated that a further 35-275g I-TEQ of dioxins contained in incinerator residues is disposed of to landfill each year. This report also draws attention to the potential importance of the brominated and mixed halogenated dioxins (Section 5.1.1) and estimates that between 2 and 60g of brominated dioxins are emitted to atmosphere from Danish MSW incinerators per year.

Table 1.2. PCDD/F air emission estimates for the UK, (numbers in bold type represent estimates calculated from measured air emissions, other number are estimates)

PROCESS	1997 Range/Low (g TEQ/annum)	1997 Range/High (g TEQ/annum)
MSW incineration	122	199
Chemical waste incineration (10 sites)	0.02	8.7
Medical Waste Incineration (5 sites)	0.99	18.3
Sewage Sludge Incineration (5 sites)	0.001	0.37
Cement Manufacture (5 sites)	0.29	10.4
Crematoria	1	35
Domestic Wood Combustion (clean)	2	18
Domestic Wood Combustion (treated)	1	5

Source: Alcock *et al.* 1998.
Footnote: The estimate for total dioxin air emissions from all sources is Range/Low 219 and Range/High 663 g TEQ/annum.

A 1997 publication cited by the commission of the European Communities (EC 1998) noted that incineration of non-hazardous waste may contribute up to 40% of all dioxin air emissions in Europe. Nevertheless, in some European countries, it has been estimated that the contribution of MSW incineration to national inventories has fallen significantly during the mid- to late 1990s. This is due to closure of old incinerators, which emitted high levels of dioxins to air and the fitting of pollution abatement equipment to both remaining plant and new installations. Estimates suggest that such improvements will have resulted in the significant reduction of dioxin emissions from incinerators to air. For instance, strong downward trends of air emissions have been identified in countries with modern technology or rigid legislation (Fiedler 1999). Considering atmospheric emissions alone, in the UK, Her Majesty's Inspectorate of Pollution (HMIP) and the Department of the Environment (DoE) estimated that the contribution to the total annual emission would fall from 53-82% in 1995, to around 4-14% in the future. Similarly, the German UBA estimated a contribution of 33% for the years 1989-1990 falling to 3% for the years 1999-2000. These estimated data remain to be confirmed with empirically derived data.

The need for confirmation is important. It has been acknowledged, for instance, in the above UK HMIP study, that there are large uncertainties in estimations of incinerator air emissions used in dioxin inventories. In the case of the UK study, this is because air emissions have generally been estimated from only very limited measurements and have also used information derived from non-UK studies. A recent UK study which corrected for these sources of uncertainty to some extent (Alcock *et al.* 1998), used a different, more precise, estimation method that included measured emission data from individual incinerators between 1995 and 1997 (see table 1.2). It currently represents the most comprehensive survey of measured UK dioxin air emission data. Importantly, the study also used data from waste incinerators, which were operating under normal everyday conditions during the periods of testing. This is more realistic than measurements being taken under "optimum" conditions that are specifically set up for testing under a "test burn" regimen which is, more often than not, the case. The study found that relative to emission data published by the HMIP for 1995, the levels of dioxin emissions from MSW incinerators between 1995-7 had fallen somewhat. Even so, they still represented a significant part of the national inventory, representing 30 to 56% of the total national air dioxin emissions. Clearly the optimistic projections of the regulatory authorities need confirmation before they can be accepted as a realistic projection of trends or as a metric of the current situation.

On the same note, Webster and Connett (1998) draw attention to uncertainties and problems in the methodology commonly employed to derive national dioxin air emission inventory data. These include several points listed below and include two points specifically mentioned in the UK study above: firstly that few empirically measured data from individual incinerators are normally used in the estimate (see first bullet point), and that data on air emissions used are most often derived from testing of incinerators under "optimum" conditions rather than normal day to day operations.

- **Methodology:** The method normally applied for estimating dioxin inventories, "the emission factor approach" relies on a limited number of specific measurements from particular types of incinerators and extrapolates these to represent all incinerators of a particular type. This is likely to underestimate emissions to all media. It does not take account of the fact that there can be enormous variability in emissions from individual incinerators of the same type. In their study,

Webster and Connett (1998) showed that the "emission factor approach" did indeed underestimate dioxin air emissions from incinerators reported in many previous US inventories over the past decade. Instead of applying the emission factor approach, Webster and Connett (1998) summed dioxin air emissions for measured facilities only – an approach that would presumably underestimate these emissions since unmonitored incinerators were not included in the calculations. Even so, this method still produced a significantly greater value for MSW incinerator dioxin emissions to air than using the emission factor approach. The authors therefore stressed the need for adopting the use of actual measurements from individual facilities for inventories.

- **Lack of Data:** On a global basis, Fiedler (1999) reported that the present number of national dioxin emission inventories is very small. Within countries that have recorded dioxin inventories, there is a general lack of comprehensive data on dioxin air emissions from incinerators. For instance, Webster and Connett (1998) identified a paucity of data in the US with specific regard to emissions from incinerators. Many US MSW incinerators had either been tested only once or had never been tested at all. Although this situation appears to be improving, operators and regulators in the past seemed quite happy to deem a plant's emissions to atmosphere acceptable based on one set of measurements derived from a pre-commissioning test-burn. Even now, the frequency and intensity of stack sampling and analysis for dioxins carried out at most incinerators is unacceptably low.

- **Monitoring:** Research has shown that taking only a limited number of measurements is not likely to accurately reflect dioxin emissions to atmosphere from incinerators over the full spectrum of operational conditions. That dioxin emissions from combustion sources may change considerably over time is well illustrated by an UK study (Alcock *et al.* 1998). The study showed that air emissions indexed by samples collected from a cement kiln stack on the same day were found to vary considerably. The first sample collected measured 4.2ng I-TEQ m^{-3} and the second sample taken 5 hours later was determined as 0.06ng I-TEQ m^{-3}.

A more accurate estimate of atmospheric dioxin emissions can only be established by continuous monitoring of emissions for extended periods of time. Start-up and shut down periods in the operation of MSW incinerators are particularly prone to result in high dioxin emissions. A study on a Belgian incinerator, using continuous monitoring, was undertaken in an attempt to demonstrate that retro-fitted modern pollution control equipment would prevent excedence of the 0.1 TEQ/Nm^3 regulatory limit at all times. In fact the results revealed that monitoring over a period of 6 hours gave an average emission concentration of 0.25 ng TEQ/Nm^3. However, the average over 2 weeks in the same period gave a result of 8.2 to 12.9 ng TEQ/Nm^3 which was substantially greater and clearly exceeded the regulatory limit (De Fre and Wevers 1998).

The above study shows, in a convincing manner, that taking measurements from individual incinerators under the normal regulatory protocols (i.e. point measurements), can significantly under-estimate the dioxin emissions to air from incinerators. In this case, point measurement under-estimated the average dioxin emissions by a factor of 30 to 50. The significance of this finding to other incineration facilities is simply not known.

- **Dioxins in ash are not considered:** Most mass balance inventories consider only dioxin emissions to atmosphere (Fiedler 1999); The output of dioxins in ash from incinerators is not included and Webster and Connett (1998) consider that the fate of dioxin captured in ash receives insufficient attention. A recent study on a Spanish incinerator showed that stack gas emissions were only responsible for a minor contribution to the total dioxin emitted compared to amounts present in fly ash (Abad *et al.* 2000). The fact that dioxins formed in incineration have become more concentrated in ashes as air pollution control technologies have evolved, thereby generating other hazards, is further discussed in section 5.3.1.

Considered together, the generally flawed sampling methodology employed in regulating releases from incinerators coupled with failure to consider the dioxin mass balance in an holistic manner suggests that it is highly probable that most, if not all, dioxin inventories greatly underestimate releases from incinerators.

Performance of Updated and New Incinerators

As indicated above, most monitoring of atmospheric dioxin releases carried out at incinerators in Europe and reported in the scientific literature has been derived on the basis of point measurements rather than continuous monitoring. This can lead to an underestimate of air emissions. This situation seems set to continue under the proposed EC legislation which specifies compliance monitoring based only two point measurements per year taken over a period of six to eight hours (EC 1999). This basis for regulation and control, as opposed to continual monitoring is unlikely to accurately describe dioxin emissions to air from these facilities.

In many cases, studies carried out on the basis of point measurements have reported that dioxin emissions to air from some European incinerators fall within the new proposed EC limit of 0.1 ng I-TEQ/m^3. For instance, a series of monthly to two-monthly point measurements taken between 1994 and 1997 from a newly constructed German MSW incinerator were below the specified limit (Gass *et al.* 1998). Two point measurements, taken within a day of each other, subsequent to initial testing of a newly constructed MSW incinerator in Venice were below the 0.1 ng I-TEQ/m^3 limit (Pietro and Giuliana 1999). A study on a German hazardous waste incinerator was performed which, in fact, used continuous long-term monitoring. Results of 11 long-term monitored samples taken between 1998-9 showed that air emissions were well within the 0.1 ng I-TEQ/m^3 limit (Mayer *et al.* 1999).

Not all studies, however, have returned data indicating compliance with the 0.1 ng I-TEQ/m^3 regulatory limit. For example, point measurements taken at 1 to 4 monthly intervals, January 1997 to April 1999, from 8 Spanish MSW incinerators revealed that 2 incinerators failed to comply (Fabrellas *et al.* 1999). Emission values were 0.7 and 1.08 ng I-TEQ/m^3. In Poland, analysis of stack emissions from 18 new or updated medical waste incinerators in 1994-7 found that almost half had emissions below 0.1 ng TEQ/m^3, but others exceeded the limit (Grochowalski 1998). For 5 of the incinerators, the limit was considerably exceeded with concentrations ranging from 9.7 to 32 ng TEQ/m3. As discussed previously, a Belgian incinerator exceeded the EC regulatory limit when emissions were measured by continuous monitoring (De Fre and Wevers 1998). The emissions were 8.2 to 12.9 ng TEQ/Nm^3.

It is important to note that the scientific literature reporting air emission levels from new and old incinerators presently operating in many countries, including developing countries is extremely limited. One study of dioxin emissions to atmosphere from the ten incinerators reported to be operating in Korea (Shin *et al.* 1998) noted a wide variation between different incinerators. Emitted levels ranged from 0.07 to 27.9 Ng TEQ/Nm^3 of dioxin in the stack gases.

Even fewer data have been published on incinerators burning wastes other than MSW. Nonetheless, in Japan, one study reported point measurements on nine industrial waste incinerators (Yamamura *et al.* 1999). Dioxin emissions to air were below 0.1 Ng I-TEQ/Nm^3 for two of the incinerators but were above this level (0.13 to 4.2 Ng I-TEQ/Nm^3) for six of them. Cement kilns in the US that were operated using coal as fuel were found to emit 0.00133 to 3.0 ng I-TEQ/dscm (Schrieber and Evers 1994). In the US, a further study reported on dioxin air emissions from mobile incinerators. (Meeter *et al.* 1997). The on-site remediation of soils at hazardous waste sites by such incinerators is carried out where sites contain compounds that are regarded as difficult to destroy. Data collected primarily from trial burns of 16 incinerators showed that 10 of the incinerators failed to meet the proposed US Environmental Protection Agency (EPA) standard of 0.2 ng TEQ /dscm. The authors commented that a significant number of mobile incinerators used in these applications could have problems meeting the proposed future EPA limit.

5.1.2 Other Organic Compounds

With a very few exceptions, very little research has been carried out on the other organic chemicals known to be emitted to air from incinerators. Of the compounds, which have been studied, the focus has largely been directed at higher molecular weight compounds rather than the less persistent volatile organic compounds that are known to be emitted (Leach *et al.* 1999). Compounds for which data have been reported include polyaromatic hydrocarbons (PAHs) and several groups of highly toxic chlorinated compounds including the polychlorinated biphenyls (PCBs), the polychlorinated napthalenes (PCNs), the chlorobenzenes and the chlorophenols.

PCBs: This group consists of 209 different individual congeners. Around half this number has been identified in the environment. PCBs are persistent, toxic and bioaccumulative. Accordingly, like the dioxins they have a tendency to build up in the fatty tissues of animals and humans where they can persist almost indefinitely. The more highly chlorinated PCB congeners are the most persistent and account for the majority of those found as environmental pollutants. PCBs have become globally ubiquitous chemicals, and they are even found at highly elevated concentrations in the tissues of animals living in what have traditionally been regarded as pristine environments. Arctic marine mammals, such as whales, seals and polar bears have been studied and the presence of PCBs together with other organochlorine contaminants confirmed (see Allsopp *et al.* 1999). PCBs are known to exert a wide range of toxic effects on health including reproductive, neurological and immunological effects. They are suspected of causing many deleterious health effects both in wildlife and in humans (see Allsopp et al. 1997, Allsopp *et al.* 1999). Some PCB congeners also cause "dioxin-like" effects on health since they are structurally similar chemicals.

PCBs produced as industrial chemicals were mainly used for insulation in electrical equipment. Production of PCBs has almost totally ceased worldwide, although there are reports of it continuing in Russia. At least one third of the PCBs that have been produced are estimated to have entered the environment (Swedish EPA 1999). The other

two thirds remain in old electrical equipment and in waste dumps from where they continue to leach into the environment. Although this is by far the major source of PCB pollution in the environment today, some PCBs are also produced as by-products of incineration and certain chemical processes involving chlorine.

PCBs are known to be formed in incinerators (Blumenstock *et al.* 2000, Wikstrom *et al.* 1998, Sakai *et al.* 1996, Fangmark *et al.* 1994) and are present in stack gases released to the atmosphere (Miyata *et al.* 1994, Wilken *et al.* 1993, Magagni *et al.* 1991). Data on levels of PCBs in stack gases are, however, somewhat sparse in the scientific literature. A study on MSW incinerators in Japan in 1992 found that emissions of the highly toxicologically significant coplanar PCBs varied considerably between different incinerators (Miyata *et al.* 1994). The mean level (1.46 ng TEQ/m^3) was greater than the guideline (0.5 ngTEQ/Nm^3) for newly constructed incinerators in Japan. The study concluded that waste incinerators were a source of PCB contamination in humans, food and environment.

PCNs: are a group of chlorinated compounds that are also persistent, bioaccumulative and toxic. When originally produced they were used in similar applications to PCBs which eventually superseded them. PCNs are known to be produced as unintentional by-products of thermal processes involving chlorine including incineration and metal reclamation (see: Falandysz and Rappe 1997). PCNs have similar properties to dioxins and PCBs and many of them have high toxic potential even at small doses (see: Abad *et al.* 1999, Abad *et al.* 1997).

PCNs have been found to be present in the stack gas of MSW incinerators. The concentration of PCN (mono-to octa-chlorinated) varied from 1.08 to 21.36 ng/Nm^3 in five MSW incinerators in Spain, while levels of dioxins varied from 0.01 to 5 ng ITEQ/Nm^3 (Abad *et al.* 1999). In addition, PCN congeners exhibiting dioxin-like toxicity have been identified in the atmospheric emissions from municipal waste incinerators (Falandysz & Rappe 1997, Takasuga *et al.* 1994).

PCNs from incineration and other combustion sources are present at detectable levels in wildlife and these processes may contribute a significant loading of these highly toxic and persistent chemicals to the environment (Falandysz and Rappe 1997, Falandysz *et al.* 1996) in addition to the environmental burden resulting from historical manufacture.

Chlorinated Benzenes: Chlorinated benzenes are formed in incinerators (Blumenstock *et al.* 2000, Wikstrom *et al.* 1998, Fangmark *et al.* 1994) as are the chlorinated phenols (Wikstrom *et al.* 1999). It has been shown that these chemicals are released in stack gases (Wilken *et al.* 1993).

The production of hexachlorobenzene (HCB), the fully substituted form of benzene is of particular significance. HCB is persistent, toxic and bioaccumulative. It is toxic to aquatic life, land plants, land animals and humans and has been used extensively as a pesticide and seed dressing. Recent research indicates that HCB can contribute significantly to the dioxin-like toxicity caused by organochlorine chemicals in human milk (van Birgelen 1998). It is listed by the IARC as a Group 2B carcinogen, i.e. it is possibly carcinogenic to humans and also appears to be a tumour promoter. HCB may damage the developing foetus, liver, immune system, thyroid, kidneys and CNS. The liver and nervous system are the most sensitive organs to its effects (ATSDR 1997, Newhook & Meek 1994).

Halogenated Phenols: including 14 chlorinated, 3 brominated and 31 mixed bromo-chloro phenols have been identified in in MSW incinerator flue gas (Heeb *et al.* 1995). These chemicals are of considerable importance since dioxins can be formed by condensation reactions of two halogenated phenol molecules. The concentrations of mixed brominated and chlorinated phenols found in the raw combustion off-gas (4nmol/Nm^3; 1.2ug/Nm^3) and stack gas (1 nmol/Nm^3; 0.5ug/Nm^3) exceeded typical raw gas concentrations of the dioxins (0.2nmol/Nm^3; 0.1ugNm^3) in MSW incineration plant.

Brominated and Mixed Halogenated Dioxins: In addition to chlorinated dioxins and furans numerous other halogenated compounds will be formed during incineration including brominated and mixed chlorinated-brominated dioxins and furans.

Polychlorinated dibenzothiophenes (PCDBTs): are sulphur containing compounds that are structurally very similar to dibenzofurans. The sulphur substitutes for the oxygen atom found in the furan moiety of dibenzofuran structure. Little is known about their toxicology, but due to their structure they are suspected to be toxic. PCDBTs have been detected in the stack gas of waste incinerators (Sinkkonen *et al.* 1991).

PAHs: are a group of compounds which are produced as by-products of incomplete combustion of organic substances. Some are persistent, toxic and bioaccumulative. Others are carcinogenic. PAHs are emitted by incinerators in stack gases (Yasuda and Takahashi 1998, Magagni *et al.* 1991). Waste composition, temperature and excess air during the incineration process determine the quantity of PAHs emitted by a given facility. High emissions to air of PAHs have been shown to occur during start-up of incinerators (see Yasuda and Takahashi *et al.* 1998). Measurements of total PAH incinerator emissions to atmosphere reported in one study were 0.02 to 12 mg/Nm^3 (see: Marty 1993).

VOCs: Few studies have been conducted on the vast array of other chemicals emitted from waste incinerators. However, one study has been undertaken specifically to identify and quantify volatile organic compounds (VOCs) in the stack gas of a MSW incinerator (Jay and Stieglitz 1995). This study identified a total of around 250 different VOC compounds for which concentrations ranged from 0.05 to 100 mg/m^3. The compounds are listed in appendix B. The list includes highly toxic and carcinogenic compounds such as benzene and the substituted phenols, together with other known toxic compounds such as phthalates. Data on the environmental and toxicological significance of many of the VOCs emitted are very limited, but VOCs are known to contribute to ozone formation in the lower atmosphere (see below).

Organic compounds emitted by incinerators are generally monitored on the basis of a group parameter which sums the total amount present in a sample of the flue gas: Total organic Carbon (TOC). In the study reported by Jay and Stieglitz (1995), the 250 compounds identified were found to account for about 42% of the TOC. The remaining 58% were shown to consist of aliphatic hydrocarbons of unknown identity.

Leach *et al.* (1999) have noted that processes which generate large quantities of VOCs are of environmental significance since, mixed with nitrogen oxides and exposed to sunlight, they aid in the formation of photochemical oxidants (ozone and peroxyacyl nitrates), with deleterious impacts upon ambient air quality. The proposed new EC limit for total VOC (expressed as carbon) is 20 mg/ Nm^3.

5.1.3 Heavy Metals

Heavy metals are emitted from all types of incinerators. Many heavy metals are known to be toxic at low concentrations and some are persistent and bioaccumulative. Further information on the toxicity of some heavy metals is given in appendix A. Heavy metals enter the incinerator as components of various materials in the raw waste. The process of incineration leads to their being concentrated by a factor of up to 10 in the waste residues (ashes) as the volume of waste is reduced through combustion (Buchholz and Landsberger 1995). A proportion of these toxic trace metals is emitted in the stack gases of incinerators to atmosphere. The major proportion is generally present in fly ash and bottom ash with the exception of mercury where the greater proportion is vented via the flue stack.

Each metal has its own major source in the raw waste. Mercury is present due to the disposal of batteries, fluorescent light bulbs and paints (Carpi 1997). Cadmium is present in paints, PVC plastics and the pigments used to colour plastics. Lead is present in batteries, plastics and pigments (Valerio *et al.* 1995, Korzan and Heck 1990), and antimony is present in flame-retardants (van Velzen and Langenkamp (1996) used in plastic items.

On a global scale, incineration contributes significantly to atmospheric emissions of many heavy metals, as shown in table 5.1 (EEA 2000). Within the EU, figures for 1990 estimated incineration to be responsible for 8% (16t/yr) of all cadmium emissions and 16% (36t/yr) of mercury emissions. Emissions of chromium amounted to 46 tonnes and over 300 tonnes of lead in addition (EC 1998). A variety of flue gas treatment systems have been devised in order to reduce stack emissions of heavy metals (EEA 2000). Stack gas data for hazardous waste incinerators indicate that the fabric filter removal efficiencies (with the metals retained in the ash arisings) are in the order of 95% for most metals except mercury.

The EEA (2000) note that control of mercury releases constitutes a special problem in incineration. Almost 100% of the elemental mercury present in waste is emitted via the stack gases because it does not adsorb to filter dusts or ashes. Elemental mercury comprises about 20-50% of the total mercury emitted. The remainder is in the form of divalent mercury which may be predominantly mercury chloride (HgCl2). After emission to the atmosphere, divalent mercury, which is water soluble, may be deposited close to the incinerator. On the other hand, elemental mercury may be transported for long distances by atmospheric currents before it is eventually converted to the divalent form. This can then become deposited on the ground (Carpi 1997).

Despite the acknowledged significance of the fate of toxic heavy metals present in the waste-streams, published data on the concentrations of heavy metals in stack emissions appears to be very limited. Nonetheless, according to an emissions inventory in the Netherlands, stack emissions of cadmium and mercury were reduced considerably between 1990 and 1995 from MSW incinerators as a result of modernisation (Born 1996). During this period, the contribution to the total Dutch air emissions of cadmium reduced from 44 to 13% and mercury from 53 to 11%. The reduction of atmospheric emissions (assuming the data are reliable) means that metals retained in the facility by pollution control devices will be retained in fly ash residues.

Table 5.3. Worldwide Atmospheric Emissions of Trace Metals from Waste Incineration

Metal	Emissions (1000 tonnes/year)	Emissions (as a % of total emissions)
Antimony	0.67	19.0
Arsenic	0.31	3.0
Cadmium	0.75	9.0
Chromium	0.84	2.0
Copper	1.58	4.0
Lead	2.37	20.7
Manganese	8.26	21.0
Mercury	1.16	32.0
Nickel	0.35	0.6
Selenium	0.11	11.0
Tin	0.81	15.0
Vanadium	1.15	1.0
Zinc	5.90	4.0

5.1.4 Particulate Matter

Minute particles of matter suspended in the air, often called particulates, are present as a result of both natural and human activities. Those of natural origin are derived from wind blown soil particles, sea salt, dusts from volcanic eruptions, spores from fungi and pollen grains from plants. Those from human activities are the result of combustion processes, such as coal-burning, incineration and vehicle exhaust. As a broad generalisation, natural particulates are generally larger in size (>2.5μm) than the finer particulates formed from combustion processes (<2.5 μm), (QUARG 1996, COMEAP 1995, EPAQS 1995). It is these finer particulates, known as "respirable particles" which are of great concern in relation to human health. Particulate pollution is implicated in the worsening of respiratory illnesses such as asthma, and increasing premature mortality from respiratory and heart diseases. This is because the respirable particulates are small enough to be inhaled into the extremities of the lung airways, whereas larger particles are prevented from reaching the deep airways by the respiratory system's protective mechanisms. In particular, those particulates sized <0.1μm, termed ultrafine particles, are of greatest concern in regard to adverse effects on human health. A more detailed description of particulates and their health impacts is given in appendix A.

Incineration gives rise to atmospheric emissions of particulates (EC 1998). Poorly controlled incineration plants can emit high levels of particulate matter and contribute to local environmental problems. Modern incinerators emit lower levels, but data suggests that the particulates emitted are fine in size and therefore would be contributing to adverse health effects (EC 1998). Indeed, the majority of particles formed from combustion processes, including all types of waste incineration, are ultrafine particles that are less than 0.1 mm in size. Even the most modern MSW incinerators do not have technology that prevents the release of ultrafine particles. Collection efficiencies for respirable particles (less than 2.5 μm) are between 5 and 30 % using current bag filter technology. For particles less than 1 mm in size, which includes all ultrafine particles, most will pass through incinerator filtration systems unabated. Furthermore, there are indications that some of the modern pollution abatement equipment installed in incinerators, particularly ammonia injection, which attempt to reduce oxides of nitrogen, may actually increase the air emissions of the finest, most dangerous particles (Howard 2000).

At present, there is only limited information on the chemical composition of particulates. Emissions to the atmosphere from incinerators include, for example, particles formed of mineral oxides and salts from the mineral constituents in the waste (Oppelt 1990). Heavy metals and organic chemicals such as dioxins, PCBs and PAHs can adhere onto the surface of the particles. Metals may absorb in a number of different forms including metal oxides, soluble salts and metal carbonates. The chemical nature of particulates, for instance, the form of metal, or the type of other potentially toxic chemical adhered to the particle surface, may ultimately influence the effects on health resulting from exposure (QUARG 1996, Seaton 1995, Marty 1993).

Ultrafine particles have been found to be highly chemically reactive, even when they originate from material which itself is not reactive. This is solely due to their minute size. Research has shown that a proportionally higher number of

surface atoms are present as the particles size decreases. This leads to their surface becoming highly charged and therefore chemically reactive. In addition, ultrafine metal particles have been shown to be especially chemically reactive (Jefferson and Tilley 1999).

MSW incinerators typically have a mixed waste input containing heavy metals and halogenated organic compounds. They emit ultrafine metal particulates. Since these particles are especially reactive, it can be argued that MSW incinerators will therefore produce a more toxic ultrafine particulate aerosol than for example a coal-fired power station (Howard 2000). In this regard, incinerators are of utmost concern regarding health of the general public.

The new EC directive on incineration of waste does not give any limits for PM10, or perhaps even more appropriately, PM 2.5, that is respirable particles, less than 2.5 μm. In this way the directive ignores the particulate pollution from incinerators which is of most relevance to public health. The directive does specify a limit for total dust emissions to air of 10 mg/m^3 from incinerators. Data published in the 1980s gave air emissions of particulate from UK MSW incinerators ranging from 18-4105 mg/m^3 (Williams 1990), and from US hazardous waste incinerators ranging from 4-902 mg/m^3 (Dempsey and Oppelt 1993). A recent report on MSW incinerators in Sweden reported particulate emissions of 0.003 to 64 mg/m^3. Four out of 21 Swedish incinerators exceeded the EC limit on dust emissions (Greenpeace Nordic 2000).

5.1.5 Inorganic Gases

Inorganic acidic gases, notably hydrogen chloride (HCl), hydrogen fluoride (HF), hydrogen bromide (HBr), sulphur oxides (SO_x), and nitrogen oxides (NO_x) are formed and emitted by incinerators. These gases arise as a consequence of the elements chlorine, fluorine, bromine, sulphur and nitrogen being present in waste (Williams 1990). NO_x are also formed as a result of the direct combination of nitrogen and oxygen, a process that is accelerated at high temperatures.

HCl is emitted in greater quantities from incinerators than from coal-fired power stations. This is due to chlorine in the waste, notably in the form of plastics such as PVC (Williams 1990). The new EC directive sets a limit (daily average value) of 10 mg/m^3 for HCl and 1 mg/m^3 for HF (EC 1998). A recent study of 21 Swedish MSW incinerators reported that HCl emissions to air from 17 of them exceeded the EC limit, often to a substantial degree (Greenpeace Nordic 2000). The average release from the 21 incinerators was 44 mg/Nm^3 with a range of 0.2-238 mg/Nm^3.

Oxides of nitrogen (NO_x), including nitrogen dioxide (NO_2), and oxides of sulphur (SO_x), including sulphur dioxide (SO_2), are emitted from industrial combustion processes including all types of incinerators. These gases can also influence the pH of rain, making it acidic. Over time, acid rain can have deleterious effects on soil and water quality, and adversely affect ecosystems. Like exposure to particulate air pollution, exposure to NO_x and SO_x is also linked to adverse effects on respiratory health of individuals with pre-existing respiratory disorders. For instance, research has shown associations between increased air pollution levels of SO_2 and increased premature deaths in individuals who had pre-existing respiratory or cardiovascular illness. Similarly an association is evident with increased hospital admissions in individuals with pre-existing respiratory illness such as asthma or chronic obstructive pulmonary disease. Studies have also shown associations between exposure to NO_2 and worsened symptoms of respiratory illness although the data is not consistent or conclusive (Ayres 1998).

NO_x and SO_x emissions also result in the formation of particulates, known as secondary particulates. The formation of secondary particulates occurs as a consequence of these gases undergoing chemical reactions in the atmosphere. They originate from the chemical oxidation of sulphur and nitrogen oxides in the atmosphere to acids, which are subsequently neutralised by atmospheric ammonia. The particles formed include ammonium sulphate and ammonium nitrate. These particles, which are generally soluble in nature, persist in the air for long periods of time. A less abundant type of secondary particle is ammonium chloride which originates from HCl gas. Like primary particles, secondary particles can have a wide variety of other potentially toxic organic compounds adsorbed onto their surfaces such as PAHs, and dioxins (QUARG 1996, COMEAP 1995, EPAQS 1995). Like primary particulates from incinerators, secondary particulates are also thought to have adverse impacts on human health (e.g. see EC 1998)

Presently, NOx emissions from incinerators are not regulated through EC limits although limits are proposed in the new EC directive. A limit (daily average value) for nitrogen monoxide and nitrogen dioxide, expressed as nitrogen dioxide, of 200 mg/m^3 is proposed (for existing incineration plants with a capacity exceeding 3 tonnes per hour, or new incineration plants). A recent study of 12 Swedish MSW incinerators documented emissions ranging from 1.2 – 236 mg/Nm^3. 4 of the 12 exceeded the EC limit.

The EC directive on incineration of wastes proposes a limit (daily average value) for sulphur dioxide of 50 mg/m^3. A recent report on 10 Swedish incinerators found that emissions ranged from 1.2 to 236 mg/Nm^3. Of the 10 incinerators, 9 of them had emissions that exceeded the EC limit (Greenpeace Nordic 2000).

5.1.6 Other Gases

Carbon dioxide (CO_2) is emitted by incinerators. Municipal waste contains around 25% by weight of carbon and this is released as CO_2 when waste is burned. Approximately one tonne of CO_2 is produced per tonne of waste incinerated. CO_2 is a greenhouse gas that affects climate change and releases have to be kept as small as possible (EEA 2000). There is no EC limit on emissions of CO_2 from incinerators.

Carbon monoxide is also released from incinerators. It is potentially toxic and is also a greenhouse gas. Research suggests that increases in CO levels in the air may be linked to health impacts in certain susceptible individuals with pre-existing heart disease (Ayres 1998). A recent study on Swedish incinerators found that of the 15 incinerators which recorded emissions, 10 exceeded the new EC limit of 50 mg/Nm^3 (Greenpeace Nordic 2000). Emissions ranged from 2.6 to 249 mg/Nm^3.

5.2 Releases to Water

Incinerators emit wastes to water from cleaning equipment. Published scientific data on these emissions is very limited. Wastewater from wet exhaust gas cleaning contains heavy metals, the most significant in terms of quantity emitted and toxicity being lead, cadmium, copper, mercury, zinc and antinomy. Wastewater from wet slag removal equipment contains high levels of neutral salts and also contains unburned organic material from the residue (EEA 2000).

5.3 Releases to Ashes

Ashes from waste incineration generally contain the same pollutants as air emissions, but may differ in concentration and composition (EEA 2000). Fly ashes and bottom ashes contain dioxins and heavy metals although, as for air emissions, little is known about many other compounds present in fly ash.

5.3.1 Organic Compounds

Information about the contents of organic compounds in bottom ashes is scarce, with the exception of dioxins for which there are some data (EEA 2000).

Dioxins

Dioxin emissions from incinerators to air and water have decreased in recent years due to improvements in pollution control equipment. However, it is difficult to tell whether the total releases of dioxins from incinerators have declined at the same time. It is highly probable that while emissions to air via stack gases have decreased, releases with the ashes have increased. Indeed, it has been proposed that the total dioxin releases from incineration probably have not been reduced greatly in recent decades (Wikstrom 1999). A theoretical assessment of the total emissions from a MSW incinerator in Sweden also found that a reduction of dioxins emitted in flue gases would result in an increase in ash (GRAAB 1996). Thus, the total dioxin releases from the plant would remain the same, regardless of improvements in air pollution abatement technology.

There are relatively few data about dioxins in fly ashes and bottom ashes because many installations are not obligated to control them (Fabrellas *et al.* 1999, Greenpeace Nordic 1999). A theoretical assessment of releases from an incinerator in Sweden suggested that 97% of the total dioxin emissions from an incinerator would be present in the ash. This is in close agreement with direct measurements from an incinerator (Spittelau) in Austria, which showed that 99.6% of the total dioxin releases were in ash residues (Greenpeace Austria 1999). A study on a Spanish incinerator also noted that only a minor proportion of dioxin emissions is through stack gases, the majority being in ashes (Abad *et al*. 2000). In addition to chlorinated dioxins, it is also likely that other halogenated dioxins and furans are present in ashes, as in flue gases, such as brominated and mixed chlorinated/brominated compounds. A study on medical and MSW incinerator fly ashes found results suggesting that iodinated dioxins are also likely to be present (Kashima *et al.* 1999).

With regard to levels of dioxins in incinerator residues, the highest levels have been found in fly ash. Levels characteristically range from parts per trillion (ppt) to parts per billion (ppb), (EEA 2000). Research on eight MSW incinerators in Spain found mean levels in fly ash between 0.07 and 3.5 ng I-TEQ/g (ppb) (Fabrellas *et al.* 1999). Another study on a MSW incinerator in Spain reported levels which fell within this range from two measurements, which were 0.37 and 0.65 ng I-TEQ/g (ppb) (Abad *et al.* 2000). Particularly high levels were reported for one Spanish incinerator in 1997 (41 ppb TEQ) although levels in 1999 were lower (Stieglitz *et al.* 1999).

Lower concentrations are apparent in bottom ash samples, typically ppt levels (EEA 2000). For instance, mean values for 3 MSW incinerators in Spain were 0.006, 0.013 and 0.098 ng I-TEQ/g (ppb), (i.e. 6, 13 and 98 ppt TEQ), (Fabrellas *et al.* 1999). Similarly, levels in bottom ashes from five MSW incinerators in Bavaria, Germany ranged from 1.6 to 24 ppt TEQ (Marb *et al.* 1997). Ash from 18 new or updated medical waste incinerators in Poland

sampled in 1994-7 had substantially higher levels of dioxins ranging from 8-45 ppb TEQ (Grochowalski 1998).

Based on limited sampling, Abad *et al.* (2000) noted that although the highest concentrations of dioxins are present in fly ash, the high production of bottom ash in incinerators means that the annual output of dioxins in bottom ash is comparable to that of fly ash. However, a study of eight MSW incinerators in Spain calculated that the overall output of dioxins was higher for fly ash (Fabrellas *et al.* 1999). The total yearly output of dioxins from 8 MSW incinerators reported to be operating in Spain, based on point measurements, was flue gases 1-1.2, fly ashes 46.6-111.6 and bottom ashes 2-19 g I-TEQ/y (Fabrellas *et al.* 1999).

As mentioned in the previous section, dioxin inventories most often underestimate releases from incinerators because ashes are not included in calculations. A report on output of dioxins from Swedish incinerators has proposed that the Swedish EPA have grossly underestimated total incinerator emissions by underestimating ash contamination in ashes (Greenpeace Nordic 1999).

Other Organic Compounds

As previously discussed in this report, emissions of organic compounds to stack gases are multitudinous and fly ashes are similarly laden with numerous compounds. The EEA (2000) note that fly ash contains concentrated organic compounds, such as PAHs and soot as well as chlorinated organic compounds. PCBs are known to be present in fly ash (see e.g. Sakai *et al.* 1996). PCBs were reported to be detected in fly ash of hospital and MSW waste incinerators (Magagni *et al.* 1994), and in sewage sludge incinerator fly ash and bottom ash (Kawakami *et al*. 1998). The level of PCBs in fly ash from sewage sludge incinerators was 7.1 ng/g with the proportion of PCBs to dioxins being similar to that found in MSW incinerators. PCNs have also been identified in incinerator fly ash (Schneider *et al*. 1998).

A study on fly ash from MSW incinerators identified 72 different phenolic compounds in the ash including many unknown ones (Nito and Takeshita 1996). Most of the compounds were hydroxy compounds of PAHs, polychlorinated PAHs, PCBs and dioxins. The study noted that some of these halogenated hydroxy compounds may be persistent and toxic and their toxicities should be evaluated because they will be leached from fly ash into the environment after disposal in landfill. Another study identified many new kinds of aza-heterocyclic hydrocarbons (azaarenes and other basic compounds in fly ash (Nito and Ishizaki 1997). These compounds are produced by incomplete combustion and this study confirmed that incinerators are a source of them. The study identified 63 and 18 kinds of azaareness from two different fractions of fly ash respectively. Of these compounds, quinoline, alkylquinoline, benzoquinoline, benzacridine, azapyrene, azabenzopyrene, phenylpyridine, biphenylamine and their isomers comprised the majority. Of concern is that many of them are known to be carcinogenic or mutagenic compounds. Leaching of such compounds from fly ash in landfill would release these toxic chemicals into the environment.

5.3.2 Heavy Metals

Both fly ash and bottom ash residues from incinerators contain many heavy metals. Fly ash generally has higher metal concentrations than bottom ash if the large, unburned metal fragments from the bottom ash are excluded (Bucholz and Landsberger 1995). Table 5.4 shows concentrations of heavy metals detected in fly ash and bottom ash from two Spanish MSW incinerators (Alba *et al.* 1997) and table 5.5 shows concentrations detected in ashes from a US incinerator (Bucholz and Landsberger 1995). The concentrations of heavy metals in incinerator ashes are very high compared to background levels in the environment. For instance, if concentrations in bulk ash (combined fly + bottom ash) are compared with average concentrations of heavy metals found in soil globally, it is clear that bulk ash contains elevated amounts of many metals (Bucholz and Landsberger 1995). In addition, the process of incineration greatly enhances the mobility and bioavailability of toxic metals compared with raw municipal waste (Schumacher *et al.* 1998). Consequently, there is greater potential for leaching of metals into the environment from ashes dumped in landfill than from ordinary waste (see section 5.4.1).

A study on incinerator ashes from veterinary college incinerators in which animal carcasses are burned found that levels of metals varied considerably between incinerators (Thompson *et al.* 1995). Generally levels of metals in the ashes were much lower than levels found in MSW incinerator ashes. One exception was zinc, which was at a similar level. It was noted that burning of plastics in the waste may contribute to lead and zinc content in the ashes.

Given that incinerator companies are not required under national laws in many countries to routinely monitor ashes, published data on heavy metal levels in ashes and exceeding of those regulatory limits which are in place are sparse. One study in the US of hazardous waste incinerators that the metals which most frequently exceeded regulatory limits were arsenic, nickel and lead (Dempsey and Oppelt 1993).

Table 5.4 Ranges of elemental abundance in MSW incinerator ashes and in soil. All concentrations are in mg/kg unless otherwise specified.

Element	Fly Ash	Bottom Ash	Soil
Ag	46-55.3	17.5-28.5	0.1
Al	3.19-7.84%	6.20-6.68%	7.1%
As	269-355	47.2-52.0	6
Br	3830-3920	676-830	5
Cd	246-266	47.6-65.5	0.06
Co	11.3-13.5	65.2-90.3	8
Cr	146-169	623-807	100
Cu	390-530	1560-2110	20
Hg	59.1-65.0	9.1-9.7	0.03
In	1.50-1.67	0.45-0.71	0.07
Mo	14-26	100-181	2
Pb	3200-4320	2090-2860	10
Se	6.7-11.2	<2.52	0.2
Sn	470-630	300-410	10
Th	2.85-3.21	4.31-4.86	5
Ti	3300-6300	7500-18100	5000
V	27-36	46-137	100
Zn	13360-13490	6610-6790	50

Source: Buchholz and Landsberger (1995).

Table 5.5 Minor and trace element concentrations in MSW incinerator residues

Element	Fly Ashes (mg/kg dry residue)	Bottom Ashes (mg/kg dry residue)
Cr	365 18	210 8
Zn	9382 208	2067 ± 9
Pb	5461 236	1693 ± 22
Ni	117 2	53 ± 3
Cu	1322 90	822 ± 4
As	<50	<50
Cd	92 2	<12.5
Hg	0.29 0.03	<0.035

Source: Alba *et al.* (1997).

5.4 Disposal of Ashes

Fly ashes are potentially toxic because of their heavy metal and salt content and consequently they require proper management (Alba *et al.* 1997). They also contain other organic toxic chemicals including dioxins. According to the EEA (2000), the disposal of fly ash from waste incineration plants is a serious problem. Under some regulations fly ashes could be classified as hazardous waste (Alba *et al.* 1997). Indeed, due to the high content of lead and cadmium in fly ash, it is classified as toxic waste under Italian law (Magagni *et al.* 1994). In response to concerns regarding incinerator ash disposal, the International Ash Working Group was established to compile and evaluate available information (Sawell *et al.* 1995) and has subsequently published its findings (Chandler *et al.* 1997).

Unlike fly ash, bottom ash is generally not classified as special waste. Nevertheless, bottom ashes also contain toxic substances and according to information cited by Brereton (1996), the potential leaching rates of metals from bottom ashes are such that there is clearly an environmental concern attached to their disposal.

Presently, fly ash is usually disposed of in landfill whilst bottom ash is disposed of in landfill or is used in

construction materials. In Canada, most European countries and Japan, bottom ash is handled separately from fly ash, whereas the current trend in the US is to combine all the residues and dispose of this waste in dedicated landfills (Chandler *et al.* 1997). The cost of disposal of ashes is a significant impact on the total cost of incineration (Brereton 1996). Utilisation of ash for construction purposes reduces the costs of ash disposal. However, the hazardous nature of incinerator ashes, and the eventual release of hazardous compounds such as persistent chemicals or heavy metals back into the environment calls into question this use. Furthermore, Shane *et al.* (1993) showed that the extent to which ashes were mutagenic varied with time. For example, samples taken at different times from the same incinerator varied in their mutagenic potential. Since it is unlikely that incinerator ashes are regularly checked for mutagenicity this again raises questions about further uses. It has been noted that another possible use for incinerator ash is as a fertiliser. However, the uptake of certain metals such as cadmium from MSW ash amended soil into edible plants, and thus into the human food supply, often precludes the use of fly ash in this manner (see Shane *et al.* 1993). The uses of fly ash and bottom ash for construction and other purposes are further discussed in section 5.4.1 below.

5.4.1 Disposal of Fly Ash

In the UK, it has been reported that fly ashes are disposed of in ordinary landfills, some of which are unlined (Mitchell *et al.* 1992). This is of great concern because toxic components in the ashes, in particular heavy metals, will contaminate subsoil above background levels. Depending on the pH of the soil, rainfall can leach metals from the landfilled ash into groundwater used for drinking. Leaching is greatest under acidic conditions. Since the ashes are frequently co-disposed with ordinary municipal waste, the surrounding soils can be acidified through organic acids which are the breakdown products of landfilled waste. This leads to greater leaching of heavy metals (Marty 1993). Furthermore, dumping of incinerator ash in landfills is of greater significance than normal waste going to landfill because not only is the concentration of metals higher in ash than normal waste, but it is also likely to be in a more soluble form and therefore more likely to leach. In one UK study, it was noted that levels of zinc, lead and cadmium were of particular concern in incinerator fly ash (Mitchell *et al.* 1992). With regard to dioxins, according to the EEA (2000), these chemicals are strongly bound to the surface of ash residues, are highly insoluble in water, and consequently they will not leach to a significant extent from landfills to groundwater.

Tests on leaching of metals from incinerator ashes have shown that the quantity of elements/heavy metals, which leach, is determined in particular by pH. The more acidic the solution used, the greater the amount of leaching (e.g. Fleming *et al.* 1996, Buchholz and Landsberger 1995). Significant releases of cadmium, lead and chromium however have been found to leach under neutral conditions with distilled water (Mangialardi *et al.* 1998). Lead has been deemed the most leachable heavy metal from fly ash (Chandler *et al.* 1997). Studies on the leaching of heavy metals from incinerator ash with water that simulates acid rain, has shown that leaching of metals to a significant degree occurs most readily with the first washing of the ashes (Buchholz and Landsberger 1995). This study noted that from this initial leaching, the metals/elements Ag, Ba, Be, Cr, Cu, Mo, Pb, S, Ti, and Zn appeared to pose the greatest threat to groundwater. Leaching over longer time periods was much less, but As, Cd, Cu, Hg, Pb, S and Zn were identified as potential long-term hazards over the lifetime of ash dumped in a landfill. In terms of very long time periods over hundreds to thousands of years, it has been noted that little is known about the long-term leaching behaviour of incinerator residues (Chandler *et al.* 1997). This is of immense concern given that landfills are unlikely to be managed indefinitely.

Currently, at landfills where leachate from the waste is collected, it is generally disposed of to municipal wastewater treatment plants. Such leachates from fly ash in landfills may be particularly high in lead and cadmium (Chandler *et al.* 1997). These and other trace metals will thus be directly discharged to the environment where leachate is disposed of via the general wastewater treatment system.

In addition to leaching of chemical contaminants from landfills, pollutants may also re-enter the environment via landfill fires. Landfill fires have been reported to be common in Finland and research has shown the release of dioxins, PCBs, PAHs and other contaminants from such fires in Sweden and Finland (see Ruokojärvi *et al.* (1995).

Pre-treatment of fly ash before disposal is being used increasingly in an attempt to reduce leaching. In their document on dangerous substances in waste, the EEA stipulate that fly ash cannot be landfilled without pre-treatment (EEA 2000). The focus on pre-treatment has been towards a minimum cost treatment which brings leachability into conformance with guidelines for disposal. This most commonly involves stabilisation of the ash in cement. According to Brereton (1996), the stabilised waste may then be suitable as some form of fill, or should be suitable for regular landfill. Chandler *et al.* (1997) report that some incinerators in Germany, Sweden, Switzerland and Austria stabilise fly ash using cement. Once stabilised, the use of fly ash in construction materials is not common in many countries. Exceptions are the Netherlands where about 50% of fly ash is used as filler in asphalt and Austria where the ash is used in concrete construction

(Greenpeace Austria 1999). On this note, it is of concern that a study on the use of fly ash for construction material has shown that these materials may subsequently leach metals (Fleming *et al*. 1996). Furthermore, whether fly ash is directly landfilled, is stabilised and then landfilled, or is stabilised in construction materials, it is important to realise that weathering and erosion will eventually result in the re-entry of persistent pollutants from the ash, including heavy metals, back into the environment.

Another treatment of fly ash has involved further thermal treatment in an attempt to reduce dioxin content. This has been successful under experimental conditions (e.g. Buekens and Huang 1998). However, nothing appears to have been reported on the formation of other potentially toxic chemicals as a result of the process. Moreover, heavy metals will remain in the waste.

5.4.2 Disposal of Bottom Ash

Like fly ash, bottom ash from incinerators is either land-filled or is used for construction purposes. Tests on leachate from bottom ash in landfills has revealed leaching of inorganic salts, but negligible leaching of heavy metals in the short term (Chandler *et al.* 1997). In some European countries, including, Denmark, France, Germany, The Netherlands, significant quantities (40 to 60% or more) of bottom ash from incinerators is being used in construction purposes (Chandler *et al.* 1997). It is largely used as base and sub-base for road construction. It is also used under cycle paths. Research on the use of bottom ashes in concrete has determined that such concrete has a lower compressive strength than concrete made with conventional aggregate (Chang *et al*. 1999). It is important to note that there are serious and legitimate concerns regarding the use of bottom ash in construction materials due to the presence of toxic components in the ash which could later enter the environment. The future release of these compounds due to weathering and degradation may have detrimental consequences for man, particularly in cases where the substances may enter the food chain (Korzun and Heck 1990).

Some of the possible dangers of utilising fly ash and bottom ash have recently become apparent in the UK (ENDS 2000a). Many MSW incinerators were compulsory closed in the UK by the end of 1996 to comply with the EC "Air Framework Directive" (84/36/EEC) and the "Incineration of Municipal Wastes Directive" (89/429/EEC), (see Leach *et al.* 1999). One of the remaining and presently operating incinerators that was deemed to comply with the EC directives was the Byker incinerator, sited in Newcastle. From 1994 to 1999, a mixture of fly ash and bottom ash from the Byker incinerator in Newcastle has been used on allotments and on paths. Concern by local residents about possible toxic substances in the ash prompted the local health authority and council to organise an analysis of dioxins and heavy metals in the ash. Initial results showed high levels of dioxins in the ash and residents were advised that children under two years of age should not play on the allotments, eggs and animal produce from the allotments should not be consumed, and all vegetables should be washed or peeled before eating. The final results of the analysis showed that levels of several heavy metals in the ash and dioxins were far higher than usual background levels. The average concentration of dioxin was very high, 1373 ng TEQ/kg, with a maximum concentration of 4224 ng TEQ/kg. These levels exceed the relevant German regulatory guidelines for dioxins. For instance, restrictions on growing of agricultural crops are recommended above 40 ng TEQ/kg, and it is recommended that remediation should be carried out if playgrounds exceed 100 ng TEQ/kg and if residential areas exceed 1000 ng TEQ/kg. With the exception of mercury, all the other heavy metals tested exceeded the Dutch trigger values for soils, as shown in table 5.6. The Dutch guidelines are used by planning authorities in Britain. As a consequence of the high levels of toxic substances in the ash, all of it had to be removed. This was at a cost to the local council of £50-70,000. It is of great concern that the use of this ash for paths and allotments was permitted to happen by the regulatory authorities and begs the question whether similar incidences have occurred but remained unnoticed in the UK or in other countries.

The new EC directive (EC 1999) does not propose any limits for the quantity of heavy metals in fly ash or bottom ash. This is of concern given the fact that most heavy metals from incineration are sequestered in the ashes and pose an environmental contamination problem. The directive does state however that

"*appropriate tests shall be carried out to establish the physical and chemical characteristics and polluting potential of the different incineration residues. The analysis shall concern in particular the total soluble fraction and heavy metals soluble fraction*".

It also states that

"*residues shall be recycled as far as possible directly in the plant or outside in accordance with relevant Community legislation and national provisions*". Thus the EC condones the use of ashes for other purposes, which could lead to future environmental contamination and threats to health as discussed and exemplified above.

Table 5.6 Levels of metals (mg/kg) and dioxins and furans (ng/kg) in 16 Byker ash samples compared with Dutch Trigger Values

Substance	**Mean (mg/kg)**	**Range (mg/kg)**	**Dutch Trigger Value (mg/kg)**
Arsenic	12	7-23	20
Cadmium	5	0.4-11	1
Chromium	88	13-182	100
Copper	1,195	10-3,620	50
Mercury	0.2	0.1-0.6	0.5
Nickel	55	14-187	50
Lead	399	17-620	50
Zinc	659	31-1,420	200
Dioxins	1,373 ng TEQ/kg	11-4224 ng TEQ/kg	

Source: Buchholz and Landsberger (1995).

6. THE SOLUTION: REDUCE, RE-USE AND RECYCLE and PHASE OUT INCINERATION.

A lack of landfill space, tighter regulations to restrict the quantity of waste going to landfill together with environmental problems with old landfills have driven municipalities in many countries to look for new methods of handling waste. Presently, 60% of waste generated throughout countries in the European Union goes to landfill (Hens *et al.* 2000). This situation is made worse by the growing amount of waste being generated. For example:

- Total waste production in the EU rose by nearly 10% between 1990 to 1995 and a further 20% increase has been predicted to occur by 2010 (EEA 1999).
- In Estonia, Slovenia, Lithuania, Slovak Republic, Bulgaria, Hungary, Czech Republic, Romania, and Poland, economic growth may lead to a doubling of municipal waste generation by 2010 (EEA 1999).
- In Asia, municipal waste from urban areas is predicted to double by 2025 (World Bank 1999).

One of the methods being chosen to deal with the current waste crisis is incineration, a method which is promoted as reducing the volume of solid waste thereby lessening the burden on landfill. However, incinerators are not the solution to the waste problem. Indeed, they are symptoms of non-existent and/or ill-conceived policies for the management of material resources. In a world of shrinking resources, it is irrational to let valuable resources "go up in smoke," and doubly so when the smoke is known to carry persistent and other hazardous chemicals. Incineration cannot be regarded as a sustainable technology for waste management and has no place in a world striving to change towards zero discharge technologies.

It is notable that incineration has already been banned by the government of the Philippines, a move primarily instigated by public opposition to incineration. The Philippines is the first country in the world to ban incineration on a national scale. The Philippine Clean Air Act of 1999 specifically bans the incineration of municipal, medical and hazardous wastes and recommends the use of alternative techniques (for municipal waste) and non-burn technologies. Waste reduction, re-use and recycling are being promoted. The Clean Air Act mandates a three-year phase out period for existing medical incinerators, and during this time, limits hospitals to incineration of infectious waste.

6.1 Problems of Incineration

6.1.1 Environment and Health

No matter how modern an incinerator is, these facilities inevitably result in the release of toxic emissions to air and the production of toxic ashes and residues. This leads to contamination of the environment and to potential exposure of animals and humans to hazardous pollutants. Many hazardous compounds are released from incinerators including organic chemicals such as chlorinated and brominated dioxins, PCBs and PCNs, heavy metals, sulphur dioxide and nitrogen dioxide. Furthermore, innumerable substances are emitted which are of unknown toxicity. The entire impact on human health of exposure to the whole mixture of chemicals emitted from incinerators is unknown. However, studies imply that individuals who work at waste incinerators and who live near incinerators have suffered from increases in the rate of mortality as well as many other diseases and effects that diminish the quality of their lives. Moreover, a prestigious scientific body has recently expressed "substantial" concern about the impacts of incinerator-derived dioxin releases on the health and well-being of broader populations, regardless of the implementation of maximum achievable control technology (NRC 2000).

6.1.2 Economics

The economics of waste management in general, and in particular incineration, are extremely complex and are outside the scope of this report. Briefly, it has been noted that incineration is a technology of the previous industrial era and is only economically feasible if much of its cost is externalised ie. borne by the general public. Pollution control constitutes a major proportion of the cost, but using such technology to reduce the toxics entering the air cannot help but redistribute them back to deposits in the ash.

A recent trend has been to generate electricity from burning waste in MSW incinerators. This can only be seen as a by-product of incineration and not a contributor to sustainable energy production. Indeed, incinerators such as MSW incinerators, are inefficient energy producers with only 20% of the energy generated by the waste usually being captured. Murray (1999) has described incineration as inefficient both as a disposal option and as an energy generator. It leads not to material conservation and hazard reduction but to material destruction and hazard creation.

In the UK, a situation has arisen whereby contracts with incinerator operators lock local authorities into long term commitments to provide huge amounts of waste each year. This works against waste prevention, re-use and recycling since local authorities would have to pay financial penalties to incinerator owners if waste was reduced and diverted to re-use/recycling schemes.

6.1.3 Sustainability

The Convention for the Protection of the Marine Environment of the North East Atlantic (the OSPAR Convention, formed from the amalgamation of the former Oslo and Paris Conventions) entered into force in March 1998 and covers the 15 States of the North East Atlantic

Region and the European Union. At the OSPAR meeting held in Sintra in June 1998, Ministers at OSPAR agreed on a clear commitment for the cessation of release of hazardous substances within one generation (by 2020). In essence, the commitment means that a target has been set for the cessation of discharges, emissions and losses of hazardous substances (or the processes that generate them) and their substitution with non-hazardous alternatives. In practice this means a shift away from dirty technologies towards clean production and zero emission strategies. Incinerators can never comply with the zero emissions strategy or be classed as a clean production technology. This old, dirty technology is not in agreement with sustainable development or political commitments already made within Europe. In effect, under the provisions of the OSPAR agreement, incineration is finally and irrevocably made obsolete.

6.2 Current EU Policy and Waste Management

Waste policy in the EU widely accepts the hierarchy of waste management to be (in order of priority): waste prevention – re-use – recycling – thermal decomposition with energy recovery (i.e. incineration with energy recovery). In spite of this general consensus, and a growing coherence of this hierarchy in policy lines of individual EU member states as a consequence of EU-Directives, the majority of waste in Europe is either landfilled or incinerated. Importantly, these are the methods which also entail the highest and most serious environmental and health risks (Hens *et al.* 2000).

A move towards a waste policy aimed at reducing health effects should put more emphasis on prevention and re-use. Presently, EU waste policy is not founded upon health data. Fortunately the available data on health effects from waste management do not conflict, and in important aspects even coincide with the hierarchy proposed by the EU (Hens 2000). For example, waste prevention is deemed to be the most important (no waste equals no health effects), followed by re-use and recycling. Despite this, the lack of consideration of the environment and human health is clearly visible in EU policy. For instance, regulations put in place for incineration by the EU together, with national limits on this issue, are based on what is technically achievable rather than on health and environmental data.

Although emission limits set in the new EU directive have resulted in the closure and upgrading of some older incinerators in European countries, the policy itself is already outdated with regard to the OPSPAR agreement to phase out the releases of all hazardous substances within one generation. The EU directive is based on the conception that small releases of hazardous substances are acceptable. This is the conventional (though misguided) approach which proposes that chemicals can be managed at "safe" levels in the environment. However, it is already known, or is a scientific opinion, that there are no "safe" levels of many environmental chemical pollutants such as dioxins, other persistent, bioaccumulative and toxic chemicals and endocrine disruptors. In addition, the abandonment of the principle is increasing in political circles. For instance, with regard to incineration, the UK environment minister, Mr. Michael Meacher, recently recognised the futility of the conventional approach to chemicals regulation when he said:

Q440…"I repeat that emissions from incinerator processes are extremely toxic. Some of the emissions are carcinogenic. We know scientifically that there is no safe threshold below which one can allow such emissions" (cited in Howard 2000).

Despite the commitment by the OSPAR Convention for the cessation of all hazardous substances by 2020, a recent trend for plans to build new incinerators by the government in the UK and other European countries continues.

6.3 The Way Forward: Adoption of the Precautionary Principle and Zero Emissions Strategy

6.3.1 Adoption of the Precautionary Principle

The precautionary principle acknowledges that, if further environmental degradation is to be minimised and reversed, precaution and prevention must be the overriding principles of policy. It requires that the burden of proof should not be laid upon the protectors of the environment to demonstrate conclusive harm, but rather on the prospective polluter to demonstrate no likelihood of harm. The precautionary principle is now gaining acceptance internally as a foundation for strategies to protect the environment and human health (Stairs and Johnston 1991).

Current regulation for incinerators is not based on the precautionary principle. Instead it attempts to set limits for the discharge of chemicals into the environment which are designated as "safe". In the current regulatory system the burden of proof lies with those who need to 'prove' that health impacts exist before being able to attempt to remove the cause of the problem and not with the polluters themselves (Nicolopoulou-Stamati *et al.* 2000). Based on knowledge regarding the toxic effects of many environmental chemical pollutants, which has accumulated over recent decades, a more legitimate viewpoint is that "chemicals should be considered as dangerous until proven otherwise".

We have now reached a situation, and indeed did some time ago, where health studies on incineration have

reported associations between adverse health effects and residing near to incinerators or being employed at an incinerator. These studies are warning signs which should not result in government inactivity, but rather to decisions being taken which implement the precautionary principle. There is already sufficient human health and environmental contamination evidence to justify a phase out of the incineration process based on the precautionary principle. To wait for further proof from a new generation of incinerators from an already harmful and dirty technology would be a blatant disregard for the environment and human health.

6.3.2 Adoption of Zero Discharge

The aim of "zero discharge" is to halt environmental releases of all hazardous substances. Although it is sometimes discussed as being simplistic or even impossible, it is a goal whereby regulation can be seen as resting places on the way to achieving it (Sprague 1991).

Zero discharge necessitates the adoption of clean production techniques both in industry and agriculture. It is essential that the change to clean production and material use should be fully supported by fiscal incentives and enforceable legislation.

The principle of clean production has already been endorsed by the Governing Council of the UNEP and has received growing recognition at a wide range of international fora. For instance, the adoption of the one generation goal for the phase out of all hazardous substances by the OSPAR Convention in 1998 necessitates instigating clean production technology under a zero discharge strategy.

In terms of waste management strategies, incineration is a dirty technology that can never fulfil the criteria of zero discharge. The way forward for waste management in line with a zero emissions strategy and hence towards sustainability, lies in waste prevention, re-use and recycling. In other words the adoption of the already well known principle of "REDUCE, RE-USE AND RECYCLE".

6.3.3 Implementation of REDUCE, RE-USE AND RECYCLE

We live in a world in which our resources are generally not given the precious status by industry and agriculture which they deserve. In part, this has led to the creation, particularly in industrialised countries, of a "disposable society" in which enormous quantities of waste, including "avoidable waste" are generated. This situation needs to be urgently changed so that the amount of waste produced both domestically and by industry is drastically reduced.

Ways to help waste reduction include the use of economic instruments and environmental taxes. The use of these measures is supported by the EC and a number environmental taxes are already in place in several European countries (Steenwegen 2000). However, far more action is presently required to stimulate the change needed for much more waste reduction to become a reality.

Current levels of recycling in European countries vary considerably. For instance, The Netherlands recycles 46% of municipal waste whereas the UK only manages 8%. Intensive re-use and recycling schemes could deal with 80% of municipal waste. It is recognised that fiscal measures can play a considerable role in encouraging re-use and recycling schemes whilst discouraging least desirable practices such as incineration and landfill (Steenwegen 2000).

Measures to be taken in the drive towards increased waste reduction, re-use and recycling, and therefore towards lessening the adverse health effects from waste management should include:

- The phase out of all forms of industrial incineration by 2020, including MSW incineration. This is in line with the OSPAR Convention for the phase out of emissions, losses and discharges of all hazardous substances by 2020.

- Financial and legal mechanisms to increase re-use of packaging (e.g. bottles, containers) and products (e.g. computer housings, electronic components).

- Financial mechanisms (such as the landfill tax) used directly to set up the necessary infrastructure for effective recycling.

- Stimulating markets for recycled materials by legal requirements for packaging and products, where appropriate, to contain minimum amounts of recycled materials.

- Materials that cannot be safely recycled or composted at the end of their useful life (for example PVC plastic) must be phased out and replaced with more sustainable materials.

- In the short term, materials and products that add to the generation of hazardous substances in incinerators must be prevented from entering the waste stream at the cost of the producer. Such products would include electronic equipment, metals and products containing metals, such as batteries and florescent lighting, and PVC plastics

(vinyl flooring, PVC electrical cabling, PVC packaging, PVC-*u* window frames etc) and other products containing hazardous substances.

and more generally:

- Further the development of clean production technologies which are more efficient in terms of material and energy usage, produce cleaner products with less wastes and which ultimately can operate in a "closed loop" configurations to serve the needs of society in a more equitable and sustainable manner;

- Implement fully the Precautionary Principle, such that, in the future, we may be better able to avoid problems before they occur. The continuation and further development of scientific research has a fundamental role to play in identification of potential problems and solutions, but we must be ready to take effective precautionary action to prevent environmental contamination and degradation even in the face of considerable and often irreducible uncertainties.

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APPENDIX A

HEALTH EFFECTS OF SPECIFIC POLLUTANTS EMITTED FROM INCINERATORS

1 Particulate Matter

1.1 Introduction

Animal and plant life as we know it evolved in the presence of particulate matter (Howard 2000). This matter consists of minute particles, sometimes called particulates. Natural sources of particulates include soil particles which are blown into the air by the wind, dusts from volcanic eruptions, particles of sea salt ejected into the air by breaking waves on the sea, spores from fungi and pollen grains from plants. These particles vary in size from those coarse-sized particles that are visible to the naked eye to tiny microscopic particles, measured in micrometers (μm). The finer particles tend to remain airborne for long periods of time whereas coarse particles, such as wind blown soil, and plant pollens, tend to fall rapidly and remain airborne for only short periods of time. Most naturally produced particles are generally greater than 20 μm in size (QUARG 1996, COMEAP 1995, EPAQS 1995).

The human respiratory system has evolved to cope with an environmental loading of such naturally produced aerial particles. The muco-cilary lining (mucus and fine hairs) of the airways operates to protect the deeper regions of the lung. Fine particles of less than 10 μm do occur naturally, mainly in the form of re-suspended sea salts. In this form, the fine particulates pose no threat to health and, if inhaled, are simply absorbed into the body (Howard 2000).

Particulates are also formed as a consequence of human activities. When humans started to use fire domestically, exposure to fine insoluble particles of material, less than 10 μm in size, must have occurred on a regular basis (Howard 2000). However, the advent of industrialized society has seen a vast increase in the level of industrial combustion processes around the world.

The major source of man-made particles arises from combustion processes such as incineration of wastes, coal-burning and vehicle exhausts and to a lesser extent from metallurgical and other industrial processes. All of these processes inevitably result in the direct emission of particles into the atmosphere. Such particles which are directly emitted into the atmosphere are called primary particles. Furthermore, as well as this direct emission of particles from combustion processes, some pollutant gases which are released from combustion processes, such as sulphur dioxide and nitrogen dioxide, undergo chemical reactions in the atmosphere in which particles (known as secondary particles) are also formed. These particles are mainly comprised of ammonium sulphate and ammonium nitrate. Secondary particles can also have a wide variety of other toxic organic compounds adsorbed onto their surfaces such as PAHs (QUARG 1996, COMEAP 1995, EPAQS 1995).

Particulates formed from human activities are generally fine particles (less than 10 μm) and even smaller ultrafine particles (less than 0.05 μm i.e. 50 nanometers (nm) in size). There is much scientific evidence which indicates that that particles less than 10 μm, have adverse effects upon the human health. Most notably, the most recent views and data suggest that it is the number of ultrafine particles, and possibly their chemical composition, which causes health impacts (QUARG 1996, Seaton *et al.* 1995). Ultrafine particles may be acidic in nature and an irritant to the lungs or carry toxic substances on their surface such as halogenated organic materials, including dioxins, metals and PAHs (COMEAP 1995).

The respiratory system has mechanisms to expel particulates to protect the deep lung regions. Naturally produced particles are generally of sizes 2.5 to 10 μm or larger. All particles below 10 μm may reach the furthest parts of the lungs but, in general, those particles sized between 2.5 and 10 μm are most likely to be deposited in the upper airways of the lungs. From here they are efficiently removed. Only a small fraction of naturally produced particles will be deposited in the deepest regions of the lungs. The body does have mechanisms which clear particles from the deep parts of the lungs, although they are less efficient than in the upper airways. However, unlike natural particles, a high proportion of particulates from industrial combustion processes are less than 2.5 μm in size. These "respirable" particles can reach and be deposited in the deep regions of the lungs and are thought to have the most significant impacts on health.

While it is recognized that there would have been naturally and man-made combustion processes in the pre-industrial age, the level of particulate production would have been very small by comparison with today. Recent studies in the UK, estimated that primary and secondary particles derived from industry each constitute about a third of the total PM10 burden in air, with the remaining third coming from marine aerosol (see Howard 2000). Differences in the levels of particulates in air are clearly influenced by local sources. For instance, a study in the city of Birmingham, UK, showed that the number of particles present in urban air ranged from 1000 to 100,000 particles per cm^3. Rural air contained lower numbers of particles, averaging 5000 to 10,000 per cm^3 but rose when influenced by nearby traffic. Less polluted air such as that over the North Atlantic Ocean typically contains around 200 particles per cm^3 (QUARG 1996, Seaton 1995).

Since the size of particles has a direct bearing on health impacts, measurements made in recent years have quantified particles in the atmosphere according to size. The most commonly used measurement is known as PM10, which estimates the mass of particles in air which are less

than 10 um in diameter. PM10 reflects the size of particles which are considered to be most likely deposited in the lung. It has been used to quantify particulate concentrations in many studies on health impacts of particulate air pollution.

1.2 Health Effects of Particulates

From the 1930s to the 1950s there were a series of severe air pollution disasters in industrialized cities including London, Meuse Valley Belgium and Donora, Pennsylvania. Dense winter smogs, caused by particulate pollution and sulphur dioxide from coal combustion, resulted in substantial increases in the death rate from respiratory and cardiovascular diseases. These incidents proved that high levels of particulate air pollution in smog caused an increase in daily death rate (see Schwartz 1994a). Such incidents became a thing of the past by the 1960s in western industrialized countries, due to changes in the types of fuel and better pollution control. The nature of air pollution also changed as a result of increases in vehicle emissions. Since the mid-1980s, much research has been carried out to determine whether the lower levels of particulates and other air pollutants to which populations are exposed nowadays also cause an increase in death rate and disease. It has been shown by many human epidemiology studies on this subject that particulate pollution is indeed linked to worsening respiratory diseases and increasing premature mortality from respiratory and heart diseases (see Pope *et al.* 1995a).

Studies on short-term fluctuations in the level of air pollution in a region and the daily death rate of the region have consistently found that small elevations in the daily death rates are associated with short-term increases in particulate air pollution (e.g. see Pope *et al.* 1995a, Schwartz 1994a). Importantly, these studies have not revealed a "safe level" of particulates, that is a threshold, below which the death rate did not increase. Thus, the association between increasing mortality with increasing particulate pollution was apparent in studies even in areas where air pollution was relatively low and was well within US air quality standards. It was estimated from the study data that for every 10 mg/m^3 increase in PM10 there was an associated 0.5-1.5% increase in daily mortality. The premature deaths were due mainly to an increase in deaths from respiratory diseases and secondarily from cardiovascular diseases. Researchers have reviewed the studies and concluded that it is unlikely the observed increased death rates were due to confounding factors that could also affect the death rate such as weather conditions or other air pollutants. It has therefore been suggested that it is reasonable to interpret results from the studies as showing a causal relationship between PM10 and daily mortality (e.g. Pope *et al.* 1995a, Schwartz 1994a).

In addition to research on short-term changes in particulate pollution and mortality rates, studies have also been conducted on the effect of exposure to long-term air pollution on mortality. These studies compared the annual average mortality rates of populations living in different areas, with the annual average concentrations of air pollution in those areas. They again revealed an association between particulate pollution and increased mortality. Death rates were found to be higher in cities with higher levels of fine or sulphate particulate pollution than those with lower levels. In fact, results showed that between 3 and 9% of all urban deaths appeared to be due to particulate air pollution (Pope *et al.* 1995b). Since this figure is so high, these studies have come under criticism for inadequate control of confounding factors. However, two further studies which overcame the problem of confounding (Pope *et al.* 1995c, Dockery *et al.* 1993), including one study on a 8-year follow up of over half a million US adults (Pope *et al.* 1995c), also found that average long-term levels of fine particulate and sulphate air particulate pollution were linked with mortality. The risk of mortality was about 15-25% higher in cities with the highest particulate pollution compared to cities with the lowest levels. The causes of death associated with particulate pollution were respiratory diseases, including lung cancer, and cardiovascular diseases, particularly amongst the chronically ill or elderly. Therefore, studies on both short-term changes in air pollution, and long-term levels of air pollution have indicated an association between mortality and particulate air pollution.

If the relationship between particulate pollution and mortality is causal, then it would be expected that morbidity (ill-health) would also be associated with particulate pollution levels (Pope *et al.* 1995b). This is indeed the case. Many studies have been conducted using a variety of endpoints to monitor health effects and have found an association between short-term changes in particulate air pollution and illness. The results are illustrated in the list of bullet points given below. Overall, these studies suggest that particulate pollution is able to cause temporary worsening of already existing respiratory illnesses.

Particulate air pollution has been associated with:

- Increase in hospital admissions for respiratory illness in winter (e.g. Schwartz 1994a, Pope 1991, Pope 1989) and in summer (e.g. Schwartz 1994a b, c and d, Thurston *et al.* 1993).
- Increase in hospital emergency department visits for respiratory diseases (see Pope 1995a, e.g. Schwartz *et al.* 1993, Sunyer *et al.* 1993).
- Exacerbation of mild and severe asthma attacks in adults and children (e.g. Pope *et al.* 1995b, Walters *et al.* 1994, Ostro 1993, Roemer *et al.* 1993,

Thurston *et al.* 1993, Pope 1989, Bates and Sizto 1987, Whittemore and Korn 1980).

- Increase in respiratory symptoms (particularly lower respiratory symptoms, such as wheezing, dry cough, phlegm, shortness of breath, chest discomfort/pain), (e.g. Sram *et al.* 1996, Schwartz *et al.* 1994b, Braun-Fahrlander *et al.* 1992, Pope and Dockery 1992).
- Small decreases in lung function (e.g. Hoek and Brunekreef 1994 and 1993, Koenig *et al.* 1993, Neas et al. 1992, Pope and Dockery 1992, Pope *et al.* 1991, Dassen *et al.* 1986, Dockery *et al.* 1982).
- Work and school absenteeism through ill health (Sram *et al.* 1996, Ransom and Pope 1991, Ostro 1990, Ostro and Rothschild 1989).

Results of epidemiology studies discussed in this section provide evidence that day-to-day variations in current air particulate concentrations are associated with transient effects on health and increases in the daily death rate. This has provoked a great deal of controversy. Critics have argued that results were artefactual and contrary to common sense and established doctrine. However, a number of independent experts have also reviewed the evidence and have suggested that the associations are causal. This view has been accepted by the UK Department of Health (Maynard 2000).

In addition to transient health effects due to short-term changes in particulate air pollution, research has also been conducted to investigate whether long-term exposure to particulate pollution over one year or more has any long-lasting or cumulative effects on health. Several studies have showed an association. For example, some studies found an association with reduced lung function (e.g. Sram *et al.* 1996, Chestnut *et al.* 1991, Schwartz 1989) and increased respiratory symptoms especially bronchitis (Abbey et al. 1995, see Pope *et al.* 1995b).

In sum, epidemiology data indicates that the link between fine particulate pollution and adverse effects on health is most likely causal (see e.g. Pope 1995b, COMEAP 1995). Less is known about the mechanisms by which particulates impact on health. Presently, it is thought that ultrafine, possibly acidic particles could cause inflammation of tissue in the lung extremities which in turn may provoke respiratory and heart related illnesses and death (e.g. Seaton 1995, see Howard 2000 and 2000b). Research suggests that it may be the smallness of ultrafine particles themselves, their high surface area for the release of transition metals, their insolubility and their possible generation of free radicals which may be the most important factors contributing to make the particle cause inflammation (Donaldson *et al.* 2000).

2. Dioxins

Dioxins are toxic, persistent in the environment and they are bioaccumulative, i.e., they build up in the tissues of animals and humans alike. Dioxins are the unintented by-products of human activities that involve the manufacture and use of elemental chlorine and the combustion of materials that contain any form of chlorine. Dioxin levels in the populations and environments of industrialised regions began increasing markedly following World War II, with the manufacture and dispersal of chlorinated pesticides and other chemicals and their associated wastes.

By the late 1970s and early 1980s, some scientists were beginning to recognize that dioxins were ubiquitous in the populations and environments of industrialised regions. Moreover, they discovered that waste incinerators were releasing dioxins in their stack gases and ashes. By this time, however, the technology was well entrenched. Once policymakers were finally convinced of the widespread nature of the dioxin problem and its link to incineration, some regions had come to depend on incineration, both as a waste disposal option as well as an economic enterprise for the industrial sector. As a consequence, these regions have become even more heavily invested in incinerator control technologies. Unfortunately, these methods primarily change the nature of the dioxin problem but do not solve it. i.e., a larger share of the dioxins that were once released into the air are captured in stack filters or in bottom ashes and placed in landfills. This will retard the dispersal of dioxins into the environment but will not, in the long-term, prevent it.

Dioxin contamination is not restricted to the local areas surrounding incinerators and other sources. Like other persistent organic pollutants, they are transported for thousands of kilometres on air currents and have consequently become globally ubiquitous pollutants. It is thought that every man, woman and child on the planet now carries dioxins in their body tissues.

The toxicology of dioxins, in particular TCDD, has been extensively researched. A considerable amount of work has also been put into conducting research on wildlife and human health effects of dioxin exposure. TCDD was recently classified as a human carcinogen by the International Agency for Research on Cancer (McGregor *et al.* 1998).

Dioxins exert a plethora of toxic effects because they act on a fundamental biochemical regulation system in the body, a system that is common to animals and humans. Dioxins exert their effects through binding to the "Ah receptor", the outcome of which affects several genes (Webster and Commoner 1994). Their toxicological effects are summarised in table 1.

Table 1 Toxicological effects of dioxin

Carcinogenesis	IARC class 1 carcinogen (carcinogenic to humans).
Immune system effects	Suppression of cell-mediated and humoral immunity; increased susceptibility to infectious challenge; auto-immune response.
Male reproductive toxicity	Reduced sperm count; testicular atrophy; abnormal testis structure; reduced size of genital organs; feminized hormonal responses; feminized behavioural responses.
Female reproductive toxicity	Decreased fertility; inability to maintain pregnancy; ovarian dysfunction; endometriosis.
Developmental impacts	Birth defects; foetal death; impaired neurological development and subsequent cognitive deficits; altered sexual development.
Modulation of hormones, receptors, and growth factors	Steroid hormones and receptors (androgens, estrogens and glucocorticoids); thyroid hormones; insulin; melatonin; vitamin A; EGF and receptor; TGF-a and TGF-b; TNF-a, IL-1b, c-Ras, c-ErbA.
Other effects	Organ toxicity (liver, spleen, thymus, skin); diabetes; weight loss; wasting syndrome; altered fat and glucose metabolism.

Source: Aapted from USEPA 1994 and Birnbaum 1994

Occupational studies have reported that exposure to dioxin has been associated with a wide range of effects including chloracne (skin lesions), changes in the levels of liver enzymes, changes in the levels of thyroid hormones, sex hormones, and cells of the immune system (reviewed by Sweeney and Mocarelli 2000). Exposure to dioxins in the workplace has also been associated with an increased risk of cancer when all cancers together are considered (eg. Fingerhut *et al.* 1991, Manz *et al.* 1991, Zorber *et al.* 1990).

For the general population of industrialised countries, research indicates that dioxins are exerting effects on people at current background levels found in the environment. For instance, dioxins can affect the levels of certain hormones, enzymes and immune system cells, at body concentrations at, or near to, the levels currently found in the human populations of industrialised countries. According to DeVito *et al.* 1995:

"Subtle changes in enzyme activity indicating liver changes in levels of circulating reproductive hormones in males, in reduced glucose tolerance potentially indicative of risk of diabetes, and in cellular changes related to immune function suggest the potential for adverse impacts on human metabolism, reproductive biology, and immune competence at or within one order of magnitude of average background body burden levels… Individuals at the high end of the general population range may be experiencing some of these effects. Some more highly exposed members of the population may be at risk for frankly adverse effects including developmental toxicity, reduced reproductive capacity based on decreased sperm counts and potential for increased fetal death, higher probability of experiencing endometriosis, reduced ability to withstand an immunological challenge and others."

Intake of dioxins in the diet of populations in Europe is frequently in excess of the tolerable daily intake (TDI) set by WHO (1-4 pg ITEQ/kg/day), particularly when dioxin-like PCBs are included (WHO 1998, see Allsopp *et al.* 2000). WHO experts acknowledged that subtle effects on health may already be occurring in the general population and that efforts should be made to ensure that intakes are at the lower end of the TDI range. Intake by breast-fed infants is very high compared with the TDI. This is of particular concern because the developing stages of life are most vulnerable to toxic insult from such chemicals.

Studies carried out during the past decade in the Netherlands have investigated the impacts of current background levels of dioxins and PCBs on foetal development, during infancy and childhood. Healthy women from the general population were selected for these studies. Results have revealed undesirable impacts on health of the immune system and nervous system during development that are associated with exposure to dioxins and PCBs during development.

For instance, some of the women had higher levels of PCBs and dioxins in their bodies and breast milk than others. It

was found that infants who were exposed to higher levels of these chemicals in the womb, and via breastfeeding, had changes in the number of certain immune system cells. It is not known what effect such changes could have on their health (Weisglas-Kuperus *et al.* 1995). Regarding effects on the nervous system, there was a slight adverse effect on psychomotor development in infants who were exposed to higher levels of PCBs and dioxins in the womb and during breastfeeding (Koopman-Esseboom *et al.* 1995). There was also a slight adverse effect on neurological development in the children detected at 18 months of age. Tests on neurological development looked at movement co-ordination, (eg. sitting, crawling, standing and walking), and are a way of measuring the quality and integrity of brain function (Huisman *et al.* 1995). At the age of 2 years and 7 months, tests on a subgroup of infants found slight changes in some measures of neurological development in the more highly exposed individuals. These changes were regarded as "unwanted" by researchers, and it was proposed that they could be due to the action of dioxins on thyroid hormones during development (Ilsen *et al.* 1996). A summary of the effects of perinatal exposure to dioxins and PCBs is presented in Table 2.

Table 2: Effects of Perinatal Exposure to Dioxins and PCBs on Infants and Children

Central nervous system	Delayed cognitive development, mildly disordered behavior, and increased activity in children of mothers who were accidentally exposed to extraordinary levels of dioxins/PCBs.
	Deficits in autonomic maturity and reflexes, less preference for a novel stimulus, and defects in short term memory in children whose mothers were exposed to background levels of PCBs and dioxins.
	Delayed motor development, hypotonia and hyporeflexia in children exposed to background levels.
	Increased hypotonia, lower psychomotor developmental indices, less optimal neurological condition, and lower cognitive scores in children whose mothers were exposed to background levels.
Immune system	More frequent occurrence of bronchitis, upper respiratory infections and ear infections among children whose mothers had extraordinary prenatal exposure.
	More frequent ear infections and altered levels of certain cells that have roles in warding off diseases in Inuit infants whose mothers had elevated exposure through their diet of traditional foods.
	Altered levels of certain cells involved in resisting diseases among children whose mothers had background exposures.
Growth, sexual development and reproductive health	Fewer boy children were born to couples in which both parents had high dioxin exposures during the seven year period following a large dioxin release at a chemical manufacturing facility.
	Lower weight at birth and continued diminished height and weight at school age among children whose mothers had extraordinaryl exposure.
	Reduced penis length among boys who were conceived in the earliest years after their mothers had extraordinary exposure.
	Altered birthweight and gestational age among infants of mothers who had occupational exposure to PCBs.
	Lower birthweight and smaller head circumference among infants of mothers whose diets included fish from the Great Lakes.

Growth, sexual development and reproductive health	Lower birthweight and slower postnatal growth until 3 months of age among infants whose mothers had background exposure.
Thyroid function	Subtle alterations in levels of thyroid hormones in pregnant mothers and their infants exposed to background levels of PCBs and dioxins.

3. Heavy Metals

3.1 Lead

Lead has no known nutritional biochemical or physiological function (Goyer 1996). The toxic effects of lead are the same, irrespective of whether it is ingested or inhaled, and blood levels less than10-100 mg/dl in children, and 10-100 mg/dl in adults have been associated with a wide range of adverse effects. These include nervous system disorders, anaemia and decreased haemoglobin synthesis, cardiovascular disease, and disorders in bone metabolism, renal function and reproduction. Of particular concern, is the effect of relatively low exposure on cognitive and behavioural development in children (Pirkle *et al.* 1998, USPHS 1997, Bernard *et al*. 1995, Goyer 1993, Nriagu 1988). It is clear that increased body burden of lead results in decreased scores on measures of intelligence from early infancy through school age. It also results in effects on behaviour of school children, including increased distractibility, short attention span and impulsivity (Rice 1996).

In 1975 the Centre for Disease control (CDC) in Atlanta recommended that the maximum permissible level of blood-lead be 30 ug/dl (for both adults and children). This levels was revised downward in 1985 to 25 ug/dl, and again in 1991, defining a blood-lead of 10 ug/l as an action or intervention level (USPHS 1997). Perhaps even more importantly is the now suggested recommendation that there may be no level of blood-lead that does not produce a toxic effect, particularly in the developing central nervous system (USPHS 1997, Goyer 1993).

3.2 Cadmium

Cadmium has no biochemical or nutritional function, and it is highly toxic to both plants and animals (USPHS 1997, WHO 1992, Alloway 1990). In humans and animals, there is strong evidence that the kidney is the main target organ of cadmium toxicity, following extended exposure (USPHS 1997, Elinder and Jarup 1996, Goyer 1996, Roels *et al.* 1993, Iwata *et al.* 1993, WHO 1992, Mueller *et al.* 1992). Renal damage includes tubular proteinuria (the excretion of low molecular weight proteins) and a decrease in the glomerular filtration rate. The latter results in a depressed re-sorption of enzymes, amino acids, glucose, calcium, copper, and inorganic phosphate. Furthermore, studies have shown that even when cadmium exposure ceases, proteinuria does not decrease, and renal tubular dysfunction and reduced glomerular filtration increase in severity (USPHS 1997, Elinder and Jarup 1996, Goyer 1996, Iwata *et al.* 1993, WHO 1992, Nriagu 1988).

Other toxic effects of cadmium, based on findings from occupation, animal, and epidemiological studies, can be summarised as follows:

Case studies indicate that calcium deficiency, osteoporosis, or osteomalacia (softening of the bones) can develop in some workers after long-term occupational exposure to high levels of cadmium. A progressive disturbance in the renal metabolism of vitamin D and an increased urinary excretion of calcium is often seen, suggesting that bone changes may be secondary to disruption in kidney vitamin D and calcium metabolism (USPHS 1997, Goyer *et al.* 1994, WHO 1992). In the Jinzu River Basin, a cadmium-contaminated area in Japan, a cadmium induced skeletal disorder known as Itai-Itai disease disabled many children born to women of middle age and poor nutrition (Alloway 1996).

The inhalation of high levels of cadmium oxide fumes or dust is intensely irritating to respiratory tissue, and acute high-level exposures can be fatal. Typical non-fatal symptoms can include severe tracheobronchitis, pneumonitis, and pulmonary oedema, which can develop within hours of exposure (USPHS 1997b, Goyer 1996, WHO 1992). At lower levels, lung inflammation have been known to cause emphysema (swelling of the lung air sacs resulting in breathlessness) and dyspnoea (difficult and laboured breathing) (USPHS 1997, Goyer 1996, WHO 1992). Animal studies have confirmed that inhalation exposure to cadmium leads to respiratory injury (USPHS 1997b, WHO 1992).

There have been a number of epidemiological studies intended to determine a relationship between occupational (respiratory) exposure to cadmium and lung and prostatic cancer, and these along with animal studies have provided considerable support for the carcinogenic potential of cadmium (Goyer 1996). Cadmium and certain cadmium compounds are therefore listed by the International Agency for Research on Cancer (IARC) as carcinogenic. The US Department of Health and Human Services in its 8th Report on Carcinogens, lists cadmium and certain cadmium

compounds as Reasonably Anticipated to be Human Carcinogens (USPHS 1998).

In addition to these toxic effects, it has also been suggested that cadmium may play a role in the development of hypertension (high blood pressure) and heart disease (USPHS 1997, Goyer 1996, Elinder and Jarup 1996). It is also known that severe oral exposure can result in severe irritation to the gastrointestinal epithelium, nausea, vomiting, salivation, abdominal pain, cramps and diarrhoea (USPHS 1997b).

3.3 Mercury

Mercury is an extremely toxic, non-essential trace metal, having no biochemical or nutritional function. Biological mechanisms for its removal are poor, and, mercury is the only metal known to biomagnify, that is, progressively accumulate as it passes though the food chain (WHO 1989).

Acute inhalation of high levels of mercury vapour may cause nausea, vomiting, diarrhoea, increases in blood pressure or heart rate, skin rashes, eye irritation, corrosive bronchitis and pneumonitis. And, if not fatal, may be associated with central nervous system (CNS) effects such as tremor or increased excitability (USPHS 1997, Goyer 1996). With chronic exposure, the major effects are on the CNS (tremor, spasms, loss of memory, increased excitability, severe depression, personality changes, even delirium and hallucination), although renal damage, associated with chronically exposed workers, has also been shown (Ratcliffe et al. 1996, Goyer 1996). These effects have also been reported in animal studies (USPHS 1997)

Acute exposure to high levels of mercury salts, or chronic low-dose exposure, is directly toxic to the kidney (Zalups and Lash 1994). In addition, nausea and diarrhoea may result after swallowing large amounts of inorganic mercury salts, and some nervous system effects have also been recorded (USPHS 1997, WHO 1989).

Once metallic mercury has entered the environment it can be methylated by micro-organisms, found for instance in aquatic sediments, to organic forms of mercury, most commonly methylmercury. In this form, it is able to cross cell membranes easily and quickly enters the aquatic food chain. From here it may enter the human food chain. Exposure to methylmercury has resulted in permanent damage to the CNS, kidneys, and the developing foetus. The levels of methylmercury that result in these effects are not usually encountered by the general population, however they were encountered by the population of Minamata, in Japan, who were exposed to high levels of methylmercury from eating contaminated fish and seafood collected from the Bay (USPHS 1997). Symptoms such as brain damage, numbness of extremities, and paralysis, along with the loss of hearing, speech and sight were reported (D'Itri 1991). However even today, the full range of neurological symptoms caused by the ingestion of methylmercury in fish and shellfish has not been fully characterised, and the total number of Minamata Disease sufferers has not been determined (D'Itri 1991). Furthermore, whilst only the Japanese cases have been confirmed as Minamata Disease, other populations in Canada (from chlor-alkali discharges) and Brazil (from gold mining) are potentially at risk. The problem of methylation of past and present inorganic mercury discharges continues, and the long retention time of mercury by sediments delays the elimination of contamination for many years (Harada 1997, Akagi *et al.* 1995, Bryan and Langston 1992, D'Itri 1991).

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APPENDIX B

INDIVIDUAL COMPOUNDS IDENTIFIED IN THE EMISSIONS OF A MUNICIPAL WASTE INCINERATION PLANT

pentane
trichlorofluoromethane
acetonitrile
acetone
iodomethane
dichloromethane
2-methyl-2-propanol
2-methylpentane
chloroform
ethyl acetate
2,2-dimethyl-3-pentanol
cyclohexane
benzene
2-methylhexane
3-methylhexane
1,3-dimethylcyclopentane
1,2-dimethylcyclopentane
trichloroethene
heptane
methylcyclohexane
ethylcyclopentane
2-hexanone
toluene
1,2-dimethylcyclohexane
2-methylpropyl acetate
3-methyleneheptane
paraldehyde
octane
tetrachloroethylene
butanoic acid ethyl ester
butyl acetate
ethylcyclohexane
2-methyloctane
dimethyldioxane
2-furanecarboxaldehyde
chlorobenzene
methyl hexanol
trimethylcyclohexane
ethyl
benzene
formic acid
xylene
acetic acid
aliphatic carbonyl
ethylmethylcyclohexane
2-heptanone
2-butoxyethanol
nonane
isopropyl benzene
propylcyclohexane
dimethyloctane
pentanecarboxylic acid
propyl benzene
benzaldehyde
5-methyl-2-furane carboxaldehyde
1-ethyl-2-methylbenzene
1,3,5-trimethylbenzene
trimethylbenzene
benzonitrile
methylpropylcyclohexane
2-chlorophenol
1,2,4-trimethylbenzene
phenol
1,3-dichlorobenzene
1,4-dichlorobenzene
decane
hexanecarboxylic acid
1-ethyl-4-methylbenzene
2-methylisopropylbenzene
benzyl alcohol
trimethylbenzene
1-methyl-3-propylbenzene
2-ethyl-1,4-dimethylbenzene
2-methylbenzaldehyde
1-methyl-2-propylbenzene
methyl decane
4-methylbenzaldehyde
1-ethyl-3,5-dimethylbenzene
1-methyl-(1-pro-penyl)benzene
bromochlorobenzene
4-methylphenol
benzoic acid methyl ester
2-chloro-6-methylphenol
ethyldimethylbenzene
undecane
heptanecarboxylic acid
1-(chloromethyl)-4-methylbenzene
1,3-diethylbenzene
1,2,3-trichlorobenzene
4-methylbenzyl
alcohol
ethylhex anoic acid
ethyl benzaldehyde
2,4-dichlorophenol
1,2,4-trichlorobenzene
naphthalene
cyclopentasiloxanedecamethyl
methyl acetophenone
ethanol-1-(2-butoxyethoxy)
4-chlorophenol
benzothiazole
benzoic acid
octanoic acid
2-bromo-4-chlorophenol
1,2,5-trichlorobenzene
dodecane
bromochlorophenol
2,4-dichloro-6-methylphenol
dichloromethylphenol
hydroxybenzonitrile

tetrachlorobenzene
methylbenzoic acid
trichlorophenol
2-(hydroxymethyl) benzoic acid
2-ethylnaphthalene-1,2,3,4-tetrahydro 2,4,6-trichlorophenol
4-ethylacetophenone
2,3,5-trichlorophenol
4-chlorobenzoic acid
2,3,4-trichlorophenol
1,2,3,5-tetrachlorobenzene
1,1'biphenyl (2-ethenyl-naphthalene)
3,4,5-trichlorophenol
chlorobenzoic acid
2-hydroxy-3,5-dichlorobenzaldehyde
2-methylbiphenyl
2-nitrostyrene(2-nitroethenylbenzene)
decanecarboxylic acid
hydroxymethoxybenzaldehyde
hydroxychloroacetophenone
ethylbenzoic acid
2,6-dichloro-4-nitrophenol
sulphonic acid
m.w. 192
4-bromo-2,5-dichlorophenol
2-ethylbiphenyl
bromodichlorophenol
1(3H)-isobenzofuranone-5-methyl
dimethylphthalate
2,6-di-tertiary-butyl-p-benzoquinone
3,4,6-trichloro-1-methyl-phenol
2-tertiary-butyl-4-methoxyphenol
2,2'-dimethylbiphenyl
2,3'-dimethylbiphenyl
pentachlorobenzene
bibenzyl
2,4'-dimethylbiphenyl
1-methyl-2-phenylmethylbenzene
benzoic acid phenyl ester
2,3,4,6-tetrachlorophenol
tetrachlorobenzofurane
fluorene
phthalic ester
dodecanecarboxylic acid
3,3'-dimethylbiphenyl
3,4'-dimethylbiphenyl
hexadecane
benzophenone
tridecanoic acid
hexachlorobenzene
heptadecane
fluorenone
dibenzothiophene
pentachlorophenol
sulphonic acid m.w. 224
phenanthrene
tetradecanecarboxylic acid
octadecane
phthelic ester
tetradecanoic acid isopropyl ester
caffeine
12-methyltetradecacarboxylic acid
pentadecacarboxylic acid
methylphenanthrene
nonedecane
9-hexadecene carboxylic acid
anthraquinone
dibutylphthalate
hexadecanoic acid
eicosane
methylhexadecanoic acid
fluoroanthene
pentachlorobiphenyl
heptadecanecarboxylic acid
octadecadienal
pentachlorobiphenyl
aliphatic amide
octadecanecarboxylic acid
hexadecane amide
docosane
hexachlorobiphenyl
benzylbutylphthalate
aliphatic amide
diisooctylphthalate
hexadecanoic acid hexadecyl ester
cholesterol.

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Cancer mortality in towns in the vicinity of incinerators and installations for the recovery or disposal of hazardous waste

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ABSTRACT

Background: Waste treatment plants release toxic emissions into the environment which affect neighboring towns.
Objectives: To investigate whether there might be excess cancer mortality in towns situated in the vicinity of Spanish-based incinerators and installations for the recovery or disposal of hazardous waste, according to the different categories of industrial activity.
Methods: An ecologic study was designed to examine municipal mortality due to 33 types of cancer, across the period 1997–2006. Population exposure to pollution was estimated on the basis of distance from town of residence to pollution source. Using Besag–York–Mollié (BYM) regression models with Integrated Nested Laplace approximations for Bayesian inference, and Mixed Poisson regression models, we assessed the risk of dying from cancer in a 5-kilometer zone around installations, analyzed the effect of category of industrial activity, and conducted individual analyses within a 50-kilometer radius of each installation.
Results: Excess cancer mortality (BYM model: relative risk, 95% credible interval) was detected in the total population residing in the vicinity of these installations as a whole (1.06, 1.04–1.09), and, principally, in the vicinity of incinerators (1.09, 1.01–1.18) and scrap metal/end-of-life vehicle handling facilities, in particular (1.04, 1.00–1.09). Special mention should be made of the results for tumors of the pleura (1.71, 1.34–2.14), stomach (1.18, 1.10–1.27), liver (1.18, 1.06–1.30), kidney (1.14, 1.04–1.23), ovary (1.14, 1.05–1.23), lung (1.10, 1.05–1.15), leukemia (1.10, 1.03–1.17), colon–rectum (1.08, 1.03–1.13) and bladder (1.08, 1.01–1.16) in the vicinity of all such installations.
Conclusions: Our results support the hypothesis of a statistically significant increase in the risk of dying from cancer in towns near incinerators and installations for the recovery or disposal of hazardous waste.

1. Introduction

Generation of waste by human activity is a matter of worldwide concern. Municipal incinerators and installations for the recovery or disposal of hazardous waste help address this problem but inevitably generate and release toxic emissions and effluents, such as dioxins – carcinogens recognized by the International Agency for Research on Cancer (IARC) (IARC, 1997) – into the environment, which then affect neighboring towns,.

Some studies have linked exposure to incinerator emissions, with adverse reproductive outcomes (Dummer et al., 2003), respiratory problems (Miyake et al., 2005) and cancer (Comba et al., 2003; Knox, 2000; Viel et al., 2008). With respect to treatment (elimination, disposal or recovery) of hazardous waste, which includes activities such as the recycling of scrap metal and end-of life vehicles (ELVs), re-refining of used oil, and physico/chemical treatment of waste, there are hardly any epidemiologic studies on these installations' health effects on the populations of nearby towns, even though they are known to release carcinogens, such as dioxins, arsenic, benzene, cadmium and chromium (Environmental Protection Agency, 2002; Landrigan et al., 1989). Accordingly, it would seem appropriate to ascertain whether residential proximity to these little-studied types of pollutant facilities might have an influence on the frequency of cancer.

Abbreviations: IARC, Agency for Research on Cancer; ELVs, End-of life vehicles; IPPC, Integrated Pollution Prevention and Control; E-PRTR, European Pollutant Release and Transfer Register; NSI, National Statistics Institute; PCBs, Polychlorinated biphenyls; RRs, Relative risks; 95% CrIs/CIs, 95% credible/confidence intervals; BYM, Besag, York and Mollié; INLAs, Integrated nested Laplace approximations; PAHs, Polycyclic aromatic hydrocarbons; NHL, Non-Hodgkin's lymphoma.

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In the case of pollution sources in Spain, the European Commission directives passed in 2002 afforded a new means of studying the consequences of industrial pollution: Integrated Pollution Prevention and Control (IPPC), governed both by Directive 96/61/CE (recently codified into Directive 2008/1/EC) and by Act 16/2002, which incorporates this Directive into the Spanish legal system, lays down that, to be able operate, industries covered by the regulation must obtain the Integrated Environmental Permit. This same enactment implemented the European Pollutant Release and Transfer Register (E-PRTR) in 2007, which makes it compulsory to declare all pollutant emissions to air, water and soil, that exceed the designated thresholds, and contains detailed information about the address and type of industrial activity in which the installations are involved. IPPC and E-PRTR records thus constitute an inventory of geo-located industries with environmental impact in Europe, which is a valuable resource for monitoring industrial pollution and, by extension, renders it possible for the association between residential proximity to such pollutant installations and health impacts, such as cancer, to be studied (Garcia-Perez et al., 2012; Lopez-Abente et al., 2012; Lopez-Cima et al., 2011).

In this context, this study sought to: (1) assess possible excess mortality attributable to 33 tumor sites among the Spanish population residing in the environs of incinerators and hazardous waste treatment plants governed by the IPPC Directive and E-PRTR Regulation; (2) analyze this risk according to the different categories of industrial activity, and for each installation individually; and, (3) perform the analysis for the population, both overall and broken down by sex, using different statistical approaches for the purpose.

2. Materials and methods

We designed an ecologic study to evaluate the association between cancer mortality and proximity to incinerators and hazardous waste treatment plants at a municipal level (8098 Spanish towns), during the period 1997–2006. Separate analyses were performed for the overall population and for each sex.

2.1. Mortality data

Observed municipal mortality data were drawn from the records of the National Statistics Institute (NSI) for the study period, and corresponded to deaths due to 33 types of malignant tumors (see Supplementary data, Table 1, which shows the list of tumors analyzed and their codes as per the International Classification of Diseases—9th and 10th Revisions). Expected cases were calculated by taking the specific rates for Spain as a whole, broken down by age group (18 groups: 0–4, …, 80–84 years, and 85 years and over), sex, and five-year period (1997–2001, 2002–2006), and multiplying these by the person-years for each town, broken down by the same strata. Person-years for each quinquennium were calculated by multiplying the respective populations by 5 (with data corresponding to 1999 and 2004 being taken as the estimator of the population at the midpoint of the study period). In addition, we specifically analyzed leukemias and brain cancer in subjects under ages 15 and 25 years, since these were the most frequent tumors in adolescents and young adults in our data.

2.2. Industrial pollution exposure data

Population exposure to industrial pollution was estimated by taking the distance from the centroid of town of residence to the industrial facility. We used the industrial database (industries governed by IPPC and facilities pertaining to industrial activities not subject to IPPC but included in the E-PRTR) provided by the Spanish Ministry for Agriculture, Food & Environment in 2007. Bearing in mind the minimum induction periods for the tumors targeted for study, generally 10 years for solid tumors and 1 year for leukemias (United Nations Scientific Committee on the Effects of Atomic Radiation, 2006), two industry databases were used:

a) for the study of leukemias, we selected the 129 installations corresponding to IPPC categories 5.1 (installations for the recovery or disposal of hazardous waste with a capacity exceeding 10 t per day) and 5.2 (installations for the incineration of municipal waste with a capacity exceeding 3 t per hour), which came into operation prior to 2002 (1 year before the mid-year of the study period), denominated "pre-2002 installations"; and,
b) for the remaining tumors, we selected the 67 installations corresponding to IPPC categories 5.1 and 5.2 which came into operation prior to 1993 (10 years before the mid-year of the study period), denominated "pre-1993 installations".

The date (year) of commencement of the respective industrial activities was provided by the industries themselves.

Each of the installations was classified into one of the following 9 categories of industrial activities, according to the type of waste involved and treatment applied:

1. "*Incineration*": incineration of solid urban (municipal) and special waste (9 pre-2002 and 5 pre-1993 installations);
2. "*Scrap metal + ELVs*": scrapping/decontamination of ELVs, and recycling of scrap metal (ferrous and non-ferrous products) and electric/electronic equipment (32 pre-2002 and 23 pre-1993 installations);
3. "*Oils + Oily waste*": treatment of used oil, oily marine pollutant (MARPOL) waste and decontamination of equipment contaminated by polychlorinated biphenyls (PCBs) (24 pre-2002 and 8 pre-1993 installations);
4. "*Packaging*": recycling of metallic and plastic industrial packaging (9 pre-2002 and 5 pre-1993 installations);
5. "*Solvents*": recovery of used solvents (7 pre-2002 and 5 pre-1993 installations);
6. "*Spent baths*": regeneration of spent acid pickling and basic baths and hydrochloric acid used in metal descaling (7 pre-2002 and 5 pre-1993 installations);
7. "*Physico/chemical treatment*": physico/chemical treatment of waste not included in the above sections (8 pre-2002 and 4 pre-1993 installations);
8. "*Industrial waste*": treatment of industrial waste not included in the above sections, such as recovery of wastes from the iron and steel industry (15 pre-2002 and 7 pre-1993 installations); and,
9. "*Wastes not otherwise specified*": treatment of waste not included in any of the above sections, such as medical wastes, lead acid batteries, photochemical wastes, or textile wastes (18 pre-2002 and 5 pre-1993 installations). This category also included installations that treated different types of waste or applied several different treatment processes.

Owing to the presence of errors in the initial location of industries, the geographic coordinates of the industrial locations recorded in the IPPC + E-PRTR 2007 database were previously validated: every single address was thoroughly checked using Google Earth (with the street-view application), the Spanish Agricultural Plots Geographic Information System (which includes orthophotos and topographic maps showing the names of the industries) (Ministerio de Agricultura Alimentación y Medio Ambiente, 2012), the Google Maps server and the "Yellow pages" web page (which allow for a search of addresses and companies), and the web pages of the industries themselves, to ensure that location of the industrial facility was exactly where it should be. 25% of the incinerators and hazardous waste treatment installation coordinates were corrected at a distance of 4471 m or more from the original location in the IPPC + E-PRTR database.

2.3. Statistical analysis

Three types of analysis were performed to assess possible excess cancer mortality in towns lying near ("near") versus those lying far ("far") from incinerators and hazardous waste treatment installations, known as a "near vs. far" analysis. In all cases, a distance of 5 km was taken as the area of proximity ("exposure") to industrial installations, in line with the distance used by other studies on these types of installations (Federico et al., 2010; Knox, 2000; Leem et al., 2006):

1) in a first phase, we conducted a "near vs. far" analysis to estimate the relative risks (RRs) of towns situated at a distance of ≤5 km from incinerators and hazardous waste treatment installations as a whole. The variable, "exposure", was coded as: a) exposed or proximity area ("near"), consisting of towns lying at a distance of ≤5 km from any incinerator or hazardous waste treatment facility; b) intermediate area, consisting of towns lying at a distance of ≤5 km from any industrial installation other than incinerators or hazardous waste treatment facilities; and, c) unexposed area ("far"), consisting of towns having no (IPPC + E-PRTR)-registered industry within 5 km of their municipal centroid (reference group);
2) in a second analysis, we decided to stratify risk of analysis anterior according to the different categories of industrial activity. To this end, we created a variable of "exposure" in which the exposed area was stratified into the following groups: Group 1, made up of towns lying close (≤5 km) to one or more installations belonging to the category "Incineration"; Group 2, if the category was "Scrap metal + ELVs", and so on, until Group 9, if the category was "Wastes not otherwise specified"; and Group 10, made up of towns lying close to two or more installations belonging to different categories of activity ("multiple pollutant categories"). Intermediate and unexposed areas were defined as in the preceding phase; and,
3) lastly, bearing in mind that characteristics tend to vary from one incinerator or hazardous waste treatment facility to the next, we conducted separate "near vs. far" analyses of the individual installations, with the analysis being confined to an area of 50 km surrounding each such installation so as to have a local comparison group.

For all the above analyses, we used two statistical approaches based on log-linear models to estimate the RRs and their 95% credible/confidence intervals (95% CrIs/CIs), assuming that the number of deaths per stratum followed a Poisson distribution:

a) a Bayesian conditional autoregressive model proposed by Besag, York and Mollié (BYM) (Besag et al., 1991), with explanatory variables:

$$O_i \sim Poisson(\mu_i), \text{with} \mu_i = E_i\lambda_i$$

$$\log(\lambda_i) = \alpha Expos_i + \sum_j \beta_j Soc_{ij} + h_i + b_i \Rightarrow \log(\mu_i) = \log(E_i) + \alpha Expos_i + \sum_j \beta_j Soc_{ij} + h_i + b_i$$

$$Soc_{ij} = ps_i + ill_i + far_i + unem_i + pph_i + inc_i$$

$$i = 1, \ldots, 8098 \ \ towns, \quad j = 1, \ldots, 6 \ \ potential \ \ confounders$$

$$h_i \sim Normal(\theta, \tau_h)$$
$$b_i \sim Car.Normal(\eta_i, \tau_b)$$
$$\tau_h \sim Gamma(\alpha, \beta)$$
$$\tau_b \sim Gamma(\gamma, \delta)$$

b) a mixed Poisson regression model (Gelman and Hill, 2007):

$$O_i \sim Poisson(\mu_i), \text{with} \mu_i = E_i\lambda_i$$

$$\log(\lambda_i) = \alpha Expos_i + \sum_j \beta_j Soc_{ij} + p_i \Rightarrow \log(\mu_i) = \log(E_i) + \alpha Expos_i + \sum_j \beta_j Soc_{ij} + p_i$$

$$Soc_{ij} = ps_i + ill_i + far_i + unem_i + pph_i + inc_i$$

$$i = 1, \ldots, 8098 \ \ towns, \quad j = 1, \ldots, 6 \ \ potential \ \ confounders$$

with λ_i being the RR in town i, the number of observed deaths in town i for each cancer site (O_i) being the dependent variable, and the number of expected deaths in town i for each cancer site (E_i) being the offset, in both cases. All estimates for the variable of "exposure" ($Expos_i$) were adjusted for the following standardized, sociodemographic indicators (Soc_{ij}), chosen as potential confounders directly from the 1991 census for their availability at a municipal level and potential explanatory ability vis-à-vis certain geographic mortality patterns (Lopez-Abente et al., 2006): population size (ps_i) (categorized into three levels: 0–2000, 2000–10,000 and ≥10,000 inhabitants); percentage illiteracy (ill_i), farmers (far_i) and unemployed ($unem_i$); average persons per household (pph_i); and mean income (inc_i) by the Spanish Market Yearbook, as a measure of income level (Ayuso Orejana et al., 1993). Their geographic patterns show the economic, demographic and social development of Spain, appreciating some spatial correspondence between illiteracy, unemployment and younger population areas. The variable of "exposure" and potential confounding covariates were fixed-effects terms in the models.

To enable the spatial autocorrelation problem (presence of geographic patterns in contiguous spatial data) to be assessed, this was estimated by applying Moran's I statistic to the Standardized Mortality Ratios (Bivand et al., 2008). The BYM Bayesian autoregressive model takes this problem into account, thanks to the inclusion of two random effects components, namely: a spatial term containing municipal contiguities (b_i); and the municipal heterogeneity term (h_i). Integrated nested Laplace approximations (INLAs) (Rue et al., 2009) were used as a tool for Bayesian inference. For this purpose, we used R-INLA (The R-INLA project, 2012), with the option of simplified Laplace estimation of the parameters. A total of 8098 towns were included, and the spatial data on municipal contiguities were obtained by processing the official NSI maps.

Furthermore, the mixed Poisson regression model includes province as a random effects term (p_i), to enable geographic variability and extra-Poisson dispersion to be taken into account and unexposed towns belonging to the same province to be considered as the reference group in each case, something that is justified by the geographic differences observed in mortality attributable to some tumors (Lopez-Abente et al., 2006).

Lastly, a residual analysis (based on deviance residuals) was performed to test the models.

3. Results

Fig. 1 depicts the geographic distribution of the 129 installations studied according to the different categories of industrial activity, together with their PRTR codes and year of commencement of operations. Supplementary data, Table 2 gives a detailed description of the type of activity undertaken by each installation and the pollutants emitted during the preceding decade. In all, the 129 installations released 525,428 t of toxic substances to air and 4984 t to water in 2007, including carcinogens such as arsenic (32 kg to air and 33 kg to water), chromium (81 kg to air and 80 kg to water) and polycyclic aromatic hydrocarbons (PAHs) (48 kg to air and 126 kg to water). More detailed information on emission amounts is provided in Supplementary data, Tables 3 and 4, which show the types of substances and amounts released by these installations to air and water, respectively.

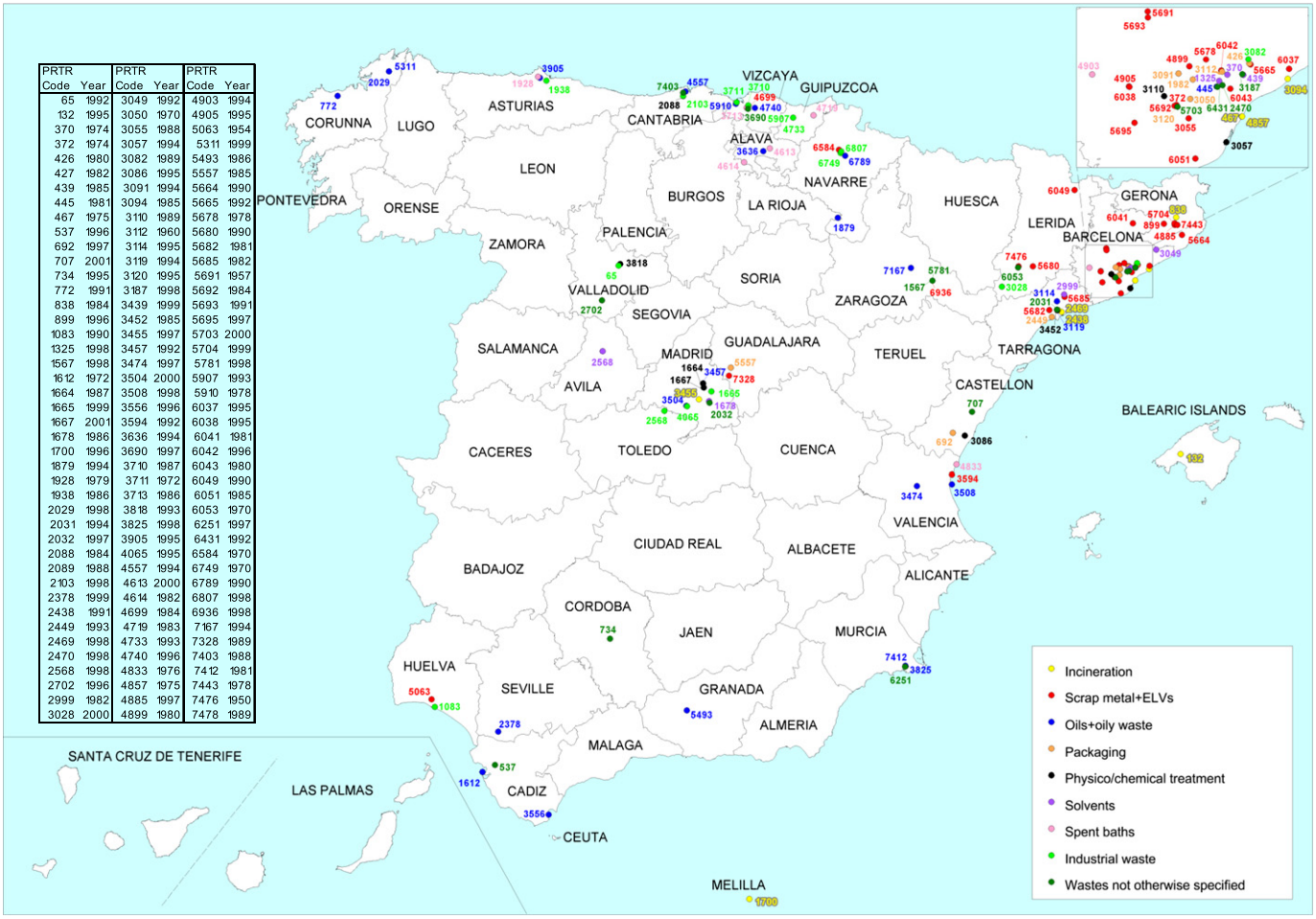

Fig. 1. Geographic distribution of Spanish-based incinerators and hazardous waste treatment installations.

Table 1 shows the RRs and 95% CrIs/CIs for cancers proving to be statistically significant in towns situated at ≤5 km from incinerators and hazardous waste treatment installations, estimated using BYM and Poisson mixed regression models and Moran's I test for spatial autocorrelation. Overall, excess cancer mortality was present in both sexes, with the two models displaying identical RRs, which were higher in men (RR = 1.08) than in women (RR = 1.03). In the case of specific tumors, the estimates yielded by both models were largely similar in general (slightly higher and significant in the mixed model in tumors of the oral cavity and pharynx, esophagus and non-Hodgkin's lymphoma (NHL), and somewhat higher in the BYM model in renal cancer). Some cancers – such as all cancers combined (in men and women) or malignant tumors of the stomach (in men and women) and lung, bladder, oral cavity and pharynx, colon–rectum, and liver (in men) – displayed a statistically significant spatial autocorrelation, and it thus seemed appropriate to use the BYM model in order to take this spatial autocorrelation into account. Based on this model, statistically significant RRs appeared for tumors of the stomach, liver, pleura and kidney (in men and women), colon–rectum, lung, bladder, gallbladder and leukemia (in men), and brain and ovary (in women). In these results, note should be taken of the high excess risk for cancer of the pleura (RR = 1.84 in men and RR = 1.52 in women). With respect to leukemias and brain cancer in the under-15- and under-25 age groups, statistically significant excess risks were not in evidence (see Supplementary data, Table 5, which shows the RR of dying from leukemia and brain cancer among the under-15 and under-25 age groups in towns situated at ≤5 km from incinerators and hazardous waste treatment installations, estimated using BYM models).

The analyses of the above table, including the two regression models and spatial autocorrelation test, were performed separately for each tumor (see Supplementary data, Tables 6 and 7, which show the RR of dying from cancer in towns situated at ≤5 km from incinerators and hazardous waste treatment installations as a whole – estimated using BYM models – and Moran's I *p*-values for spatial autocorrelation analyses, respectively). In the residual analysis of the BYM model for all tumors under study, the graphs plotting deviance residuals against distance to the nearest installation displayed an apparently random scatter pattern, consistent with a well-fitted model (see Supplementary data, Fig. 1).

Table 2 shows the RRs and 95% CrIs estimated with BYM models for cancers that yielded statistically significant results in the analysis of risk stratified by category of industrial activity. For all cancers combined, statistically significant excess risks were observed in the environs of multiple pollutant categories (men and women), incinerators and installations for the recycling of scrap metal + ELVs (total population), and installations for the regeneration of spent baths (men), though in no case were these higher than 10%. Insofar as the remaining tumors were concerned, attention should be drawn to the significant excess risks found for the following (we have highlighted the highest statistically significant RRs for each tumor): stomach and colorectal cancers in men, in the vicinity of packaging recycling industries (RRs = 1.53 and 1.29, respectively); cancers of the liver and ovary in women, in

Table 1
Relative risk of dying from cancers with significant results in towns situated at ≤5 km from incinerators and hazardous waste treatment installations as a whole, estimated using BYM and Poisson mixed regression models, and Moran's I test for spatial autocorrelation. Significant results are in bold.

	T[a]	Obs[b]	Exp[c]	BYM model		Mixed model		Moran's I test
				RR[d]	95%CrI[e]	RR[d]	95%CI[f]	*p*-Value
All cancers[g]								
Total	163	91,708	85,109.6	**1.06**	**1.04–1.09**	**1.06**	**1.05–1.07**	**0.0001**
Men	163	58,275	53,071.8	**1.08**	**1.05–1.11**	**1.08**	**1.07–1.10**	**0.0001**
Women	163	33,433	32,037.8	**1.03**	**1.01–1.06**	**1.03**	**1.01–1.04**	**0.0006**
Oral and pharyngeal cancer								
Total	163	2482	2178.7	1.04	0.95–1.14	**1.11**	**1.05–1.19**	**0.0039**
Men	163	2056	1804.5	1.03	0.94–1.13	**1.11**	**1.04–1.19**	**0.0031**
Women	163	426	374.2	1.09	0.94–1.26	1.07	0.93–1.24	0.4660
Esophageal cancer								
Total	163	1960	1733.3	0.99	0.90–1.09	**1.07**	**1.00–1.15**	0.0725
Men	163	1710	1504.0	1.01	0.91–1.11	**1.08**	**1.00–1.16**	0.0979
Women	163	250	229.4	0.92	0.74–1.13	1.02	0.84–1.24	0.7441
Stomach cancer								
Total	163	6123	5646.0	**1.18**	**1.10–1.27**	**1.07**	**1.03–1.11**	**0.0001**
Men	163	3822	3461.8	**1.18**	**1.09–1.28**	**1.09**	**1.04–1.15**	**0.0073**
Women	163	2301	2184.3	**1.16**	**1.06–1.27**	1.04	0.98–1.11	**0.0049**
Colorectal cancer								
Total	163	12,265	11367.2	**1.08**	**1.03–1.13**	**1.06**	**1.03–1.09**	**0.0004**
Men	163	7084	6343.6	**1.12**	**1.06–1.18**	**1.08**	**1.04–1.12**	**0.0131**
Women	163	5181	5023.6	1.04	0.98–1.10	1.03	0.99–1.08	0.6319
Liver cancer								
Total	163	2929	2310.4	**1.18**	**1.06–1.30**	**1.23**	**1.15–1.31**	**0.0012**
Men	163	2075	1678.6	**1.17**	**1.05–1.30**	**1.22**	**1.13–1.31**	**0.0014**
Women	163	854	631.8	**1.20**	**1.02–1.40**	**1.24**	**1.10–1.40**	0.8100
Gallbladder cancer								
Total	163	1339	1262.6	1.10	0.99–1.21	**1.10**	**1.01–1.19**	0.2574
Men	163	511	432.5	**1.26**	**1.08–1.45**	**1.23**	**1.07–1.41**	0.5436
Women	163	828	830.1	1.02	0.90–1.15	1.04	0.94–1.15	0.6723
Lung cancer								
Total	163	19,214	17,394.4	**1.10**	**1.05–1.15**	**1.10**	**1.07–1.12**	**0.0001**
Men	163	17,156	15,336.5	**1.12**	**1.06–1.18**	**1.12**	**1.10–1.15**	**0.0001**
Women	163	2058	2057.8	0.92	0.84–1.00	0.91	0.85–0.97	0.9473
Pleural cancer								
Total	163	394	206.8	**1.71**	**1.34–2.14**	**1.74**	**1.44–2.11**	0.1093
Men	163	284	147.0	**1.84**	**1.39–2.40**	**1.86**	**1.48–2.34**	0.0688
Women	163	110	59.7	**1.52**	**1.04–2.14**	**1.51**	**1.07–2.14**	0.8281
Skin cancer								
Total	163	354	424.0	1.11	0.93–1.31	1.10	0.94–1.27	0.3792
Men	163	209	226.5	1.23	0.99–1.50	**1.26**	**1.03–1.53**	0.4815
Women	163	145	197.5	0.97	0.75–1.23	0.88	0.70–1.10	0.2312
Ovarian cancer								
Women	163	1852	1770.0	**1.14**	**1.05–1.23**	**1.12**	**1.05–1.21**	0.8134
Bladder cancer								
Total	163	4131	3809.9	**1.08**	**1.01–1.16**	**1.07**	**1.02–1.12**	**0.0140**
Men	163	3419	3138.4	**1.10**	**1.02–1.18**	**1.09**	**1.03–1.14**	**0.0092**
Women	163	712	671.5	1.02	0.91–1.15	1.02	0.91–1.13	0.7499
Renal cancer								
Total	163	1918	1651.3	**1.14**	**1.04–1.23**	**1.07**	**1.00–1.15**	0.6497
Men	163	1268	1094.0	**1.12**	**1.02–1.24**	1.07	0.98–1.17	0.4631
Women	163	650	557.4	**1.16**	**1.02–1.31**	1.11	0.99–1.26	0.9937
Brain cancer								
Total	163	2380	2245.9	1.04	0.97–1.12	1.03	0.97–1.10	0.9354
Men	163	1285	1248.8	1.00	0.91–1.09	1.00	0.92–1.08	0.1687
Women	163	1095	997.0	**1.11**	**1.00–1.22**	**1.10**	**1.00–1.20**	0.2573
Non-Hodgkin's lymphoma								
Total	163	2396	2240.2	1.02	0.94–1.11	**1.09**	**1.02–1.16**	0.3802
Men	163	1274	1171.1	1.07	0.97–1.19	**1.12**	**1.03–1.22**	0.7342
Women	163	1122	1069.1	0.96	0.87–1.07	1.03	0.94–1.13	0.1000
Leukemia								
Total	237	5378	4947.1	**1.10**	**1.03–1.17**	**1.06**	**1.01–1.11**	0.6310
Men	237	2956	2713.8	**1.12**	**1.04–1.21**	**1.09**	**1.02–1.16**	0.1279
Women	237	2422	2233.4	1.07	0.98–1.17	1.04	0.97–1.20	0.2602

[a] Number of towns situated at ≤5 km from incinerators and hazardous waste treatment installations as a whole.
[b] Observed deaths.
[c] Expected deaths.
[d] RRs adjusted for population size, percentage illiteracy, farmers and unemployed persons, average persons per household, and mean income.
[e] 95% credible interval.
[f] 95% confidence interval.
[g] Sum of the 33 types of cancer analyzed.

areas surrounding installations for the regeneration of spent baths (RRs = 1.55 and 1.29, respectively); cancers of the gallbladder, lung and pleura in men living near incinerators (RRs = 1.43, 1.19 and 1.98, respectively); skin cancer in men, in the vicinity of solvent treatment installations (RR = 3.30); Hodgkin's lymphoma and kidney cancer in men, in the areas around physico/chemical treatment installations

Table 2
Relative risk of dying from cancers with significant results in towns situated at a distance of 5 km or less from incinerators and hazardous waste treatment installations as a whole, estimated using BYM models and shown with a breakdown by category of industrial activity. Significant results are in bold.

	T[a]	Total			Men			Women		
		Obs[b]	RR[c]	95%CrI[d]	Obs	RR[c]	95%CrI[d]	Obs	RR[c]	95%CrI[d]
All cancers[e]										
Incineration	12	13,051	**1.09**	**1.01–1.18**	8385	1.09	0.99–1.19	4666	1.06	0.98–1.14
Scrap metal + ELVs	52	11,981	**1.04**	**1.00–1.09**	7668	**1.06**	**1.00–1.12**	4313	1.03	0.98–1.08
Oil + oily waste	7	8277	1.08	0.99–1.18	5214	1.09	0.99–1.21	3063	1.07	0.98–1.16
Packaging	2	2471	1.09	0.97–1.22	1591	1.13	0.98–1.29	880	1.02	0.91–1.14
Solvents	6	1108	0.97	0.87–1.08	693	0.98	0.87–1.11	415	0.95	0.84–1.08
Spent baths	15	12412	1.06	0.98–1.14	7833	**1.09**	**1.00–1.18**	4579	1.03	0.95–1.11
Physico/chemical treatment	5	369	1.11	0.97–1.26	230	1.08	0.92–1.27	139	1.15	0.95–1.37
Industrial waste	7	8261	1.07	0.98–1.17	5166	1.09	0.99–1.21	3095	1.01	0.92–1.11
Wastes not otherwise specified	1	144	0.98	0.74–1.26	93	0.99	0.71–1.33	51	0.98	0.70–1.31
Multiple pollutant categories	56	33,634	**1.08**	**1.04–1.13**	21402	**1.10**	**1.05–1.15**	12232	**1.04**	**1.00–1.09**
Stomach cancer										
Incineration	12	801	1.21	0.98–1.47	492	1.11	0.89–1.36	309	**1.38**	**1.09–1.72**
Scrap metal + ELVs	52	794	**1.14**	**1.00–1.29**	508	**1.17**	**1.01–1.34**	286	1.11	0.94–1.31
Oil + oily waste	7	522	1.22	0.97–1.51	326	**1.30**	**1.01–1.64**	196	1.10	0.83–1.42
Packaging	2	193	**1.38**	**1.02–1.82**	134	**1.53**	**1.20–2.04**	59	1.08	0.76–1.49
Solvents	6	76	1.10	0.79–1.47	50	1.15	0.79–1.60	26	1.01	0.63–1.50
Spent baths	15	842	**1.23**	**1.00–1.48**	523	1.20	0.97–1.48	319	1.20	0.96–1.49
Physico/chemical treatment	5	17	0.90	0.50–1.41	15	1.24	0.67–1.99	2	0.35	0.06–0.94
Industrial waste	7	700	**1.33**	**1.05–1.67**	407	1.22	0.94–1.55	293	**1.33**	**1.03–1.68**
Wastes not otherwise specified	1	10	1.25	0.52–2.41	7	1.47	0.53–3.01	3	1.01	0.22–2.51
Multiple pollutant categories	56	2168	**1.17**	**1.05–1.29**	1360	**1.14**	**1.01–1.28**	808	**1.17**	**1.03–1.33**
Colorectal cancer										
Incineration	12	1645	1.07	0.95–1.20	933	1.08	0.94–1.24	712	1.04	0.91–1.18
Scrap metal + ELVs	52	1583	1.05	0.97–1.14	894	1.09	0.98–1.19	689	1.04	0.93–1.14
Oil + oily waste	7	1072	1.09	0.95–1.25	576	1.09	0.92–1.27	496	1.12	0.95–1.31
Packaging	2	347	1.16	0.97–1.37	215	**1.29**	**1.05–1.55**	132	0.99	0.79–1.22
Solvents	6	148	1.05	0.85–1.27	85	1.10	0.85–1.39	63	0.99	0.74–1.28
Spent baths	15	1763	1.11	0.99–1.25	1045	**1.20**	**1.05–1.37**	718	1.04	0.90–1.20
Physico/chemical treatment	5	43	1.03	0.73–1.37	20	0.84	0.51–1.26	23	1.31	0.82–1.91
Industrial waste	7	1201	1.11	0.96–1.28	710	1.15	0.97–1.34	491	1.05	0.88–1.23
Wastes not otherwise specified	1	15	0.90	0.48–1.49	9	0.93	0.41–1.69	6	0.92	0.34–1.81
Multiple pollutant categories	56	4448	**1.09**	**1.02–1.16**	2597	**1.13**	**1.04–1.21**	1851	1.03	0.95–1.12
Liver cancer										
Incineration	12	521	1.26	0.96–1.63	375	1.28	0.97–1.66	146	1.28	0.87–1.81
Scrap metal + ELVs	52	364	1.08	0.90–1.29	273	1.13	0.92–1.36	91	0.97	0.71–1.29
Oil + oily waste	7	290	1.19	0.85–1.60	181	1.14	0.80–1.56	109	1.43	0.88–2.18
Packaging	2	80	1.24	0.83–1.78	59	1.28	0.85–1.85	21	1.14	0.60–1.93
Solvents	6	43	1.17	0.76–1.70	30	1.19	0.74–1.79	13	1.37	0.66–2.42
Spent baths	15	326	**1.43**	**1.09–1.83**	240	1.30	0.98–1.68	86	**1.55**	**1.01–2.25**
Physico/chemical treatment	5	11	1.52	0.72–2.65	8	1.51	0.64–2.81	3	1.75	0.40–4.25
Industrial waste	7	186	1.03	0.73–1.39	133	1.00	0.70–1.37	53	1.14	0.66–1.79
Wastes not otherwise specified	1	3	1.84	0.37–4.84	2	1.61	0.23–4.68	1	3.71	0.22–13.91
Multiple pollutant categories	56	1105	**1.18**	**1.02–1.36**	774	**1.18**	**1.01–1.37**	331	1.20	0.96–1.49
Gallbladder cancer										
Incineration	12	201	1.24	0.98–1.55	81	**1.43**	**1.04–1.92**	120	1.11	0.83–1.44
Scrap metal + ELVs	52	172	1.10	0.90–1.32	65	1.24	0.91–1.62	107	1.04	0.81–1.30
Oil + oily waste	7	116	1.04	0.77–1.36	43	1.23	0.79–1.78	73	1.01	0.71–1.39
Packaging	2	33	1.01	0.66–1.46	12	1.09	0.54–1.87	21	1.01	0.59–1.55
Solvents	6	17	1.22	0.69–1.92	6	1.31	0.49–2.57	11	1.21	0.59–2.07
Spent baths	15	177	1.07	0.83–1.35	64	1.25	0.85–1.76	113	0.97	0.71–1.29
Physico/chemical treatment	5	7	1.75	0.71–3.28	4	2.90	0.85–6.33	3	1.27	0.30–3.02
Industrial waste	7	104	0.94	0.69–1.23	44	1.23	0.78–1.80	60	0.84	0.57–1.17
Wastes not otherwise specified	1	3	2.08	0.47–5.13	1	2.60	0.17–9.31	2	2.24	0.35–6.31
Multiple pollutant categories	56	509	1.13	0.98–1.29	191	**1.25**	**1.01–1.53**	318	1.06	0.89–1.25
Lung cancer										
Incineration	12	2960	**1.17**	**1.01–1.34**	2682	**1.19**	**1.01–1.38**	278	0.94	0.75–1.16
Scrap metal + ELVs	52	2496	1.05	0.96–1.14	2255	1.07	0.98–1.17	241	0.88	0.74–1.05
Oil + oily waste	7	1772	1.13	0.97–1.31	1618	1.15	0.98–1.35	154	0.90	0.67–1.17
Packaging	2	474	1.03	0.85–1.24	414	1.05	0.85–1.28	60	0.96	0.67–1.32
Solvents	6	229	0.94	0.77–1.14	204	0.96	0.77–1.18	25	0.80	0.49–1.19
Spent baths	15	2485	1.12	0.99–1.27	2132	1.13	0.99–1.29	353	1.07	0.84–1.33
Physico/chemical treatment	5	82	1.24	0.94–1.58	69	1.18	0.88–1.54	13	1.72	0.89–2.82
Industrial waste	7	1570	1.09	0.94–1.26	1388	1.13	0.96–1.32	182	0.82	0.62–1.07
Wastes not otherwise specified	1	35	1.20	0.71–1.88	31	1.21	0.69–1.95	4	1.29	0.36–2.95
Multiple pollutant categories	56	7111	**1.14**	**1.06–1.22**	6363	**1.17**	**1.08–1.26**	748	0.91	0.80–1.03
Pleural cancer										
Incineration	12	55	1.55	0.94–2.39	42	**1.98**	**1.09–3.29**	13	1.16	0.52–2.15
Scrap metal + ELVs	52	38	1.37	0.87–2.01	22	1.13	0.63–1.83	16	1.93	0.99–3.27
Oil + oily waste	7	49	**3.45**	**1.97–5.54**	43	**4.85**	**2.50–8.34**	6	1.25	0.41–2.71
Packaging	2	9	1.64	0.66–3.19	7	1.88	0.65–3.98	2	1.44	0.22–4.04
Solvents	6	2	0.93	0.15–2.57	1	0.74	0.05–2.64	1	2.28	0.15–8.12
Spent baths	15	43	1.50	0.86–2.41	35	1.87	0.98–3.18	8	0.93	0.35–1.88

Table 2 (*continued*)

	T[a]	Total			Men			Women		
		Obs[b]	RR[c]	95%CrI[d]	Obs	RR[c]	95%CrI[d]	Obs	RR[c]	95%CrI[d]
Pleural cancer										
Physico/chemical treatment	5	1	1.78	0.12–6.32	0	0	0-inf	1	7.11	0.46–25.47
Industrial waste	7	30	1.78	0.95–2.98	20	1.82	0.83–3.35	10	1.85	0.77–3.56
Wastes not otherwise specified	1	0	0	0-inf	0	0	0-inf	0	0	0-inf
Multiple pollutant categories	56	167	**1.79**	**1.31–2.36**	114	**1.90**	**1.32–2.64**	53	**1.83**	**1.13–2.76**
Connective and soft tissue cancer										
Incineration	12	57	1.04	0.74–1.41	24	0.85	0.53–1.26	33	1.29	0.81–1.92
Scrap metal + ELVs	52	58	1.10	0.81–1.45	30	1.13	0.74–1.60	28	1.12	0.71–1.63
Oil + oily waste	7	52	**1.48**	**1.01–2.06**	22	1.32	0.80–1.99	30	1.47	0.85–2.28
Packaging	2	13	1.18	0.61–1.94	9	1.59	0.72–2.81	4	0.82	0.24–1.81
Solvents	6	2	0.40	0.07–1.08	1	0.46	0.03–1.61	1	0.52	0.04–1.83
Spent baths	15	53	1.11	0.77–1.54	27	1.15	0.72–1.70	26	1.04	0.61–1.62
Physico/chemical treatment	5	0	0	0-inf	0	0	0-inf	0	0	0-inf
Industrial waste	7	41	1.03	0.69–1.46	23	1.23	0.74–1.86	18	0.97	0.53–1.57
Wastes not otherwise specified	1	1	1.94	0.13–6.89	0	0	0-inf	1	4.38	0.28–15.77
Multiple pollutant categories	56	156	1.06	0.85–1.29	84	1.13	0.86–1.44	72	1.00	0.73–1.32
Skin cancer										
Incineration	12	39	1.12	0.71–1.66	22	1.07	0.61–1.69	17	1.15	0.58–1.99
Scrap metal + ELVs	52	35	0.92	0.61–1.30	18	0.86	0.49–1.33	17	0.99	0.55–1.59
Oil + oily waste	7	54	1.50	0.95–2.22	38	**2.14**	**1.31–3.22**	16	1.06	0.50–1.88
Packaging	2	9	1.05	0.45–1.96	8	1.70	0.71–3.18	1	0.36	0.02–1.29
Solvents	6	10	**2.34**	**1.06–4.20**	7	**3.30**	**1.30–6.34**	3	1.49	0.33–3.70
Spent baths	15	47	1.04	0.65–1.55	25	1.12	0.65–1.77	22	0.96	0.48–1.66
Physico/chemical treatment	5	0	0	0-inf	0	0	0-inf	0	0	0-inf
Industrial waste	7	41	1.00	0.60–1.55	28	1.40	0.82–2.19	13	0.68	0.29–1.29
Wastes not otherwise specified	1	1	1.76	0.11–6.44	0	0	0-inf	1	3.75	0.21–14.19
Multiple pollutant categories	56	116	1.14	0.88–1.46	62	1.07	0.77–1.45	54	1.14	0.77–1.60
Vulvar and vaginal cancer										
Incineration	12							42	1.01	0.70–1.40
Scrap metal + ELVs	52							40	1.03	0.72–1.41
Oil + oily waste	7							47	**1.85**	**1.28–2.56**
Packaging	2							6	0.81	0.30–1.59
Solvents	6							6	1.68	0.63–3.27
Spent baths	15							37	0.89	0.58–1.29
Physico/chemical treatment	5							1	1.33	0.09–4.65
Industrial waste	7							41	**1.55**	**1.02–2.24**
Wastes not otherwise specified	1							0	0	0-inf
Multiple pollutant categories	56							96	0.89	0.69–1.12
Ovarian cancer										
Incineration	12							251	1.13	0.95–1.34
Scrap metal + ELVs	52							228	1.08	0.92–1.25
Oil + oily waste	7							151	1.08	0.87–1.33
Packaging	2							59	1.34	0.99–1.75
Solvents	6							23	1.07	0.67–1.56
Spent baths	15							281	**1.29**	**1.07–1.53**
Physico/chemical treatment	5							8	1.32	0.58–2.37
Industrial waste	7							158	1.08	0.86–1.33
Wastes not otherwise specified	1							2	0.94	0.16–2.56
Multiple pollutant categories	56							691	**1.15**	**1.03–1.27**
Bladder cancer										
Incineration	12	567	1.13	0.95–1.34	474	1.13	0.94–1.36	93	0.98	0.75–1.24
Scrap metal + ELVs	52	573	1.11	0.98–1.25	483	**1.16**	**1.02–1.32**	90	0.99	0.77–1.24
Oil + oily waste	7	413	1.09	0.88–1.33	348	1.11	0.88–1.38	65	1.11	0.80–1.48
Packaging	2	128	1.27	0.98–1.62	102	1.24	0.93–1.61	26	1.43	0.90–2.09
Solvents	6	46	0.98	0.69–1.33	36	0.95	0.64–1.33	10	1.26	0.60–2.16
Spent baths	15	528	1.01	0.84–1.20	431	1.02	0.84–1.23	97	0.99	0.74–1.28
Physico/chemical treatment	5	15	1.09	0.60–1.72	13	1.16	0.61–1.89	2	1.03	0.17–2.78
Industrial waste	7	363	1.05	0.84–1.28	302	1.04	0.82–1.30	61	1.00	0.71–1.35
Wastes not otherwise specified	1	5	0.87	0.28–1.84	3	0.66	0.15–1.60	2	2.46	0.40–6.73
Multiple pollutant categories	56	1493	1.09	0.99–1.20	1227	1.09	0.98–1.21	266	1.02	0.86–1.19
Renal cancer										
Incineration	12	240	1.08	0.88–1.30	150	1.04	0.83–1.28	90	1.18	0.88–1.53
Scrap metal + ELVs	52	290	**1.36**	**1.17–1.58**	198	**1.39**	**1.16–1.64**	92	**1.33**	**1.03–1.67**
Oil + oily waste	7	151	1.14	0.90–1.44	99	1.10	0.83–1.42	52	1.17	0.81–1.63
Packaging	2	55	1.24	0.90–1.67	36	1.19	0.80–1.66	19	1.32	0.77–2.03
Solvents	6	21	0.98	0.59–1.46	12	0.84	0.43–1.39	9	1.33	0.61–2.35
Spent baths	15	284	1.06	0.86–1.27	189	1.04	0.83–1.28	95	1.16	0.86–1.52
Physico/chemical treatment	5	14	**2.25**	**1.22–3.61**	10	**2.43**	**1.16–4.17**	4	2.15	0.64–4.66
Industrial waste	7	165	0.95	0.75–1.19	107	0.95	0.73–1.22	58	1.03	0.72–1.39
Wastes not otherwise specified	1	3	1.16	0.27–2.80	3	1.77	0.41–4.26	0	0	0-inf
Multiple pollutant categories	56	695	1.11	0.99–1.25	464	1.11	0.97–1.26	231	1.12	0.93–1.33
Brain cancer										
Incineration	12	322	0.99	0.84–1.16	178	0.97	0.79–1.18	144	1.03	0.82–1.27
Scrap metal + ELVs	52	288	1.00	0.86–1.15	160	0.96	0.80–1.14	128	1.04	0.85–1.26

(*continued on next page*)

Table 2 (*continued*)

	T[a]	Total			Men			Women		
		Obs[b]	RR[c]	95%CrI[d]	Obs	RR[c]	95%CrI[d]	Obs	RR[c]	95%CrI[d]
Brain cancer										
Oil + oily waste	7	193	1.00	0.80–1.22	90	0.85	0.65–1.09	103	1.24	0.94–1.59
Packaging	2	82	1.31	0.99–1.68	51	**1.41**	**1.01–1.90**	31	1.18	0.77–1.68
Solvents	6	35	1.07	0.73–1.48	22	1.14	0.70–1.69	13	0.98	0.52–1.59
Spent baths	15	300	0.99	0.82–1.19	153	0.94	0.75–1.17	147	1.07	0.84–1.35
Physico/chemical treatment	5	7	0.75	0.31–1.39	6	1.11	0.42–2.15	1	0.37	0.03–1.30
Industrial waste	7	233	1.12	0.90–1.37	132	1.11	0.86–1.41	101	1.15	0.87–1.48
Wastes not otherwise specified	1	9	1.99	0.88–3.60	3	1.32	0.31–3.17	6	**3.29**	**1.20–6.56**
Multiple pollutant categories	56	911	1.06	0.96–1.17	490	1.01	0.89–1.14	421	1.14	0.99–1.30
Thyroid cancer										
Incineration	12	31	0.93	0.59–1.36	7	0.63	0.25–1.20	24	1.09	0.64–1.69
Scrap metal + ELVs	52	52	**1.63**	**1.16–2.20**	22	**1.97**	**1.17–3.00**	30	1.42	0.91–2.06
Oil + oily waste	7	20	1.05	0.59–1.66	6	0.89	0.33–1.77	14	1.13	0.57–1.93
Packaging	2	10	1.51	0.70–2.66	3	1.37	0.32–3.29	7	1.66	0.65–3.18
Solvents	6	5	1.68	0.57–3.45	1	1.20	0.08–4.22	4	2.16	0.63–4.75
Spent baths	15	39	1.14	0.73–1.66	14	1.31	0.67–2.22	25	1.14	0.66–1.79
Physico/chemical treatment	5	2	2.42	0.40–6.57	0	0.00	0-inf	2	3.82	0.62–10.43
Industrial waste	7	25	1.09	0.65–1.68	8	1.08	0.45–2.03	17	1.08	0.57–1.78
Wastes not otherwise specified	1	0	0	0-inf	0	0.00	0-inf	0	0	0-inf
Multiple pollutant categories	56	98	1.06	0.81–1.35	30	0.94	0.59–1.37	68	1.12	0.81–1.49
Hodgkin's lymphoma										
Incineration	12	32	0.87	0.56–1.26	18	0.90	0.51–1.41	14	0.95	0.49–1.56
Scrap metal + ELVs	52	45	1.41	0.99–1.91	27	1.52	0.97–2.21	18	1.38	0.79–2.15
Oil + oily waste	7	15	0.81	0.43–1.32	9	0.89	0.40–1.59	6	0.74	0.27–1.46
Packaging	2	4	0.63	0.19–1.38	3	0.87	0.21–2.06	1	0.53	0.04–1.84
Solvents	6	4	1.14	0.34–2.48	3	1.50	0.36–3.58	1	0.98	0.07–3.43
Spent baths	15	25	0.93	0.56–1.42	11	0.72	0.35–1.25	14	1.12	0.58–1.90
Physico/chemical treatment	5	3	3.39	0.81–8.05	3	**5.64**	**1.34–13.43**	0	0	0-inf
Industrial waste	7	16	0.78	0.42–1.26	10	0.89	0.41–1.58	6	0.71	0.26–1.41
Wastes not otherwise specified	1	1	3.46	0.23–12.26	1	5.95	0.40–21.08	0	0	0-inf
Multiple pollutant categories	56	93	1.04	0.79–1.32	48	0.96	0.67–1.30	45	1.21	0.83–1.68
Leukemia										
Incineration	16	416	1.05	0.97–1.13	245	1.08	0.98–1.18	171	1.03	0.93–1.13
Scrap metal + ELVs	56	430	**1.14**	**1.01–1.28**	227	1.09	0.93–1.26	203	**1.23**	**1.04–1.43**
Oil + oily waste	24	387	1.08	0.90–1.28	216	1.14	0.91–1.39	171	1.03	0.80–1.28
Packaging	9	135	1.15	0.89–1.44	79	1.11	0.80–1.48	56	1.21	0.85–1.64
Solvents	4	33	1.29	0.94–1.70	16	1.28	0.83–1.82	17	1.35	0.85–1.98
Spent baths	14	195	1.01	0.85–1.18	112	1.12	0.92–1.35	83	0.86	0.68–1.06
Physico/chemical treatment	8	1573	1.33	0.74–2.08	840	0.97	0.37–1.87	733	1.95	0.90–3.39
Industrial waste	13	354	1.01	0.84–1.21	188	1.04	0.83–1.28	166	0.99	0.77–1.24
Wastes not otherwise specified	11	22	1.03	0.30–2.25	12	1.40	0.33–3.34	10	0.79	0.06–2.77
Multiple pollutant categories	82	1833	**1.13**	**1.04–1.23**	1021	**1.14**	**1.02–1.26**	812	1.12	0.99–1.26

[a] Number of towns situated at ≤5 km from incinerators and hazardous waste treatment installations as a whole.
[b] Observed deaths.
[c] RRs adjusted for population size, percentage illiteracy, farmers and unemployed persons, average persons per household, and mean income.
[d] 95% credible interval.
[e] Sum of the 33 types of cancer analyzed.

(RRs = 5.64 and 2.43, respectively); bladder and thyroid cancer in men and leukemias in women in the vicinity of scrap metal + ELV recycling installations (RRs = 1.16, 1.97 and 1.23, respectively); brain cancer in women living near other waste treatment installations (RR = 3.29); and cancers of the pleura in men, vulva and vagina in women, and connective tissue in the total population (RRs = 4.85, 1.85 and 1.48, respectively), in the environs of oil and oily waste treatment installations. If we analyze the results on stratifying risk by category of industrial activity, the following associations were found between malignant tumors and residential proximity to certain types of installations: a) "Incinerators", and tumors of the lung, pleura and gallbladder (men) and stomach (women); b) "Installations for the recycling of scrap metal and ELVs", and cancer of the kidney (men and women), tumors of the stomach, bladder and thyroid (men) and leukemia (women); c) "Installations for the treatment of used oil and oily waste", and cancer of the connective tissue (total population), tumors of the stomach, pleura and skin (men), and of vulva and vagina (women); d) "Packaging recycling installations", and tumors of the stomach, colon–rectum and brain (men); e) "Installations for the recovery of used solvents", and skin cancer (men); f) "Installations for the regeneration of spent baths", and cancer of the stomach (total population), colorectal cancer (men), and tumors of the liver and ovary (women); g) "Installations for physico/chemical treatment of wastes", and cancer of the kidney (men); h) "Industrial waste treatment installations", and tumors of the stomach, vulva and vagina (women); and, i) "Installations for the treatment of wastes not otherwise specified", and cancer of the brain (women). In addition, towns situated near several installations of "Multiple pollutant categories" displayed significant results for malignant tumors of the stomach and pleura (men and women), colon–rectum, liver, gallbladder, lung and leukemia (men), and ovary (women).

Table 3 shows the RRs in the vicinity of specific incinerators and hazardous waste treatment facilities which registered statistically significant excess risks in the "near vs. far" analysis and a number of observed deaths ≥15. There are a total of 3 incinerators, 15 installations for the recycling of scrap metal and ELVs, 6 installations for the treatment of used oil and oily waste, 3 packaging recycling installations, 2 installations for the recovery of used solvents, 3 installations for the regeneration of spent baths, 3 installations for physico/chemical treatments of wastes, 4 industrial waste treatment installations, and 6 installations for the treatment of wastes not otherwise specified, with significant results. Many of the installations displayed considerably high RRs for more than one tumor simultaneously, and this was especially true for installations '372', '4699' and '5692' ("Scrap metal + ELVs"), '3710' ("Industrial waste"), and '6053' ("Wastes not otherwise

specified"), with statistically significant results for 6 tumors, and installations '3055' and '7476' ("Scrap metal + ELVs"), '3713' ("Spent baths"), '3110' ("Physico/chemical treatment"), '3711' ("Industrial waste"), and '7478 ("Wastes not otherwise specified"), with statistically significant results for 5 tumors. It is also noteworthy to note that there are 11 facilities with significant excess risk for all cancers combined: installations '372' (RR = 1.28 in women), '3055' (RR = 1.10 in the total population), '5692' (RR = 1.30 in women), '6051' (RR = 1.21 in women), '3050' (RR = 1.19 in women), '3110' (RR = 1.30 in women), and '7478' (RR = 1.10 in the total population), located in the province of Barcelona; installations '4699' (RR = 1.13 in men), '5910' (RR = 1.27 in men), '3710' (RR = 1.13 in men), and '3711' (RR = 1.33 in men), located in the province of Vizcaya); and, installation '5493' (RR = 1.20 in men), located in the province of Granada.

4. Discussion

This study is one of the first to use IPPC- and E-PRTR-registered industrial data to explore the effects of industrial waste-treatment on cancer mortality in neighboring towns. In general, our results suggest that there is a moderate increased risk of dying of all cancers combined, higher among men than among women, in the vicinity of Spanish incinerators and hazardous waste treatment plants as a whole. Stratifying the risk by industrial activity, high statistically significant excess risks were detected in towns lying near "Incinerators" (total population), "Installations for the recycling of scrap metal and ELVs", "Installations for the regeneration of spent baths" (men), and various installations of "Multiple pollutant categories" (men and women).

On analyzing cancers individually, significant excess risks were observed for malignant tumors of the stomach, liver, pleura and kidney (men and women), colon–rectum, lung, bladder, gallbladder and leukemia (men), and brain and ovary (women). Furthermore, on stratifying risk by category of industrial activity, the following associations were found between other malignant tumors and residential proximity to certain types of installations: "Installations for the recycling of scrap metal and ELVs", and tumors of the stomach and thyroid (men); "Installations for the treatment of used oil and oily waste", and cancer of the connective tissue (total population), tumors of the skin (men), and of the vulva and vagina (women); "Installations for the recovery of used solvents", and skin tumor (men); and, "Industrial waste treatment installations", and tumor of the vulva and vagina (women).

The fact that statistically significant results, with RRs $\geq$1.10, appeared mainly for tumors of both the digestive and respiratory system (in total population), leads us to suspect two possible routes of exposure to the pollution released by these installations, namely: direct exposure to pollutants released to air; and indirect exposure, both to pollutants and liquid effluents which are released to water and can then pass into the soil and aquifers, and pollutants which are released to air and then settle on plants. In such cases, the toxins may pass into the trophic chain, affecting the population.

The hypothesis that some excess cancer mortality may be due to population exposure to industrial pollution is reinforced by recent studies that have reported associations between residential proximity to certain types of industrial installations and certain malignant tumors (Garcia-Perez et al., 2010, 2012; Lopez-Abente et al., 2012; Musti et al., 2009; Tsai et al., 2009). As regards incinerators and hazardous waste treatment plants, studies have almost exclusively focused on the environs of incinerators, where associations have been found with some tumors, such as NHL (Floret et al., 2003; Viel et al., 2011), soft tissue sarcomas (Comba et al., 2003), and childhood tumors (Knox, 2000).

Ecologic studies, such as that reported here, are proposing new hypotheses and lines of research with respect to population exposure to industrial pollution. In this regard, one of the principal strengths of our study resides in the completeness of its exploratory analysis, which consisted of an in-depth examination of mortality due to 33 types of cancer with reference to different categories of industrial activity. Another strength was its use of different methodological approaches to perform the statistical analysis: one, based on a hierarchical spatial model at a municipal level, with inclusion of explanatory variables (BYM model), in which the use of spatial terms in the model, not only meant that it was less susceptible to the presence of the ecological fallacy (Clayton et al., 1993), but also ensured that the geographic heterogeneity of the distribution of mortality was taken into account; and the other, based on a Poisson mixed regression model, was justified by its ease of adjustment and shorter computation times. Although the results in the two models used are not very different in general, the presence of spatial autocorrelation in some of the tumors studied renders the use of spatial models advisable. Moreover, the method of estimation afforded by INLA, as an alternative to Markov chain Monte Carlo methods, amounts to a qualitative leap in the use of hierarchical models with explanatory variables (Rue et al., 2009). A consideration to bear in mind is that mixed models seem to be more sensitive to detect potential statistical associations than spatial models, which are more restrictive. An example of the above mentioned can be seen in our results on NHL in males, where the mixed model provided statistically significant results (RR = 1.12, 95%CI = 1.03–1.22) whereas the model BYM did not show a statistically significant association (RR = 1.07, 95%CrI = 0.97–1.19).

Further advantages of the study are: its high statistical power, thanks to the inclusion of a great number of reported deaths, a factor that enables it to identify excess mortality of a lower magnitude, in line with the expected effects of environmental exposures; analysis of risk in the vicinity of industrial activities such as ELV-disposal or scrap-metal recycling plants, which had never before been studied as a whole, as well as detailed individual analyses of the respective installations; elimination for study purposes of those installations that had come into operation most recently, and whose possible influence on tumor development is debatable if the minimum latency periods of the tumors analyzed are taken into account; and inclusion of towns lying close to industries other than incinerators and hazardous waste treatment installations, as the "intermediate category" in the analyses, something that avoids the confounding effect of such industries (which release toxic substances that could be related to the tumors under study) and allows for the establishment of a "clean" reference group made up of towns having no industry in their vicinity.

Aside from the limitations inherent to all ecologic studies, in our case mention should also be made of the following: the inclusion of many variables in the models that could make the analyses very susceptible to type I error; the non-inclusion of possible confounding factors that might be associated with distance (though adjustment for socioeconomic variables goes some way to mitigating this lack of information, since many life-style-related risk factors, such as smoking, alcohol consumption, type of diet or infectious agents, show a distribution correlated with socioeconomic status (Prattala et al., 2009; Woitas-Slubowska et al., 2010)); the use of distance from town of residence to industrial centers as a "proxy" of population exposure to industrial pollution, based on the assumption of an isotropic model, since real exposure may depend on prevailing wind patterns or geographical landforms (though this would limit the capacity for detecting positive results, without invalidating the associations found); and the use of mortality rather than incidence data, due to the absence of a national population-based incidence register (though in Spain, tumors with lower survival rates are well represented by death certificates (Perez-Gomez et al., 2006)).

A critical decision when designing the study was the choice of categories of industrial activity for stratifying risk in the analyses. In this respect, we chose to construct the categories according to the characteristics of the waste applicable and type of treatment used (Agència de Residus de Catalunya, 2012; Special Territorial Plan of Waste Management (PTEOR), 2012). Furthermore, landfills, composting

Table 3
Relative risk of dying from cancers with significant results and a number of observed deaths ≥15 in towns situated at a distance of less than 5 km from specific incinerators and hazardous waste treatment installations, estimated using BYM models. Significant results are in bold.

Industrial activity[a]	PRTR Code		T[b]	Obs[c]	BYM model		Industrial activity[a]	PRTR code		T[b]	Obs[c]	BYM model	
					RR[d]	95% CrI[e]						RR[d]	95% CrI[e]
All cancers[f]							Pancreatic cancer						
2	372	Total	4	949	**1.11**	**1.01–1.23**	2	5680	Total	5	24	**1.94**	**1.11–3.09**
		Men	4	591	1.03	0.91–1.17			Men	5	12	1.35	0.61–2.47
		Women	4	358	**1.28**	**1.10–1.48**			Women	5	12	**3.15**	**1.38–5.95**
2	3055	Total	5	1370	**1.10**	**1.00–1.20**	2	5691	Total	3	27	**2.08**	**1.27–3.14**
		Men	5	916	1.10	0.98–1.22			Men	3	15	**2.06**	**1.07–3.47**
		Women	5	454	1.09	0.95–1.25			Women	3	12	**2.16**	**1.02–3.84**
2	4699	Total	6	4803	1.10	0.99–1.21	2	7476	Total	2	137	1.36	0.90–1.95
		Men	6	3184	**1.13**	**1.00–1.27**			Men	2	71	1.09	0.62–1.77
		Women	6	1619	1.04	0.91–1.19			Women	2	66	**1.86**	**1.04–3.06**
2	5692	Total	3	864	1.09	0.98–1.21	7	3110	Total	3	32	1.27	0.77–1.94
		Men	3	531	0.99	0.87–1.13			Men	3	23	**1.86**	**1.05–3.00**
		Women	3	333	**1.30**	**1.11–1.51**			Women	3	9	0.76	0.31–1.45
2	6051	Total	3	2441	**1.11**	**1.00–1.23**	8	65	Total	2	388	**1.67**	**1.01–2.60**
		Men	3	1612	1.06	0.94–1.20			Men	2	202	**2.08**	**1.08–3.66**
		Women	3	829	**1.21**	**1.04–1.39**			Women	2	186	1.30	0.61–2.41
3	5493	Total	3	561	**1.18**	**1.00–1.38**	8	6749	Total	9	299	1.30	0.85–1.90
		Men	3	350	**1.20**	**1.00–1.42**			Men	9	153	**1.79**	**1.03–2.89**
		Women	3	211	1.11	0.90–1.36			Women	9	146	0.93	0.49–1.58
3	5910	Total	3	472	**1.25**	**1.08–1.43**	9	6053	Total	2	137	1.36	0.90–1.95
		Men	3	309	**1.27**	**1.07–1.51**			Men	2	71	1.10	0.62–1.78
		Women	3	163	1.21	0.97–1.47			Women	2	66	**1.84**	**1.03–3.05**
4	3050	Total	3	1308	**1.12**	**1.01–1.24**							
		Men	3	847	1.08	0.95–1.23	Peritoneal cancer						
		Women	3	461	**1.19**	**1.02–1.38**	6	3713	Total	6	42	2.00	0.59–5.00
7	3110	Total	3	654	1.09	0.97–1.22			Men	6	19	**9.04**	**4.80–32.66**
		Men	3	398	0.99	0.86–1.14			Women	6	23	0.93	0.22–2.80
		Women	3	256	**1.30**	**1.09–1.52**							
8	3710	Total	6	4803	1.10	0.99–1.21	Laryngeal cancer						
		Men	6	3184	**1.13**	**1.00–1.27**	2	372	Total	4	21	**1.91**	**1.02–3.21**
		Women	6	1619	1.04	0.91–1.19			Men	4	21	**2.11**	**1.12–3.58**
8	3711	Total	4	713	**1.26**	**1.11–1.42**			Women	4	0	0	0-inf
		Men	4	478	**1.33**	**1.14–1.54**	2	3055	Total	5	31	**1.88**	**1.09–3.01**
		Women	4	235	1.13	0.93–1.35			Men	5	30	**1.99**	**1.13–3.23**
9	7478	Total	5	1370	**1.10**	**1.00–1.20**			Women	5	1	1.49	0.13–5.00
		Men	5	916	1.10	0.98–1.22	2	5692	Total	3	20	**1.96**	**1.03–3.32**
		Women	5	454	1.09	0.95–1.25			Men	3	20	**2.17**	**1.13–3.70**
									Women	3	0	0	0-inf
Esophageal cancer							9	7478	Total	5	31	**1.88**	**1.09–3.01**
2	3055	Total	5	45	**1.59**	**1.00–2.38**			Men	5	30	**1.99**	**1.13–3.23**
		Men	5	44	**1.74**	**1.08–2.64**			Women	5	1	1.49	0.13–5.00
		Women	5	1	0.47	0.05–1.51							
9	7478	Total	5	45	**1.59**	**1.00–2.38**	Lung cancer						
		Men	5	44	**1.74**	**1.08–2.64**	2	4699	Total	6	990	1.20	0.96–1.48
		Women	5	1	0.47	0.05–1.51			Men	6	893	**1.30**	**1.02–1.64**
									Women	6	97	0.77	0.47–1.19
Stomach cancer							2	7476	Total	2	566	**1.39**	**1.05–1.81**
2	6049	Total	5	49	1.63	0.96–2.58			Men	2	511	**1.43**	**1.07–1.90**
		Men	5	27	1.35	0.73–2.29			Women	2	55	1.04	0.55–1.84
		Women	5	22	**2.26**	**1.00–4.29**	3	5493	Total	3	135	**1.39**	**1.04–1.81**
3	5493	Total	3	36	1.31	0.82–1.94			Men	3	120	1.33	0.98–1.76
		Men	3	25	**1.73**	**1.02–2.68**			Women	3	15	**2.27**	**1.06–4.14**
		Women	3	11	0.82	0.36–1.51	3	7412	Total	1	819	1.31	0.97–1.71
6	4719	Total	8	43	**1.72**	**1.03–2.69**			Men	1	743	**1.40**	**1.02–1.87**
		Men	8	31	1.60	0.87–2.70			Women	1	76	0.81	0.43–1.38
		Women	8	12	1.98	0.77–4.07	5	1678	Total	2	164	1.24	0.98–1.56
6	4833	Total	2	94	**1.59**	**1.05–2.32**			Men	2	143	**1.29**	**1.00–1.63**
		Men	2	44	1.27	0.73–2.05			Women	2	21	0.89	0.50–1.46
		Women	2	50	**2.26**	**1.24–3.78**	8	3710	Total	6	990	1.20	0.96–1.48
									Men	6	893	**1.30**	**1.02–1.64**
Colorectal cancer									Women	6	97	0.77	0.47–1.19
2	372	Total	4	134	1.25	0.98–1.58	8	3711	Total	4	141	**1.38**	**1.02–1.82**
		Men	4	71	1.14	0.82–1.52			Men	4	126	**1.45**	**1.06–1.95**
		Women	4	63	**1.41**	**1.00–1.92**			Women	4	15	1.09	0.51–2.00
2	4699	Total	6	605	1.19	0.95–1.47	9	6053	Total	2	566	**1.37**	**1.05–1.79**
		Men	6	380	**1.35**	**1.02–1.74**			Men	2	511	**1.42**	**1.07–1.88**
		Women	6	225	1.00	0.72–1.37			Women	2	55	1.04	0.55–1.84
2	7476	Total	2	433	**1.35**	**1.04–1.72**							
		Men	2	247	**1.48**	**1.09–1.96**	Pleural cancer						
		Women	2	186	1.23	0.81–1.75	2	4699	Total	6	30	4.75	0.74–13.97
6	3713	Total	6	1976	1.19	0.96–1.46			Men	6	25	**4.33**	**4.56–13.64**
		Men	6	1182	**1.34**	**1.03–1.72**			Women	6	5	inf	0-inf
		Women	6	794	1.03	0.74–1.38	6	3713	Total	6	61	2.82	0.73–9.02

Table 3 (*continued*)

Industrial activity[a]	PRTR Code		T[b]	Obs[c]	BYM model		Industrial activity[a]	PRTR code		T[b]	Obs[c]	BYM model	
					RR[d]	95% CrI[e]						RR[d]	95% CrI[e]
Colorectal cancer							Pleural cancer						
7	3110	Total	3	87	1.14	0.86–1.49			Men	6	50	**2.44**	**3.64–7.75**
		Men	3	41	0.92	0.62–1.31			Women	6	11	inf	0-inf
		Women	3	46	**1.49**	**1.01–2.09**	8	65	Total	2	27	**8.73**	**1.32–35.97**
8	3710	Total	6	605	1.19	0.95–1.47			Men	2	19	**12.23**	**1.41–41.46**
		Men	6	380	**1.35**	**1.02–1.74**			Women	2	8	NE[g]	NE[g]
		Women	6	225	1.00	0.72–1.37	8	3710	Total	6	30	4.75	0.74–13.97
9	6053	Total	2	433	**1.35**	**1.04–1.71**			Men	6	25	**4.33**	**4.57–13.64**
		Men	2	247	**1.47**	**1.08–1.95**			Women	6	5	inf	0-inf
		Women	2	186	1.23	0.81–1.75	8	6749	Total	9	25	3.44	0.86–9.74
									Men	9	10	1.24	0.26–4.47
Liver cancer									Women	9	15	**18.61**	**3.58–79.24**
2	7476	Total	2	99	**2.40**	**1.40–3.87**							
		Men	2	73	**2.59**	**1.42–4.36**	Bone cancer						
		Women	2	26	2.29	0.75–5.34	1	467	Total	3	29	**2.89**	**1.04–6.64**
3	1612	Total	1	176	**2.25**	**1.23–3.77**			Men	3	23	**12.40**	**11.67–47.49**
		Men	1	102	1.91	0.92–3.56			Women	3	6	0.88	0.14–2.63
		Women	1	74	**3.79**	**1.32–8.43**	1	4857	Total	3	29	**2.89**	**1.05–6.64**
6	4833	Total	2	58	2.51	0.98–5.29			Men	3	23	**12.29**	**17.17–46.76**
		Men	2	34	2.12	0.69–4.96			Women	3	6	0.88	0.14–2.63
		Women	2	24	**3.65**	**1.08–9.53**	6	3713	Total	6	28	2.31	0.02–7.79
9	6053	Total	2	99	**2.36**	**1.37–3.79**			Men	6	20	**3.18**	**2.61–11.22**
		Men	2	73	**2.56**	**1.40–4.30**			Women	6	8	**14.49**	**1.79–73.89**
		Women	2	26	2.17	0.71–5.08	8	65	Total	2	26	**6.90**	**1.65–22.49**
									Men	2	15	3.26	0.40–13.19
									Women	2	11	NE[g]	NE[g]
Connective and soft tissue							Ill-defined tumors						
3	6789	Total	2	34	2.55	0.62–7.25	2	5664	Total	2	36	**1.74**	**1.15–2.49**
		Men	2	19	**9.41**	**3.10–35.45**			Men	2	28	**2.47**	**1.51–3.74**
		Women	2	15	0.90	0.02–3.65			Women	2	8	0.88	0.36–1.67
8	6749	Total	9	36	2.28	0.52–6.03	2	5682	Total	6	168	**1.36**	**1.00–1.81**
		Men	9	19	**6.65**	**4.82–23.45**			Men	6	94	1.28	0.86–1.83
		Women	9	17	0.93	0.11–3.55			Women	6	74	1.53	0.97–2.28
							6	4833	Total	2	115	1.41	0.82–2.22
Melanoma									Men	2	**70**	**2.16**	**1.17–3.64**
2	5063	Total	1	16	**19.55**	**10.16–79.17**			Women	2	45	0.88	0.43–1.62
		Men	1	10	NE[g]	NE[g]							
		Women	1	6	NE[g]	NE[g]	Non-Hodgkin's lymphoma						
6	3713	Total	6	114	1.80	0.82–3.46	1	467	Total	3	215	**1.49**	**1.02–2.12**
		Men	6	56	1.54	0.55–3.49			Men	3	113	1.63	0.95–2.64
		Women	6	58	**2.58**	**1.18–6.89**			Women	3	102	1.45	0.85–2.35
							1	4857	Total	3	215	**1.49**	**1.02–2.12**
Skin cancer									Men	3	113	1.64	0.96–2.67
3	7412	Total	1	39	**6.39**	**1.35–17.89**			Women	3	102	1.44	0.84–2.33
		Men	1	29	**17.38**	**2.92–52.97**	2	5692	Total	3	30	**1.67**	**1.00–2.59**
		Women	1	10	3.04	0.35–10.62			Men	3	18	1.86	0.93–3.26
									Women	3	12	1.52	0.68–2.85
Vulvar and vaginal cancer							2	6051	Total	3	82	**1.60**	**1.03–2.39**
3	7412	Women	1	21	**6.66**	**1.06–23.49**			Men	3	49	**2.15**	**1.15–3.75**
									Women	3	33	1.23	0.65–2.14
Uterine cancer							3	5910	Total	3	15	**2.22**	**1.04–4.04**
4	5557	Women	1	27	**2.12**	**1.00–3.94**			Men	3	9	**3.96**	**1.45–8.36**
8	3711	Women	4	15	**2.27**	**1.05–4.17**			Women	3	6	1.26	0.36–2.94
							8	3711	Total	4	21	**2.01**	**1.02–3.50**
Ovarian cancer									Men	4	12	**3.40**	**1.36–6.91**
1	2438	Women	2	51	**1.95**	**1.09–3.29**			Women	4	9	1.22	0.42–2.66
2	5685	Women	4	17	**2.72**	**1.38–4.70**							
2	7328	Women	3	15	**2.68**	**1.39–4.48**	Myeloma						
3	445	Women	8	156	**1.49**	**1.03–2.09**	2	372	Total	4	21	**2.08**	**1.11–3.49**
4	3050	Women	3	28	**1.82**	**1.04–2.94**			Men	4	10	1.70	0.68–3.42
4	5557	Women	1	36	**2.45**	**1.24–4.31**			Women	4	11	**2.72**	**1.09–5.53**
5	2999	Women	3	16	**2.58**	**1.29–4.52**	2	3055	Total	5	31	**1.91**	**1.11–3.04**
7	3110	Women	3	17	**1.98**	**1.02–3.39**			Men	5	20	**2.25**	**1.09–4.09**
7	3452	Women	4	57	**2.39**	**1.39–3.84**			Women	5	11	1.56	0.64–3.09
9	6431	Women	7	151	**1.46**	**1.00–2.06**	2	5692	Total	3	21	**2.28**	**1.21–3.84**
									Men	3	10	1.87	0.74–3.75
Prostate cancer									Women	3	11	**2.98**	**1.19–6.08**
3	5493	Men	3	43	**1.66**	**1.10–2.38**	6	3713	Total	6	227	**1.69**	**1.02–2.65**
									Men	6	115	**2.62**	**1.25–4.92**
Bladder cancer									Women	6	112	1.21	0.60–2.20
2	5680	Total	5	24	**2.39**	**1.34–3.86**	7	3110	Total	3	16	**2.37**	**1.18–4.18**
		Men	5	21	**2.68**	**1.45–4.45**			Men	3	7	1.87	0.64–4.05
		Women	5	3	1.36	0.24–3.76			Women	3	9	**3.19**	**1.18–6.76**
2	7476	Total	2	116	1.48	0.93–2.19	7	3452	Total	4	54	**1.93**	**1.10–3.22**

(*continued on next page*)

Table 3 (*continued*)

Industrial activity[a]	PRTR Code		T[b]	Obs[c]	BYM model		Industrial activity[a]	PRTR code		T[b]	Obs[c]	BYM model	
					RR[d]	95% CrI[e]						RR[d]	95% CrI[e]
Bladder cancer							Myeloma						
		Men	2	97	**1.68**	**1.03–2.56**			Men	4	23	1.73	0.78–3.34
		Women	2	19	0.85	0.33–1.81			Women	4	31	**2.24**	**1.05–4.32**
9	6053	Total	2	116	1.47	0.92–2.19	9	7478	Total	5	31	**1.91**	**1.11–3.04**
		Men	2	97	**1.67**	**1.01–2.55**			Men	5	20	**2.25**	**1.09–4.09**
		Women	2	19	0.86	0.33–1.81			Women	5	11	1.56	0.64–3.09
Brain cancer							Leukemia						
1	2438	Total	2	69	1.14	0.70–1.79	2	372	Total	4	42	**1.59**	**1.03–2.30**
		Men	2	27	0.78	0.37–1.66			Men	4	22	1.27	0.72–2.05
		Women	2	42	**2.05**	**1.01–3.72**			Women	4	20	**2.28**	**1.16–3.98**
2	372	Total	4	30	1.49	0.89–2.31	2	3055	Total	5	59	**1.58**	**1.08–2.23**
		Men	4	13	0.99	0.47–1.78			Men	5	36	1.56	0.96–2.38
		Women	4	17	**2.59**	**1.17–4.92**			Women	5	23	1.69	0.90–2.87
2	4699	Total	6	111	1.42	0.92–2.10	2	3594	Total	10	50	**1.63**	**1.06–2.40**
		Men	6	59	**1.90**	**1.04–3.20**			Men	10	28	1.70	0.96–2.79
		Women	6	52	1.12	0.59–1.90			Women	10	22	1.65	0.84–2.88
2	5692	Total	3	27	1.43	0.84–2.25	2	4699	Total	6	136	1.24	0.82–1.80
		Men	3	12	0.98	0.45–1.78			Men	6	77	0.96	0.58–1.51
		Women	3	15	**2.50**	**1.09–4.86**			Women	6	59	**1.97**	**1.01–3.49**
3	5910	Total	3	16	**2.25**	**1.11–3.95**	2	5680	Total	5	16	**2.31**	**1.18–3.95**
		Men	3	8	2.63	0.94–5.52			Men	5	10	**2.83**	**1.16–5.51**
		Women	3	8	2.05	0.73–4.36			Women	5	6	1.92	0.63–4.13
4	3050	Total	3	46	**1.60**	**1.02–2.40**	2	5692	Total	3	39	**1.60**	**1.03–2.35**
		Men	3	25	1.36	0.76–2.24			Men	3	20	1.26	0.69–2.05
		Women	3	21	**2.14**	**1.03–3.92**			Women	3	19	**2.37**	**1.19–4.18**
7	2088	Total	3	37	**1.91**	**1.02–3.24**	2	6051	Total	3	81	1.28	0.85–1.86
		Men	3	22	1.97	0.84–3.86			Men	3	43	1.01	0.60–1.60
		Women	3	15	1.88	0.68–4.12			Women	3	38	**2.02**	**1.02–3.67**
8	3710	Total	6	111	1.42	0.92–2.10	3	6789	Total	2	147	**2.11**	**1.13–3.65**
		Men	6	59	**1.90**	**1.04–3.20**			Men	2	85	**2.87**	**1.25–5.82**
		Women	6	52	1.12	0.59–1.90			Women	2	62	1.57	0.63–3.27
8	3711	Total	4	25	**2.42**	**1.31–4.03**	4	3120	Total	5	49	**1.60**	**1.07–2.29**
		Men	4	14	**3.42**	**1.47–6.66**			Men	5	25	1.25	0.72–1.97
		Women	4	11	1.78	0.70–3.60			Women	5	24	**2.37**	**1.26–4.05**
9	2089	Total	3	43	**1.92**	**1.03–3.28**	8	3710	Total	6	136	1.24	0.82–1.80
		Men	3	22	1.52	0.62–3.07			Men	6	77	0.96	0.58–1.51
		Women	3	21	2.47	0.94–5.31			Women	6	59	**1.97**	**1.01–3.49**
9	7403	Total	3	43	**1.92**	**1.03–3.28**	9	5703	Total	5	49	**1.60**	**1.07–2.29**
		Men	3	22	1.52	0.62–3.07			Men	5	25	1.25	0.72–1.97
		Women	3	21	2.47	0.94–5.31			Women	5	24	**2.37**	**1.26–4.05**
							9	6053	Total	2	109	**1.65**	**1.00–2.51**
Thyroid cancer									Men	2	57	1.69	0.89–2.88
1	467	Total	3	21	1.11	0.38–2.59			Women	2	52	1.78	0.97–3.00
		Men	3	6	0.66	0.13–2.18	9	7478	Total	5	59	**1.58**	**1.08–2.23**
		Women	3	15	**2.05**	**1.52–6.14**			Men	5	36	1.56	0.96–2.38
1	4857	Total	3	21	1.10	0.38–2.57			Women	5	23	1.69	0.90–2.87
		Men	3	6	0.65	0.14–2.14							
		Women	3	15	**2.04**	**1.49–6.13**							

[a] 1 = incineration. 2 = scrap metal + ELVs. 3 = oil + oily waste. 4 = packaging. 5 = solvents. 6 = spent baths. 7 = physico/chemical treatment. 8 = industrial waste. 9 = wastes not otherwise specified.
[b] Number of towns situated at ≤5 km from specific incinerators and hazardous waste treatment installations.
[c] Observed deaths.
[d] RRs adjusted for population size, percentage illiteracy, farmers and unemployed persons, average persons per household, and mean income.
[e] 95% credible interval.
[f] Sum of the 33 types of cancer analyzed.
[g] Not estimated: risk could not be estimated using INLA.

plants, and waste water treatment facilities were not included in our study, since they do not come under IPPC categories 5.1 and 5.2.

Another aspect to consider is that poor communities are forced to live in polluted areas, near waste and industrial sites (Parodi et al., 2005), so it is particularly important to emphasize that the results and conclusions are not simply a reflection of socioeconomic status.

4.1. Incinerators

Incineration is a thermal treatment that generates recognized and suspected carcinogens such as dioxins, arsenic, chromium, benzene, PAHs, cadmium, lead, tetrachloroethylene, hexachlorobenzene, nickel, and naphthalene (European Commission, 2006).

Epidemiologic studies addressing increases in cancer in towns lying in the vicinity of incinerators have provided limited evidence (Porta et al., 2009): the results of a study on incidence of cancer in the environs of 72 incinerators in the United Kingdom (Elliott et al., 1996) which showed statistically significant increases in certain cancers, were critically reviewed (Elliott et al., 2000) and, according to the authors, these results could be affected by different biases, which would in turn mean that the observed effects would not be attributable to incinerator emissions. Nevertheless, studies undertaken in other countries have reported excess risks for hematologic tumors, lung cancer, and some cancers of the digestive system (Biggeri et al., 1996; Comba et al., 2003; Floret et al., 2003; Knox, 2000; Ranzi et al., 2011; Viel et al., 2011).

The results reported in our study show excess risks for all cancers combined and for lung cancer, and in particular, marked increases in risk of tumors of the pleura and gallbladder (men) and stomach (women). Individualized analyses of the installations revealed statistically significant RRs in NHL in the vicinity of installations '467' and '4857' situated in the same town, as well as high excess risks of tumors of the ovary and brain in women in the environs of incinerator '2438'.

4.2. Installations for the recycling of scrap metal and scrapping of motor vehicles

One of the most surprising results of our study is the excess risk detected – statistically significant in all cancers combined, malignant tumors of the stomach, bladder, and thyroid (in men), renal cancer (in men and women), and leukemia (in women), and very close to statistical significance in malignant tumors of the colon–rectum and lung (in men), pleural cancer (in women), and Hodgkin's lymphoma (in the total population) – in the vicinity of installations engaged in the recycling of scrap metal and the scrapping/decontamination of ELVs. The reason for pooling these activities into one category for analysis purposes was because, until relatively recently, these types of waste came within the scope of the Spanish scrap metal sector (Muñoz et al., 2011). In Europe, ELVs have been defined as hazardous waste since 2002, due to the toxic composition of their constituent materials, i.e., used oils, brake liquid, oil filters, absorbent materials, batteries, and fuel. The treatment applied by these types of installations (Joung et al., 2007; Nourreddine, 2007; Santini et al., 2012) generates recognized and suspected carcinogens, such as dioxins, furans, dioxin-like PCBs, lead, chromium, PAHs, cadmium or nickel, and other hazardous substances, such as shredder dusts.

To the best of our knowledge, no epidemiologic studies have been conducted on populations living near these types of installations. Insofar as occupational exposure is concerned, some studies have reported associations between organic dust exposure and gastrointestinal (e.g., stomach) and respiratory problems among workers at material recovery and recycling facilities (Gladding et al., 2003; Ivens et al., 1997). The point should be made, however, that there are studies which have assessed exposure to ionizing radiation and radioactive materials among scrap metal-processing and -recycling workers (Lubenau and Yusko, 1998; Vearrier et al., 2009); these agents are recognized carcinogens for leukemia and thyroid cancer and could be related with significant excess risk of these tumors detected in the proximity of these installations by our study.

4.3. Installations for treatment of used oils and oily waste

These installations include the treatment (cleaning, re-refining, thermal fractionation, gasification and distillation) of all types of used oils and oily waste, and decontamination of equipment contaminated by PCBs, a group of organochlorine substances defined as oil waste by the European Waste Catalogue and Hazardous Waste List (Environmental Protection Agency, 2002). Among the substances released by these installations are recognized and suspected carcinogens, such as dioxins, arsenic, PAHs, benzene, chromium, nickel, lead, naphthalene or tetrachloroethylene.

To our knowledge, there are no epidemiologic or occupational studies of populations living near these types of installations. In this respect, therefore, our study is a pioneer in terms of analyzing the risk of dying due to cancer in the environs of such pollution sources and, indeed, detecting high excess risks for malignant tumors of the connective tissue (total population), pleura, skin, and stomach (men), and vulva and vagina (women). Some of these installations carry out oil re-refining, an activity which may involve significant levels of polycyclic aromatic compounds and PCBs derived from comingling used cutting oils with used engine and transformer oils (Hewstone, 1994). Long-term exposure to certain cutting fluids and mineral oils is known to be associated with an increase in certain occupational cancers, such as those of stomach and skin (DHHS (NIOSH), 1998; Mackerer, 1989). This could account for the excess risks observed in these tumors, given that they were only found in men, and would suggest a possible occupational exposure, assuming that workers' residence was homogenously distributed.

4.4. Installations for the regeneration of spent baths

In metal-scaling operations (i.e., immersion of metals, such as stainless steel, in acid baths to eliminate the layer of oxides formed on their surface after thermal treatments), a large quantity of effluents is discharged from spent baths in Europe every year (Frias and Perez, 1998). These effluents represent a serious environmental problem, as they are a type of waste that contains nitrates, fluorides, acids, and heavy metals (Singhal et al., 2006; Vijay and Sihorwala, 2003). In addition, treatment of such wastes gives rise to exposure to radioactive materials among workers at these plants (Donzella et al., 2007). Our study observed a statistically significant increase in the overall risk of dying from all cancers (men) in the vicinity of these installations, and particularly so in the case of malignant tumors of the stomach (total population), colon–rectum (men), liver (women) and ovary, and close to statistical significance in tumors of the lung and pleura (men).

5. Conclusion

Our results support the hypothesis of a statistically significant higher risk, among men and women alike, of dying from all cancers in towns situated near incinerators and hazardous waste treatment plants, and specifically, a higher excess risk in respect of tumors of the stomach, liver, pleura, kidney, and ovary. Furthermore, this is one of the first studies to analyze the risk of dying of cancer related with specific industrial activities in this sector at a national level, and to highlight the excess risk observed in the vicinity of incinerators and installations for the recycling of scrap metal and scrapping of ELVs, regeneration of spent baths, and treatment of oil and oily waste.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.envint.2012.10.003.

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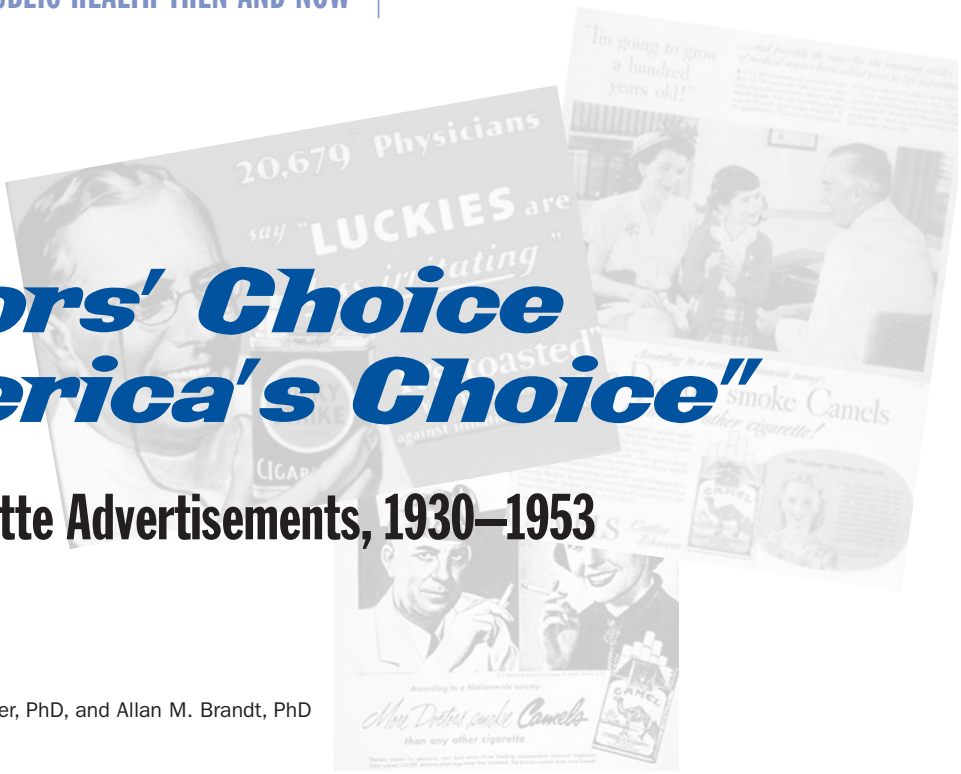

"The Doctors' Choice Is America's Choice"

The Physician in US Cigarette Advertisements, 1930–1953

In the 1930s and 1940s, smoking became the norm for both men and women in the United States, and a majority of physicians smoked. At the same time, there was rising public anxiety about the health risks of cigarette smoking. One strategic response of tobacco companies was to devise advertising referring directly to physicians. As ad campaigns featuring physicians developed through the early 1950s, tobacco executives used the doctor image to assure the consumer that their respective brands were safe.

These advertisements also suggested that the individual physicians' clinical judgment should continue to be the arbiter of the harms of cigarette smoking even as systematic health evidence accumulated. However, by 1954, industry strategists deemed physician images in advertisements no longer credible in the face of growing public concern about the health evidence implicating cigarettes. (*Am J Public Health.* 2006; 96:222–232. doi:10.2105/AJPH.2005.066654)

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IN 1946, THE RJ REYNOLDS Tobacco Company initiated a major new advertising campaign for Camels, one of the most popular brands in the United States. Working to establish dominance in a highly competitive market, Reynolds centered their new campaign on the memorable slogan, "More doctors smoke Camels than any other cigarette." This phrase would be the mainstay of their advertising for the next 6 years. Touting surveys conducted by "three leading independent research organizations," one typical advertisement proclaimed that according to "nationwide" surveys of 113 597 doctors "from every branch of medicine," Camel was the brand smoked by most respondents. It also asserted that this statistic was an "actual fact," not a "casual claim."

In reality, this "independent" surveying was conducted by RJ Reynolds's advertising agency, the William Esty Company, whose employees questioned physicians about their smoking habits at medical conferences and in their offices. It appears that most doctors were surveyed about their cigarette brand of choice just after being provided complimentary cartons of Camels.[1]

Even without the suspect nature of the data used in the "More Doctors" campaign, the frequent appearance of physicians in advertisements for cigarettes in this and many other ad campaigns is both striking and ironic from the vantage point of the early 21st century. Any association between physicians and cigarettes—the leading cause of death in the United States—is jarring given our current scientific knowledge about the relationship of smoking to disease and the fact that fewer than 4% of physicians in the United States now smoke.[2]

In 1930s and 1940s, however, smoking had become the norm for both men and women in the United States—and a majority of physicians smoked.[3] At the same time, however, rising public and scientific anxiety existed about cigarettes' risks to health, creating concern among the tobacco companies. The physician constituted an evocative, reassuring figure to include in their advertisements. In retrospect, these advertisements are a powerful reminder of the cultural authority physicians and medicine held in American society during the mid-20th century, and the manner in which tobacco executives aligned their product with that authority.

Even before modern epidemiological research would demonstrate the health risks of smoking at mid-century, there had already arisen considerable concern about the health impact of cigarette use.[4] Questions of the moral and health consequences of cigarette smoking that had been prevalent at the beginning of the 20th century still lingered. Although many physicians were unconvinced by this older research, some had begun to recognize a disturbing increase in lung cancer, and some had also started to consider the respiratory and cardiovascular effects of smoking. A common theory held that cancer resulted from chronic irritation to the affected tissue, and many wondered whether cigarette smoke "irritated" lung tissue in this manner.[5]

Well aware of these concerns—and their impact on cigarette sales—the tobacco companies devised advertising and marketing strategies to (1) reassure the public of the competitive health advantages of their brands, (2) recruit physicians as crucial allies in the ongoing process of marketing tobacco, and (3) maintain the salience of individual clinical judgments about the health effects of smoking in the face of categorical scientific findings.

These elements would be of growing importance as the health effects of smoking came to be more fully elucidated. One aspect of these promotional strategies was to refer directly to physicians in both images and words. We explored how physicians were depicted in these advertisements and how the ad campaigns developed as health evidence implicating cigarette smoking accumulated by the early 1950s.

EARLY MEDICAL CLAIMS

American Tobacco, the leader in the splashy ad campaigns that had made its Lucky Strike brand dominant by the late 1920s, was the first to mention physicians in advertisements. The physician was just one piece of a much larger campaign on behalf of American Tobacco. As cigarette sales grew exponentially in the United States in the early 20th century, Lucky Strikes had become the preeminent brand largely because of its massive promotional efforts. Company president George Washington Hill worked with ad man Albert Lasker to develop a "reason why" consumers should purchase their brand. With no real scientific evidence to back their claims, American Tobacco insisted that the "toasting" process that Lucky Strikes tobacco underwent decreased throat irritation.[6] In fact, Lucky Strikes' curing process did not significantly differ from that of other brands.

Related campaigns emphasized that "Luckies" would help consumers—especially women, their new market—to stay slim, since they could "Reach for a Lucky instead of a sweet." Along with these persistent health claims, a typical advertisement from 1930 boldly stated that "20,679 Physicians say 'LUCKIES are less irritating'" and featured a white-haired, white-coated doctor with a reassuring smile (Figure 1).[7]

> *"American Tobacco, the leader in the splashy ad campaigns that had made its Lucky Strike brand dominant by the late 1920s, was the first to mention physicians in advertisements."*

FIGURE 1—Advertisement: "20,679* physicians say 'LUCKIES are less irritating.'"
***Source.* *Magazine of Wall Street.* July 26, 1930.**

In this manner, American Tobacco advertisements reflected an awareness of ongoing public concern about the potential health effects of cigarette smoking. Referring to a large number of physicians who they claimed backed up the superiority of Lucky Strikes, the ad text noted in small print that their accounting firm had "checked and certified" this number, independently validating the claim.[8] Their advertising agency, Lord, Thomas and Logan, had sent cartons of cigarettes to physicians in 1926, 1927, and 1928 and asked them to answer whether "Lucky Strike Cigarettes . . . are less irritating to sensitive and tender throats than other cigarettes."

Touting the toasting process in the accompanying cover letter, advertising executive Thomas Logan pointed out the virtues of Lucky Strikes and claimed that

they had heard from "a good many people" that they could smoke Lucky Strikes "with perfect comfort to their throats." American Tobacco used the physicians' responses to this survey to validate their claim that Lucky Strikes were "less irritating," claiming it confirmed their enduring assertion that their "toasting" process made cigarettes less irritating. Toasting, the advertisement went on to explain, was "your throat protection against irritation—against cough."[9] Although there was no substantive evidence that this process of curing tobacco was superior to the methods used by other companies, American Tobacco made the bold claim and tied it to physicians.

By the mid-1930s, Philip Morris, a newcomer to the market, took the use of health claims a step further, designing a campaign that used a new strategy of referring directly to research conducted by physicians. Both in magazines targeted to the general public and in medical journals, Philip Morris claimed that their cigarettes were *proven* to be "less irritating." For example, in a 1937 *Saturday Evening Post* advertisement, Philip Morris's hallmark spokesman, bellhop Johnny Roventini, announced that according to "a report on the findings of a group of doctors . . . when smokers changed to Philip Morris, every case of irritation cleared completely and definitely improved" (Figure 2). The text referred specifically to faithful doctors "day after day. . . [keeping] a record" to "prove conclusively" the decrease in irritation.[10]

FIGURE 2—Advertisement: "A report on the findings of a group of doctors.*" *Source.* *Saturday Evening Post*. October 16, 1937.

These "findings" resulted from an aggressive pursuit of physicians and focused on the concept that adding a chemical to their cigarettes, diethylene-glycol, made them moister and less irritating than other brands. As Alan Blum, editor of the *New York State Journal of Medicine*, explained in his 1983 assessment of cigarette advertisements that had appeared in the journal from 1927 to 1953, Philip Morris—armed with papers written by researchers that the company had sponsored—attempted to use "clinical proof" to establish the superiority of their brand.[11] Specifically, Columbia University pharmacologist Michael Mulinos and physiologist Frederick Flinn produced findings (on the basis of the injection of diethylene-glycol into the eyes of rabbits) that became the centerpiece of the Philip Morris claim that diethylene-glycol was less irritating, although other researchers *not* sponsored by Philip Morris disputed these findings.[12]

This highly successful campaign made Philip Morris into a major brand for the first time.[13] As a 1943 advertisement in the *Saturday Evening Post* proclaimed, Philip Morris provided "[f]ull *reports in medical journals* by men, high in their profession—regularly offered to physicians on request."[14]

These advertisements used physicians and science to make their particular brand appeal to the broader public while at the same time they curried favor with physicians. Company operatives appeared at medical conventions and in physicians' private offices, providing physicians with free cigarettes and reprints of scientific articles on the subject. As a 1936 *Fortune Magazine* profile of Philip Morris & Company made clear:

> The object of all this propaganda is not only to make doctors smoke Philip Morris cigarettes, thus setting an example for impressionable patients, but also to implant the findings of Mulinos so strongly in the medical mind that the doctors will actually advise their coughing, rheumy, and fur-tongued patients to switch to Philip Morris on the ground that they are less irritating.[15]

With careful, deferential appeals to physicians, Philip Morris aimed to gain their approval. The specific positive references to clinical evidence that had appeared in medical journals helped to establish and maintain this connection between physicians and tobacco companies, and between health and cigarettes.

TOBACCO INDUSTRY COURTS DOCTORS

According to a number of accounts, medical professionals—having themselves joined the ranks of inveterate smokers—doubted the connection between smoking and disease after 1930.[16] Although hygienic and physiological concerns continued

to be voiced, clinical medicine claimed that individual assessment and judgment was required.[17] During this era, there was a strong tendency to avoid altogether causal hypotheses in matters so clearly complex. There was—and would remain—a powerful notion that risk is largely variable and thus, most appropriately evaluated and monitored at the individual, clinical level.[18]

According to this logic, some people could smoke without risk to health, whereas others apparently suffered untoward and sometimes serious consequences. As cigarette smoking became increasingly popular in the early decades of the 20th century, medicine offered no new insight into how best to evaluate such variability other than on an individual post hoc basis. If, and when, an individual developed symptoms, a physician might appropriately advise restricting or eliminating tobacco. As a result, rather than being located within the sphere of public health, cigarette use remained within the domain of clinical assessment and prescription. The tobacco industry would actively seek to keep cigarettes within this clinical domain.

For the tobacco companies, physicians' approval of their product could prove to be essential, especially since patients often brought smoking-related symptoms and health concerns to the attention of their doctors. Through advertisements appearing in the pages of medical journals for the first time in the 1930s, tobacco companies worked to develop close, mutually beneficial relationships with physicians and their professional organizations. These advertisements became a ready source of income for numerous medical organizations and journals, including the *New England Journal of Medicine* and the *Journal of the American Medical Association (JAMA)*, as well as many branches and bulletins of local medical associations.[19]

Coming during the Great Depression, the placement of advertisements in medical journals helped to keep medical organizations financially solvent when resources were scarce. Philip Morris praised physicians in these advertisements with taglines like "Every doctor is a doubter" and "Doctor as judge" as they appealed to physicians' expert ability to evaluate the evidence, referring them to scientific articles that they claimed illustrated the superiority of their brand. As one such advertisement explained in its entirety in 1939, "If you advise patients on smoking—*and what doctor does not*—you will find highly important data in the studies listed below. May we send you a set of reprints?"[20]

Not only, then, did physicians' findings help to make the Philip Morris brand appear superior in the eyes of the public, but the company also turned to physicians with great effect. Physicians became, through this process, an increasingly important conduit in the marketing process.

RJ REYNOLDS'S MEDICAL RELATIONS DIVISION

Although Philip Morris may have created this strategy—and gained a leg up in the competitive cigarette market—RJ Reynolds became the leading force in soliciting physicians. Reynolds created a Medical Relations Division (MRD) in the early 1940s that became the base of their aggressive physician/health claims promotional strategy. They directly solicited doctors in a 1942 advertisement that appeared in medical journals describing the MRD. Declaring that "[t]he most significant medical data is derived from the every-day records of practising [sic] physicians," the text asserted "your office record reports in such cases should prove interesting to study."[21]

The MRD, including its longtime director, A. Grant Clarke, was in fact a part of RJ Reynolds's advertising firm, rather than any kind of professional scientific division of the company. The MRD's mailing address was the side door of the William Esty Advertising Company.[22] The work of the MRD focused on promoting Camels mainly through finding and courting researchers to help substantiate the health claims RJ Reynolds made in their advertisements.

In the late 1930s and early 1940s, Clarke—who had no medical or scientific training—corresponded with many researchers who were pursuing questions relating to smoking and health. The MRD financed research that Reynolds then referred to in advertisements. Rather than emphasizing claims of moistness as Philip Morris had done, RJ Reynolds focused on nicotine absorption, insisting that Camels were the slowest burning of all cigarettes. The safety of nicotine—like the issue of chronic irritation—was a source of ongoing concern; Reynolds maintained that nicotine was "the chief component of pharmacologic and physiological significance." Camels' slow burning rate, their advertisements now asserted, decreased nicotine absorption; as a result, Camels offered smokers an advantage over other, faster-burning brands.[23]

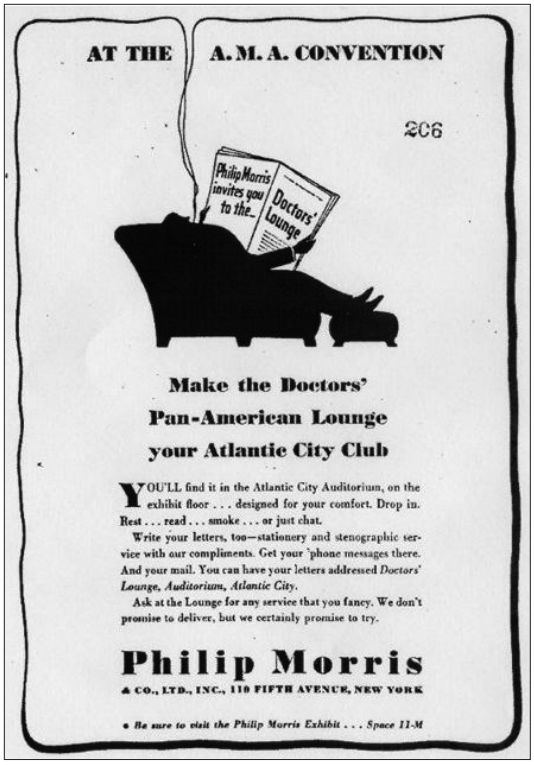

FIGURE 3—Advertisement: "Philip Morris invites you to the . . . Doctor's Lounge."[26]

As they made this claim, RJ Reynolds also asked physicians to use the information when advising their patients. They referred to "a number of reports from physicians who recommend Camels" and called on those reading the advertisement to send in their own clinical experiences and to request copies of medical journal articles from the MRD that proved their assertions. The offer served to legitimate RJ Reynolds's claims. The main article cited did not in fact address Camels specifically, although it did make the claim that slow-burning cigarettes were superior.[24] With no clear knowledge about whether nicotine absorption was even an area that should concern smokers, and with very little data showing Camels' slower absorption, the scientific basis for Reynolds's claim remained obscure.

Nonetheless, such health claims would become the basis for the aggressive recruitment of physicians as allies in the promotion of their products and brands. Tobacco companies' participation in medical conventions provided a clear example of their efforts to appeal to physicians. For example, social commentator Bernard Devoto described the exhibit hall of the 1947 American Medical Association (AMA) convention in Atlantic City, where doctors "lined up by the hundred" to receive free cigarettes.[25] At the 1942 AMA annual convention, Philip Morris provided a lounge in which doctors could relax and socialize. The lounge, an advertisement explained, was "designed for your comfort. Drop in. Rest . . . read . . . smoke . . . or just chat"[26] (Figure 3).

Besides welcoming physicians to the convention, Reynolds touted their scientific research into cigarettes. In an advertisement that appeared in medical journals across the country in the weeks before the 1942 AMA meeting, Reynolds reiterated their claim that "[t]he smoke of slow-burning CAMELS contained less nicotine than that of the 4 other largest-selling brands tested," and continued to direct its health theme at doctors. The advertisement also referred to "the interesting features of the Camel cigarette exhibit," including "the dramatic visualization of nicotine absorption from cigarette smoke in the human respiratory tract" and "giant photo-murals of Camel laboratory research experiments." At a time when laboratory science had garnered especial admiration, the advertisement linked clinical medicine to the authority of investigative science.[27]

Along with directly soliciting physicians, the tobacco advertisements portrayed a glowing image of physicians in both medical journals and popular magazines. In advertisements that were precursors to the "More Doctors" slogan, RJ Reynolds specifically featured dedicated physicians serving their country and its soldiers during World War II. As a 1944 advertisement that appeared in *Life Magazine* entitled "Doctor of Medicine . . . and Morale" illustrated, doctors on the front received hero status:

> He wears the same uniform. . . . He shares the same risks as the man with the gun. . . . Yes, the medical man in the service today is a fighting man through and through, except he fights without a gun. . . . [H]e's a trusted friend to every fighting man. . . .[H]e well knows the comfort and cheer there is in a few moments' relaxation with a good cigarette . . . like Camel . . . the favorite cigarette with men in *all* the services.[28]

With this and similar advertisements, the positive place that physicians held in American culture was both exploited and underlined by RJ Reynolds's advertising scribes. Linking physicians to wartime patriotism further elevated their status and, with it, Camel cigarettes.

THE "MORE DOCTORS" CAMPAIGN

When the "More Doctors" campaign began in January 1946, it also focused on the respected and romantic image the medical profession had achieved in American society.[29] Featuring 6 illustrations of physicians with patients—in the laboratory or sitting back with cigarette in hand—this first advertisement personalized the physician for the readers of such popular magazines as *Ladies' Home Journal* and *Time*.[30] Prefaced with the bold statement that "Every doctor in private practice was asked:—family physicians, surgeons, specialists . . . doctors in every branch of medicine," the

advertisement touted the thoroughness of their survey and insisted that "yes, your doctor was asked . . . along with thousands and thousands of other doctors from Maine to California."

By linking their depiction of physicians to the consumer's own physician, Reynolds brought immediacy to their claims. Any fears that smoking might be harmful were also easily contradicted by the physician's being a smoker himself. Admirable, forthright physicians—including the consumer's own—had "named their choice," and that choice, the advertisement insisted, was Camels, hands down.

Even though a few of these advertisements did appear in print, the Reynolds advertising department soon realized that they might have overstepped their evidence. With the Federal Trade Commission already challenging suspected health claims in cigarette advertisements, RJ Reynolds toned down their copy, quickly shifting their claim to "113,597 physicians" surveyed rather than *all* physicians.[31]

At least some individual physicians questioned the original claim. In a letter to Howard T. Behrman, a physician who had requested "more specific information concerning the survey of physicians' smoking preference," RJ Reynolds advertising executive W. T. Smither assured him that the surveying had been thorough and scientific. Explaining that the question about brand preference had been embedded in a survey that included less relevant topics—such as medical journals, medical conventions, and numerous consumer products—Smither emphasized how 3 independent surveys had garnered "similar findings, and in doing so, served to confirm the accuracy of each other."[32]

Beyond the questionable methods used to gather data, Reynolds was also careful how they described the survey findings in advertising copy, making sure to avoid conflating doctors' choice of a cigarette with any belief on their part that Camels were healthier. In their advertisements, they asserted, "Doctors smoke for pleasure just like the rest of us."[33] Internally, Reynolds's advertising executives cautioned William Esty, their advertising company, to be careful of what they claimed, insisting that "in no way [should] the copy. . . intimate that doctors recommended smoking of CAMELS, [or] that CAMELS are good for health."[34] This cautionary approach reflected the growing industry concern about potential regulation and litigation.[35]

Even so, the "More Doctors" campaign resonated effectively with American cultural values about contemporary medicine. Throughout 1946, the slogan flooded print, radio, and television media. Doctors were often idealized, as in the 1946 advertisement "I'll be right over!" Here, a middle-aged physician, in bed in his pajamas, telephone in hand, is about to grab the black bag lying ready on his bedside table and make a middle-of-the-night visit to a patient in need:

> 24 hours a day your doctor is "on duty." . . . [I]n his daily routine he lives more drama, and displays more devotion to the oath he has taken, than the most imaginative mind could ever invent. And he asks no special credit. When there's a job to do, he does it. A few winks of sleep. . . a few puffs of a cigarette. . . and he's back at the job again.[36]

This neighborhood family physician is saintly and deserving of trust, representing (as another 1946 advertisement explained) "an honored profession . . . his professional reputation and his record of service are his most cherished possessions."[37] The importance of professional autonomy loomed large, and the industry was eager to sustain this view. As physicians geared up to fight the Truman administration's national health insurance proposals, their image as loyal and deserving of respect was especially important.[38]

Along with providing images of professional trustworthiness and dedication, the "More Doctors" ad campaign also exploited the popular faith and admiration of medical science and technology. In one such "More Doctors" ad, a 5-year-old girl sits next to her mother in a doctor's office and proclaims, "I'm going to grow a hundred years old" to the kindly man in white (Figure 4).

FIGURE 4—Advertisement from the Camels "More Doctors" series: "I'm going to grow a hundred years old!"
Source. Good Housekeeping. **July 1946.**

Referring to the "amazing strides in medical science [that] have added years to life expectancy," the advertisement goes on to "thank medical science for that. Thank your doctor and thousands like him. . . toiling ceaselessly. . . that you and yours may enjoy a longer, better life."[39] With medical advances having captured popular imagination, connections drawn between scientific discovery and Camel-smoking doctors added to the appeal of their cigarette of choice.[40]

Figure 5—Advertisement: "How mild can a cigarette be?"
***Source. Ohio State Journal of Medicine.* July 1949;45:670.**

MEDICAL AUTHORITY AND TOBACCO

After the initial onslaught of heroic physicians and medical miracles in 1946, the "More Doctors" advertisements in 1947 and 1948 continued to remind readers about the survey as the focus of the advertisements shifted. The main slogan of one such campaign was "Experience is the best teacher." In this series of advertisements, RJ Reynolds explained that the cigarette shortage created by the war had forced many to smoke whatever brands were available, and this experience, they claimed, had made the superiority of Camels' quality clearly evident. The smoker was able to tell the difference between brands, and such "experience" translated to other areas where someone might have know-how. When the slogan appeared in magazines like *Life* and *Saturday Evening Post*, the "experience" cited might be that of a talented celebrity athlete able to discern quality in his or her sport. In medical journals, the references were to famous scientific researchers. These advertisements championed physicians and medicine and reminded their audience again that "More doctors smoked Camels" as they also continued to praise science.[41]

But the idea of "experience" also figured into another prevalent theme communicated in RJ Reynolds's advertising—that of individual authority, both the physician's and the individual consumer's. The question of throat irritation so central to many 1920s and 1930s ad campaigns again emerged here as RJ Reynolds introduced a "mildness" theme. With the central claim that Camels did *not* irritate the throat, Reynolds featured both the physician-researcher and the everyday smoker to convince readers of Camels' mildness.

In July 1949 issues of both local and national medical journals, RJ Reynolds asked, "How mild can a cigarette be?" In answering this question, the advertisement juxtaposed a "doctors report"—illustrated with a physician, cigarette in hand and head mirror strapped around his brow—with a "smokers report"—illustrated with a smiling "Sylvia MacNeill, secretary." Physicians, the advertisement explained, had concluded after scientific investigation that there was "not one single case of throat irritation" from smoking Camel cigarettes. In fact, "noted throat specialists" had conducted "weekly examinations" of patients in making this determination. Reynolds used this depiction of careful, clinical observation to substantiate their health claim (Figure 5).[42]

The advertisement went beyond medical authority, however, asserting that smokers didn't even have to take their physicians' word for it. Instead, they could take their "own personal 30-day test," as Sylvia MacNeill had done. She concluded that she "knew" that "Camels are the mildest, best-tasting cigarette I ever smoked." Advertisements in popular magazines took smokers' ability to judge for themselves even further, with Elana O'Brian, real estate broker, declaring in a typical example, "I don't need my doctor's report to know Camels are *mild*." The advertisement underlined her assertion with photos of 6 other smokers from various walks of life under the heading "Thousands more agree!"[43]

In another example, Anne Jeffreys, a stage and screen star, insisted, "The test was fun and it was *sensible*!" Parallel to earlier solicitation of physicians' opinions, in this series of advertisements RJ Reynolds requested that smokers determine the safety of Camels on their own and praised their acumen. With some advertisements calling on

smokers to "*Prove it yourself!*" and even guaranteeing a money-back guarantee for dissatisfied customers, Reynolds insisted on the superiority of their product.[44] These advertisements worked to subvert the emerging population-based epidemiological findings by emphasizing the primacy of "individual" judgment.

By 1952, advertising copy went beyond the typical individual smoker to emphasize the sheer volume of people who chose Camels as their cigarette. Highlighting that Camel was "America's most popular cigarette *by billions*," the ad copy mentioned that "long before Camel reached those heights, repeated surveys showed that *more doctors* smoke Camels than any other cigarette."[45] The cigarette's popularity in itself became a selling point: how could so many people be wrong? And physicians' cigarette choice served to confirm this popularity. As the heading of a similar advertisement explained, "The doctors' choice is America's choice."[46]

THE DISAPPEARING DOCTOR

Ultimately, however, the use of physicians in Camel advertisements could not be sustained as the health evidence against cigarettes accumulated. When disturbing scientific results connecting lung cancer and cigarettes began to emerge, Camel advertisements shifted away from physicians' judgment and authority. In 1950, the publication of the now-famous work of Evarts Graham and Ernst Wynder in the United States—as well as that of A. Bradford Hill and Richard Doll in the United Kingdom—showed that there was cause for alarm.[47] The reporting of their findings connecting lung cancer to cigarette smoking in national magazines like *Time* and *Reader's Digest*—and the corresponding declines in sales and stock prices—forced tobacco executives to assess strategies for responding to growing medical and public concerns about their product.[48]

By 1953, when Wynder, Graham, and their colleague Adele Croninger published laboratory findings confirming that cigarettes were carcinogenic, scientific findings constituted a critical threat to the industry.[49] Tobacco executives were well aware both of these findings and of the public attention they were receiving, and their statements and actions reflected an understanding that this new scientific evidence constituted a full-scale crisis for their corporations.

Most notably, company executives realized that they would have to work together in the face of the scientific evidence. Although each company still sought an advantage over its competitors, the new health evidence threatened the future of the entire industry. In December 1953, the tobacco executives met to devise a joint strategy. They hired prominent public relations firm Hill & Knowlton to aid in this effort. As a planning memo makes clear, health claims were considered to be no longer viable. According to Edward Dakin, a Hill & Knowlton executive, it would be critical to

> Develop some understanding with companies that, on *this* problem, none is going to seek a competitive advantage by inferring to its public that *its* product is less risky than others. (No claims that special filters or toasting, or expert selection of tobacco, or extra length in the butt, or anything else, makes a given brand less likely to cause you-know-what. No "Play-Safe-with-Luckies" idea—or with Camels or with anything else.)[50]

Hill & Knowlton's advice was that the industry as a whole must desist from health claims that had been a centerpiece of the advertising that featured physicians. Such claims, the agency now contended, would now draw attention to the "health scare," as they professed to call it.[51]

> *"Tobacco executives were well aware both of these findings and of the public attention they were receiving, and their statements and actions reflected an understanding that this new scientific evidence constituted a full-scale crisis for their corporations."*

In popular magazines, the last notable reference to doctors in an advertisement came in 1954. After the other tobacco companies had left such marketing techniques behind, Liggett and Myers (which had declined participation in the joint industry program directed by Hill & Knowlton) made the claim that their L&M filter cigarette was "Just what the doctor ordered!" In a typical advertisement that appeared in a February issue of *Life* magazine, Hollywood star Fredric March made this assertion after having read the letter written by a "Dr Darkis" that was inset into the advertisement. Darkis explained in this letter that L&M filters used a "highly purified alpha cellulose" that was "entirely harmless" and "effectively filtered the smoke" (Figure 6).

Dr Darkis was in fact not a medical doctor at all but a research chemist, yet another example of misrepresentation in a tobacco ad.[52] More significantly,

this use of implicit doctor endorsement of cigarettes would not occur again in American advertising after this campaign. Much in the way that the industry had used doctors to reassure smokers in the 1940s, filter cigarettes were becoming the industry's new strategy for appealing to consumers, whose concerns about the health risks of smoking would be repeatedly confirmed by new research studies. In 1950, filter cigarettes were 2% of the US cigarette market; by 1960, they were 50%.[53]

In medical journals, the last-gasp attempt by a tobacco company to ally itself with physicians came in 1953, when the Lorillard Company appealed to physicians as they promoted their new filter cigarette, Kent. These advertisements queried, "Have you tried *this* experiment, doctor?"

FIGURE 6—Actor Fredric March in an advertisement for L&M Filters: "This Is It."
Source. Life Magazine. **February 22, 1954.**

and "Why is it, doctor, that one filter cigarette gives so much more protection than any other?" One advertisement mentioned how "thousands" of physicians at a recent AMA convention witnessed "a convincing demonstration. . . [of] the effectiveness of the MICRONITE FILTER" and included photos of the experiment demonstrated there. In their marketing of Kent, Lorillard had created a campaign reminiscent of those designed by Philip Morris and RJ Reynolds in the 1930s and early 1940s.[54] Just as in those earlier advertisements, Lorillard called on physicians to interpret scientific results using their individual, clinical judgment. But the swift and vehement reaction to these advertisements clearly illustrated how the social and scientific climate had shifted. A 1954 *JAMA* editorial labeled the reference to physicians and the AMA convention an "unauthorized and medically unethical use of the prestige and reputation of the American Medical Association."[55] No longer could tobacco companies count on physicians to serve as public advocates of their product.

In fact, in 1953 *JAMA* had decided to stop accepting cigarette advertisements in its publications and banned cigarette companies from exhibiting their products at AMA conventions.[56] After conducting its own survey of physicians, the AMA explained in a letter to tobacco companies that "a large percentage of physicians interviewed expressed their disapproval" of cigarette advertisements in medical journals. Other *JAMA* advertisers had come to dislike having their products appear next to cigarette advertisements as well.[57] With the AMA publicly condemning the Kent ad campaign in 1954 as "hucksterism," it became even more clear to tobacco companies that the purported allegiance with physicians was no longer feasible or effective.

One additional indicator of the growing medical disdain for cigarettes was the very fact that many physicians who followed the emerging health evidence began the process of giving up smoking. According to one study of physicians' smoking practices in Massachusetts, nearly 52% had reported being regular smokers in 1954 (over 30% reported smoking at least a pack per day); just 5 years later, only 39% were regular smokers. Additionally, only 18% now reported consumption of a pack or more per day.[58]

Although the industry would continue to solicit physicians with materials disputing the relationship between smoking and disease and would also seek out physicians who doubted the harmfulness of cigarettes in order to undermine emerging scientific findings, such efforts would be greeted with rising skepticism.[59] The era of explicit use of physicians and health claims to promote smoking had ended even though the AMA would not publicly acknowledge the harms of cigarette smoking until 1978.[60] The smoking physician had become a visual oxymoron. The industry would turn to new images and more sophisticated strategies to hawk their dangerous product. ■

About the Author

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Endnotes

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Toxic Fallout

Waste Incinerator Bottom Ash in a Circular Economy

Research Report - January 2022

gaia
Global Alliance for Incinerator Alternatives
Global Anti-Incinerator Alliance

ZERO WASTE EUROPE

Abstract

Bottom ash is fallout from the grate of mass-burn waste incinerators. Large quantities are produced and this residue has negative value. Visible proportions of sand, glass, and stones make it appear, on the surface, to be low hanging fruit for use in a circular economy; but bottom ash also contains appreciable quantities of toxic 'high level of concern' elements and persistent organic pollutants.

A secondary 'fallout' occurs when these substances leach from bottom ash into its surroundings across a range of conditions and timescales. The waste incineration industry fails to mention these facts when advertising bottom ash as a 'green' building material. In comparison to direct airborne pollution from waste incinerators, bottom ash has gone somewhat under the radar, making it ripe for greenwash.

This report uses independent empirical research to evidence that incinerator bottom ash is insidiously hazardous and underregulated. Risk is heightened by the fact that testing methods for its use as a building material are outdated. A list of fifteen concerns for public health and safety is provided in relation to the use of waste incinerator bottom ash in cement-based products and as road/pathway aggregate. Calls for the support of its use within a circular economy are premature, and, as per the precautionary principle, all ongoing usage should cease. Examination of independently analysed bottom ash provides a diagnostic on the operational steady state of waste incinerators, incidentally raising concerns about operational compliance with emissions legislation and the capacity of incinerators to produce benign bottom ash when fed with municipal solid waste.

1 Introduction

In Nature's biosphere, something's discarded effluence is something else's resource. All naturally occurring 'waste' is readily consumed in the efficient process of elemental recycling that operates at the Earth's surface. Within moments, creatures set about its consumption in earnest. Waste does not occur in nature because nature abhors inefficiency.

In contrast, civilisation in the 21st century has implemented an economic system which is proactively inefficient in terms of how it utilises its natural resources (the finite budget of chemicals that form the Earth's lithosphere and biosphere and the energy contained within their chemical bonds) by seeking to expedite disorder and create temporary, localised financial gain. In doing so, it has taken human endeavour above and beyond stability - i.e. the natural recycling of elements within a finite budget –; and has thrown it into the unstable realm of a throwaway society where, in an attempt to satiate this requirement, greater consumption of goods, services, and fuel must occur in greater volumes than the year before, thus creating increasingly larger amounts of waste.

Prompted by numerous environmental concerns directly arising as a result of this system, and of the logic to transition away from it, a number of ideas have been proposed which, rather than directly challenging the fundamentals of the system, suggest a reconciliation. One of these is 'sustainable development' (Spaiser et al., 2017). Another is 'circular economy' (Ellen MacArthur Foundation, 2014). Waste incineration is considered to be outside of the circular economy (ibid.). Reasons are that it is a destructive process which provides 'back-end pull' for waste generation accentuated by contractual lock-ins (Muznik, 2017).

Currently, the European Union (EU) is examining whether the use of modern waste incinerator bottom ash could be worthy of investment support within a future circular economy. The matter is being discussed as part of a wider EU Taxonomy (EU, 2020). To be aligned, suitable activities must make a "substantial contribution" to at least one of six objectives:

1. Climate change mitigation,
2. Climate change adaptation,
3. Sustainable use and protection of water and marine resources,
4. Circular Economy Transition,
5. Pollution prevention and control, and
6. Protection and restoration of biodiversity and ecosystems;

while simultaneously they must *"do no significant harm"* to any of the other objectives; in other words, progress towards one objective must not be made at the expense of another. Compliance is assessed against specified "technical screening criteria", which require that evidence is *"science based"* [sic], and *"developed via a robust methodology"* (PSF, 2021).

It is not the objective of this report to consider the de-merits of waste incineration within the circular economic model. It is its aim to provide evidence against the aforementioned criteria: specifically, the use of bottom ash in both 'unbound' aggregates (i.e. for roads and paths) and bound composites (e.g. cement based products like concrete and blocks). The topic has wider relevance to the legislative, permitting, and planning sectors where claims are put forward by the incinerator industry that bottom ash can have *"many applications"*, can be *"carbon negative"*, and even that it can assist with *"Climate change adaptation and greenhouse gas emissions"* (Powerfuel, 2020).

In this report, the hazard (if any) posed by the use of incinerator bottom ash is assessed using independent, empirical, peer-reviewed scientific literature. Specifically, the total concentrations of toxic substances in bottom ash and their propensity to leach out into the environment from subsequent products and applications. Current regulatory and testing safeguards within a European context are investigated, while drivers and motivations for the proposed use of bottom ash are also discussed.

2 Background to Bottom Ash

In the mid-1800's, prior to the first municipal solid waste incinerator (MSWI) patent (Clark, 2007) societal waste comprised mainly dust, ashes, and cinders (ca. 80% - the residue from fire grates), along with lesser quantities of vegetative matter, excrement, bones, and animal carcasses; plus minor amounts of ceramics, rags, paper, and metals (Tanner, 2006). This detritus was frequently piled up within the boundaries of rapidly expanding urban areas, and these refuse heaps were considered to be of some value (Dickens, 1865). People lived among them, scavenging was permitted, and in one city at least a fee was charged for the privilege (Melosi, 1973). Once all 'valuables' had been removed, the leftover ash and cinders were commonly used as a sub-base for paths and carriageways; indeed, in 1848, the whole of London's Great Dust Heap (Figure 1) was reportedly sold to Russia for building the streets of Moscow (Tilley, 2014).

Figure 1. King's Cross, London: the Great Dust-Heap, next to Battle Bridge and the Smallpox Hospital. Watercolour painting by E. H. Dixon, 1837 (Wellcome Collection, no date).

Modern incinerator bottom ash is markedly different from the ash and cinders which were used as a road base in the 1800s. Municipal solid waste (MSW) now includes ubiquitous quantities of plastics and their additives, along with plastic/metal composites such as printed circuit boards and other petrochemically coated substances like paper, packaging, and waste wood (Conesa et al., 2021). A recent report listed over 2400 substances in waste plastic that are identified as of potential concern because they meet one or more of the persistence, bioaccumulation, and toxicity criteria in the EU (Wiesinger et al., 2021).

The majority of modern waste incinerators are mass-burn, grate-fired systems, and the most massive quantity of residue that they produce is 'fallout' from the main grate - 'bottom ash'. Though incinerators are not built to harvest bottom ash, their purpose is to create it: the word's etymological route is a process for 'converting to cinders'. Some incinerators recover a quantity of the energy contained in waste, so-called Waste-to-Energy (WtE) or Energy-from-Waste (EfW) plants. But the waste to electricity efficiency is very low, at $\eta \leq 0.3$, essentially meaning that at least 70% of the chemical functionality in waste is lost in the process of 'converting to cinders' (Neuwahl et al., 2019).

In modern incinerators, approximately a third of the input waste is incombustible or goes uncombusted (Bielowicz, et al. 2021). This equates to about a quarter of the input mass becoming bottom ash (Bunge, 2019; Hulgaard and Vehlow, 2011). The balance - a smaller amount of solid residue - becomes entrained in the combustion gases and is either emitted into the atmosphere (Particulate Matter Research Group, 2019) or captured by gas cleaning modules (Vehlow, 2015). These entrained particles are termed fly ash and air pollution control residues (APCr) and are not part of this report.

The focus of legislation in Europe has been to minimise these airborne emissions, lately implemented via the Industrial Emissions Directive (IED) (EU, 2010). This requires that the post-combustion gas [author's emphasis] must be subjected to at least 850°C for a minimum of 2 seconds even under the most unfavourable of conditions, and that the bottom ashes/slag have total organic carbon (TOC) content of <3 wt% or their loss on ignition (LOI) is less than 5 wt%. Limit values exist only for pollutant concentrations in the airborne emissions and APCr system wastewater. The combustion environment above an incinerator grate is a hostile one to monitor and, though little is known about localised variations, temperatures above the grate are believed to oscillate around 900°C (Bunge, 2019).

At the macro-scale, bottom ash is mostly (between 50 – 97%) amorphous material, stones, shards of glass, chunks of metal, and sandy grit (Buchholz and Landsberger, 1995; Caviglia et al., 2019). The amorphous fraction is often referred to as 'slag' and 'clinker'; a product of high temperatures in the combustion zone at which substances melt, aided by elements from groups 1 and 2 of the periodic table which are fluxing agents (Miles et al., 1995). The words 'slag' and 'clinker' are often used as synonyms for bottom ash. Chemically, bottom ash has a pH in the 11-12 range (Bunge, 2019). Major constituents (ca. 90%) are oxides of sulphur (S), silicon (Si), calcium (Ca), iron (Fe) and aluminium (Al) bound, among which are numerous minor elements from different chemical groups, many of which are very toxic (Simon et al., 2021; Vateva, and Laner, 2020). Bottom ash also has some pure metals and a fraction of these are commercially extractable (Bunge, 2019).

Commercial extraction of metals is influenced by how bottom ash is temporarily stored upon discharge. Some incinerators have quenching systems (a water-filled tank) while others operate dry capture, often with a period of open air stockpiling known as weathering or ageing, each of which can alter bottom ash form and chemistry. Both ferrous (Fe) and non-ferrous (NFe) metals are extractable, but this refers to only unoxidised constituents (i.e. pure, native metals) and not to metal oxides which are grouped with the mineral constituents. Full recovery of all metals is not possible, with the remainder along with metal oxides left within what is sometimes called the 'mineral fraction' of bottom ash. This 'left-over' bottom ash residue is the subject of this report.

3 Method and Hazard Identification

Research was framed by two hypotheses:

1. The use of incinerator bottom ash could substantially contribute to the transition to a circular economy; and
2. Its utilisation will do no significant harm.

The research methodology was a literature review, with papers selected by date of publication from 2019 onwards, and only those which contained results derived from empirical research. Datasets were limited to samples of bottom ash produced by the incineration of MSW, i.e. household and commercial/industrial waste; studies reporting on special 'hazardous waste' incinerators were excluded. Also excluded were publications either directly commissioned by industry, co-authored by, or co-funded by industrial sponsors. The scope was set within Europe, defined geographically; but, for organic substances, it was extended to include empirical studies from other continents which evidenced compliance with EU legislated operational minima and/or Best Available Techniques (BAT).

The potential hazards of bottom ash are a function of its intrinsic chemistry. Further hazards are created by the interaction of bottom ash with the chemistry of its external environment when applied in product form. Risk is assessed also as a function of the legislative framework of safeguards, if any, which govern product manufacture and point of use. Literature commonly expresses the chemical hazard by two metrics: a) the 'total concentration', which is the quantity per unit mass of specific elements and compounds; and b) the mobility of these elements and compounds from bottom ash, termed 'leaching concentration', and defined as the mass of substance per unit volume of liquid used to irrigate the sample.

In this report, chemical hazard identification was based on EU REACH (Registration, Evaluation, Authorisation and Restriction of Chemicals). All the substances listed in Table 1 were present in the bottom ash as reported by the studies which comprise this report. All are considered as High Level of Concern by fulfilling one or more of hazard criteria under EU REACH (namely: very bioaccumulative; carcinogenicity; mutagenicity; reproductive toxicity; endocrine disruption; specific target organ toxicity upon repeated exposure; and chronic aquatic toxicity), combined with the large volumes produced, as identified by Wiesinger et al. (2021). Phase change data is provided in Table 1 so that inferences can be drawn on the conditions inside the waste incinerator and by the presence and form of the substances in bottom ash.

Table 1. Selection of elements* found in MSWI bottom ash from studies in this report, and which are considered as High Level of Concern (Wiesinger et al., 2021). * = Cl^- is an ion and SO_4^{2-} is an ionic compound. ** Halkidiskis et al., 2019; Wiesinger et al., 2021.

Element	Melting Point	Boiling Point	Origin in MSW**
Arsenic (As)	Sublimes at 616°C		Used in electronics and glass, wood preservative. Biocide in plastics.
Barium (Ba)	729°C	1637°C	Antioxidant, colourant, filler, heat and UV stabiliser in plastics.
Bromine (Br)	-7°C	59°C	Major constituent of flame retardants in plastics, foams and textiles.
Cadmium (Cd)	321°C	756°C	Heat stabiliser, antioxidant and pigment in plastics. Used in metal plating and batteries.
Cobalt (Co)	1495°C	2870°C	Catalyst and pigment in plastics. Widely used in magnets and metal alloys.
Chloride (Cl^-)*	n/a	n/a	Plasticiser, heat stabiliser, colourant, antioxidant and catalyst in plastics. Major constituent of polyvinyl chloride (PVC). Wood preservative.
Chromium (Cr)	1860°C	2672°C	Catalyst and pigment in plastics. Used in metal plating.
Copper (Cu)	1084°C	2567°C	Biocide and pigment in plastics. Present as wiring in most electrical goods.
Lead (Pb)	334°C	1740°C	Colourant, antioxidant, UV and heat stabiliser in plastics. Present in batteries, metal goods, glass, electronics.
Mercury (Hg)	-39°C	357°C	Catalyst, colourant, cross-linking agent, filler and biocide in plastics.
Molybdenum (Mo)	2617°C	4612°C	Catalyst, cross-linking agent and flame retardant in plastics.

Nickel (Ni)	1453°C	2732°C	Catalyst and biocide in plastics.
Antimony (Sb)	631°C	1635°C	Main use is as a flame retardant in plastic, Also plastic catalyst, antioxidant and pigment.
Sulphate (SO_4^{2-})*	n/a	n/a	Filler, colourant, heat and UV stabiliser in plastics.
Tin (Sn)	232°C	2270°C	Biocide and antioxidant in plastics. Used as flame retardant, and in metal plate, glass, ceramics.
Vanadium (V)	1887°C	3377°C	Antioxidant in plastic. Also a lubricant in plastic manufacture. Level of concern = vanadium oxide.
Zinc (Zn)	420°C	907°C	Multiple uses as plastics additive: filler, heat stabiliser, flame retardant, slip agent, pigment.

In addition, a number of organic chemical groups are present in bottom ash and are also considered as hazardous. These are commonly known as Persistent Organic Pollutants (POPs) and most are listed within the Stockholm Convention, though some are not. POPs are long-lasting, toxic, known to bio-accumulate within higher trophic levels, and each may comprise many species with similar properties (known as congeners). These are detailed separately within §5.2.

4 Results: Legal Framework of Standards and Testing Methods

Currently in Europe, the use of waste incinerator bottom ash as a building material is fragmented (Blasenbauer et al., 2020). Some countries (Norway) prohibit its use, while one (The Netherlands) uses all that it creates. Some nations use none even though it is permitted, while many have regional variations which range in usage from 0 ≤ wt % ≤ 100. Some have use-specific requirements (e.g. in Italy bottom ash can be used in cement, bricks and expanded clay without treatment or testing, but for road use leaching tests specify only the presence of some heavy metals (Caviglia et al., 2019). Five out of twenty six EU countries provide no regulation at all, while two (Ireland and Luxembourg) prohibit domestic use but allow export (Blasenbauer et al., 2020).

The collective European Standards for building aggregate and cement-based products (EN12620, EN13139, EN13043, and EN13242) employ the general term ***"dangerous substances"***. This refers to total concentration only (no leachate), is unspecific, and is marked as ***"informative"***, rather than ***"normative"***, for the purposes of linking with other EU Directives, none of which provide adequate safeguards - Art. 53 of EU (2010) discusses the minimisation of residues, stating that ***"appropriate tests"*** should be carried out to establish their polluting potential. The matter is delegated to countries, yet no harmonised testing method exists, an issue that was identified more than a decade ago (Blasenbauer et al., 2020).

Of those countries which require leaching tests for bottom ash prior to use as a building aggregate, details are summarised below (Blasenbauer et al. 2020):

- Eleven countries use batch tests, based on EN-12457 - a method not designed for bottom ash as a building aggregate – and which uses water as leachant[1] (EN,2002/2003). This varies between countries via the following parameters:
  - the volume of eluent per mass of sample, quantified as liquid to solid ratio (L/S). Countries use either 2 or 10 $l.kg^{-1}$, and one country (Belgium) requires a two-stage test. All have a duration of 24 hours . Seven countries apply this test to grain sizes of <4mm, while three countries (plus one region in Belgium) apply it to grain sizes <10 mm.
- Four countries use a column or percolation test, with an elution rate from 0.1 to 10 $l.kg^{-1}$. Particle size is not specified in Finland and Sweden, while it is set at <4 mm in The Netherlands and two regions of Belgium. Test duration is not specified.
- The Netherlands are unique in also having a monolith test which applies only to bound aggregates. This involves placing a sample of bottom ash in water for 64 days. Particle size is not specified and L/S varies depending on the monolith size.

[1] 'Eluent' or 'leachant' is the known mass of liquid which flows through, or is agitated within a close vessel in contact with, the known mass of bottom ash, in turn capturing some portion of the toxins. Following the analysis, the quantity of toxins retained by the 'eluate' are determined as their test specific 'leaching/leachate concentration'.

The broad opinion among scientific authors is that the leaching tests are inadequate and provide insufficient safeguards. For example (Tiberg et al., 2021):

"To what extent the leaching from and presence of metals in the mineral fraction of bottom ash poses a threat to the environment and how this should be evaluated is still not entirely clear."

And, (Simon et al., 2020):

"The standard leaching tests mobilize only small amounts of the complete reservoir of certain substances in incinerator bottom ash."

Further independent observations are provided in the following section which compares empirical bottom ash analyses with the various European limit values for total concentration and leaching concentration. Information on limit values was obtained from Blasenbauer et al. (2020) and Glauser et al. (2021) unless otherwise stated. No comment is made on the suitability of limit values because (Blasenbauer et al., 2020):

"It cannot be concluded whether a specific limit value is too high or too low, since it is unknown how limit values were defined in each country."

5 Results: Empirical Research

5.1 Potentially Toxic Elements

Table 2. shows the studies which met the scope of the literature review. These were from six European countries. Note in particular in Table 2 the detail of sample preparation (ageing, metals extraction, weathering).

Table 2. Datasets which report empirical studies analysing potentially toxic elements in MSW incinerator bottom ash.

Reference	Sample provenance	Sample preparation	Method	Analyte
Bielowicz et al., 2021.	WtE plant, Poland.	Stockpiled indoors for two weeks, post Fe and NFe metals extraction. Sampling of 20kg from 350-400kg each week, analysis over 36 weeks.	Leaching to EN12457 with L/S of 10 $l.kg^{-1}$.	Ba, Cl^-, Cr, Cu, Mo, Pb, Sb, SO_4^{2-}, Zn.
Glauser et al., 2021	Two Swiss MSWI/bottom ash treatment plants: 1 wet discharge, ageing and metals extraction; 2. 'enhanced' dry treatment and metals extraction.	1 tonne, post-metal, sampled at 5-6 times over two days from each plant. Pieces of metal and unburned matter >5 mm removed. All size fractions crushed to <0.25 mm.	Leaching tests: 1. Batch at 10 $l.kg^{-1}$, a) with de-ionized water, and b) with CO_2 saturated water (reduced pH). 2. Column with de-ionized water at L/S of $0.1 \leq l.kg^{-1} \leq 10$.	Cl^-, Cu, Pb, Zn (leachate) As, Cl^-, Cd, Cr, Co, Cu, Ni, Pb, Sb, Sn, V, Zn (total concentration).
Mantovani et al., 2021.	WtE plant, Italy	Five samples totalling 30kg taken over five days. Dried for 24 hours then sieved to various grain sizes.	Total concentrations only.	Numerous.
Simon et al., 2021.	BA treatment facility, Germany.	Fe and NFe separated, no ageing. Wet separation and	6 year leach test experiment using	Cl^-, Cr, Cu, Mo, Sb, SO_4^{2-}, V.

		sieving to grain size range of 25 ≤ mm ≤ 45.	simulated rainwater in a lysimeter.	
Tiberg et al., 2021.	Six different Swedish WtE plants.	Metal extraction then aged outdoors for at least four months or treated to pH 10.	pH dependent leach test on grain sizes: 5.5 ≤ mm ≤ 8.5.	Al, Cu, Fe, Zn.
Kalbe and Simon, 2020.	Bottom ash treatment facility, Germany.	Post Fe and NFe metals extraction, no ageing. Wet separation and sieving to size range 25 ≤ mm ≤ 45.	Four different leaching tests: simulated rainwater in a lysimeter of 2.96 $l.kg^{-1}$, column up to 9.6 $l.kg^{-1}$, batch tests of 2 $l.kg^{-1}$, and 10 $l.kg^{-1}$.	Numerous
Vateva and Laner, (2020)	German MSWI plant (wet discharge system).	1900 kg taken in 9 days over a four week period. Compared 'as received', with 4 month aged, grain sizes: <0.063 ≤ mm ≤ 31.	Batch leaching test EN12457 at 10 $l.kg^{-1}$.	As, Cd, Cl^-, Cr, Cu, Hg, Ni, Pb, SO_4^{2-}, Zn.
Alam et al., 2019a	Different incinerators in The Netherlands.	Natural weathering for six weeks, then dried, and sieved to ≤ 0.125 mm.	Batch leaching test EN12457-2, at 10 $l.kg^{-1}$di-ionized water.	Numerous
Alam et al., 2019b	Different incinerators in The Netherlands.	No metals extraction, weathering for six weeks, dried at 105°C, and sieved to 3 grain sizes in range: 0.125 ≤ mm ≤ 4.	Column leaching test to EN 7383:2004, compared to sequential test of acidic, reducing, oxidising stages.	Numerous
Caviglia et al. 2019	Incinerator in Italy.	Single grab sample. Grain size sieved to 0.063 ≤ mm ≤ 20.	Batch leaching test to EN12457 with de-ionized water at L/S of 10 $l.kg^{-1}$.	Numerous

5.1.1 Comparison between Leaching Test Methods

Glauser et al. (2021) showed how the different leaching test methods of Switzerland and The Netherlands produced incompatible results for numerous potentially toxic elements from the same sample. Statistically significant correlations ($R^2 \geq 0.95$) were only found for Cu and Cl^- using deionized water as eluent. There was no statistically significant correlation for Zn ($R^2 = 0.65$), while for Pb the average concentration in the leachate from the Swiss batch test was 2.5 times higher than in the Dutch column test ($R^2 = 0.55$), thus evidencing the relative leniency of the Dutch test.

None of the bottom ash samples met Swiss regulations for landfill due to the total concentrations of heavy metals Cr, Cu, and Pb in some grain size fractions, but in particular Sb which exceeded the threshold for all size fractions, by 11 times in fine fractions, and even by 1.5 to 3 times when the bottom ash was subjected to 'enhanced treatment'. To this, the authors dryly observed that it is for a *"good reason"* that the Swiss landfill regulations only have limit values for TOC and certain non-ferrous metals, otherwise bottom ash samples would fail to comply. In general, their results showed that (Glauser et al., 2021):

"Disposal on landfills with lower requirements and recycling of bottom ash as raw material for cement clinker is not possible without applying further treatment steps."

With regard to the Dutch column leaching tests, only Cl^-, Cu, Pb, and Zn were reported, but all size fractions of bottom ash failed for Cl^-, while 62% of the samples also failed for Cu, and 38% failed for Pb. Of this, the authors commented how the Dutch column test, using deionized water, was unrepresentative because it neglected changing pH conditions over time.

Different leaching test methods using the same sample were also compared by Kalbe and Simon (2020). Some toxic elements such as Cd, Co, Ni, and Pb were shown to leach out of the bottom ash in greater quantities under column and lysimeter tests than under batch tests; while others such as Cl^-, Sb, and Sn leached out in greater quantities under batch rather than column testing, supporting the findings of Glauser et al. (2021). They explained their use of a lysimeter in comparison to the common batch or column tests (Kalbe and Simon, 2020):

"The results from lysimeter experiments are closer to real field conditions than the column test due to larger sample size and overhead irrigation rather than up-flow conditions".

As shown in Table 2, no European country uses a lysimeter for leaching tests.

Quantifying the range of results highlights the great variations that exist between batch and column test methods and hence how these permit some countries to use bottom ash while others cannot. Measured as a percentage difference in total concentration (C) across the batch and column tests, each using the same sample $((C_{batch} - C_{column})/C_{batch}) \times 100$, values differed by the following amounts: As = 63%, Cl^- = 44%, Sn = 52%, and Sb = 52%. This again quantifies the relative leniency of the Dutch column test, though for some other elements the values were negative evidencing the reverse.

Again of relevance to The Netherlands' use of column or monolith leaching tests, Allam et al. (2019a) assessed a bottom ash sample using the deionized water batch tests commonly applied in other countries. Their results showed that leachate concentrations of Cu, Cr, Mo, and Sb, along with Cl^- and SO_4^{2} from batch tests, would have put the samples above the legal threshold in The Netherlands for the use as a building aggregate. They went on to discuss options to improve bottom ash, and did not advocate weathering because:

"The formation of weathering phases reduces the leaching of potentially toxic elements for the short to mid-term; however these weathering phases are stable only in a limited pH range."

The same research group compared the Dutch column leaching tests for granular building material with a sequential leach test method (Allam et al., 2019b). In a sequential test, the sample is exposed to different conditions and therefore provides a 'worst case scenario'. No European country uses the sequential test.

Allam et al. (2019b) found that the ≤4 mm sample breached the Dutch leach test limit values for Cl^-, Cu, and Mo. They also referred to their own previously published studies from 2016 and 2017, which showed that leaching of Cr, Ni, Pb, Sb, and Zn ***"commonly exceeds the limit for use in non-isolated applications"***. With the sequential leach tests, Zn in particular was highly mobile during the conditions of low pH, while Cr, Cu, Ni, and Sb all showed high mobility during the oxidising conditions. The authors attributed this to the complexation of many toxic elements with humic material, which then become mobile when organic matter is destroyed under oxidisation. These results further evidence that the sterile leach tests are unrepresentative of real life conditions, in this case due to interactions with organic matter, and particularly as an unbound aggregate for road and footpath construction.

More results are provided below, drawing from other studies. To avoid a bland repetition of summaries, further results have been grouped to focus on specific limitations of the leach tests used across Europe in relation to the hazards of bottom ash.

5.1.2 Ageing, pH, Buffering, and Humic Material

When Glauser et al. (2021) changed the eluent of the batch test to lowered pH (using CO_2 saturated water) the differences in leachate concentrations were much greater: Zn mobility increased in the batch test by 15 times in comparison with deionized water

eluent. It was also observed that Cu leachate concentrations were higher when dissolved organic matter was present even under alkaline conditions (statistical correlation of R^2 = 0.90 between Cu concentration and dissolved organic carbon). The authors noted a high buffering capacity within the smaller fractions when CaO was present, resulting in temporary stability of both Cu and Zn.

Vateva and Laner (2020) compared their leach test results against two current and one draft German standard for building aggregate. Leachate concentrations of Cl^- and SO_4^{2-} in the samples were above the limit values for most grain fractions, while leachate concentrations of Cr, Cu, and Pb were also above limit values. They concluded that the ageing process of four months did not improve the quality of the bottom ash with regard to Cl^- and SO_4^{2-}. Interestingly, the authors observed, though did not explore, a *"substantial variation in the content of unburned organic matter over the whole nine days of sampling"*. In summary:

"The processed bottom ash was not suitable as a whole, neither as a construction material in constrained structures nor as an aggregate in concrete because it did not comply with limit values in current German regulations...Legal compliance of bottom ash as a construction material was not facilitated by longer aging [and] further manipulation or processing would be required to reduce the contents of soluble salts as well as to minimise residual metal contents."

And:

"The processed bottom ash, as a mixture, did not comply with current German limit values for use as a construction material mainly due to excessive soluble salt contents. Coarser grain size fractions were less contaminated, resulting in an utilisable potential of less than 30% of the bottom ash as a construction material."

The same draft German aggregate limit values were used as a reference by Simon et al. (2021), results which were the culmination of a six-year experiment with irrigation of a bottom ash sample by simulated rainwater (600 $mm.a^{-1}$) in a lysimeter. The lysimeter allows for temporal analyses which batch tests do not (Kalbe and Simon, 2020). In these tests, the pH did not change (9.9 ± 0.5), thus substances in the bottom ash acted as a temporary pH buffer. Even with ageing, short term release of high quantities of Cl^- and SO_4^{2-} were observed, which, along with Cu and Mo, greatly exceeded one of the limit value categories. Both Sb and V continued to be mobile throughout the six years and approached the limit values, as stated:

"Our experiment shows that the release of Sb and V from incinerator bottom ash is not minimised over the time of almost six years. Thus long-term use of incinerator bottom ash e.g. in secondary building materials can pose a potential risk to the environment".

They went on to say that no economically viable technical measures for the targeted depletion of Sb and V are available. And, in comparison to the new draft ordinance for mineral waste, the limit values were not reachable even for wet treated bottom ash (Simon et al. 2021).

In the experiments of Kalbe and Simon (2020), the substrate in the lysimeter was maintained in the alkaline range (8.5 ≤ pH ≤ 10.5). Though the authors say that in this limited range no correlation can be made between pH and Sb mobility, they did say that, at the end of the experiment, (six years) the cumulative release of Sb was still increasing. Sb release was shown to increase with decreasing Ca, as Ca forms less soluble compounds such as $CaCO_3$ (calcium carbonate) due to ageing. Thus, ageing to stabilise bottom ash will not only have an inverse effect on Sb mobility, but pro-active treatments such as adding Ca compounds would be unlikely to solve the problem as, in the long term, these will transform to $CaCO_3$ anyway (Kalbe and Simon, 2020).

Tiberg et al. (2021) confirmed that both Zn and Cu were more mobile below pH 8.5 often as much as four orders of magnitude. They also noted that leachability of these elements above neutral pH was governed by other parameters, but that humic matter increases their mobility. The authors did not refer the results to legal implications because "*clear guidance is lacking, and practice differs between countries*". They specifically referred to the limitations of the Waste Framework Directive.

5.1.3 Incinerator Instability and Hazard as a Function of Grain Size

Bielowicz et al. (2021) sampled bottom ash over a period of 36 weeks and during this time all elements studied at some stage exceeded the national leachate limit values for *"processing outside the plant"* by Polish legislation.[2] For Cl^-, the value was never below the limit at all and reached a maximum at seven times the limit value, while other significant maxima were for Ba (11.5x above the limit) and Sb (21x above the limit), with Zn also exceeding the limits on four dates; while the mean for Sb was more than twice the limit value, and exceeding the limits more often than not. The mean value for Pb concentration in the leachate exceeded the limit value by 70%. This broad variability in concentrations evidenced that the incinerator plant provided little or no constraints on toxic substances in the bottom ash. This was corroborated elsewhere by Simon et al. (2020) who observed that: *"The concentrations especially of trace metals can vary by an order of magnitude".*

Some authors suggest that removing the finer fractions of bottom ash would lead to reduced toxicity (Alam et al., 2019a). But this is not always correct. Kalbe and Simon (2020) showed that the smallest fractions (<25 mm) did not necessarily contain the most toxic elements, with approximately twice the amount of Br, Co, Cr, and Ni in the largest (0.25 ≤ mm ≤ 45) cut. Vateva and Laner (2020) also measured Cr and Pb in greater quantities in larger (>31.5 mm) and smaller (4–8 mm) fractions; Cd, Ni and Zn in smallest fractions; and, for Sn and Sb, partitioning exhibited a random spread. They also found greater concentrations in the >4 mm fraction for Cl^-, Cr, and SO_4^{2-} when the sample had been aged.

Mantovani et al. (2021) detected higher concentrations of Cr and Pb in the largest (>16 mm) fraction, while the highest concentration of Ni was in the 8 – 16 mm cut. A surface coating of finer fragments was found to cover the larger clasts and these were easily separated during transport and sieving, evidencing the friable nature of potentially toxic element release after sieving. Another interesting finding was that the LOI values measured as a function of size fraction: all greatly exceeded the EU (IED) requirements, with the lowest being 9.4% and the highest 26.4%. They concluded against the worth of sieving and screening because of potentially toxic elements (such as Cu, Ni, Zn, Pb, Sn) in all categories of carbonates, sulphates, amorphous, and LOI residues.

Caviglia et al. (2019) found that due to the presence of Cu, Italian limit values were exceeded for all size fractions below 10 mm. Other interesting findings were that LOI values exceeded EU regulations (at 6.2%); and that concentrations of most heavy metals (namely Cu, Ni, Pb, Sr, Zn) were in the mid-range (2 - 8 mm) grain sizes.

5.2 Persistent Organic Pollutants and Microplastics

Only one paper met the scope, from a research group in Norway. The methodology was therefore extended by year and to encompass recent references from outside Europe. Datasets are shown in Table 3.

Table 3. Datasets of empirical studies on organic toxins in MSW incinerator bottom ash.

Reference	Sample provenance	Sample preparation	Method	Analyte
Arp, et al., 2020	Norwegian incinerators via a waste handling facility	Samples taken over a year.	Leaching test with distilled water, as per EN12457 at 10 $l.kg^{-1}$. Shake time increased to 28 days.	PCBs (7 congeners).
Morin et al., 2017	Twelve Norwegian waste handling facilities	Sampled over one year.	Batch leaching test with distilled water, compliant with EN12457 at 10 $l.kg^{-1}$.	BFRs (PBDEs)

[2] Following personal correspondence with author: "Construction, reconstruction or renovation of railway structures and track beds, embankments, railway and road embankments, road foundations in motorways, impermeable coverings, bowls of earth settlements, cores of hydrotechnical structures and other structures and construction works, including foundations".

			Shake time increased to 28 days	
Lin et al., 2014	Two incinerators in Taiwan	Four samples per day (every 6 hours) over one week. Size fractions < 0.075 ≤ mm ≤ 9.5	Batch leaching test (shaking bottle) agitated for 18 hours with 'fluid' of pH 2.9.	PBDEs
Wang et al., 2010	Incinerator in Taiwan	Three samples per day (every 8 hours) over 3 days. Extractable Fe and NFe metals, 'stones and glass' removed.	Total concentrations. Samples compared to other ash samples downstream of the combustion chamber.	PBDD/Fs (12 congeners). PBDEs (30 congeners).
Liu et al., 2021	Three Chinese incinerators	24 samples taken over two years.	Total concentrations plus supernatant after centrifuge and pH 'stabilisation' to 7-8.	PFAS.
Hsieh et al., 2018	Incinerator in Taiwan	Taken directly from the incinerator over four years.	Total concentrations.	PCDD/Fs.
Chen et al., 2006	Two incinerators in Taiwan	Sampled 4 times per day at 2-hour intervals over one week.	Total concentrations. Size < 0.25 ≤ mm ≤ 9.5	PCDD/Fs.
Yang et al., 2021	Sixteen MSWIs and one bottom ash treatment plant in China	31 bottom ash sampled loads during stable operation.	Total concentrations.	Microplastics

5.2.1 PCBs

Polychlorinated biphenyls (PCBs) were widely used in paint and electronic capacitors until this was restricted in the 1970s. Many remain in circulation as legacy pollutants, but only three European countries (The Netherlands, Belgium, and Czech Republic) assess for the hazard of PCBs in building aggregate, and none assess for PCBs in leachate.

Arp et al. (2020) compared total concentrations of PCBs and their leachability from bottom ash and fly ash against a number of other substances from waste recycling facilities in Norway. They found that mean total concentration of PCBs were much greater in bottom ash (28 ± 34 µg kg^{-1}) than in fly ash (0.3 ± 0.8 µg kg^{-1}) - 93 times greater taken at the mean, while bottom ash showed the second highest leachability of PCBs for all samples studied. These results likely underestimate the true leach hazard, for the study only considered the freely dissolved fraction and not those PCBs associated with particles or dissolved organic carbon: the authors cite previous research that PCBs are mainly associated with particles and dissolved organic carbon by *"up to 80 - 99.9%"*. These mean total concentrations would have breached the limit value for the Czech Republic and one region in Belgium, which, if considering the full range, all limit values were exceeded. Of note is the wide variation in concentrations over the one year period of sampling.

5.2.2 PCDD/Fs

Polychlorinated dibenzo-p-dioxins/dibenzofurans (PCDD/Fs) are halogenated poly-aromatic compounds, commonly termed 'dioxins'. They produce a range of harmful effects including carcinogenicity, teratogenicity, immunotoxicity, and embryotoxicity. But, of the twenty six countries reported by Blasenbauer et al. (2020), only one of these (Germany) stipulated an assessment of PCDD/F

total concentration in bottom ash prior to its use as a building aggregate. No country required an assessment of PCDD/Fs in bottom ash leachate.

PCDD/Fs are more concentrated in fly ash than in bottom ash, but in bottom ash they are still appreciable (Hsieh et al., 2018). Over a four-year sampling period, the mean concentrations of PCDD/Fs in bottom ash (1.48 $ng.g^{-1}$) were approximately three fifths that of fly ash (2.56 $ng.g^{-1}$). Importantly, since the quantities of bottom ash produced were (mean average) three times greater than the quantities of fly ash, bottom ash was the main repository of incinerator PCDD/Fs. Again of note was the wide range of total PCDD/F concentrations in the bottom ash over the sampling period, evidencing the inconstancy of the MSWI process to stabilise PCDD/F fallout. This was expressed by the authors on a WHO_{2005}-TEQ basis[3], with concentrations varying by a factor of seven.

Chen et al. (2006) analysed total concentrations of PCDD/Fs in post metals extraction and air-dried bottom ash from two modern incinerators in Taiwan, described as *"the most effective technique for PCDD/F emission control"*. Measured on an I-TEQ basis, they found that particles ≤0.6 mm accumulate amounts of PCDD/Fs at a level potentially hazardous to the environment and so unsuitable for use in *"soil, road sub-base and construction blocks"*. Based on fertilizer limit values in Germany, one sample of bottom ash (grain sizes ≤0.6 mm) exceeded the limit values for application to pasture land, while larger grain sizes (≥2.36 mm) were borderline.

5.2.3 BFRs: PBDEs and PBDD/Fs

Brominated flame retardants (BFRs) are widely used in electronic circuit boards and plastic packaging such as microwave trays and coated textiles (Weidlich, 2021). Their concentrations in plastic range from 3 to 15% (Hennebert, 2021). Some are banned and regulated in the EU, but legacy products are in circulation, so will likely remain in waste for decades to come (ibid.). Polybrominated diphenylethers (PBDEs) are a subgroup of BFRs; many are endocrine disruptors; immune system toxicants; and form PBDD/Fs during low temperature, low oxygen regions above a MSWI grate (Weidlich, 2021). Evidence is that PBDD/Fs are as highly toxic as their better known chlorinated analogues, and more hazardous than the PBDEs from which they formed (Conesa et al., 2021). No European country currently tests for PBDEs or PBDD/Fs in building aggregate, while PBDD/Fs are neither monitored nor is there any limit for these substances in waste incineration residues.

After assessing the total concentration and leachability of PBDEs from a variety of waste handling sites, two of which were bottom ash from a modern Norwegian waste incinerator, Morin et al. (2017) found that total concentrations were approximately an order of magnitude higher in bottom ash than in fly ash, while leachate concentrations were also approximately an order of magnitude higher from bottom ash than fly ash. The study, again, did not consider the leached PBDEs sorbed to colloids or humic acids, but noted that the leaching results were likely *"biased low"* due to equilibrium not being reached over the experimental time period. The authors observed that the total concentration of BFRs remained constant between bottom ash and feedstock MSW indicating that they were not destroyed by incineration:

"Bottom ash contains concentrations of flame retardants that cannot be considered negligible [and that] this may need to be considered when landfilling bottom ash, or utilizing it in other purposes, such as filling materials."

In Lin et al. (2014), the PBDE total concentrations were 2 to 19 times higher in bottom ash than in fly ash, further evidencing that PBDEs were not completely destroyed during the incineration process. These total concentrations were two orders of magnitude higher than in urban and rural soils, more distributed in larger (0.25 ≤ mm ≤ 1) particles, and leached out at a rate approximately four orders of magnitude higher than the original waste material. Total concentrations in the bottom ash were 29 ≤ $ng.g^{-1}$ ≤ 243, but there are currently no limit values against which to compare them. The authors qualified the hazard by stating that after passing through the incinerator, the highly brominated leachable PDBEs readily degrade to lighter brominated variants, resulting in enhanced toxicity via increased uptake and bioaccumulation. They also commented that humic solutions enhance the leachability of PBDEs from

[3] TEQ = Toxicity equivalent. The sum of the products of the concentration of each compound multiplied by its Toxic Equivalence Factor (TEF). TEQ is applied due to the commonly shared toxicity across a suite of compounds to represent how their combined toxicity is additive. Two common metrics exist: I-TEQ based on TEF determined by an expert international group, and WHO-TEQ based on TEFs determined by the World Health Organisation.

bottom ash impacting on its use in ground works. Consequently, they advised caution on the proposed use of incinerator bottom ash as construction material.

The most highly cited study on POP leachability from bottom ash is by Wang et al. (2010). The authors sampled residues of bottom ash and fly ash from two air pollution control modules, and ash from two post combustion zones (economizer and superheater) three times per day at eight-hour intervals for three days, on a system *"recognized as the most effective technique for PCDD/F emissions control"*. They found that bottom ash had the highest PBDD/F and PBDE content of all residues, supporting that PBDEs were not destroyed by the incineration process; and also that PBDD/Fs were created by it giving higher concentrations than in the input waste. The content of PBDEs in bottom ash were between one and three orders of magnitude higher than in reference soils. They concluded that reutilization of incinerator bottom ash would contribute these substances to the environment.

5.2.4 PFAS

Polyfluoroalkyl substances (PFAS) have been produced since the 1940s and are widely used as flame retardants, and as water and oil repellents in and on plastics (Liu, et al., 2021). They are sufficiently soluble in water to be taken up by plants (Ghisi et al., 2019). None of the European countries reported on by Blasenbauer et al. (2020) tested for PFAS in bottom ash for use as a building aggregate.

PFAS are also not completely destroyed during the incineration process. Liu et al. (2021) found that bottom ash from two out of three incinerators sampled was enriched in PFAS at three times greater concentration than in fly ash. The authors concluded that bottom ash constitutes an important vector for PFAS into the environment; that more study is needed on the thermal transformation of PFAS; and that techniques to destroy PFAS within incinerators need to be developed. Another important finding was the widely varying concentrations of PFAS sampled in bottom ash over the two years.

5.2.5 Microplastics

The hazard posed by microplastics in bottom ash is a very new and underresearched topic. Yang et al. (2021) showed that incineration does not terminate microplastics and their presence in bottom ash ranged from 1.9 to 565 particles per kg, or up to 102,000 microplastic particles per metric ton of waste incinerated. The study was from sixteen modern MSW incinerators established or upgraded to *"advanced technology"* in the last ten years and obtained during stable operation. For all samples the LOI was below 3.2%, thus well within the <5wt% stipulated by the EU. The largest fractions of microplastic in bottom ash were identified as from packaging and building materials (polypropylene and polystyrene) indoctrinated with flame retardants thus making them resilient to high temperatures. Unfortunately, the study did not analyse for specific flame retardants or any POPs within the microplastics. There are currently no standardised test methods for determining plastic content in solid matrices, not least bottom ash, and there are no bottom ash/aggregate limit values for microplastics.

6 Discussion

6.1 Processing Influences and Implications

To understand bottom ash composition and its production mechanisms, one must first understand that waste is a poor fuel (Hulgaard and Vehlow, 2011). MSW is highly heterogeneous making its combustion a very complex phenomenon involving thousands of chemical reactions (Chagger et al., 2000). In theory, elements such as Cd and Hg with boiling points lower than the grate temperature should not be present in bottom ash, while others such as Pb and Zn with higher boiling points should always fall out through the grate. However, this does not follow, with also As, Br, Cd, even Hg in bottom ash (studies in this report, plus Buchholz and Landsberger, 1995; Meima et al., 1999; Klymko et al., 2016). Even recent authors question how volatile substances find their way

there (Glauser et al., 2021). But it is basic reactor engineering that localised that hot and cold regions occur due to endothermic drying and pyrolysis reactions that sap the internal temperature, along with rich and lean (i.e. oxygen enriched and oxygen depleted) pockets, combined with physical mass and heat transfer limitations, resulting in improper conditions for burning both spatially and temporally; while the presence of many metallic elements such as Cu and Sb create catalysis, speeding up unfavourable reactions that form chlorinated and brominated dioxins (Ebert and Bahadir, 2003; Weidlich, 2021). Added to this there are numerous plastic goods in waste which are impregnated with flame retardants (Table 1), thus not only resistant to thermal treatment, but many of which convert to more toxic forms. This is seen by the range of substances which bottom ash contains, evidencing exceedingly high temperatures (molten Cu) along with substantially low (even unburned sewage sludge is emitted) (Bunge, 2019).

Bottom ash treatment is still a fledgeling industry, progressing largely since the 1990s, and with no two treatment plants the same (Bunge, 2019). Processing is usually undertaken offsite, by a different commercial entity, and frequently after shipping across regional or national borders (Arkenbout, 2019). Mehr et al. (2021) describe how modern plants have extraction efficiencies of between $29 \leq \% \leq 92$ for five metals only, while for Pb it is only 16%. As Simon et al. (2021) state:

"The recovery of elemental metals is still a challenge in terms of recovery rate and purity".

Increased recovery of metals can be achieved by extra comminution, but this would adversely affect the residue's value as an aggregate where the integrity of larger particles must be maintained (Bunge, 2019). It would also increase the risk of toxic dust creation, a problem which is somewhat mitigated by wet discharge predominating in Europe. Wet capture creates new mineral phases, which in theory leads to some stabilisation of heavy metals, but at the same time binds elements into a mineral matrix making them unextractable with current technology (Vateva and Laner, 2020). Dry extraction creates its own problems, one of which is friability and dust containment since the dust is loaded with heavy metals (Bunge, 2019). Airborne dispersal of Pb is identified as a particular critical risk factor with road and sub-base applications (Van Praagh et al., 2018). Adverse environmental consequences of bottom ash dust release have also been reported in recent case studies from The Netherlands where it is claimed that open air transportation and handling (without precautions) resulted in marine fauna endocrine disruption at a UNESCO site (Arkenbout, 2019).

In the EU BAT document for waste incineration (EU, 2019), screening and sieving of bottom ash is recommended. However, the findings of this study do not totally concur with these recommendations, since toxic elements are widely spread across all size fractions and types of bottom ash clast. Similarly, the same document recommends weathering/ageing, and while formerly it was perceived that longer duration was better, Germany for example is moving towards shorter ageing periods, which leads to higher metal extraction potential but also higher solubility of salts and less stable mineral phases (Vateva and Laner, 2020).

Weathering/ageing can also lead to the detachment of finer particles and hence increase the mobilisation of toxins (Alam et al. 2019a). Furthermore, weathering increases the mobility of Sb from bottom ash over long timescales (Kalbe and Simon, 2020). Scientific understanding on the subject is still weak and cannot yet adequately guide praxis.

Some authors tested secondary treatment but this is not totally beneficial and has associated climate/cost impacts. Caviglia et al (2019) found that, after subjecting bottom ash to temperatures up to 1000°C, leachate concentrations of Cu, Pb, and Zn were reduced, but there were two orders of magnitude increase in leachate concentrations of Cr and Ba, and one order of magnitude increase for Al.

The increased mobility of Al is interesting and important for those companies who promote cement bound 'green' products – such as building blocks – made from incinerator residues. The matter is of concern not just for toxic substances leaching out but also structural safety, particularly where high treatment temperatures are used in manufacture (as per Caviglia et al., 2019). Aluminium reacts with water and, over time, releases hydrogen, which can lead to both fires and swelling of the block product from which it is made, thus deteriorating compressive strength (Allegrini et al., 2015; Bunge, 2019). This was supported by studies such as Vateva and Laner (2020) where metallic Al content was above 1% in all grain fractions of bottom ash treated with current BAT; and, even at this level, it was said to impair the utilisation of bottom ash as bound aggregate. Elsewhere, Tiberg et al. (2021) reported concentrations of Al of 5-6% of the total composition of the bottom ash even after metals separation.

When such products are brought to sale in Europe, they come via a weak, outdated, fragmented, and therefore unsatisfactory regulatory system of standards and testing methods. The few nationally approved leach tests represent a compromise between the desire to replicate real conditions and the wish to minimise testing time and avoid crushing (Blasenbauer, 2020). But, as shown in this report, they are not comprehensive of all toxic substances, they only represent short-term and relatively sterile conditions, thereby providing at best only a makeshift snapshot rather than a long-term assurance against product integrity and public safety. By discounting changes in pH and the influence of humic matter they also provide spurious results: High acidity (low pH) of the eluent increases the concentration of metals in the leachate and therefore overestimates the leaching potential, but slightly acidic conditions are buffered by the alkalinity and therefore temporarily stabilise it, leading to underestimation of species mobility. This also undermines the safety of bound products, since Portland cement is known to continue ageing through its lifetime and convert to $CaCO_3$ (Haselbach, 2009). Thus, bound bottom ash products are unlikely to remain stabilised over time within cement-based blocks or concrete claimed to be initially safe. This would also then lead to the fallout of toxins.

An unexpected and incidental finding of this report was the range in LOI values in excess of the legal minima. Surprisingly, this report is believed to be the first to use bottom ash (made available by independent testing) as an incinerator diagnostic. It casts doubts on whether incinerators fed with modern MSW are fit for the purpose of creating environmentally-benign, and therefore usable, bottom ash; but also on the efficacy of current incinerator monitoring and operational stability. They supplement previous concerns expressed by bottom ash treatment plant operators regarding large fluctuations in the quality of ash sent to them and general trends of decreasing quality (Arkenbout, 2019). Others have suggested that sources of POPs could result from MSW incinerator operators not applying BAT (Weber et al. 2019); while periods of Other Than Normal Operating Conditions (OTNOC) have been offered as a further supposition to account for high pollutant emissions (Arkenbout et al., 2018). This report suggests that, in fact, instability may be commonplace even during periods of steady state, a matter which requires further investigation.

6.2 Drivers and Other Motivations

In one of the more readable practical accounts on bottom ash treatment, its author concludes that the push for using bottom ash after metals extraction has nothing at all to do with environmental concerns (Bunge (2019):

"using dry processed [sic] *BA as a construction material is exclusively driven by commercial interest hiding behind a fig leaf of environmental commitment."*

He is referring to the high cost of landfill and, perhaps, also the legal impositions of reporting and monitoring prior to disposal of a commodity which has negative value (EU, 2004). Similar findings were reported by Arkenbout (2019), where the Dutch environment inspectorate concluded that a high risk of fraud comes from the waste industry due to the negative market value of bottom ash, and indicated a clear problem with current implementation of regulations (Arkenbout, 2019). It went on to say that:

"Due to a lack of commercially viable options to 'clean' the bottom ash to acceptable levels of toxins (POPs and heavy metals), it is simply not done."

There have been case studies, such as Byker (Newcastle), and Jezera (Czech Republic) where the application of incinerator ash led to local soil contamination and POP bioaccumulation, the latter accidentally supported by EU funding (Petrlik and Bell, 2020; Swedish Environmental Protection Agency, 2011; Arnika, 2021). More recently, drawing on further case studies from The Netherlands, it was concluded that (Arkenbout, 2019):

"Though research is limited, what exists indicates strong concerns for public safety and the environment" [It urgently called for further research and...] "until then, any 'useful' application of bottom or fly ash should be suspended."

Trade appears to be a driver for the use of bottom ash rather than domestic usage. Austria has no need for bottom ash as a construction material (Blasenbauer et al., 2020). The same applies to Switzerland, where vast amounts of unpolluted aggregate are

generated as a surplus to excavation work (Glauser et al., 2021). When one looks at European trading statistics, a handful of countries (particularly The Netherlands, Germany, Norway, and Belgium) are major traders in natural gravel and sand, with an economic turnover in billions of US dollars per year (Leal Filho et al., 2021). The Netherlands in particular is the third largest global exporter of sand (OEC, 2021). Combined with the unharmonised and fragmented testing methods already reported, the building aggregate standards take a laissez-faire approach which puts the onus of risk assessment on the producer, advising testing only *"when required or in case of doubt"*, while product control merely reverts back to the inappropriate EU Directives (EN, 2008):

"It is the producer's responsibility to ensure that if any dangerous substances are identified their content does not exceed limits in force according to the provisions valid in the place of use of the aggregate."

All of which creates a high level of risk to public and environmental health. The main factor controlling bottom ash use in Europe currently is not whether it is scientifically shown to be safe, but rather which leaching test method is chosen and which country or region is the point of sale. Any financial support for 'green' use of bottom ash would likely encourage the movement of hazardous material away from containment in landfill and towards countries with either no, or at least more lenient, environmental regulations. Developing countries seem particularly vulnerable. Some protection is offered by bottom ash being listed on Annex II of the Basel Convention, which seeks to minimise the transboundary movements of hazardous wastes; but not all countries are party to this, and transboundary shipping is permissible with prior consent.

Bottom ash does not have End of Waste status (Blasenbauer et al., 2020). But, according to EU rules, it may be classified as non-hazardous if proven by testing in relation to fifteen hazard classes (EU, 2014). However, there is no harmonised testing method (Blasenbauer et al., 2020). In one Dutch study for the incinerator industry, it was shown that bottom ash was in breach of limit values for the EU waste classification by Pb total concentration, being particularly vulnerable to H10 (toxic for reproduction) and H14 (ecotoxicity) (Klymco, et al. 2016). This confidential report was followed one year later by guidance explaining how the different methodologies for determining H14 could provide different (i.e pass or fail) hazard classifications, despite the limit value breach (Klymko et al., 2017).

The hazards of bottom ash were identified in the 1990s. Meima et al. (1999) found that Cd, Cu, Mo, and Pb showed leachability which was independent of pH, with Zn (all samples) and Cd showing highest leachability at low pH. The sequential leaching methodology is also not new. It was used by Buchholz and Landsberger (1995) who found that leaching of Zn was deemed to be particularly significant due to relatively high quantities in bottom ash; As, Cd and Pb leached in mildly acidic conditions and were assigned as *"long term leaching hazards"*; while As and Pb were present in the greatest range of compounds, making their containment more difficult.

Yet, nearly three decades later, and despite the evidence presented in this report, one still finds that the facts about toxins in bottom ash are totally missing in waste incinerator industry bottom ash 'fact sheets' (CEWEP, 2019). While they also go unmentioned in waste incinerator planning/permitting applications (see §1).

Recently, the United Nations Environment Programme Special Rapporteur published a document on the environmentally sound management and disposal of hazardous substances and waste. Particular criticism was directed at the waste industry (Orellana, 2021):

"Examples abound of disinformation campaigns developed by companies and industries in order to retain their market share at the expense of the rights of people, including workers, consumers, individuals and communities who are exposed to hazardous substances."

Among thirty nine recommendations, the following seem relevant (ibid.):

"Design policy interventions to address the risks and harms of hazardous substances on the basis of the best available scientific evidence."

"Respond to scientific breakthroughs by updating and revising protection measures regarding toxics in a timely manner."

"Apply the precautionary principle in all policy-making and regulatory contexts in which the relevant scientific evidence concerning hazardous substances is inconclusive."

6.3 Findings in Relation to the EU Taxonomy

This research finds that considerations of bottom ash making a substantial contribution to a circular economy transition are premature and unproven, and the hypothesis that the use of bottom ash in civil engineering applications would *"do no significant harm"* is refuted. This is by reference to the following EU Taxonomy objectives:

- Sustainable Use and Protection of Water and Marine Resources;
- Pollution Prevention and Control; and
- Protection and Restoration of Biodiversity and Ecosystems.

The use of bottom ash is seen to be excluded by Art 17, 1, d, iii of EU (2020):

"The long-term disposal of waste may cause significant and long-term harm to the environment."

With respect to Climate Change Mitigation from the use of bottom ash, and while excluding the greenhouse gas emissions directly associated with waste incineration, only one author commented on it, though some mentioned the energy demands of bottom ash treatment without comparison (e.g. Mehr et al., 2021). Bunge (2019) stated that the CO_2 emissions between landfill and the use of bottom ash in place of gravel and sand in building applications cannot be expressed in any meaningful way due to the environmental damage caused by the leaching of metals once *in situ.*

Of relevance is an earlier study from Allegrini et al. (2015) who used empirical data from leaching tests based on bottom ash obtained from a Danish bottom ash processing plant. With this they modelled the toxicity impact for metal leachate only (not POPs) via three categories: carcinogenic human toxicity; non-carcinogenic human toxicity; and freshwater ecotoxicity from the proposed utilisation of bottom ash in nine different construction scenarios. Adverse impact was shown for all three categories in eight out of the nine scenarios: a significant contribution to human carcinogenic toxicity and freshwater ecotoxicity was identified with the inclusion of bottom ash in concrete; while Cr dominated the human carcinogenic impact; and As and Zn were more influential in the non-carcinogenic toxicity category. The high impact of Cr was caused after carbonating concrete specimens were used as a road sub-base, leachate values for Cr and selenium (Se) did not comply with release limit values from concrete specimens set by the Danish government, while leachate from the same sample was not compliant prior to ageing due to excessive release of Pb, and Cu had the greatest adverse impact on freshwater ecotoxicity.

6.4 Alternatives and Recommendations

If waste incineration (a linear activity) remains for some time within a circular economy transition, then better upstream source separation of waste to remove plastics which contain toxins and/or which are known to produce POPs during waste incineration seems essential. Removal of plastics from waste incinerators would, however, have adverse consequences for internal temperature because plastics are of relatively high calorific value. Also halogens - the precursors to many POPs - will remain widely distributed in other waste substances such as coated wood and textiles (see Table 1). A robust system of traceability for toxins in waste would appear to be a preferential activity for funding, and this would offer wider benefits to a circular economy by providing a mechanism to reduce dissemination and repeated cycling of toxins. A better option is, however, to reduce the generation of waste.

Until then, an overhaul of regulatory standards and best practice is needed to better reflect science and the precautionary principle before any further use of bottom ash is sanctioned. This should involve sequential (worst case scenario) leaching test methods and

more comprehensive testing of toxic substances in leaching and total concentration analyses with a focus on long-term, life-cycle stability, along with greater European harmonisation.

6.5 Limitations

It is possible that some empirical studies were overlooked during this review. The absence of recent European-based empirical research on POPs is perplexing, and cannot be explained other than by the hypothesis that it reflects a prevalent academic funding environment in Europe, with a trend for research with commercial enterprise potential – in this case, increased bottom ash metals extraction. This would also explain the lack of research studies on toxic organics, which would likely elicit no such immediate benefit.

Only one study reported on a single grab sample. The rest provided results of long-term bottom ash sampling and analysis (maximum six years), thus strengthening the value of these results. Variations in waste composition could increase the potential for error, but this is constrained by European nations having broadly similar waste demographics (Hoornweg and Bhada-Tata, 2012). Results for organic toxins from Asian incinerators might need treating with some caution, though the plastic fraction in waste is similar globally (ibid.).

A number of studies are considered as underrepresenting the true hazard of bottom ash. For example, Glauser et al. (2021) picked out unburned organic material >5mm from the bottom ash prior to sampling; while Bielowicz et al. (2021) excluded certain samples which contained Pb exceeding national limit total concentrations, and despite this their results still showed Pb concentrations which exceeded the leachate limit value by 70%. Some studies were limited by the range of elements analysed: Vateva and Laner (2020) did not analyse Sb, an element which was shown to exceed the leachate limit values elsewhere for similar tests (Glauser et al. 2021, Simon et al. 2021).

7 Conclusions

Arising from this study, there follows a list of concerns for public health and safety relating to the use of incinerator bottom ash residues in 'circular' applications:

1. Current standards for safety are outdated. In the EU, the use of bottom ash is inadequately regulated; rather there exists a hotchpotch of, at best, autonomic rules and guidelines, with many countries having no requirement for testing.
2. Bottom ash contains significant total concentrations of elements which are a 'high level of concern' based on EU REACH hazard classifications.
3. Bottom ash test methods have inconsistently prescribed total concentration values, with regulations only requiring the determination of a handful of toxic substances.
4. Bottom ash leaching test methods have inconsistently prescribed limit values, with regulations only requiring the determination of a handful of toxic substances.
5. Bottom ash leaching test methods are not based on current science and underrepresent real conditions:
    a. They consider short-term leaching only, with some toxic elements mobile after six years of experimentation.
    b. They give spurious results due to pH buffering. This makes the sample appear to be more stable than it actually is.
    c. They fail to consider the influence of humic matter, which is shown to accelerate leaching.
    d. For bound applications they fail to consider the long-term effects of cement carbonation due to atmospheric CO_2 uptake and weathering. This gives a false estimate of stability.
6. There is a likelihood of hazardous bottom ash export to countries with more lenient regulations.
7. The waste incinerator industry fails to mention the hazards associated with bottom ash in its 'fact sheets' and in permit/planning applications.
8. Microplastics are not destroyed by the incineration process, with up to 565 microplastic particles per kg of bottom ash.

9. PCDD/Fs are present in bottom ash in larger volumes than in fly ash and in concentrations of ca. 3/5 that of fly ash. For bottom ash to be used as building aggregate, only one European country assesses for PCDD/F total concentration and no European country assesses for PCDD/Fs in leachate.
10. PBDE concentrations are an order of magnitude higher in bottom ash than in fly ash, and are not destroyed by the incineration process. No European country assesses for PBDEs in bottom ash to be used as a building aggregate, either with total concentration or leachate.
11. PCBs concentrate in bottom ash in quantities almost two orders of magnitude higher than in fly ash (taken by the mean), and they also leach from bottom ash in higher concentrations than fly ash. Only three countries in Europe assess for the total concentration of PCBs in bottom ash for use as a building aggregate, and none assess for PDBs in leachate.
12. PFASs accumulate at three times greater the total concentration in bottom ash than in fly ash. No European country assesses PFASs in bottom ash for use as a building aggregate either by total concentration or leachate.
13. The EU Best Available Techniques for bottom ash processing are outdated and do not represent current scientific knowledge:
    a. Sieving/screening to remove smaller grain size fractions is not satisfactory, with many potentially toxic elements found in larger quantities in larger grain sizes. It leads to a higher risk of toxic dust exposure and airborne toxin dissemination.
    b. Weathering/ageing is not wholly beneficial. It can lead to higher toxin mobility and indirectly increase the hazard of bottom ash by binding more metals within the mineral fraction.
14. The bottom ash treatment industry is still at a fledgling stage and it is currently incapable of removing all metals. The presence of some, such as Al, causes swelling and hydrogen release, along with a possible fire hazard in cement-bound applications over the long term. Even after treatment, Al is present in bottom ash in quantities liable to disrupt the structural integrity of cement-based (blocks and concrete) products over time, creating long-term risks associated with the use of these products.
15. Many independent studies showed that waste incinerators were not operating at a steady state in compliance with the Industrial Emissions Directive. This impacts not only on the capacity of waste incinerators to produce benign bottom ash, but also raises concerns about the efficacy of waste incinerator monitoring and policing.

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Zero Waste Europe is the European network of communities, local leaders, experts, and change agents working towards the elimination of waste in our society. We advocate for sustainable systems and the redesign of our relationship with resources, to accelerate a just transition towards zero waste for the benefit of people and the planet.

GAIA is a global network of more than 800 grassroots groups, NGOs, and individuals. We envision a just, zero waste world built on respect for ecological limits and community rights, where people are free from the burden of toxic pollution, and resources are sustainably conserved, not burned or dumped. We work to catalyze a global shift towards environmental justice by strengthening grassroots social movements that advance solutions to waste and pollution.

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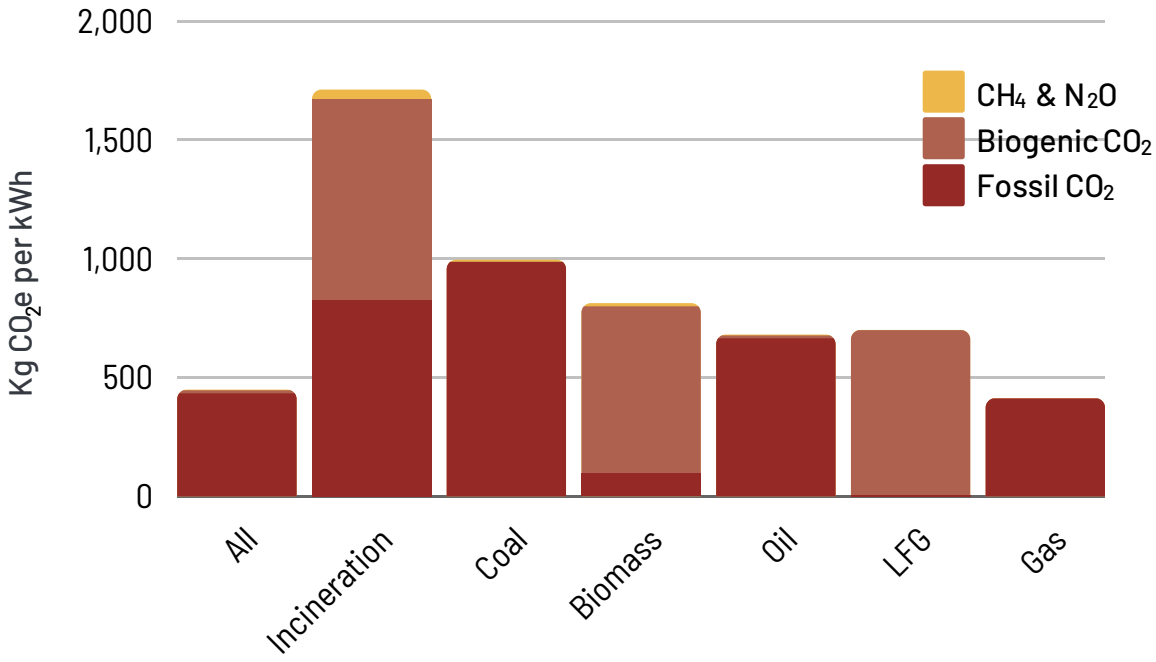
WASTE INCINERATORS UNDERMINE CLEAN ENERGY GOALS

Renewable energy is defined as energy produced by natural resources — such as sunlight, wind, and geothermal heat — that are naturally replenished within a certain time span. Municipal solid waste is derived from finite natural resources and burning these materials for energy significantly hinders resource conservation, while burdening communities with pollution and climate impacts.

A new study reaffirms that waste incineration is neither a renewable nor clean source of energy through a thorough comparison with other energy sources. Waste incinerators, the dirtiest source of energy on the grid today, must not be part of national or state climate plans.



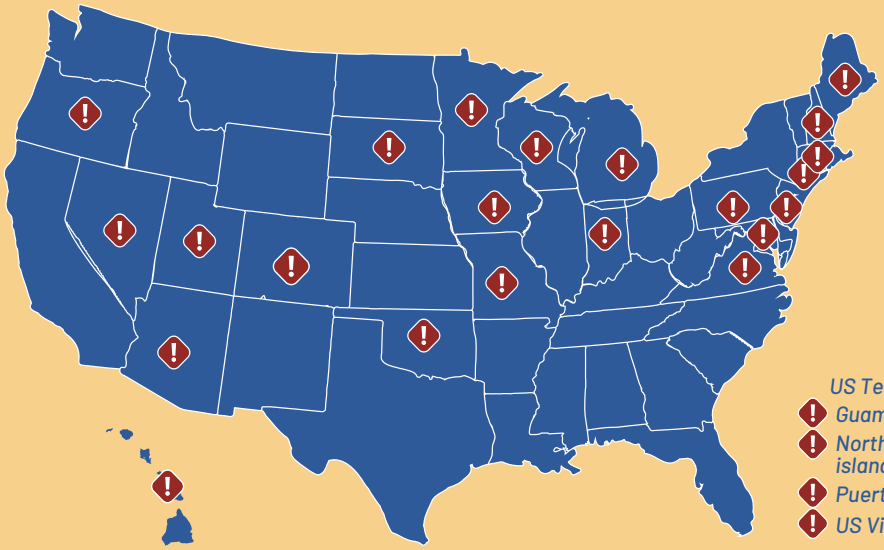
Incinerators are dirtier than the rest of the grid: per unit of electricity generated, they emit **3.8 times** as much GHGs (1.9 times as much fossil CO_2, **15 times** as much N_2O & CH_4, and **66 times** as much biogenic CO_2) as the grid average. They also emit **14 times** as much NO_x and **1.3 times** as much SO_2.

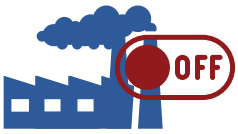
79% of incinerators are located in environmental justice communities: WTE facilities add to the cumulative burden of pollution on low-income, and Black, Brown, and Indigenous communities that causes long-term, multi-generational health impacts from toxic air pollution.

Waste incinerators stand out as the only generation source that emits large quantities of both fossil and biogenic emissions — in fact, **incinerators emit more biogenic than biomass plants!**

42 US states & territories have Renewable Portfolio Standards; **26** of them include incineration as a form of "renewable" energy. In 2018, incinerators earned **$41-44 million** in subsidies as "renewable energy."

Arizona, Colorado, Connecticut, Hawaii, Indiana, Iowa, Maine, Maryland, Massachusetts, Michigan, Minnesota, Missouri, Nevada, New Hampshire, New Jersey, Oklahoma, Oregon, Pennsylvania, South Dakota, Utah, Virginia, Wisconsin





The average age of the US incinerator fleet is 32 years — in other words, they are rapidly nearing retirement.

Without renewable energy subsidies, most incinerators will probably close in the next few years. But if these subsidies are extended, or if incinerators are subsidized through a new federal Clean Energy Standard, they could continue polluting for another **20 or more years.**

If all US incinerators retire at 35 years old, they would still put an extra 157 million tonnes CO_2e, 16.8 million tonnes NO_x, and 39,700 tonnes SO_2 into the air, which is equivalent to:

emissions from **40** coal power plants per year

or **34 million** cars on the road for one year,

or **18 million** homes' energy use for one year

If their lives are extended another 20 years, these excess emissions would go up to 637.7 million tonnes CO_2e, 61.9 million tonnes NO_x and 161,200 tonnes SO_2.

Both NO_x and SO_2 cause harm to human health, including increased asthma risks, reduced lung function, and greater likelihood of hospital admissions, especially among children and people with respiratory conditions.

[Read the study: Tangri, N. (2021). Waste Incinerators Undermine Clean Energy Goals. doi: 10.31223/x5vk5x]