Report

Onondaga County
Resource Recovery Facility

NYSDEC Part 360 Permit ID No. 7-3142-00028/00011
Title V Air Permit ID No. 7-3142-00028/00009

Annual Report of Facility Performance
Operating Year 2010

Onondaga County
Resource Recovery Agency
WWW.OCRRA.ORG

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Section 1 – Introduction

In the 1980s, the Onondaga County Solid Waste Management Program developed a plan to deal with the community's mounting garbage crisis. Realizing that there were no easy answers, they set out to design a safe, reliable, and cost-effective program that would serve the community's needs, at that time and into the future. They carefully analyzed the environmental impacts of different trash disposal alternatives and determined that no single method of disposal would solve the trash dilemma. Ultimately, a comprehensive, finely balanced, and integrated solid waste management system was required to manage the County's waste. The final plan consisted of four parts:

1) a waste reduction program,
2) an aggressive recycling program,
3) a state-of-the-art mass burn waste-to-energy (WTE) facility, and
4) a modern, lined landfill.

To manage this new County-wide waste management system, the County created a public authority – the Onondaga County Resource Recovery Agency (OCRRA). OCRRA would administer the County's solid waste management with a prioritization of management methods that exactly mirrored New York State's Solid Waste Management Plan: 1) waste reduction, 2) recycling, 3) recovery of useful energy through solid waste combustion (i.e., modern waste-to-energy facilities), and 4) use of permitted landfill facilities.

After a rigorous procurement process in 1988 and 1989, Ogden Martin Systems was selected to design, build, and operate the Onondaga County Resource Recovery Facility (Onondaga County WTE Facility). OCRRA entered into a service agreement with Ogden Martin Systems of Onondaga (currently Covanta Onondaga) in 1990. On December 18, 1992, with environmental permits in place and project revenue bonds totaling $178 million, formal groundbreaking ceremonies were held for the construction of the waste-to-energy facility. By late 1994 the Facility had its first official burn and by early 1995 the Facility was commercially operational.

Today, the Onondaga County WTE Facility continues to be an integral part of OCRRA's solid waste management system, or perhaps more aptly termed, OCRRA's resource recovery system. About 47% of materials that could otherwise go to the WTE Facility are source separated for recycling. The remaining non-recyclable portion goes to the WTE Facility, which uses a mass burn combustion system (and temperatures of 1800° - 2000° F) to safely and efficiently convert non-hazardous, non-recyclable trash into steam.

The steam is then used to generate electricity that is sold to National Grid, providing enough electricity for approximately 30,000 households and the Facility itself. Ferrous and non-ferrous metals that would otherwise have gone to a landfill are recovered at the WTE Facility for recycling. The by-product of the combustion process is a non-hazardous ash residue, which is about 10% of the original volume of the trash processed at the Facility. The ash residue is sent to a landfill for use as alternative daily cover.

Incorporated into the operations of the Facility is an air pollution control system, which helps the Facility comply with one of the strictest air permits in the nation, meeting federal and state emissions requirements. Emissions from the Facility are carefully monitored through continuous emissions monitors (CEMs) and annual stack testing.

Since its start-up in 1994 the facility's operational and environmental performance has exceeded expectations. In fact, the Facility has received several national awards and, in 2008, POWER Magazine ranked the Onondaga WTE Facility as one of the top five renewable energy facilities. By generating power for use by homes and businesses, the Onondaga County WTE Facility offsets the burning of fossil fuels by using an alternative, domestically-generated fuel: non-recyclable solid waste.

This report presents a summary of operational, environmental, and financial performance of the Onondaga County WTE Facility, located at 5801 Rock Cut Road (Town of Onondaga), Jamesville, New York for calendar year 2010. The Facility operates in accordance with NYSDEC Part 360 Permit ID No. 7-3142-00028/00011 (issued 11/16/01) and USEPA Title V Air Permit ID No. 7-3142-00028/00009 (issued 1/8/02). 2010 was the 16th full year of Facility operation since initial start-up on November 10, 1994. Commercial operation began on February 25, 1995.

The report is organized as follows:

- Section 2 of the report presents an Executive Summary.
- Section 3 presents a summary of the Facility's operational performance.
- Section 4 presents a summary of the Facility's environmental performance.
- Section 5 presents a summary of the Facility's financial performance.
- Section 6 provides a list of references.

Section 2 – 2010 Highlights

2010 Overview

- OCRRA's system is exceptionally consistent with the New York State and U.S. Environmental Protection Agency waste management hierarchy, which includes (in order of preference): 1) waste reduction, 2) recycling, 3) recovery of useful energy through solid waste combustion (i.e., modern waste-to-energy facilities), and 4) use of permitted landfill facilities.

- 2010 was another year of low trash tonnage due to the continuing recession. Trash tonnage was about 9% below historical levels. Electricity rates averaged 4.5¢ per kilowatt hour (kWh); better than 2009's average rate of 3.7¢ per kWh, but still much less than the rate of about 6¢ per kWh for the previous few years. On a positive note, the Facility's operational and environmental performance remained strong and consistent with historical performance.

2010 Operational Performance

- The Facility has been for the past 16 years, and continues to be, well operated and maintained by Covanta Onondaga.

- The Facility processed 315,385 tons of non-hazardous, non-recyclable trash (enough to overfill the Syracuse Carrier Dome) or 87% of capacity and, in doing so, generated 218,118 megawatt hours (MWh) – enough electricity to power approximately 30,000 homes, as well as the Facility itself. As mentioned above, the amount of trash processed was down about 9% due to the ongoing recession, and therefore, the total amount of electricity generated was also down from historical generation.

- The Facility had a net electricity production of 605 kWh per ton of refuse processed. This rate is below the Facility's 16-year average of 628 kWh/ton; however the difference is primarily due to the turbine-generator system being out of service during the 21-day major maintenance overhaul.

- In 2010, the Facility's metal recovery systems recovered 8,932 tons of metal for recycling.

- Overall boiler availability for 2010 was 93.3%. This value reflects less downtime for scheduled maintenance and equipment malfunctions than the historical Facility average.

- Turbine-generator availability was at a Facility-low of 94.4% due to a thorough and detailed inspection of the turbine-generator system; the second major overall since Facility start-up.

2010 Environmental Performance

- The 2010 annual stack testing results indicate that the Facility is performing strongly. All parameters met the corresponding air permit limits, and most were significantly below the permit limit.

- Levels of mercury in the incoming waste stream continue to trend downward, indicating that OCRRA's mercury removal programs are effective. Furthermore, the Facility demonstrates high mercury removal efficiency. Mercury emissions from the Facility were 5% of the permit limit.

- In 2010, the estimated annual total dioxin toxic equivalence (TEQ) emissions were 0.00009 lbs (90 millionths of a pound) – an amount equivalent to 3% of the weight of a standard paper clip. Dioxin/furan emissions from the Facility were less than 5% of the permit limit.

- By sending the community's non-recyclable trash to the WTE Facility, rather than to a landfill, greenhouse gas emissions are avoided. As a general rule of thumb, approximately 1 ton of trash processed prevents 1 ton of carbon dioxide emissions. So in 2010, the WTE Facility avoided 315,385 tons of carbon dioxide emissions, which is the equivalent of taking about 50,000 passenger vehicles off the road.

- The WTE Facility utilizes a locally-generated feedstock – the community's non-recyclable trash to generate a significant amount of electricity; this not only reduces dependence on fossil fuels, it also achieves goals of energy independence. In 2010, the WTE Facility generated enough energy to displace nearly 400,000 barrels of oil or 80,000 tons of coal – enough energy to satisfy the needs of approximately 30,000 homes in OCRRA's service area.

- With one of the highest recycling rates in New York State, Onondaga County demonstrates that WTE and recycling are highly compatible; it also supports many studies that have concluded communities with WTE facilities often have higher rates of recycling.

- In 2010, all ash residue from the Facility was used as alternative daily cover at the landfill. This beneficial use of the ash ultimately means that other materials, such as clean soil, do not need to be used for daily cover at the landfill.

2010 Financial Performance

- Due to the continuing recession, trash tonnage were still down about 9% and electricity rates remained relatively low. As a result, OCRRA's 2010 Facility-related expenses were $1.57 million more than Facility revenues. Total operating revenues were approximately $24.29 million and total (operating and bond) expenses were $25.86 million. As evident, WTE facilities like the local Facility have tremendous fixed costs. If those fixed costs are not offset by sufficient electricity revenue and tipping fees, there may be facility-related net losses, as in 2009 and 2010.

Section 3 – Operational Performance

3.1 Summary of Operations

Based on the 2010 operating data, overall Facility operations continued at high levels for the 16th year of continuous operation despite a second year of relatively low trash levels. The Facility processed 315,385 tons of municipal solid waste (MSW), 87% of the Facility's permitted throughput limit of 361,350 tons. Overall boiler availability for 2010 was 93.3%, which is higher than the 16-year Facility average of 91.4%. Turbine-generator availability was at a Facility-low of 94.4% due to a thorough and detailed inspection of the turbine-generator system; the second major overall since Facility start-up.

The average higher heating value (HHV) of waste processed in 2010 was 5,399 British thermal units per pound (Btu/lb). The 2010 HHV, which indicates the energy embodied in the incoming waste stream, was slightly above the Facility's 16-year average (1995-2010) average HHV of 5,377 Btu/lb. The Facility had a net electricity production of 605 kilowatt-hours per ton of refuse processed (kWh/ton). This rate is lower than the Facility's 16-year average of 628 kWh/ton, however the difference is primarily due to the turbine-generator system being out of service during the 21-day major overhaul.

In 2010, the WTE Facility generated 77,534 tons of combined ash residue, which were hauled by OCRRA to Seneca Meadows Landfill in Waterloo, NY. Based on the waste tonnage processed, this amount of ash was 24.6%; therefore the Facility reduced the weight of the refuse by over 75%. The 2010 ash ratio is a percentage less than the 16-year Facility average of 25.7%. For all of 2010, ash residue from the Facility was used as alternative daily cover at the landfill. This beneficial use of the ash ultimately means that other materials, such as clean soil, do not need to be used for daily cover at the landfill.

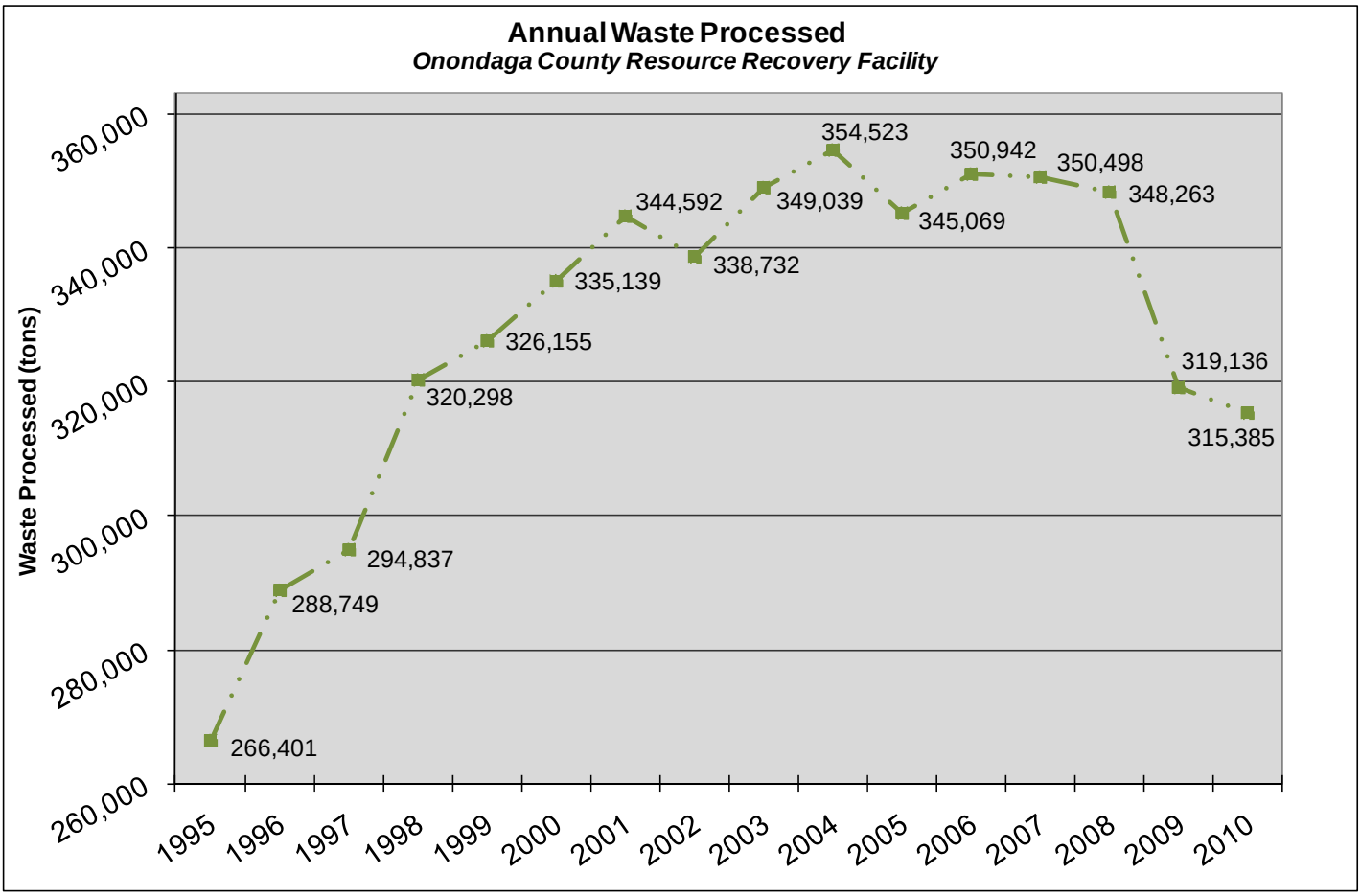
In 2010, the Facility recovered approximately 8,599 tons of ferrous metal, or 2.7% of the refuse processed, for shipment to recycling markets. The non-ferrous metal recovery system, which uses an eddy-current separator, recovered 403 tons of material, of which 333 tons were deemed to be non-ferrous metal – about 0.11% of the refuse processed.

In 2010, the average boiler utilization was 95.8%, indicating that while the boilers were operational, they operated at slightly less than full design levels (due to low trash tonnage). Whenever the boilers are operated at less than full capacity, their efficiency and, therefore steam production, drops. Often times, when there is not enough trash to run all three units at full capacity, one unit is taken offline so that the other units may be operated at full capacity, thereby still maximizing boiler utilization. However, it is not ideal to bring units online and offline too frequently. Another term, steam capacity, is also used to compare boiler utilization, and is defined as the ratio of actual steam to the maximum amount of steam that could be generated if the unit were running full time. For 2010, the Facility's average steam capacity was 78.5%.

3.2 Refuse Processed

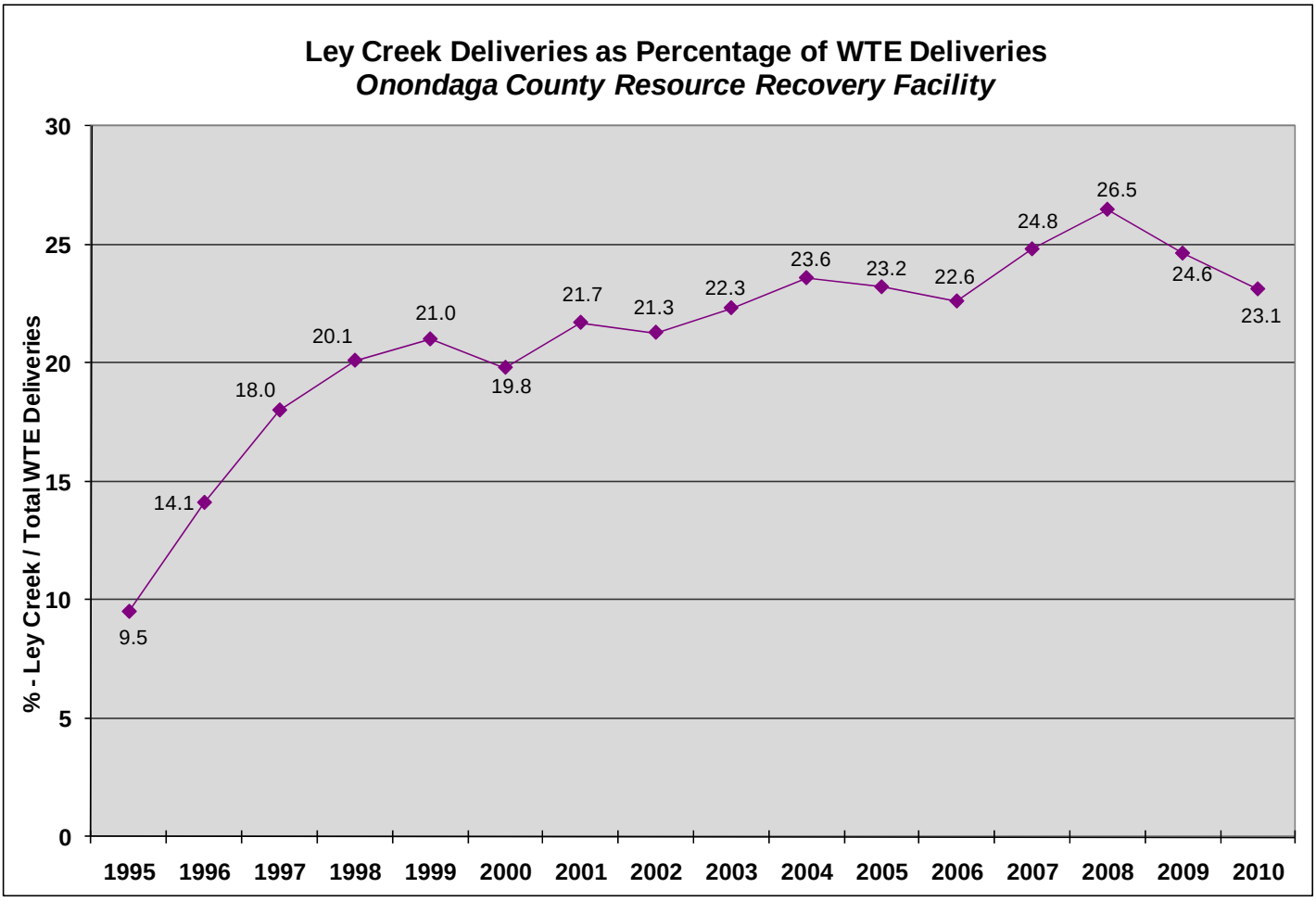
The WTE Facility received 314,474 tons of refuse during 2010, or 95% of OCRRA's total non-recyclable waste tonnage. Only 7 tons, or less than 0.002% of the incoming waste stream, were rejected from the Facility as non-processable waste. Taking into consideration the refuse received and the beginning and ending refuse pit inventory, 315,385 tons of solid waste were processed in 2010. This represents 87.3% of the Facility's permitted throughput limit of 361,350 tons, leaving 45,965 tons of unused processing capacity.

Due to the continuance of the recession from 2009, waste received and processed in 2010 was 1.2% less than the tonnage processed in 2009 and 9.4% less than the tonnage processed in 2008. The figure below shows the historical annual waste processed at the Facility.

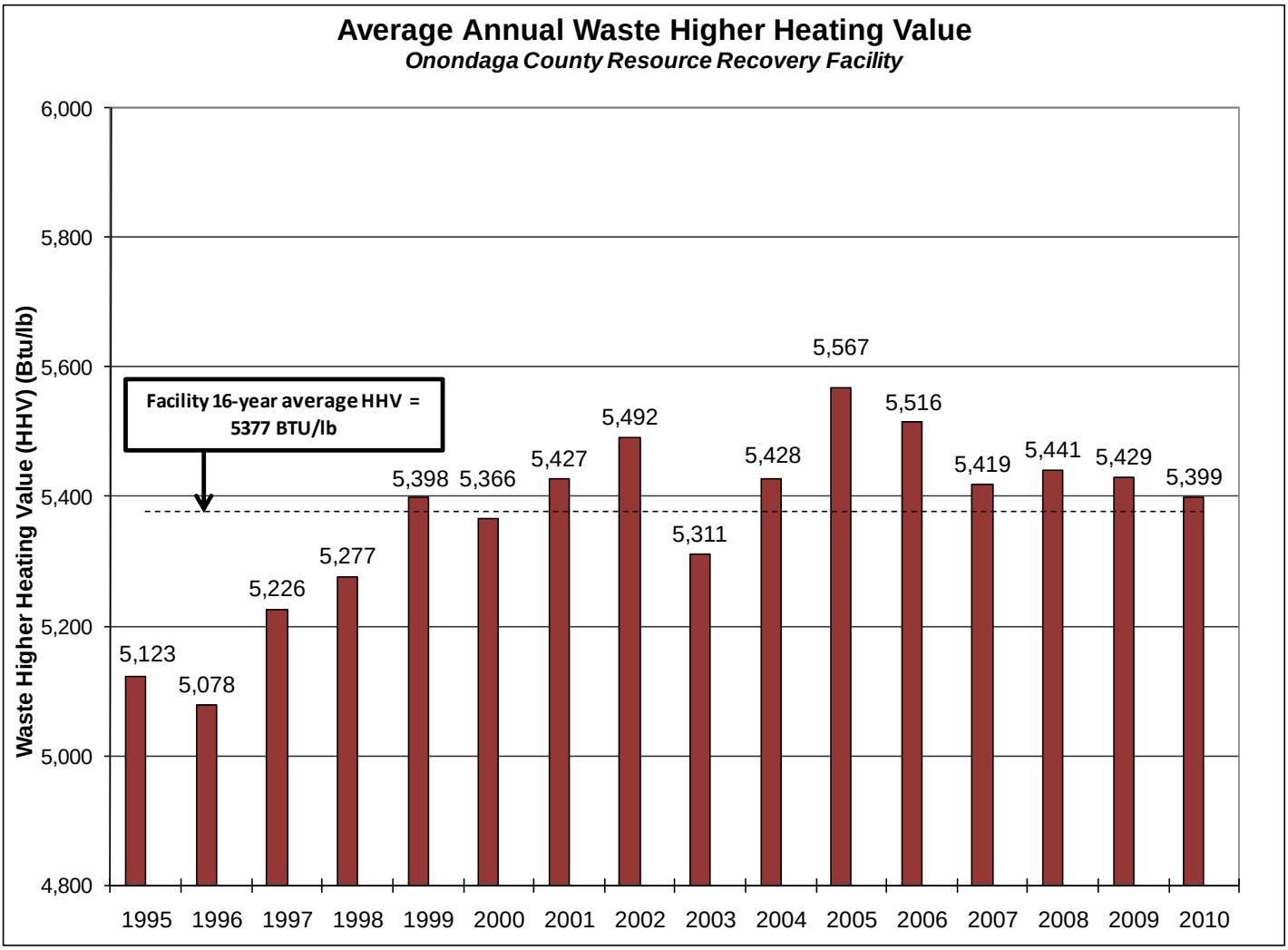


The refuse delivered to the Facility consists primarily of MSW and processable construction and demolition debris (C&D), including roofing. Licensed haulers collect Onondaga County (with the exception of the Town and Village of Skaneateles) MSW and deliver it directly to the Facility. Direct hauler deliveries generally account for about 75% of the tonnage processed. Direct hauler deliveries accounted for 75.6% of the tonnage delivered to the plant in 2010.

In addition to direct hauler MSW deliveries, OCRRA delivers MSW and processable C&D to the Facility from the Ley Creek and Rock Cut Road transfer stations (with the majority from Ley Creek). These deliveries generally account for about 25% of the tonnage processed at the Facility. The 2010 MSW and C&D tonnage delivered to the Facility from OCRRA's transfer stations was 24.4% of the total material delivered to the plant. Ley Creek deliveries as a percentage of total deliveries are shown below.



The average higher heating value (HHV) of waste processed in 2010 was 5,399 British thermal units per pound (Btu/lb). The 2010 average HHV was slightly above the Facility's 16-year average (1995-2010) average HHV of 5,377 Btu/lb (see figure on next page). HHV, which is mainly determined by waste composition and moisture content, is a measure of the amount of energy contained in the waste being combusted. If other boiler operating parameters remain the same, the net effect of a greater waste HHV is increased steam production and, in turn, increased electricity generation.



For 2010, Covanta Energy reported an average HHV of 5,024 Btu/lb for all of its domestic WTE facilities (Covanta, 2011). According to a study of 13 mass burn facilities (including the Onondaga Facility), the average HHV was about 5,200 Btu/lb for years 2003-2008 (LoRe and Oswald, 2009).

OCRRA's average HHV is likely higher than the other averages for two main reasons – 1) the proportion of processable C&D materials and 2) OCRRA's high recycling rate. Other facilities may not process C&D materials, which generally have a higher heating value than MSW, and therefore, if present, tend to increase a facility's average HHV. In contrast, some recyclable materials, such as glass and metal, tend to have a low heating value. By removing these materials from the waste stream, a facility's average HHV will increase. Therefore, OCRRA's highly effective recycling program also plays a role in the Facility's higher-than-average HHV.

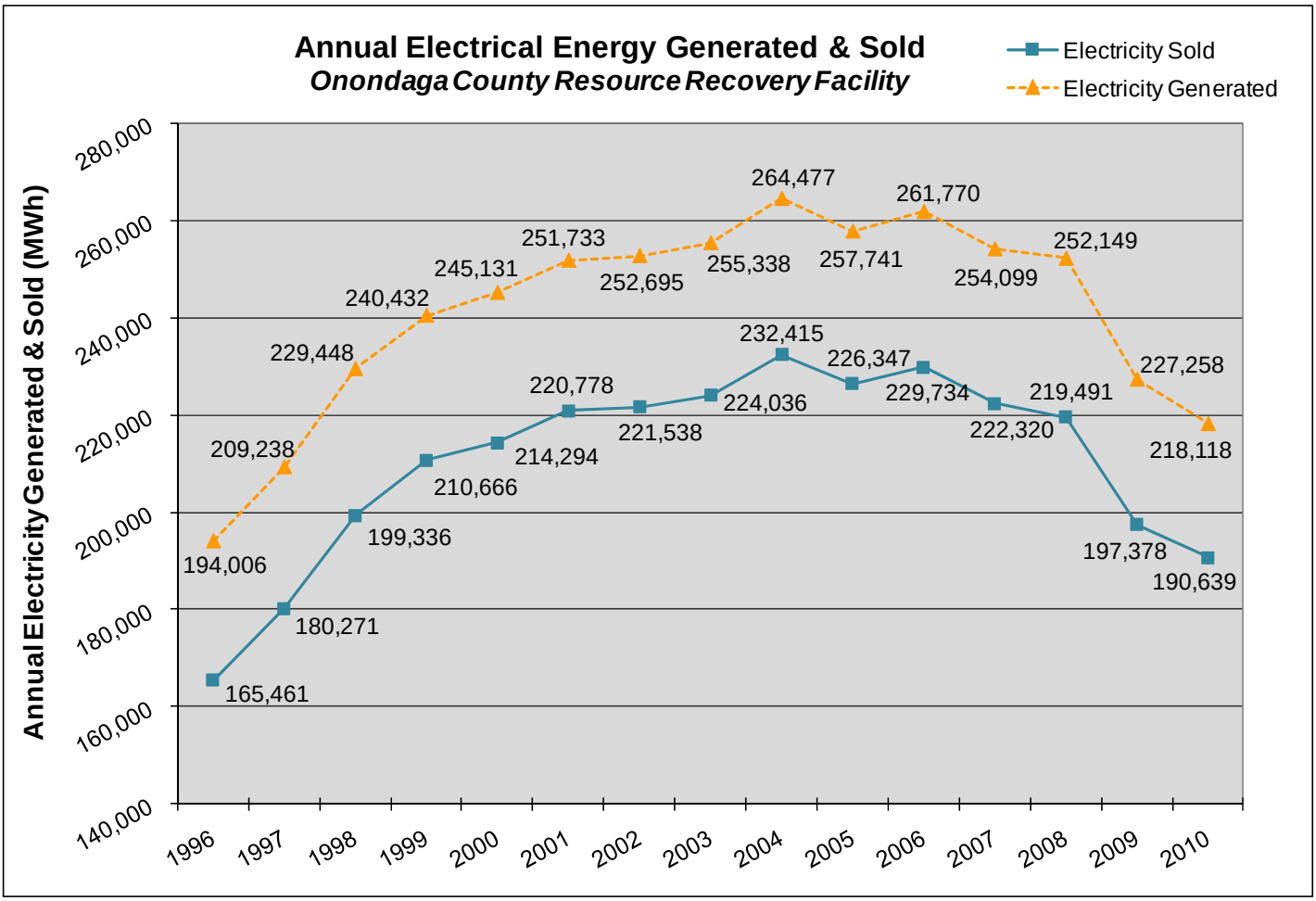
3.3 Steam Generated

Steam generated in 2010 was 2,145,735 kilopounds (klb), or 3.4 pounds of steam per pound of refuse processed. The amount of steam generated depends on the boiler efficiency and HHV of the waste being combusted. Of the total amount of steam generated, 1,981,043 klb were used by the Facility's turbine-generator for electricity production. About 5% is generally consumed for the Facility's internal needs, such as preheating combustion air and heating boiler feedwater. Also, during the 2010 major maintenance event for the turbine-generator, steam bypassed the turbine and was shed to the steam desuperheater and then the air cooled condenser.

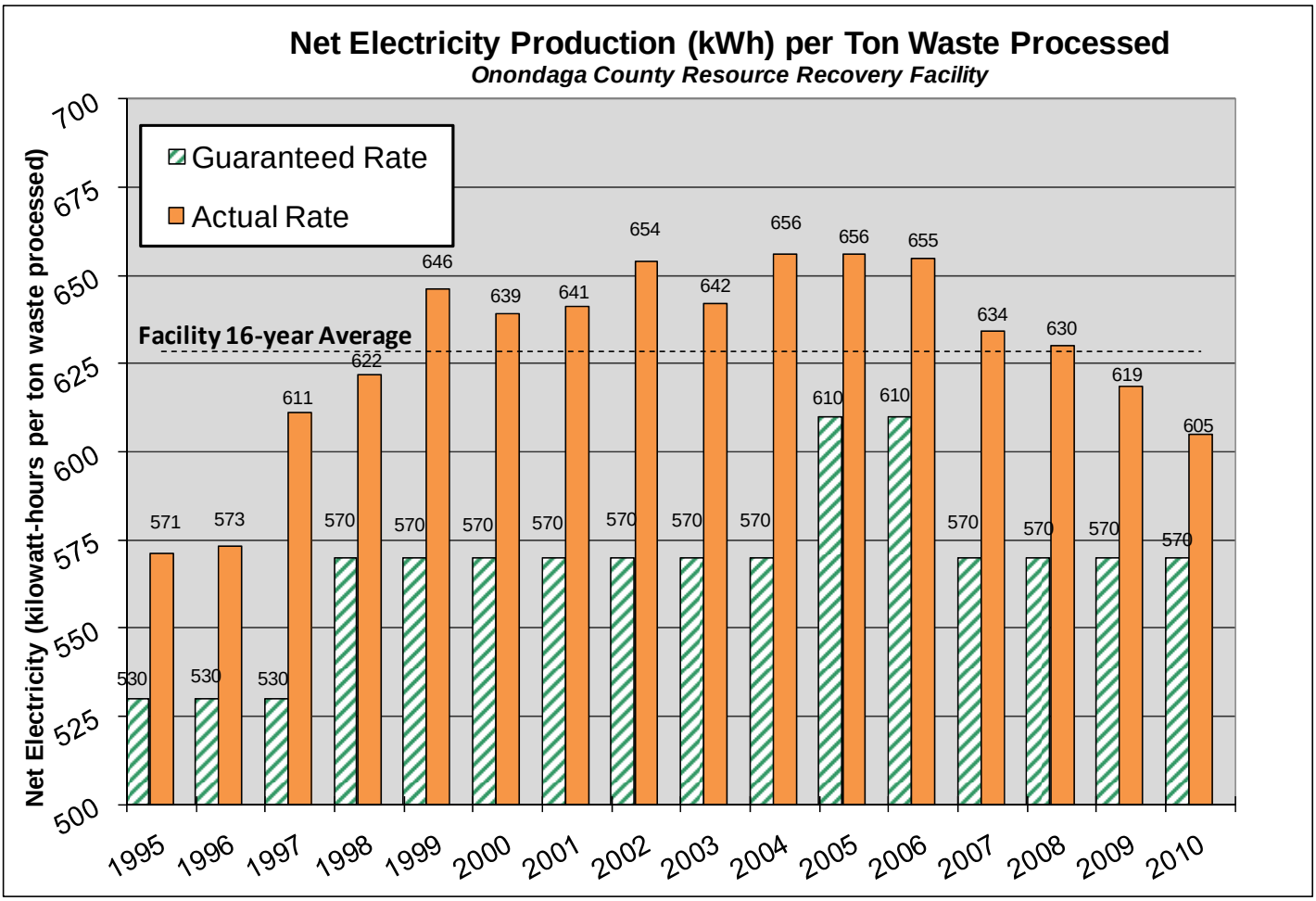
Boiler efficiency, in simplest terms, is the difference between the energy input (HHV of waste being combusted) and energy output (quantity of steam generated). Using monthly data, the 2010 overall boiler efficiency was 71.2%, a value consistent with historical levels and reported literature values.

3.4 Electricity Production

Total (gross) electricity generated for 2010 was 218,118 megawatt-hours (MWh). Of this amount, 190,639 MWh, or 87%, was sold to National Grid (net electricity). The balance, or 13%, was used for the Facility's electrical needs. The amount of electricity sold in 2010 decreased by 3.4% from 2009.



The Facility had a net electricity production of 605 kilowatt-hours per ton of refuse processed (kWh/ton). This rate is below the Facility's 16-year average of 628 kWh/ton; however the difference is primarily due to the turbine-generator system being out of service during the 21-day major overhaul. Furthermore, this rate exceeds the net electricity production guarantee of 570 kWh/ton (based on the average annual HHV of the waste processed, which was 5,399 Btu/lb for 2010). In their benchmarking report, LoRe and Oswald (2009) suggest an average 14-facility (including Onondaga County) net electricity production of 500 kWh/ton.

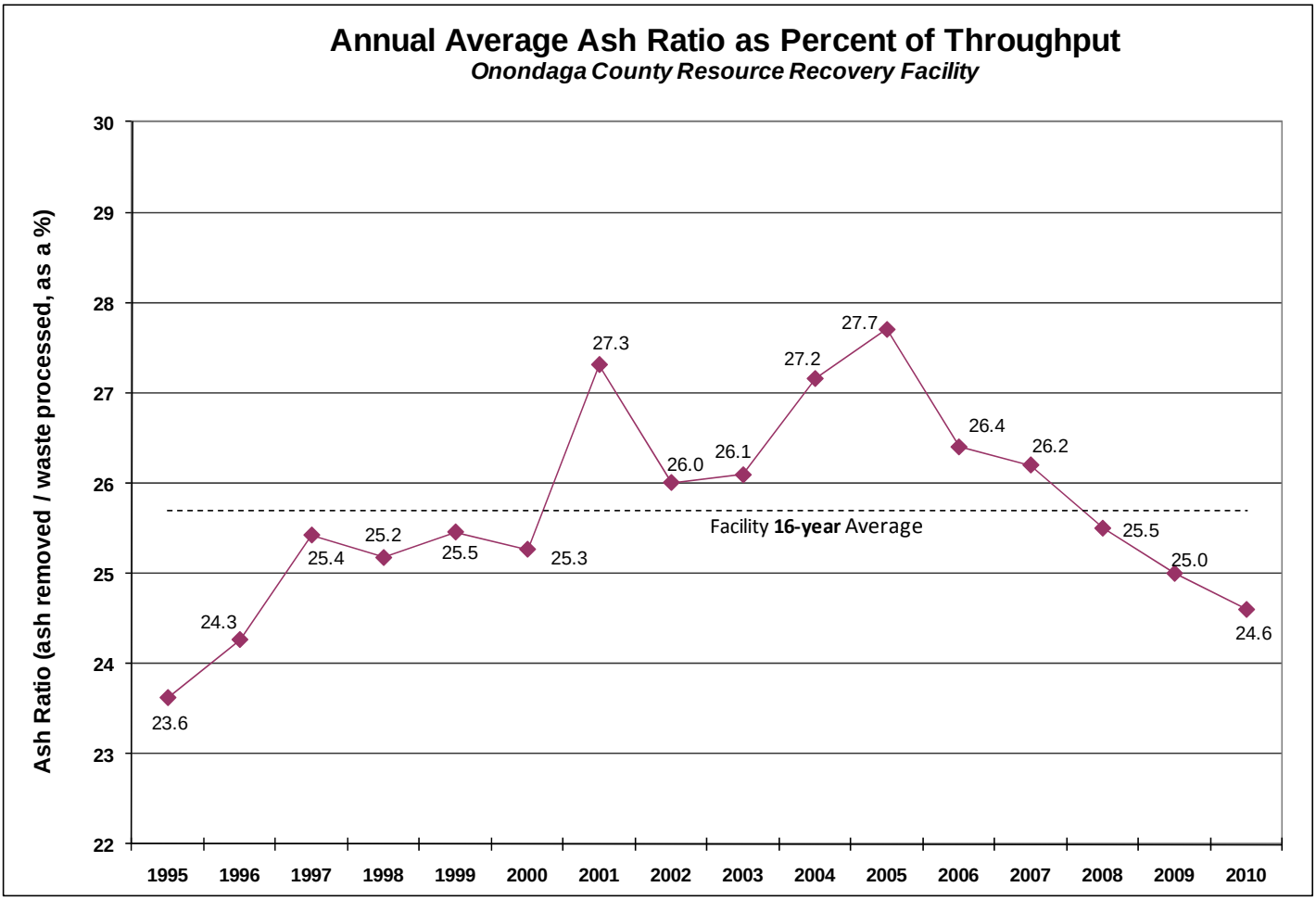


During normal Facility operation, the Facility's electrical demand is satisfied by the Facility's turbine-generator system, with the excess electricity being exported to the grid. Thus, the difference between the gross electricity produced by the turbine-generator and the net electricity sold to the grid is the Facility's electrical demand. In 2010 the Facility used an average of 93 kWh per ton of refuse. This is consistent with the Facility's long-term average, as well as that for other similar facilities. Lore and Oswald (2009) suggest a 14-facility average electricity usage of 90.4 kWh per ton.

3.5 Ash Residue Generation

In 2010, the WTE Facility generated 77,534 tons of combined ash residue, which were hauled by OCRRA to Seneca Meadows Landfill in Waterloo, NY. Based on the waste tonnage processed, this amount of ash was 24.6%; therefore the Facility reduced the weight of the refuse by 75.4%. The 2010 ash ratio (weight basis) is less than the 16-year Facility average of 25.7% and well below the annual contractual limit of 32% (see figure below).

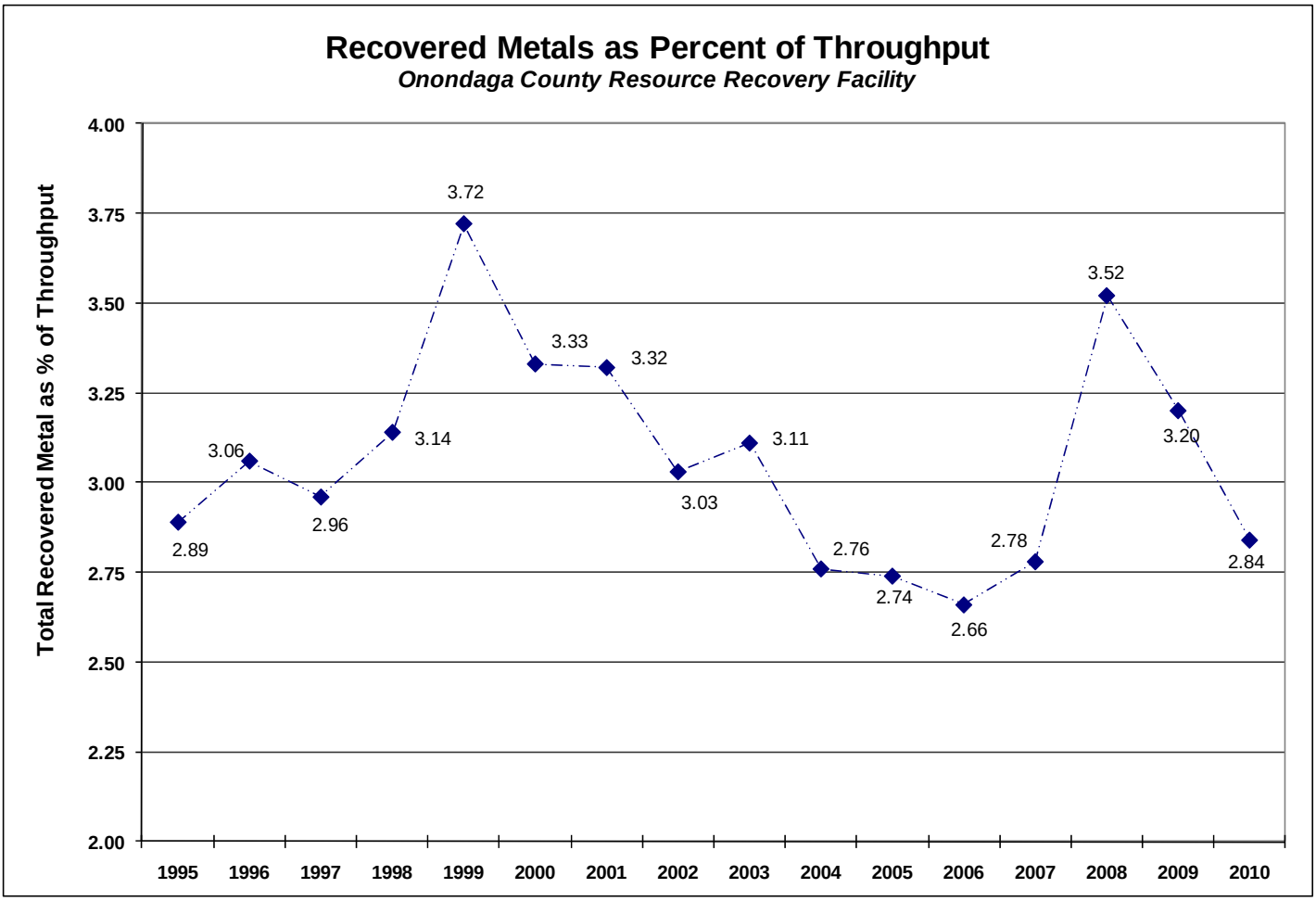
For all of 2010, ash residue from the Facility was used as alternative daily cover at the landfill. This beneficial use of the ash ultimately means that other materials, such as clean soil, do not need to be used for daily cover at the landfill.



3.6 Metal Recovery

In 2010, the Facility recovered approximately 8,599 tons of ferrous metal, or 2.7% of the refuse processed, for shipment to recycling markets. The non-ferrous metal recovery system, which uses an eddy-current separator, recovered 403 tons of material, of which 333 tons were deemed to be non-ferrous metal – about 0.11% of the refuse processed.

The graph below shows the metal recovery over the life of the Facility.



As shown, the annual quantity of recovered metal has varied over time. Recovery of metal is dependent upon the amount of metals in the incoming waste stream, as well as on the effectiveness of the Facility's metal recovery systems.

3.7 Boiler and Turbine-Generator Availability

Though the boilers and turbine-generator are designed to operate 24 hours a day, 365 days per year, a WTE facility cannot realistically achieve 100% boiler availability because of necessary routine and periodic maintenance. Boiler and turbine-generator availability are generally defined as the percentage of hours that the boiler/turbine-generator is available for operation, taking into account downtime related to scheduled and unscheduled maintenance. Downtime related to low refuse deliveries is not generally counted against availability. This is consistent with industry standards (LoRe and Oswald, 2009).

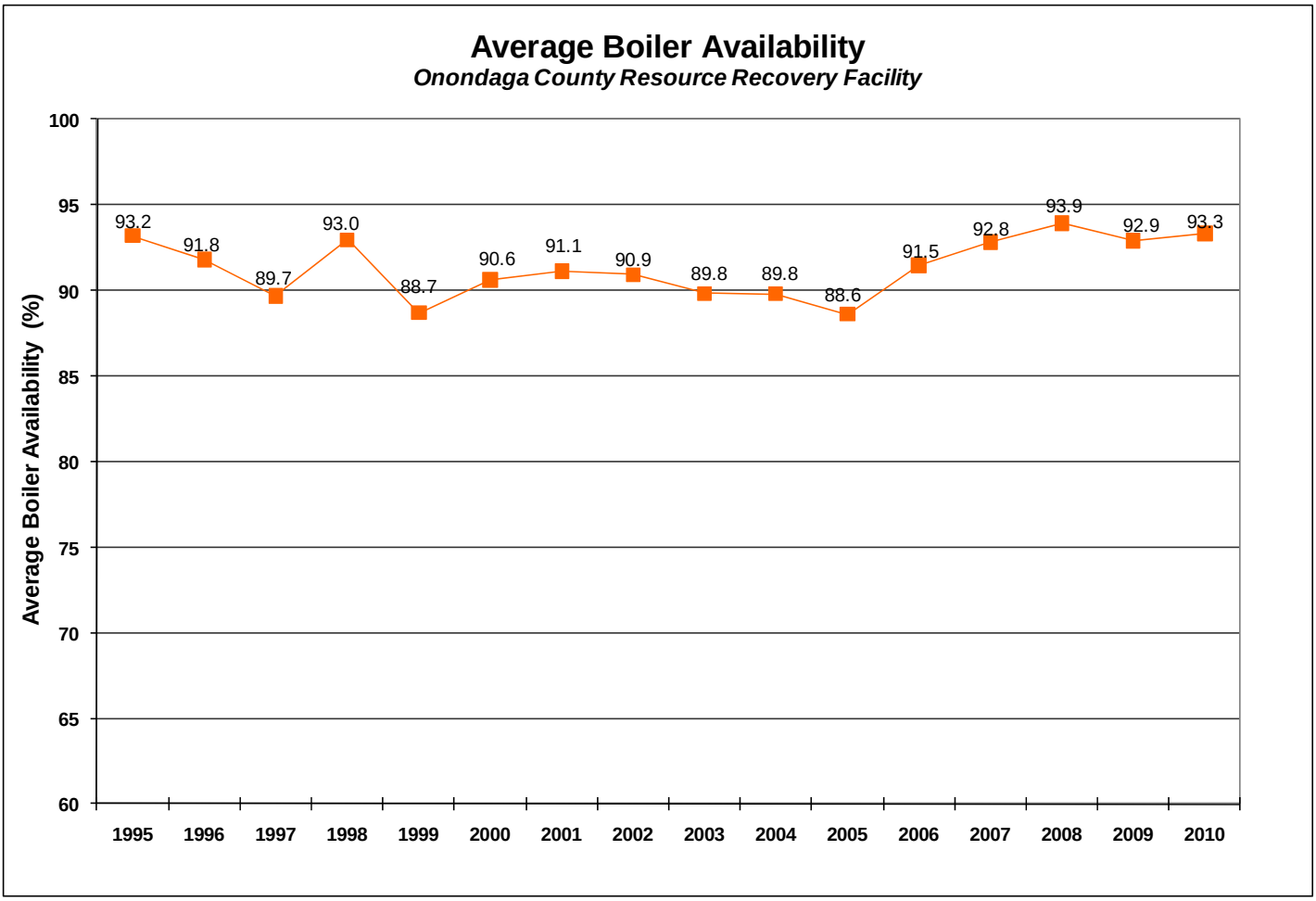
Facility boiler and turbine-generator availability are reported monthly and annually. 2010 availability information is presented below:

	Boiler Unit #1	Boiler Unit #2	Boiler Unit #3	Turbine/Generator
Total Scheduled Downtime (hr)	463.2	515.1	475.1	445.4
Total Unscheduled Downtime (hr)	172.9	42.4	83.6	41.6
Total Downtime (hr)	636.1	557.5	558.7	487.0
Total Downtime (days)	26.5	23.2	23.3	20.3
Availability (%)	92.7	93.6	93.6	94.4

Overall average boiler availability for 2010 was 93.3%, a fair bit higher than the Facility's 16-year (1995-2010) average of 91.4%. The 2010 average boiler availability reflects less downtime for scheduled boiler maintenance and equipment malfunctions. For comparative purposes, Covanta reported that their domestic WTE facilities had an average boiler availability of 91.2% for 2010 (Covanta, 2011) and LoRe and Oswald (2009) suggest a 15-facility average (including Onondaga County) of 90.3%.

Covanta has historically performed, and continues to perform, necessary boiler maintenance consistent with industry standards. Performing preventative maintenance remains critically important in prolonging the useful life of the boiler; replacing and repairing worn components prevents unscheduled downtime, thereby increasing boiler availability. Scheduled maintenance accounted for 73%, 92%, and 85% of downtime for Unit 1, 2, and 3, respectively. Unscheduled boiler downtime in 2010 resulted mainly from waterwall and superheater tube leaks.

The figure on the next page shows the Facility's historical average boiler availability.



The table below presents a summary of historical scheduled and unscheduled total boiler downtime.

Historical Boiler Operating Data (total hours for three boilers)

Year	Scheduled Maintenance (hours)	Unscheduled Maintenance (hours)	Total Maintenance (hours)	Total Maintenance Downtime* (%)	Downtime due to low trash deliveries (hours)	Low Trash Downtime* (%)	Total Downtime (hours)	Total Downtime* (%)
1996	1,964	196	2,160	8.2	6,954	26.5	9,114	34.7
1997	2,124	586	2710	10.3	5,985	22.7	8,695	33.0
1998	1,262	588	1850	7.0	3,541	13.5	5,391	20.5
1999	1,873	1,101	2974	11.3	3,585	13.6	6,559	25.0
2000	1,728	745	2473	9.4	1,652	6.3	4,125	15.7
2001	1,991	338	2329	8.9	2,011	7.6	4,340	16.5
2002	1,998	383	2381	9.1	1,052	4.0	3,433	13.1
2003	1,958	714	2672	10.2	1,034	3.9	3,706	14.1
2004	1,954	738	2692	10.2	777	3.0	3,469	13.2
2005	2,373	790	3163	12.0	218	0.8	3,381	12.8
2006	1,688	551	2239	8.5	171	0.7	2,410	9.2
2007	1,321	565	1886	7.2	151	0.6	2,037	7.8
2008	1,337	264	1,601	6.1	920	3.5	2,521	9.6
2009	1,546	318	1,864	7.1	1,859	7.1	3,723	14.2
2010	1,453	299	1,752	6.7	2,978	11.3	4,730	18.0

* Total Maintenance Downtime, Low Trash Downtime, and Total Downtime computed as a percentage of total unit-hours in calendar year.

The 2010 unscheduled and scheduled downtime represented 6.7% of total annual hours. The downtime due to low trash levels represents an additional 11.3%. Total boiler downtime, including downtime due to low trash deliveries, for 2010 was 4,730 hours, or 18% of the unit-hours in the calendar year.

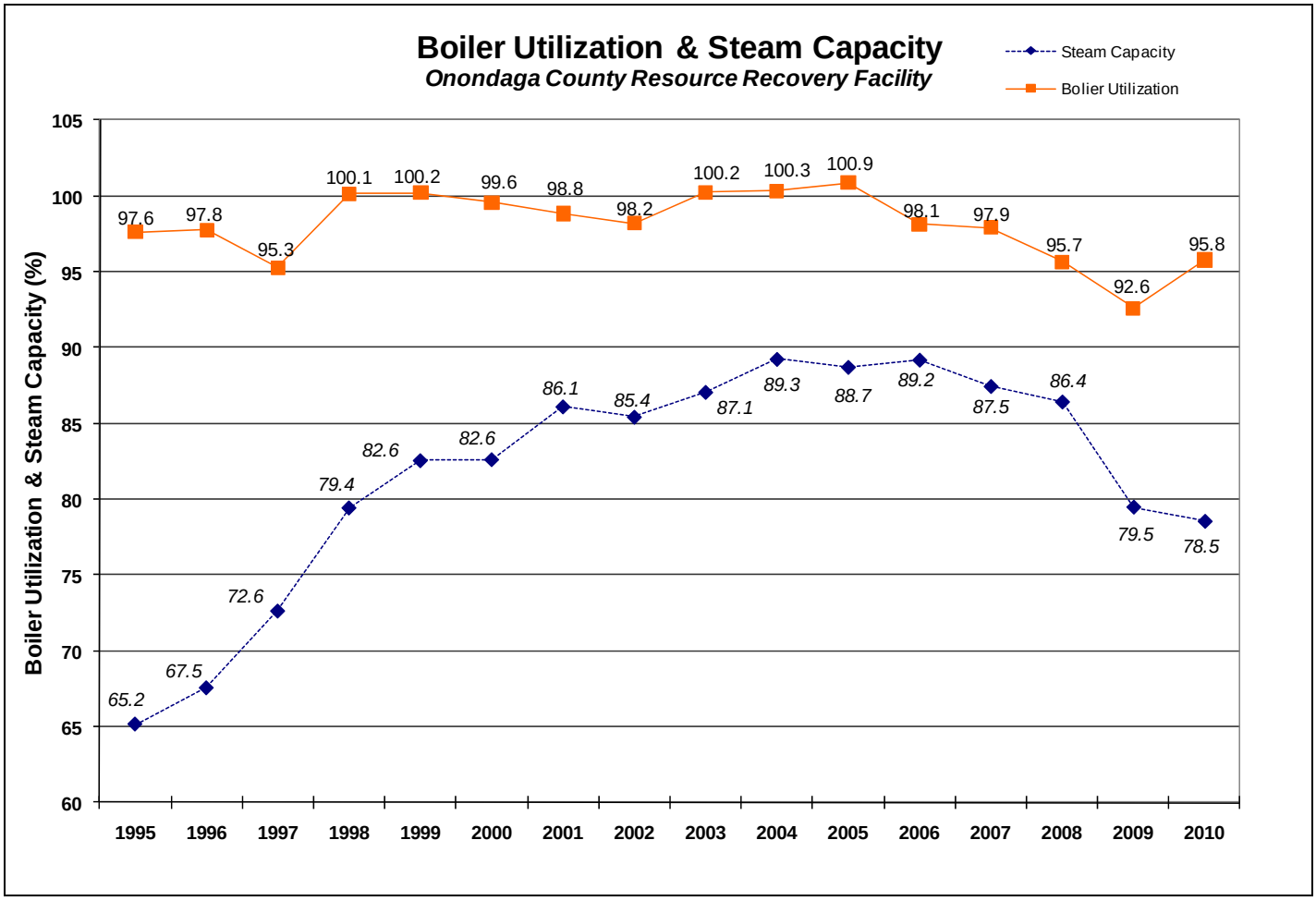
Turbine-generator availability for 2010 was 94.4% with 487 hours of downtime. Unscheduled outages accounted for only 8.5% of the downtime, while the the major turbine-generator outage accounted for the rest. For comparative purposes, LoRe and Oswald (2009) suggest a 14-facility average (including Onondaga County) of 96.6%.

3.8 Boiler Utilization and Steam Capacity

Another metric used to evaluate Facility efficiency is boiler utilization. Each boiler is designed with a maximum steam rate (pounds per hour) at which the unit is intended to be operated. This is referred to as the "maximum continuous rating" (MCR). The maximum design steam rating for the Onondaga Facility is 103,950 lb of steam per hour per boiler, or 311,850 lb of steam per hour for all three boiler units. Boiler utilization is the ratio of actual steam generated by the boilers to the MCR. It is important to note that boiler utilization only takes into account boiler operating time; that is, it does not include boiler downtime. Another term, steam capacity, is also used to evaluate Facility efficiency, and is defined as the ratio of actual steam to the maximum amount of steam that could be generated if the unit were running full time.

For 2010, the average boiler utilization was 95.8%. Boiler utilization of 100% represents the most efficient mode of Facility operation, and will maximize steam production and thus electrical energy generation. Anything less than 100% indicates that while the boilers were operational, they were being utilized at less than their full steaming capacity. It is not optimal to frequently bring boilers on- and off-line, so an alternative for dealing with low trash levels is to run the boilers at less than full capacity. This was the case for 2010. For comparative purposes, LoRe and Oswald (2009) suggest a 14-facility average (including Onondaga County) boiler utilization of 96.0%. Steaming capacity, which also takes into consideration steam "lost" from boiler downtime, for 2010 was 78.5%. Due to low trash tonnage, the steaming capacity was at the lowest level since 1998 (at which point the permitted throughput capacity of the Facility was significantly lower).

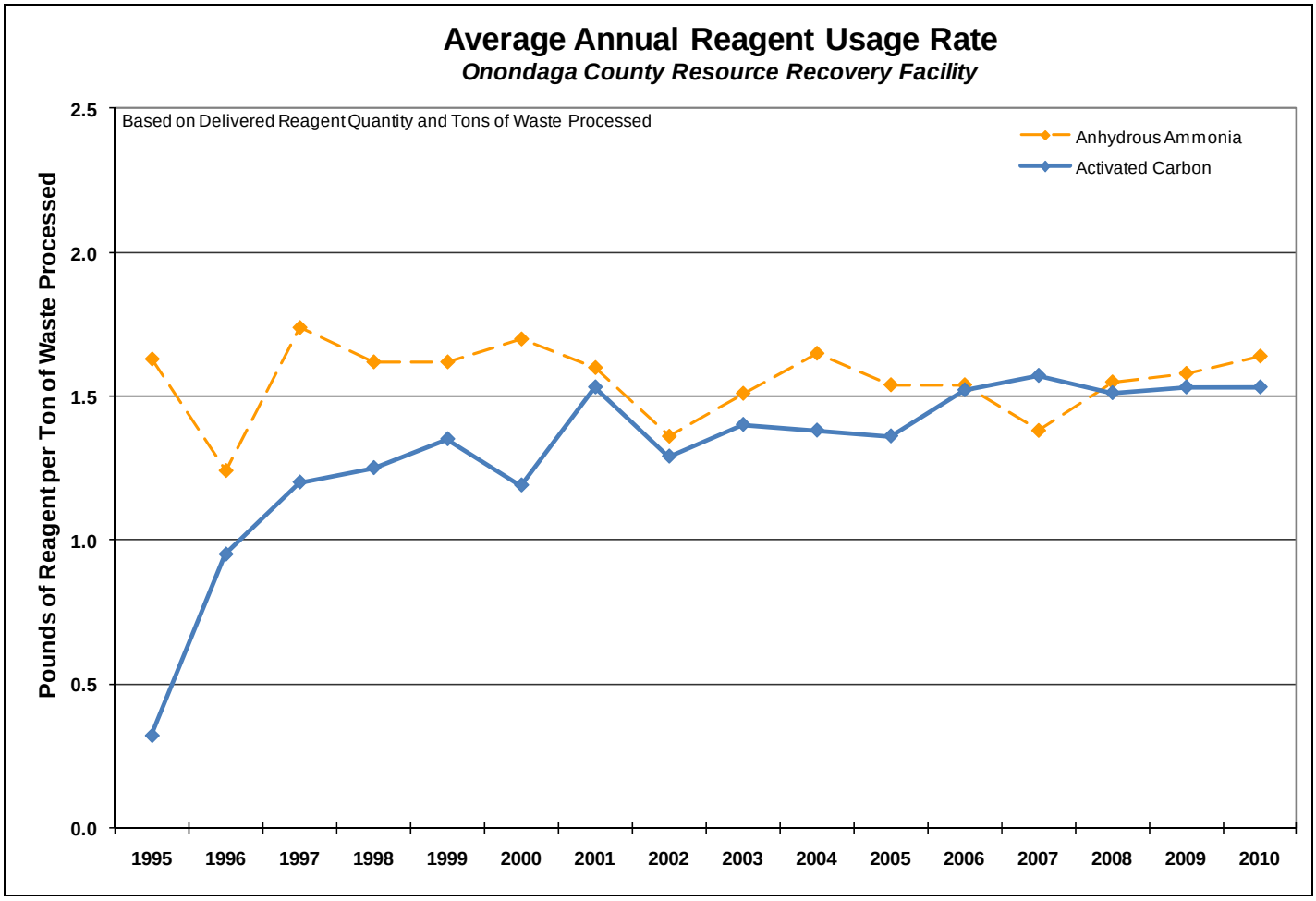
Historical data for boiler utilization and steam capacity are shown in the figure on the next page.



3.9 Pollution Control Reagent Consumption

The Facility uses several reagents for pollution control including anhydrous ammonia for control of nitrogen oxides (NO_x), activated carbon for mercury and dioxin/furan control, and lime for control of acid gases (as well as ash conditioning).

To control NO_x emissions, anhydrous ammonia is injected into the combustion chamber of each boiler unit. To control mercury emissions, as well as dioxin and furan emissions, powdered activated carbon is mixed into slurry and injected into the spray-dry scrubbers through the rotary atomizer. The rotary atomizer creates tiny droplets for optimal reaction. The average annual 2010 reagent usage rates for ammonia and carbon were 1.64 lb and 1.53 lb per ton of waste processed, respectively. As evident in the chart below, anhydrous ammonia and carbon usage rates have been consistent with historical rates. According to Lore and Oswald (2009), the Facility's anhydrous ammonia usage rate is consistent with other facilities that use anhydrous ammonia and the carbon usage is a bit higher than a 12-facility average (including Onondaga County) of 1.01 lb per ton.



To neutralize acid gases, namely sulfur dioxide (SO_2), hydrogen chloride (HCl), hydrogen fluoride (HF), and sulfuric acid (H_2SO_4), a calcium-based lime, referred to as pebble lime, is injected into the spray-dry scrubbers through the rotary atomizer. In 2010, the average reagent application rate was 29.4 lb of pebble lime per ton of waste processed. This is consistent with 2002 (31.0 lb of pebble lime per ton of waste processed) and 2007–2009 (29.2, 28.4, and 30.3 lb of pebble lime per ton of waste processed, respectively) when pebble lime was the only form of lime used.

Prior to making the decision to solely use pebble lime, dolomitic lime, a lime with a higher magnesium content than pebble lime, was added to the fly ash prior to combining with the bottom ash to provide additional conditioning of the fly ash. In August 2006, dolomitic lime use was discontinued and the reagent application rate for pebble lime increased above that needed for acid gas control. While still providing satisfactory ash conditioning, this change was implemented to improve housekeeping conditions, reduce OCRRA's overall ash conditioning costs, and produce a drier, more manageable combined ash residue for disposal. In 2009, Covanta also experimented with another type of lime (in conjunction with pebble lime) called carbide lime but found it to be too abrasive.

3.10 Electricity, Natural Gas, and Water Utilization

During normal Facility operation, the Facility's electrical demand is satisfied by the Facility's turbine-generator system, with the excess electricity being exported to the grid. During those times when the turbine-generator is off-line due to maintenance or malfunction, electricity is purchased from National Grid to operate the Facility and continue combusting the incoming MSW. In 2010, 1,239,156 kWh of electricity was purchased from National Grid for in-plant needs, with the majority of the usage occurring during the planned major turbine-generator outage. The amount of electricity purchased during 2010 is significantly more than the Facility average, due to the relatively long maintenance period that the turbine-generator was offline. The Service Agreement allows for 3,348,000 kWh over a three-year rolling period and at the end of 2010 the Facility had used 2,035,350 kWh for 2008-2010.

Natural gas is an auxiliary fuel used for boiler start-ups and shutdowns, and for maintaining minimum furnace temperatures when processing overly wet waste. 2010 natural gas usage was 137,520 therms, which is consistent with historical consumption. Under the Service Agreement, OCRRA is responsible for the first 110,000 therms and Covanta pays for usage in excess of 110,000 therms.

City water satisfies all potable and process needs of the Facility, with the majority being for process use. However, the Facility is a zero discharge plant relative to process wastewater; meaning that only sanitary sewage is discharged off-site. 30,360,000 gallons were used in 2010. This amount of water translates into about 96 gallons per ton of waste combusted or approximately 58 gallons per minute. 2010 water usage remained consistent with historical levels and design parameters following initial start-up. The Onondaga Facility's water use is much lower than that of similar facilities because it is a zero-process water discharge Facility, meaning that all process water gets treated and reused thereby requiring less potable water. According to LoRe and Oswald (2009), a ten-facility average water consumption rate is 422 gallons per ton of waste processed.

3.11 Facility Inspections

In accordance with NYSDEC Part 360 regulations, an annual general Facility inspection must be undertaken to determine the operating condition of the safety, emergency, security, process, and control equipment. Covanta must have this inspection performed under the direction of a New York State licensed professional engineer. With the approval of the NYSDEC, Covanta performed the required Facility annual inspection on November 19, 2010. Covanta's Regional Vice President, Steven J. Bossotti, P.E., certified: "*Based upon the above inspections and information, the safety, emergency, security, process and control equipment at the Onondaga County Resource Recovery Facility operated by Covanta Onondaga at 5801 Rock Cut Road, Jamesville, NY 13078 are considered to be in acceptable operating condition.*" This annual inspection report was submitted to the NYSDEC on February 25, 2011 as part of the Facility's 2010, 4th Quarter & Annual Solid Waste Report.

NYSDEC also conducted several facility inspections in 2010. The Division of Air Resources was on site for the annual stack testing activities. On July 22, 2010 the Division of Solid and Hazardous Materials visited the Facility for an inspection.

In 2010, OCRRA had its independent consultant, CDM, conduct a comprehensive 2-day site inspection. This visit focused on all various aspects of plant operations and maintenance, and coincided with the Unit #2 spring boiler outage. Based on the results of their visual inspection and experience at other WTE facilities, CDM opined that the Onondaga Facility continues to be well maintained, and is in overall good operating condition. The routine preventative maintenance and major repairs performed are consistent with the type and level of repairs observed at other facilities. The systems inspected were in good operating condition, and all equipment appeared to be operating properly. The level of daily repair and preventative maintenance observed was considered normal for facilities of the same type and age.

CDM also conducted a comprehensive 2-day site inspection during the major turbine-generator outage. The purpose of this site visit was to inspect the condition of the steam turbine and electrical generator and to observe the maintenance activities that were being undertaken during the outage. Based on this inspection, CDM concluded that the turbine-generator was in good overall condition, with no major problems reported. They also concluded that the type and level of maintenance work performed during this major outage was consistent with the age of the equipment and similar work observed at other facilities.

In addition to Facility inspections, CDM performed oversight for the annual air emissions stack testing and semi-annual ash residue testing. CDM concluded that testing was conducted in accordance with required procedures and protocols.

Section 4 – Environmental Performance

4.1 Summary of Environmental Performance

Operating under one of the strictest WTE air permits in the country, the Onondaga County WTE Facility turns the County's non-recyclable trash into energy. Over 15 million dollars worth of air pollution control equipment is in place to make sure this is done safely. Results of the Facility's annual air emissions and ash residue test results consistently demonstrate the Facility's exemplary environmental track record. Coupled with Onondaga County's nationally high recycling rate of 60% in 2010 (nearly double the national average), the Onondaga County WTE Facility generates enough renewable energy to satisfy the needs of approximately 30,000 homes in OCRRA's service area while also reducing the volume of trash that needs to be landfilled by 90%.

4.2 2010 Stack Test Results

Stack testing is an important tool that measures the amount of regulated pollutants being emitted from the Facility. Stack testing consists of a series of sampling events, in which a probe is inserted into the stack gases to collect a representative sample, over a defined amount of time. Sampling and laboratory analysis are conducted in accordance with NYSDEC and USEPA protocols. NYSDEC oversees stack testing at the WTE Facility.

In addition to annual stack testing, the Facility has a continuous emission monitoring system (CEMS) that measures equipment performance and stack emissions in order to constantly track Facility performance. The CEMS tracks carbon monoxide, carbon dioxide, oxygen, sulfur dioxide, and NOx as well as ammonia, opacity, and combustion temperatures.

The 2010 stack testing consisted of the 10 parameters that are tested annually. The results from the 2010 stack testing indicate that the Facility is operating acceptably and that the air pollution control devices are functioning properly. As shown by the summary data on the next page, many of the parameters were considerably below the permit limit.

2010 ANNUAL STACK TEST RESULTS

		Constituent	Average Measured Emissions[1] Unit 1	Unit 2	Unit 3	Permit Limit[2]	Pass/Fail? P/F
TESTED ANNUALLY	FEDERAL	Cadmium (mg/dscm @ 7% O_2)	5.50E-04	6.51E-03	1.26E-03	3.50E-02	P
		Cadmium (lb/hr)	8.41E-05	1.06E-03	2.05E-04	1.90E-03	P
		Carbon Monoxide (lb/hr)	1.01E+00	1.09E+00	1.01E+00	8.04E+00	P
		Dioxins/Furans (ng/dscm @ 7% O_2)	2.42E+00	1.03E+00	1.30E+00	3.00E+01	P
		Hydrogen Chloride (ppmdv @ 7% O_2)	3.11E+00	2.37E+00	2.53E+00	2.50E+01	P
		Hydrogen Chloride (lb/hr)	7.14E-01	5.94E-01	6.39E-01	5.24E+00	P
		Hydrogen Chloride Removal Efficiency (%)	99.5	99.7	99.7	>=95	P
		Lead (mg/dscm @ 7% O_2)	1.44E-02	1.13E-01	2.36E-02	4.00E-01	P
		Lead (lb/hr)	2.22E-03	1.84E-02	3.84E-03	3.81E-02	P
		Mercury (lb/hr)	< 9.49E-05	4.21E-04	1.35E-04	4.00E-03	P
		Nitrogen Oxides (lb/hr)	5.44E+01	5.25E+01	5.32E+01	5.80E+01	P
		Particulates (gr/dscf @ 7% O_2)	2.99E-04	6.81E-03	1.52E-03	1.00E-02	P
		PM_{10} (gr/dscf @ 7% O_2)	4.65E-04	3.18E-03	5.33E-04	1.00E-02	P
		PM_{10} (lb/hr)	1.87E-01	1.29E+00	2.17E-01	3.16E+00	P
		Sulfur Dioxide (lb/hr)	3.47E+00	6.91E+00	4.50E-01	1.62E+01	P
	STATE	Ammonia (ppmdv @ 7% O_2)	< 8.90E-01	< 8.44E-01	1.47E+00	5.00E+01	P
		Ammonia (lb/hr)	< 9.55E-02	< 9.87E-02	1.73E-01	4.88E+00	P
		Dioxins/Furans-2,3,7,8 TCDD TEQ (ng/dscm @ 7% O_2)	2.79E-02	2.27E-02	1.58E-02	4.00E-01	P
		Dioxins/Furans-2,3,7,8 TCDD TEQ (lb/hr)	4.47E-09	3.38E-09	2.61E-09	1.29E-07	P
		Mercury (µg/dscm @ 7% O_2)	< 6.20E-01	2.59E+00	8.20E-01	2.80E+01	P
		Mercury Removal Efficiency (%)	> 99.5	98.2	99.1	>=85	P
		Zinc (lb/hr)	6.73E-03	6.97E-02	4.21E-02	1.42E-01	P

NOTES:

[1] Based on three test runs

[2] NYSDEC Title V Permit #7-3142-00028/00009 - Draft Renewal

UNITS:

gr/dscf = grains per dry standard cubic foot

ppmdv = parts per million dry volume

lb/hr = pounds per hour

ng/dscm = nanograms per dry standard cubic meter

µg/dscm = micrograms per dry standard cubic meter

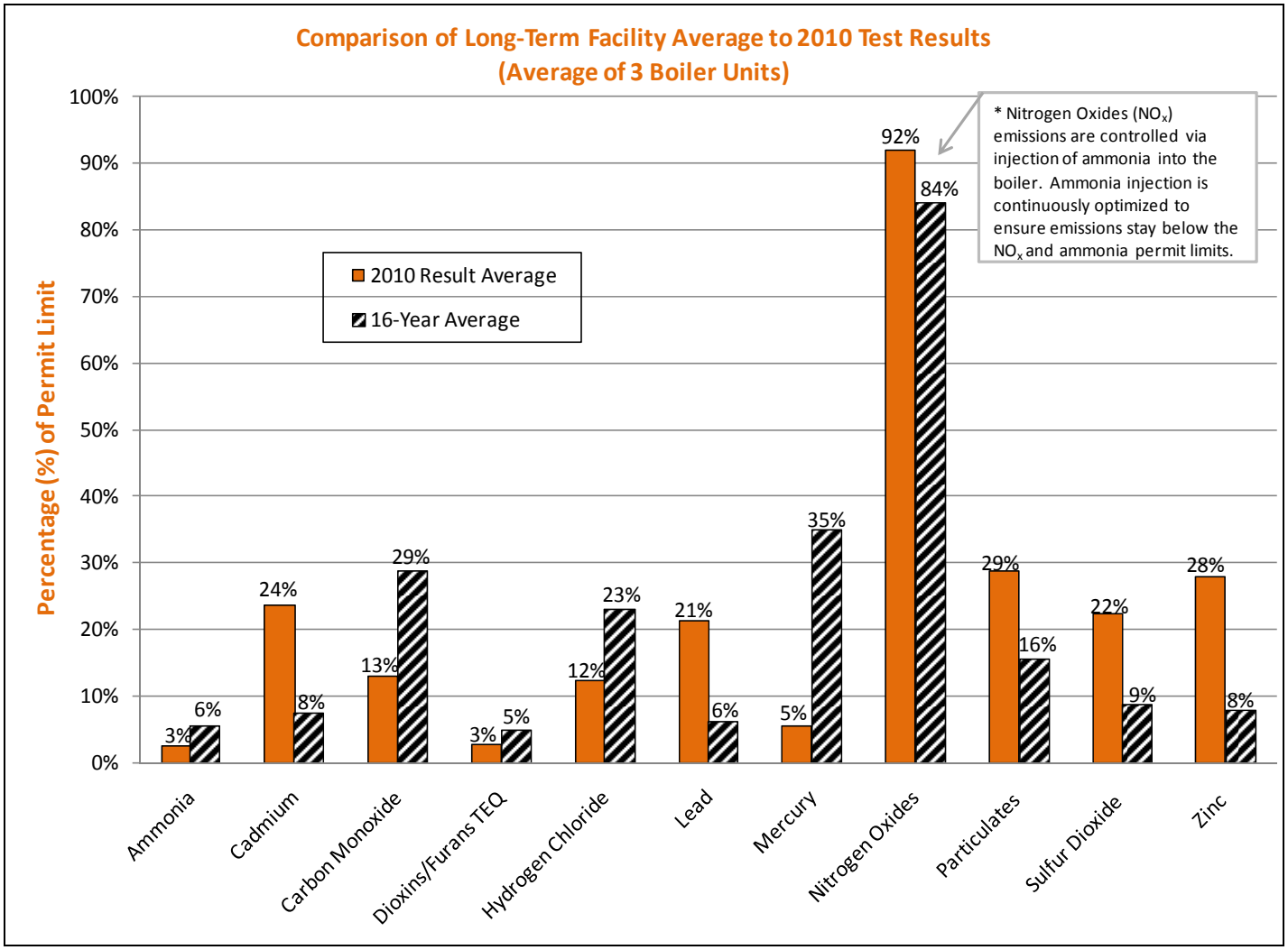
mg/dscm = milligrams per dry standard cubic meter

@ 7% O_2 = concentration corrected to 7% oxygen

4.2.1 Parameters Tested Annually

The figure below presents a comparison of the 2010 stack test results with their respective long-term (16-year) Facility averages (1995 through 2010) for the parameters tested annually. The results are graphed as a percentage of their respective permit limits. The graph shows that the 2010 results continue to be well below regulatory limits. These results indicate that the Facility's air pollution control system continues to operate effectively, and that OCRRA's efforts in screening the incoming waste continue to be effective.

Compared to the other parameters, NO_x emissions are much closer to the permit limit. This is because NO_x emissions are controlled via injection of ammonia into the boiler. Ammonia injection is continuously optimized to ensure emissions stay below the NO_x and ammonia permit limits.



WTE facilities have significantly reduced emissions over the past decade. In 1997 a memorandum by the United States Environmental Protection Agency (USEPA) documented this progress. The table from USEPA's memorandum is provided on the following page.

Emissions From Large and Small MWC Units

Pollutant	*1990 Emissions (tpy)*	*2005 Emissions (tpy)*	*Percent Reduction*
CDD/CDF, TEQ basis*	4400	15	99+ %
Mercury	57	2.3	96 %
Cadmium	9.6	0.4	96 %
Lead	170	5.5	97 %
Particulate Matter	18,600	780	96 %
HC1	57,400	3,200	94 %
SO_2	38,300	4,600	88 %
NO_x	64,900	49,500	24 %

(*) dioxin/furan emissions are in units of grams per year toxic equivalent quantity (TEQ), using 1989 NATO toxicity factors; all other pollutant emissions are in units of tons per year.

Source: USEPA Memorandum dated 1997

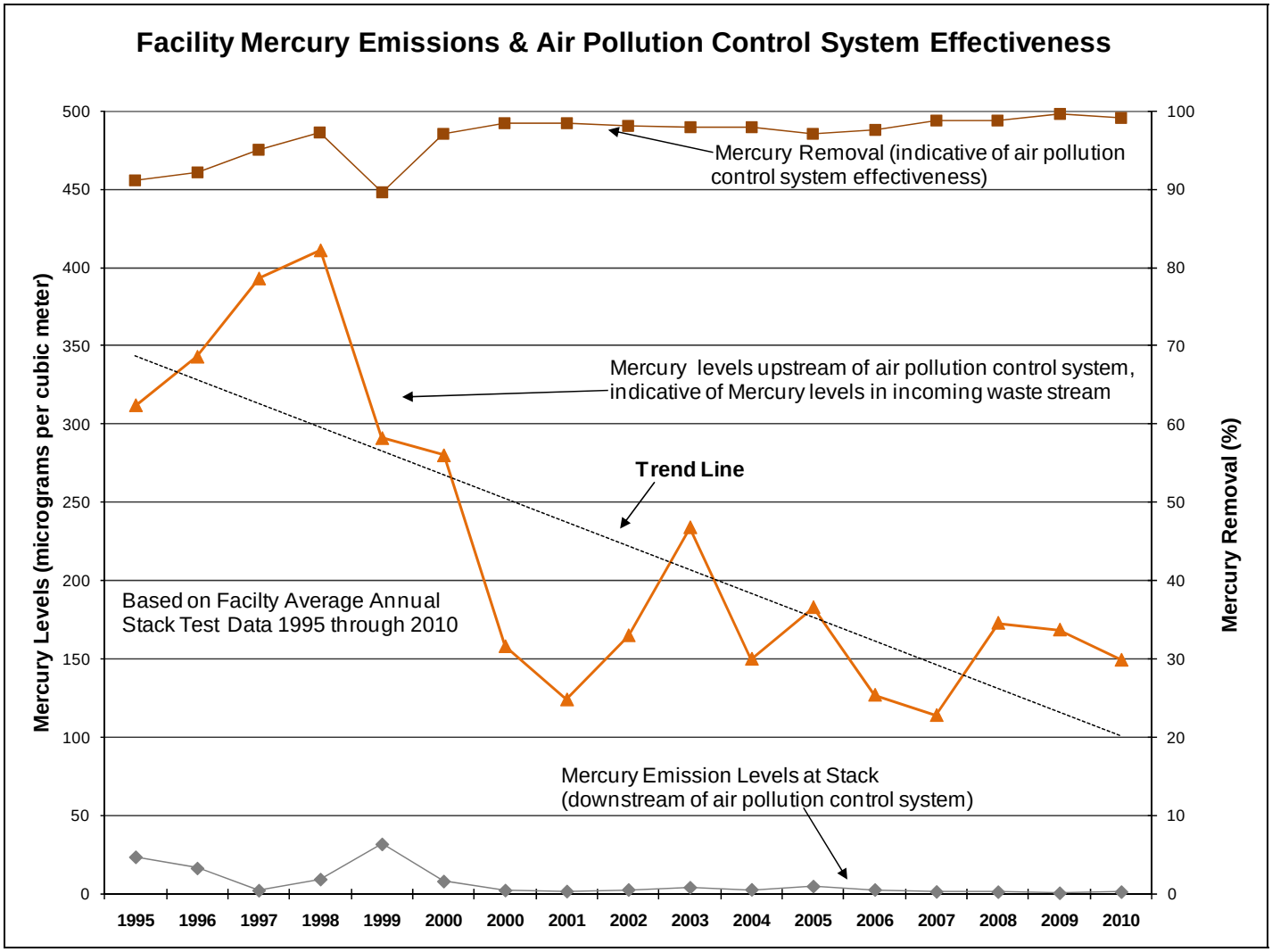
Some of these parameters will be discussed in further detail the following sections.

4.2.2 Mercury

To control mercury emissions, powdered activated carbon is mixed into slurry and injected into the spray-dry scrubbers through a rotary atomizer, which creates tiny droplets. The activated carbon reacts with the mercury in the gas exiting the boiler and forms particles that are captured in the baghouse. Still considered the most highly advanced control technology, activated carbon injection has been used at WTE facilities for the past decade; however activated carbon injection is just beginning to be used at coal-fired power plants.

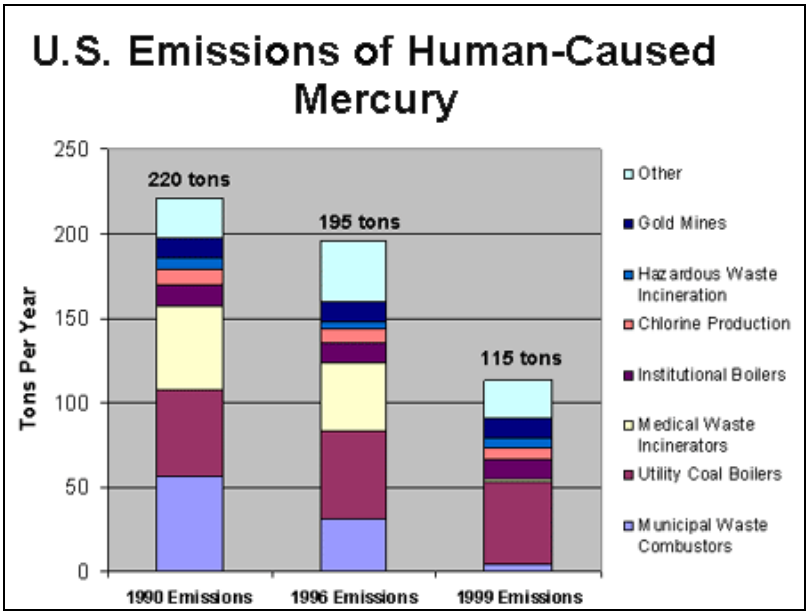
In addition to advanced control technologies, it's important to limit the amount of mercury in the incoming waste stream. OCRRA has multiple programs in place to do just that. These programs include household hazardous waste collection events, an ongoing mercury-containing thermostats and thermometer exchange at OCRRA's Rock Cut Road Transfer Station (a joint program with Covanta), partnerships with local businesses for electronic waste and household fluorescent collections, active daily sorting activities at OCRRA's transfer stations, and active daily screening at the Facility itself. Coupled with extensive public education efforts, these programs have had a significant impact.

The figure on the following page shows the effectiveness of the Facility's mercury control system, as well as the inlet and outlet (stack) average mercury concentrations. Inlet concentrations indicate the level of mercury in the incoming waste stream. As shown, inlet mercury levels since 1995 have exhibited a dramatic decrease, which has been the result of OCRRA's programs to remove mercury from the local waste stream, as well as restrictions on the mercury content of many products, most notably, alkaline batteries.



Average mercury emissions measured during 2010 annual stack testing event were 5% of the Facility's current permit limit of 28 micrograms per dry standard cubic meter and the average effectiveness of the Facility's carbon injection system for removing mercury was 99.1 % (85% removal efficiency is required).

In 1990, the contribution of atmospheric mercury from coal-fired power plants and WTE facilities were similar and substantial. During the following decade, Maximum Achievable Control Technology (MACT) emission standards were imposed on municipal waste combustors (MWCs) and the contribution to atmospheric mercury from MWCs relative to coal-fired power plants dropped dramatically. According to the USEPA Memorandum mentioned previously, mercury emissions from MWCs were reduced by 96% from 1990 to 2005. While coal-fired plants still contribute over 40% of all domestic human-caused mercury emissions in the U.S., according to the USEPA, mercury emissions from WTE plants have decreased to about 4% of the total. The following chart has been provided from USEPA's website.



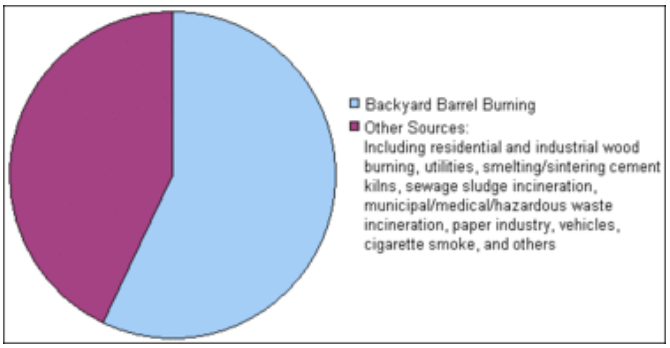
Source: USEPA website: www.epa.gov/mercury/control_emissions/emissions.htm

4.2.3 Dioxin/Furan

Like mercury emissions, dioxin and furan emissions constitute considerable environmental concern. The Onondaga County WTE Facility has several permit limits associated with dioxin/furan emissions. The 2010 results were all at least 95% below the associated permit limits. These levels are exceptionally small and indicative of effective combustion and air pollution controls.

2,3,7,8-Tetrachlorodibenzo-p-dioxin (TCDD) is the most toxic congener of dioxin. The total dioxin toxic equivalence (TEQ) value expresses the toxicity as if the mixture were pure TCDD. In 2010, the estimated annual TEQ dioxin/furan emissions are 0.00009 lbs (90 millionths of a pound); or 3% of the weight of a standard paper clip.

Over the past 20 years, the WTE industry has drastically reduced dioxin/furan emissions – by more than 99% (see table from referenced EPA memo). Today, backyard burn barrels emit more dioxins and furans than all other sources combined. The pie chart below is from NYSDEC's website and it provides data from an EPA study during 2002 to 2004.



Source: NYSDEC website - http://www.dec.ny.gov/chemical/32065.html

Fortunately, in 2009, NYSDEC passed and enacted new open burning regulations that prohibit burning household trash in burn barrels or piles statewide.

4.3 2010 Ash Testing Results

Semi-annual ash testing determines whether residual ash, the byproduct of turning non-recyclable trash into energy, should be managed as a non-hazardous or hazardous material. A representative sample of residual ash is collected according to NYSDEC and USEPA protocols. The sample is then analyzed by an independent laboratory for leachable metals, according to USEPA's Toxicity Characteristic Leaching Procedure (TCLP). TCLP analysis simulates landfill conditions (the final disposal site for the ash) and determines whether the ash exhibits hazardous characteristics. Over the life of the Facility (including 2010 results), TCLP analysis has always indicated that the ash is non-hazardous. A summary of the ash residue test results for 2010 is provided below.

2010 ASH RESIDUE CHARACTERIZATION TEST RESULTS

Semi-Annual Test Results - May 2010

Constituent	*Test Result*	*Permit Limit*	*Pass or Fail*
Cadmium	0.12 mg/L	1 mg/L	Pass
Lead	0.25 mg/L	5 mg/L	Pass

Semi-Annual Test Results - Sept./Oct. 2010

Constituent	*Test Result*	*Permit Limit*	*Pass or Fail*
Cadmium	0.60 mg/L	1 mg/L	Pass
Lead	0.25 mg/L	5 mg/L	Pass

CONCLUSION

Ash residue does NOT exhibit a hazardous characteristic. As such, it should continue to be managed as a non-hazardous solid waste.

In 2010, 77,534 tons of combined ash residue (consisting of mixed fly and bottom ash) were sent to Seneca Meadows Landfill in Waterloo, NY. Based on waste processed, this amount of ash was 24.6% of the waste tonnage combusted; therefore the Facility reduced the weight of the refuse by more than 75%. Since October 2009, ash residue from the Facility has been used as alternative daily cover at the landfill. This beneficial use of the ash ultimately means that other materials, such as clean soil, do not need to be used for daily cover at the landfill.

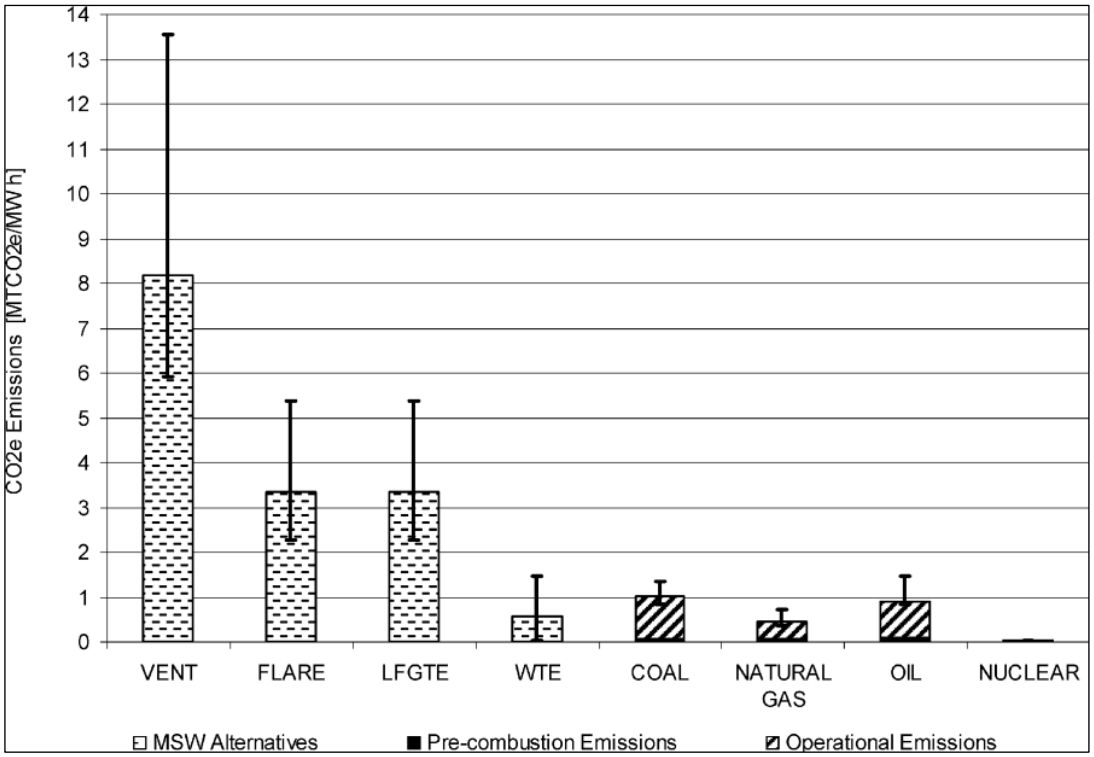
4.4 Combustion versus Landfilling

A recent USEPA-authored journal article published in Environmental Science and Technology applies a life-cycle analysis model to evaluate whether it's better to burn or bury MSW. The article is titled, "Is It Better to Burn or Bury Waste for Clean Energy Generation?" and the analysis compares greenhouse gas emissions and emissions of other pollutants for WTE and landfill gas-to-energy (LFGTE), using a life-cycle analysis model. The study states that MSW is a viable source for electricity generation and finds that WTE is a better option than LFGTE because WTE generates significantly more electricity from the same amount of waste, with fewer emissions. Though not immediately intuitive, emissions from LFGTE are due to fugitive methane emissions in a landfill, as well as emissions from combusting landfill gas in an internal combustion engine. The last paragraph of the article provides a good summary (Kaplan, Decarolis, and Thornloe, 2009):

"Despite increased recycling efforts, U.S. population growth will ensure that the portion of MSW discarded in landfills will remain significant and growing. Discarded MSW is a viable energy source for electricity generation in a carbon constrained world. One notable difference between LFGTE and WTE is that the latter is capable of producing an order of magnitude more electricity from the same mass of waste. In addition, as demonstrated in this paper, there are significant differences in emissions on a mass per unit energy basis from LFGTE and WTE. On the basis of the assumptions in this paper, WTE appears to be a better option than LFGTE. If the goal is greenhouse gas reduction, then WTE should be considered as an option under U.S. renewable energy policies. In addition, all LFTGE scenarios tested had on the average higher NOx, SOx, and PM emissions than WTE. However, HCl emissions from WTE are significantly higher than the LFGTE scenarios."

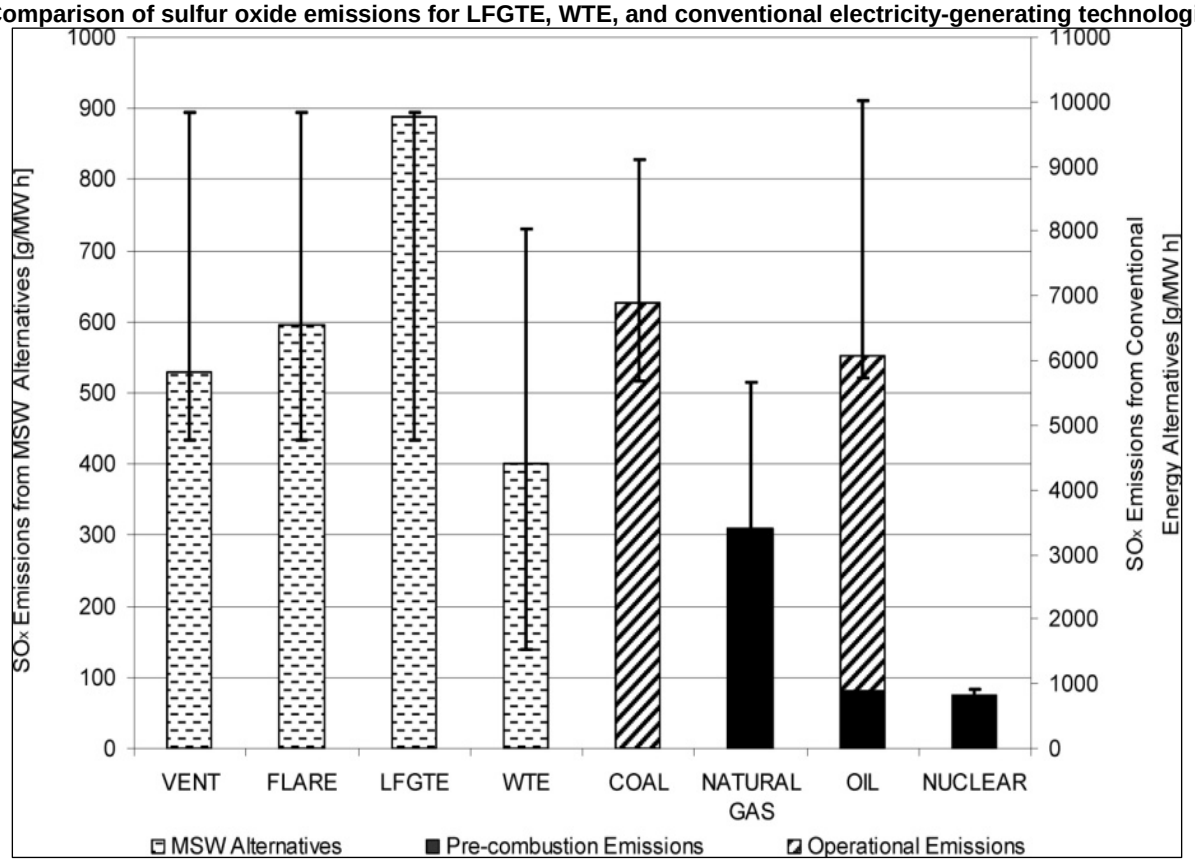
Several graphs from the article are provided below and on the next page. These graphs compare the relative emissions of greenhouse gas emissions, NO_x, and sulfur oxide (SO_x) for WTE, LFGTE, and several conventional electricity generating technologies.

Comparison of greenhouse gas emissions for LFGTE, WTE, and conventional electricity-generating technologies



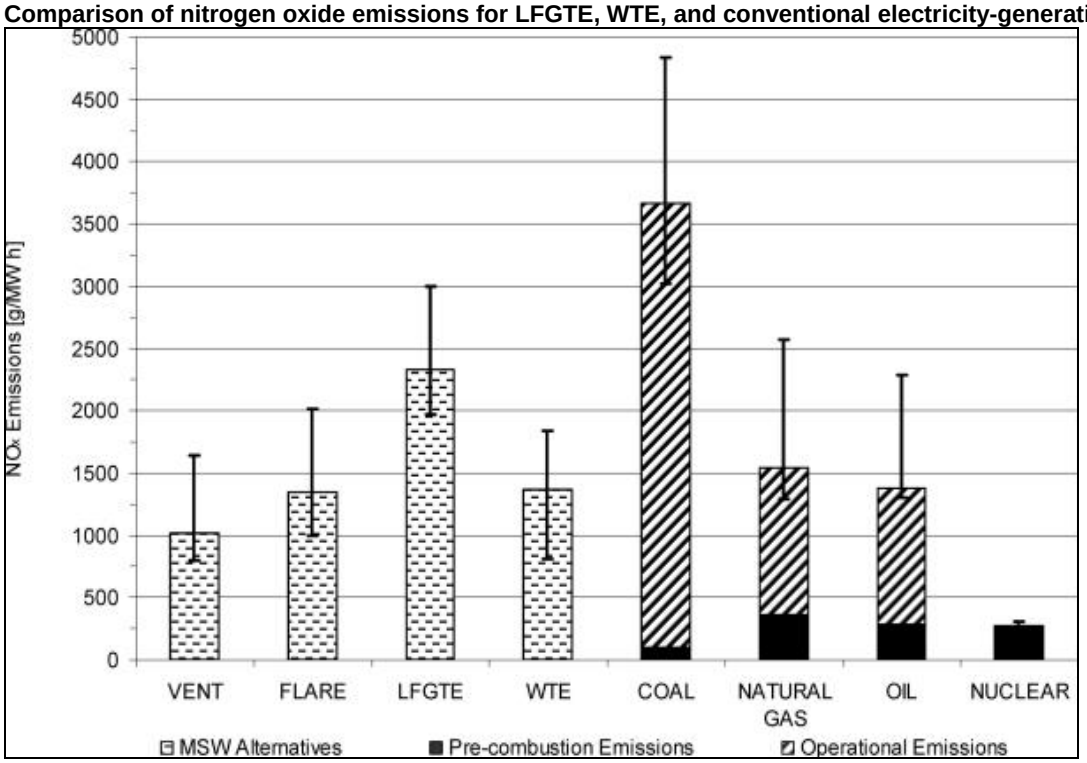
Source: Kaplan, Decarolis, and Thornloe, 2009 (Figure 2)

Comparison of sulfur oxide emissions for LFGTE, WTE, and conventional electricity-generating technologies



Source: Kaplan, Decarolis, and Thornloe, 2009 (Figure 3)

Comparison of nitrogen oxide emissions for LFGTE, WTE, and conventional electricity-generating technologies



Source: Kaplan, Decarolis, and Thornloe, 2009 (Figure 4)

4.5 Greenhouse Gas Emissions

Managing what happens to the County's non-recyclable trash is about choices. If Onondaga County did not have a WTE Facility, the County's non-recyclable trash would be destined for a landfill. Landfills generate methane (a potent greenhouse gas) as the trash degrades anaerobically. Although many landfills now have landfill gas collection systems and, ultimately, flare the landfill gas (and convert the methane to carbon dioxide), or preferably, generate electricity from the gas (landfill gas-to-energy), there are still fugitive landfill gas emissions because the landfill gas collection systems are not 100% effective. Although the Onondaga County WTE Facility generates carbon dioxide as a result of the complete combustion processes, when compared to emissions associated with landfilling, the emissions from the WTE Facility are significantly less.

In addition to having lower emissions (in terms of carbon dioxide equivalents), the WTE Facility offsets electricity that would have otherwise been generated using coal, natural gas, or nuclear fuels. According to the latest 2010 USEPA eGRID data (for 2007), New York's electricity generation resources (with associated percentages) are natural gas (30.6%), nuclear (29.4%), hydropower (16.9%), coal (14.8%), oil (5.7%), biomass (1.3%), other fossil (0.7%), and wind (0.6%). The carbon dioxide equivalent emissions associated with this profile are 755 lb/MWh. Assuming a given energy demand, the WTE Facility generates electricity that would have otherwise by generated by an alternative source.

Lastly, every year the WTE Facility recovers roughly 9,000 - 10,000 tons of metals that would have otherwise gone to a landfill. The recovered metal is then recycled, which saves considerable energy and prevents greenhouse emissions that would have resulted from virgin metal production.

When all of these factors are considered, the Onondaga County WTE Facility reduces greenhouse gas emissions (in carbon dioxide equivalents) by one ton for every ton of waste processed. Thus, in 2010, the Facility prevented 315,385 tons of carbon dioxide equivalent greenhouse gas emissions, which is the equivalent of taking nearly 50,000 cars off the road!

USEPA recently released a study entitled, "Opportunities to Reduce Greenhouse Gas Emissions through Materials and Land Management Practices" (September 2009). The study highlights several waste management practices, including waste prevention (source reduction), reuse/recycling, and WTE (energy recovery), that can lead to significant reduction in the country's greenhouse gas emissions. The study indicates there is significant GHG reduction potential associated with WTE facilities (*i.e.,* energy recovery).

4.6 Renewable Energy and Energy Independence

The Facility utilizes a locally-generated feedstock – the community's non-recyclable trash to generate a significant amount of electricity. This not only reduces dependence on fossil fuels, it also achieves goals of energy independence. In 2010 alone, the WTE Facility generated enough energy to displace nearly 400,000 barrels of oil or 80,000 tons of coal – enough energy to satisfy the needs of approximately 30,000 homes in OCRRA's service area. That is in addition to reducing the volume of non-recyclable trash by 90% and recovering ferrous and non-ferrous metal for recycling.

In many European countries and about half of the U.S. states, WTE (or energy from waste, as it is referred in Europe), is considered a renewable energy source. Most recently, Maryland Governor Martin O'Malley signed into law a bill elevating waste-to-energy to a Tier 1 renewable status in Maryland's Renewable Portfolio Standard. WTE was also highlighted as one of eight "key renewable energy sectors" by the World Economic Forum's recent (January 2009) report, "Green Investing – Towards a Clean Energy Infrastructure."

In a February 2003 letter to the Integrated Waste Services Association (IWSA) (currently the Energy Recovery Council), USEPA assessed WTE as "…clean, reliable, renewable power…"; "These plants produce 2,800 megawatts of electricity with less environmental impact than almost any other source of electricity." The Onondaga County Resource Recovery Facility is leading the way in providing an environmentally sound and cost-effective method of solid waste disposal while partially providing the energy needs of a community of 450,000 people.

4.7 Preservation of Landfill Capacity and Greenfields

In the United States, landfills are the predominant disposal alternative for MSW, with 54.3% of MSW ending up in landfills, 11.9% going to WTE facilities, and 33.8% being recycled or composted (Municipal Solid Waste Generation, Recycling and Disposal in the United States: Facts and Figures for 2009, USEPA). Over the past couple of decades, the number of landfills has decreased dramatically, however the size of the remaining landfills is substantially larger. Due to economies of scale, these "mega-landfills" are becoming the norm. However, as you can imagine, "mega-landfills" take up massive amounts of open space.

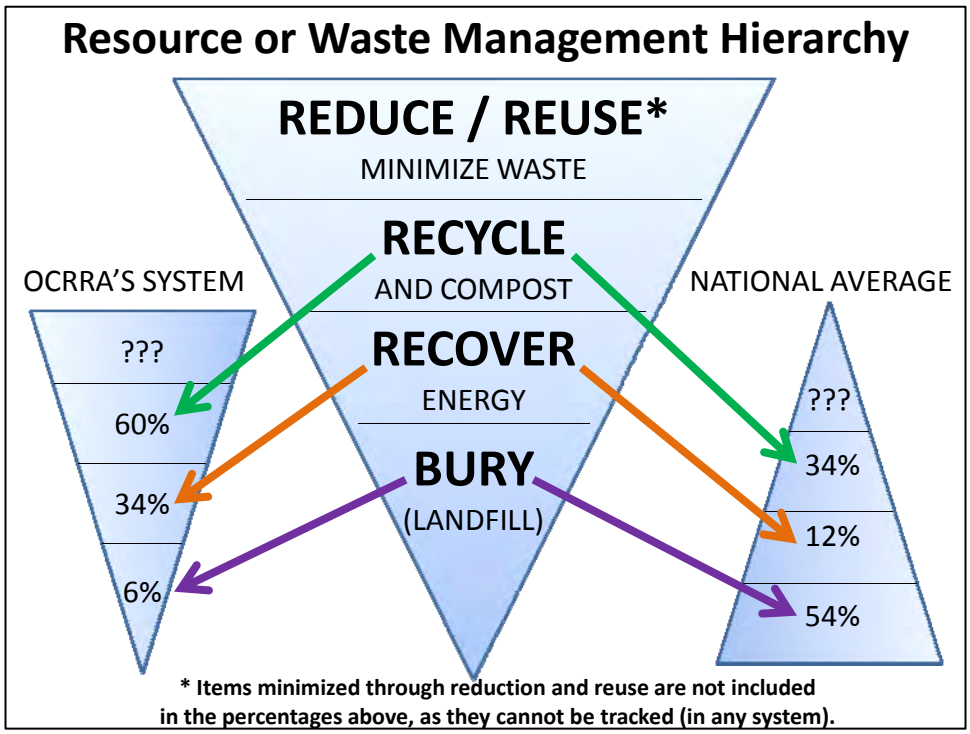
WTE facilities preserve existing landfill capacity by reducing the volume of MSW by 90%. This means that the current landfill capacity will last longer, and that "greenfields" will not be utilized for landfill expansion projects. Had the 315,385 tons of waste processed at the Facility in 2010 been landfilled, it would have utilized close to a million cubic yards of landfill space. To put this into perspective, if the waste was compacted to a 20-foot height, the landfilled waste would consume about 31 acres of land.

4.8 Compatibility with Recycling

In Onondaga County, which has one of the highest recycling rates in the State and perhaps in the nation, it seems trivial to question the compatibility of WTE and recycling. However, WTE facilities are often thought to compete with recycling. Interestingly, study after study, it has been shown that communities with WTE facilities often have higher recycling rates than communities that landfill their non-recyclable trash, both in Europe and the United States. A recent study (June 2009) entitled "A Compatibility Study: Recycling and Waste-to-Energy Work in Concert, A 2009 Update" again indicates the same conclusion.

4.9 Consistency with Waste Management Hierarchy

The waste management hierarchy set forth in New York State's 2010 "Beyond Waste" Solid Waste Management Plan, as well as in USEPA guidelines, includes (in order of preference): 1) waste reduction, 2) recycling, 3) recovery of useful energy through solid waste combustion (i.e., modern waste-to-energy facilities), and 4) use of permitted landfill facilities. This hierarchy, supported by our state and the nation, considers the environmental impacts of each level and prioritizes them accordingly, with the most preferred option being waste reduction/reuse and the least preferred option being landfilling. It also provides a good measuring stick for evaluating OCRRA's system. As indicated in the figure below, OCRRA's system is extremely consistent with the hierarchy. On the other hand, the national average doesn't do nearly as good a job with its low recycling rate and heavy reliance on landfilling. In fact, the national numbers are upside down.



National data from USEPA Report, "Municipal Solid Waste Generation, Recycling, and Disposal in the United States: Facts and Figures for 2009."

Section 5 – Financial Performance

5.1 Waste-to-Energy Facility Financial Summary

A simplified financial summary of OCRRA's revenues and expenses associated with the WTE Facility for 2010 is provided below. Please note that the presentation of information in this report is different from the presentation in OCRRA's financial statements. The information in this report should not be used for financial accounting purposes and is only intended to provide a simplified perspective on OCRRA's costs and expenses associated with the WTE Facility. It should be emphasized that the revenues and expenses described in this report pertain specifically to OCRRA; Covanta Onondaga also has Facility-related operating revenues and expenses that are not described in this report.

Waste-to-Energy Financial Summary for 2010

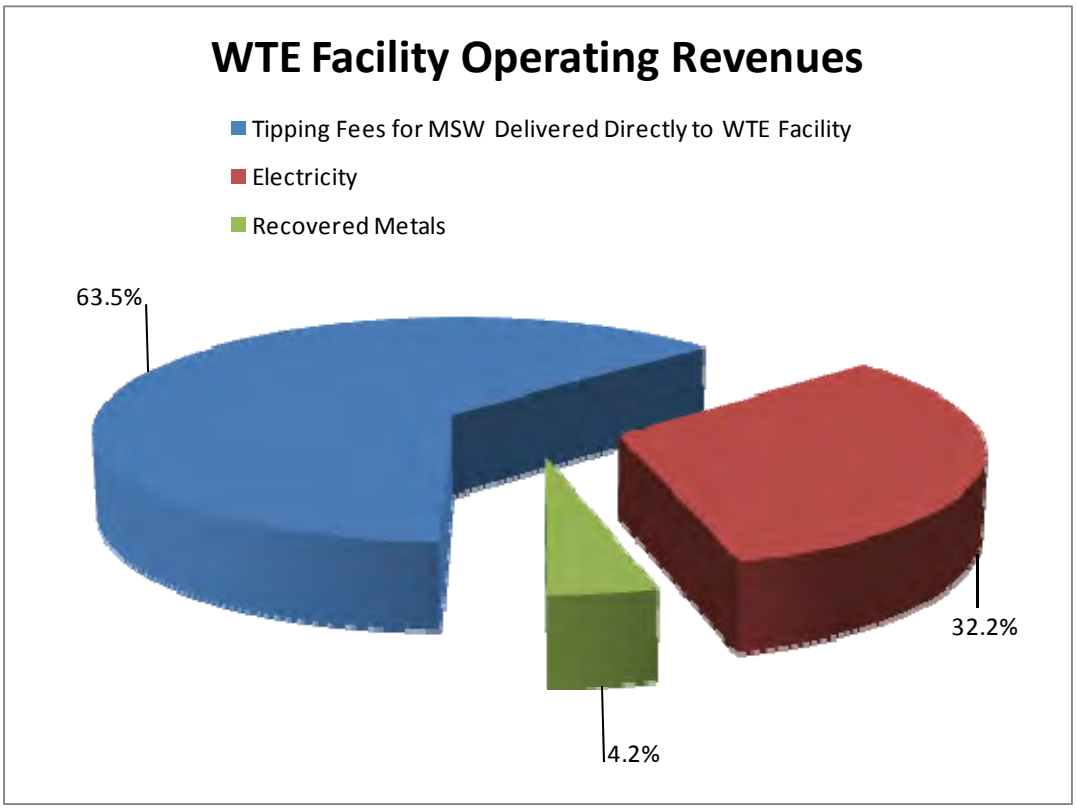
Operating Revenues	
Tip Fee for MSW Delivered Directly to Facility……	$15,427,490
OCRRA's Electricity Share……	$7,832,026
OCRRA's Recovered Metals Share……	$1,031,263
Supplemental Waste Tip Fee……	$4,100
***Total*……**	**$24,294,879**
Operating Expenses	
Operations and Maintenance Service Fee	$11,137,824
Ash Transportation and Disposal	$3,485,172
Excess Waste Fee	$110,594
Pollution Control Reagents	$719,728
Taxes/Fees	$331,674
Utilities	$414,833
Other Expenses (Mainly insurance)	$340,371
***Total*……**	**$16,540,197**
***Bond Expenses*……**	**$9,320,217**
***Total Expenses*……**	**$25,860,414**

As evident, OCRRA's 2010 WTE-related expenses exceeded the WTE-related revenues (net loss of $1.6 million). These Facility-related revenues and expenses constitute a significant portion of OCRRA's total Agency revenues and expenses. To provide some perspective, in the 2010 budget, WTE Facility-related operating and bond expenses accounted for about two thirds of OCRRA's total Agency expenses. Similarly, WTE Facility-related operating revenues accounted for roughly two thirds of OCRRA's total Agency revenues.

In 2010, total cost per ton of MSW processed was approximately $82 and total revenue per ton of MSW processed was approximately $77. As evident, WTE facilities like the local Facility have tremendous fixed costs. If those fixed costs are not offset by sufficient electricity revenue and tipping fees, there may be facility-related net losses, as in 2010. In 2010, the average electricity rate (including the capacity factor) was 4.5¢ per kWh; better than 2009's average rate of 3.7¢ per kWh, but still much less than the rate of about 6¢ per kWh for the previous few years. The low electricity rate, compounded by low trash tonnage, resulted in a net loss.

5.2 Waste-to-Energy Facility Operating Revenues

OCRRA's operating revenues associated with the WTE Facility include tipping fees for waste delivered *directly* to the Facility (not including tipping fees for waste delivered to OCRRA's transfer stations), sale of electricity generated by the Facility, the sale of metals recovered by the Facility, and revenue derived from the supplemental waste program, which was negligible for 2010. A summary of the relative contribution of these revenues is provided in the pie chart below. It should be emphasized that the revenues described in this report are revenues that pertain to OCRRA. Covanta Onondaga also receives Facility-related operating revenues that are not described in this report.



Although MSW and C&D from OCRRA's transfer stations are delivered to the WTE Facility, tipping fees are collected at the transfer stations and are therefore not included in this financial summary. Similarly, the cost of processing MSW and C&D at the transfer stations is not included in this report. However, it should be noted that electricity generated from the transfer station MSW and C&D is included in the electricity revenue.

5.2.1 Tip Fee for MSW Delivered Directly to Facility

In 2010, tipping fees for MSW delivered directly to the Facility accounted for just under two thirds of the revenues associated with the WTE Facility. In previous years, when electricity rates had been higher, tipping fees generally accounted for about half of the Facility-related revenues.

OCRRA receives the full tipping fee for MSW delivered directly to the Facility. In 2010, tipping fees were $69 per ton, with a $4 prompt payment discount. Most haulers take advantage of the prompt payment discount; therefore OCRRA generally received revenues of $65 per ton. OCRRA's office staff is responsible for billing and collecting payments from haulers.

5.2.2 OCRRA's Electricity Share

Electricity sales represent the other major revenue component associated with the WTE Facility. Historically, electricity had accounted for about 40-45% of Facility-related revenues. However, due to the low electricity rates in 2010, electricity sales accounted for 32% of Facility-related revenues. OCRRA receives 90% of the electricity revenues, with Covanta Onondaga receiving the remaining 10%.

For 2010, the total amount of electricity sold was 190,639 MWh, with an average electricity rate (including the capacity factor) of 4.5¢ per kWh. While this was better than the average rate of 3.7¢ per kWh in 2009, prior to 2009, a contract between OCRRA/National Grid (formerly Niagara Mohawk), provided minimum floor pricing of 6¢ per kWh. Ironically, the historical annual average electricity rate had generally exceeded the floor pricing. Unfortunately, in 2009, when electricity prices took a sharp decline, the minimum floor pricing had expired. In 2010, total energy revenues were $8,575,583, with OCRRA's share generating $7,832,026 in revenue. For comparison, 2008 energy revenues were $15,006,122, with OCRRA's 90% share generating $13,505,512 in revenue.

5.2.3 OCRRA's Recovered Metal Share

In 2010, recovered metal revenue accounted for 4% of Facility-related revenues. Unlike electricity rates, metal prices rebounded in 2010. OCRRA and Covanta Onondaga evenly split metal recovery revenues, each receiving 50%. A breakdown of 2010 tonnage and revenues for the non-ferrous and ferrous recovery systems is provided below.

	Tonnage	*OCRRA's Revenue*
Ferrous Metal	8,599	$816,087
Non-Ferrous Metal	333	$215,176

In 2010, average ferrous and non-ferrous prices were about $190 and $1,290 per ton, respectively. In comparison, average ferrous and non-ferrous pricing for 2008 were about $250 and $1000 per ton, respectively. In 2009, average ferrous and non-ferrous pricing were about $90 and $550 per ton, respectively.

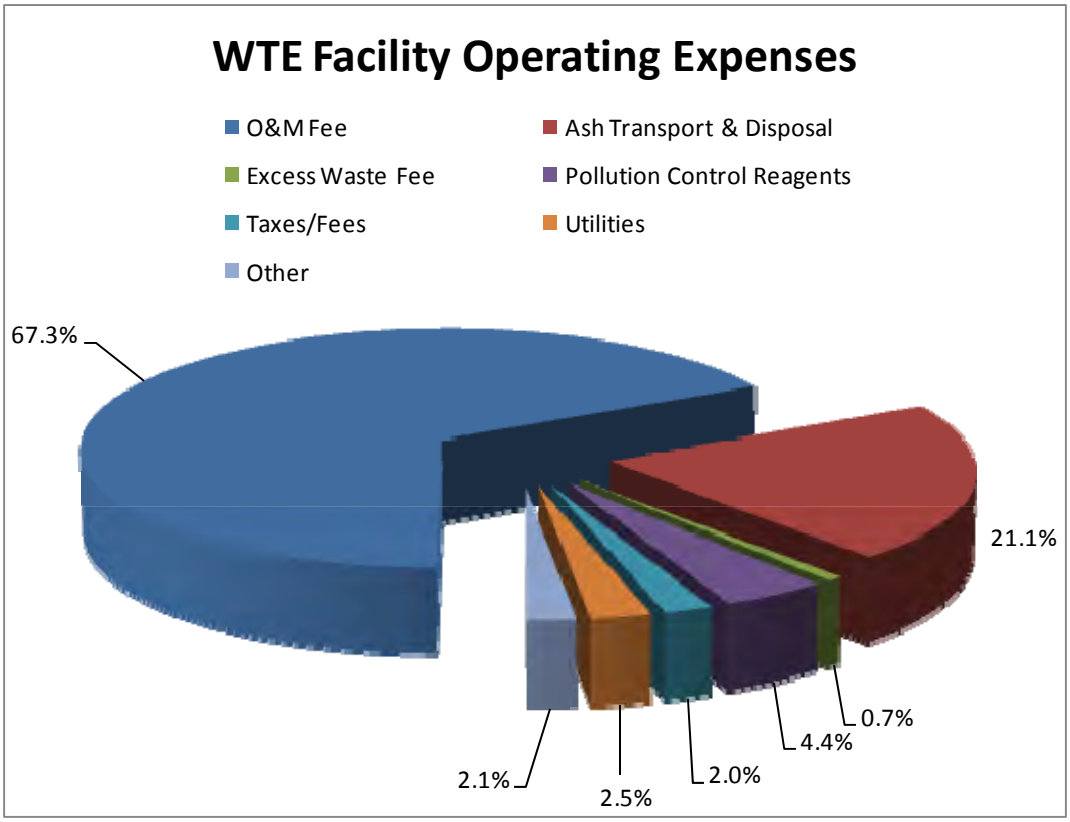
5.2.4 Supplemental Waste Tip Fee

The supplemental waste program is in place to provide proper disposal for waste streams other than MSW that may need special handling, secure destruction, or other special provisions. These wastes are carefully screened and evaluated to ensure that they will not impact Facility operations, including air emissions and ash residue characteristics. Covanta Onondaga administers the supplemental waste program with oversight from NYSDEC and OCRRA. As such, Covanta receives the established tipping fee less $10 (which OCRRA receives) for the first 500 tons of waste and thereafter Covanta receives the established tipping fee less OCRRA's tipping fee, which OCRRA receives.

In 2010, 544 tons of supplemental waste were processed, generating $4,100 in revenue for OCRRA. The types of waste processed in 2010 include pill bottles with labels (which under the HIPAA regulations require secure destruction); confiscated drugs, uniforms, and other paraphernalia from drug enforcement agencies; and pharmaceutical laboratory debris.

5.3 Waste-to-Energy Facility Operating Expenses

The operating expenses associated with the WTE Facility include an operations and maintenance (O&M) service fee paid to Covanta to maintain the Facility, the costs to transport and dispose of ash generated by the Facility, an excess waste fee payment to Covanta if more than 310,000 tons of MSW is processed at the Facility, costs associated with pollution control reagents, taxes/fees, utilities, and other miscellaneous expenses (described further below). A summary of the relative contribution of these expenses is provided in the pie chart below. It should be emphasized that the operating expenses described in this report are expenses that pertain to OCRRA. Covanta Onondaga also has Facility-related operating expenses that are not described in this report.



5.3.1 Operations and Maintenance Service Fee

OCRRA pays an operations and maintenance (O&M) service fee for Covanta Onondaga to operate, repair, and maintain the Facility in accordance with the 2003 Service Agreement between OCRRA and Covanta Onondaga. This is, by far, the largest Facility-related expense. Each calendar year the O&M fees are adjusted according to several indices (skilled labor index, producer price index, and employment cost index) and OCRRA's annual tipping fee. In 2010, the base O&M service fee was $11,097,888 and the non-ferrous O&M fee was $39,936, for a total of $11,137,824.

5.3.2 Ash Transportation and Disposal

OCRRA is responsible for transporting and disposing of ash residue generated at the Facility. The associated costs were estimated from a unit cost report and include all costs associated with handling and disposal of ash residue (salaries, fuel, tolls, tip fees, social security, insurance, and maintenance). The average unit cost for 2010 was approximately $44.95 per ton, with 77,534 tons of ash being managed. Therefore, the total ash transportation and disposal costs for 2010 were approximately $3,485,172.

5.3.3 Excess Waste Fee

According to the 2003 Service Agreement between OCRRA and Covanta, OCRRA is required to pay Covanta an excess waste fee if the Facility processes more than 310,000 tons of material in the calendar year. The unit fee per ton of waste greater than 310,000 is adjusted annually, based on the same indices as the O&M Service Fee adjustment. For 2010, the unit fee was $22.85. The excess waste fee is not applicable for supplemental waste; therefore the quantity of supplemental waste is subtracted from the amount of waste processed in excess of 310,000 tons. In 2010 the Facility processed 4,840 tons of excess waste, resulting in an excess waste fee payment to Covanta of $110,594. Historically, the excess waste fee has been between $500,000 and $800,000. The 2010 fee was significantly less because of the low waste tonnage in 2010.

5.3.4 Pollution Control Reagents

The Facility uses several reagents for pollution control including anhydrous ammonia for control of NO_x, carbon for mercury and dioxin/furan control, and lime for control of acid gases. The cost of these reagents is generally a pass-through cost to OCRRA, with the exception of lime for which OCRRA only pays a portion of the cost.

To control NO_x emissions, anhydrous ammonia is injected into the combustion chamber of each boiler unit. There are no contractual maximum levels for ammonia usage, so OCRRA is solely responsible for the expense of all ammonia used. In 2010, the cost for ammonia reagent was $188,527 for 259 tons of anhydrous ammonia at an average cost of about $725/ton. Given the 2010 waste tonnage processed, these figures translate into an application rate of 1.64 lb per ton of waste processed and a unit cost of $0.60 per ton of waste processed.

To control mercury emissions, as well as dioxin and furan emissions, powdered activated carbon is mixed into slurry and injected into the spray-dry scrubbers through the rotary atomizer. The rotary atomizer creates tiny droplets for optimal reaction. There are no contractual maximum levels for carbon usage, so OCRRA is solely responsible for the expense of all carbon used. In 2010, the cost for activated carbon was $339,033 for 241 tons of activated carbon at an average cost of $1,406 per ton. The average carbon reagent application rate for 2010 was 1.53 lb per ton of waste processed, a rate within the historical range, and the unit cost was $1.07 per ton of waste processed. In 2009 and 2010, the average unit cost for activated carbon was nearly 23% over the 2008 unit cost.

To neutralize acid gases, namely sulfur dioxide (SO_2), hydrogen chloride (HCl), hydrogen fluoride (HF), and sulfuric acid (H_2SO_4), a calcium-based lime, commonly referred to as pebble lime, is injected into the spray-dry scrubbers through the rotary atomizer. According to an agreement between OCRRA and Covanta, OCRRA is responsible for the cost associated with the pebble lime usage in excess of 21 pounds of pebble lime per ton of waste processed, up to a maximum of 32 lb per ton of waste processed. Covanta is responsible for pebble lime reagent costs up to 21 lb per ton of waste processed and above 32 lb per ton of waste processed. In 2010, OCRRA's cost for lime was $192,168 and the average reagent application rate was 29.4 lb of lime per ton of waste processed. The cost of the lime reagent for 2010 was about $137 per ton.

5.3.5 Taxes/Fees

OCRRA is contractually responsible for the cost of the following taxes/fees:

- State and local sales taxes on Facility-related purchases – $34,496 in 2010
- Regulatory operating permit annual fees – $26,733 in 2010
- Host Community Agreement payments to the Town of Onondaga – $144,747 in 2010
- Special fire district tax assessments – $119,433 in 2010
- Special water district tax assessments – $6,265 in 2010

5.3.6 Utilities

During normal Facility operation, the Facility's electrical demand is satisfied by the Facility's turbine-generator system, with the excess electricity being exported to the grid. During those times when the turbine-generator is offline due to maintenance or malfunction, electricity is purchased from National Grid (NG) to operate the Facility and continue combusting the incoming MSW. OCRRA is financially responsible for paying for the electricity purchased during these periods. The contractual threshold levels beyond which Covanta is responsible for such costs are as follows:

Electrical Energy	3,348,000 kWh/rolling 3-year period (maximum)
Electrical Demand	4,400 kW (maximum per billing period)

In 2010, 1,239,156 kWh of electricity was purchased from National Grid for in-plant needs during turbine-generator outages. The amount of electricity purchased during 2010 is significantly more than the average annual electricity purchased and is mainly due to the occurrence of the second major turbine-generator overall since Facility start-up. The 3-year rolling period total for 2008-2010 was 2,035,350 kWh, significantly less than the contractual maximum amount stated above. For 2010, the maximum monthly metered electrical demand was 3,888 kW. The cost of purchased power paid by OCRRA for 2010, including electrical usage and demand charges, was $231,010.

City water satisfies all potable and process needs of the Facility, with the majority being for process use. 30,360,000 gallons, representing 76% of the contractual maximum (40 million gallons per year) for which the Agency is financially responsible, were purchased in 2010. This amount of water translates into about 96 gallons per ton of waste combusted or approximately 58 gallons per minute. 2010 water usage remained consistent with historical levels and design parameters following initial start-up. Total 2010 water costs were $63,805, or $2.10 per thousand gallons, a 10% increase from 2009.

Natural gas is an auxiliary fuel used for boiler start-ups and shutdowns, and for maintaining minimum furnace temperatures when processing overly wet waste. 2010 natural gas usage was 137,520 therms, which is consistent with historical usage. The contractual maximum amount of natural gas OCRRA is financially responsible for is 110,000 therms per year, with Covanta being responsible for usage over 110,000 therms. Covanta exceeded the usage threshold in October 2010, at which point OCRRA was no longer responsible for natural gas costs. OCRRA's total annual natural gas costs were $120,019.

5.3.7 Other Expenses

In 2010, OCRRA was financially responsible for several other Facility-related expenses totaling $340,371, which consisted of:

- Facility-related insurance premiums ($290,024);
- System telecommunications between Facility and National Grid ($6,355);
- Traffic signalization for the hauler entrance to the Facility ($1,585);
- OCRRA's WTE engineering consulting services related to providing technical assistance and annual stack and ash testing on-site observations ($34,742);
- Trustee fees ($8,620); and
- Other miscellaneous (-$955).

5.4 Bond Expenses

Until May 2015, OCRRA is responsible for paying debt service on the bonds for the Facility. At that point, the Series A bonds will have been paid off and the responsibility of the Series B bonds will be transferred to Covanta Onondaga. OCRRA pays a set amount for the principal and interest on the Series A bonds; however the amount paid on the Series B bonds depends on the profitability of OCRRA in any given year. OCRRA did not have a profitable year in 2010 (similar to 2009); therefore, OCRRA did not make payments on the principal of the Series B bonds. The total payment on the Series A bonds in 2010 was $9,320,217.

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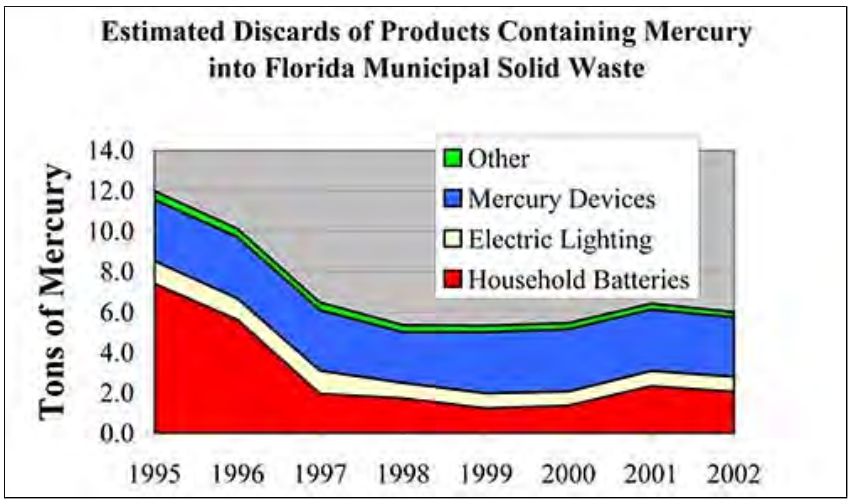
Quantifying Mercury in Florida's Solid Waste

Most discarded MCLs (about 80%) and MCDs come from commercial and governmental facilities, while the rest are generated from households and other residential buildings. Many commercial and governmental facilities do periodic relamping to improve the energy efficiency of their buildings and to save on energy costs. These efforts are to be encouraged as reductions in energy use may also have benefits in respect to lower emissions (including mercury) from certain fossil fuel power plants.

Mercury Containing Lamps (MCLs): Using US EPA and manufacturer data, the annual discards of fluorescent lamps, which make up the vast majority of MCLs, and MCDs for Florida have been estimated since 1995. About 39 million fluorescent lamps were estimated to be discarded in Florida in 2002. Four-foot fluorescent lamps, the most common type of fluoresent lamp sold, were estimated to have an average mercury composition of about 10 milligrams (mg) per lamp based on the lamp manufacturing industry's data. Prior to 1994, new fluorescent lamps sold in the US were reported by this industry to contain an average of about 41.6 mg. Beginning in 1994, this average dropped to about 22.8 mg of mercury per this lamp type. Fluorescent lamp manufacturers have continued to reduce the amount of mercury their lamps contain through source reduction, and the average mercury content in four-foot fluorescent lamps dropped to about 8 mg in lamps made in 2001. This would be about an 80% reduction in the mercury content of these lamps over the last ten years. Based on an average lamp life of four years, fluorescent lamps that are discarded beginning in 2005 will be assumed to have an average mercury concentration of about 8 mg.



Mercury Containing Devices (MCDs): Florida's current estimates of discarded MCDs and the mercury they contain comes from a variety of sources including US EPA reports and manufacturer-reported information. The types of MCDs included in Florida's estimates are mercury-containing thermostats, thermometers, and electric switches (e.g., light switches, displacement relays, and boat bilge pump float switches). Since 1997, MCDs have been the largest source of mercury in Florida's MSW stream. The chart above and table below show the mercury in MCLs and MCDs that are estimated to have been discarded

in Florida's MSW since 1995 in comparison with other sources. For more information on the sources of mercury in Florida's MSW, see Chapter 5 "Special Wastes" of the "Solid Waste Management in Florida 2001-2002 Annual Report."

Table I
Estimated Mercury in Florida Municipal Solid Waste, 1995 - 2002
Estimate, in Tons

Product Category	1995	1996	1997	1998	1999	2000	2001	2002	2002%
Household Batteries	7.4	5.6	2.0	1.7	1.3	1.4	2.3	2.1	34%
Electric Lighting	1.1	1.1	1.1	.07	.07	0.7	0.7	0.3	12%
Mercury Devices	3.0	3.0	3.0	2.5	3.0	3.1	3.1	3.0	49%
Other	0.4	0.4	0.4	0.4	0.3	0.3	0.3	0.3	4%
TOTALS	12.0	10.1	6.5	5.4	5.3	5.5	6.4	6.0	100%

Fluorescent Lamps
» Rules & Related Laws
» See Mercury Vapors on Video
» Sources and Quantities of Mercury in Florida's Solid Waste
» Spill Cleanup
» State Recycling Contract for Government Agencies

Last updated: January 25, 2012

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Review: Mercury in waste incineration

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Abstract

The paper investigates the sources of mercury (Hg) in municipal/industrial waste and the consequences of the presence of this pollutant for the incineration of this waste.

About 1990 the average mercury concentration of the feed stream to incinerators was about 4 mg kg^{-1}. The concentration decreased considerably during the last decade thanks to a considerable reduction of the application of mercury and to the introduction of effective battery return systems. Presently the mercury concentration in municipal SOLID waste is approximately 2 mg kg^{-1}.

During incineration mercury passes practically for 100% in the flue gas. The techniques for mercury removal from flue gases are discussed at the hand of practical examples. It is concluded that there are a number of processes which guarantee mercury concentrations of <50 µg Nm^{-3} in the clean gas, the present emission limit concentration.

All mercury control processes produce a new solid or liquid waste stream that contains the mercury removed from the flue gas. This stream has to be disposed of as hazardous waste in a qualified landfill.

The flue gas from waste incinerators undergoes very rapid dispersion and dilution after leaving the incinerator stack. It follows that the maximum mercury concentration in the ambient air will remain at least five to six orders of magnitude below the lowest MAC value (=Maximum Admissible Concentration in work spaces) and that public health will not be threatened.

Mercury | mercury in waste | waste incineration | flue gas treatment | abatement processes | mercury di

United States
Environmental Protection
Agency

Solid Waste and
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April 1992

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Characterization of Products Containing Mercury in Municipal Solid Waste in the United States, 1970 to 2000

Executive Summary

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CHARACTERIZATION OF PRODUCTS CONTAINING MERCURY IN MUNICIPAL SOLID WASTE IN THE UNITED STATES, 1970 TO 2000;

EXECUTIVE SUMMARY

April, 1992

U.S. Environmental Protection Agency
Office of Solid Waste
Municipal and Industrial Solid Waste Division

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CHARACTERIZATION OF PRODUCTS CONTAINING MERCURY IN MUNICIPAL SOLID WASTE IN THE UNITED STATES, 1970 TO 2000

Executive Summary

THE PURPOSE AND SCOPE OF THIS REPORT

The purpose of this report is to identify the products in municipal solid waste (MSW) that may contain mercury and to quantify, to the extent that data are available, the mercury present in these products. Since the data are presented in a time trend (1970 to 1989), the report helps to identify which products in MSW are making declining contributions of mercury and which are increasing. The information in this report can thus be used to identify opportunities for source reduction and removal of mercury from the municipal solid waste stream.

As in earlier municipal solid waste characterization reports published by EPA*, the characterization of mercury in MSW relies on a material flows methodology. By definition in the referenced reports, municipal solid waste is generated from residential, commercial, and institutional sources. Some wastes from industrial facilities, such as office waste and packaging, are also included. MSW as characterized in the referenced reports does *not* include other Subtitle D wastes such as municipal sludges, municipal waste combustion ash, industrial nonhazardous process wastes, small quantity generator wastes, construction and demolition wastes, agricultural wastes, oil and gas production wastes, and mining wastes. Subtitle C (hazardous) wastes also are not included.

HEALTH AND ENVIRONMENTAL EFFECTS OF MERCURY

Human Health Effects

Mercury is a heavy metal with a high toxicity and strong tendency to bioaccumulate in the food chain. Worldwide, the major route for entry of mercury in humans is ingestion of mercury-contaminated food, especially fish. Long-term exposure, or exposure during developmental stages, to either organic or inorganic mercury can permanently damage the brain, kidneys, and fetuses. Short-term exposure to high levels of inorganic or organic mercury can cause similar health effects, which may be reversible. Pregnant women, fetuses, and children appear to be at highest risk.

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* The most recent of these reports is **Characterization of Municipal Solid Waste in the United States: 1990 Update.** EPA/530-SW-90-042. June 1990.

Environmental Effects

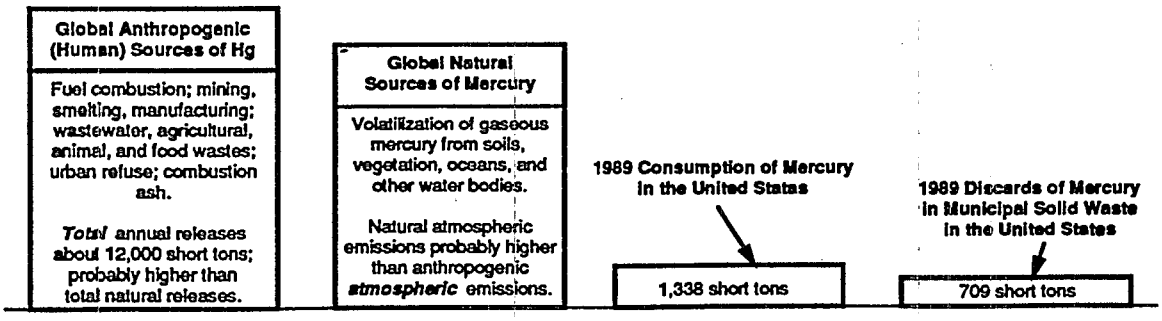
In addition to potential effects on human health, mercury poisoning can also affect other living organisms. Mercury is unique among the metals in that it is consistently biomagnified within the aquatic food chain. Organisms eating mercury-contaminated fish, such as birds, wild mink, and otter, have been found to have mercury poisoning. In addition, several countries have reported poisoning of birds through ingestion of seeds treated with mercury compounds, and of predatory animals through ingestion of contaminated birds.

MERCURY RELEASES IN PERSPECTIVE

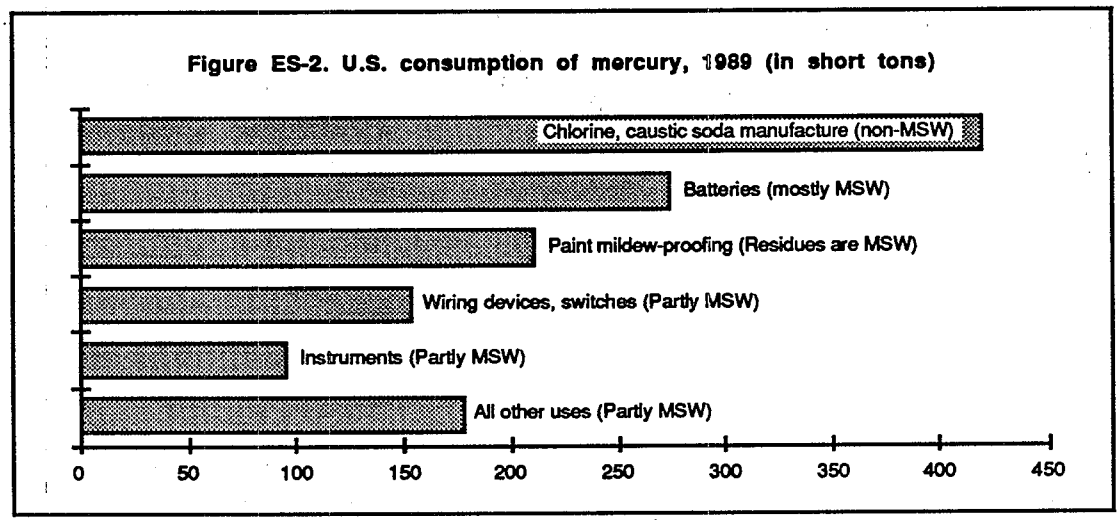
While the products containing mercury in municipal solid waste are an important source of mercury releases in the environment, they are far from the only source. Global releases of mercury in the environment are both natural and anthropogenic (caused by human activity). Relative sources of mercury are shown in Figure ES-1.

While global releases are not well documented, the best estimate available is that about 12,000 short tons of mercury are released annually to the air, soil, and water through anthropogenic sources. These sources include combustion of various fuels; mining, smelting, and manufacturing activities; wastewater; agricultural, animal, and food wastes; urban refuse; combustion ash, and other human activities. Global natural sources of mercury include volatilization of gaseous mercury from soils, vegetation, oceans, and other water bodies. The natural sources are thought to release less mercury overall than the anthropogenic sources, but natural atmospheric emissions may be higher than anthropogenic atmospheric emissions.

Figure ES-1. Global and United States sources of annual mercury releases.



Note: Heights of columns are for comparative purposes only. Global anthropogenic and natural sources of mercury are not well documented. Due to time lag before products in MSW are discarded, mercury in discards may have been consumed several years previously.



Discards of mercury in products in municipal solid waste in the United States are derived from consumption of mercury, with appropriate adjustments for manufacturing losses, imports and exports of products containing mercury, and the lifetimes of the relevant products. In 1989, an estimated 709 short tons of mercury were discarded in the U.S. in municipal solid waste compared to the 1,338 short tons reported to be consumed in the U.S. the same year (Figure ES-1). The MSW discards are less than consumption because mercury is used in several products and processes that are not discarded as MSW. (The adjustments listed above also partially account for differences between consumption and discards in any given year.)

There are several uses of mercury in products and processes that are not classified as MSW discards. The largest use of mercury in the U.S. is in chlorine and caustic soda manufacture (Figure ES-2); mercury wastes from these processes are classified as industrial process waste. Another example is mercury-containing paint that has been applied to indoor or outdoor surfaces. If the structure is demolished, the waste would be classified as demolition waste, not MSW. Many batteries, instruments, and electrical devices containing mercury are used in industrial, communications, transportation, or military applications that also are not classified as MSW.

SOURCES OF MERCURY IN MUNICIPAL SOLID WASTE

Research performed in the preparation of this report identified a number of sources of mercury in municipal solid waste, with total discards of mercury in 1989 estimated to be 709 short tons. A summary of the results is shown in Tables ES-1 and ES-2, and Figure ES-3.

Table ES-1

DISCARDS* OF MERCURY IN PRODUCTS IN THE MUNICIPAL SOLID WASTE STREAM, 1970 TO 2000
(In short tons)**

Products	1970	1980	1989	2000
Household Batteries	310.8	429.5	621.2	98.5
Electric Lighting	19.1	24.3	26.7	40.9
Paint Residues	30.2	26.7	18.2	0.5
Fever Thermometers	12.2	25.7	16.3	16.8
Thermostats	5.3	7.0	11.2	10.3
Pigments	32.3	23.0	10.0	1.5
Dental Uses	9.3	7.1	4.0	2.3
Special Paper Coating	0.1	1.2	1.0	0.0
Mercury Light Switches	0.4	0.4	0.4	1.9
Film Pack Batteries	2.1	2.6	0.0	0.0
TOTAL DISCARDS	421.8	547.5	709.0	172.7

* Discards before recovery.
** Weights in this report are converted to short tons of 2000 pounds.
Source: Franklin Associates, Ltd.

The tables show that batteries discarded from households and other sources of MSW are by far the largest current source of mercury. Light bulbs, paint residues, thermometers, thermostats, and pigments are estimated to contribute most of the remainder of mercury in MSW. A few other uses, such as dental mercury and light switches, were also identified, but these totaled less than one percent of mercury in MSW in 1989.

Mercury discards in MSW peaked in 1986, and are declining rapidly (Figure ES-4). In particular, there is projected to be a significant decrease in mercury in alkaline batteries and paint residues over the next few years. The decrease in batteries is due to a long-term commitment to research and development by the battery industry to remove mercury from alkaline batteries. The removal of mercury from paint residues is the result of bans by the Environmental Protection Agency, and voluntary cancellations of registrations by the paint industry, of mercury-based biocides in 1990 and 1991.

The only products identified to be increasing in total tonnage of mercury discarded are electric lighting and mercury light switches. Fever thermometers and thermostats, while formerly increasing in tonnage discarded, are projected to be fairly stable as sources of mercury.

Table ES-2

DISCARDS* OF MERCURY IN PRODUCTS IN THE MUNICIPAL SOLID WASTE STREAM, 1970 TO 2000
(In percent of total discards)

Products	1970	1980	1989	2000
Household Batteries	73.7	78.4	87.6	57.0
Electric Lighting	4.5	4.4	3.8	23.7
Paint Residues	7.2	4.9	2.6	0.3
Fever Thermometers	2.9	4.7	2.3	9.7
Thermostats	1.3	1.3	1.6	6.0
Pigments	7.7	4.2	1.4	0.9
Dental Uses	2.2	1.3	0.6	1.3
Special Paper Coating	0.0	0.2	0.1	0.0
Mercury Light Switches	0.1	0.1	0.1	1.1
Film Pack Batteries	0.5	0.5	0.0	0.0
TOTAL DISCARDS	100.0	100.0	100.0	100.0

* Discards before recovery.

Source: Franklin Associates, Ltd.

Each identified source of mercury in MSW is discussed briefly in this section.

Household Batteries

Batteries containing mercury that are assumed to be discarded into MSW are mostly of two types:

- Alkaline batteries, which are usually the cylinder-shaped batteries used in flashlights, radios and other electronics, and toys.

- Mercury-zinc batteries, which are usually in a "button" form, are used in hearing aids, watches, calculators, cameras, and similar applications. Mercury-zinc cylinder-type batteries are also used in some medical applications that were assumed to be discarded in MSW.

A few other kinds of batteries—carbon zinc, silver oxide, and zinc air—account for relatively small amounts of mercury in MSW.

Alkaline Batteries. Alkaline batteries accounted for about 419 short tons, or over 59 percent, of discards of mercury in MSW in 1989. While the

amount of mercury used in each battery has been quite small, the large numbers of alkaline batteries sold have caused these batteries to become the leading source of mercury in MSW.

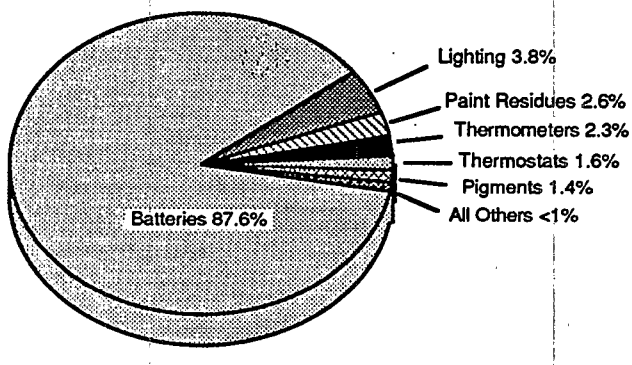
The battery industry has been under intense pressure to reduce the amounts of mercury (and other heavy metals) discarded into MSW. The industry has announced its intention to reduce mercury in alkaline batteries to 0.025 percent by weight by 1992, and to eventually eliminate all mercury from these batteries. Projections made for this report take these goals into account.

Mercury-Zinc Batteries. Mercuric oxide is used as the cathode material in mercury-zinc batteries, so mercury comprises a relatively high percentage of the material in these batteries. They contributed over 196 tons, or nearly 28 percent, of mercury discards in 1989. The amount of mercury discarded in mercury-zinc batteries has declined over the years as other kinds of batteries (silver oxide, zinc air) have taken some of their market share.

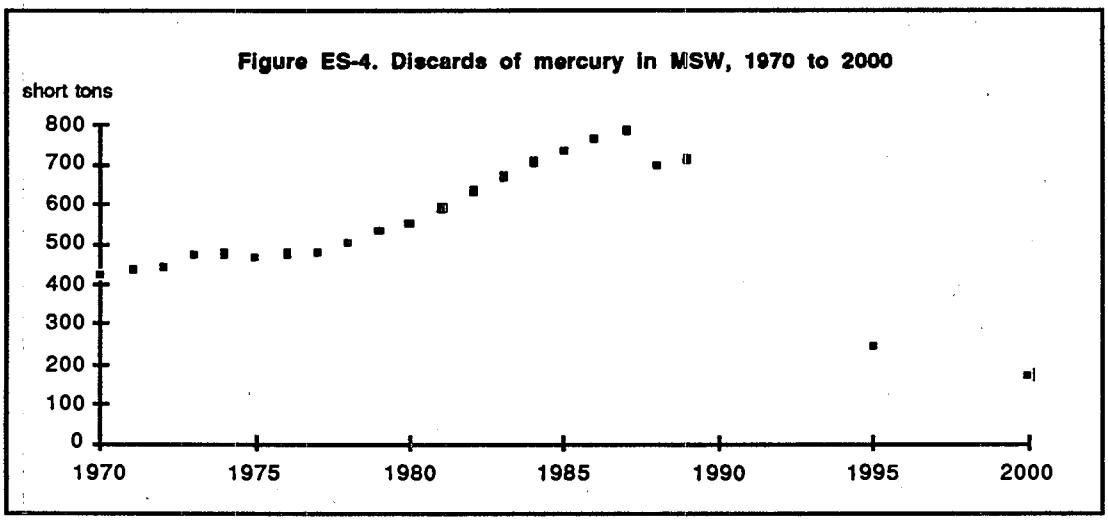
While mercury can be eliminated from alkaline batteries, it is an integral part of mercury-zinc batteries. Discards of mercury from this source were thus projected to decrease but not be eliminated. Based on the projections, only mercury-zinc batteries will be found in MSW in the year 2000.

Other Batteries. Other batteries that contributed about 5 tons, or less than one percent, of mercury discards in MSW in 1989 include carbon-zinc batteries, silver oxide batteries, and zinc air batteries. Production of carbon-zinc batteries is declining, while use of silver oxide and zinc air batteries has

Figure ES-3. Discards of mercury in MSW, 1989



Total mercury discards = 709 short tons



been increasing. It is projected that use of mercury in these batteries will be discontinued.

Recovery for Recycling. While some programs to recover batteries, either for recycling or simply to keep them out of the waste stream, were identified, the quantities recovered were not believed to be significant enough to affect discards in 1989. It was assumed for this report that 5 percent of the mercury in batteries will be recovered in 1995 and that 20 percent will be recovered in the year 2000. These recovery rates are consistent with recovery rates achieved by many other products in MSW, and lower than some.

Electric Lighting

The second largest source of mercury in MSW in 1989 was estimated to be electric lighting. This mercury came from two sources:

- The ordinary fluorescent lamps (bulbs) used in residences, offices, and other commercial and institutional buildings
- Certain high intensity lamps (bulbs) used in lighting streets, parking lots, and similar sites.

Of these two sources, fluorescent lamps are by far the largest, accounting for 26 tons of mercury in MSW in 1989, or 3.7 percent of total discards. All lighting sources were estimated to contribute nearly 27 tons of mercury in 1989, or almost 4 percent of total discards.

The mercury content of these lamps has been reduced over the past 5 years, but increasing sales will cause the amount of mercury from this source to continue to increase. New energy-efficient fluorescent lamps are being promoted as a replacement for incandescent lamps at this time, but it is too early to determine whether sales and discards of these lamps will further increase the amount of mercury discarded.

While a few attempts to recover mercury from fluorescent lamps were identified, no basis for projecting a significant amount of recovery from lamps in MSW in the future was found.

Paint Residues

By 1991, EPA had banned the use of mercury as a biocide in paints for exterior or interior use. Even though mercury is no longer used in paint manufacture, paint cans containing residues including mercury will continue to be discarded. It was estimated that about 18 tons of mercury were discarded in paint residues in 1989, or 2.6 percent of total discards. These discards are projected to decline rapidly as paints made after the ban on mercury took effect begin to be discarded. (Note that these estimates do *not* include mercury in paints applied to interior or exterior surfaces, which are not classified as municipal solid waste.)

Fever Thermometers

The familiar fever thermometer was identified as a source of mercury discarded from homes and medical establishments. In 1989, an estimated 16.3 tons of mercury were discarded in thermometers, or just over 2 percent of total discards.

Mercury fever thermometers are being replaced by digital thermometers, especially in medical applications. It therefore was projected that there will be a gradual decline in discards of mercury from this source.

Thermostats

The typical thermostat used for temperature control in residences and other buildings contains mercury that could enter MSW if the thermostat is discarded. (This mercury could also become demolition waste if the thermostat is in a demolished house.) An estimated 11 tons of mercury entered MSW in thermostats in 1989; this was less than 2 percent of total discards.

Thermostats have a long life—estimated to be 20 years—so there is a long lag time before they are discarded. Thus, even though mercury

thermostats are gradually being replaced by digital thermostats, they are projected to continue to be a source of mercury in MSW through 2000.

Pigments

Published data on the end uses for pigments containing mercury was not found. It appears that most of the mercury in pigments is used in plastics, often in combination with cadmium, but other uses could include paints, printing inks, rubber, textiles, and others. Based on the data available, it was estimated that 10 tons of mercury in pigments were discarded in 1989. This was less than 2 percent of total discards.

Use of mercury in pigments has been declining steadily. Cadmium-mercury pigments are no longer manufactured in the United States, but some imports were identified. Since there is continuing pressure on pigment makers to eliminate metals, it was projected that use of mercury in pigments will continue to decline rapidly.

Other Sources of Mercury in MSW

Other sources of mercury in products discarded in MSW include dental amalgams, a special paper coating used with cathode ray tubes, and mercury electric light switches. Together these uses amounted to less than one percent of mercury in MSW discards in 1989.

Use of mercury in dentistry is declining, and the manufacturers of the special paper have announced plans to discontinue use of mercury by 1995. Of this group, only mercury light switches are a growing source of mercury in MSW. Their discards are projected to total about 2 tons in 2000, or about one percent of total discards in that year.

Mercury was formerly a component of batteries used in instant camera film packs, but this use was discontinued in 1988.

Discontinued Sources of Mercury in MSW

Research for this report identified several products that can be classified as MSW where mercury has been used in the past. These sources were not quantified, but are listed below:

- Mirrors (discontinued about 50 years ago)
- Glass in highly specialized applications
- Felt (discontinued in the 1950s)
- Textiles intended for outdoor use
- Paper (discontinued about 1972).

MERCURY IN NON-MUNICIPAL SOLID WASTE PRODUCTS

While the purpose of this report was to quantify sources of mercury in municipal solid waste, other products containing mercury were identified in the research. Some of these wastes could very likely be managed in a landfill or combustor intended primarily for MSW.

Agricultural Products

Mercury and mercury compounds have been used as fungicides for agricultural purposes. These uses were greatly restricted by FIFRA, and presently only applications for treatment of outdoor textiles, to control brown mold on freshly sawn lumber, to control Dutch elm disease, and to control snow mold are allowed. No use is permitted on food crops.

Paints

In the past mercury compounds were widely used as biocides or preservatives in paint, especially in latex paints. Mercury was also formerly used in antifouling paints for marine use, but this use was banned in 1972. By regulatory action taken in 1990 and 1991, EPA prohibited further use of mercury in indoor or outdoor paints manufactured in the U.S.

Paint containing mercury manufactured before the ban may, however, still be discarded as a residue (see above), and demolition waste including mercury in paint will undoubtedly still be discarded.

Chlorine and Caustic Soda Production

Mercury is used in the manufacturing process for the production of chlorine and caustic soda. In fact, this use was the largest consumer of mercury in the United States in 1989. This use was classified as industrial, not MSW.

Other Non-MSW Sources of Mercury

Other uses for mercury not classified as entering MSW include the catalysis of various plastics, explosives, laboratory uses, residues of pharmaceuticals and cosmetics, and certain electrical apparatus.

Discontinued Uses of Mercury

A number of non-MSW applications for mercury that have been discontinued were identified. These include embalming fluid, photographic development, soap, and wood preservatives.

LIMITATIONS OF THIS REPORT

The purpose of this report was to characterize the sources of mercury in municipal solid waste. The characterization applies to the United States as a whole, and should not be construed to be representative of mercury in MSW in a particular locality. Local variations in waste composition and in waste management practices may cause the mercury content at any particular facility to vary from the United States average.

In many cases, the amounts of historical, current, and projected mercury in products in MSW are not well documented in any available data source. The estimates in this report are, therefore, often based on assumptions, which are documented in the report.

Identification of alternatives and substitutes for mercury in products was not part of the work scope for this report. Information on these topics must come from other sources.

A SUMMARY OF MERCURY EMISSIONS AND APPLICABLE CONTROL TECHNOLOGIES FOR MUNICIPAL WASTE COMBUSTORS

A SUMMARY OF MERCURY EMISSIONS AND APPLICABLE CONTROL TECHNOLOGIES FOR MUNICIPAL WASTE COMBUSTORS

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1.0 INTRODUCTION

This report provides information on mercury emission rates and control technologies applicable to municipal waste combustors (MWC's). Section 2.0 presents emissions data for MWC's located in North America. Discussions on apparent relationships between mercury removal and various parameters such as particulate matter (PM) control device inlet temperature and levels of carbon in the fly ash are included.

Section 3.0 discusses mercury control technologies currently being used in Europe and Canada, and includes a review of MWC's that use sodium sulfide (Na_2S) injection, activated carbon injection, and wet scrubbing to limit mercury emissions. Section 4.0 provides a listing of the references used.

2.0 EXISTING MERCURY DATA EMISSIONS

2.1 REVIEW OF EXISTING DATA

Table 2-1 lists mercury emissions data for MWC's in North America that are not equipped with specific mercury control technologies. The data cover 47 MWC units at 32 different plants. For those facilities reporting both inlet and outlet mercury concentrations (in micrograms/dry standard cubic meter [µg/dscm] at 7 percent O_2), mercury removal efficiencies are calculated. Table 2-1 also lists the type of combustor, type of air pollution control device (APCD), PM control device inlet temperature, and inlet dioxin/furan (CDD/CDF) concentrations.

Uncontrolled (i.e., APCD inlet) mercury concentrations are reviewed in Section 2.2. Section 2.3 reviews the controlled (i.e., APCD outlet) concentrations for different APCD types and discusses the apparent relationship between controlled mercury levels versus APCD type and PM control device temperature. The temperature entering the PM control device is important because mercury exists in a vaporous form at temperatures greater than 300 °F and does not effectively condense onto PM. The relationship with inlet CDD/CDF concentrations is also examined in this section. Inlet CDD/CDF levels serve as a surrogate for estimating residual carbon in the fly ash, which may enhance mercury removal due to the adsorption of mercury onto carbon. A summary of the results and conclusions drawn from these relationships is presented in Section 2.4.

2.2 UNCONTROLLED MERCURY LEVELS

As listed in Table 2-1, mercury levels prior to APCD's range from roughly 200 to 1400 µg/dscm. Most of the concentrations range from 400 to 1000 µg/dscm and average roughly 650 µg/dscm. Based on the data, there is no clear distinction in inlet mercury levels at mass burn plants and refuse-derived fuel (RDF) plants.

TABLE 2-1. EXISTING MERCURY EMISSIONS DATA

MWC NAME	RUN NUMBERS	COMBUSTOR TYPE	APCD TYPE	APCD TEMP. (F)	INLET CDD/CDFa (ng/dscm)	INLET Hg AVERAGEa (ug/dscm)	OUTLET Hg AVERAGEa (ug/dscm)	REMOVAL EFFICIENCY (%)	REFERENCE
Detroit (7/89)	9 Runs	RDF	ESP	600	--	--	653	--	1a
Detroit (11/89)*	3 Runs	RDF	ESP	600	--	--	96	--	1a
Detroit (11/89)**	14 Runs	RDF	ESP	600	--	--	193	--	1a
Detroit (12/89-1/90)***	13 Runs	RDF	ESP	600	--	--	172	--	1a
Detroit (3/90)	9 Runs	RDF	ESP	600	--	--	194	--	1b
Hillsborough	3 Runs	MB/WW	ESP	--	--	--	823	--	1c
Oneida County	1-3	M/S	ESP	NM	--	--	2060	--	2
Pigeon Point	1-3	M/E	ESP	410	--	--	363	--	2
Pinellas County	1-3	MB/WW	ESP	543	54	--	847	--	2
Pope/Douglas	1-3	M/E	ESP	482	--	--	133	--	2
Quebec City	2,10,11	MB/WW	ESP	406	--	--	918	--	2
Quebec City	5,6,12	MB/WW	ESP	417	--	--	685	--	2
Tulsa	1-3	MB/WW	ESP	375	--	--	418	--	2
Tulsa	3 Runs	MB/WW	ESP	--	--	--	1000	--	1c
Tulsa	3 Runs	MB/WW	ESP	--	--	--	746	--	1c
Tulsa	3 Runs	MB/WW	ESP	--	--	--	600	--	1c
Tulsa	3 Runs	MB/WW	ESP	--	--	--	711	--	1c
Tulsa	3 Runs	MB/WW	ESP	--	--	--	97	--	1c
Tulsa	2 Runs	MB/WW	ESP	--	--	--	466	--	1c
Dayton	1-3	MB/R	ESP	560	252	962	1016	-5.6	2
Dayton	4-6	MB/R	ESP	401	328	1055	1150	-9.0	2
Dayton	10-12	MB/R	FSI/ESP	394	38	973	757	22.2	2
Dayton	13-15	MB/R	FSI/ESP	298	14	907	709	21.8	2
Dayton	16-18	MB/R	DSI/ESP	308	5	716	491	31.4	2
Alexandria, Unit 1 (12/88)	1-3	MB/WW	FSI/ESP	--	--	--	517	--	1c
Burnaby, Unit 3 (11/88)	1-3	MB/WW	DSI/FF	313	78	527	485	8.0	3
Burnaby, Unit 1 (4/89)	1-3	MB/WW	DSI/FF	307	--	1360 b	--	--	3
Burnaby, Unit 1 (9/89)	1-3	MB/WW	DSI/FF	324	--	--	470	--	3
Burnaby, Unit 2 (9/89)	1-3	MB/WW	DSI/FF	325	--	--	368	--	3
Burnaby, Unit 3 (9/89)	1-3	MB/WW	DSI/FF	319	--	--	1088	--	3
Dutchess County, Unit 1	1-3	MB/WW	DSI/FF	430	--	--	1080	--	2
Dutchess County, Unit 2	1-3	MB/WW	DSI/FF	365	--	--	85	--	2
Quebec City - Pilot	5-6	MB/WW	DSI/FF	400	1597	451	614	-36.1	2
Quebec City - Pilot	1,2,11	MB/WW	DSI/FF	285	2277	320	16	95.0	2
Quebec City - Pilot	3-4	MB/WW	DSI/FF	250	2361	480	13	97.3	2
Quebec City - Pilot	12-13	MB/WW	DSI/FF	231	887	445	40	91.0	2

TABLE 2-1. EXISTING MERCURY EMISSIONS DATA

MWC NAME	RUN NUMBERS	COMBUSTOR TYPE	APCD TYPE	APCD TEMP. (F)	INLET CDD/CDFa (ng/dscm)	INLET Hg AVERAGEa (ug/dscm)	OUTLET Hg AVERAGEa (ug/dscm)	REMOVAL EFFICIENCY (%)	REFERENCE
Charleston, Units A & B	1-3	MB/WW	SD/ESP	--	-- c	--	723	--	4
Haverill, Unit A (6/89)	1-3	MB/WW	SD/ESP	285	--	--	247	--	1c
Haverill, Unit B (6/89)	1-3	MB/WW	SD/ESP	285	--	--	208	--	1c
Haverill, Unit B (3/90)	1-3	MB/WW	SD/ESP	--	--	--	567	--	1c
Honolulu, Unit 1	1-3	RDF	SD/ESP	300	-- d	--	5	--	5
Honolulu, Unit 2	1-3	RDF	SD/ESP	293	-- d	--	7	--	5
Millbury, Unit 1	1-6	MB/WW	SD/ESP	249	--	--	565	--	2
Millbury, Unit 2	1-3	MB/WW	SD/ESP	240	170 e	--	954	--	2
Portland, Unit 1 (12/89)	4-6	MB/WW	SD/ESP	308	-- f	--	550	--	6
Portland, Unit 2 (12/89)	1-3	MB/WW	SD/ESP	285	-- f	--	382	--	6
SEMASS, Unit 1	1-3	RDF	SD/ESP	287	-- g	--	59	--	7
SEMASS, Unit 2	2-4	RDF	SD/ESP	293	-- g	--	105	--	7
West Palm Beach, Unit 1	3 Runs	RDF	SD/ESP	275	--	--	56	--	8
West Palm Beach, Unit 2	3 Runs	RDF	SD/ESP	278	--	--	23	--	8
Babylon, Unit 2	1-3	MB/WW	SD/FF	331	--	--	451	--	9
Babylon	3 Runs	MB/WW	SD/FF	--	--	--	323	--	1c
Biddeford	1-3	RDF	SD/FF	278	903	389	ND	>99	2
Bristol	3 Runs	MB/WW	SD/FF	--	--	--	99	--	1c
Bristol	3 Runs	MB/WW	SD/FF	--	--	--	105	--	1c
Bristol	3 Runs	MB/WW	SD/FF	--	--	--	64	--	1c
Bristol	3 Runs	MB/WW	SD/FF	--	--	--	399	--	1c
Commerce (1987)	11,13,14	MB/WW	SD/FF	270 h	28	450	570	-26.7	2
Commerce (1988)	3,5,9	MB/WW	SD/FF	290 h	446 e	453	39	91.4	2
Commerce (1988)	13,16,18,29	MB/WW	SD/FF	290 h	783 e	261	68	74.0	2
Fairfax	3 Runs	MB/WW	SD/FF	--	--	--	406	--	1c
Fairfax	3 Runs	MB/WW	SD/FF	--	--	--	466	--	1c
Fairfax	3 Runs	MB/WW	SD/FF	--	--	--	331	--	1c
Fairfax	3 Runs	MB/WW	SD/FF	--	--	--	514	--	1c
Hempstead, Unit 1 (9/89)	1-3	MB/WW	SD/FF	310 h	--	--	9	--	10
Hempstead, Unit 2 (9/89)	1-3	MB/WW	SD/FF	310 h	--	--	25	--	10
Hempstead, Unit 3 (10/89)	1-3	MB/WW	SD/FF	310 h	--	--	25	--	10
Huntsville	3 Runs	MB/WW	SD/FF	--	--	--	1275	--	1c
Huntsville	3 Runs	MB/WW	SD/FF	--	--	--	463	--	1c

TABLE 2-1. EXISTING MERCURY EMISSIONS DATA

MWC NAME	RUN NUMBERS	COMBUSTOR TYPE	APCD TYPE	APCD TEMP. (F)	INLET CDD/CDFa (ng/dscm)	INLET Hg AVERAGEa (ug/dscm)	OUTLET Hg AVERAGEa (ug/dscm)	REMOVAL EFFICIENCY (%)	REFERENCE
Indianapolis, Unit 1	1-3	MB/WW	SD/FF	307	--	--	283	--	11
Indianapolis	3 Runs	MB/WW	SD/FF	--	--	--	200	--	1c
Indianapolis	3 Runs	MB/WW	SD/FF	--	--	--	277	--	1c
Long Beach	1-3	MB/WW	SD/FF	298	305	--	180	--	2
Marion County	4-6	MB/WW	SD/FF	272	43 e	--	239	--	2
Mid-Connecticut (7/88)	1-3	RDF	SD/FF	276	1019	1008	--	--	2
Mid-Connecticut (7/88)	1-3 (Hg)i	RDF	SD/FF	284	--	884	50	94.3	2
Mid-Connecticut (2/89)	12-14	RDF	SD/FF	--	436	668	9	98.7	2
Quebec City - Pilot	7-8	MB/WW	SD/FF	282	1764	187	10	94.7	2
Quebec City - Pilot	9-10	MB/WW	SD/FF	284	2157	360	19	94.7	2
Stanislaus County, Unit 1	14,16,19	MB/WW	SD/FF	295	--	--	499	--	2
Stanislaus County, Unit 2	38,40,42	MB/WW	SD/FF	290	--	--	462	--	2
Stanislaus County	3 Runs	MB/WW	SD/FF	--	--	--	508	--	1c
Stanislaus County	3 Runs	MB/WW	SD/FF	--	--	--	481	--	1c
Stanislaus County	3 Runs	MB/WW	SD/FF	--	--	--	427	--	1c
Kent	3 Runs	MB/WW	SD/FF	--	--	--	166	--	1c
Kent	3 Runs	MB/WW	SD/FF	--	--	--	248	--	1c

M/E - Modular/Excess Air
M/S - Modular Starved Air
MB/R - Mass Burn/Refractory
MB/WW - Mass Burn/Waterwall
ND - Not detected.
RDF - Refuse Derived Fuel

* Secondary trommel bypass.
** Lime fed at 1200 lb/hr.
*** Both "*" and "**" apply.

a Results reported at 7% O2.
b Inlet values reported for comparison purposes; outlet values influenced by Na2S injection.
c Outlet CDD/CDF values: Unit B - 44.2 ng/dscm (average of three runs conducted during same test campaign).
d Outlet CDD/CDF values: Unit 1 - 6.3 ng/dscm; Unit 2 - 3.8 ng/dscm.
e Inlet CDD/CDF samples collected during separate runs from Hg, but during same test campaign and at similar operating conditions.
f Outlet CDD/CDF values: Unit 1 - 36.5 ng/dscm; Unit 2 - 43.6 ng/dscm.
g Outlet CDD/CDF values: Unit 1 - 9.3 ng/dscm; Unit 2 - 12.3 ng/dscm.
h Temperatures not reported for Hg runs; temperatures estimated based on other runs during same test campaign.
i Additional inlet and outlet mercury samples were collected by Method 101A. Not measured simultaneously with other metals.

2.3 CONTROLLED MERCURY LEVELS

2.3.1 Electrostatic Precipitators (ESP's)

Nine of the facilities reviewed are equipped with ESP's. With the exception of the data from the Pope/Douglas and the Oneida County MWC's, and one set of runs from the Detroit and Tulsa MWC's, mercury outlet levels range from approximately 200 to 1200 µg/dscm. The Dayton MWC was the only ESP-equipped plant that measured inlet mercury concentrations, and the corresponding removal efficiencies for the ESP-only averages at Dayton were both negative (-9.0 and -5.6 percent).

Based on typical uncontrolled levels, the measured outlet mercury levels at ESP-equipped MWC's suggest little or no removal of mercury by this control device type. Flue gas temperatures reported for these tests range from 375 °F to 600 °F. At these temperatures no mercury condensation will have occurred; therefore, no relationship between flue gas temperature and mercury outlet levels were observed.

2.3.2 Sorbent Injection/ESP

Two facilities equipped with sorbent injection for acid gas control followed by an ESP have been tested for mercury emissions. One set of data was collected from the stack of the Alexandria MWC, which is equipped with a furnace sorbent injection (FSI) system. The system generally operates at an ESP inlet temperature of 360 to 370 °F. The average emission concentration from three runs was 517 µg/dscm. Inlet CDD/CDF levels were not measured, but were probably low based on information from other Martin grate systems.[2] This suggests that levels of carbon in the fly ash at Alexandria were probably low.

Three sets of data were collected at the Dayton MWC. Two of the data sets were with FSI, and the third data set was with duct sorbent injection (DSI). Inlet and outlet mercury levels were measured during all three test sets. The mercury removal efficiencies for Runs 10-12 (FSI at an average ESP inlet temperature of 394 °F) and Runs 13-15 (FSI at an average ESP inlet temperature of 298 °F) are both 22 percent, which

correspond to average mercury outlet levels of 757 µg/dscm and 709 µg/dscm, respectively. Mercury reductions during Runs 16-18 (DSI at an average ESP inlet temperature of 306 °F) average 31 percent, corresponding to a mercury outlet level of 491 µg/dscm. Inlet CDD/CDF levels reported for all of these tests average less than 40 ng/dscm.

The data from Alexandria and Dayton indicate that sorbent injection/ESP systems may have a small impact on controlling mercury emissions, and that little or no additional benefits are achieved by lowering flue gas temperature. The limited mercury control achieved by these sorbent injection/ESP systems may have been influenced by the lack of carbon in the fly ash.

2.3.3 Sorbent Injection/Fabric Filter

Test data are available for six MWC units at three sites with DSI/fabric filter (FF) systems. Operating conditions and outlet mercury emissions varied considerably during these tests.

During testing of a pilot-scale DSI/FF system at Quebec City, reported outlet mercury emissions varied from 13 to 614 µg/dscm. At the lower readings (13 to 40 µg/dscm), the flue gas temperature at the FF inlet was less than 300 °F, and the inlet CDD/CDF levels averaged between 900 and 2400 ng/dscm. The high mercury reading (614 µg/dscm) was at a FF inlet flue gas temperature of 400 °F, and the inlet CDD/CDF concentration averaged 1600 ng/dscm. Based on measured inlet mercury concentrations, mercury removal efficiencies range from essentially zero at FF inlet temperatures of 400 °F or more, to over 90 percent at FF temperatures less than 300 °F.

At Dutchess County, average stack mercury emissions reported from Unit 1 (operating at 430 °F) and Unit 2 (operating at 365 °F) are 1080 µg/dscm and 85 µg/dscm, respectively. No inlet mercury or CDD/CDF measurements were made.

At Burnaby, outlet mercury emissions range from 368 to 1086 µg/dscm. Inlet mercury levels were measured during two sets of runs and suggest little or no mercury reductions. Flue gas inlet temperatures at the FF reported for these tests are between

307-325 °F. Inlet CDD/CDF levels were measured during the first set of tests only, and were relatively low at 78 ng/dscm.

These data suggest that mercury removal may be a function of flue gas temperature and inlet fly ash carbon content. At high flue gas temperatures (greater than 400 °F) or low CDD/CDF levels (less than 200-300 ng/dscm), mercury control is low. At lower temperatures and higher CDD/CDF levels, however, the level of mercury control increases. At Quebec City, mercury reductions exceed 90 percent at flue gas temperatures of less than 300 °F and inlet CDD/CDF levels exceeding 800 ng/dscm.

2.3.4 Spray Dryer/ESP

Outlet mercury emissions data for SD/ESP systems are available for 14 units at seven MWC facilities. Four of the MWC's, Charleston, Haverill, Millbury, and Portland, are mass burn combustors. The other three, Honolulu, SEMASS, and West Palm Beach, are RDF units. Outlet mercury levels at the mass burn MWC's range from approximately 210 to 950 µg/dscm. Reported ESP inlet temperatures are approximately 300 °F or less. The RDF units operated at similar temperatures. Outlet mercury levels for these units, however, range from 5 to 105 µg/dscm. Due to the suspension firing of fuel in the combustor, RDF units generally have higher PM loadings and higher carbon contents at the combustor exit than do mass burn units. The results from the RDF units support the theory that increased levels of carbon in the fly ash enhance mercury removal.

2.3.5 Spray Dryer/Fabric Filter (SD/FF)

Mercury data were obtained from 14 MWC's (17 units) that use SD/FF's. With the exception of one test average from the Huntsville MWC, outlet mercury levels from these plants vary from below detection to 570 µg/dscm. The high test average from the Huntsville MWC (1275 µg/dscm) was due to one high run of over 2700 µg/dscm.

Flue gas temperatures entering the FF's were less than 300 °F at all of the SD/FF-equipped facilities. As with the

SD/ESP data, the lowest mercury outlet levels and highest removal efficiencies occurred at the two RDF plants, Biddeford and Mid-Connecticut. The average inlet CDD/CDF levels at these two plants were 903 µg/dscm and 436 µg/dscm, respectively. The Quebec City mass burn/waterwall (MB/WW) pilot-scale SD/FF test achieved 94.7 percent removal and had an inlet CDD/CDF level of 2157 ng/dscm.

The lowest reported mercury removal efficiency (-27 percent) was during the 1987 tests at the Commerce MWC. During this testing, inlet CDD/CDF levels were low, averaging 28 ng/dscm. During subsequent testing the following year, average mercury outlet levels were approximately 40 µg/dscm (91 percent removal) and 70 µg/dscm (74 percent removal). During the 1988 testing the mercury inlet levels were similar to those during the 1987 test, but the CDD/CDF levels were much higher, averaging 450 ng/dscm and 780 ng/dscm. These data further support the theory that increased levels of carbon in the fly ash enhance mercury removal.

It should be noted that the Hempstead MWC, which is a mass burn MWC, also had low mercury emissions -- below 25 µg/dscm. Inlet CDD/CDF levels were not reported.

2.4 SUMMARY

The test data indicate that mercury emission levels from municipal solid waste combustion vary significantly from site to site. Factors such as waste composition, combustion efficiency (carbon burnout), and APCD type and operation may affect mercury removal. Their exact effects are unclear. However, it appears that good PM control, low temperatures in the APCD system, and significant carbon in the fly ash are necessary to achieve mercury control. As discussed previously, the combination of a low PM control device operating temperature and a high level of carbon in the fly ash (as indicated by the level of CDD/CDF at the combustor exit) enhance mercury adsorption onto particles which are removed by the PM control device.

Based on compliance test data, the PM operating temperature that is critical for mercury capture is not clear, but appears to be in the range of 300 to 400 °F. Based on published literature citing theoretical calculations, however, a temperature of around 300 °F appears to be necessary.[12] The data indicate that RDF units (which generally have higher uncontrolled PM and CDD/CDF at the combustor exit, and thus, higher expected fly ash carbon levels) can achieve mercury emissions less than 110 µg/dscm when equipped with a good PM control system operating at less than 300 °F. The mercury emissions data for mass burn units range from 10 to over 1000 µg/dscm, even when equipped with the best acid gas/PM control systems. These units are generally characterized as having low organic emissions, low fly ash carbon content, and low PM loadings relative to RDF units.

3.0 MERCURY EMISSION CONTROL TECHNOLOGIES

Mercury control technologies include the injection of Na_2S, activated carbon or modified activated carbon into the flue gas prior to the DSI or SD-based acid gas control system. Alternatively, wet scrubbing can be used for mercury control. These technologies have not been used on U.S. MWC's, but have been applied to MWC's in Europe, Canada, and Japan. Brief discussions of these technologies are presented in this section.

3.1 SODIUM SULFIDE

3.1.1 Chemistry

Sodium sulfide is a crystalline solid that dissolves in water to form a solution of up to 10 weight percent Na_2S in 34 ^oF water and 15 weight percent Na_2S at 60 ^oF. The resulting Na_2S solution is sprayed into the flue gas prior to the acid gas control device. Aqueous Na_2S is caustic and will off-gas toxic hydrogen sulfide (H_2S). The reaction of Na_2S and Hg precipitates solid HgS that can be collected in the PM control device. The specific reactions of Na_2S and Hg are not totally understood, but appear to be:

$$HgO\ (gas) + Na_2S \cdot H_2O \rightarrow HgS\ (solid) + NaOH, \text{ and}$$

$$HgCl_2\ (gas) + Na_2S \cdot H_2O \rightarrow HgS\ (solid) + NaCl \cdot H_2O$$

While (1) flue gas temperature, or (2) lime or ammonia injection for acid gas or NO_x control may affect these mercury reactions, their effects are uncertain. Testing is currently being conducted that should provide insight on some of these relationships.

3.1.2 Existing Use of Na_2S By MWC's

Sodium sulfide is or has been used for mercury control by MWC's in Avesta, Koping, and Hogdalen, Sweden; Kempten, and Munich (South), Germany; and Burnaby, British Columbia. Injection of Na_2S has been used at the Hogdalen MWC since 1986. The Avesta, Koping, and Kempten plants began Na_2S injection in 1989. The Munich plant began operation with Na_2S injection in 1990. The Burnaby MWC began testing of Na_2S in 1989 and began

continuous operation with a temporary system in December 1989. In October 1991, however, the Burnaby plant is intending to switch to activated carbon injection for mercury control.

All of these facilities use DSI/FF systems supplied by Fläkt for acid gas and PM control. Injection of Na_2S occurs prior to the DSI system at flue gas temperatures of 265-480 °F. Hogdalen reduces flue gas temperatures prior to Na_2S injection with a heat exchanger that provides hot water for district space heating. The Burnaby and Munich MWC's use water quench towers for flue gas cooling. Flue gas temperatures at the stack at Burnaby normally range from 260-300°F. Additional information on the Munich plant was not available.

Fläkt reports that Na_2S feed rates vary from 0.05 to 0.5 kg/Mg (0.1 to 1 lb/ton) of MSW, depending on site-specific conditions such as the amount of mercury in the flue gas, the level of control required, and the level of carbon present in fly ash.[13] As discussed in Section 2.0, residual carbon in the fly ash is believed to promote mercury removal through adsorption onto the carbon. As a result, if a plant has little carbon in its fly ash, it may be necessary to increase the amount of Na_2S injected.

Mercury control performance data with Na_2S injection are shown in Tables 3-1 and 3-2. The data have been compiled from information provided by Fläkt, the Burnaby MWC facility owner (the Greater Vancouver Regional District (GVRD)), and from trip reports to the Hogdalen and Burnaby MWC's.[3,13,14,15,16,17]

Mercury levels prior to Na_2S injection at the Burnaby MWC (400-1400 µg/dscm) are higher than general inlet values reported at European MWC's (55-560 µg/dscm). The objective of the testing conducted at the Burnaby MWC was to evaluate key system parameters. During the initial tests, 1 to 3 kg/hr (2 to 7 lb/hr) of Na_2S was fed as 10-15 percent concentration solutions and achieved mercury reductions of 50-65 percent. Subsequent tests conducted at a feed rate of 2 to 6 kg/hr (4 to 13 lb/hr) of

TABLE 3-1. MERCURY EMISSIONS DATA FROM THE BURNABY MWC

MWC PLANT	WASTE TPD	APCD TYPE	Na2S FEED RATE (kg/hr)	TESTED Hg EMISSIONSa INLET (ug/dscm)	OUTLET (ug/dscm)	REDUCTION EFFICIENCY (%)
Burnaby	265	DSI/FF	Run 1 = 1.0	1465	570	
3/89b			Run 2 = 2.0	993	407	
(10% Na2S)			Run 3 = 2.0	1151	393	
				AVG 1203	AVG 457	AVG 62
Unit 1, 4/89			All runs =	1423	670	
(15% Na2S)			3.0	1443	750	
				1205	473	
				AVG 1357	AVG 632	AVG 53
8/89b			Run 1 = 2.5	406	98	
(2-4% Na2S)			Run 2 = 6.0	775	91	
			Run 3 = 2.0	670	84	
			Run 4 = 3.0	793	101	
			Run 5 = 6.0	661	103	
				AVG 661	AVG 95	AVG 86
Unit 1, 12/89			All runs =	NR	138	
(2% Na2S)			4.0	NR	67	
				NR	146	
					AVG 117	NA
Unit 2, 12/89			All runs =	NR	149	
(2% Na2S)			4.0	NR	115	
				NR	118	
					AVG 127	NA
Unit 3, 12/89			All runs =	NR	152	
(2% Na2S)			4.0	NR	159	
					AVG 155	NA

NR - Not reported; NA - Not applicable

a Results reported at 12% CO2 (assumed to be equal to 7% O2).

b Unit not specified.

TABLE 3-2. MERCURY EMISSIONS DATA FROM THE HOGDALEN AND KEMPTEN MWC's

MWC PLANT DATE	WASTE TPD	APCD TYPE	APCD INLET TEMP (F)	Na2S FEED RATE (kg/hr)	TESTED Hg EMISSIONS[a] INLET (ug/dscm)	OUTLET (ug/dscm)	REDUCTION EFFICIENCY (%)
Hogdalen, Unit 3	265	DSI/FF					
(02/86)			343	--	--	96	
(05/86)			316	--	181	168	7
(08/86)			304	1.2	--	65	
(08/86)			343	1.2	344	37	89
(08/86)			345	1.2	463	57	88
(09/86)			304	0.9	--	51	
(12/86)			334	1.2	--	28	
(02/87)			253	1.2	310	3	99
(03/87)			334	--	124	105	15
(04/87)			334	1.2	207	54	74
(09/87)			336	1.2	366	30	92
(12/87)			336	--	--	139	
(04/88)			336	--	266	136	49
(05/88)			325	1.2	369	22	94
(08/88)			311	1.2	121	18	85
(08/88)			311	1.2	--	8	
(08/88)			311	--	--	22	
(10/88)			291	1.8	102	13	87
(10/89)			270	1.2	158	142	10
(11/89)			280	2.4	192	4	98
(02/90)			280	1.2	--	9	
(02/90)			262	1.2	--	8	
(02/90)			261	2.4	--	6	
(06/90)			266	2.4	--	26	
Kempten (Germany)	210	DSI/FF	--	--	--	<56	

a Results reported at 12% CO2 (assumed to be equal to 7% O2).

Na_2S and a solution concentration of 2-4 percent achieved average mercury reductions of 86 percent and outlet mercury concentrations between 84 and 103 µg/dscm. Testing conducted at a Na_2S feed rate of 4 kg/hr (9 lb/hr) and a solution concentration of 2 percent achieved average outlet mercury concentrations between 117 and 155 µg/dscm; inlet mercury concentrations were not measured during these tests, therefore percent reductions could not be calculated. The improved mercury reduction at lower Na_2S concentrations is believed to be the result of improved atomization and mixing when feeding higher volumes of low concentration solution versus lower volumes of high concentration solutions.

Mercury performance data for the Hogdalen (Unit 3) and Kempten MWC's are presented in Table 3-2.[14,15,16,17] Mercury testing with Na_2S injection at the Hogdalen facility began in the summer of 1986. Testing prior to the installation of the Na_2S injection system indicated mercury levels as high as 165 µg/dscm. Testing in 1986 with a Na_2S feedrate of 1.2 kg/hr (2.6 lb/hr) and 0.9 kg/hr (2 lb/hr) decreased mercury emissions to between 37 and 65 µg/dscm. When inlet levels were measured, emission reductions were 88 and 89 percent. Subsequent testing with Na_2S injection in 1987 resulted in similar emission levels, with emission reductions between 74 and 99 percent. For those tests conducted without Na_2S injection, minimal mercury control was achieved.

In the summer of 1988, a heat exchanger was installed, replacing a precooler, which resulted in improved performance of the Na_2S system. Other changes to the system have been made over the course of operation (see discussion in Section 3.1.3) which also improved performance. Test results from 1989 and 1990, excluding the October 1989 testing, show mercury levels between approximately 5 and 25 µg/dscm. The poor performance during the October 1989 testing may be due to the operating conditions during the test--the boiler load was constantly increased from low load to full load, which increased the boiler wall temperatures and potentially volatilized mercury adsorbed on

collected soot. Also note that the high level of performance during the August 1988 testing without Na_2S injection was measured immediately after a measurement with Na_2S injection. Therefore, the results are uncertain.[17]

Injection of Na_2S on the two older units at the Hogdalen plant was also investigated, but was discontinued since these units were able to achieve low mercury levels (< 4 µg/dscm) without using the Na_2S system. The high collection of mercury has been attributed to the high content of unburned carbon in the flue gases from these two older units.[14,17]

Limited information is available on the Kempten, Germany MWC, but outlet mercury levels at Kempten when using Na_2S injection averaged less than 56 µg/dscm.

As indicated by the data, typical inlet mercury levels at the European facilities are lower than those in the U.S. and Canada. This may be the reason for Hogdalen's lower Na_2S feedrates and the resulting Hg outlet levels at Hogdalen and Kempten. The reduction efficiencies, however, are generally similar for all facilities currently using Na_2S injection for mercury control.

3.1.3 Potential Technical Limitations with Na_2S Injection

All of the existing MWC's using Na_2S injection are equipped with DSI/FF systems. As a result, some uncertainty exists regarding the applicability of Na_2S injection to other APCD configurations, such as SD/FF's and SD/ESP's. Potential problems related to application of Na_2S to spray drying systems include the existence of adequate time for reaction between mercury and aqueous Na_2S and possible reactions between Na_2S and acid gas control sorbent. If Na_2S is injected into hot flue gas (e.g., 450 ^{o}F), the associated water may evaporate rapidly, leaving a dry Na_2S particle, which may be less reactive with mercury. If Na_2S and calcium in the sorbent react to form calcium sulfide (CaS), the availability of S for reaction with mercury would be diminished and reduce the mercury collection efficiency. Fläkt stated that they do not believe this was a problem, but do not

have any actual operating experience with application of Na_2S to spray drying systems. Fläkt did indicate, however, that it would probably be necessary to have separate Na_2S and calcium sorbent feed and injection systems to avoid CaS scaling of the sorbent feed line. A concern related to the use of ESP's is the collection capability of an ESP if the HgS precipitates as a very fine particulate.[3,13]

The only operating problem reported by Fläkt was gas-side corrosion of cool surfaces (such as the Na_2S piping and nozzles) by condensation of HCl from the humid flue gas. To prevent this, hastalloy steel has been used on these surfaces. An initial concern raised by the Burnaby MWC operator was an apparent increase of roughly 50 percent in lime consumption rates following installation of the Na_2S system. However, due to limited operating experience at that time, the plant operator was not certain whether this increase was due to Na_2S use or was caused by other changes in plant operations. The plant operator had carefully inspected the DSI/FF system during the plant's last outage prior to beginning continuous injection of Na_2S. The plant had not been out of service since restarting operation, therefore, the operator had not been able to re-examine the unit for corrosion or other problems.[3,13] Recent information from the plant did not indicate the occurance of any such problems.[18] Further, recent discussions with plant personnel indicated that the lime consumption increase was not related to the use of NA_2S injection.[19]

The Hogdalen plant encountered a problem due to moisture buildup which resulted in the clogging and plugging in the screw conveyor that was used to transport the sodium sulfide to the mix tank. Additionally, the sodium sulfide caked up and solidified due to pressurization of the storage silo when transferring the sodium sulfide. To overcome these problems, Hogdalen eliminated the storage silo and developed a system in which 500 kg bags of sodium sulfide are emptied directly into the mix tank.[14,17]

Another problem encountered at Hogdalen was sludge buildup occurring in the mix tank as a result of the presence of inorganic salts in the mixing water. To remedy this, treated boiler feed water is now used for mixing. Other operational modifications at the Hogdalen plant include the use of piston-type pumps rather than impeller pumps in order to maintain a more consistent feed rate, and the injection of the sodium sulfide downstream of the heat exchanger to prevent clogging of the nozzle.[14,17]

None of the problems discussed above appear to be of a magnitude to raise concerns about the ability of Na_2S injection to control mercury on a continuous basis.

3.1.4 Cost Estimates for Na_2S Injection

Available cost data are based on estimates from Fläkt, information provided for the Burnaby plant, and supporting chemical costs from PPG. The Burnaby MWC operator estimated capital costs for a Na_2S system for the Burnaby plant, which has a MSW combustion capacity of 800 TPD, at $150,000-250,000 (1990 dollars).[3] The chemical costs for the sodium sulfide, as quoted by Fläkt, range from $0.10-0.50/ton of MSW.[13] This cost is dependent upon the uncontrolled mercury level and the level of reduction required. The chemical cost reported for the Burnaby MWC is $0.30/ton of MSW, and the chemical cost (without shipping) reported by PPG is $0.30/ton of MSW, both of which are consistent with Fläkt's estimate. Based on this information, annualized costs (based on a capital recovery factor of 0.1315 and 8,000 hours of operation per year) for Burnaby are estimated at $0.20-0.60/ton of MSW.

3.2 ACTIVATED CARBON INJECTION

Another mercury control technology used in Europe is the injection of powdered activated carbon prior to the APCD. It is believed that the activated carbon is a catalyst for the oxidation of elemental mercury to mercuric oxide, which can be captured in the APCD.[20] This technology has been used commercially on an MWC located in Zurich, Switzerland, and during

test programs at MWC's in Amager, Denmark; Kassel, Germany; and Burnaby, British Columbia.

The Zurich MWC is equipped with an SD/ESP system. Powdered activated carbon is injected into the flue gas ahead of the SD, and the temperature entering the SD is between 430 and 540 °F. Test results from the Zurich plant are shown in Table 3-3. Testing was conducted with and without activated carbon injection and at SD outlet temperatures between 230-284 °F. For tests run without activated carbon injection, the lowering of temperature did not result in a substantial increase in mercury capture. The addition of activated carbon, however, increased the average percent removal efficiencies from the mid-40's to over 87 percent. It was observed that fluctuating mercury inlet levels did not affect performance when activated carbon was used. The affect of increasing additive was investigated during the testing at a SD outlet temperature of 248 °F. The increase from 9 mg/dscm to 20 mg/dscm to 39 mg/dscm did not have a significant impact on mercury outlet levels or removal efficiency. In all cases, the average removal efficiencies were greater than 85 percent, and average outlet levels were between roughly 30 and 90 µg/dscm.[21]

The Amager MWC is equipped with a SD/FF system and operates similarly to the Zurich plant. Testing was conducted with temperatures at the SD exit of 284 °F and at 260 °F. As shown in Table 3-4, results from the testing with activated carbon injection at the higher temperature indicate outlet mercury levels between 23 and 77 µg/dscm, corresponding to removal efficiencies between 82 and 95 percent. Without activated carbon injection, outlet mercury emissions were between 67 and 195 µg/dscm, with removal efficiencies between 15 and 65 percent. The highest removal efficiencies when using activated carbon occurred with increased additive levels (70 mg/dscm vs. 7 mg/dscm).[21]

Testing at the lower APCD inlet temperatures shows greater control of mercury, especially when activated carbon injection

TABLE 3-3. MERCURY EMISSIONS DATA FROM THE ZURICH MWC

ADDITIVE (mg/dscm)	SD OUTLET TEMPERATURE (F)	INLET MERCURY (ug/dscm)	OUTLET MERCURY (ug/dscm)	REMOVAL EFFICIENCY (%)
0	284	703	510	27
		449	310	31
		890	546	39
		730	542	26
		531	438	17
		1403	1003	28
		---	---	--
	AVERAGE	784	559	29
9	248	656	122	81
		574	62	89
		---	--	--
	AVERAGE	615	92	85
20	248	300	43	86
		230	14	94
		313	29	91
		---	--	--
	AVERAGE	281	29	90
39	248	705	51	93
		771	41	94
		202	26	87
		148	30	80
		---	--	--
	AVERAGE	457	37	92
0	239	648	304	53
		841	271	68
		306	153	50
		1242	877	29
		963	623	35
		525	327	38
		---	---	--
	AVERAGE	754	426	44
39	239	461	58	88
		462	58	88
		368	38	90
		---	--	--
	AVERAGE	430	51	88

TABLE 3-3. MERCURY EMISSIONS DATA FROM THE ZURICH MWC

ADDITIVE (mg/dscm)	SD OUTLET TEMPERATURE (F)	INLET MERCURY (ug/dscm)	OUTLET MERCURY (ug/dscm)	REMOVAL EFFICIENCY (%)
0	230	326	162	50
		293	173	41
		453	277	39
		---	---	--
	AVERAGE	357	204	43
	230	636	89	86
		851	59	93
		171	58	66
		546	52	90
		352	67	81
		---	--	--
		511	65	87

*All concentrations are at 7 percent O2.

TABLE 3-4. MERCURY EMISSIONS DATA FROM THE AMAGER MWC

ADDITIVE (mg/dscm)	SD OUTLET TEMPERATURE (F)	INLET MERCURY (ug/dscm)	OUTLET MERCURY (ug/dscm)	REMOVAL EFFICIENCY (%)
0	284	203	154	24
		229	195	15
		219	86	61
		202	74	63
		165	67	59
		---	---	--
	AVERAGE	204	115	44
7	284	378	58	85
		227	40	82
		---	--	--
	AVERAGE	303	49	84
20	284	214	31	86
		248	35	86
		336	36	89
		---	--	--
	AVERAGE	266	34	87
70	284	1516	77	95
		318	23	93
		---	--	--
	AVERAGE	917	50	94
0	260	421	32	92
		196	48	76
		163	30	82
		189	54	71
		---	--	--
	AVERAGE	242	41	80
23	260	201	24	88
83	260	198	6	97
		220	7	97
		---	--	--
	AVERAGE	209	6.5	97

*All concentrations are at 7 percent O2.

was not used. With activated carbon, outlet mercury levels ranged from 6 to 24 µg/dscm (88 to 97 percent removal), and without activated carbon the outlet levels were between 30 and 53 µg/dscm (72 to 92 percent removal).[21]

The MWC in Kassel is equipped with an ESP followed by a SD/FF, and the system has the capability of operating in either the single-pass of partial-product recycle mode. (Zurich and Amager are single-pass systems.) The recycle design of the Kassel system results in an increased chloride content of the lime slurry, therefore, the temperature exiting the SD must, at a minimum, be kept at 275 °F. Recent test results at a temperature of 279 °F and carbon additive feedrates of between 0 and 84 mg/dscm are in shown in Table 3-5. Without activated carbon injection average mercury outlet levels exceeded 750 µg/dscm (35 percent removal). By using carbon injection at levels of 25 mg/dscm and higher, mercury levels less than 75 µg/dscm were achieved, with removal efficiencies exceeding 80 percent.

Recently available information on the use of carbon injection at the Burnaby MWC indicate significant mercury reductions. Results of testing from June 1990 through January 1991 show mercury removal efficiencies averaging 84 percent. It has been determined that the final installation at the Burnaby plant will be a carbon based injection system. The annual operating costs with such a system will be less than an Na_2S injection system, and the health risk to employees (caused by the offgassing of Na_2S) will be reduced. It is expected that the final mercury control system will be operational in October 1991.[18]

List prices for activated carbon range from $0.50-1.00 per pound depending on the raw material used to produce the carbon and the available surface area.[22] Based on a carbon feed rate of 1 to 2 kg/hr, these costs correspond to $0.15-0.35/ton of MSW. Estimates of capital costs for the construction of an activated carbon injection system at the Burnaby MWC are on the order of $200,000 (1990 dollars).[18] With this information, the annualized

TABLE 3-5. MERCURY EMISSIONS DATA FROM THE KASSEL MWC

ADDITIVE (mg/dscm)	SD OUTLET TEMPERATURE (F)	INLET MERCURY (ug/dscm)	OUTLET MERCURY (ug/dscm)	REMOVAL EFFICIENCY (%)
0	279	1175	762	35
12	279	440	229	48
26	279	424	75	82
62	279	234	25	89
84	279	389	68	82

*All concentrations are at 7 percent O2.

costs (based on a capital recovery factor of 0.1315 and 8000 hours of operation per year) for Burnaby are estimated at $0.30 - 0.50/ton of MSW. One of the criteria for selecting the additive used at the Zurich MWC was its low cost.[23]

3.3 ACTIVATED CARBON/LIME INJECTION

Another mercury control technique in use in Europe is the injection of an activated carbon scrubber additive consisting of approximately 95-97 percent lime and 3-5 percent activated carbon. One of the first tests using this additive* was on the MWC in Geiselbullach, Germany, in January 1989. The plant is equipped with a DSI/FF system and has two lines, each capable of combusting 158 tpd of MSW. Despite high operating temperatures during the initial testing (sometimes as high as 465 °F exiting the combustor), mercury emissions were reduced from inlet levels of 250 to 330 µg/dscm, to outlet levels under 110 µg/dscm. Subsequent testing with inlet levels between 140 and 640 µg/dscm resulted in outlet levels between 12 and 46 µg/dscm.[24,25]

Along with the Geiselbullach MWC, activated carbon/lime injection has been used on other MWC's in Germany including the Berlin-Ruhleben, Würzburg, and Siemens-KWU MWC's. During full-scale testing using this additive at the Berlin-Ruhleben MWC, which is equipped with a SD/FF system, inlet mercury levels averaged 444 µg/dscm for Boiler 2 and 402 µg/dscm for Boiler 3. Outlet levels were reduced to average levels of 99 µg/dscm and 83 µg/dscm for Boilers 2 and 3, respectively.[25,26]

At the Würzburg MWC (DSI/FF), mercury emissions were reduced by over 80 percent to levels under 65 µg/dscm when using activated carbon/lime injection. At the Siemens-KWU MWC, which is equipped with a wet scrubber followed by a FF, injection of this additive is used as a final purification stage. Outlet levels of mercury during testing were less than 25 µg/dscm.[25]

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* Sorbalit (Manufactured by Marker Zementwerk GmBH) was the commercial produced used. All test results discussed in this section are based on the use of this product.

Activated carbon/lime injection has also been used on special waste incinerators in Germany (Schöneiche and Schwel-Brenn-Anlage), where substantial reductions in mercury were observed.[24,25]

Details on costs associated with the use of this additive were not available, but similar to activated carbon, low investment costs and easy management are cited as two of the advantages of the product.[24]

3.4 WET SCRUBBING

Wet scrubbing is a form of acid gas and metals control that has been used primarily at MWC's in Europe and Japan. Wet scrubbing of MWC flue gases typically involves passing the flue gas through an ESP to reduce PM, followed by a two-stage absorber where flue gas is contacted with water to remove HCl in the first stage and an alkaline solution in the second stage to remove SO_2. The absorber also saturates the gas stream and reduces flue gas temperatures to as low as 130 ^{o}F. Several wet scrubber designs include a fine PM collection system following the second absorber to reduce aerosol and fine particulate emissions. The alkaline solution, typically containing calcium hydroxide [$Ca(OH)_2$], reacts with the acid gas to form salts, which are generally insoluble and may be removed by sequential clarifying, thickening, and vacuum filtering. The dewatered salts or sludges are then landfilled.

Due to the low absorber operating temperature that promotes mercury condensation, wet scrubbing technology achieves high mercury reduction. Mercury emissions can be reduced by up to 90 percent. The use of liquid chelating agents enhances the coagulating sedimentation and the fixation of the mercury compound in the sludge.[27] Disadvantages of wet scrubbing, however, include the quantity of water required and potential difficulties with waste handling. To stabilize condensed mercury compounds, use of additives, such as TMT (trimercapto-s-triazine) is required in some wet scrubber designs. Failure to stabilize and remove collected mercury compounds from the scrubber solution

can result in revolatilization of mercury from the scrubber solution and, thus, reduced collection efficiency. Further, while mercury control may be higher for wet systems, control of organic emissions may be lower than that achieved with dry acid gas controls.

Test results from three wet scrubber-equipped MWC plants located in France and Switzerland are available. The two French plants, Lyon-Nord and Lyon-Sud, began commercial operation in 1989 and 1990, respectively, and they are equipped with ESP's followed by wet scrubbers. Mercury emissions results from these plants are shown in Table 3-6. Average mercury outlet emissions at Lyon-Nord were under 50 µg/dscm for Unit 1 and 62 µg/dscm for Unit 2. Average removal efficiencies were greater than 82 percent for Unit 1 and 62 percent for Unit 2. At Lyon-Sud, average mercury outlet emissions were less than approximately 60 µg/dscm for both units, and average removal efficiencies were greater than 86 percent.[28]

The Basel, Switzerland MWC, which was originally equipped with only ESP's, was retrofitted with wet scrubbing in 1989. Mercury outlet emissions, listed in Table 3-6, ranged from 16 µg/dscm to 20 µg/dscm at Unit 1, and from less than 13 µg/dscm to 34 µg/dscm at Unit 2. This corresponds to average removal efficiencies between 90 and 96 percent for Unit 1, and between 82 and 96 percent for Unit 2.[28] The higher mercury control efficiency at the Basel MWC may reflect higher unburned carbon levels in the fly ash from the older Basel unit as compared to the new combustors in Lyon.

3.5 OTHER TECHNOLOGIES

In addition to the mercury control technologies discussed in the previous sections, final stage activated carbon beds are being investigated as an applicable mercury control technique. Activated carbon beds are "back end" controls, positioned after all other APCD's in the system. As the flue gases pass through the bed, pollutants adsorb onto the porous surface.[26]

TABLE 3-6. MERCURY EMISSIONS DATA FROM WET SCRUBBING SYSTEMS

MWC NAME	UNIT	RUN NUMBER	INLET MERCURY (ug/dscm)	OUTLET MERCURY (ug/dscm)	REMOVAL EFFICIENCY (%)
LYON-NORD, FRANCE	1	1	168	<49	>71
		2	289	<50	>83
		3	578	<49	>91
	AVERAGE		345	<49	>82
	2	4	177	49	72
		5	177	76	57
		6	140	60	57
		AVERAGE	165	62	62
LYON-SUD, FRANCE	1	3	457	72	84
		4	568	<49	>91
		AVERAGE	513	<61	>88
	2	1	438	69	84
		2	373	<49	>87
		AVERAGE	406	<59	>86
BASEL, SWITZERLAND	1	1	252	16	94
		2	168	17	90
		4	401	17	96
		7	513	20	96
		8	187	19	90
	2	1	186	<13	>93
		2	224	<14	>94
		3	261	32	88
		4	224	<13	>94
		5	168	<13	>92
		6	168	21	88
		7	140	<13	>91
		20	363	13	96
		21	270	33	88
		23	75	<13	>82
		24	196	<13	>93

*All concentrations are at 7 percent O2.

A potential disadvantage to the carbon bed technology includes the possibility of fires, since activated carbons are naturally flammable and may self-ignite at temperatures as low as 175 °F.[26] Also, if the bed is regenerated, special precautions must be taken to ensure the capture of any resulting mercury emissions.

Another emerging mercury control technology includes the use of selenium filters. Such filters are used in metallurgical smelting operations, and consist of a cylindrical shell which contains graded porous material impregnated with selenium. Selenium has a strong affinity of mercury. Flue gas exiting an ESP would pass through the filter prior to the stack. The filters would need to be replaced once they are spent.[29]

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Heavy Metal Partitioning in a Municipal Solid Waste Incinerator

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PEER-REVIEW

INTRODUCTION

Waste Management in Norway

Norway has the following priorities for management of municipal solid waste (MSW)[1]:

1) Reduce waste generation and toxic components in waste
2) Encourage re-use, recycling and energy recovery
3) Secure an environmentally safe management of residues

MSW consists of household waste and waste from the service and trade industry delivered to municipal waste treatment plants or recycling schemes. In 1995, a total of 2.7 million tons of MSW (1.26 million tons of household waste and 1.44 million tons of waste from service and trade industry) was handled as follows[2]: 68% was deposited on landfills, 18% was combusted, 13% recycled and 1% composted. Combustion of MSW is handled in five larger plants with energy recovery located in different cities in Norway. In addition, a new incinerator for MSW is planned. This incinerator will have to meet the new emission regulations given by the European Union which are more stringent than the present regulations. Hence, Norway is moving towards more stringent regulations, leading to an increased interest in the environmental aspects of MSW incinerators.

Heimdal Heating Central (HHC)

One of the largest MSW incinerators in Norway is situated in Trondheim and is owned and operated by Trondheim Energy Company (TEV). HHC consists of two lines each with a maximum capacity of 6.5 tonne/hour. The furnaces are moving grate units delivered by VonRoll, Switzerland. The flue gas cleaning system consists of an electrostatic precipitator (ESP) and a wet scrubber. The MSW incinerator is a base load energy production unit in a district heating system. In 1996, a total of 85600 tonne of MSW was incinerated in this plant with a heat production equivalent to 222 GWh. Figure 1 show a sketch of the incinerator. The MSW is delivered by trucks to a bunker for storage and mixing. The waste is collected from the bunker and into the feeding hopper by a waste crane. The waste is fed to the furnace by a dosing pusher. While the waste is transported through the furnace by a moving grate, primary air is supplied for combustion from below the grates and from the side walls. The side walls are cooled and insulated with refractory. Recirculated flue gas is injected into the flame above the grate before the flue gas enters the secondary combustion chamber. The flue gas first flows through a hot water generator and the ESP, which removes dust particles, before it enters the wet scrubber. In the wet scrubber, the flue gas pass through a quencher which reduces the temperature before it enters the wet scrubber where most of the pollutants in gas phase are removed. The water from the scrubber is first treated in a neutralization tank and then in a flocculation tank where particles are separated from the scrubber water. Then the scrubber water passes a sediment reactor where the solid phase is collected in the bottom. The solid phase is transported to a filter press where filter cakes are made. The water from the cleaning process is filtered in a sand filter and a ion exchanger system, before it goes to the drain. The clean flue gas is emitted through the 70 meter high stack. The bottom ash is deposited on a landfill, while the ESP dust and filter cakes are deposited in a landfill as hazardous waste.

Objective of the project

During 1995, TEV carried out an investigation program to examine the residues from the incinerator. Primary attention was on the heavy metals in the bottom ash, fly ash and the landfill leachate. The program was conducted in order to establish more information about characteristics of the residues and thus be able to undertake a sounder evaluation of the environmental aspects of the final treatment of these products. This program was supplementary to the emission analysis done periodically for the flue

gas and drain water. The objective of this work has been to establish knowledge about the partitioning of heavy metals through the incinerator and calculate the concentrations of heavy metal in the input MSW. A comparison of the results obtained from this study with other studies such as the WASTE Program (Burnaby)[3] and Brunner/Mönch[4] is of interest to see equalities and differences in the heavy metal partitioning through the incinerator and also the estimated heavy metal concentrations of the input MSW. All of the three different MSW incinerators (HHC, Burnaby, Brunner/ Mönch) have similar furnaces (moving grate), but different cleaning systems.

Heavy metal characteristics

Heavy metals are environmental toxics which accumulates in the environment. Heavy metals do not break down, but will remain in the environment forever. Cadmium, lead, mercury, vanadium, chromium, nickel, copper, zinc and arsenic are the most important heavy metals found in emissions from combustion of MSW. Cadmium, lead and mercury got most attention due to their relative toxicity[5,6]. Some of the heavy metals, especially cadmium, can deposit in the soil and be absorbed by plants. Mercury can be transformed into methyl mercury in sediments and be accumulated in the food chain, especially through fresh water. Excessive levels of heavy metals can provoke a number of health effects. Excessive amounts of lead and mercury are especially dangerous with regards to damage to the nervous system and fetal life. Lead can also give cardiovascular diseases and anaemia. Excessive amounts of cadmium can damage the kidney after long term exposure and accumulating in the body[6,7]. In order to reduce emissions of heavy metals and other pollutant emissions, the Norwegian government has signed several agreements, both national and international, which have the aim of reducing emissions in the future. For example: the Parliament report on national reduction of 70% of 13 selected environmental toxics, the North Sea Declaration states that the emission of 40 selected substances to both air and water shall be reduced and the Montreal Protocol include reduction of ozone destructive.

EXPERIMENTAL METHODS AND PROCEDURES

The experiments performed during this investigation were mainly conducted in 1995 and 1996. The input MSW has the average composition given in table1[9]. The waste originates from the urban and rural areas in and around Trondheim municipality. The degree of material recovery of different components is estimated to[9]: paper 35%, food waste 22%, plastic/textiles/rubber 11%, glass 25%, metals 63% and other non-combustibles 50%. The total material recovery rate from MSW is estimated to 30%. In the sections below, a description of the sampling and experimental methods and procedures are explained.

Mass Balance

In this study the mass balance has been established for bottom ash, filter ash and filter cake. All of the values are mean values taken over a whole year and are on a dry basis.

All of the MSW delivered to the incinerator is weighed before dumped into the waste bunker. The weight of the waste is continuously recorded every year. The bottom ash, filter ash (ESP dust) and filter cakes are always weighed (on a wet basis) before sent to the landfill. However, the average moisture content has been determined for each of the different residues. All residues were expressed on a dry basis for the mass balance. The amount of flue gas has been calculated by measuring the flow through the stack with a pitot tube and micromanometer according to the Norwegian standard method (NS 4862). All of the moisture from the input waste was considered leaving with the flue gas.

Chemical Analysis

Chemical analysis of the different waste streams have been done according to different standard tests methods. The heavy metals presented in this investigation is mercury (Hg), zinc (Zn), chromium (Cr), cadmium (Cd), lead (Pb) and iron (Fe). Some of the residues and the flue gas has also been analysed for other heavy metals such as arsenic (As), nickel (Ni) and copper (Cu).

Bottom ash. For ten weeks samples were taken every Monday, Wednesday and Friday. The samples were taken from the bottom ash conveyor belt with a spade. Particles larger than 50 mm were sieved out and magnetic metals were removed with a magnet. Other easy visible pieces of metal was also picked out. The samples were air dried for 70 hours at 20°C before shipment to the laboratory, in order to enable crushing of the ash before analysis. Every sampling day six samples, each of approximately 2 kg, were taken with 1-1.5 hours intervals. The samples from one day were mixed and split into two parts and a sample of approximately 4 kg was sent to the laboratory for chemical analysis and another sample was analyzed for grain size distribution. The samples were sent to a certified laboratory in The Netherlands, Tauw Milieu bv, and the chemical analysis was done according to the standard test method ICP NPR 6425 for cadmium, chromium, lead and zinc; mercury was determined by the Cold Vapour-method NEN 5779. Full details of the chemical analysis of bottom ash is given in the report from Kummeneje[10].

The amount of iron in the bottom ash was determined by sieving and magnet separation test in a pilot plant with a capacity of 36 tons/hour[11]. Approximately six tons of bottom ash was handled in the sieving/magnet separation test. The bottom ash was first sieved on 50 mm sieve and this fraction was put through the magnet separator to determine the magnetic fraction. The rest fraction after sieving (<50 mm) was also separated for magnetic materials in the same system. The total amount of magnetic materials from the magnetic separation test was considered as iron.

Filter Ash. Samples of approximately 2 kg were taken once a week for three weeks. The samples were taken in the intermediate container below the conveyor belt from the filter ash bin. The samples were stored in plastic bags just above room temperature in order to prevent interaction with the surroundings before shipment to the laboratory. The laboratory, Tauw Milieu bv in The Netherlands, is certified for these chemical analysis. ICP-method NPR 6425 was used to determine the content of cadmium, chromium, iron, lead and zinc. The Cold Vapour-method NEN 5779 was used to determine the content of mercury. Full description of these chemical analysis of the filter ash is given in the report by Kummeneje[12].

Filter Cakes. Some samples of the filter cakes from the washing process were analyzed with Atom Absorption Spectrophotometry (AAS) to determine the content of lead, zinc and chromium [13]. Iron was estimated from values found in the literature. The content of mercury and cadmium was determined by measuring the difference in concentration in the water before and after the water cleaning process. For a whole year one sample was taken every day and all of the samples from one week were put together and analysed. The NS4768 and NS4781 methods were used to determine the content of mercury and cadmium in the samples.

Drain Water. The drain water has been analyzed for mercury, cadmium and lead according to Norwegian standard methods NS4768, NS4781 and NS4781. The amount of iron, zinc and chromium was estimated on the basis of values found in the literature.

Flue Gas. The Norwegian standard method NS 4863 was used to determine the content of lead, cadmium, mercury and chromium in the flue gas. The procedure is described in a SINTEF report[14]. The

content of iron and zinc was estimated on basis of values found in the literature. Four samples of the flue gas were taken for chemical analysis.

Sources of error in mass balances

In addition to the usual measurement inaccuracy's connected to every measurement done in this investigation, the closure of the mass balance also represent a source of error. Since the measurement inaccuracy's are given by the standarized measurement methods, those will not be commented on further. However, the mass balance closure need further explanation. The examination of the bottom ash, filter ash, filter cakes, drain water and flue gas has served different purposes. In addition to the determination of heavy metal balances and concentrations in input MSW, determination of heavy metal concentrations in residues and flue gas with regards to soil, water and air pollution has been one of TEV's objectives with these investigations. All of the measurements used in the mass balances has, therefore not been performed within the same period of time and with different sampling rates. However, this investigation has tried to give relatively long term mass balances and determined the concentrations of heavy metals in input MSW on basis of these findings. Short term investigations within the same time period has both the advantages and disadvantages of stable conditions with regards to operational parameters and composition of input MSW. The closure of the mass balances is therefore more likely to happen in investigations with parallel sampling of the residues and flue gas rather than sampling in series. However, the heavy metal balances and concentrations in input MSW is also a function of the variations in operational conditions and composition of input MSW. More extensive investigations with sampling in parallel would be very interesting and would provide a more complete picture of the mass balances and concentrations of heavy metals in input MSW. The cost of such comprehensive measurements is one of the major problems.

HEAVY METAL PARTITIONING THROUGH THE INCINERATOR

The results of the mass balance, heavy metal partitioning through the incinerator, heavy metal content in the residues and flue gas are presented in the sections below. Further, calculated heavy metal concentrations in the input MSW on the basis of the heavy metal content of the residues and flue gas are presented.

Mass Balance

The result of the mass balance is given in figure 2. The figure show that 83 % of the input MSW is converted to CO_2 and H_2O and is emitted as moist flue gas. The mass flux of bottom ash was found to be 16.5 % and the mass flux of filter ash was 0.37 %. The dust found in the filter cakes was almost negligible (0.02%). Brunner and Mönch[4] found in their investigation a bottom ash portion of 20.5% and a flue gas portion of 77%. The flux of filter ash was the same as this study.

Heavy Metal Balance

Figure 3 and table 2 show the heavy metal balance between the different residues and flue gas for the incinerator.

Cadmium (Cd). Cadmium is a quite volatile metal in a combustion context with a boiling point of 767°C. Figure 3 show that 63% of the cadmium remain in the bottom ash, 24% is captured in the filter ash, 8% is emitted with the flue gas, 6% is captured in the scrubber (filter cakes) and virtually nothing is emitted via the drain water. In this study 37% of the cadmium was evaporated from the combustion process and entered the flue gas cleaning system. Several other studies on the evaporation of cadmium

from MSW incinerators have been done and they have shown quite different results. The Brunner and Mönch[4] investigation stated that only 12% of the cadmium remained in the bottom ash, while the Burnaby project[3] could report an even lower content of cadmium in the bottom ash (3.7%). A Swedish investigation[15] of four different MSW incinerators found large variations in cadmium remaining in the bottom ash, varying from 13 to 83%. These large variations can originate from two different important parameters, namely combustion temperature and chlorine content in the waste. Cadmium has a boiling point which is close to the combustion temperature and the content of cadmium in the bottom ash will be dependent on the operating furnace temperature. The operating furnace temperature is a parameter which is strongly dependent on furnace construction, MSW composition and fuel/air ratio. Of these parameters the fuel/air ratio is the easiest to control. Two important factors in the evaporation of cadmium are: chlorine available to form cadmium-chloride and the chemical form of cadmium. With chlorine present, cadmium can form relatively volatile chlorides that will follow the hot flue gases[16]. When the temperature drops, volatile cadmium adsorbs on the relatively small particles which have the largest surface area. The emission of cadmium will therefore be highly dependent on the efficiency of the flue gas cleaning system and it's ability to capture particle emissions.

Table 2 shows the heavy metal balance, the estimated heavy metal content in input MSW and concentrations of heavy metals for the different residues and the flue gas. The total yearly input of cadmium for the incinerator is 434 kg and 33 kg is leaving with the flue gas. The total efficiency of the flue gas cleaning system regarding cadmium is 80% (63% in ESP and 17% in scrubber) in this study. The Burnaby study[3] which had a flue gas cleaning system consisting of a conditioning tower, reactor with lime injection and fabric filter showed an efficiency of 99.8% for cadmium. The Brunner and Mönch[4] study which only had a electrostatic precipitator as flue gas cleaning system had an efficiency factor of 86% for cadmium.

Of the heavy metals in this investigation, cadmium is the one with the highest portion emitted through the stack. The calculated cadmium concentration in MSW was 5 mg/kg. The calculated cadmium concentration in MSW in the Burnaby project[3] was 13.5 mg/kg while Brunner and Mönch[4] reported 8.7 mg/kg. The city of Trondheim is currently implementing a source separation system where environmentally harmful waste such as electrical and electronic waste is separated and sent to a special landfill site for hazardous waste. A study has shown that electrical and electronic waste contributes with a large fraction of cadmium in MSW[17]. The effect of separating this fraction from the input MSW will be followed with great interest.

Chromium (Cr). Chromium with a boiling point of 2672°C can be regarded as a non-volatile heavy metal from a MSW combustion point of view. This is also shown by the results of the heavy metal balance for the combustion plant (table 2). Of the total input of 1829 kg/year to the incinerator, nothing is emitted through the stack. Approximately 2% is captured in the flue gas cleaning system the rest is left in the bottom ash. The Burnaby study reported that 93% of the chromium was left in the bottom ash. Only 0.06% was emitted with the flue gas, the rest was captured in the flue gas cleaning system. Calculated concentration of chromium in MSW is 21.1 mg/kg. The Burnaby project[3] calculated the chromium concentration in MSW to 92.5 mg/kg.

Iron (Fe). Like chromium iron can be regarded as a non-volatile heavy metal in MSW combustion, with a boiling point of 2750°C. From the input of 764954 kg/year virtually nothing is emitted through stack or drain water, 0.4% is captured in the flue gas cleaning system while the rest is found in the bottom ash.

Brunner and Mönch[4] found in their study 99% of the iron in the bottom ash and 1% in the filter ash, while 0.02% were stack emissions. Calculated iron concentration in MSW in this study was 8823 mg/kg.

Mercury (Hg). Mercury is the most volatile of the heavy metals with a boiling point of 357°C. The volatile behaviour of mercury was confirmed in this study with only 6% remaining in the bottom ash. Mercury content in the bottom ash between 0.2-19% has been found in other studies[3,4,15]. Mercury differs from the other heavy metals by the fact that it is in gas-phase in a combustion plant[16]. Mercury is converted to a gaseous metal and gaseous chloride salt in the combustion process. The flue gas cleaning system had an efficiency of 95% for mercury. The wet scrubber captured 92% of the volatile mercury. Brunner/ Mönch[4] had a mercury removal efficiency of 25% with electrostatic precipitator, while the Burnaby study[3] had an efficiency of 40% where practically all captured mercury was found in the fabric filter. The calculated concentration of mercury in MSW in this study was 1.6 mg/kg, which is exactly the same concentration as a similar mercury balance study gave in 1994[18]. Other studies have indicated mercury concentrations in the range of 0.7 - 1.5 mg/kg[3,4,19]. Given the volatile nature of mercury, emphasis should be put into removing mercury containing waste from the input MSW in order to reduce mercury emission to the environment. A study has shown that electrical and electronic waste contributes with a large fraction of mercury in MSW[17]. The potential for reduction of mercury entering the incinerator by removing the electrical and electronic waste fraction should therefore be considerable.

Lead (Pb). Lead, with a boiling point of 1750°C, should in combustion of MSW normally be considered as a non-volatile metal. However, studies have shown that small portions of chlorine in the waste will decrease the volatility temperature with several hundred degrees[20]. From the total of 38061 kg/year input of lead to the incinerator, 94% is captured in the bottom ash, 5% is captured in the flue gas cleaning system and 0.2% is stack emission. Other studies have shown a large variation of lead in the bottom ash ranging from 58 to 89%[3,4,15]. This and other studies have shown that most of the volatile part of lead is captured on ash particles and cleaned in the filter units. The calculated concentration of lead in MSW was 439 mg/kg. Other studies have reported values in the range of 160 to 430 mg/kg[3,4,19]. As for cadmium and mercury, lead is also strongly represented in the older electronic and electrical waste fraction although not relatively as much as mercury and cadmium. The combination lead and PVC (resin PVC contains over 50% of chlorine[21]) is highly possible for the electronic and electrical waste fraction, giving an increase in the volatile release of lead in the combustion process.

Zinc (Zn). Zinc, with a boiling point of 907°C, is one of the more volatile metals in this investigation. Zinc is also the heavy metal with largest yearly stack emissions (76 kg/year). Most of the zinc was found in the bottom ash (86%), while most of the residual 14% is captured in the flue gas cleaning system. Only 0.1% is emitted through the stack. This study differ from other studies with a large portion of the zinc remaining in the bottom ash. Other studies[3,4] have reported 42 and 51% of zinc in the bottom ash, but like this study most of the volatile fraction of zinc is captured in the ESP. The large variation in volatility can be ascribed to differences in combustion temperature due to the relatively low boiling point which is in the area of the combustion temperature. The calculated concentration of zinc in MSW was 1044 mg/kg while others have reported 1873 and 2000 mg/kg[3,4].

CONCLUSIONS

In this study most of the cadmium (63%), zinc (86%), lead (94%), chromium (98%) and iron (100%) was found in the bottom ash. The filter ash captured most of the volatile cadmium (24%), zinc (13%), lead (5%) and chromium (2%). The wet scrubber captured most of the mercury (87%). From these results it is evident that a large portion of the heavy metals are captured in the bottom ash compared to other studies, indicating a relatively low combustion temperature. The calculated concentrations of heavy metal in the input MSW was 1.6 mg/kg for mercury, 5 mg/kg for cadmium, 1044 mg/kg for zinc, 439 mg/kg for lead, 21.1 mg/kg for chromium and 8823 mg/kg of iron. Long term monitoring of the flue gas and residues seem to be a better and easier way to measure the content of heavy metals in MSW rather than sampling the input waste. Reduction in heavy metal emissions is expected after implementation of source separation of environmental harmful waste including electronic and electrical waste in 1997. The potential for reduction of mercury is confirmed by the relatively high concentration in MSW compared to other studies. The continuing work on identifying and separation of MSW fractions with large concentrations of heavy metals seem to be an important method to reduce heavy metal emissions further. Improved flue gas cleaning systems, furnace constructions and combustion control will also contribute to reduced emissions. This work has contributed to a better understanding of the behaviour of heavy metals in MSW combustion and given complementary understanding of the efficiency of various flue gas cleaning systems for different heavy metals.

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Table 1. Average composition of MSW delivered to HHC[9].

COMPONENT	FRACTION [wt%]
Paper	30
Food wastes	17
Plastic, textiles,rubber, leather	6
Wood	8
Glass	7
Metal	12
Other combustibles	6
Other non-combustibles	14
Total	**100**

Table 2. Concentrations in residues, flue gas and waste and heavy metal balance.

	Cd	Cr	Fe	Hg	Pb	Zn
Concentration in residues, flue gas and waste						
Flue gas [mg/Nm^3 dry]@11%O_2	0.057	0.00075	0	0.013	0.162	0.159
Bottom ash [mg/kg dry]	19	125	53250	0.62	2513	5464
Filter ash [mg/kg dry]	323	127	9833	8	5833	37667
Filtercake [mg/kg dry]	1331	10	2600	5805	7027	8567
Drain water [Tg/m^3]	1.3	0.9	0	3.6	48.1	100
Input MSW [mg/kg]	5	21.1	8823	1.6	439	1044
Heavy metal balance for the combustion plant						
Flue gas [kg/year]	32.7	0.44	0	7.2	91.8	76.3
Bottom ash [kg/year]	272	1788	761768	9	35950	78165
Filter ash [kg/year]	104	41	3154	3	1871	12083
Filtercakes [kg/year]	27.7	0.2	54.1	120.8	146.2	178.3
Drain water [kg/year]	0.02	0.02	0	0.06	0.83	1.7
Input MSW [kg/year]	434	1829	764954	139	38061	90515

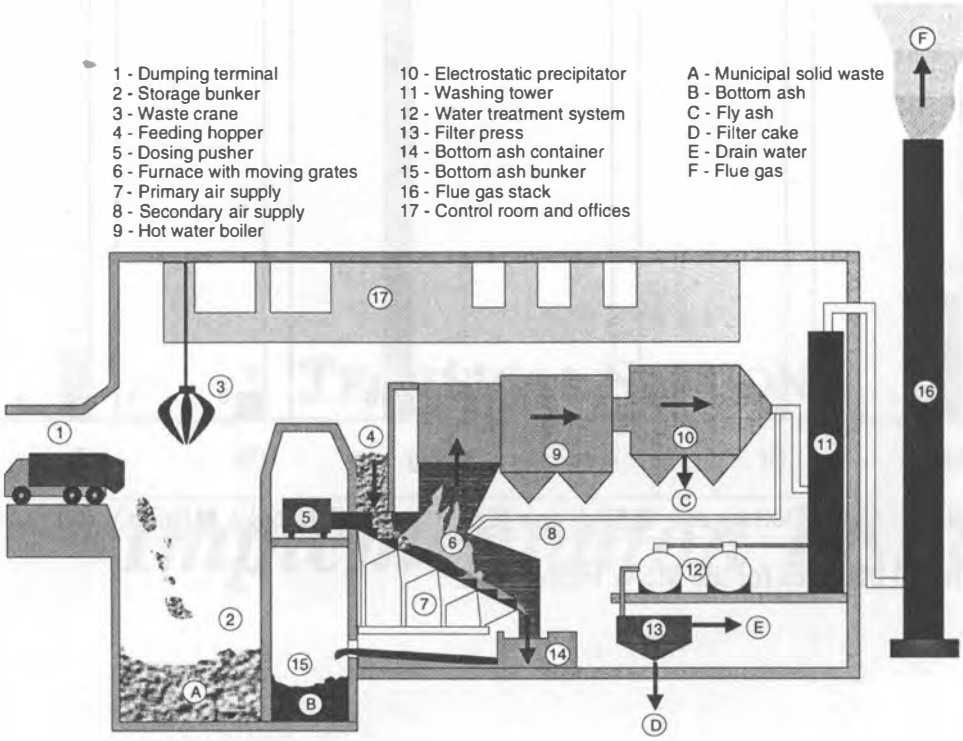

Figure 1. Sketch of the waste incinerator with major parts and mass fluxes indicated.

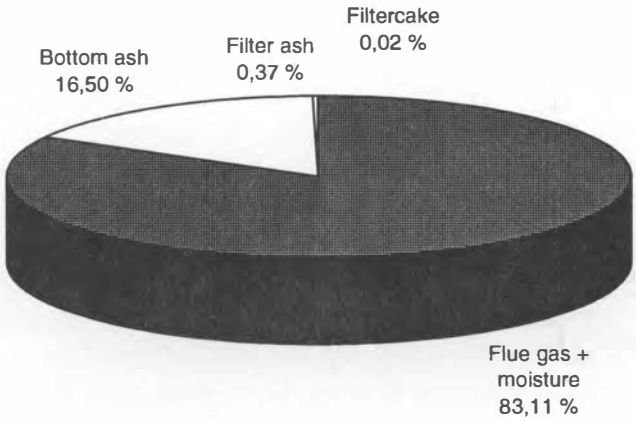

Figure 2. Overall mass balance for the MSW incinerator (dry basis).

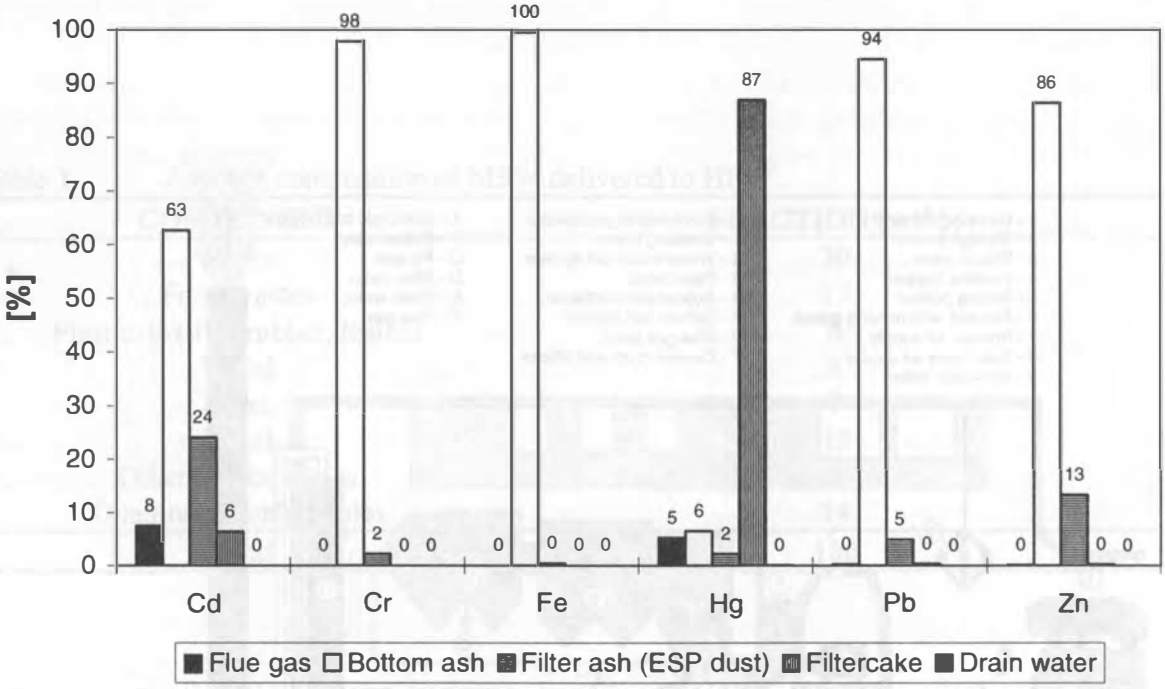

Figure 3. Heavy metal balances for the MSW incinerator.

EMISSIONS OF HEAVY METAL AND PAH COMPOUNDS FROM MUNICIPAL SOLID WASTE INCINERATORS: CONTROL TECHNOLOGY AND HEALTH EFFECTS

Report on a WHO meeting

Florence
12–16 October 1987

WORLD HEALTH ORGANIZATION
Regional Office for Europe
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FOREWORD

In recent years, the incineration of municipal solid waste and the resultant emission of toxic substances have generated worldwide concern about possible adverse health effects. Such concern in Italy prompted two meetings organized by the WHO Regional Office for Europe at the request of Italian regional authorities in 1986 and 1987. The first meeting focused on the emissions of dioxins and furans and the second meeting on the emissions of PAH and heavy metal compounds.

This document reports on the deliberations, conclusions, and recommendations of the second meeting held in October 1987 in Florence to discuss control technology for and health effects from incinerator emissions of heavy metal and PAH compounds. The meeting was initiated and financed by the Regional Government of Tuscany, in collaboration with the Government of Italy.

This report is aimed at the public and environmental health authorities of the Tuscany Region, but it should be equally useful to such authorities elsewhere as well as to professonals in this area.

Michael J. Suess
Regional Officer for Environmental
Health Hazards

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INTRODUCTION

Emissions consisting mainly of dust, smoke, heavy metal compounds, and noxious gases from municipal solid waste (MSW) incinerators, sewage sludge incinerators, hazardous waste incinerators, heating facilities, and thermal power plants have long aroused concern. More recently, the potential danger of hazardous organic compounds that may be present in emissions, such as polycyclic aromatic hydrocarbons (PAH), has been recognized. The high toxicity of some of these compounds has called into question the safety of incinerating MSW. Answering these questions requires a better understanding of the emission levels and potential adverse public health effects of heavy metal and organic compounds, such as PAH, emitted from MSW incinerators.

The WHO Regional Office for Europe organized a Working Group at the request of the Regional Government of Tuscany, in collaboration with the Government of Italy. The purpose of the meeting was to review and discuss data and public health aspects related to the emission of heavy metal and PAH compounds during incineration of MSW. The Working Group's objective was to give national and local health authorities, as well as other interested parties, guidance on the public health significance of these emissions and, when possible or necessary, to point out practicable approaches to minimize the emissions and the related public health impacts. The meeting was attended by 26 participants from 13 countries (Annex 2).

Dr M. Piscator was elected Chairman, Mr A. Finkelstein and Dr K. Kostial were elected Vice-Chairmen, and Mr C.O. Velzy and Dr A. Wadge each served as Co-Rapporteur. Dr M.J. Suess acted as Scientific Secretary.

This meeting is the second on municipal waste incinerators organized by the WHO Regional Office for Europe in Italy. A working group on dioxin and furan emissions met in Naples in 1986. The report on the first meeting is available from the WHO Regional Office for Europe [1].

The Working Group focused on the following major items:

- influence of the composition of the material being incinerated, the conditions of the incineration process (temperature, mixing, retention time of combustion products, oxygen, and the operational aspects), and pollution control technology and its operation on the quality and quantity of emissions;

- environmental pathways of emitted heavy metal and PAH compounds as related to human exposure;

- relative importance of MSW incineration compared with other sources of human exposure to heavy metal and PAH compounds;

- public health significance of emissions of heavy metal and PAH compounds during MSW incineration.

The Working Group broke into two subgroups (Annex 1). Subgroup 1 reviewed available data on emissions of heavy metal and PAH compounds from MSW incinerators. This subgroup also considered and evaluated aspects of design and operation to minimize incinerator emissions. The results of these analyses and considerations were organized into tables of expected emission concentrations from incinerators of different types and with different operational characteristics. Much of the descriptive text material on waste and incineration in this report was extracted from the WHO Naples meeting report [1].

Subgroup 2 summarized the potential human health aspects of exposure to cadmium, mercury, lead, and PAH. It then evaluated the relative contribution that emissions from incinerators may make to the overall exposure of the general population to these pollutants. As a starting point, subgroup 2 compared estimates of ground level concentrations of emissions from MSW incinerators, developed from measured stack emissions of these pollutants, with observed ambient concentrations when available. Pollutant emissions were also compared with inventories of regional or national emissions when available. These comparisons indicated different degrees of importance for each pollutant of concern.

WASTE AND INCINERATION

MSW is a diverse and heterogeneous material generally containing domestic and commercial waste. Domestic waste is solid waste produced in the normal course of household activity or waste of a similar character from institutions. Commercial waste is waste from shops and offices, usually collected with domestic waste. MSW can vary widely in physical composition (e.g. form, size, and consistency) and chemical composition (e.g. carbon, hydrogen, oxygen, nitrogen, chlorine, sulfur, metals, pesticides, plastics, and halogenated hydrocarbons). Industrial waste can introduce substantial quantities of contaminants into the solid waste stream that would otherwise exist only in trace amounts in MSW collected from residential and commercial areas. Only domestic and commercial solid wastes are considered in this report.

A number of options exist for the processing and disposal of MSW. In the past, some countries disposed of these materials at sea. This method is now generally considered to be unsatisfactory because of obvious adverse effects on, for example, marine life, and less

obvious effects, such as the introduction of contaminants into the food chain. Alternate means of MSW disposal include deposition in carefully designed, constructed, and operated landfills, composting or separating component materials for recycling or reuse, or incineration. All of these disposal options produce residuals that must ultimately be disposed of in properly managed landfills. In this report, only incineration is considered from a detailed processing standpoint.

Incineration is defined in the WHO Glossary on solid waste [2] as "the process of reducing combustible waste to an inert residue by high temperature burning". This process has several outputs, such as solid, liquid, and gaseous emissions, and heat energy. The primary function of an incinerator is to reduce waste to an inert residue with minimum adverse impact on the environment. Thermal efficiency, while important, is secondary to this function. MSW incinerators are usually classified according to their capacity, the nature of the waste to be burned, the type of grate system used, and the method of waste heat extraction, flue gas cooling, and gas cleaning. The capacity of an incinerator in terms of quantity of waste processed per unit of time is roughly inversely proportional to the heat value of the waste.

Modern incineration plants reflect significant advances made in solving the technical and practical difficulties of material handling and combustion control. In the early days of waste incineration when air pollution regulations were undemanding or nonexistent, relatively simple, fixed-grate plants operating on a single- or two-shift basis were common. With increasingly stringent regulations to control air pollution, more complex plants requiring continuous operation have become the norm. Frequently, regulations and/or requirements make small plants prohibitively expensive to operate.

In this report, the removal efficiencies and emission test results from different installations are presented in terms of "traditional" and "advanced" incinerators. Traditional incinerators include all facilities incorporating normal combustion controls plus dry particulate collection devices with an efficiency of approximately 95 to 97%. Advanced plants include facilities incorporating modern combustion control systems and acid gas control scrubbing equipment with a particulate control efficiency of 99%+.

Types and characteristics of waste

Two types of waste are considered in this report: as-collected or as-received MSW and processed MSW.

As-received MSW

Some typical compositions of as-collected MSW can be calculated (Table 1). In western Europe, waste composition varies over a wide

range. A typical analysis of MSW samples in western Europe and United States is shown in Table 2.

Recent tests on MSW at two plants in the United States conducted by the US Bureau of Standards [5] have shown average chlorine contents ranging from 0.45 to 0.89% by mass of the total dry as-received waste. Of the total chlorine, approximately 0.14 to 0.46% by mass was from plastic materials, while approximately 0.31 to 0.43% by mass was from all other material. Thus, the variation in chlorine from plastics was about 2.3 times the minimum level at 0.14% by mass, while the variation in chlorine from all other material was about 0.4 times the minimum level at 0.31% by mass. Therefore, the chlorine content in MSW probably depends greatly on the total plastic content.

Processed MSW

Waste processing or separation technologies can affect the chemical composition of MSW, which can vary widely and include or-

Table 1. *Average composition of as-collected MSW*

Type of waste	Western Europe		USA	Middle East city
	%[a]	Range in %[b]	%[a]	%[a]
Organic	21.3	15-40	22.6	62.3
Paper	27.4	8-43	45.6	25.3
Textiles	3.5	1- 6	4.5	1.4
Plastics	3.1	3- 7	2.6	5.8
Glass	9.5	5-11	6.2	1.0
Metals	8.5	3- 8	9.1	2.8
Ash dust	19.8	2-60	7.6	NA
Miscellaneous	6.9	4	1.8	1.4
Total	100.0		100.0	100.0

[a] [3].

[b] Personal communication from experts at the WHO meeting on PCDD and PCDF Emissions from Incinerators for Municipal Sewage Sludge and Solid Waste - Evaluation of Human Exposure, Naples, 17-21 March 1986.

NA = Not available.

Table 2. *Analysis of MSW*

Type of analysis	Weight by %[a]	
	Western Europe[b]	United States[c]
Proximate analysis		
Combustible	42.1	50.3
Water	31.0	25.2
Ash and inert material	26.9	24.5
Total	100.0	100.0
Ultimate (elemental) analysis of combustibles		
Carbon	51.1	50.9
Hydrogen	7.1	6.8
Oxygen	40.1	40.3
Nitrogen	1.2	1.0
Sulfur	0.5	0.4
Chlorine	NA	0.6
Total	100.0	100.0

[a] Gross heat value, as fired = 9000 kJ/kg.
[b] [3].
[c] [4]; Domalski, E.S. et al., paper given at the 1986 National Waste Processing Conference.
NA = Not available.

ganic and inorganic compounds. MSW can contain the following inorganic compounds: cadmium, as a pigment, a stabilizer in plastics or a surface coating, or in alloys and batteries; lead, in batteries, lead-based paints, various metal items, and dusts; mercury, in batteries, and in electrical and control equipment, and organomercuric compounds in paper manufacturing; beryllium, in fluorescent light bulbs; and inorganic metallic salts, in paper manufacturing [6]. Some organic compounds can be potentially troublesome: hexane, toluene, and xylene in oil and acrylic paints, turpentines and other solvents, and lithographic inks; methylene chloride, in paint strippers and propellents for spray paint aerosol cans; formal-

dehyde, in particle board and plywood; dioxins, a contaminant in pentachlorphenol wood preservatives; and various organic compounds, in paper manufacturing.

Some of these substances, such as lead, mercury, and cadmium, can contaminate the environment, regardless of the MSW disposal method. Therefore, when developing a management plan for the disposal of MSW, consideration should be given to collecting separately some material containing these compounds, such as batteries, and disposing of this material under tightly controlled conditions. In other cases, consideration should be given to substituting less hazardous materials. This approach can best be achieved at a national or international level.

Trace organic compounds are widely distributed in the total mass of MSW, and most of them can be destroyed in a properly designed and operated waste combustion unit. However, some types of waste processing, such as composting or materials recovery (e.g. reclamation of paper fibre) may not destroy these substances [6].

Many countries have voluntary programmes of source separation, including Italy, the United Kingdom, and the USA. Such programmes have not had a significant impact on the composition of MSW. In areas where bottles are returned and reused, as much as 50% of the glass may be recycled [7]. However, as glass usually represents 10% or less of MSW, the combustion process will be little affected. Paper collection at the source is usually restricted to newspapers and magazines. In countries with advanced separate collection systems, up to 45% of the paper consumed may be recycled. This reduction may considerably lower the heat value of MSW.

Several programmes have been initiated to remove certain potentially toxic substances from the solid waste stream prior to incineration. In Zurich, Switzerland, mercury emissions from incinerators were essentially the same before and after implementation of a public education programme for removal of mercury-containing batteries from the waste stream [8]. Sweden has a similar programme for removal of batteries at source. While the effectiveness of the programme to reduce incinerator emissions of mercury has not been fully evaluated, results from the 2nd and 3rd quarters of 1987 (when the programme started) indicate that approximately 50% of mercury and 20% of cadmium in batteries were collected. The goal is to recover 75% of the mercury in batteries (K. Hasselgren, Swedish Association of Public Cleansing and Solid Waste Management, personal communication, 1988).

Information is limited on the impact on emissions of removing certain waste constituents from burning refuse-derived fuel (RDF). Few plants use waste processing or material separation technologies. In Italy, a very small proportion of plants (approximately 10%) use mechanical processing for sorting metals, plastics, glass, or organic matter (the last processed by composting for agricultural use). Canada has only one mechanical processing system in oper-

ation; it uses shredding and magnetic metal separation. In the USA, many RDF plants have been shut down [9].

Lorber [10] describes a testing programme investigating emissions from combustion of as-received MSW and RDF. Emission tests indicate that the concentration of hydrogen chloride, cadmium, and lead in the untreated flue gas from the RDF combustion unit was in the same range as that found in the flue gas of an MSW incinerator before electrostatic precipitation and scrubbing [11]. These pollutants were removed when the RDF was burned in a cement kiln where the pollutants were fixed in the cement clinker.

In the United Kingdom, experiments are using mechanical sorting in a rotary drum to produce a "cleaner" product to be used in RDF production [12]. Preliminary processing results (Table 3) produced 160 tonnes of product (>40 mm) and 110 tonnes of reject material (<40 mm). Most of the glass, approximately half of the putrescible material, and about one third of the nonferrous metal were removed. A significant portion of the lead and mercury (about two thirds and one half, respectively) from nonmetallic sources was discharged to the rejects (Table 4). Rejection of heavy metals contained within metallic items can only be inferred from the category data (Table 3); a magnetic separation stage would be helpful in this respect. Emissions have not been analysed for heavy metal compounds.

However, considering Lorber's results, one would expect to see a major difference in heavy metal content in the stack emissions. Boiler efficiency was calculated to increase from 68.3% when burning raw waste to 73.2% when burning RDF. To the extent to which this increase in boiler efficiency indicates improved combustion conditions, it may also indicate lower emissions of organics, but no test results are available to evaluate this theory.

Thus, careful consideration should be given to determining, in detail, the trace chemical content of solid wastes before choosing an overall solid waste management strategy.

MSW incineration technology

Waste combustion is a complex process. MSW is heated by contact with hot combustion gases or preheated air and by radiation from the furnace walls. The temperature range for drying is 50 to 150 °C. At higher temperatures, volatile matter is formed by thermal decomposition reactions. This volatile matter is generally combustible and, after ignition, produces flames. The remaining material is further degassed and burns much more slowly.

The combustion of MSW proceeds without supplementary fuel when the heat value is at least 5000 kJ/kg. This occurs when the waste is within the following general compositional limits: ash content, <60%; moisture content, <50%; and combustible matter, >25%.

The complexity of the combustion of solids stems from the nature of the decomposition and burning reactions and their

Table 3. *Analysis of RDF* [12]

Material	Feed	Reject (>40 mm)	Product (<40 mm)
	(% by weight)		
Ferrous metals	6.4	41.0	99.0
Nonferrous metals	0.9	24.0	76.0
Glass	6.4	79.0	21.0
Putrescible material	20.8	30.0	50.0
Textiles	2.6	2.0	98.0
Paper and board	24.3	4.0	96.0
Plastics	7.3	5.0	95.0
Miscellaneous			
Combustible	4.6	5.0	95.0
Noncombustible	4.0	73.0	27.0
Fines (<10 mm)	22.7	92.0	8.0
Total	100.0	41.0	59.0

association with heat transfer, air flow, and diffusion. In most incinerators, combustion takes place while the solids are supported on and conveyed by a grate. Since the early 1960s, most MSW incinerators have incorporated one of a number of proprietary grate systems that allow a continuous feed of unscreened waste into and movement through furnaces with integral boiler facilities. The grate performs several functions: providing support for the burning bed of refuse; admitting underfire air through openings in the grate surface; transporting the solid waste from feed mechanism to ash quench; agitating the bed; and redistributing the burning mass.

Combustion air is introduced as primary air and secondary air. Primary air passes through the grate and the fuel bed and assists in burning the fuel and cooling the grate. Secondary air is blown into the furnace through nozzles above the fuel bed to promote mixing and complete combustion of the burning gases. In some plants, tertiary air is added to control the temperature of the flue gases to required levels before discharge from the combustion area.

During combustion, the heat value of the fuel is converted into high-temperature flue gas and ashes, and into

Table 4. *RDF product analysis* [12]

Analysis	Feed	Reject (>40 mm)	Product (<40 mm)	% of total Reject	% of total Product
Gross calorific value (MJ/kg)	9.2	6.62	11.0	29.6	70.4
Net calorific value (MJ/kg)	7.91	5.57	9.53	28.9	71.1
Moisture content (% by weight)	31.4	32.1	31.0	41.7	58.3
Ash content (% by weight)	30.5	42.8	21.9	57.6	42.4
Volatiles (% by weight)	31.6	14.4	43.7	18.4	81.6
C (% by weight)	28.8	28.7	29.0	40.8	59.2
H_2 (% by weight)	2.52	1.04	3.55	17.0	83.0
O_2 (% by weight)	12.4	4.9	17.6	16.3	83.7
N (% by weight)	0.71	0.65	0.74	38.0	62.0
S (% by weight)	0.28	0.43	0.18	62.0	38.0
Cl (% by weight)	0.35	0.14	0.49	17.4	82.6
Pb (% by weight)	0.011	0.019	0.006	57.8	32.2
Cd (% by weight)	0.0012	0.0004	0.0017	16.4	83.6
Hg (% by weight)[a]	0.00004	0.00003	0.00004	41.0	59.0

[a] Identifiable metal pieces were removed from samples prior to chemical analysis. Hence metal content and distribution refer only to nonmetallic sources.

heat losses through the furnace walls. With an estimate of heat losses and the unburned carbon in the residue, the theoretical combustion temperature can be calculated. This temperature may be controlled by introducing varying quantities of excess air into the furnace.

When waste with a low heat value is incinerated or when high combustion temperatures are desirable, heat losses may be kept to a minimum by adequate insulation of the furnace. The combustion temperature may also be increased by preheating the combustion air or by predrying or heating the waste products.

Control of incinerator emissions

Gas cleaning is required to reduce the level of pollutants in

the raw flue gases (Table 5). Electrostatic precipitators (ESP) are the most commonly used gas cleaning device for particulate emission control in municipal incinerators as they can be designed to achieve very high collection efficiencies (99% or higher) and their operating costs are moderate. However, the investment cost and volumetric space required are high.

Fabric filters can operate at high efficiency even in the submicron particle size range. Until recently, they were not widely used in solid waste incineration plants because of their relatively high investment and operating costs, and limited life at high temperatures. However, with the increasing emphasis of regulatory agencies on acid gas control and an apparent trend to lower particulate emission levels, baghouses, usually in conjunction with addition of lime, and lower flue gas temperatures are being used more widely with municipal waste incinerators in the USA.

Wet collectors were initially used to control acid gas. Various semiwet and dry methods of cleaning flue gases have also been used. These methods of gas treatment, based on the injection of slurried or powdered lime, limestone, or dolomite, eliminate a large fraction of the acid gases from the flue gas by adsorption and absorption, followed by chemical conversion. Because the reactivity of these lime materials is rather low, a multiple of the stoichiometric quantity is normally required to obtain a satisfactory cleaning effect. High removal efficiencies can be achieved for hydrogen chloride, but reduction of sulfur dioxide and sulfur trioxide is

Table 5. *Typical composition of raw flue gases (wet basis, NTP*[a]*)*[3]

Substance	Amount
H_2O	10 - 18 % by volume
CO_2	6 - 12 % by volume
O_2	7 - 14 % by volume
CO	< 0.1 % by volume
Dust	2 - 15 g/m^3
Cl^-	250 - 2000 mg/m^3 (as HCl)
F^-	0.5 - 5 mg/m^3 (as HF)
$SO_2 + SO_3$	100 - 1000 mg/m^3 (mainly SO_2)
$NO + NO_2$	100 - 400 mg/m^3 (largely as NO)

[a] NTP = normal temperature and pressure.

only partial. However, slaked lime is highly reactive and stoichiometric ratios of 1.2 to 1.7 have been used for 97-99% of hydrogen chloride removal and 60-90% reduction of sulfur dioxide, depending on operating conditions and particulate collector. Lime injection into a scrubber/fabric filter system has removal efficiencies of 90-99% for hydrogen chloride and 70-90% for sulfur dioxide, provided that the flue gas temperature, certain equipment design features, and stoichiometric ratio for lime addition are suitable.

Test results for MSW incinerators equipped with the conventional two-field ESP have shown a wide range of particulate emissions, varying from 50 to 300 mg/Nm^3 (0.07 to 0.13 grains per dry standard cubic foot (gr/dscf)). The three- and four-field ESP (see Table 6) can achieve emissions of 20 to 75 mg/Nm^3 (0.009 to 0.03 gr/dscf). An emission level below 20 mg/Nm^3 (0.009 gr/dscf) is technically possible and has been demonstrated at Zurich (J.T. Möller, personal communication, 1986) where lime addition was retrofitted on a plant with an existing two-field ESP.

The scrubber/fabric filter control systems have also been shown (see Table 6) to be capable of operating at a particulate emission level of 20 mg/Nm^3 (0.009 gr/dscf) and lower. The material selected for the filter bags can have an important effect on filtering efficiency and the emission level thus achieved. In general, test results for the scrubber/fabric filter indicate lower particulate emissions than those for ESP on MSW incinerators, though ESP have not normally been designed to meet emission levels as low as those specified for fabric filter installations. In Europe, ESP following spray drying absorbers have been tested at particulate emission levels of 1 to 8 mg/Nm^3 (0.00045-0.0036 gr/dscf). The reliability and overall economics of the various control processes must be considered when making a selection of equipment to meet these very low emission control requirements.

Because the condensation point for most metals, such as compounds of lead, cadmium, chromium, and zinc, is above 300 °C, the removal efficiency of such metals depends highly on the removal efficiency of the particulate control system used. Some heavy metal compounds, particularly chlorides, have condensation points below 300 °C. For such compounds, particulate collection temperatures will be a factor in collection efficiency. High removal (over 99%) has been observed for most metals for highly efficient (over 99%) particulate removal systems operating at appropriate temperatures [13].

Traditional incineration facilities have relatively inefficient particulate control equipment and emissions of metals may be relatively high in such facilities (Table 7).

Mercury, a very volatile metal, exists in vapour phase at temperatures as low as 20 to 50 °C. Several studies have suggested that both sufficient cooling of the flue gas (typically below 140 °C, based on test results) and a highly efficient particulate

removal system to remove the particles on which the mercury has been actively absorbed [14,15] (H. Vogg et al., paper presented at Fifth International Recycling Congress, 1986) are required to achieve high mercury removal. High mercury removal has also been obtained using the scrubber/fabric filter and semidry/ESP systems, provided that the flue gas is adequately cooled [13]. Properly operated wet scrubbers are also capable of considerably reducing mercury emissions (Goepfert, Reimer and Partners, unpublished report, 1986).

Data on emissions of PAH and other chlorinated organic compounds are limited. Most sampling programmes for PCDD/PCDF, which have been fairly extensive, have unfortunately neglected to analyse for these compounds. The scrubber/fabric filter technology generally demonstrated removal rates of 80-99% for these compounds [13].

Impact of incinerator operation on atmospheric emissions

An advanced incinerator is designed to achieve 99% conversion of organic matter to carbon dioxide [16]. This objective is achieved by providing adequate residence time, postcombustion temperature, and turbulent mixing. When this objective is achieved, the concentration

Table 6. *Particulate emissions from MSW incinerators* [1]

Plant	Dry filterable material (at 12% CO_2) gr/dscf	mg/Nm^3
G (1983); ESP	0.0321	72
T (1984); DS,FF	0.0120	27
M (1984); DS,EP	0.0104	23
W (1985); DS,FF	0.0040	9
P (1985); ESP	0.0163	37
T (1986); ESP	0.0070	16
M (1986); DS,FF	0.0070	16

[1] Unpublished reports: Cooper Engineers, 1984, 1985; Hahn et al., 1986; Velzy, 1986, 1987).
DS = Dry or semidry scrubber (lime addition).
ESP = Electrostatic precipitator.
FF = Fabric filter (bag house).
gr/dscf = Grains per dry standard cubic foot.

level of carbon monoxide remains consistently below 0.1% by volume. Generally, levels of 125 mg/Nm^3 (100 ppm) and less are achievable in advanced design plants with modern combustion control systems.

Combustion may be incomplete when the incinerator is improperly operated in the following ways:

- inadequate mixing of the feed material so that large quantities of material with extremely high or low heat value enter the furnace;

Table 7. *Removal efficiencies for certain metals from flue gases from MSW incinerators*

Equipment	Particulate-bound (Cd, Pb)	PAH	Vapour-bound (Hg)
	Removal efficiency (%)		
Traditional			
High eff. ESP[1]	Cd 95-99+ Pb 95-99+		NA 20%
Advanced			
1. Scrubber + ESP[2] (Semidry)	Cd 98-99++ Pb 98-99++		80-90% (at 120-140 °C)
2. Scrubber + FF[3] (Dry)	Cd 99++ Pb 99++	99 99	85-95% (at 140 °C) Much uncertainty
3. Wet[4]	Cd 99+ Pb 99+		70[a]-90%[b]

ESP = Electrostatic precipitator. FF = Fabric filter (baghouse).
NA = Not applicable.
[1] Data from measurements in Europe and North America.
[2] J.T. Möller, personal communication, 1986 and 1987.
[3] R. Bergström, personal communication, 1986.
[4] A. Nottrodt, personal communication.
[a] Dry endproduct 60-75%
[b] Liquid effluent 80-95%

- operation at excessively low temperatures (some regulatory agencies require the use of auxiliary burners when the temperature at the exit from the combustion chamber drops below 800 °C);

- quenching of the flames caused by improper distribution of secondary air;

- gross overloading of the plant.

Incomplete combustion can be detected by monitoring the flue gas composition. Conventional combustion gas measurements include carbon monoxide, total hydrocarbons, and oxygen or carbon dioxide. Both carbon monoxide and total hydrocarbons have been of interest as potential surrogates to indicate incomplete combustion or periods of poor combustion. Under these poor combustion conditions, carbon monoxide levels of several thousand mg/Nm^3 have been observed and total hydrocarbon levels have risen from a typical 1.25 to 6.25 mg/Nm^3 (1-5 ppm) to 125 mg/Nm^3 (100 ppm) and more.

Poor combustion conditions in incinerators can lead to local oxygen-deficient conditions, resulting in the emission of hazardous organic compounds. PAH, for example, are formed during fuel-rich combustion as a consequence of free radical reactions in the high-temperature flame zone. In addition, in the presence of water-cooled surfaces, such as found in oil-fired home-heating furnaces, a high fraction of the polycyclic compounds is oxygenated. Similar free radical reactions probably take place in fuel-rich zones of incinerator flames, yielding PAH, oxygenated compounds (e.g. phenols), and, in the presence of chlorine, some PCDD and PCDF.

The amount of heavy metal compounds, other than those with low vaporization such as mercury, discharged from an MSW incinerator is largely a function of the amount of fly ash discharged. The amount of ash carried with the flue gases from a burning refuse bed increases with increasing underfire air rate and with bed agitation. For this reason, starved or controlled air incinerators with low underfire air flow rates tend to have lower particulate emissions from the furnace than conventional mass-burn units. In addition, the particle size of the inorganic content of the MSW will influence the amount of ash carried from the combustion chamber.

High-efficiency particulate control devices can effectively reduce the atmospheric emission of most heavy metal compounds. As the efficiency of particulate control improves, metal emissions generally decrease and, in most cases, decrease significantly. Even though ample evidence from tests [17,18] (W. Dannecker, personal communication, 1983) indicates that heavy metals tend to concentrate on the finer particles, evidence also suggests that more than 99% metal removal from flue gases can be achieved by using high efficiency (99%) particulate control devices (Cooper Engineers, un-

published report, 1985; Hahn et al., paper given at National Waste Processing Conference, 1986).

DISCHARGE OF METALS AND PAH FROM INCINERATORS

Organic materials composed entirely of carbon, hydrogen, and oxygen will produce water and carbon dioxide as the final products of complete combustion in the presence of oxygen. However, MSW includes many compounds or elements in addition to the three elements noted above. Therefore, discharges to the environment emanating from the burning of MSW include substances resulting directly from combustion and other substances present in the bed of solid waste. Discharges may be in the form of stack gas and particulate emissions, bottom ash from the furnace grates, and fly ash from the removal of particulates in the flue gases.

Metals are not produced by the incineration process but rather are changed, concentrated or redistributed among the various process effluents. This is in contrast to PAH, which are largely generated as a consequence of incomplete combustion.

Redistribution of elements during combustion

The distribution of elements between bottom ash and ash carried to the air pollution control device depends upon the design and operation of the incinerator and the composition of the feed. The distribution of elements among the different components of refuse also strongly influences the fate of the elements. For example, titanium dioxide, a pigment in paper products, has a particle size of about 0.2 um and will tend to be carried off by the flue gases passing through the refuse bed, whereas titanium dioxide in glass will end up in the bottom ash. About 10% of the inorganic content of the waste will be entrained from burning refuse beds to form fly ash particles [19]. The remainder will end up in the residue.

Volatile elements and their compounds, usually present in trace amounts in the feed, will vaporize from the refuse and condense in the cooler portions of a furnace either as an ultrafine aerosol (< 1 um in diameter) or on the surface of fly ash particles. A large fraction of highly volatile elements in the feed, such as mercury, will be volatilized and emitted to the atmosphere in vapour form unless special preventive measures are taken. Some elements may be converted to more volatile chemical forms during combustion. For instance, the relatively high chlorine content of MSW results in an adequate supply of hydrogen chloride to form substantial quantities of volatile metal chlorides [17] as well as chlorinated organic compounds.

PAH concentrations in MSW incineration residues are relatively uniform (Table 8). However, the metals increase in concentration

from bottom ash, to boiler/economizer ash, to precipitator or fly ash. Greenberg et al. [17] suggest that trace element enrichment on fine particles is primarily due to a volatilization-condensation mechanism similar to that proposed by Natusch et al. [20] for coal-fired power plants. The proportionately greater surface area of fine fly ash particles is expected to result in an enrichment of the volatile trace elements on these particles. However, Wadge et al. [18] suggests that the elevated levels of certain trace elements in incinerator fly ash may result not from a volatilization-condensation mechanism but, in part at least, from the loss of organic fraction in the furnace. This would leave an inorganic residue enriched in trace elements.

Using the information in Tables 7 and 8, the distribution of selected metals and PAH in incinerator residues can be calculated (Table 9). The concentrations of refractory elements such as iron, carbon monoxide and titanium will be higher in the bottom ash.

Atmospheric emissions and deposition

Cadmium, lead, and mercury

A review of data on cadmium, lead, and mercury indicated that emissions were significant (Tables 10-12). The emission levels or toxicities of other metals were so low that they were considered to be of minor importance.

Table 8. *Concentrations of PAH, certain metals, chlorine, and sulfur in MSW-incinerated residues*[1]

Substance	Bottom ash	Boiler/economizer ash	Precipitator ash
PAH (ug/g)	0.1-0.5	0.03-0.2	0.2-0.5
Cd (ug/g)	2.0-7.5[a]	115-350	250-1100[a]
Pb (ug/g)	750-6000[a]	5000-16000	5000-26000[a]
Hg (ug/g)	0.01-1.0[a]	1-8	5-70[a]
Cl (mg/g)	3-6[a]	ND	40-80[a]
S (mg/g)	2-4[a]	ND	20-40[a]

[1] Personal communications: P.H. Brunner, A. Nottrodt, J.T. Möller.

[a] [19,21].

ND = Not detected.

Table 9. Emission factors and distribution (% of input) for selected metals and PAH [22]

Compound	Bottom ash		Boiler ash	Fly and boiler ash		Fly ash from stack gas cleaning		
	g/t of waste	%	g/t of waste	g/t of waste	%	Traditional g/t of waste	Advanced g/t of waste	%
Cadmium	0.4-2	10-40	0.1-4	3-30	55-80	0.4-9	0.1-0.25	1-15
Lead	200-2000	55-80	5-200	50-800	20-45	0.6-300	0.06-5	1-5
Mercury	0.1-0.3	2-6	0.001-0.06	0.05-2	15-35	0.6-10	0.2-1.3	60-85
PAH[a]	20-150	45-12	0.03-2.0	1.5-14	4-1	20-1000		50-80

[a] Distribution in percentage is not based on PAH concentrations in MSW input.

Table 10. *Emission data on cadmium from stacks*

Traditional dry collection 95% +/- efficiency	Advanced acid gas control scrubber (dry or wet/dry) 99% + efficiency
Canada[a]	Canada[a]
0.6-1.0 mg/Nm^3 - 12% CO_2 SA 0.02-1.0 mg/Nm^3 - 12% CO_2 MB w/ESP	N.D.-0.6 ug/Nm^3 - 8% O_2 MB w/scrubber and FF
France[b]	France[b]
0.06-0.1 mg/Nm^3 - 7% CO_2 (wet) MB 1-1.5 mg/Nm^3 - 7% CO_2 (wet) MB	0.03 mg/Nm^3 - 7% CO_2 (wet) MB w/ESP and scrubber 0.001-0.003 mg/Nm^3 - 7% O_2 (wet) MB, DS and FF
Italy[c]	Italy[c]
0.83 mg/Nm^3 - 10% O_2 (wet) MB	0.02 mg/Nm^3 - 10% O_2 (wet) MB
Switzerland[d]	Switzerland[g]
0.1-2.0 mg/Nm^3 - 7% CO_2	0.01 mg/Nm^3 - 11% O_2
United Kingdom[e]	Federal Republic of Germany[h]
0.16-0.45 mg/Nm^3 - 10% CO_2 (dry) MB	00.2 mg/Nm^3 - 11% O_2 MB w/SDA and ESP 0.04 mg/Nm^3 - 12% O_2 MB w/wet scrubber
United States[f]	
0.2-0.5 mg/Nm^3 - 12% CO_2 (dry) MB 0.2-1.2 mg/Nm^3 - 7% O_2 (dry) SA 0.5-0.7 mg/Nm^3 - 7% O_2 (dry) RDF w/ESP	

DS = Dry scrubber. ESP = Electrostatic precipitator. FF = Fabric filter. MB = Mass burn. ND = Nondetectable. RDF = Refuse-derived fuel. SA = Starved air. SDA = Spray drier-absorber.
[a] [13,26,27]. [b] Dimitrov, paper given at Third National Seminar of the Air Quality Agency, 1987. [b] [28]. [c] [29]. [d] [30]. [e] [31]. [f] [32]. [g] J.T. Möller, personal communication, 1986. [h] J.T. Möller, personal communication, 1987.

Table 11. *Emission data on lead from stacks*

Traditional dry collection 95% +/- efficiency	Advanced acid gas control scrubber (dry or wet/dry) 99% + efficiency
Canada[a]	Canada[a]
0.6–2.5 mg/Nm^3 – 12% CO_2 MB w/ESP 8.0–15.0 mg/Nm^3 – 12% CO_2 SA	0.002–0.006 mg/Nm^3 – 8% O_2 MB w/scrubber and FF
France[b]	France[b]
34–49 mg/Nm^3 – 7% CO_2 (wet) MB 1.3–1.9 mg/Nm^3 – 7% CO_2 (wet) MB w/ESP	0.055–0.07 mg/Nm^3 – 7% CO_2 (wet) MB, DS and FF 0.78 mg/Nm^3 – 7% CO_2 (wet) MB w/ESP and scrubber
Italy[c]	Italy[c]
19 mg/Nm^3 – 10% O_2 (wet) MB	0.54 mg/Nm^3 – 10% O_2 (wet)
Switzerland[d]	Switzerland[e]
1–18 mg/Nm^3 – 7% CO_2	0.38 mg/Nm^3 – 11% O_2 MB w/SDA and ESP
United Kingdom[f]	Federal Republic of Germany[g]
2.4–10.7 mg/Nm^3 – 10% CO_2 (dry) MB	0.6 mg/Nm^3 – 12% O_2 MB w/wet scrubber; 0.01 mg/Nm^3 – 11% O_2 MB w/SDA and ESP
United States[h]	
0.15–22.0 mg/Nm^3 – 7% O_2 (dry) SA 1.5–1.8 mg/Nm^3 – 7% O_2 (dry) RDF w/ESP	

DS = Dry scrubber. FF = Fabric filter. MB = Mass burn. RDF = Refuse-derived fuel. SA = Starved air. SDA = Spray drier-absorber. [a] [13,26,27]. [b] Dimitrov, paper given at Third National Seminar of the Air Quality Agency, 1987. [b] [28]. [c] [29]. [d] [30]. [e] J.T. Möller, personal communication, 1986. [f] [31]. [g] J.T. Möller, personal communication, 1987. [h] [32].

Table 12. *Emission data on mercury from stacks*

Traditional dry collection 95% +/- efficiency	Advanced acid gas control scrubber (dry or wet/dry) 99% + efficiency
Canada[a]	Canada[a]
0.7–0.85 mg/Nm^3 - 12% CO_2 MB w/ESP 0.5–0.7 mg/Nm^3 - 12% CO_2 SA	0.01–0.6 mg/Nm^3 - 8% O_2 MB w/scrubber
Federal Republic of Germany[b]	Federal Republic of Germany[b]
0.42–0.62 mg/Nm^3 - 11% O_2 MB w/SDA and ESP	0.07–0.08 mg/Nm^3 - 12% O_2 MB w/wet scrubber
France[c]	France[c]
0.5–0.66 mg/Nm^3 - 7% CO_2 (wet)	<0.06 mg/Nm^3 - 7% CO_2 (wet) MB w/ESP and scrubber <0.22 mg/Nm^3 - 7% CO_2 (wet) MB, DS and FF
Italy[d]	Italy[d]
0.1 mg/Nm^3 - 10% O_2 (wet) MB	0.01 mg/Nm^3 - 10% O_2 (wet) MB
Switzerland[e]	Switzerland[f]
0.3–1.5 mg/Nm^3 - 7% CO_2	0.035 mg/Nm^3 - 11% O_2 MB w/SDA and ESP
United Kingdom[g]	
0.5–0.6 mg/Nm^3 - 10% CO_2 (dry) MB	

Table 12. *(cont'd)*

USA[h]	USA[h]
1.2-2.2 mg/Nm^3 - 7% O_2 (dry) MB w/ESP 0.4-0.6 mg/Nm^3 - 7% O_2 (dry) RDF w/ESP 2.0 mg/Nm^3 - 7% O_2 (dry) MB 2.2 mg/Nm^3 - 12% CO_2 (dry) SA	0.028 mg/Nm^3 - 12% CO_2 (dry) MB w/wet/dry scrubber and FF

DS = Dry scrubber. ESP = Electrostatic precipitator.
FF = Fabric filter. MB = Mass burn.
RDF = Refuse-derived fuel. SA = Starved air.
SDA = Spray drier-adsorber.
[a] [13,26,27]. [b] J.T. Möller, personal communication, 1987.
[c] Dimitrov, paper given at Third National Seminar of the Air Quality Agency, 1987. [c] [28]. [d] [29]. [e] [30]. [f] Möller, personal communication, 1986. [g] [31]. [h] [32].

Cadmium emissions range between 0.06 and 2.0 mg/Nm^3 for traditional plants and 0.001 and 0.04 mg/Nm^3 for advanced plants (Table 10). The lowest specific test result was 0.00004 mg/Nm^3 for an advanced plant [23]. However, as the specific conditions of this plant operation and its design features are unknown, this result should not be included in the range of anticipated test results. Test results for lead ranged between 0.2 and 50 mg/Nm^3 for traditional plants and 0.01 and 0.8 mg/Nm^3 for advanced plants (Table 11). Test results for mercury ranged between 0.1 and 2.2 mg/Nm^3 for traditional plants and 0.03 and 0.2 mg/Nm^3 for advanced plants (Table 12). Results for advanced plants for mercury removal depend largely on temperature and technology.

Table 13 gives ranges of stack emission rates of cadmium, lead, and mercury from incinerators in Europe, Canada, and the USA. To calculate the increase in ambient air concentration given in Table 13, a dilution factor of 5×10^{-5} was used. This factor is lower than the one used in the dioxin report of the WHO meeting in Naples [1]. This value corresponds to the value based on the model of the TA-Luft [24]. To calculate the dilution factor for deposition rates, two different values have been chosen:

cadmium and lead $\frac{(2 \times 10^{-2}\ ug/m^2/d)}{ug/m^3}$

mercury and PAH $\frac{(2 \times 10^{-3}\ ug/m^2/d)}{ug/m^3}$

The value of 2×10^{-2} has been derived from preliminary research on the dry and wet deposition of cadmium and lead in the vicinity of MSW incinerators [25]. The value of 2×10^{-3} refers to the model of the German TA-Luft 1986 for industrial incinerators.

In conclusion, the results indicate that traditional MSW incinerators may have a significant impact on the ambient air concentration as well as on the deposition rates of cadmium and lead. In contrast, the contributions from advanced MSW incinerators are in the range of the background levels or even less.

Table 9 summarizes the element emission factors (in grams per tonne of input MSW) and distributions (in percent of input MSW) for the data available for analysis at this meeting. The distribution of the elements depends on the element properties as well as on the design and operation of the incinerator and gas cleaning system, and the composition of the MSW.

PAH

A wide range of PAH has been identified in the various effluents arising from MSW incineration [33,34]. A number of studies have assessed PAH in various parts of incinerators. However, there is a

Table 13. *Emission, ground-level concentration, and deposition rates for selected metals and PAH*

Compound	Stack emissions (mg/Nm^3)	Calculated increase in ambient air concentration (ground level) (ng/Nm^3)[a]	Rural background ambient air (ng/Nm^3)	Calculated increase in deposition rates (wet and dry)[b] ($ug/m^2/d$)	Background deposition rates ($ug/m^2/d$)
Cd (2)	0.06-2	3-80	<1-5	1-40	0.5-2
(3)	0.001-0.04	0.05-2		0.02-0.8	
Pb (2)	0.2-50	8-3000	50-300	3-1000	40
(3)	0.01-0.8	0.5-40		0.2-20	
Hg (2)	0.1-2.2	5-10	1-4	0.2-4	1-<1
(3)	0.03-0.2	2-10		0.02-1	
PAH (4)	0.003-0.2(5)	0.1-<0.10(5)	<1	0.006-0.4	-

[a] Dilution factor for ambient air concentration, 5×10^{-5} (see text).
(1) Dilution factor for deposition of Cd, Pb (2×10^{-2}), Hg and PAH (2×10^{-3}) (see text).
(2) Traditional incinerator.
(3) Advanced incinerator.
(4) No differentiation between traditional and advanced except as noted.
(5) Large value stands for small-scale incinerator (<500 kg/h), which may produce high air concentration due to low stack height.

lack of consistency in the individual PAH reported and the methods of sampling and analysis used. Consequently, data are insufficient to derive good mass balance determinations that relate the amount of material entering, synthesized in, and leaving the incinerator system.

In a well-run plant, PAH in the refuse feed material are unlikely to persist through the combustion process. More probably, PAH in the various effluent streams result from incomplete combustion of ordinary non-PAH carbonaceous material or high-temperature free radical mechanisms [35,36].

The most important emission pathways of PAH are via the flue gases and the bottom ash, due to the significantly greater mass of bottom ash compared to that of other plant residuals. PAH account for less than 1% of the total organic carbon (TOC) in the products of incineration.

Based on a review of test results, stack emissions were tabulated for PAH and benzo(a)pyrene (BaP) (Table 14). Very little information is available that would allow a separate projection of PAH emissions for "traditional" and "advanced" plants.

The contribution of incinerator emissions to background ambient air concentrations and deposition rates of PAH can be estimated (Table 13) using dilution factors of 5×10^{-5} and 2×10^{-3}, respectively, based on the model of the German TA-Luft [24].

Incinerator ash

During incineration, 98 to 99% of the organic matter of MSW is converted to carbon dioxide and other volatile compounds, resulting in a mass reduction of 65 to 75%. Depending on the composition of the MSW, about 200 to 300 kg of bottom ash is produced per tonne of MSW. The volume of this bottom ash corresponds to about 10% of the initial MSW volume. The amount of filter ash is approximately 10% of the mass of the bottom ash, depending on the operating conditions of the furnace and the gas cleaning system. Ash deposited in the economizer and boiler will vary from 1.0 to 10.0 kg/t of initial MSW. If dry or wet/dry acid gas control scrubber systems are used for emission control, the mass of filter dust or combined fly ash and chemical treatment additive (usually lime) will be increased by 25 to 150%.

As noted earlier, fly ash contains higher concentrations of volatile metals compared with bottom ash (Table 8). Precipitator ash contains higher chloride and sulfur concentrations than bottom ash (Table 8), which together with the smaller particle size and the different chemical form of the element indicates a greater tendency for metals to leach from the precipitator ash than from the bottom ash. Experiments (P.H. Brunner & P. Baccini, paper given at Second International Conference on New Frontiers for Hazardous Waste Management, 1987; S.E. Sawell, unpublished report, 1987) [25] show a

Table 14. *Emission data on PAH from stacks*

Data by country	Technology
Total PAH	
Canada[a]	
0.005–0.02 mg/Nm^3	Traditional MB
0.00002–0.00013 mg/Nm^3	Advanced MB, w/scrubber and FF
0.007–0.012 mg/Nm^3	Traditional MB (starved air)
Italy	
0.0075–0.208 mg/Nm^3	Traditional MB
Japan	
0.04 mg/Nm^3	Traditional MB (batch fed)
0.002 mg/Nm^3	Traditional MB
Sweden[b]	
0.001 mg/Nm^3	Advanced
BaP only	
Canada[b]	
0.008–0.062 ug/Nm^3	One traditional plant
0.000015–0.00013 ug/Nm^3	One advanced plant
Federal Republic of Germany	
0.0005–0.005 ug/Nm^3	Five traditional and advanced plants
Italy	
0.05–33[a] ug/Nm^3	Four traditional and advanced plants

Table 14. *(cont'd)*

Japan	
0.5-1.5[b] ug/Nm^3	Two traditional plants
United States[c]	
5.9-19 ug/Nm^3	One traditional plant

[a] BaP and BeP combined data.
[b] Assumed particulate emission rate of 100 mg/m^3.
MB = Mass burn. FF = Fabric filter.
[a] [13,26,27] [b] [37] [c] [32]

high leachability at first water contact of the precipitator ash (10% cadmium, 0.5% lead, 20% zinc with deionized water). With a high content of calcium oxide, as used in acid gas treatment, the pH of an aqueous suspension of filter ash rises to 9-10. At this pH, most metals are present as insoluble hydroxides. However, a question remains about the long-term behaviour of ash deposits. In a laboratory study designed to examine long-term leaching patterns at landfill sites, Wadge & Hutton [38] report a relatively rapid movement of cadmium and lead from incinerator fly ash.

Several factors have the ability to change the mobility of metals in an ash deposit: pH changes, complexing agents, and microbial activity. At pH 3-4, 85% of the cadmium, 5% of the lead, and 50% of the zinc in ash are mobilized (Sawell, unpublished report, 1987) [25]. If lime or other products are added during gas purification, the pH and the salt content change, which alters the physicochemical behaviour of the filter residue (mobility of the amphoteric elements). Thus, to ensure safe long-term behaviour in a natural environment, further treatment of flue gas cleaning residues may be required.

The content of organic matter in the filter residues is about 10 to 40 g/kg. Less than 1% of the compounds in this fraction has been identified. Of those identified, the largest group consists of chlorinated benzenes (approximately 1 ug/g), PAH (0.2-0.5 ug/g), and chlorinated phenols (approximately 0.5 ug/g) [16]. The organic compounds are tightly bound to particle surfaces and are not mobile in an aqueous environment if organic solvents are absent. An exception is chlorinated phenols, which may go into solution. Still unknown is the extent to which carbon can be used by microbial activities, reducing pH levels and changing the leaching characteristics of ash

deposits (P.H. Brunner & P. Baccini, paper given at Second International Conference on New Frontiers for Hazardous Waste Management, 1987).

Research on disposal of MSW incinerator ash residues addresses the following topics:

- concentration of the hazardous elements in the ash, including controlled leaching of the metals at different pH;
- fixation of the metals in the residues by vitrification or solidification (cementor silica-based) [39,40];
- destruction of carbonaceous residues by optimizing combustion and reheating filter dust;
- leach tests to determine the long-term behaviour of residues from MSW incineration in the natural environment;
- plant availability of metals in incinerator residues [41].

Government regulations

Canada, the USA, and most countries in Europe have regulations covering emissions from MSW incinerators. Regulations normally establish specific control levels for the emissions of, for example, dust, carbon monoxide, oxides of nitrogen, heavy metal compounds, and acid gases, together with stipulations for the method of monitoring and enforcement.

Pollution control agencies consider a number of factors that influence their decisions on how to control MSW incineration, including the following ones:

Plant-related factors. Some countries, such as France, feel that large plants should be more strictly controlled than small ones. Similarly, plants within residential areas may have to be regulated more carefully than those well removed from towns. Other factors within this category would include the type of plant, local topography, and prevailing meteorological conditions.

Pollutant-related factors. Pollution control decisions will also depend upon the nature of the substances to be controlled. Some, such as specific metal compounds, can be relatively easily sampled and analysed directly. Others, such as PAH, are very costly to determine as individual compounds within this entire group of compounds. Even with metals, it is arguable whether those metals of similar toxicity should be identified and controlled as a group, as is the case in France, rather than establishing emission limitations for individual metals as is generally the case in the USA. Also,

while the total amount of a metal is generally determined in a test, sometimes the species are of interest.

Pollution control options. Another choice to be made is whether action should be taken to control the process using specific control limits or by specifying the equipment technology to be used.

Control limits can be applied either at the source directly or as ambient air quality standards at the point of maximum source impingement on a receptor. When applied at the source, operators are allowed to choose the technology to control their plant emissions, provided that set control limits are met. These limits would specify the maximum quantities of a substance that could be emitted by any emission route, whether it be to air, water, or land. In the case of a metal, the control limit could relate directly to the pollutant, as is the case in many countries. In the case of PAH, which are difficult to measure as a total listing of all compounds, indirect indicators might have to be used as surrogates. These are analytes (such as BaP), which are relatively easy to determine and vary in concentration as the target species would. This approach has some attractions, but the identification of suitable surrogates is difficult and, in the case of PAH, would require further research work.

In some cases, ambient air quality standards have been adopted: for example, the Council of the European Communities Directive on the limit value for lead in the air (OJ L 378, 31 December 1982). This approach is useful when the pollutant is emitted from a large number of small sources.

Specific control procedures are a commonly used alternative or addition to setting control limits. The control equipment or technology that an operator should adopt is specified (i.e. best available control technology, BACT), bearing in mind that this should not entail excessive costs. This approach is being adopted increasingly by the Council of the European Communities. It is also somewhat similar to the best practicable means (BPM) system used in the United Kingdom.

When both the substances to be controlled and a well-defined control technology can be identified, legally binding emission standards can be set, as many countries have with metals. But in the case of PAH, the compounds most hazardous to health can probably not be identified or a clear consensus arrived at as to the most cost-effective control process. In these circumstances, recommendations or guidelines are commonly used. While they are not legally binding, they do indicate target values to be aimed at. This approach has proved useful for dioxin control in the past (i.e. Sweden, Denmark, and Italy) and has encouraged interested parties to identify the control issues and adopt appropriate control practices.

An implicit component of any control strategy is the need to test whether a control requirement is being met; this is called compliance testing. Broadly speaking, compliance testing has two main approaches: monitoring to check emission levels and/or visits by qualified inspection personnel.

The problems of monitoring emissions are considerable and should not be underestimated. Neither should monitoring instrumentation and procedures be assumed to be sufficiently developed to be accepted (as proof of plant malfunction) in a court of law.

Both regulatory pressures and technological advances have succeeded in transferring the environmentally harmful species from stack emissions into the water effluents and residual ashes from MSW incinerator. The legislation for the disposal of these effluents is often less well developed than those for air pollution control and may have to be modified in the future.

Regulations for MSW incinerator emissions

Many countries have national regulations for controlling emissions of heavy metal compounds. These are constantly under review, and listing the provisions of all the countries is impossible. Instead, the current proposals of the Commission of the European Communities for new municipal waste incineration plants can be given as an indication of possible regulations (Table 15). The emission limit values in Table 15 are standardized at the following conditions: temperature, 273 K; pressure, 101.3 KPa; and 11% oxygen or 9% carbon dioxide (17% O_2 in case of incineration plants of a nominal capacity less than 1 tonne per hour).

Water and solid effluent provisions

Incinerators produce liquid effluents from ash quench tanks, wet scrubbers, and other areas of water usage in the processing of MSW. They also produce large quantities of residual ash for disposal. Many countries have guidelines for the handling and disposal of these effluents, but few have specific legislation that treats solid and liquid effluents from MSW incinerators differently from any other pollutant discharge. While a general discussion of water treatment and solid waste disposal legislation is outside the scope of this report, the specific problems of MSW ash handling have prompted some governments, including France, the Federal Republic of Germany, Sweden, Switzerland, and the USA, to consider legislation calling for specific regulations for MSW incineration plants. These proposals range from requirements to contain and treat wastewater produced by MSW incineration prior to discharge, to requirements for the segregation of ashes and the separate storage/disposal of fly ash in special landfills.

Table 15. *Proposals of the Commission of the European Communities for regulations for MSW incinerator emissions*

Pollutant	Emission limit value[1] (mg/Nm^3)
Total dust	50[a]
Heavy metals	
Pb + Cr + Cu +Mn	5[b]
Ni + As	1[b]
Cd + Hg	0.2[b]
Hydrochloric acid	50[a]
Hydrofluoric acid	2[b]
Sulfur dioxide	300[b]

[1] These values do not apply to plants designed to burn refuse-derived fuel and that use such fuels exclusively. The Commission shall submit to the Council as soon as possible proposals for limit values to be applied to these plants.

[a] 100 mg/Nm^3 in case of incineration plants of a nominal capacity less than 3 tonnes per hour.

[b] These limits do not apply to incineration plants of a nominal capacity less than 1 tonne per hour.

HEALTH-RELATED BACKGROUND

Major episodes of ill-health have been reported among populations acutely and chronically exposed to cadmium, lead, and mercury arising from non-occupational sources other than MSW incinerators [42-44]. Epidemiological studies have not been conducted in the vicinity of MSW incinerators, but considerable amounts of cadmium and mercury are emitted from these facilities [30,45,46] and lead is a major constituent of the metals in MSW [47]. For these reasons, the metals of prime concern with respect to emissions from MSW incinerators are cadmium, lead, and mercury.

This section summarizes the environmental pathways, metabolism, toxicity, and human health effects of these three metals. Comprehensive reviews of their health effects are available [42,48-51]. The relative significance of metal and PAH emissions from MSW incinerators compared with other high-temperature combustion and industrial processes is considered later.

Human health concerns of cadmium, mercury, and lead

Exposure pathways

Generally, dietary intake is the major route of exposure for the three metals. In the case of lead, inhalation is also an important source of exposure in urban-dwelling populations [51]. Infants may also ingest considerable quantities of lead by inadvertently consuming dusts or deliberately ingesting lead-rich paint or soil [52,53]. Tobacco smoking can be an important source of cadmium intake [54]. Methylmercury is the chemical species of mercury of importance in environmental exposure, and, in the general population, the major (>90%) source of this is fish consumption [55].

The three metals differ greatly in the percentage that is absorbed across the gastrointestinal tract. Methylmercury is absorbed efficiently (approximately 95%) while the fractional absorption of cadmium is only about 5%. This latter value increases considerably in cases of iron deficiencies [56]. Lead absorption is often reported to be about 10% in adults, but a higher value (approximately 50%) is often assumed in children [57,58].

Metabolism

Once absorbed, the three metals display characteristic tissue distributions. Cadmium selectively accumulates in the kidney and liver where it binds to the protein metallothionein [42]. In contrast, methylmercury is initially distributed throughout the body, readily crossing the blood-brain barrier, although the kidneys retain the highest tissue concentration [59]. Most of the lead (approximately 90%) is stored in the skeleton, but the liver and kidney also contain relatively high concentrations. An important feature of lead and methylmercury is that both easily cross the placental barrier; this is particularly efficient for methylmercury. The placental barrier is more efficient for cadmium; the median blood cadmium of neonates was about 50% lower than in mothers [60].

Health effects and critical populations

Lead may affect two endpoints: the haematological system and the central nervous system. The haematological effects arise at the beginning and final steps of haem synthesis. Interference in the last stage results in the accumulation of protoporphyrin in the blood, which may occur at blood lead concentrations of about 15 ug/100 ml; that is, at concentrations regarded as "normal". At higher levels of lead exposure, anaemia can develop. The haematological effects of lead are reversible.

Of potentially greater concern is the reported association between certain neurobehavioural effects and lead exposure in ur-

ban-dwelling children [61]. Decreases in measurements of intelligence and behaviour have been found in children from the general population subjected to commonly occurring levels of lead exposure. These effects are widely believed to reflect the greater susceptibility of the fetus and child to lead resulting from greater sensitivity of the developing central nervous system and higher intake and uptake of the metal as compared with adults.

Methylmercury intoxication is also characterized by effects on the central nervous system, the major areas affected being associated with the sensory, visual, and auditory functions as well as those concerned with coordination [55]. The developing nervous system of the fetus is more sensitive than that of the adult, and prenatal exposure can subsequently result in neurotoxic effects in the infant in the absence of effects in the mother. Populations with heavy fish consumption have been identified as being exposed to elevated levels of methylmercury, which is enriched to a high degree in the aquatic food chain [55].

The kidney is the critical organ of intoxication after long-term exposure to cadmium [42]. Increased urinary levels of B_2-microglobulin (B_2M), retinol-binding protein (RBP), and N-acetyl-*B*-D-glucosamidase (NAG) appear to be the most sensitive parameters of early changes in tubular function. The concentrations of B_2M in serum and albumin in urine appear the most sensitive parameters of changes in glomerular function. However, these forms of proteinuria are not specific to cadmium [62-64]. Nevertheless, measurement of B_2M is being increasingly adopted to monitor for the early effects of cadmium exposure.

Human health concerns of PAH

Environmental data and exposure pathways

About 500 PAH have been detected in air (Pott, address before the Federal Council of Environment, Berlin (West), 1983), but reports of ambient air levels of PAH dating from the 1950s to about 1970 are limited only to BaP only. These data are not necessarily comparable with more recent reports because many different sampling procedures and analytical methods have been used. The natural background level of BaP is determined by the production of PAH in vegetation [65]. PAH could be detected even in borings out of 140-m depth more than 100 000 years old [66]. Compared with the anthropogenic load, the natural background level of PAH in air amounts to not more than 1%.

The annual average of BaP concentrations in urban European locations often exceeded 100 ng/m^3 in the 1960s [67], while more recent measurements in the Federal Republic of Germany and USA indicate that levels are now mostly below 10 ng/m^3 [67,68].

Daily intake of BaP by inhalation of polluted ambient air containing a relatively high concentration of 50 ng/m^3, which is similar to the levels found in the vicinity of coke oven plants [69], is estimated at 400 ng BaP [51]. However, the BaP intake in clean rural areas and cities where adequate smoke control has been achieved may be only a few percent of this or less. Tobacco smoke will add to the inhalation of PAH. Modern "low-tar" cigarettes deliver 10 ng BaP [70], while Grimmer et al. [71] measured 22 ng/m^3 of BaP in a room heavily polluted with cigarette smoke.

Concentrations of BaP in drinking water range from 0.1 to 23 ng/l [72]. The possible sources of PAH in foodstuffs are varied and extremely widespread. Curing smokes, contaminated soils, polluted air and water, modes of cooking, food additives, food processing, and endogenous sources have been investigated. Some 100 PAH have been identified and numerous reviews are available [72-74]. BaP concentrations up to 50 ug/kg were measured in broiled and smoked food [73], and BaP concentrations ranging from 2 to 540 ug/kg have been reported in shellfish [75]. A person in the German Democratic Republic is estimated to ingest 24 to 85 mg BaP during his or her lifetime (70 years) [72]. The daily intake of BaP from food is estimated to range from 0.16 to 3.0 ug in the USA [76].

For most people, oral intake of PAH is considerably greater than the amount inhaled. Much attention has been given to the role played by inhalation of PAH in causing respiratory tumours, but the evidence is not clear whether, and to what extent, ingested PAH may be implicated in respiratory or other tumours.

Uptake and metabolism

PAH are highly lipid-soluble and are absorbed across the lungs, gut, and skin of mammals. As most of the PAH in the ambient air are attached to fine particles, the potential health effects of PAH after inhalation depends on the deposition rate in the respiratory tract, residence time in the lung, the dissolution rate of PAH from the particles, and the chemical structure. Furthermore, the overall composition of the particulate matter (i.e. the ratio of the mass of the carbonaceous core to the mass of the attached organic material) is a factor in determining the biological half-life of the particle-related PAH in the lungs. Particles that consist mainly of organic material, such as coal tar aerosols, will be eliminated from the lungs almost completely within 24 hours. In contrast, the lung clearance half-life of soot particles, which consist mainly of carbonaceous material, such as diesel soot, is relatively long [77]. PAH associated with the latter particle types may exert a stronger carcinogenic effect in the lungs because their residence time is longer.

In general, PAH metabolism consists of a series of oxidation steps catalysed by monoxygenases and epoxide hydrase. Metabolites

include epoxide intermediates, dihydrodiols, phenols, quinones, and their combination. After being metabolized to dihydrodiols, some of the epoxides can be further oxidized to diol epoxides. These latter compounds are generally thought to be the ultimate carcinogens [72].

Oxidized metabolites of PAH are later conjugated with glucuronic acid, glutathione and sulfonic acid, forming polar metabolites that are more easily excreted from the body [78,79].

Toxicity data and biological effects in laboratory animals

Toxic effects of PAH have not been studied as well as their oncogenic effects. Longer-term oral exposures to PAH have resulted in various systemic effects in animals, but overt signs of toxicity are not usually produced until the dose is sufficiently high to produce tumours [72].

Although embryotoxic, teratogenic, and immunosuppressive effects have been observed in some assays, PAH are not acutely toxic compounds.

Several PAH and some of their metabolites have been shown to be mutagenic in mammalian cells and bacterial assays [72,78-80].

The ability of individual PAH to produce neoplasia has been intensively researched for some 50 years, particularly on induction of skin tumours in rodents by topical application of PAH. The carcinogenicity of selected PAH in rodents by different routes of administration has been recently summarized [81].

BaP-induced tumours usually develop at the site of application. For example, rats fed BaP develop forestomach tumours [82] while intraperitoneal infection leads to hepatic tumours when phenobarbital, a promoter, is added to the diet [83], and inhalation induces respiratory tract tumours [84]. Respiratory tract carcinogenicity of BaP is enhanced when this compound is coadministered with particulates, such as ferric oxide, or is absorbed to the administered particles [85].

Many substituted PAH, such as nitro-, methyl- and oxygenated PAH, are also likely in combustion products. Many of these compounds are mutagenic in in vitro tests, and some have been reviewed for evidence of human carcinogenicity [72].

Human health effects

Data summarized by IARC [49] demonstrate carcinogenicity to humans from industrial exposures to mixtures containing PAH. These occupational exposures include aluminium production, coal gasification, coke production, and iron and steel founding. Bitumens, coal tars and derived products, shale oils, and soots have also been evaluated for their human health effects (skin cancer) [86]. A detailed evaluation of the carcinogenicity of PAH in coke oven emissions has been published [87].

Epidemiological studies of these workplaces provide sufficient evidence of the role of inhaled PAH in inducing lung cancer [88-92]. However, levels of PAH in these workplaces were very high, often exceeding 10 ug/m^3 BaP [49,93,94], which is several orders of magnitude greater than in ambient air. In addition to high PAH levels, other carcinogenic gases and metals were present in the occupational settings studied. For example, coke oven emissions may contain other carcinogenic agents, including arsenic, chromium, nickel(VI), 2-naphthylamine, and benzene.

Cigarette smoking is strongly and consistently associated with cancer at several sites, including the lung, larynx, oral cavity, oesophagus, bladder, and kidney [95,96]. Thirty-four PAH, some of which are recognized carcinogens, have been identified in mainstream and sidestream cigarette smoke [72].

Little is known about the possible implications for human health of PAH in food and water, and the role of PAH in toxic endpoints other than carcinogenicity has not been well studied. Therefore, on the basis of available experimental results, the carcinogenic potential of PAH is considered the most significant health effect for the general population.

Characterization of potential human health risk from PAH

A major difficulty in hazard assessment of PAH is that the data for compounds other than BaP are very limited, and that from most emissions, such as those from municipal waste combustion, the full range of PAH has not been analysed. The hazard assessment of this diverse but related group of components would benefit from a model of biological effects compared with those of a well-studied compound, as has been done for dioxins. To this end, several attempts have been made to compare carcinogenic activity of various PAH with that of BaP [81]. At present, however, such a comparative ranking seems prudent only for the most general of classification.

Because several PAH are carcinogenic [49,87] and data for BaP are most extensive, BaP is normally used as an index of the carcinogenic potential of all PAH in a sample. However, PAH profiles detected in different types of industrial emission and workplace air can differ widely.

Several approaches have been taken to provide quantitative estimates of potential human risk from PAH exposure. WHO [51] calculated a unit risk of 9×10^{-5} cases of cancer for a lifetime exposure to 1 ng/m^3 BaP. This upper bound of estimates is derived by application of the linearized multistage procedure to data for lung tumour incidence among coke oven workers [81,91] and using a reported value of 0.71% BaP in benzene-soluble coke oven emissions [97]. This risk assessment implies that about 9 per 100 000 exposed people may die from cancer of the respiratory tract as a result of spending a lifetime in ambient air containing an average level of

1 ng/m^3 BaP in a mixture with all the other PAH and related substances of coke oven emissions.

SIGNIFICANCE OF MSW INCINERATOR EMISSIONS FOR HUMAN HEALTH

Emissions from MSW incinerators, in common with many high-temperature combustion and other industrial processes, contain a wide range of inorganic and organic compounds, including metals and PAH [31,32,34,45]. When assessing potential health hazards, emissions of metals and PAH from incinerator plants must be considered in conjunction with pollution from other pertinent sources in the area. In Czechoslovakia, for example, coal combustion has been identified as an important source of this type of emissions [98-100].

This section considers the significance of MSW incinerator emissions for human health Occupational hazards are not considered here, but workers in incinerator plants will be exposed to dust and fumes containing metals and PAH [101]. The problems encountered will be similar to those in other industrial plants where furnaces are operated.

Technological aspects

MSW incinerators initially included less efficient air pollution control equipment than required by most current regulations. Thus, the emissions of pollutants were high, as can be seen from examples cited in Tables 10-12 and 14. The impact of such emissions on the environment can be assessed only in rough terms, using data on (a) the likely concentrations of toxic chemicals in the vicinity of the plants, resulting from modelling computations, (b) actual measurements of the concentrations in the air close to incinerators as compared with the background levels, and (c) measurements of toxic chemicals in soil and vegetation near incinerators.

In cases (b) and (c), measurements at various distances from the plant are helpful in evaluating the real impact, that is, the increase above the local background level.

Although such data on the impact of traditional incinerators are incomplete, they point to the possibility of some impact on the environment in the immediate vicinity of the plant. Such assessment, even if incomplete and based largely on modelling computations, justify legislation restricting emission from MSW incinerators. Advanced control technology can markedly reduce the emission of metals (Table 7). The following considerations seem to be relevant in this respect:

(a) the air concentrations (increase above background level) in the immediate vicinity as compared with the urban back-

ground levels; here, the direct toxic action via inhalation should be considered

(b) the indirect impact in the vicinity of the plant, the indicators of which may be the increase of the concentration in soil and plants grown for human consumption

(c) direct impact in the immediate vicinity on humans, as judged by biological monitoring of human samples (e.g. blood, urine, and hair)

(d) indirect impact at regional and global level, assessed by comparing of emission sources of varying strength.

Populations under consideration

Two distinct populations can be identified when considering exposure to refuse incinerator emissions. The first consists of those individuals who are close to incinerator plants while the second is the general population.

Populations close to incinerators

In many countries, refuse incinerators are located in densely populated areas. Therefore, the identification of the potential routes and intensity of exposure of local residents as a result of emissions from these plants are pertinent. Unfortunately, little information is available on the environmental levels of cadmium, lead, mercury, and PAH around municipal incinerators. However, their emissions could elevate air concentrations, deposition rates, and soil and dust levels. The deposition of mercury is especially difficult to predict because of the mobility of this element in the environment through re-vaporization that gives rise to the "bounce off" effect reported for other point sources [102]. Mercury emissions may increase mercury levels in local freshwater, especially in water systems with low pH. Elevation of ambient air levels of metals is unlikely to be large near advanced incinerator plants for several reasons: stringent emission controls are imposed on new incinerator plants; modelling data indicate minimal impact around advanced incinerators; and high urban background values may mask any contribution from refuse incinerators.

Air concentration increases will have to be relatively large to increase noticeably total exposure levels, as the dietary intake of these metals is normally greater than the respiratory intake. This is particularly important for cadmium where even a doubling of urban background levels (approximately 10 ng/m^3) would produce only minor increases in total cadmium exposure.

Deposition of cadmium, lead, and PAH onto soil and garden vegetables around incinerators, either from stack emissions or fugitive release, could increase dietary exposure. Once again, little information is available.

Finally, street and possibly household dusts could contain elevated concentrations of metals (cadmium and lead) and PAH as a result of incinerator emissions. Lead levels in urban dust are markedly elevated (approximately 1000 ug/g) compared with those of rural dust, and incinerator emissions are unlikely to cause detectable increases, although fugitive releases may have an impact in the immediate vicinity. The relative contribution of incinerators will increase only when the use of unleaded petrol is widespread.

General population

The exposure pathway of relevance with respect to the general population is dietary uptake. The tall stacks (commonly about 100 m) of modern incinerators will widely disperse and dilute the plume. Thus, the metal and PAH emissions may contribute to the deposition of these contaminants at considerable distances from the source. The deposition of contaminants on agricultural land or in water systems can have two effects: a long-term increase in the soil or sediment concentration of the contaminant in question; and a direct deposition onto crop plants and the subsequent ingestion of the contaminant. In the first case, if the soil level increases, the crop level may also rise. This is the case for cadmium, but soil uptake of lead and PAH by plants is considered to be less important than direct deposition [103].

The question needs to be answered, however, whether incineration contributes significantly to the deposition of these contaminants at sites distant from point sources. Such an assessment requires information on the emission strengths of other sources to construct an inventory of atmospheric release. In the case of lead, motor vehicle emissions dwarf incinerator emissions [104]. Therefore, this pathway is unlikely to be of current importance in Europe for lead in crops. The situation for PAH is uncertain given the lack of information. Only for cadmium is the pathway considered important because of the relatively large discharges of this metal from incineration [105].

Local health impact of cadmium

The Air Quality Guidelines for Europe [51] quote a concentration of 2 ug/m^3 of cadmium (2000 ng/m^3) as being likely to cause, through life-long exposure, renal accumulation of this metal sufficient to reach the critical concentration in the kidney. The recommended levels were are <1-5 ng/m^3 for rural areas and 10-20 ng/m^3 for urban areas, with emphasis on the qualifying

statement that "an increase in present levels should not be tolerated".

Direct measurements of cadmium concentrations in the air close to incinerators seem to be lacking. A rough estimate can be obtained in two ways. One way is through modelling, in which one assumes for the ground level a fraction of the cadmium concentrations directly measured in the effluent gases. Such calculations, although burdened with a high uncertainty, lead to the expected increase (over background levels) of the order of up to 80 ng/m^3 for traditional incinerators and less than 2 ng/m^3 for advanced incinerators (Table 13). Such values, neither alone nor in combination with the background levels for urban areas, would be expected to lead to adverse health effects by inhalation. The second way is through recomputing the existing data on the rate of deposition. In a pathway analysis, 1 ng/m^3 is estimated to cause a deposition of 1.6 g/ha/y [106].

Deposition rates of cadmium in the vicinity of an incinerator [25] in a rural area of the Federal Republic of Germany were about 20 g/ha/y for a traditional incinerator, and half this value after advanced control equipment was added. The first value corresponds roughly to an ambient air concentration of 10 ng/m^3. Further similar examples could be considered, but the conclusion would be valid that these concentrations are at least one order of magnitude lower than the critical dose for inhalation. The above example, if applied to an urban area where the background concentrations of cadmium are generally about 5 to 10 ng/m^3, would lead to a concentration between 10 and 20 ng/m^3, and the conclusion remains valid.

The possible influence of incineration on environmental levels of cadmium in the immediate vicinity of the plant has, until recently, received little attention. Hutton et al. [107] record, in the vicinity of a traditional incinerator in the London area, deposition rates of about 3 to 70 g/ha/y in the downwind area as compared with 1 to 16 g/ha/y in the upwind area. This deposition has not, after 15 years of operation, markedly elevated cadmium in either street dust, soil, or grass as compared with the upwind area and other London locations. Studies summarized by Brunner & Zobrist [30] reported elevated levels of cadmium in spinach grown near two refuse incinerators, though only trace quantities were detected in grass samples near other plants.

At present, the evidence does not indicate that the level of contamination of the environment by cadmium in the immediate vicinity of incinerators has created a health hazard. However, only limited data are available and this situation may change with time. Therefore, locally grown food should be periodically monitored for cadmium levels.

Local health impact of mercury

Urban ambient air concentrations of mercury are usually in the range of 4 to 5 ng/m^3 [55]. For occupational exposure, a concentration in air of 50 ug/m^3 is allowed. For non-occupational exposure, the outdoor and indoor air concentrations of mercury are proposed to be safe (with an appreciable safety factor) at 1 ug/m^3 [51].

The concentrations of mercury in ambient air in the vicinity of incineration plants have not been monitored. However, concentrations are unlikely to exceed 100 ng/m^3 (see Table 13) and cannot be considered of direct importance to the health of the inhabitants, irrespective of age, sex, and health status.

Inclusion of mercury in this report is based solely on one pathway. This involves deposition of mercury from the atmosphere onto soil and into surface waters: only the latter is relevant. Here, through conversion into methylmercury, it accumulates in fish that are eaten by people.

The only situation in which this pathway could have local impact is an area with rivers or lakes in the vicinity where fishing is an important activity and where the fish are consumed locally. The exposure assessment will be constrained by several factors, such as the history of past contamination of the area from other sources (as mercury in the lake sediments has a long residence time) and the background of the given area (region) dependent on the global mercury cycle, which basically dominates the deposition of the metal. Another confounding factor, however, is the acidification of surface waters increasing in parallel with the overall contamination of ambient air from anthropogenic sources, enhancing the solubility of mercury deposits [102].

Local health impact of lead

Inhalation represents an important source of lead uptake for the general adult population. Uptake by inhalation is in the range of 20 to 70% of the total, depending on the ambient air concentration. Based on the assumed total uptake, the Air Quality Guideline value for lead in ambient air is 0.5 to 1.0 ug/m^3 and this includes a safety factor of about 2 [51].

The ambient air levels of lead in European areas vary from 0.1 to 0.3 ug/m^3 in rural areas and from 0.5 to 3.0 ug/m^3 in most European cities [51]. The degree to which waste incineration increases these concentrations is not known, and some estimates could be made only from modelling data and the few existing data on lead deposition rates.

In rural areas of the Federal Republic of Germany, the background deposition rate of lead is 150 g/ha/y, whereas downwind at <1 km of an incinerator, this value is 500 g/ha/y for the traditional

type and about 275 g/ha/y for the same plant after modernization [25]. Assuming that these data on lead concentrations in ambient air in rural areas are of the order of 0.1 ug/m^3, the lead levels near an incinerator (located in a rural area) could be of the order of 0.2 to 0.3 ug/m^3. Such values, if applied to an urban area, would elevate the background levels by some 0.1 to 0.2 ug/m^3. Thus, waste incineration alone does not seem likely to cause locally significant elevations of the ambient air lead concentration sufficient to lead to health effects by inhalation.

Other available data [107] do not point to significantly elevated levels of lead in street dust, soil, or grass in the vicinity of an incinerator in the London area. No impact on human health from the deposition of lead from incinerators onto locally grown food is expected.

For children, general contamination of the environment (e.g. street dust and soil) may play a dominant role in their exposure pattern. At present, the increased exposure derived from incinerator emissions via this pathway cannot be quantified but it is not expected to play a major role in urban areas.

To summarize, in general no local health impact of lead from incinerator emissions alone should be envisaged. The relative significance of lead emissions from incinerators to total exposure levels among local populations will be greater in more rural areas, where other sources, particularly motor vehicle emissions, make a smaller contribution.

Contribution of incinerator emissions to metal exposure in the general population

This section assesses the extent to which MSW incineration influences metal exposure in the general population. The issues of concern are the fate and significance of the long-range transport of emitted metals and the metal build-up in soils and sediments in rural areas.

Mercury

Fish consumption is by far the major source of methylmercury exposure in the general population. In most countries, marine fish are the dominant source but in some countries, freshwater fish are also regularly consumed. MSW incineration is a significant source of inorganic mercury in the atmosphere, but in most industrialized countries, coal combustion and other sources (e.g. chlor-alkali works) dominate. Nevertheless, MSW incineration is expected to make an important contribution to the mercury cycle at the regional and global level. What is not clear, however, is the significance of this perturbation on the behaviour and accumulation of methylmercury in edible fish species. Direct discharges of inorganic mercury to

aquatic systems are expected to be more important in this respect and to conceal any effect from atmospheric inputs, at least in the short term. Continued release of large quantities of airborne mercury by human activities may progressively increase concentrations of mercury in freshwater and possibly marine systems. If this is the case, MSW incineration is only one contributor to the overall anthropogenic total of airborne mercury; coal combustion is much more important.

Cadmium and lead

The issue of concern is whether airborne emissions of cadmium and lead from MSW incineration bring about significant atmospheric inputs of these two metals to agricultural land. For both metals, dietary intake is a major source of exposure for the general population. Even in rural areas, direct atmospheric deposition is an important source of lead in certain crop species [108].

However, most of the lead deposited in such areas is derived from petrol combustion [104]. Thus, refuse incineration will make only a minor contribution to lead levels in crops in rural areas.

For cadmium, the situation is different because, in contrast to lead, plants take up cadmium relatively efficiently from the soil [103]. Thus, the accumulation of cadmium in agricultural soils is of particular concern as this can lead to enhanced crop concentrations and hence increases in dietary intake for the general population. Important sources of cadmium inputs to arable land are phosphate fertilizers and atmospheric deposition. At local level, sewage sludge application is also important. In the United Kingdom, one third of the contribution from atmospheric deposition is estimated to originate from MSW incineration [103].

Many countries have taken steps to reduce cadmium emissions, while others are trying to prevent any increases in the average dietary exposure to the metal. At the same time, countries are increasingly recognizing that, in contrast to lead, the multiple source nature of cadmium necessitates control measures on a wide range of sources to reduce effectively inputs to agricultural land.

In view of this, refuse incineration is considered by several authorities to be a sufficiently large source of cadmium to merit further controls.

Environmental release of metals from refuse incineration compared with other sources

Numerous human activities result in the environmental release of potentially toxic trace elements. The identity of these sources is well known, and attempts have been made in recent years to assess their quantitative importance. Such studies have been carried out at the local level for a single city or county as well as at the

regional and global level. The inventories produced by these investigations provide an insight into the relative importance of traditional refuse incineration as an environmental source of metal release. These studies refer to emissions from traditional MSW incinerators; the contribution from incineration to overall metal emissions will decline as incinerators of advanced design become widely used.

Environmental release of metals in the United Kingdom

About 34 x 10^6 t of MSW are generated annually in the United Kingdom, of which about 10% is incinerated. Emission factors (gram metal released per tonne material consumed) were derived from experimental in-stack studies of incinerators in Europe and the USA. Table 16 shows the resulting atmospheric emission discharges. The residual trace metal burdens associated with the fly ash and furnace clinker are landfilled. Table 16 also shows the estimated quantities of the trace metals present in the remaining 90% of the MSW that is landfilled directly.

The data in Table 16 may be compared with the inventories of trace metal release to the atmosphere and landfill for all anthropogenic sources in the United Kingdom. In the case of cadmium, Table 17 indicates that traditional refuse incineration is the single largest source of atmospheric release, being responsible for about one third of the total. Atmospheric lead discharges are also considerable from traditional incineration, being greater than emissions from nonferrous metal production. Nevertheless, lead emissions from petrol combustion currently dwarf all other sources of lead. Mercury emissions from incineration account for about one tenth of the total anthropogenic discharge in the United Kingdom.

Table 16. *Environmental release of trace elements from municipal waste disposal in the United Kingdom, 1983* [103]

Disposal route	Trace elements release (t/y)		
	Cadmium	Lead	Mercury
Direct landfill	479	31 673	62.0
Incineration			
Atmosphere	5	142	5.9
Landfill	46	3 241	0.7

The landfill inputs of metals in Table 16 reveal the much larger tonnage associated with the direct disposal of MSW compared with the incinerator residues. However, the latter is likely to be of greater environmental significance at a local level because trace elements are generally present at much higher concentrations.

Environmental release of metals in Sweden

Several emission inventories have been carried out in Sweden in the last 5 years. A detailed investigation into the sources of atmospheric mercury is summarized in Table 18. These data indicate much smaller total emissions of mercury in Sweden than in the United Kingdom. In addition, refuse incineration emerges as a more important source of atmospheric mercury, mainly because of the relatively small quantity of coal combusted in Sweden. Refuse incineration accounted for over half the total released in 1985 [45]. The release of other metals from MSW incineration was less significant, accounting for only 0.3% of the total cadmium emissions.

As a result of a national programme for collecting used batteries and installing advanced flue gas treatment at some plants, the emission of mercury from waste incineration in Sweden is expected to be reduced from 3.3 t/y (Table 18) to about 1 t/y from 1987 (K. Hasselgren, personal communication, 1987).

Table 17. *Estimated atmospheric emissions of trace elements in the United Kingdom* [103]

Source	Atmospheric emission (t/y)		
	Cadmium	Lead	Mercury
Nonferrous metal production	3.7	51	5.0
Production and use of articles containing the element	ND[a]	6802	10.1
Iron and steel production	2.3	478	1.8
Fossil fuel combustion	1.9	80	25.5
Cement manufacture	1.0	36	2.5
Municipal waste incineration	5.0	142	5.9
Sewage sludge incineration	0.2	1.2	0.6
Total	14.1	7590.2	51.4

ND = not determined.
[a] Atmospheric emission not determined but probably negligible.

Table 18. *Estimated emissions of mercury to air in Sweden, 1984* [45]

Source of emission	Emission of mercury kg/y
Combustion of coal and peat	200
Chlor-alkali industry	400
Crematoriums	300
Smelting (mainly copper and zinc)	900
Incineration of household waste	3300
Other	200
Total	5300

Cadmium release in the European Community

A cadmium inventory survey was carried out in 1980 for the then nine Member States of the EEC [105]. Table 19 presents a breakdown of the amounts of MSW incinerated, and the estimated discharges of cadmium to landfill and the atmosphere. Atmospheric emissions of cadmium from traditional refuse incineration were estimated to exceed 30 t in 1977; the largest contributors were, in decreasing order, the Federal Republic of Germany, France, and the United Kingdom. In comparison with other human activities and natural sources, traditional refuse incineration emerges as a major source of atmospheric release of cadmium, being responsible for about 25% of the regional total, and is the second largest single source category.

Metal release at the global level

Nriagu [109] estimates the total worldwide atmospheric emissions of several metals from human activities. Most of this information is not available at present, but Table 20 summarizes the reported emission strengths for cadmium. Traditional refuse incineration also makes a considerable input to the global release of cadmium, being the largest single source after the nonferrous metal industries.

Emissions of metals other than cadmium, mercury, and lead

In addition to the three metals covered in this report, emissions from municipal incinerators, in common with many combustion processes, will contain trace quantities of several other metals. Traditional MSW incineration may be responsible for major

Table 19. *Quantity of incinerated refuse and estimated discharges of cadmium in the European Community*[a] *in 1977* [105]

Country	Waste incinerated (x 10^6 t)	Cadmium in discharge medium (t)	
		Atmospheric emissions	Ash
Belgium	0.5	0.8	7.3
Denmark	2.5	3.7	33.8
France	4.9	7.4	66.0
Germany, Federal Republic of	5.4	8.1	72.9
Italy	2.6	3.9	35.1
Luxembourg	0.1	0.2	1.4
Netherlands	1.7	2.6	23.0
United Kingdom	3.2	4.8	43.0
Total	20.9	31.5	282.5

[a] Except Ireland.

Table 20. *Worldwide anthropogenic emissions of cadmium to the atmosphere* [109]

Source	Annual release (10^3 t/y)
Mining and production of nonferrous metals	4.71
Secondary nonferrous metal production	0.60
Iron and steel production	0.07
Industrial applications	0.05
Coal combustion	0.06
Waste incineration	1.40
Other sources	0.23
Total	7.12

fractions of zinc, antimony, silver, indium, and tin found in urban aerosols [17].

Some compounds of inorganic arsenic, hexavalent chromium, and nickel have caused lung cancer after occupational exposure to very high concentrations [51]. Such compounds are suspected of being potential carcinogens for the general population, even at the low concentrations found in ambient air. However, based on the emission data, the emissions of such compounds from incinerators would probably contribute very little to ambient air levels. If there is a risk for lung cancer from ambient air levels, the additional risk from incinerator emissions would not be of any practical significance.

Contribution of incinerator emissions to PAH exposure

This section considers the relative importance of MSW incineration as a source of human exposure to PAH. As the data on PAH other than BaP are severely limited, this discussion is restricted mostly to BaP.

Direct exposure via inhalation

The background concentrations of BaP quoted in the Air Quality Guidelines [51] range from less than 1 ng/m^3 in rural areas, less than 10 ng/m^3 in urban areas, and up to 40 ng/m^3 in industrial areas. The last value was recorded in the vicinity of a coke oven plant.

The concentrations of BaP reported in stack gases of incinerators range from less than 1 ng/m^3 for advanced facilities and to 60 ug/m^3 for traditional incinerators (Table 14). Maximum ground-level concentrations of pollutants from individual incinerator plant emissions can be estimated by using mathematical models. These models attempt to allow for several aspects, such as the stack height, local topography, and meteorological conditions in the given area. However, these models have yet to be validated by comparing the estimated concentrations with measurements taken in the field.

A simple dilution factor of 5×10^{-5} [24] may be applied to the stack gas emissions of incinerators to gauge the order of magnitude of ambient air concentrations of BaP at ground level. On this albeit simplistic basis, the worst case emission of BaP given in Table 14 (208 ug/m^3) for traditional incinerators would result in a maximum ambient air concentration at ground level of 10 ng/m^3. This is somewhat less than the concentration of BaP recorded in the vicinity of a coke oven plant [51]. However, for most incinerators of traditional design cited in Table 14, the maximum ambient air concentration of BaP at ground level is likely to be less than 10 ng/m^3 (Table 13), which is similar to the background concentrations in urban ambient air.

Most MSW incinerators are situated in urban areas, so the estimated maximum concentrations of PAH at ground level should be added to the prevailing background values. On the basis of the worst-case examples presented here, a poorly operated incinerator of traditional design may, at most, double the ambient air concentration of PAH. However, the impact of most traditional incinerator plants will likely be much less.

The stack emission data for advanced incinerators (Table 14) show PAH concentrations considerably less than the background concentrations in urban ambient air. This type of incinerator is therefore most unlikely to increase PAH concentrations in air close to the plant. Some recent field measurements also support this conclusion [110]. These workers recorded a maximum concentration of BaP in air at ground level of less than 1 ng/m^3, measured at two sampling sites situated 900 m and 2500 m from an incinerator. Most values recorded were less than 0.4 ng/m^3, and days with peak values of PAH in the ambient air were not correlated to days with peak PAH values in the stack gases of the incinerator plant. What type of incinerator was examined in this study is not clear, but if it were of the traditional type, it would only strengthen the conclusion that advanced incinerators will have little, if any, impact on ambient air levels of PAH.

Mutagenic activity in emissions

Besides analysis of specific chemical components in the flue gas, the biological activity may also be tested, thereby offering complementary information on potential health effects. The *Salmonella*/microsomal assay has been used to detect mutagenic activity in the emissions from different combustion sources, including MSW incineration and ambient air [111-113].

The mutagenic activity is distributed between fractions containing PAH and more polar fractions, probably containing oxygenated PAH components [113]. Thus, other substances besides unsubstituted PAH are of importance. As the identity of these compounds is not known, their potential health impact cannot be assessed.

Technical measures aimed at optimizing the combustion efficiency may reduce the emission of mutagenic substances, as well as the emission of other organic compounds. The results from mutagenicity testing can be used, for example, to compare the effect of different control devices on emissions, especially in combination with chemical analyses. However, the data generated with the *Salmonella*/-microsomal assay are insufficient to estimate the mutagenic potency, and even less the carcinogenic risks of various emissions. On its own, this test can give only an insight into the potential for mutagenicity; it cannot produce quantitative data.

Indirect exposure via deposition onto food crops

The estimated increase in the deposition rate of PAH in the vicinity of MSW incinerators (Table 13) is 0.02 to 1.5 g/ha/y. The influence of this small increase in deposition of PAH may slightly increase the PAH concentrations in food crops grown in the vicinity of a traditional incinerator, particularly one poorly situated with respect to the local geography. However, this source is not expected to increase appreciably the total daily intake of PAH.

Emissions of PAH from MSW incinerators compared with other sources

Table 21 shows that incineration and open burning of refuse contributes about 25% of the global BaP emissions to the atmosphere. The emissions from open burning of refuse and agricultural wastes will heavily outweigh the contribution from MSW incinerators, which together make up this estimate.

More recent estimates for the Nordic countries (Table 22) indicate that the contribution of MSW incinerators to emissions of PAH is probably less than 5% of the total. In Norway, aluminium smelting is the largest single source, whereas in Denmark, agricultural burning is the dominant source. If agricultural burning, forest fires, and industrial emissions are excluded, residential heating with wood contributes from 44 to 68% of the total emission in different countries. Motor vehicles is the other dominant source.

Table 21. *Estimated emissions of BaP to the atmosphere* [114]

Source	BaP in t/y		
	USA	Worldwide (excl. USA)	Worldwide
Heating and power generation	475	2 129	2 604
Industrial processes	198	847	1 045
Refuse and open burning including incinerations	588	762	1 350
Vehicles	22	23	45
Total	1 283	3 761	5 044
Contribution from incineration (%)	46	-20	27

Table 22. *Estimated emissions of PAH from different sources in the Nordic countries (kg/y)* [115]

Source	Norway	Sweden	Denmark	Finland
Residential combustion (wood, oil, gas)	49 000	105 000	29 000	42 000
Industrial production	230 000[a]	48 000	1 200	2 700
Power and heat generation (coal, oil, biomass)	230	6 600	550	1 100
Municipal incineration	250	2 200[b]	3 500	170
Open burning, forest fires agricultural burning	13 000	7 800	33 000	5 500
Mobile sources (gasoline and diesel automobiles)	20 000	47 000	22 000	26 000
Total	312 480	216 600	89 250	77 470
Contribution from waste incineration (%)	0.08	1.0	4.0	0.22

[a] 180 000 kg from aluminium production.
[b] Later estimated to be 840 kg/y [45].

CONCLUSIONS AND RECOMMENDATIONS

Conclusions

1. Cadmium, lead, and mercury are the metals of prime concern because of their toxicity and the relative importance of municipal solid waste (MSW) incineration as a source of these metals emitted to the atmosphere. Considerable amounts of these metals may be emitted from incinerators of traditional design, which in turn may increase exposure among residents in the vicinity of such an incinerator.

2. As most MSW incinerators in operation are of traditional design, they are an important source of cadmium and mercury emissions to the atmosphere at a regional level and can contribute to the total human intake of these metals through the food chain. However, the contribution of lead from incinerators through this pathway is negligible in comparison with emissions from other sources, such as motor vehicles and smelters.

3. Incinerators can be designed and operated in a manner that will affect the physical and chemical properties of metallic combustion products and their distribution into bottom ash, boiler ash, fly ash, and stack emissions. Emissions of metals, such as cadmium, lead, and mercury, can be considerably reduced by use of a proper flue gas cleaning system.

4. On the basis of the limited data available, metal emissions from advanced MSW incinerators may not increase exposure levels in the population living near such facilities.

5. Advanced air pollution control systems result in increased quantities of dry or wet residues. Gas treatment residues, quench waters, and other liquid effluents require appropriate treatment prior to disposal.

6. Management of MSW incinerator residues must be done properly to prevent dispersal of ashes by fugitive emissions and contamination of surface and groundwaters due to the leachability of metals, particularly cadmium and lead.

7. The level of PAH emissions from MSW incinerators varies widely, and the factors causing these variations are not thoroughly understood. Nevertheless, tests show that advanced incinerators operating at sufficiently high temperatures and with adequate oxygen, good mixing, and adequate retention time (together reflecting good combustion conditions) result in lower formation levels of PAH than in traditional incinerators and much lower than in poorly maintained and poorly operated incinerators. The data on advanced incinerator designs indicate that highly efficient destruction of PAH can be achieved in a single-chamber incinerator.

8. The available data indicate that advanced MSW incinerators will make only a minor contribution to the PAH concentrations in ambient air.

9. At a local level, poorly operated MSW incinerators may, in the worst case, increase the ambient air concentration of PAH in urban locations by a factor of 2, and, consequently, may add to the risk of cancer induction. Furthermore, the likelihood of PAH contami-

nation of food crops in the vicinity of such incinerators cannot be excluded. However, PAH are only a fraction of the potentially toxic organic compounds emitted from incinerators.

10. At a regional level, MSW incineration makes only a small (up to a few percent) contribution to the total PAH emission and is therefore of minor importance compared with other sources.

Recommendations

1. The local environmental impact of MSW incinerators should be investigated at a few selected facilities by direct determination of cadmium, mercury, lead and PAH concentrations in air, deposited dust, soil, and vegetation. These values should be compared with appropriate background values. Studies should also be conducted at ash residue landfill sites, with particular reference to the potential contamination of surface and groundwaters by metal leachates.

2. Major sources of metals in MSW, such as cadmium, lead, and mercury, should be identified, their amounts evaluated, the significance of their removal on incineration emissions assessed, and alternative disposal routes investigated.

3. The evaluation of metal and PAH emissions should be based on a series of measurements of emissions from MSW incinerators, and the recording of corresponding operating conditions of the incinerator and flue gas cleaning devices and MSW characteristics.

4. Additional studies should be conducted to validate sampling techniques and analytical methods for the measurement of PAH and metal emissions, especially mercury, from incinerators.

5. A better and more comprehensive scientific database should be developed to refine criteria for the selection of operating conditions for traditionally designed incinerators to control more adequately PAH emissions from such facilities.

6. Many PAH are known animal carcinogens and may also be carcinogenic to humans. As the cancer risk from carcinogenic PAH is assumed to depend on dose, the emissions of these substances, even at low levels over a prolonged time, should be kept to a minimum.

7. Substituted PAH compounds emitted from MSW incinerators, such as methylated, nitrated, and oxygenated PAH, and organic compounds other than PAH, should be identified and their biological effects investigated.

8. Investigations should be continued to increase combustion efficiency that would, in turn, maximize the destruction of organic compounds and their conversion to carbon dioxide.

9. Advanced technologies of MSW incinerators and emission control systems should be implemented rapidly to reduce environmental dispersal of metals and PAH.

10. Application of genotoxicology testing in conjunction with chemical analysis should be considered as a valuable tool for assessing the impact of operating conditions and new technologies on the potential toxicity of pollutant mixtures emitted in the stack gas.

11. The location of new MSW incinerators should be carefully selected to minimize human and environmental impact from metal and PAH emissions.

12. Safe handling of ash residues, both in the incinerator plant and during transport to disposal sites, should be practised to minimize occupational exposure and fugitive emissions; assessments of the metal and PAH exposure levels of the operators should be undertaken. Liquid effluents from wet scrubbers and quenches should be properly treated prior to discharge to receiving waters.

13. The construction of small incinerators should be discouraged when adequate number of properly trained personnel cannot be provided on a cost-effective basis, and should be permitted only when proper pollution control equipment is incorporated in the incinerator facility.

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Annex 1

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Statistical estimate of mercury removal efficiencies for air pollution control devices of municipal solid waste incinerators

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ABSTRACT

Although representative removal efficiencies of gaseous mercury for air pollution control devices (APCDs) are important to prepare more reliable atmospheric emission inventories of mercury, they have been still uncertain because they depend sensitively on many factors like the type of APCDs, gas temperature, and mercury speciation. In this study, representative removal efficiencies of gaseous mercury for several types of APCDs of municipal solid waste incineration (MSWI) were offered using a statistical method. 534 data of mercury removal efficiencies for APCDs used in MSWI were collected. APCDs were categorized as fixed-bed absorber (FA), wet scrubber (WS), electrostatic precipitator (ESP), and fabric filter (FF), and their hybrid systems. Data series of all APCD types had Gaussian log-normality. The average removal efficiency with a 95% confidence interval for each APCD was estimated. The FA, WS, and FF with carbon and/or dry sorbent injection systems had 75% to 82% average removal efficiencies. On the other hand, the ESP with/without dry sorbent injection had lower removal efficiencies of up to 22%. The type of dry sorbent injection in the FF system, dry or semi-dry, did not make more than 1% difference to the removal efficiency. The injection of activated carbon and carbon-containing fly ash in the FF system made less than 3% difference. Estimation errors of removal efficiency were especially high for the ESP. The national average of removal efficiency of APCDs in Japanese MSWI plants was estimated on the basis of incineration capacity. Owing to the replacement of old APCDs for dioxin control, the national average removal efficiency increased from 34.5% in 1991 to 92.5% in 2003. This resulted in an additional reduction of about 0.86 Mg emission in 2003. Further study using the methodology in this study to other important emission sources like coal-fired power plants will contribute to better emission inventories.

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1. Introduction

Since the outbreak of Minamata disease in 1953, the release of mercury to the environment and corresponding adverse effects on human health and the ecosystem have attracted great attention. Although mercury is a natural trace component in the environment, the bioaccumulation of mercury via the food chain, especially in fish, concentrates the toxicity of mercury and is a serious hazard to the biosphere (Harada, 1995).

The anthropogenic release of mercury is principally from the production of metals and combustion processes (van Velzen et al., 2002). The main sources are copper and lead smelters, coal-fired utilities, medical waste combustors, municipal solid waste incinerators (MSWI), cement kilns and chloralkali plants. Pacyna and Pacyna (2002), Pacyna et al. (2006) estimated the global release of mercury from anthropogenic sources in 1995 and 2000 to be 1912.8 Mg and 2190 Mg, respectively. Asia contributes about 50% of the global emission according to inventories. Streets et al. (2009) estimated 2390–4860 Mg of global mercury emission in 2050. There have been recent international efforts for the global assessment and management of mercury. United Nations Environment Programme Chemicals began a global assessment of mercury in 2001.

Since the early 1980s, air emissions of mercury from combustion plants and mercury removal efficiencies for several air pollution control devices (APCDs) have been researched. Removal efficiencies for APCDs of coal-fired power plants, which are major emission sources, have been reported by Yokoyama et al. (2000), Srivastava et al. (2001), Meij et al. (2002), Ito et al. (2006), Pudasainee et al. (2009) and reviewed by Pavlish et al. (2003). In the case of municipal solid waste incineration (MSWI), Reimann (1986) reported 0.42 g-Hg/Mg-waste as air emission and 88.5% mercury removal efficiency for an electrostatic precipitator (ESP) with a wet scrubber. Tanikawa and Urano (1998) reported 27, 35 and 60% removal efficiencies for an ESP, an ESP with dry sorbent injection (DSI) and an ESP with a wet scrubber, respectively. Chang et al. (2000) investigated the speciation and removal efficiencies of mercury in MSWI. They reported 29.56

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